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ROSSANO BELLADONA

THE OPPORTUNITY COST OF WATERSHED CONSERVATION

THE DECISIONS ON URBAN WATER SUPPLY MANAGEMENT

PORTO ALEGRE, BRAZIL

July 6th, 2023

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This thesis was submitted to the Post-Graduation Program in Water Resources and Environmental Sanitation at the Federal University of Rio Grande do Sul, as a partial requirement for the PhD degree.

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For an efficient, equitable and sustainable urban water management.

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SUMMARY

Land use intensification and the discharge of urban waste within the watershed have significantly contributed to the contamination of water sources. The growing concern over deteriorating source water quality has led to increasingly stringent drinking water standards over the years. Consequently, the demand for more advanced and expensive water treatment technologies has risen, resulting in an escalation of water tariffs. One alternative policy to mitigate the additional costs associated with water treatment is the implementation of land use restrictions within the watershed. By regulating land activities, it becomes possible to prevent further degradation of water quality. However, it is important to acknowledge that such conservation measures, aimed at maintaining or restoring watershed quality to ensure the production of raw water, impose financial burdens on landowners residing within the watershed.

Decision makers should prioritize the careful consideration of costs and benefits when planning urban water management, even in the face of changing conditions. An efficient, equitable and sustainable approach to urban water management aims to benefit both city residents, who rely on the distribution of drinking water, and watershed landowners, who must preserve their properties to safeguard the water resources that supply the city. However, the decision-making process is complex and influenced by a myriad of uncertainties. Factors such as land use regulations, political interests, climate change, population growth, economic development, the costs of the chemicals used in water treatment plants and technological advancements all contribute to this complexity. Additionally, modest short-term gains in avoided water treatment costs can blur the perception of the outcomes in the long-term, potentially leading to decisions that are not truly efficient, equitable, or sustainable.

In the dichotomous context between watershed conservation and low water treatment costs for urban water security in a world under deep uncertainties, it becomes imperative to engage in extensive analysis to identify trade-offs and comprehend the consequences associated with each chosen decision pathway. Consequently, the central question of this thesis emerges as the focal point: Which decisions regarding water treatment technologies, aimed at maintaining low tariffs, have the greatest impact on the opportunity cost of watershed conservation? Building upon this fundamental query, the overarching objective of this research is to ascertain a dynamic decision-making

framework that can adapt over a specified timeframe, ensuring affordable water tariffs while simultaneously reducing the economic burden on landowners who contribute to watershed conservation.

Nonetheless, there are additional secondary objectives that also need to be addressed. These include (1) identifying the extent to which treatment costs are influenced by the quality of the source water watershed, (2) estimating the additional water treatment costs when advanced technologies need to be incorporated into the existing treatment process due to watershed degradation, (3) quantifying the costs and benefits associated with watershed conservation and (4) developing an adaptive pathways map to facilitate robust decision-making in various future scenarios. To achieve these objectives, a methodological approach was employed and tested in a real-world case study: the municipality of Caxias do Sul, located in the State of Rio Grande do Sul, southern Brazil. This municipality supplies drinking water to over 500,000 inhabitants through five conventional treatment plants, which receive raw water from five small watersheds ranging from 5 to 100 km^2 in size. Detailed explanations of the applied methods, materials used, and the results obtained can be found in Chapters 4, 5, 6, and 7.

Existing literature is limited in its comprehensive consideration of how water treatment costs are influenced by the quality of the watershed, particularly in the context of short and long-term planning. No references have been found that specifically associate this effect with planning considerations. Chapter 4 of the thesis sheds light on this aspect by illustrating how the short-term economic benefits derived from improved raw water quality can blur the perception of long-term treatment costs. The results indicate that, in the studied area, a one-unit improvement in the water quality index leads to a marginal reduction in treatment costs of only 0.0002 $USD \cdot m^{-3}$. However, over the long-term, the cumulative effect of water quality degradation may result in an alarming increase of 242% in treatment costs. These findings underscore the importance of considering the long-term implications of watershed quality on treatment costs, emphasizing the need for holistic planning approaches that account for the interplay between raw water quality and treatment expenses.

Obtaining socioeconomic data for small watersheds can be a challenging task, often resulting in a lack of available information. However, such data are crucial for making informed decisions regarding land use and water resources management, particularly when assessing their economic value. Recognizing the significance of addressing this knowledge gap, Chapter 5 of the thesis focuses on estimating key socioeconomic indicators, namely Gross Domestic Product (GDP), population, and jobs, for small areas. To overcome the data limitations, Chapter 5 employs nighttime light (NTL) satellite images and utilizes existing socioeconomic records from larger localities. By establishing a relationship between the radiance quantified from the NTL images and the three socioeconomic indicators, a simple regression analysis is conducted across the 497 municipalities in the State of Rio Grande do Sul. The findings are further validated using data from 50 municipalities in the neighboring State of Santa Catarina. The results demonstrate that this innovative application of NTL for estimating socioeconomic data can serve as a valuable tool in supporting land use and water resources management for small watersheds.

In the context of the aforementioned dichotomy, it is often observed that those who bear the burden of watershed conservation rarely reap the direct benefits, because they live away from the water distribution network. Chapter 6 introduces a novel methodological framework, inspired by Data Envelopment Analysis, which explores the relationship between the opportunity cost of watershed conservation and changes in consumer surplus to quantify the benefits associated with such efforts. The results obtained from this analysis demonstrate that the economic benefits derived from watershed conservation outweigh the associated costs in all evaluated scenarios. These findings highlight that, when policies are oriented towards improving source water quality, there is a positive economic balance in the aggregate. This provides evidence that prioritizing conservation efforts has a beneficial outcome, both in economic and environmental terms.

Urban water security is influenced by various factors, including the quantity and quality of raw water both in space and time. However, an often-overlooked aspect is the value of drinking water tariffs. Ensuring that tariffs remain affordable for users is crucial for achieving urban water security. In this regard, robust decision-making plays a significant role as it enables a thorough evaluation of the trade-offs between optimal land use practices within the watershed and optimal treatment costs. By considering potential future scenarios, decision-makers can make informed choices that are more likely to yield successful outcomes. Chapter 7 of this thesis focuses on the exploration of robust decisions in urban water supply. It employs the Dynamic Adaptive Policy Pathways methodology, coupled with a heuristic mathematical approach, which is a novel application in the field of watershed conservation and water treatment technology. The result is an adaptive decision map that presents various technological options, enabling decision-makers to understand the consequences associated with different alternatives. The purpose is not to determine the correct pathway, but to provide decision-makers with a comprehensive understanding of the potential outcomes associated with different decisions. This approach empowers decision-makers to make informed choices based on a deeper understanding of the implications of their actions within an adaptive context.

The pursuit of low water treatment costs can come at a significant economic price. This "shadow price" of urban water must be acknowledged, and actions should be taken to compensate landowners for their role in preserving watersheds. Addressing this incongruence requires further investigation, and one potential solution could be the implementation of a payment for watershed services, similar to existing mechanisms of payment for environmental services. Throughout the thesis, it is recognized that urban water security is accompanied by inherent uncertainties. The research demonstrates that there is no singular optimal decision that perfectly balances costs and benefits in urban water management. Instead, a wide range of efficient alternatives exists, each involving trade-offs. Decision-makers and stakeholders must be prepared to adapt their planning approaches to accommodate evolving future conditions. This allows for the exploration of diverse and efficient alternatives while considering the trade-offs inherent in each option.

Keywords: Opportunity cost. Benefits. Watershed. Urban water security. Uncertainty.

SUMÁRIO (PORTUGUESE SUMMARY)

A intensificação do uso do solo e os resíduos urbanos na bacia têm contribuído para o aumento de contaminantes na água bruta. A preocupação com a baixa qualidade da água dos mananciais tem deixado os padrões de água potável mais rigorosos ao longo dos anos, o que exige um tratamento mais tecnológico e caro, culminando no aumento das tarifas de água. Para mitigar o aumento nesses custos, diversas ações devem ser planejadas e implementadas na bacia hidrográfica, que podem acabar por restringir alguns usos do solo e gerar a necessidade de melhores práticas de manejo. No entanto, essa conservação (ações para manter ou recuperar a qualidade da bacia hidrográfica para garantir a água ao abastecimento público) gera custos para os proprietários de terras nessas bacias.

Os responsáveis pela tomada de decisão devem buscar avaliar custos e benefícios para as diversas partes envolvidas (por exemplo, os moradores de áreas ruais e urbanas, os produtores rurais e industriais) e identificar soluções que equilibrem esses custos e benefícios a longo prazo, para garantir segurança hídrica sob condições de incerteza futura. Aqui os responsáveis pela tomada de decisão incluem a administração pública municipal (prefeitos e secretarias), as empresas de saneamento, as entidades reguladoras e os comitês de bacia hidrográfica. Uma gestão eficiente, equânime e sustentável das águas favorece tanto os moradores das cidades, que recebem água potável nas suas casas, quanto os proprietários de terras na parte rural das bacias hidrográficas (por exemplo, os produtores agrícolas, as agroindústrias e os pequenos aglomerados e comunidades) cujos usos do solo e da água devem envolver ações e práticas de conservação para a salubridade dos recursos hídricos para abastecimento de todos. No entanto, a tomada de decisão é cercada por uma grande quantidade de incertezas, como a restrição do uso do solo, os interesses políticos, as mudanças climáticas, o crescimento populacional, o desenvolvimento econômico e o avanço tecnológico. Além disso, ganhos modestos de curto prazo com os custos evitados no tratamento de água podem ofuscar a percepção dos resultados de longo prazo, levando a decisões que não sejam eficientes, equânimes e sustentáveis.

Nessa dicotomia entre conservação de bacias hidrográficas e baixos custos de tratamento de água para garantir a segurança hídrica urbana em um mundo sob profundas incertezas, identificar *trade-offs* e mostrar as consequências de cada caminho escolhido é um procedimento que deve ser avaliado. Assim, a pergunta central desta tese vem à tona: quais decisões sobre tecnologias de

tratamento de água para manter a tarifa baixa mais influenciam no custo de oportunidade da conservação de bacias hidrográficas destinada ao abastecimento público? Com base nessa pergunta, o objetivo geral desta pesquisa é identificar uma combinação de caminhos dinâmicos de decisão que possam se adaptar ao longo de um determinado período de tempo para garantir tarifas de água acessíveis e, ao mesmo tempo, reduzir as perdas econômicas que os proprietários de terras têm para a preservação da bacia hidrográfica.

Essa tese possui ainda outros objetivos secundários, como (1) identificar até que ponto o custo do tratamento é influenciado pela qualidade da bacia hidrográfica, (2) estimar o custo adicional do tratamento da água quando uma tecnologia avançada deve ser adicionada àquela existente em um cenário de degradação da bacia, (3) quantificar o custo e o benefício da conservação de bacias hidrográficas e (4) desenvolver um mapa de caminhos adaptativos para produzir decisões robustas em diferentes cenários futuros. Para atingir todos esses objetivos, a abordagem metodológica foi testada em um caso real, o município de Caxias do Sul no Estado do Rio Grande do Sul. O município abastece mais de 500.000 habitantes por meio de cinco estações de tratamento convencionais cuja água é recebida de cinco pequenas bacias hidrográficas (de 5 a 100 km^2). Os Capítulos 4, 5, 6 e 7 detalham os métodos aplicados, o material utilizado e os resultados obtidos.

Existe uma literatura limitada que explica adequadamente como os custos de tratamento de água são impactados pela qualidade da bacia, nenhuma referência foi encontrada associando esse efeito ao planejamento de curto e longo prazo. O Capítulo 4 demonstra que muitas vezes o benefício econômico de curto prazo da qualidade da água bruta pode ofuscar a percepção dos custos de tratamento de longo prazo. Os resultados demonstram que há uma redução marginal no custo de tratamento de apenas 0,0002 $USD{\cdot}m^{-3}$, dada a melhoria de uma unidade no índice de qualidade da água na área de estudo. No entanto, a longo prazo, o efeito cumulativo pode aumentar os custos do tratamento em 242%. Esses resultados ressaltam a importância de considerar as implicações de longo prazo da qualidade das áreas de mananciais nos custos de tratamento, enfatizando a necessidade de abordagens de planejamento holístico que levem em conta a interação entre a qualidade da água bruta e os custos de tratamento.

Dados socioeconômicos para pequenas bacias são difíceis de encontrar, dependendo do tipo de informação requerida eles simplesmente não existem. No entanto, esses dados são essenciais na tomada de decisões sobre o uso do solo e na gestão dos recursos hídricos, principalmente na determinação do seu valor econômico. A fim de contribuir para preencher essa notável lacuna de conhecimento, o Capítulo 5 estima o Produto Interno Bruto (PIB), população e empregos para pequenas áreas, aplicando imagens de satélite de luz noturna (NTL) e registros socioeconômicos disponíveis de localidades maiores. A relação entre esses três indicadores socioeconômicos e a radiância quantificada nas imagens NTL foi obtida por meio de análise de regressão simples aplicada nos 497 municípios do Estado do Rio Grande do Sul e validada com 50 municípios do vizinho Estado de Santa Catarina. Os resultados indicam que esta nova aplicação da NTL para estimar dados socioeconômicos pode ser uma ferramenta útil para a gestão dos recursos hídricos e do uso do solo em pequenas bacias hidrográficas.

Dentro da dicotomia supracitada, quem conserva costuma arcar com o ônus e não necessariamente compartilha todos os benefícios, pois habita região fora da cobertura da rede de distribuição de água. O Capítulo 6 apresenta uma nova estrutura metodológica, seguindo a Análise Envoltória de Dados, para abordar a relação entre o custo de oportunidade da conservação de bacias hidrográficas, enquanto segue as mudanças no excedente dos consumidores para calcular os benefícios de tal preservação. Os resultados demonstraram que o benefício econômico da conservação de bacias hidrográficas excede seu custo em todos os cenários avaliados. Essa descoberta revela que, no agregado, o saldo econômico é positivo quando as políticas são voltadas para a qualidade da água dos mananciais. Isso evidencia que priorizar os esforços de conservação tem um resultado benéfico, tanto em níveis econômicos quanto ambientais.

A segurança hídrica urbana é influenciada por vários fatores, incluindo a quantidade e a qualidade da água bruta, tanto no espaço quanto no tempo. No entanto, um aspecto muitas vezes esquecido é o valor das tarifas de água potável. Garantir que as tarifas permaneçam acessíveis para os usuários é crucial para alcançar a segurança hídrica urbana. Decisões robustas são importantes para a segurança hídrica urbana, pois as compensações entre o uso ideal do solo na bacia e o custo ideal do tratamento são mais bem avaliadas e os resultados são propensos ao sucesso em possíveis cenários futuros. Ao considerar possíveis cenários futuros, os tomadores de decisão podem fazer escolhas propensas a gerar resultados bem sucedidos. O Capítulo 7 foca nas decisões robustas no abastecimento de água urbana, empregando a metodologia *Dynamic Adaptive Policy Pathways*, seguida de uma abordagem matemática heurística, em uma nova aplicação no campo da conservação de bacias hidrográficas e tecnologia de tratamento de água. O resultado é um mapa de decisão adaptativo que apresenta possíveis ações de tecnologia que permitem aos tomadores de decisão entender as consequências de diferentes alternativas, em vez de dizer qual é o caminho correto. Essa abordagem permite aos tomadores de decisão fazer escolhas com base em uma compreensão mais profunda das implicações de suas ações dentro de um contexto adaptativo.

A busca por baixos custos de tratamento de água pode ter um preço econômico significativo. Esse preço sombra (*shadow price*) da água urbana precisa ser contabilizado e algum tipo de ação deve ser tomada para compensar os proprietários de terras devido ao seu papel na preservação das áreas de mananciais. A forma de ajustar tal incongruência deve ser melhor investigada, e uma possível solução poderia ser a implementação de um pagamento por serviços de conservação da bacia hidrográfica cuja água é destinada ao abastecimento público, semelhante aos mecanismos existentes de pagamento por serviços ambientais ou serviços ecossistêmicos. Esta tese reconhece que a segurança hídrica urbana é acompanhada de intrínsecas incertezas. A pesquisa demonstrou que não existe uma única decisão ótima capaz de equilibrar perfeitamente custos e benefícios na gestão da água urbana. Em vez disso, mostrou um conjunto de diversas alternativas eficientes, cada uma envolvendo *trade-offs*. Os tomadores de decisão e as partes interessadas devem estar preparados para adaptar o planejamento para condições futuras que vão surgindo. Isso permite a exploração de alternativas diversas e eficientes, considerando os *trade-offs* inerentes a cada opção.

Palavras-chave: Custo de oportunidade. Benefícios. Bacia hidrográfica. Segurança hídrica urbana. Incerteza.

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ATP	Adaptation Tipping Point
BCC	Banker, Charnes and Cooper method
BRL	Brazil Real (R$)
Cap	Volume of treated water [$L.s^{-1}$]
CCR	Charnes, Cooper and Rhodes method
CN	Curve Number
CPI	Consumer Price Index
CRS	Constant Returns to Scale
CS	Consumers Surplus
DAPP	Dynamic Adaptive Policy Pathways
DBP	Disinfection By-Products
DEA	Data Envelopment Analysis
DMU	Decision-Making Unit
DO	Dissolved Oxygen
DWTP	Drinking Water Treatment Plant
GDP	Gross Domestic Product
MF	Microfiltration
NF	Nanofiltration
NSE	Nash-Sutcliffe
NTL	Nighttime Light (radiance)
NTU	Nephelometric Unit
O&M	Operational and Maintenance
OC	Opportunity Cost
PAC	Powder Activated Carbon
PES	Payment for Environmental Services
RMSE	Root Mean Square Error
RO	Reverse Osmosis
RSR	Standard Deviation Ratio
SAMAE	Public Water and Wastewater Service of Caxias do Sul
SBEI	Slacks-based Efficiency Index
SBM	Slacks-based Efficiency Measure
SDWA	Safe Drinking Water Act
TC	Treatment Cost

UF	Ultrafiltration
USD	United States Dollar (US$)
VRS	Variable Returns to Scale
WCI	Water Compliance Index
WCIL	WCI technology lower bound
WTP	Willing to Pay
θ_1	Economic efficiency
θ_2	Environmental efficiency

CONTENTS

CHAPTER 1

1 INTRODUCTION

In different branches of the scientific community, water is referred to as a dilute aqueous solution of inorganic and organic compounds (Snoeyink and Jenkins, 1980). Unfortunately, land use intensification and urban waste have contributed to the addition of allochthonous constituents to water. The transport of nutrients from the uppermost portions of the watershed to the water system inlet, resulting from the replacement of natural vegetation such as trees, shrubs, and grasses with anthropic activities, leads to an increase in soluble salts and the concentration of these elements in raw water (Liu et al., 2015; Meneses et al., 2015; Nielsen et al., 2012; Simedo et al., 2018). Furthermore, land use interactions with natural variables such as climate, relief, and aquatic life can deteriorate water quality, even in watersheds with the best conservation practices (Lee and Biggs, 2015).

Since the cholera outbreak in mid-nineteenth-century London, which was controlled after John Snow linked the infection cases to the urban water from local pumps (Arlinghaus and Nystuen, 1988), regulations on drinking water quality have become more rigorous, and the general public has become both more knowledgeable and more discriminating about water quality (Crittenden et al., 2012). Paradoxically, this has led to increased technological demands and ever-growing investments in treatment, imposing an undue burden on residents through additional increases in water tariffs.

In 1977, a significant milestone was reached when the initial set of Brazilian standards aimed at protecting public drinking water and its sources was published. This milestone closely followed the introduction of the Safe Drinking Water Act (SDWA) in the United States, which occurred just three years earlier. Since 1914, water standards in the United States have undergone frequent updates, particularly between 1998 and 2004, when new regulations were established, including the implementation of the Disinfection By-Products (DBP) Rule (Libânio, 2010). As a consequence of the growing presence of undesirable elements, the number of regulated contaminants has experienced a significant increase (Howe et al., 2012), as depicted in Figure 1.1. Notably, in Brazil, the number of controlled parameters has more than tripled between 1977 and 2021. In 1977, there were 36 controlled parameters (Brasil, 1977), whereas the current count stands at 116 (Brasil, 2021a). Moreover, non-legal requirements (Heberling et al., 2015) have become more stringent due to heightened consumer awareness and demands regarding drinking water quality. Furthermore, there is a genuine concern for the reputation and public perception of the company responsible for the water system.

In Brazil, 75.1% of the drinking water volume that supplies approximately 121 million people comes from conventional treatment plants, while 20% of the total volume is treated by simple disinfection, with the latter typically associated with municipalities with populations of fewer than

5,000 inhabitants (IBGE, 2020a). Only in special cases, where the natural water quality meets potability standards and the conditions established by the sanitary authorities, is the treatment waived (Brasil, 2021a; Shammas and Wang, 2016). Conversely, low-quality raw water requires the addition of more complex units to the existing treatment train.

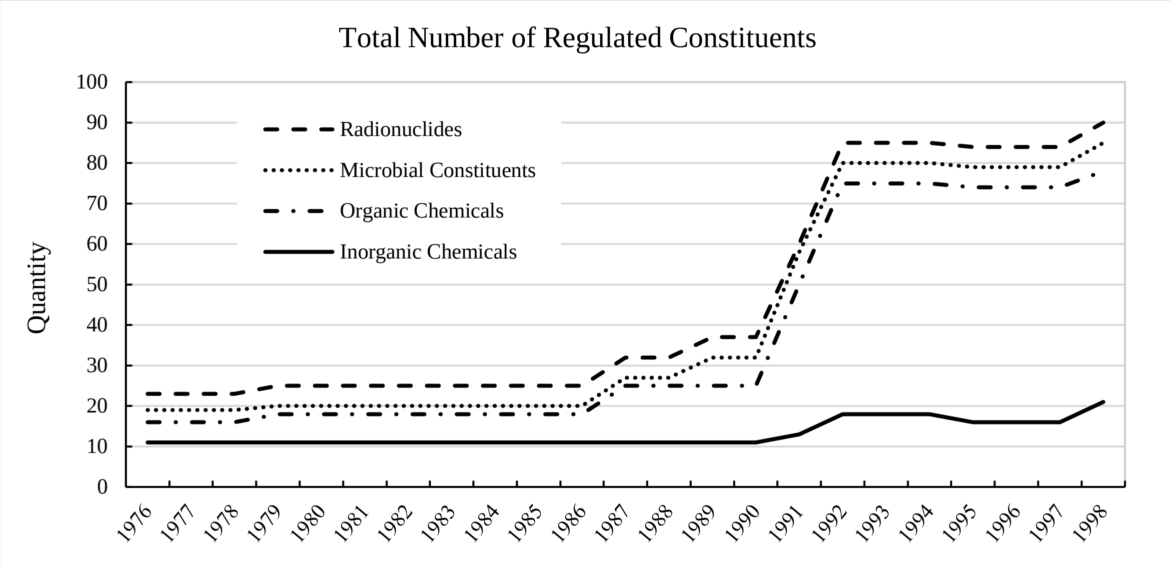

Figure 1.1. Regulated drinking water parameters.
The increasing number or regulated parameters since the inception of the SDWA (Howe et al., 2012).

An operational alternative for purifying raw water with a high concentration of inorganic and organic compounds, especially in watershed where surface water conservation is not prioritized in public policies, involves the utilization of advanced treatment technologies, such as membranes (McDonald et al., 2016). Low pressure filtering membranes (microfiltration and ultrafiltration) can be employed either as a replacement for conventional treatment methods when the concentration of particulate matter in the raw water is low, or in conjunction with conventional treatment to enhance the removal of particles and microorganisms, reducing the formation of DBP by minimizing the required chlorine dosage (EPA, 2005). On the other hand, high-pressure membranes, such as nanofiltration and reverse osmosis, require high quality pre-treatment procedures, particularly when dealing with water obtained from surface sources (Ratnayaka et al., 2009).

Moreover, it has been demonstrated that there is a modest short-term gain in avoided water treatment costs when conservation actions are held at the watershed (Price et al., 2017; Price and Heberling, 2018). In the short-term, it is less costly to reduce the marginal increase in contaminant loads during treatment than the economic and political efforts required to reduce them at the watershed (Heberling et al., 2015). Such low short-term gains can blur the perception of the outcomes in the long-term, because decision makers may prioritize policies that exhibit a myopic attitude towards watershed conservation. Consequently, this can compromise the efficiency of water treatment plants that rely on the conventional processes, ultimately posing a threat to urban water security. In this thesis, the concept of watershed conservation will be treated as those actions aimed at maintaining or restoring the physical and biological processes that are associated with raw water production (both in terms of quantity and quality) that are necessary to maintain water supply sources without compromising the costs of treatment. Within the context of this problem, and as a preliminary step preceding the exploration of the central research question (Chapter 1.2), the first sub-question for investigation emerges:

> *What are the long-term economic effects on water treatment costs when current decisions are being blurred by modest short-term gains in avoided treatment costs?*

In this thesis, the aforementioned question is comprehensively examined and analyzed in detail within Chapter 4.

This blurring effect is particularly concerning in countries such as Brazil, where the average monthly income is 449 United States dollars (USD) and the unemployment rate among the working-age population is 14.1% (IBGE, 2021a). The long-term increase in treatment costs in a scenario of watershed degradation can be the tipping point for an unrestricted rise in water tariffs, potentially leading to a state of “water tariff insecurity”. This term implies that although there may be ample water available within the watershed to meet the demand and technologies exist to treat water to a certain quality level, a significant portion of urban residents would be unable to afford the required tariff. Given these circumstances, one might assume that the solution to address "water tariff insecurity" lies in focusing on policies that prevent anthropic interventions and activities within the watershed, thereby preserving its pristine state as much as possible. Unfortunately, achieving this objective is not as straightforward as it may seem. While it is true that these activities are crucial for the socioeconomic development of the locality, they also have adverse impacts, both on the quantity (Dibaba et al., 2020; Dunn and Mackay, 1995; Yira et al., 2016) and on the quality (Gorgoglione et al., 2020; Kändler et al., 2017; Rodrigues et al., 2018) of water, resulting in negative externalities.

Achieving a balance between the various functions of land, namely the social, ecological, and economic functions, is crucial (Liu et al., 2021). However, therein lies a challenge. Despite its positive externalities, watershed conservation poses a negative economic impact on landowners who bear the costs of conservation efforts. Furthermore, they are the ones who rarely benefit from low tariff, as they are far away from the drinking water distribution system. Consequently, the second relevant subquestion arises:

> *What is the opportunity cost of watershed conservation to keep affordable water tariff to city dwellers?*

The aforementioned question is thoroughly addressed and examined in its entirety within Chapter 6.

Even if the short and long-term water treatment costs are accountable, albeit difficult to be quantified, there is also a cost associated with maintaining the watershed preserved. This conservation cost can be represented in terms of opportunity cost. Opportunity cost is the cost associated with opportunities that are forgone by not putting the watershed's resources to their best alternative economic use (adapted from Pindyck and Rubinfeld, 2018), and depends on how much of a good is being produced to the detriment of another. The opportunity cost of something is what is sacrificed in order to obtain it, which can be easily represented in a production possibilities frontier (Mankiw, 2018; Nicholson and Snyder, 2017). The economy can produce any combination of, for instance, watershed conservation and water treatment cost inside or on the frontier, with the latter representing the Pareto front, as points outside the frontier are not feasible given the available resources (Mankiw, 2018). It should be noted that, in the considered case, greater watershed conservation (resulting in higher opportunity cost) corresponds to lower treatment costs, as it is assumed that pristine watersheds yield high-quality raw water. Agricultural rent serves as another example of the opportunity cost of land, because it is an important indicator of urban growth. In this case, when agricultural rent increases, the growth rate of cities decreases and vice versa. When agricultural rent becomes higher, the city's expansion is curtailed at its periphery as individuals will move out of the area with high agricultural rent to avoid the decrease in utility (Jiang, 2011).

Regardless of the innumerous economic studies having the opportunity cost as the main scope, one major knowledge gap in this field is acknowledged in this thesis: quantifying the opportunity cost of watershed conservation. Despite its concept being deeply referred to in the literature of water resources (Dinar et al., 2015; Griffin, 2006; Loucks and van Beek, 2017; Young and Loomis, 2014), there is still a dearth of researches devoted to studying the application and effectively quantifying this cost in water resources management (Belladona et al., 2019). Hence, this thesis aims to delve deeper into the discussion surrounding the opportunity cost of watershed conservation, making a significant contribution towards narrowing this knowledge gap. Moreover, this research will also unfold the trade-off between the optimal economic activities within the watershed and the optimal water treatment cost, giving rise to a third pertinent subquestion:

> *How can the opportunity cost of watershed conservation be compared to the economic benefits of such preservation?*

This thesis proposes the use of a fundamental concept applied in economics to evaluate the benefit of watershed conservation: consumer surplus. Because consumer surplus is the amount a buyer is willing to pay for a cubic meter of water minus the amount the buyer actually pays for it, it measures the benefit present consumers receive from participating in a market (Jehle and Reny, 2011; Mankiw, 2018). Any fluctuation in water tariff or water quantity, resulting from factors such as degraded watershed quality leading to higher costs for water treatment, can have a direct impact on consumer surplus. Watersheds with lower quality provide raw water of inferior quality, necessitating the implementation of more advanced water treatment trains. As a result, the cost of treatment increases, subsequently leading to higher water tariffs.

Production possibility curves illustrate the various combinations, including the efficient ones, between two decisions. However, decision-making at the watershed scale is significantly more complex compared to the choices made by individual consumers between two consumer goods. This complexity arises from the fact that decision-making at the watershed scale involves a multitude of interested parties, and thus the number of decision variables is as extensive as the number of these stakeholders. The balance of these two conflicting interests involves more than hydrology, economics and technology, it involves people's values and priorities, and the optimum combination might depend on who is asked (Loucks and van Beek, 2017). Moreover, at this scale, policies must consider their implications for many decades to come (Lempert et al., 2003), thus surrounding decision-making with deep uncertainties.

Deep uncertainty is not simply the absence of knowledge (Walker et al., 2013), it also exists when those responsible for decision-making cannot reach an agreement on a model capable of describing the interactions between system variables (Lempert et al., 2003) and the optimal trajectory is hardly found. The challenges of urban water security are compounded by inherent uncertainties surrounding potability standards, escalating anthropic pressure on watersheds, and the costs associated with water treatment technologies. These challenges present significant threats to the decision-making process in ensuring urban water security. Additionally, uncertainties will arise from many other external factors, such as climate change, population growth, and economic developments (Haasnoot, 2013). Under this myriad of severe uncertainties analysts are blind, the parties cannot agree on a decision, probability distributions cannot be used (uncertainty is incalculable and uncontrollable) and the desirability of alternative outcomes is not possible to be valued (Lempert et al., 2003; Marchau et al., 2019). This condition is referred to as deep uncertainty. In light of this, the fourth pertinent subquestion arises:

> *What decisions can be made today and during the course of time to ensure urban water security in the future in a world of deep uncertainties?*

The ability to anticipate and adapt to changes is crucial in long-term decision-making processes. However, the presence of deep uncertainties makes it increasingly challenging to effectively anticipate future developments. Within this context, Chapter 8 of this thesis focuses on addressing the aforementioned question by presenting technology pathways that can be flexibly adapted over time as the future unfolds. These pathways aim to establish robust strategies capable of addressing various future scenarios. Such strategies must consider the quality of the water resources and the land use policies within the watershed as central factors in ensuring affordable water tariffs.

1.1 Research topic

Based on the considerations mentioned above, this thesis centers on the following topic:

The opportunity cost of watershed conservation and the decisions to ensure urban water security in a world under deep uncertainties.

1.2 Research question

Having the research topic as a general guideline, this thesis attempts to answer to the following central research question:

Which decisions regarding water treatment technologies, aimed at maintaining low tariffs, have the greatest impact on the opportunity cost of watershed conservation?

1.3 Research hypotheses

The research question can be addressed through the formulation of hypotheses. Thus, this thesis proposes the following hypotheses:

- Decision-making based on short-term land use policies within the watershed leads to a significant increase in long-term water treatment costs.
- The opportunity cost of watershed conservation is high.
- The benefits of watershed conservation outweigh the costs.
- It is possible to identify decision pathways that enable the identification of the most suitable technological water treatment train, ensuring low tariffs and thereby guaranteeing urban water security.

1.4 Research objectives

1.4.1 General objective

The objective of this PhD research is to develop decision pathways that can be dynamic over time in order to ensure the provision of affordable public drinking water supply while minimizing the economic loss experienced by landowners who preserve the source water watershed.

To achieve this objective, the thesis employs five full-scale treatment systems. Each system comprises (1) a dam that retains water from small streams and forms (2) a reservoir, (3) the watershed area bounded by the dam and upstream boundaries, and (4) the drinking water treatment plant.

The scope of urban water security addressed in this thesis encompasses three dimensions: water quality, water quantity, and water cost. Urban water security is compromised when one or more of these dimensions are not met. For example, simply having an adequate quantity of water and employing efficient technology to meet potability standards is insufficient if the resulting treatment costs drive water tariffs to unaffordable levels.

1.4.2 Specific objectives

The specific objectives of this thesis are as follows:

- Determine the correlation, if any, between water treatment costs in conventional drinking water treatment plants (DWTPs) and the quality of the watershed.
- Estimate the additional costs incurred in conventional water treatment due to the integration of advanced treatment methods using filtering membranes.
- Quantify the opportunity cost associated with watershed conservation.
- Quantify the benefits derived from watershed conservation.

- Identify and assess the scarcity rent for water.
- Conduct a comparative analysis of the costs and benefits associated with watershed conservation.
- Develop water treatment technology pathways that are adaptive and capable of creating robust strategies in different future scenarios.

1.5 Outline of this thesis

Along with this Introduction Chapter, this thesis is divided into 11 Chapters. In Chapter 2, it is provided a concise description of the methodological framework that was applied throughout the study. Chapter 3 details the area in which this framework was tested. The municipality of Caxias do Sul in the State of Rio Grande do Sul, southern Brazil, was selected as the case study due to its unique water system arrangement. About half million people are supplied daily with drinking water that is provided from five different watersheds located either in urban or rural areas. Chapter 8 presents a brief discussion of the findings and insights gained during the course of the study. It discourses about the scientific innovation of the thesis and its contributions to the field of water management science. Chapter 9 summarizes the most significant conclusions drawn from the research and provides suggestions for future studies related to water supply and watershed management. The extensive bibliography used to support this research and to aid in its findings is compiled in Chapter 10. Finally, Chapter 11 brings the appendix of Chapter 6 where it is given a more in-depth explanation of the methodological procedure employed for the Slacks-based efficiency index (SBEI).

Notwithstanding, the central part of this thesis comprises four core Chapters: Chapters 4, 5, 6, and 7. These Chapters have either been published, submitted to scientific journals, or are forthcoming. While efforts have been made to avoid repetitive content, as it is inevitable when formatting them separately for journal submission, the reader might still find some overlap in the content between the Chapters. However, this redundancy is intended to ensure the coherence and completeness of each individual Chapter, even when read independently.

Chapter 4 examines the potential distortion caused by modest short-term gains with avoided treatment costs, when compared to the investments on watershed conservation, and how it may blur the perception of significant long-term increases in water treatment costs. In Chapter 5, socioeconomic data for the studied watersheds is estimated using nighttime light satellite images and available records from larger locales. Official socioeconomic data is often unavailable at the specific physical basin boundaries, especially for small watersheds with varying areas ranging from 5 to 100 km², such as the case of Caxias do Sul. Chapter 6 introduces a novel methodological framework in the field of water resources management. This framework quantifies the opportunity cost of watershed conservation and assesses the economic benefits associated with this practice in ensuring urban water security. Chapter 7 assesses the trade-offs between optimal land use and optimal treatment cost that can lead to success across possible future scenarios. The success is reached when city dwellers can afford tariff and landowners at the watershed are not incurred additional opportunity cost. For that, this Chapter draws an adaptive decision map presenting alternative technological actions that can be taken to cope with watershed conservation/degradation.

CHAPTER 2

2 METHODOLOGICAL FRAMEWORK

To achieve the objective of this thesis, a methodological framework (schematically represented in Figure 2.1), which is divided into four parts, was applied to (1) estimate short and long-term treatment costs and establish their correlation with the quality of the watershed, (2) estimate socioeconomic data for small watersheds based on nighttime light satellite image and available records from larger areas, (3) quantify the opportunity cost of watershed conservation and the economic benefits associated with this conservation and (4) evaluate the trade-offs between this opportunity cost and the costs of water treatment.

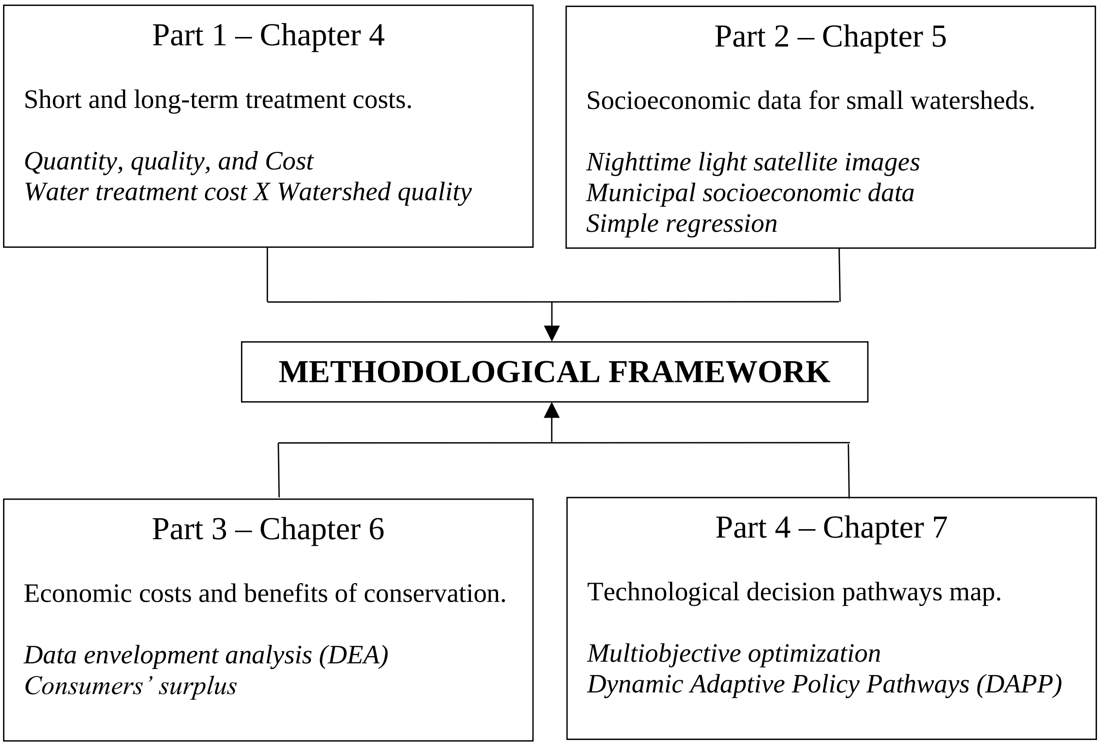

Figure 2.1. Methodological framework.

2.1 Methodological part 1

Chapter 4 introduces a methodology for quantifying the disparity between short-term and long-term water treatment expenses, taking into account the state of the quality of the watershed. The study focuses on evaluating the short-term costs associated with five conventional water treatment plants that utilize powdered activated carbon, considering the quality of their respective watersheds. The analysis establishes a correlation between watershed quality and short-term costs. To approximate the watershed quality, the study employs the water quality index proposed by the Canadian Council of Ministers of the Environment (CCME, 2001).

To evaluate the long-term costs of water treatment, particularly in cases where the short-term quality-cost relationship is insignificant or very weak, a future scenario is considered. This scenario assumes intensified land use, resulting in a compromised watershed quality. In this scenario, the conventional treatment process can no longer ensure compliance with potability standards. Consequently, an advanced treatment method involving membrane filtration is integrated into the existing treatment process, leading to augmented costs.

2.2 Methodological part 2

In the second part (Chapter 5), a simple regression analysis is employed to establish correlations between three socioeconomic indicators - Gross Domestic Product (GDP), population, and jobs - and encoded nighttime images from the DMSP-OLS and NPP-VIIRS sensors for the years 2011, 2014, and 2018. These images are sourced from the study conducted by (Chen et al., 2020). The analysis encompasses 497 municipalities in the State of Rio Grande do Sul. The regression equations yielding the highest Coefficient of Determination (R^2) are subsequently validated using a subset of 50 municipalities from the neighboring State of Santa Catarina. Finally, the validated equations are applied to estimate these indicators for the watersheds corresponding to each of the five water treatment systems. Additionally, data for two other areas, although not part of the water treatment system, are also estimated to verify the accuracy of the estimation results.

2.3 Methodological part 3

The third part (Chapter 6) introduces a novel application of Data Envelopment Analysis (DEA) in the context of watershed conservation. DEA, originally proposed by Charnes et al. (1978) and Banker et al. (1984), and later modified by Tone (2001) and Zhou et al. (2006), is utilized to quantify the opportunity cost associated with watershed conservation. Each watershed is considered a Decision-Making Unit (DMU) in this analysis. Since there are five watersheds and their socioeconomic indicators are represented for three different years (2011, 2014, and 2018), a total of 15 DMUs are included in the model. In the DEA model, two input variables ($\mathbf{x}$) are used: (1) the number of jobs and (2) the productive area of each DMU. The model incorporates two output variables: (1) a desirable product ($\mathbf{g}$), represented by GDP, and (2) an undesirable product ($\mathbf{b}$), represented by the impermeable area for each DMU.

While DEA quantifies the costs associated with watershed conservation, the benefits of conservation efforts are calculated based on the concept of total consumer surplus. This approach draws upon the works of Nicholson and Snyder (2017), Mankiw (2018) and Pindyck and Rubinfeld (2018). Specifically, the total benefit is determined by assessing changes in total consumer surplus.

Prior to this, the Marshallian demand curve for municipal potable water is generated. However, this thesis deviates from the existing literature by considering the value of the water source and incorporating it into the known tariff before defining the demand curve. This adjustment in the value of the tariff is based on the scarcity rent method proposed by Moncur and Pollock (1988).

2.4 Methodological part 4

The final methodological part (Chapter 7) focuses on identifying various technological actions or technological decisions that can be adopted to achieve success within a specified timeframe. The notion of success in this thesis pertains to maintaining low treatment costs while simultaneously minimizing the opportunity cost of watershed conservation. To address this multiobjective optimization problem, the ε-Constraint method is employed. This technique involves selecting one objective function for optimization while converting the other objective into a constraint by setting an upper bound. Consequently, the problem is transformed into a scalar or mono-objective optimization problem (Haimes et al., 1971; Lin, 1976; Miettinen, 1999). The evolutionary algorithm model is implemented using an electronic spreadsheet, which generates a Pareto front that identifies non-dominated solutions. The decision-maker is then responsible for selecting the most suitable combination from a wide range of optimal solutions.

To assist decision-makers in situations with numerous alternatives, an approach called Dynamic Adaptive Policy Pathways (DAPP) is introduced (Haasnoot, 2013; Haasnoot et al., 2013; Lawrence et al., 2019). DAPP enables decision-making to occur over time and facilitates the development of robust and adaptive strategies based on unfolding future conditions. DAPP recognizes the concept of an Adaptation Tipping Point, which signifies the condition where the current treatment technology fails to achieve success. At this point, management actions are ineffective, and an alternative strategy becomes necessary. Since optimization serves as a valuable tool in supporting decision-making, the evolutionary algorithm is employed to assist the DAPP approach.

CHAPTER 3

3 THE AREA OF STUDY

3.1 General description

In order to implement the provided methodological framework outlined in Chapter 2, a decision was made to utilize five water treatment systems situated within the municipality of Caxias do Sul, located in the State of Rio Grande do Sul, Brazil (Figure 3.1). The selection of these systems was primarily influenced by the geomorphological characteristics of the municipality and the type of water supply infrastructure in place. Furthermore, the case of Caxias do Sul is particularly suitable for this study due to the presence of regulated land use practices and local legislation governing the five watersheds, in addition to the utilization of surface water and conventional treatment processes.

Caxias do Sul, with a Gross Domestic Product (GDP) of US$ 6.56 billion in 2017 (FEE, 2020), is the second largest economy in the state of Rio Grande do Sul. It encompasses a total area of 1,625 km². Geographically, it is situated on the geomorphological units known as the Planalto dos Campos Gerais and Serra Geral (IBGE, 1986). The region exhibits significant variations in altitude, with the northern portion adjacent to the Rio das Antas valley ranging from 290 to 430 meters, the southern portion in the Rio Caí valley at approximately 30 meters above mean sea level, and reaching altitudes close to 1,000 meters in the northeast region, which is part of Campos Gerais (Belladona and De Vargas, 2017). Due to these geomorphological conditions, the area lacks rivers capable of meeting the volumetric water requirements for supplying an urban population of over half a million people (IBGE, 2020b). Consequently, the municipality is dependent on damming small existing creeks to form reservoirs that regulate the flow.

The annual precipitation variability in Caxias do Sul, spanning the period between 1970 and 2007, ranges from 1,700 to 1,800 mm (Rossato, 2011). However, the distribution of rainfall in terms of both spatial and temporal patterns is highly heterogeneous (Belladona and De Vargas, 2017). The spatial variation of rainfall is particularly pronounced towards the west and southwest directions. For instance, in 2013, the total precipitation in the northeastern portion of the municipality was 579 mm less than that recorded in the southwest region, and this difference decreased to 251 mm in the subsequent year. The orography of the municipality is attributed as a contributing factor to this variation (Belladona and De Vargas, 2017). Unfortunately, it is worth noting that the site with the lowest precipitation levels is where the two largest watersheds, Faxinal and Marrecas, are situated. These watersheds are currently responsible for supplying raw water to nearly half a million inhabitants.

The annual variability of the total precipitation over Caxias do Sul, between 1970 and 2007, ranges from 1,700 to 1,800 mm (Rossato, 2011), however the spatial and temporal distribution of rainfall is very heterogeneous (Belladona and De Vargas, 2017). The spatial variation of rainfall is more prominent towards the west and southwest. In 2013, for example, the total precipitation in the northeast portion of the municipality was 579 mm less than that recorded in the southwest region and 251 mm less in the following year, a fact attributed to the orography of the municipality (Belladona and De Vargas, 2017). Unsuitably, the site with the lowest precipitation volumes is the one where the two largest watersheds are located in (Faxinal and Marrecas), currently responsible for supplying raw water to almost half a million inhabitants.

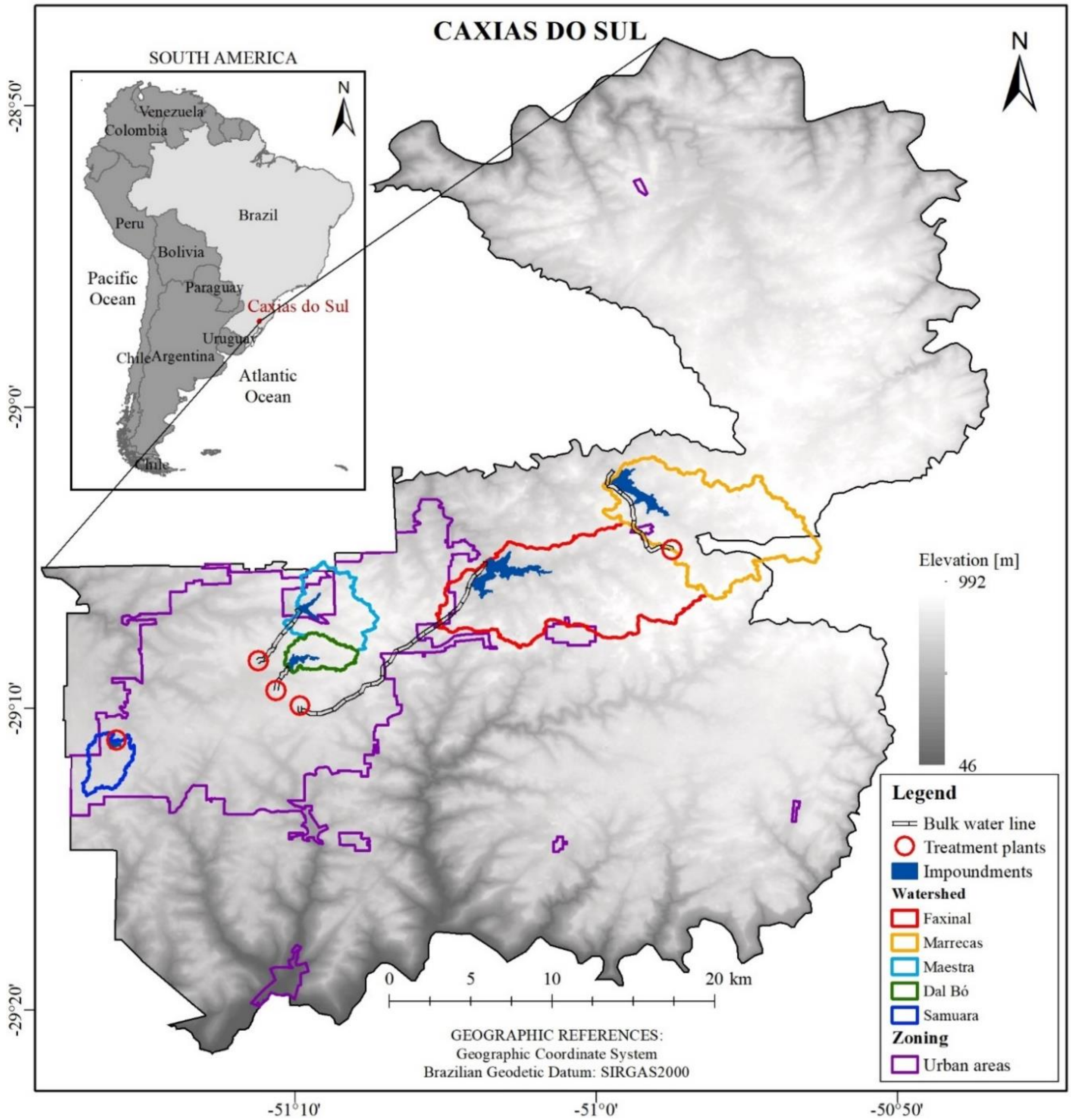

Figure 3.1. The area of study.

The natural pedological conditions of the watersheds in the area are characterized by soils with moderate to low infiltration rates. These soils have a moderately fine to fine texture and display a moderate to slow rate of water transmission (Dal Bosco et al., 2020; Flores et al., 2007). According to Mishra and Singh (2003), these soil characteristics classify them as belonging to the B and C hydrologic soil groups. These conditions hold significant importance in managing the hydrological state of the region, as these factors directly influence the water availability within the watersheds.

3.2 Water systems

Currently, the Public Water and Wastewater Service of Caxias do Sul (SAMAE) operates five surface water treatment systems in the municipality. These systems cater to 99% of the population, providing them with drinking water. However, some residents in rural areas rely on private or community wells to fulfill their water requirements (SAMAE, 2022a). Due to its geographical location and geomorphology, Caxias do Sul depends on the supply of water from a collection of relatively small reservoirs created by damming small creeks. Each reservoir serves a local water treatment plant, operating independently. Consequently, each treatment system consists of the following components: (1) the watershed, (2) the respective lake formed by damming the main creek, (3) the pipeline, and (4) the drinking water treatment plant (DWTP), while only during distribution can the same city area receive water from one or more systems, depending on the sectioning operations. However, the reservation systems for drinking water and the distribution network are outside the scope of this study and will not be considered in this thesis. Figure 3.1 illustrates each watershed across the municipality, indicating the pipeline and the location of the respective DWTP.

The capacity of each system is directly influenced by the water availability within each watershed and the flow regulation of the respective reservoirs (Table 3.1). The volumes of each reservoir exhibit significant variation, primarily due to the size of the watershed. This, in turn, is associated with the construction date of the dam and the population served at that time. Therefore, the variation in reservoir volumes can be attributed to the historical urban expansion of the city. The Dal Bó system was constructed in the 1920s when the watershed was situated far from the urban periphery. However, as the population increased and the city expanded, the area became incorporated into the urban region. A similar scenario unfolded with the Samuara system, which was built in the 1950s, and the Maestra system, constructed in the 1970s. In response to the municipality's rapid economic and population growth during the 1990s, the Faxinal system was established. More recently, in the 2010s, the Marrecas system was built (SAMAE, 2022b). Both the Faxinal and Marrecas systems remain predominantly situated in the rural area of the municipality.

Table 3.1. Conventional DWTP systems in Caxias do Sul.

DWTP System	*Watershed Area [ha]	Zoning	*Reservoir Area [ha]	*Reservoir Volume [hm³]	**DWTP Capacity [$L \cdot s^{-1}$]	Population Served
Faxinal	6,679	Rural	248	24.8	813 / 1,100	270,130
Marrecas	5,325	Rural	201	29.9	253 / 760	111,900
Maestra	1,526	Urban	49	5	219 / 325	96,800
Dal Bó	643	Urban	46	1.9	63 / 80	20,280
Samuara	687	Urban	19	0.7	22 / 38	6,260

* Source: (SAMAE, 2022b)
** Capacity: current treatment flow / full treatment capacity

Within the Faxinal system, the transportation of raw water to the DWTP involves the utilization of two pipelines. One pipeline has a diameter of 500 mm, while the other has a diameter of 700 mm. As for the DWTP within the Dal Bó system, being the oldest among the systems mentioned, it receives its water supply via a 350 mm pipeline. Similarly, the Maestra system receives water through a 600 mm pipeline. Conversely, in the specific scenario of the Marrecas system, the treatment facility is situated in a rural area. Subsequently, the treated water is conveyed to the urban area through a pipeline with a diameter of 1000 mm, utilizing gravity as the sole driving force. All of these pipelines are constructed using ductile iron pipes. In contrast to the previously discussed systems, the Samuara system benefits from the natural gravitational flow, as the reservoir is conveniently situated immediately upstream of the DWTP.

While water bodies in Brazil can fall under either state or federal jurisdiction (refer to Marques (2022) for detailed information), the responsibility for regulating land use, zoning, and environmental licensing policies primarily lies with the respective municipalities. These regulations play a crucial role in implementing watershed conservation measures aimed at restoring and preserving the water's quantity and quality.

The management of these five watersheds involves imposing restrictions on land use. A local legislation, known as the Water Law (Caxias do Sul, 2005), governs the permissible occupation and economic activities within these areas. It designates different zones with varying levels of vulnerability and emphasizes in-situ water conservation, both in terms of quantity and quality. Preserving water quantity contributes to ensuring water security, while maintaining water quality directly impacts treatment costs and tariffs, which also plays an important role in water security.

The Water Law is aligned with certain provisions of the Brazilian Forest Law (Brasil, 2012). These legal instruments establish the requirement for designated buffer zones around perennial and intermittent natural water courses. The buffer varies based on the water course's width, with courses wider than 10 meters necessitating a buffer zone of 50 meters, while narrower courses require a buffer zone of 30 meters. Additionally, buffer zones of 50 meters are designated around springs and marshes, and a buffer zone of 30 meters is established around artificial or natural ponds. Artificial reservoirs that form part of the water systems are assigned a buffer zone of 100 meters. Moreover, hilly areas with slopes exceeding 45° are also mandated to be preserved. Within these designated conservation zones, human activities are severely restricted, and only activities with minimal environmental impact are permitted.

Apart from the imposed restrictions on the buffering zones, certain economic activities are prohibited within the watersheds of Caxias do Sul, specifically industries and some service activities with high pollution potential. Furthermore, urbanization within these areas differs from the rest of the city. In general, the minimum lot size for urban development in Caxias do Sul is 300 m^2 (Caxias do Sul, 2007). However, within the watersheds falling under urban zoning, the lot sizes are required to be 1,000, 2,500, or 10,000 m^2, depending on the hydrogeological vulnerability of the specific location (Caxias do Sul, 2005). These regulations inevitably lead to additional costs for landowners and residents situated within these areas, as compared to those outside the watersheds. Consequently, the conservation measures impose an opportunity cost on these areas.

CHAPTER 4

4 THE SHORT-TERM BLURRING EFFECT ON THE LONG-TERM URBAN WATER SECURITY[1]

ABSTRACT

The importance of proper land use management in preventing additional treatment costs is widely recognized. However, in the short-term, the cost savings resulting from such measures may be relatively modest, failing to signal the need for improved land and water management in the long-term. Limited research currently exists that comprehensively examines how water treatment costs are influenced by water quality using detailed field data, particularly in relation to short-term and long-term planning. The objective of this Chapter is to utilize available information to illustrate that the immediate benefits of improved water quality in treatment processes can blur the perception of the long-term treatment costs. The research findings indicate that a marginal reduction of just 0.0002 USD·m^{-3} in treatment costs is observed with a one-unit improvement in the water quality index. However, over the long-term, the cumulative impact of water quality deterioration may necessitate the adoption of more advanced treatment methods, potentially leading to an increase in treatment costs ranging from 83% to 242%.

[1] This Chapter will be published as *Impacts of changing raw water quality to treatment costs: can urban water security be compromised in the long-term by short-term decisions on land use management?*

4.1 Introduction

Land use intensification and urban waste have contributed to add several allochthonous constituents to water. Additionally, since the first confirmed waterborne disease outbreaks in the mid-nineteenth century (Arlinghaus and Nystuen, 1988), water quality regulations have become more rigorous and the general public has become both more knowledgeable and more discriminating about water quality (Crittenden et al., 2012). This interconnectedness between source water quality and potable water standards has since created concern about whether the treatment technology can guarantee safe water without imposing an undue burden on dwellers with additional water tariff.

Since the 1970s, the number of regulated contaminants has experienced a significant and continuous increase (Howe et al., 2012). This trend is also evident in Brazil (Brasil, 2021a, 2011, 2004, 1990, 1977), where the majority of drinking water treatment plants (DWTPs) employ conventional treatment methods. Libânio (2010) further suggests that international regulatory trends indicate a growing focus on new organic and inorganic contaminants, as well as disinfection by-products (DBPs), which is expected to result in future standards incorporating a larger number of parameters and stricter maximum contaminant levels.

It should be noted that conventional DWTPs are not designed to effectively remove synthetic organic contaminants, such as pesticides and herbicides, as well as substances causing taste and odor issues (Elder and Budd, 2011). Moreover, conventional treatment processes are generally not effective in removing cyanotoxins or DBPs precursors (Davis, 2010; Summers et al., 2011). To address the removal of these constituents, the application of powdered activated carbon (PAC) as an adsorbent has shown promising results in enhancing the efficiency of conventional treatment methods (Teixeira et al., 2020; Walker, 2015). Furthermore, conventional DWTPs often encounter challenges in minimizing odor-related issues, particularly during the summer months (Bruce et al., 2002; McGuire et al., 1981).

Source water protection plays a crucial role in preventing the need for additional chemical treatments and technological interventions. It is widely acknowledged that the higher the quality of the raw water, the simpler and more cost-effective the treatment process becomes (Crittenden et al., 2012), resulting in lower water tariffs. Although this general trend is recognized, there is still a knowledge gap regarding the precise extent of the impact that improved raw water quality can have on treatment costs. Understanding this impact is essential in making informed decisions about land and water allocation to ensure water quality and avoid future increases in treatment costs.

As land and water allocation is adjusted to maintain and improve water quality, it often involves implementing land management practices, reducing water usage, and restricting occupation in certain watershed areas to mitigate both point and non-point source pollution. However, in doing so, some of the economic benefits derived from land and water use are foregone, particularly in the short and mid-terms. The challenge arises when the benefits associated with improved water quality and reduced treatment costs appear relatively small within this short-term timeframe, leading to a blurred perception of the long-term outcomes. In the short-term, the economic gains forgone due to watershed protection outweigh the immediate savings in water treatment costs, which can result in suboptimal decision-making concerning the water quality. This can lead to increased future risks and compromise long-term water security. Loucks and van Beek (2017) explain that decision-makers rely on their own beliefs, qualitative facts, and quantitative data provided by mathematical or computer-based models to anticipate the outcomes of their decisions. However, if the interpretation of model results is limited by a focus on short-term information, decisions that may seem suitable for addressing immediate issues may have unintended consequences in the long-term.

Several contributions in the field have quantified the avoided water treatment costs when source water protection policies are put into practice (Dearmont et al., 1998; Heberling et al., 2015; Price and Heberling, 2018; Warziniack et al., 2017). For instance, Price and Heberling (2018) studied the relationship between treatment costs and source water quality and discovered that marginal changes in water quality at the watershed result in modest gains in avoided treatment costs. Price et al. (2017) further emphasize the challenge of assessing the impact of increased treatment costs on water prices and consumers' welfare. However, the existing literature in this field often fails to explore the long-term implications and consequences of prioritizing short-term cost savings, which is crucial for avoiding suboptimal decision-making.

This Chapter aims to address this knowledge gap by conducting a comprehensive analysis of the potential impacts of enhancing water quality on both short-term and long-term water treatment costs. Additionally, it demonstrates how seemingly insignificant short-term benefits and decisions regarding watershed conservation can impact the perception of the long-term outcomes. To describe this phenomenon, the term "blurring effect" is introduced, which is described as a reduction of visual resolution immediately following defocus (Jain et al., 2012; Smith, 1998). In the context of water resources management, the blurring effect can be understood as data, analysis, or circumstances that misguide short-term decisions and lead to detrimental long-term consequences.

The primary objective of this Chapter is to estimate the long-term economic losses in terms of incremental operational and maintenance (O&M) costs for DWTPs resulting from a scenario of degradation in watershed quality, which may lead to increased treatment costs. It is quantified the chemical costs associated with five conventional DWTP and correlate them with the corresponding watershed quality, which is expressed by the water quality index introduced by CCME (2001). Subsequently, it is estimated the future additional O&M cost for a scenario where the conventional treatment alone proves insufficient to ensure safe drinking water, necessitating the incorporation of advanced membrane treatment in the existing treatment train. Finally, it is integrated all economic losses, additional costs, and expenditures to illustrate the blurring effect. The findings and methodologies presented in this Chapter can provide valuable support for integrating sanitation and water supply planning into broader water resources management at the watershed scale, with the aim of safeguarding urban water supply sources. By highlighting the blurring effect, this study underscores the significance of supporting and prioritizing current water and land allocation decisions from a long-term perspective, rather than focusing solely on short-term economic gains.

4.2 Methodological framework

The approach employed in this Chapter encompasses three key steps. The first step involves quantifying the water availability within the watershed to determine whether water quantity poses a significant concern to urban water security. Then, a water quality index is established to assess the existing water quality status of each watershed and establish a correlation with the corresponding chemical costs incurred by the DWTPs. Finally, to demonstrate the potential long-term economic losses resulting from the degradation of watershed quality, this Chapter estimates the costs associated with implementing advanced membrane treatment as a supplementary post-treatment to the conventional treatment. This estimation enables an assessment of the incremental O&M costs attributed to varying water quality conditions.

4.2.1 Area of study

Chapter 3 of the thesis offers a detailed and comprehensive overview of the study area. For a more in-depth understanding of the area, it is recommended to referring to the aforesaid Chapter. Additionally, Figure 4.1 provides a visual representation of the geographical location of the study area. It is important to note that Figure 4.1 differs from Figure 3.1 as it incorporates the identification of two additional watersheds. These additional watersheds are instrumental in providing hydrological data required for quantifying the water availability within the remaining five watersheds.

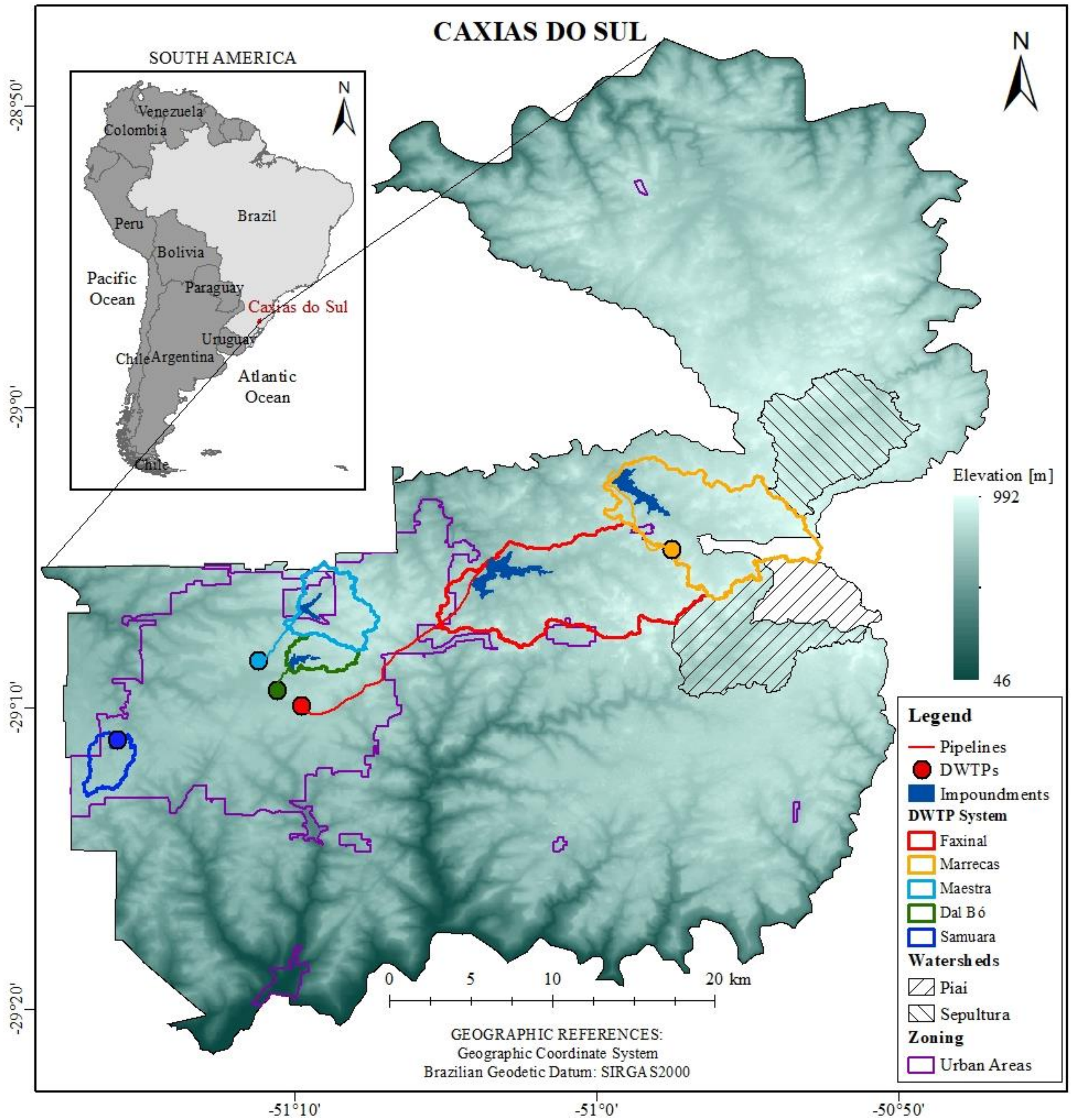

Figure 4.1. Area of study with the Piaí and Sepultura watersheds.

4.2.2 Source water availability

The estimation of water availability for the rural and non-urbanized areas of each watershed involved employing rainfall-runoff hydrological modeling based on the hierarchical testing scheme. The Proxy-basin Differential Split-sample Test, as proposed by Klemes (1986), was used in this study. This test is applicable when the transposability of the model's data is feasible, specifically in cases where similar edaphoclimatic attributes are observed among the watersheds.

To implement this testing scheme, two gauged watersheds, labeled A and B, were selected. These two gauged watersheds exhibit characteristics similar to those of the ungauged watershed, labeled C. Additionally, segments representing different climatic conditions, namely wet (ω) and dry (δ), were identified. The modeling process involved a series of steps to assess the impact of a dry climate scenario.

Firstly, the hydrological model was calibrated using data from watershed A under wet climate conditions (Aω), and then validated using data from watershed B under dry climate conditions (Bδ). Subsequently, the model was calibrated using data from watershed B under wet climate conditions (Bω) and validated using data from watershed A under dry climate conditions (Aδ), or vice versa. The adequacy of the model was evaluated by comparing the errors in both validation runs, Aδ and Bδ, to ensure they were acceptable and not significantly different (Klemes, 1986).

The performance of the hydrological model was assessed using two objective functions. The root mean square error (RMSE) was minimized to measure the goodness-of-fit for the computed outflow. Whereas the goodness-to-fit statistic was maximized by applying the Nash-Sutcliffe objective function.

To model the rainfall-runoff process in the urbanized areas of the watersheds, the calibrated and validated parameters as presented by Missiaggia (2018) were used. These parameters, which have undergone calibration and validation processes specific to the urbanized areas, were employed in running the rainfall-runoff model. Table 4.1 provides a comprehensive summary of information pertaining to the level of instrumentation and the sources of the modeling parameters for the five watersheds. This table outlines the specific details regarding the availability of data and the origin of the parameters used in the modeling process for each watershed.

Table 4.1. Parameter applied in the rainfall-runoff modelling.

Watershed	Stream Gauge	*Parameter	Use in the Model
A - Sepultura	gauged	-	Calibration/Validation
B - Piaí	gauged	-	Calibration/Validation
C - Faxinal	ungauged	1	Modelling
C - Marrecas	ungauged	1	Modelling
C - Maestra	ungauged	1 and 2	Modelling
C - Dal Bó	ungauged	1 and 2	Modelling
C - Samuara	ungauged	1 and 2	Modelling

*Calibrated and validated parameters used: 1 from Klemes (1986) for rural-like areas, and 2 from Missiaggia (2018) for urban-like areas. The last column indicates the role of each watershed for determining the discharges.

The rainfall-runoff model employed in this study was based on the SCS Curve Number Loss Model, as described by Mishra and Singh (2003). To facilitate the modeling process, the HEC-HMS

4.3 software (USACE, 2018) was used. It is important to note that the effect of reservoir weir flow regularization was not taken into consideration during the modeling process. Therefore, the computed discharges only represent the direct natural outflow, without accounting for any flow regulation by reservoir weirs.

4.2.3 Water compliance index (WCI) and DWTP chemical costs

Given the common occurrence of turbidity in treatment facilities, the assessment of water quality typically revolves around this specific parameter. However, relying only on turbidity fails to provide a comprehensive representation of the overall source water quality or accurately reflect the direct treatment costs. This is because the addition of polymer during conventional treatment's coagulation process does not exhibit a linear relationship with nephelometric units (NTU) (Crittenden et al., 2012). To circumvent this issue, it is suggested to employ an index as a surrogate for assessing both raw water quality and land use.

Water quality indexes serve the purpose of summarizing complex data related to water quality, thereby reducing processing time and enhancing comprehension for both administrative and general audiences (CCME, 2001; Ho et al., 2019). The approach proposed by CCME (2001) employs commonly monitored data available at water utilities, consisting of three factors and a minimum of four variables that are sampled at least four times within a specified period. In this study, the quality objective considered meeting (being in compliance with) is the attainment of Level 2 of the Brazilian water quality guidelines regulation (Brasil, 2005). Therefore, we refer to this index as the water compliance index (WCI). The WCI is assigned a range from 0 (indicating the poorest quality) to 100 (representing the highest quality).

To compose the water compliance index (WCI), a selection of six variables was made, which are commonly found in conventional water plants and encompass biological, chemical, and physical pollutants. These variables include dissolved oxygen (DO), cyanobacteria, turbidity, manganese, nitrite, and pH. The presence of DO and manganese can impact the taste and odor of water, while cyanobacteria have the ability to produce biologically active substances with high toxicity (Goncharuk, 2014). Turbidity levels exceeding 4 NTUs render water visibly unacceptable for consumption (Pandit and Kumar, 2019). Although pH typically does not directly affect consumers, it is preferable for effective water clarification and disinfection that the pH remains below 8 (WHO, 2017). Nitrite, on the other hand, poses a health risk in drinking water, potentially causing disorders such as methemoglobinemia, which can lead to cyanosis (Semitsoglou-Tsiapou et al., 2016). Further information on the background and detailed calculations for constructing this index can be found in CCME (2001).

Pearson's correlation coefficient, which measures the strength of linear relationship between variables (Frost, 2019), was utilized to assess the correlation between the WCI and the annual chemical costs of the five DWTP systems. In this analysis, the costs associated with electricity and labor were not included, given their relationship with watershed degradation and water quality also depends on other factors, for which there was not enough data available. The relationship between the two variables was elucidated through the application of the least-squares regression line ($f(x) = \alpha \cdot x + \beta$) (Spiegel and Stephens, 2018). Table 4.2 presents the conventional guidelines typically used to interpret the Pearson's correlation coefficients (R).

Table 4.2. Conventional interpreting approach of the Pearson's correlation coefficients.

Coefficients		Interpretation
-1	+1	Perfect relationship
-0.91 to -0.99	0.91 to 0.99	Very strong correlation
-0.71 to -0.90	0.71 to 0.90	Strong correlation
-0.41 to -0.70	0.41 to 0.70	Moderate correlation
-0.11 to -0.40	0.11 to 0.40	Weak correlation
0 to -0.10	0 to 0.10	Negligible relationship

Adapted from (Akoglu, 2018; Frost, 2019; Schober et al., 2018).

4.2.4 Estimating membrane O&M treatment costs

The estimation of the mean O&M cost for an advanced treatment utilizing membrane filtration is based on information sourced from relevant literature. The selection of studies included in our database followed *ex ante* criteria: (1) the studies should present cost results of two or more plants with distinct flow capacities, and (2) the studies should incorporate data from real-scale plants. Prior to conducting the analysis, all costs and indexes were adjusted to the December 2019 value in United States Dollars (USD), utilizing the Consumer Price Index (CPI) provided by the Bureau of Labor Statistics (BLS, 2020).

O&M cost curves were developed for membrane advanced treatment for each type of process (microfiltration, ultrafiltration, nanofiltration, and reverse osmosis) and for different water sources, including sea water, brackish water, reclaimed water, ground water, and surface water. By using the literature data and considering the non-linear nature of the distribution, it was applied a non-linear regression approach based on the power law model $f(x) = a \cdot x^b$, where coefficients a and b represent the parameters of the model (Gallant, 1975; Rhinerhart, 2016). It was utilized the respective equations derived from this model to quantify the O&M cost for membrane advanced treatment across various plant capacities. Finally, it was performed an exploratory data analysis, followed by the construction of box-and-whisker plots as suggested by Tukey (1977).

4.3 Data

The data for this particular segment of the study primarily originated from SAMAE. O&M treatment costs were obtained from comprehensive internal records maintained by the utility spanning the period from 2011 to 2019. In addition, various other data sources were utilized, including precipitation data, observed stream discharges, water quality information, and treated volume data, which were collected in collaboration with the municipal utility. Furthermore, the O&M treatment cost data specific to membrane advanced treatment were derived from relevant literature sources.

4.3.1 O&M treatment cost data

O&M treatment expenses encompass both fixed and variable costs. Fixed costs typically involve labor and administration, whilst variable costs are associated with chemicals, electricity, repairs, and other supplies and services necessary for the operation of the treatment plants (McGivney and Kawamura, 2008). In this particular thesis segment, the analysis of costs spans from the watershed to the treatment process. Therefore, the definition of O&M costs is limited to the cumulative expenses related to labor, chemicals, and electricity. Distribution, administration, and opportunity costs of the systems were not taken into account for the purposes of this study. Additionally, capital costs were excluded, as the objective of this study focuses specifically on incremental O&M costs in relation to variable water quality.

The fixed costs comprised wages for engineers, technicians, watershed control personnel, and operators. On the other hand, the variable costs included the expenses for chemicals and electricity, including conveyance and pumping costs from the impoundment to the DWTP, as well as the costs associated with the treatment process itself. Figure 4.2 depicts the average O&M costs for the period from January 2011 to December 2019. All costs are presented in United States Dollars (USD), and the historical costs have been adjusted to December 2019 based upon the IPCA (IBGE, 2020c), and the exchange-rate was considered 1USD:4.10BRL according to (BCB, 2020).

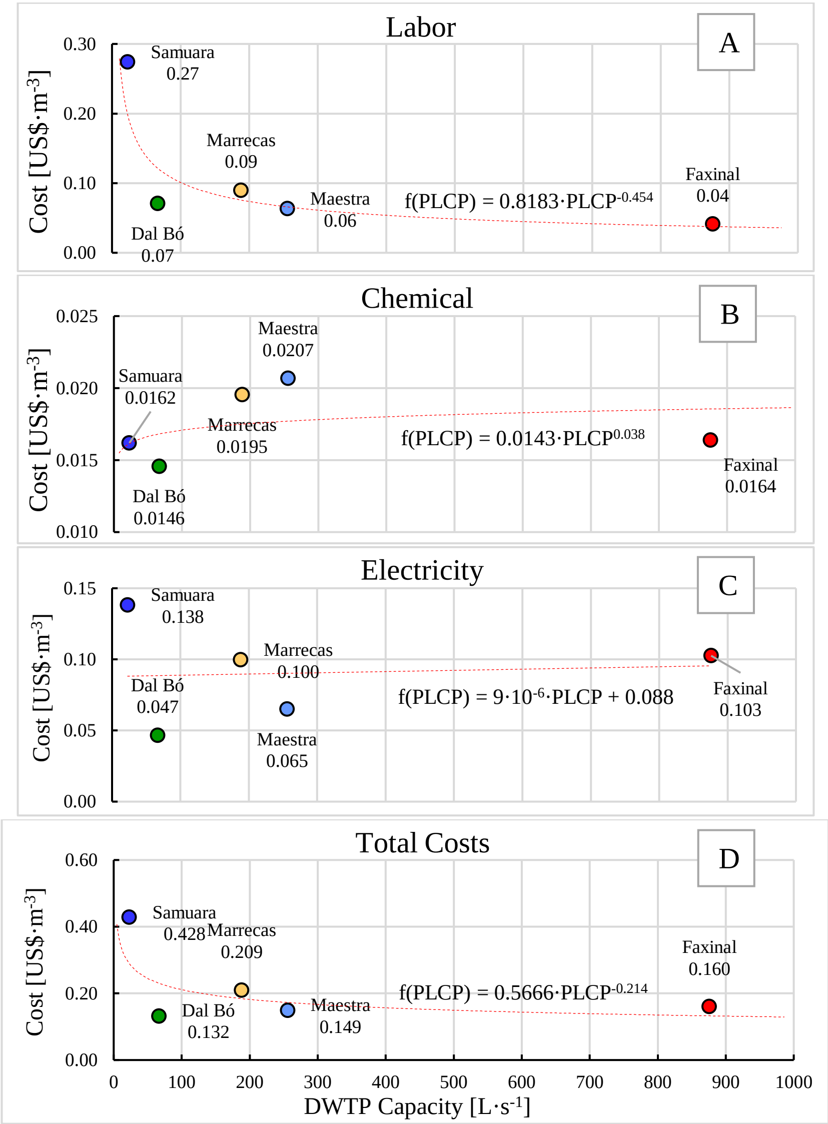

Figure 4.2. O&M costs of each studied DWTP.

Figure 4.2A shows the mean cost of labor of the personnel involved in the system, from watershed to treatment. Figure 4.2B presents the mean expenses with chemicals used in the plant. Figure 4.2C reveals the mean energy costs involved with conveyance and treatment. Figure 4.2D shows the mean total cost with O&M. The dashed red line represents the most suitable regression curve and the correspondent regression equation is a function of the plant capacity (PLCP), expressed in liters per second.

4.3.2 Precipitation and stream discharge data

A hydrology model was employed to estimate the streamflow for each watershed. Local data specific to the 2011-2020 timeframe were utilized for the hydrological modeling. Precipitation data were collected from automatic rain gauges strategically distributed across the municipality, as documented by Belladona and De Vargas (2017). To model the discharge of the Samuara watershed, data from rain gauge 2 had to be used since rain gauge 8 commenced its operation only in 2012. Observed stream discharge data were acquired by employing water level sensors programmed to record information on an hourly basis.

4.3.3 Water quality data

To comply with environmental and water quality regulations, regular testing of raw water quality is necessary. Sampling is conducted at the water inlet tower of the reservoir, prior to the pumping station, as part of routine monitoring procedures. However, only one analysis per month is required for each system to establish the WCI as per the regulations.

4.3.4 Membrane O&M treatment cost data

The design of the treatment train is significantly influenced by the quality requirements of both raw water and potable water (McGivney and Kawamura, 2008). However, the understanding of the interaction between membrane properties and water quality is generally limited (Alspach et al., 2008). In the database used, it has been compiled the costs associated with treating various water sources, including sea water, brackish water, reclamation water, ground water, and surface water. Additionally, we have incorporated data on the most commonly used filtration systems, namely microfiltration (MF), ultrafiltration (UF), nanofiltration (NF), and reverse osmosis (RO) (Zoubeik et al., 2018).

4.3.5 DWTP treated flow data

Data on the monthly volumes of treated water were collected for the period from January 2011 to December 2019. This information was obtained from the local utility and pertains to each individual DWTP within the study area, including Faxinal, Marrecas, Maestra, Dal Bó, and Samuara.

4.4 Results and discussions

4.4.1 Quantity: water availability and urban water security

The hydrological model underwent calibration during a dry period at the Piaí Watershed (Bδ) and subsequently underwent validation during a wet period at Sepultura (Aω). During the calibration phase (Bδ), the model's performance was evaluated using the Nash-Sutcliffe efficiency (0.76), percent bias (-0.10%), and the root mean square error (RMSE) value of 0.48. These performance metrics fell within the "very good" range, as defined by Moriasi et al. (2007). In contrast, the validation phase (Aω) yielded slightly lower performance ratings, although they still fell within the "good" range. The Nash-Sutcliffe efficiency was 0.65, and the RMSE was 0.59.

However, the percent bias was -39.95%, which is considered "unsatisfactory." This discrepancy can be attributed to the fact that percent bias values for streamflow tend to exhibit greater variation during dry years compared to wet years, especially when employing a split-sample evaluation approach involving separate calibration and validation periods (Gupta et al., 1999; Moriasi et al., 2007). The urban-like areas of the model were also assessed using the Nash-Sutcliffe efficiency, resulting in values of 0.83 and 0.70 for the calibration and validation phases, respectively (Missiaggia, 2018).

Figure 4.3 illustrates the flow duration curves for the analyzed watersheds, accompanied by the volume of water treated at each DWTP. A notable observation from the right-hand side of the figure is the consistent pattern observed in the treated flow. Specifically, the Faxinal and Maestra systems experienced a considerable decline in water supply after 2015, coinciding with the initiation of the new Marrecas system. This decline in supply for the Faxinal and Maestra systems can be attributed to the ongoing expansion of distribution made possible by the implementation of the Marrecas system. The expansion resulted in an increased production capacity, ultimately affecting the water supply from the Faxinal and Maestra systems.

The historical data of the treatment flow, as depicted in Figures 4.3B, 4.3D, and 4.3E, indicates that three DWTP systems (Marrecas, Dal Bó, and Samuara) have maintained production levels below the watershed capacity of the Q_{50} discharge, which corresponds the median flow (Chowdhury and Eslamian, 2014; Davie and Quinn, 2019). This information suggests that the urban areas supplied by these systems have not faced significant threats in terms of water quantity. On the other hand, for the Maestra and Faxinal watersheds (Figures 4.3A and 4.3C), which account for approximately 75% of the total water consumption in the city of Caxias do Sul, withdrawals exceed the Q_{50} discharge but remain below the Q_{45} threshold. These values allow the inference that water quantity is not an issue that pose a significant threat to the urban water security in the short-term, but at the same time, not leaving the city in a comfortable position.

The current situation, where there is no immediate perception of water shortage risk in the short-term, can have a detrimental effect on urban water security in the long-term. When water supply is limited to meet only the current demand, there is a diminished incentive for public or private water utilities to invest in new political, technological, or economic initiatives. In the public sphere, where decisions often prioritize short-term goals due to government mandate periods, and citizens are not directly experiencing immediate consequences (González-Gómez et al., 2011), urban water security tends to focus on satisfying the demands of the general voting population. Whereas, in the private sector, new investments are discouraged under this scenario as quantity is not considered inefficient, and financial resources are channeled to reduce inefficiencies (Embid-Irujo, 2005). The key concern is that urban planners may not adequately consider the long-term implications, given their decisions are based on current quantitative data set in an environment full of uncertainties and unpredictable socioeconomic changing conditions.

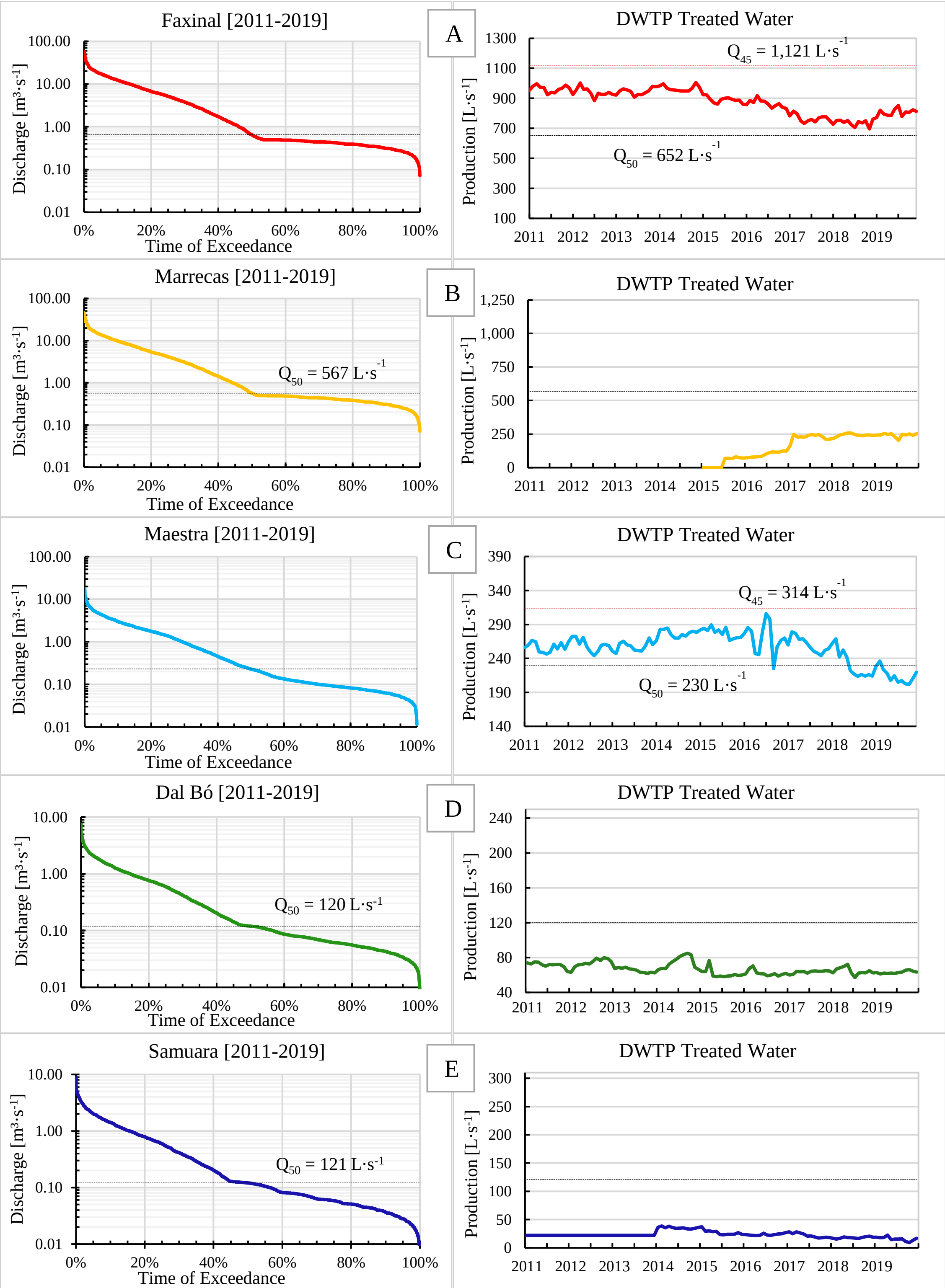

Figure 4.3. Flow duration curve and DWTPs production from Jan/2011 to Dec/2019.

4.4.2 Water quality and treatment costs

The relationship between the WCI and chemical costs varied among the five systems, as shown in Figure 4.4. The decision to increase the use of treatment chemicals, such as PAC, is affected not only by the quality of the raw water, which is assessed by the WCI index, but also by the perceived risk from the perspective of the public utility. When a particular watershed is considered to have a higher risk of fluctuating raw water quality, treatment is enhanced by employing more chemicals, resulting in increased costs. This approach is taken to avoid potential penalties, such as fines, that may be imposed for failing to meet water quality standards.

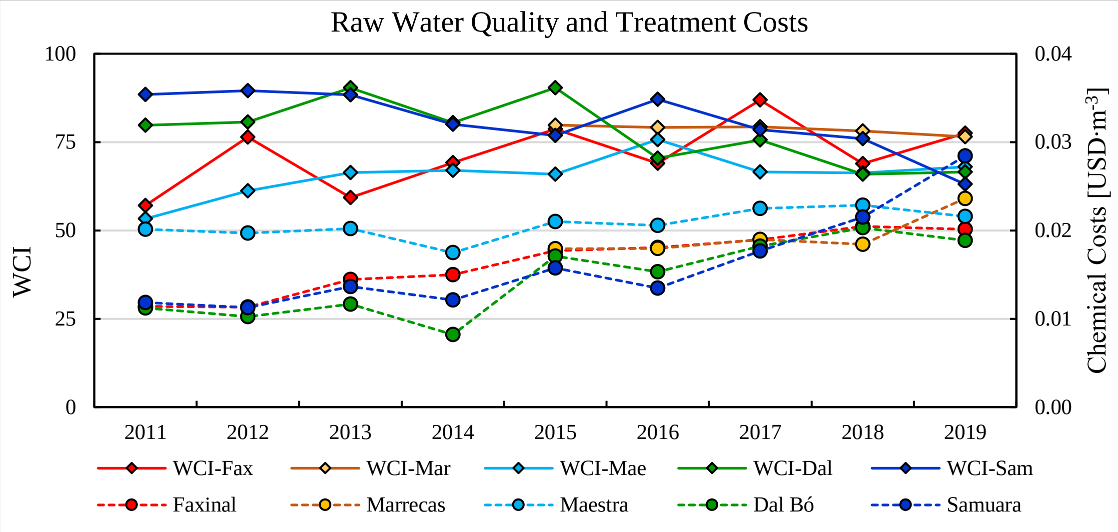

Figure 4.4. WCI and chemical costs behavior for each DWTP system.
The dashed line indicates the behavior of chemical costs throughout the studied period, whilst the continuous line shows how the WCI varied between Jan/2011 and Dec/2019.

This perception of risk is particularly evident in the case of the Faxinal system. Over the course of nine years, the Faxinal system experienced an overall increase in WCI. However, costs also consistently increased due to the additional usage of chemicals, primarily PAC, driven by concerns about water quality. The Maestra system, on the other hand, consistently had the lowest WCI and the highest costs, but the values remained relatively constant during this period.

In contrast, the Samuara system witnessed a significant reduction in WCI from 2016 to 2019, which coincided with a doubling of expenses on chemicals. This suggests that the Samuara system, with its small impoundment and watershed, is more vulnerable to pollution. As for the Dal Bó system, a clear trend was observed where WCI decreased from 79.7 in 2011 to 66.5 in 2019, indicating a deterioration in watershed quality. Consequently, chemical costs increased from 0.011 to 0.019 USD·m^{-3}. Due to the short working period of the Marrecas system, further insights are limited. The Marrecas system's WCI has not changed significantly since 2015, decreasing slightly from 79.8 to 76.5. However, chemical costs have increased from 0.018 to 0.023 USD·m^{-3}. This can be attributed to the expansion of production and the perceived risk regarding the quality of the raw water.

Table 4.3 provides insights into the relationship between the WCI and chemical costs for the Faxinal, Marrecas, Dal Bó, and Samuara systems. The table reveals that this relationship is significant, although not always negative, as indicated by the sign of α, which represents the rate of change in chemical costs as the WCI varies. In the case of the Faxinal system, a positive correlation is observed, meaning that chemical costs increased as the WCI improved. However, this relationship exhibits high variability over the studied period, while maintaining a moderate correlation, as denoted by the R value of 0.488. For the Maestra system, there is slight evidence of a relationship between chemical usage and watershed quality. Although the correlation is weak (R=0.135), the low p-value confirms the existence of a relationship between these variables.

Table 4.3. Least-squares equation constants and the Pearson's correlation coefficients.

System	α	β	R	*p-value
Faxinal	0.0002	0.0036	0.488	<0.0001
Marrecas	-0.0017	0.1491	0.896	<0.0001
Maestra	0.00004	0.0183	0.135	<0.0001
Dal Bó	-0.0003	0.0347	0.542	<0.0001
Samuara	-0.0006	0.0646	0.914	<0.0001

*The p-values fall far below $1x10^{-4}$, indicating that the null hypothesis (H_0) was rejected. The H_0 (p-value = 0) considers that there is no relation between the dependent and the independent variables (Frost, 2019). The smaller the p-value the stronger the sample evidence is for rejecting H_0. Least-squares equation corresponds to $f(x) = \alpha{\cdot}x+\beta$.

The results indicate an inverse correlation between the WCI and chemical costs for the Marrecas, Dal Bó, and Samuara systems. This suggests that a decrease in the WCI, indicating poorer water or watershed quality, is associated with an increase in chemical costs. On the other hand, an increase in the WCI, reflecting better water or watershed quality, is associated with lower chemical costs. The Pearson's coefficient further confirms the strength of the relationship between the variables. There is a strong relationship between the WCI and chemical costs in the Marrecas and Samuara systems, while the Dal Bó system shows a moderate correlation between these variables. The negative sign of the α coefficient reinforces the inverse relationship, indicating that as water or watershed quality deteriorates (lower WCI values), the expenditure on chemicals at the treatment plants increases, or vice versa (higher WCI values correspond to less expensive treatment).

When jointly assessed (Figure 4.5), the five DWTP systems of Caxias do Sul indicate that there are potential risks to urban water security. The degradation of watersheds can result in higher water tariffs as more resources are required for the treatment process. The least-squares regression equation derived from the data is $y = -0.0002{\cdot}x + 0.0326$, indicating a moderate correlation (R=0.45) and a p-value far below 0.0001. This equation reveals that, in Caxias do Sul, there is a marginal change of just 0.0002 USD·m^{-3} in treatment costs for every one-unit change in the WCI. Similar patterns were observed in previous studies by Dearmont et al. (1998) and Freeman et al. (2007). Dearmont et al. (1998) estimated that a 1% reduction in turbidity would only lead to a 0.27% decrease in chemical costs (equivalent to 0.000083 USD·m^{-3}·year^{-1}, updated to December 2019) in their studied plant. Freeman et al. (2007) found that higher treatment costs were associated with lower water quality index, with chemical costs ranging from 0.0046 to 0.1274 USD·m^{-3}·year^{-1} (also updated to December 2019). This wide range of costs was attributed to variations in operation procedures, economies of scale, and regional pricing. Additionally, Abildtrup et al. (2013) estimated the economic value of the ecological service provided by forests in maintaining raw water quality, and their results align with the findings mentioned above. They found that a 1% increase in forest cover could potentially save 0.00904 USD·m^{-3} in treatment costs (value also updated to December 2019).

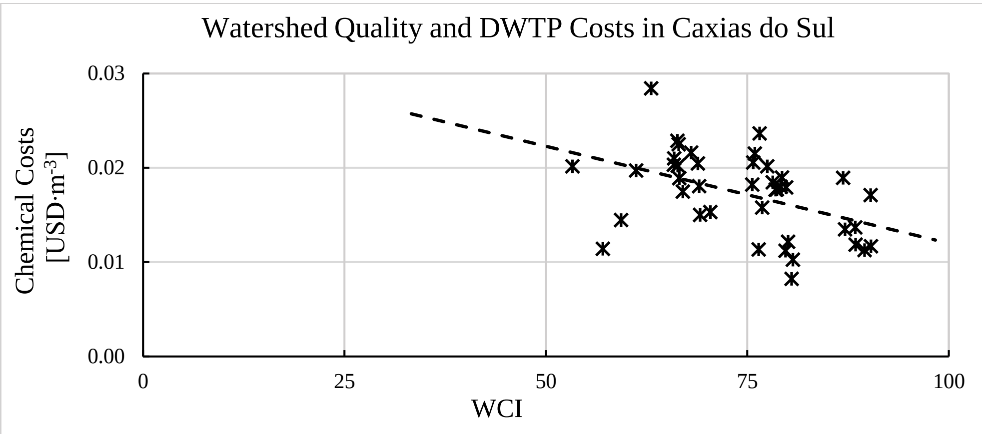

Figure 4.5. Correlation between the chemical costs and the WCI. for all DWTPs.

The minimal cost reduction associated with a one-unit improvement in the WCI may not appear attractive in the short-term, making efforts to preserve the watershed and improve raw water quality discouraging for both public and private utilities. However, it can cause a blurring effect on urban water security in the long-term. The conventional treatment train is capable of copying with only so much surface water degradation. It is not designed to effectively remove synthetic organic contaminants, cyanotoxins, DBP precursors, taste and odor compounds (Crittenden et al., 2012; Davis, 2010; Teixeira et al., 2020). Consequently, if watershed conservation is not prioritized in the short-term and raw water quality deteriorates to a level where the conventional treatment train fails to meet potability standards, alternative measures must be considered. In such scenario, two options emerge: either conveying water from a farther location with better water quality or incorporating advanced treatment techniques, such as membrane filtration, to supplement the conventional treatment. In both cases, costs are expected to increase.

4.4.3 Additional treatment costs with membrane filtration

To estimate the O&M expenses associated with membrane filtration, which can assist in conventional treatment under conditions of degraded watershed quality, it was conducted a comprehensive review of relevant literature to gather data on membrane filtration costs that met our specific criteria (refer to Figure 4.6). It should be noted that the compiled data includes various water sources beyond fresh surface water, as well as four distinct types of pressure-driven membranes. This is primarily due to a limited availability of literature focusing on membrane filtration at a real-scale level for the treatment of fresh surface water. Furthermore, as previously mentioned, there is a lack of comprehensive understanding regarding the relationship between membrane properties and water quality (Alspach et al., 2008). For reference, Table 4.4 presents the regression equations with all costs adjusted to December 2019.

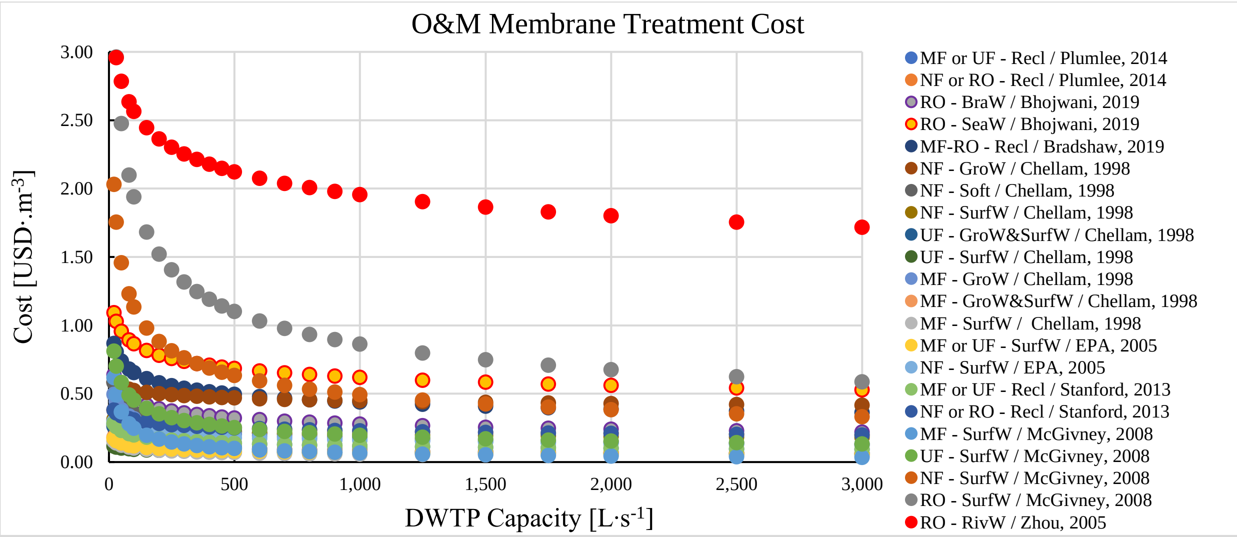

Figure 4.6. O&M membrane treatment costs from around the world.
The abscissa represents de DWTP production capacity and the ordinate shows the normalized O&M cost in terms of US dollar per cubic meter. In the legend, each abbreviation means: Reclamation water (Recl), Seawater (SeaW), Brackish water (BraW), Water softening process (Soft), Groundwater (GroW), Surface water (SurfW) and River water (RivW). Sources: (Bhojwani et al., 2019; Bradshaw et al., 2019; Chellam et al., 1998; EPA, 2005; McGivney and Kawamura, 2008; Plumlee et al., 2014; Stanford et al., 2013; Zhou and Tol, 2005).

The dependent variable represents the normalized cost, expressed as the amount spent per cubic meter of treated water, while the independent variable is the plant capacity (see Figure 4.6). To illustrate, the O&M cost for membrane advanced treatment in a DWTP with a capacity of 3,000 liters per second ranges from 0.04 to 1.72 USD per cubic meter. Similarly, for DWTPs with capacities of 30 and 1,000 liters per second, the O&M cost ranges from 0.11 to 2.96 USD per cubic meter and from 0.06 to 1.96 USD per cubic meter, respectively. McGivney and Kawamura (2008) state that actual O&M costs can vary significantly between two plants with the same flow and instrumentation, even within the same company.

Table 4.4. Membrane advanced O&M treatment costs curve regression equations.

Process	Water Source	*Equation [$USD \cdot m^{-3}$]	**Adapted from
MF	Ground and surface water	$0.335 \times PLCP^{-0.205}$	(Chellam et al., 1998)
MF	Groundwater	$1.140 \times PLCP^{-0.278}$	(Chellam et al., 1998)
MF	Surface water	$0.328 \times PLCP^{-0.251}$	(Chellam et al., 1998)
UF	Ground and surface water	$0.618 \times PLCP^{-0.287}$	(Chellam et al., 1998)
UF	Surface water	$0.175 \times PLCP^{-0.135}$	(Chellam et al., 1998)
NF	Groundwater	$0.717 \times PLCP^{-0.068}$	(Chellam et al., 1998)
NF	Surface water	$0.436 \times PLCP^{-0.111}$	(Chellam et al., 1998)
NF	Softening	$1.420 \times PLCP^{-0.297}$	(Chellam et al., 1998)
MF	Surface water	$0.346 \times PLCP^{-0.233}$	(EPA, 2005)
UF	Surface water	$0.346 \times PLCP^{-0.233}$	(EPA, 2005)
NF	Surface water	$0.477 \times PLCP^{-0.159}$	(EPA, 2005)
RO	Brackish water	$4.539 \times PLCP^{-0.051}$	(Zhou and Tol, 2005)
RO	River water	$4.418 \times PLCP^{-0.118}$	(Zhou and Tol, 2005)
MF	Surface water	$3.386 \times PLCP^{-0.568}$	(McGivney and Kawamura, 2008)
UF	Surface water	$2.404 \times PLCP^{-0.362}$	(McGivney and Kawamura, 2008)
NF	Surface water	$6.009 \times PLCP^{-0.362}$	(McGivney and Kawamura, 2008)
RO	Surface water	$9.813 \times PLCP^{-0.352}$	(McGivney and Kawamura, 2008)
MF	Reclamation water	$0.548 \times PLCP^{-0.221}$	(Stanford et al., 2013)
UF	Reclamation water	$0.548 \times PLCP^{-0.221}$	(Stanford et al., 2013)
NF	Reclamation water	$0.562 \times PLCP^{-0.130}$	(Plumlee et al., 2014; Stanford et al., 2013)
RO	Reclamation water	$0.562 \times PLCP^{-0.130}$	(Plumlee et al., 2014; Stanford et al., 2013)
MF	Reclamation water	$0.538 \times PLCP^{-0.220}$	(Plumlee et al., 2014)
UF	Reclamation water	$0.538 \times PLCP^{-0.220}$	(Plumlee et al., 2014)
RO	Brackish water	$1.203 \times PLCP^{-0.212}$	(Bhojwani et al., 2019)
RO	Sea water	$1.687 \times PLCP^{-0.145}$	(Bhojwani et al., 2019)
RO	Reclamation water	$1.460 \times PLCP^{-0.174}$	(Bradshaw et al., 2019)

* PLCP = plant capacity [given in $L \cdot s^{-1}$].
** Equations were adapted either to fit to the metric system or to consider O&M costs exclusively.

The box-and-whisker plot (Figure 4.7) reveals that costs exceeding 1 USD per cubic meter are considered outliers. These outliers are associated with the costs of reverse osmosis (RO) obtained from Zhou and Tol (2005), McGivney and Kawamura (2008) and Bhojwani et al. (2019), as well as the cost of nanofiltration (NF) from McGivney and Kawamura (2008) for low-capacity plants. In this particular study, the median costs fall within a range of 0.50 $USD \cdot m^{-3}$ for a DWTP with a capacity of 20 liters per second to 0.13 $USD \cdot m^{-3}$ for a DWTP with a capacity of 3,000 liters per second. This finding can primarily be attributed to the economies of scale. Additionally, Figure 4.7 illustrates the positive skewness of each distribution, especially towards higher larger production plants, confirming that the majority of the data are concentrated closer to the median value and towards lower membrane treatment costs.

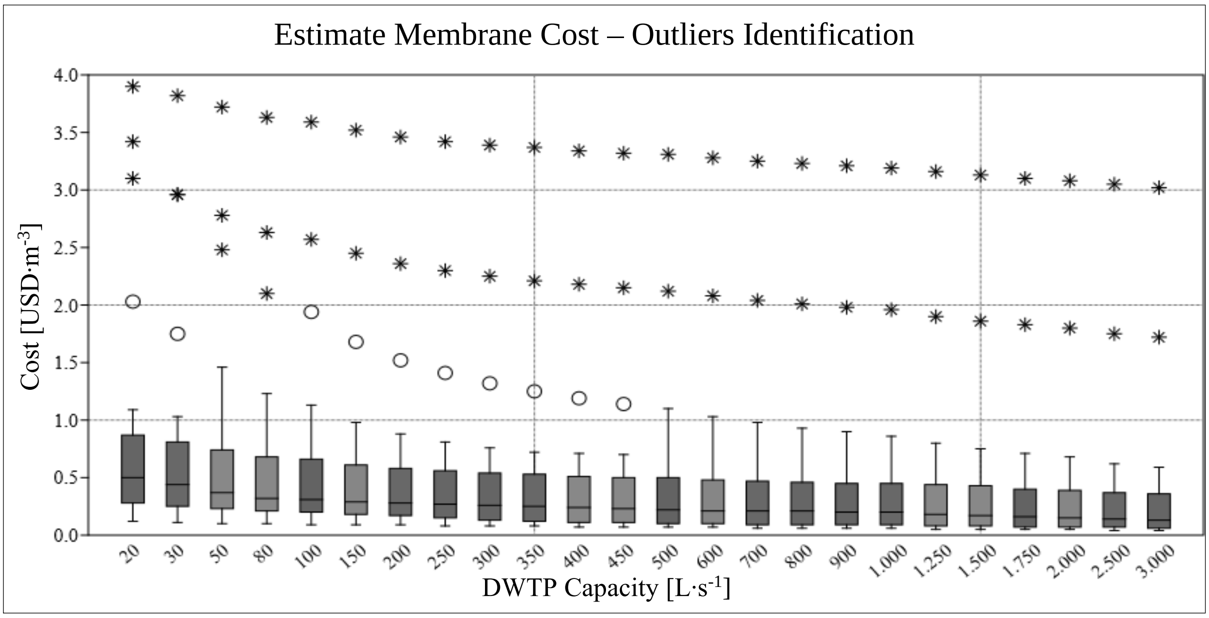

Figure 4.7. Box-and-whisker plot.

The symbol "✱"represents the far out value outliers and the symbol "○" corresponds to the outside value outliers (Tukey, 1977).

In light of the potential significant impact of outliers on the data and the risk of obtaining misleading results (Barnett and Lewis, 1978; Pearson, 2018), they were excluded from the analysis. Following the removal of outliers, the median costs ranged from 0.38 to 0.13 USD·m^{-3}. Consequently, the mean and median value curves were best fitted using the following power law equations, accompanied by their respective Pearson's correlation coefficients: $f(PLCP) = 0.744 \times PLCP^{-0.164}$, R = 0.982, and $f(PLCP) = 0.755 \times PLCP^{-0.208}$, R = 0.992.

To illustrate the impact of excluding outliers, Table 4.5 provides a comparison of the median, mean, and standard deviation computed from the complete dataset and the dataset obtained after the removal of outliers. Eliminating the outlying costs results in a 48.1% decrease in the mean value, despite removing only 11.2% of the records. The reduction in the standard deviation is even more substantial, amounting to 71.3%.

Table 4.5. Influence of outliers in the membrane treatment costs.

	All records	Data without outliers	Change
Median	0.24	0.21	-12.5%
Mean	0.54	0.28	-48.1%
Standard Deviation	0.80	0.23	-71.3%

Records are expressed in terms of USD·m^{-3}.

4.4.4 The short-term blurring effect

The intensification of land use, growing concerns about new organic and inorganic contaminants, and the likelihood of stricter water quality standards pose challenges to the feasibility of conventional water treatment. This trend suggests the search for alternative water sources, when and if available, or the implementation of additional treatment measures to achieve potable water standards in cases watershed protection is neglected. Short-term decisions often prioritize immediate economic gains from land use, while underestimating the apparently modest benefits of avoiding water treatment costs. This can blur the perception of the long-term consequences, where water treatment costs may escalate significantly, particularly when conventional water treatment can no longer ensure the required water standards.

Table 4.6 provides a comprehensive review of the total O&M costs for the five examined DWTPs. It presents the estimated median cost of incorporating additional membrane filtration for each plant capacity. The results highlight that the long-term economic impact of O&M costs at the DWTP can range from 83% to 242% when advanced membrane treatment needs to be implemented alongside the conventional treatment process to ensure the production of drinking water. This finding aligns with the study conducted by McDonald et al. (2016) and emphasizes the substantial economic burden associated with such modifications.

Table 4.6. Total O&M costs and the estimate additional advanced membrane treatment costs.

DWTP	Total O&M conventional treatment cost [USD·m^{-3}]	*Estimate O&M membrane treatment cost [USD·m^{-3}] **	Additional cost **
Faxinal	0.160	0.187 / 0.176	117 / 110%
Marrecas	0.209	0.239 / 0.190	114 / 91%
Maestra	0.149	0.246 / 0.227	165 / 152%
Dal Bó	0.132	0.319 / 0.303	242 / 230%
Samuara	0.428	0.397 / 0.354	93 / 83%

*Costs based on the median value, according to the following equation: $f(PLCP) = 0.755 \times PLCP^{-0.208}$.
** Costs for current treatment flow / full treatment capacity [L·s^{-1}].

Thus, this short-term blurring effect poses a long-term threat to urban water security. The marginal increase in treatment costs remains relatively low, not exceeding 0.0017 USD·m^{-3} (for a one-unit change in the WCI in the case of the Marrecas system). Decision-makers may overlook the degradation of the watershed due to such marginal changes, particularly when there are political and economic costs associated with preserving the watershed, whether for public or private water utilities. The additional expenses incurred from implementing advanced membrane treatment would inevitably be passed on to water tariffs, impacting the social and economic welfare of the municipality. This diverges from the primary objective of ensuring water security (UN, 2013).

Although the analysis provided in this study is valuable, it is important to acknowledge its limitations. The correlation between chemical costs at the DWTP and raw water quality is influenced by various factors, including land use variability and risk perception, which were not fully accounted for in this analysis. Additionally, as the number of regulated contaminants continues to increase, and more stringent maximum contaminant levels are established, conventional DWTPs may face challenges in meeting drinking water standards. It is necessary to consider the inclusion of additional contaminants in future analyses. To enhance our understanding in this field, future research should aim to gather additional data that can provide more comprehensive insights. This would enable the utilization of more advanced analytical techniques, such as artificial neural networks or multiple regression models, to better associate costs with multiple influencing factors. By employing these sophisticated methods, a more accurate and nuanced understanding of the relationships between costs, water quality parameters, and other relevant factors can be obtained.

4.5 Conclusions

One challenge in long-term water and land use planning is accurately assessing water treatment costs and understanding the potential impacts of improving water quality in both the short-term and long-term. The utilization of the Water Compliance Index (WCI) as a proxy for water quality and land use has proven to be a valuable tool for conducting economic and environmental analyses, and can contribute to enhancing urban water security.

The conservation of watersheds may not be immediately perceived as cost-effective, particularly when the estimated benefits are only considered in the short-term. The findings in this Chapter emphasize that even small marginal reductions in treatment costs resulting from watershed conservation efforts may create a short-term blurring effect on decision-makers. This is because the cumulative deterioration of the watershed and the subsequent increase in treatment costs only become apparent in the long-term. However, it is important to acknowledge that while some of these costs may decrease over time due to technological advancements, there are still significant uncertainties and risks associated with watershed degradation that may not be adequately accounted for in current decision-making processes.

By developing a deeper understanding of this blurring effect, it becomes possible to identify the beneficiaries of improved watershed quality and assess the extent to which urban water security is compromised. This serves as a foundation for proposing more effective criteria for the development and enhancement of water and land use policies that can accurately anticipate and address the future impacts of present decisions.

CHAPTER 5

5 ESTIMATING SOCIOECONOMIC DATA BASED ON NIGHTTIME LIGHT SATELLITE IMAGE[2]

A B S T R A C T

Small watersheds lack socioeconomic data. These data are essential for effective land use decision-making and water resources management, particularly in determining their economic value. To address this significant gap, this Chapter introduces an approach to estimate such information for small watersheds ranging from 5 to 100 km². The method utilizes nighttime light (NTL) satellite images and available socioeconomic records from larger localities. Three socioeconomic indicators, namely Gross Domestic Product (GDP), population, and jobs, were selected to test the proposed method. The relationship between these three indicators and the radiance extracted from the NTL images was established through simple regression analysis, applied to the 497 municipalities in the State of Rio Grande do Sul (RS), located in southern Brazil. The polynomial fit equations demonstrated the best Coefficient of Determination, thus being subsequently validated using data from 50 municipalities in the neighboring State of Santa Catarina. The validation process exhibited excellent estimation performance. The validated equations were then employed to estimate the socioeconomic indicators for small watersheds within the municipality of Caxias do Sul, RS, for three different years: 2011, 2014, and 2018. Findings indicate that this novel application of NTL for estimating socioeconomic data can be a helpful tool towards land use and water resources management of small watersheds.

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5.1 Introduction

Instruments, such as water resources policy action plans, serve to highlight strategies pertaining to land use and investment allocation that should take into account the desired socioeconomic standards within a watershed, as noted by Santos et al. (2020). However, determining these desired standards becomes challenging in the absence of socioeconomic information. It is crucial to establish a more quantitative understanding of water demand and the economic benefits derived from its utilization to ensure effective water resources management. For example, various hydro-economic assessment and modeling methods require data to estimate water demand and assess its value (Harou et al., 2009), calculate the opportunity cost of conservation measures (Belladona et al., 2019), and manage land use in small watersheds that supply water for public consumption. Given that watershed scale often serves as the fundamental planning unit in water resources management, it becomes imperative to incorporate socioeconomic indicators at this level, enabling decision makers to evaluate the performance of implemented policies within these areas.

However, there are two limitations in accessing socioeconomic information for water resources management. Firstly, such information is not readily available at the watershed scale, but rather at larger geographic regions such as countries, states, and municipalities. This poses a challenge for smaller areas that require more detailed spatial scale, such as areas situated between multiple municipalities or small watersheds, resulting in an information gap for refining socioeconomic indicators (Ruijs et al., 2017). Secondly, there is a lack of socioeconomic information available on an appropriate time scale, which makes it difficult to assess changes over time (Taylor et al., 2021).

Geospatial remote sensing enables significant advancements in measuring the relationship between the economy and water resources (Booker et al., 2012). This tool has been applied to estimate socioeconomic data of municipalities and countries from nighttime light (NTL) satellite images, such as the Gross Domestic Product (GDP) (Dai et al., 2017a; Huang et al., 2021; Li et al., 2013), population (Archila Bustos et al., 2015; Doll and Pachauri, 2010; Dória, 2015) and to monitor urbanization spatial and temporal changes in cities (Liu et al., 2016; Ma et al., 2012; Nel·lo et al., 2017; Pandey et al., 2013; Zhang and Seto, 2011). However, the study of this Chapter departs from the previous ones in one relevant aspect: the study focuses on small watersheds ranging from 5 to

100 km², as opposed to larger scales such as municipalities, counties, provinces, or other regional scales.

This Chapter presents an approach to estimate socioeconomic indicators for small watersheds by utilizing NTL satellite images and incorporating known socioeconomic information from larger geographical areas. These data are further used in Chapter 6 to calculate the opportunity cost for watershed conservation. The methodology employs a simple regression method to quantify indicators such as Gross Domestic Product (GDP), population, and jobs, as they reflect the pressures exerted on land and water resources (Pozzebon et al., 2022). Additionally, these indicators assist in decision-making processes concerning land use within these areas. GDP represents the total value of goods and services produced in a specific location and timeframe (Williamson, 2016) and corresponds to the overall income generated within the economy (Mankiw, 2018). Population plays a crucial role in the economy as it is linked to the labor supply and consumption dynamics (Castro et al., 2020). The number of jobs also serves as an indicator of the economic well-being of a region, revealing periods of growth or recession (Hall, 2005).

5.2 Methodology

The methodology was divided into three steps. In the first step, the encoded nighttime images captured by the DMSP-OLS and NPP-VIIRS sensors (Chen et al., 2020) for the years 2011, 2014, and 2018 were processed. These images were associated with three socioeconomic indicators, namely GDP, population, and jobs, for the 497 municipalities in the State of Rio Grande do Sul (RS), located in southern Brazil. Simple regression techniques, including linear, exponential, and polynomial regression, were applied for the analysis. The DMSP-OLS and NPP-VIIRS sensors possess unique capabilities to detect low levels of radiance at night across visible and infrared wavelengths (Elvidge et al., 1997), enabling the identification of lighting in urban areas, industrial parks, and rural communities. By examining the temporal changes in the radiance of nighttime lights captured in these images, various processes can be observed. For instance, city expansion leads to a distinct increase in illumination at the periphery, while in other cases, the enrichment of adjacent neighborhoods contributes to enhanced nighttime light emission (Kyba et al., 2017).

In the second step, the regression equations with the highest Coefficient of Determination (R^2) values were validated by applying them to 50 municipalities in the adjacent State of Santa Catarina. The results obtained from applying these equations were then compared with the observed data to assess their accuracy and reliability.

In the third step, the validated equations were utilized to estimate the GDP, population, and number of jobs in a central urban area and six small watersheds situated in both urban and rural regions of the municipality of Caxias do Sul, RS. This estimation was performed for the years 2011, 2014, and 2018, providing insights into the socioeconomic indicators within these specific areas and time periods.

5.2.1 First step - Socioeconomic indicators and nighttime light satellite images relationship

The independent variable (X) in the regression functions was represented by the radiance values obtained from the nighttime light satellite images, while the dependent variable (Y) was represented by the socioeconomic indicators including GDP, population, and the number of jobs. Linear, exponential and polynomial fits (Gupta et al., 2020; Ross, 2021) were examined to identify the most suitable model for the data analysis. The performance of each regression fit was evaluated by calculating the corresponding Coefficient of Determination (R^2), which indicates the goodness-of-fit of the model to the data.

To construct the database, official data from reputable sources were collected. Specifically, the GDP data (IBGE, 2021b), population data (IBGE, 2022), and the number of jobs data (Brasil, 2021b) for the 497 municipalities in the State of Rio Grande do Sul (RS) were obtained from relevant governmental institutions. These data were compiled for the years 2011, 2014, and 2018 to compose the basis of the analysis.

The radiance (or simply NTL), was measured in nanowatts per square centimeter per steradian ($nW \cdot cm^{-2} \cdot sr^{-1}$). These measurements were derived from nighttime satellite images captured by the DMSP-OLS and NPP-VIIRS sensors (Chen et al., 2020) (Figure 5.1), with a spatial resolution of 15 arc-seconds (approximately 500 meters). To enhance the quality of the images, a decoding-encoding process was employed, resulting in a composite image that mitigated the effects of incandescence and light saturation (Chen et al., 2021).

Geoprocessing techniques were applied to determine the radiance for each municipality for 2011, 2014, and 2018. This process involved considering the municipal grid (IBGE, 2021c), available in shapefile format and georeferenced in the WGS84 coordinate system. The radiance of all pixels falling within the boundaries of each municipality was summed, enabling the quantification of the respective NTL values for each of the three years. Table 5.1 presents a statistical summary of the dataset for the evaluated years.

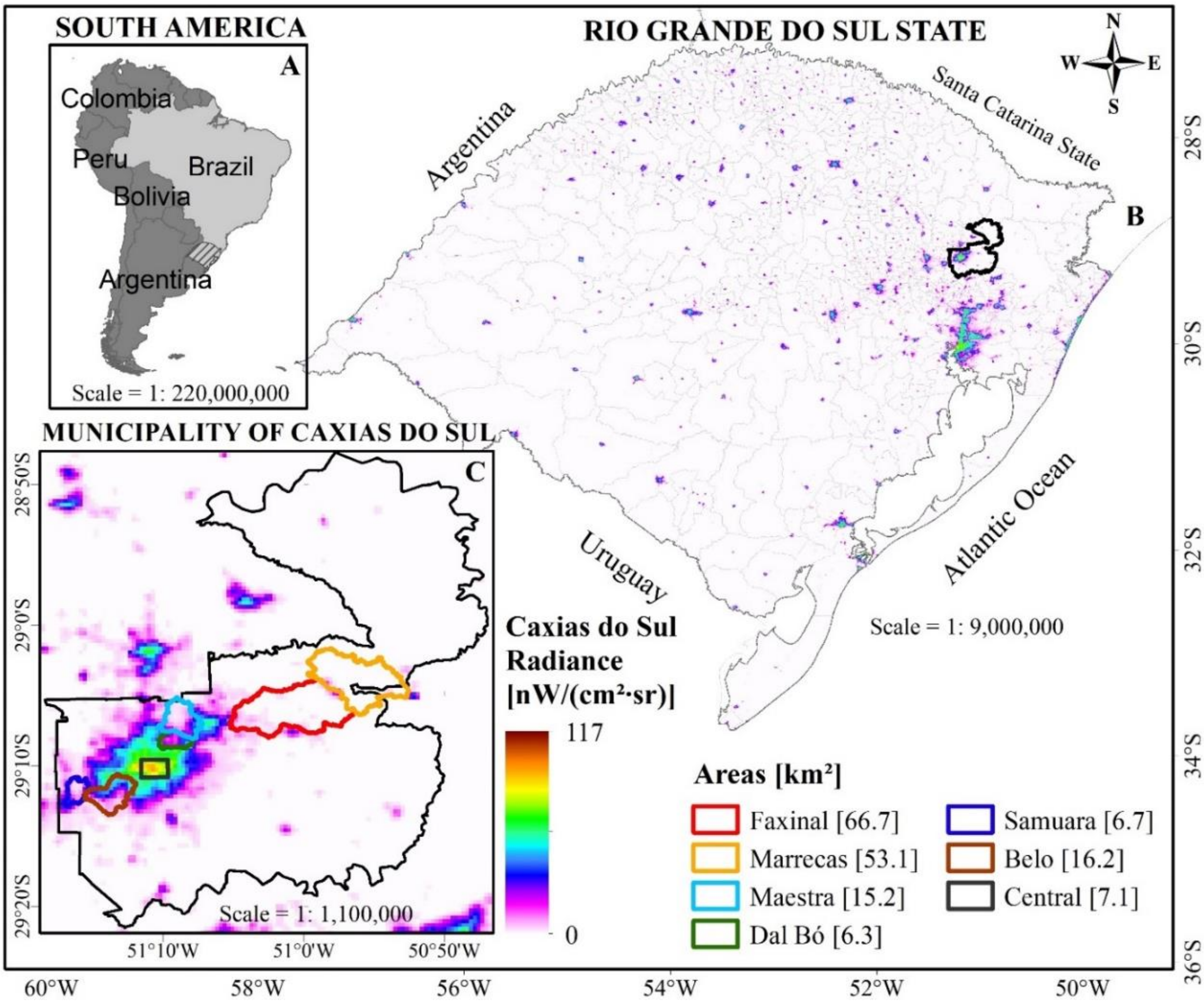

Figure 5.1. Radiance of the municipalities of RS in 2018.
(A) identifies the location of Rio Grande do Sul State at the southmost region of Brazil; (B) demonstrates the concentrated radiance (NTL) over the municipalities' polygons, which coincides with their respective urban areas, and highlights the position of the municipality of Caxias do Sul. (C) highlights the NTL distribution in Caxias do Sul (1,625 km²), where the lower or null values are identified in its rural portions. Source of the nighttime light satellite image: Chen *et al.* (2020).

Table 5.1. Statistics for radiance, GDP, population and jobs of the 497 municipalities of RS.

	2011	*2014*	*2018*
[1]*Maximum radiance*	40331	54792	48145
Minimum radiance	0	2.5	2.4
Mean radiance	398.4	858.3	891.1
Standard deviation radiance	2024.5	3077.2	2848.7
[2]*Maximum GDP*	48288171	63989576	77134613
Minimum GDP	18772	25996	31548
Mean GDP	534387.9	719952.5	920108.5
Standard deviation GDP	2498680.5	3257103.0	3966447.1
Maximum population	1413094	1472482	1479101
Minimum population	1224	1286	1088
Mean population	21638.6	22549.8	22795.9
Standard deviation population	76001.2	79301.9	80253.8
Maximum jobs	741196	780126	689598
Minimum jobs	102	106	58
Mean jobs	5888.2	6255.8	5835.8
Standard deviation jobs	35373.8	37104.7	32923.4

[1]Radiance expressed in $nW \cdot cm^{-2} \cdot sr^{-1}$ and [2]GDP expressed in x1,000 R$. Source: Radiance (Chen et al., 2020), GDP (IBGE, 2021b), population (IBGE, 2022) and jobs (Brasil, 2021b).

5.2.2 Second step - Validation

The regression equations with the highest R^2 values were chosen for validation purposes. The validation process involved 50 randomly selected municipalities located in the State of Santa Catarina (SC) (Figure 5.2), which shares a border with Rio Grande do Sul. These municipalities were selected under the condition that they all possessed available data on GDP, population, and jobs. The radiance values were obtained following the methodology described in Section 2.1. Table 5.2 presents a statistical summary of the dataset for the 50 selected municipalities during the years 2011, 2014, and 2018.

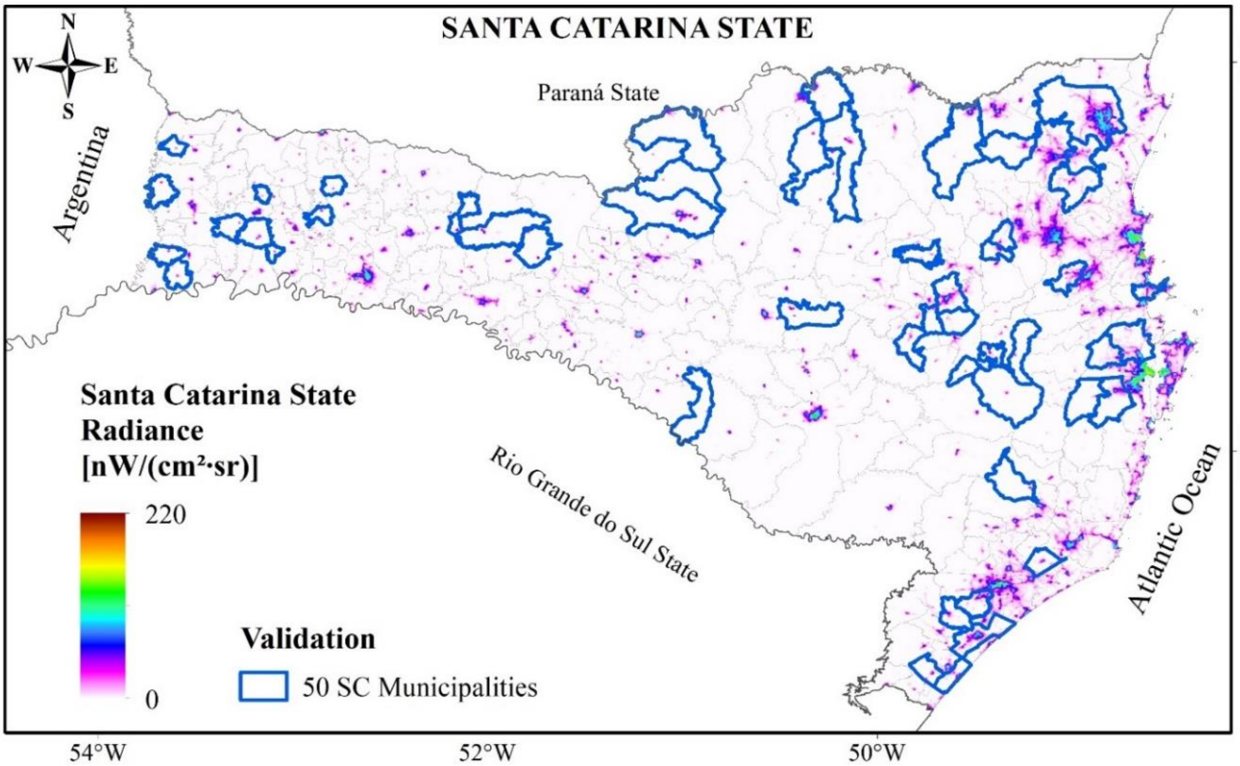

Figure 5.2. Spatial distribution of the 50 municipalities of SC selected for validation
The NTL in this Figure corresponds the 2018 image. Source of the nighttime light satellite image: Chen *et al.* (2020).

The performance comparison between the observed/existing data and the estimated data was evaluated using three metrics: the Nash-Sutcliffe Efficiency (NSE) (Equation 5.1), the Root Mean Square Error (RMSE) (Equation 5.2), and the RMSE-Standard Deviation Ratio (RSR) (Equation 5.3). According to Moriasi et al. (2007), the NSE performance can be classified into four categories: very good for $0.75 < NSE \leq 1$, good for $0.65 < NSE \leq 0.75$, satisfactory for $0.50 < NSE \leq 0.65$, and unsatisfactory for $NSE \leq 0.50$. Similarly, RSR is divided into four categories: very good for $0.0 \leq RSR \leq 0.50$, good for $0.50 < RSR \leq 0.60$, satisfactory for $0.60 < RSR \leq 0.70$, and unsatisfactory for $RSR > 0.70$.

Table 5.2. Statistics for radiance, GDP, population and jobs of the 50 municipalities of SC.

	2011	***2014***	***2018***
[1]Maximum radiance	14203	20295	22283
Minimum radiance	4.7	23.8	35.3
Mean radiance	492.5	963.3	1164.3
Standard deviation radiance	1979.4	2867.0	3169.1
[2]Maximum GDP	18675102	25136136	30785682
Minimum GDP	22231	30170	42286
Mean GDP	689409	940350	1136821
Standard deviation GDP	2602978	3506529	4290969
Maximum population	520905	554601	583144
Minimum population	1748	1733	1646
Mean population	24427.9	25888	27084.7
Standard deviation population	72898.7	77632.8	81654.1
Maximum jobs	191924	208493	213318
Minimum jobs	167	203	204
Mean jobs	7420.1	8048.5	8137.4
Standard deviation jobs	26946.8	29188.6	29823.4

[1]Radiance expressed in $nW{\cdot}cm^{-2}{\cdot}sr^{-1}$ and [2]GDP expressed in x1,000 R$. Source: Radiance (Chen et al., 2020), GDP (IBGE, 2021b), population (IBGE, 2022) and jobs (Brasil, 2021b). Selected municipalities: Agrolândia, Agronômica, Águas Mornas, Alfredo Wagner, Antônio Carlos, Araranguá, Ascurra, Aurora, Balneário Gaivota, Biguaçu, Caçador, Calmon, Cerro Negro, Chapadão do Lageado, Corupá, Cunha Porã, Cunhataí, Dona Emma, Formosa do Sul, Forquilhinha, Grão-Pará, Guabiruba, Guaramirim, Imbuia, Iraceminha, Joinville, Leoberto Leal, Major Vieira, Massaranduba, Meleiro, Papanduva, Paraíso, Ponte Serrada, Porto Belo, Porto União, Princesa, Rio do Sul, Rio Negrinho, Rodeio, Santa Rosa do Sul, Santo Amaro da Imperatriz, São Cristóvão do Sul, São João do Oeste, Tigrinhos, Três Barras, Treze de Maio, Tunápolis, União do Oeste, Vargeão and Vargem Bonita.

In the evaluation of the method, performances classified as very good or good based on the NSE and RSR metrics were considered successful. Additionally, the RMSE value was required to be less than 10% of the range between the maximum and minimum values of the observed target data. This criterion, applied by Al-Murad et al. (2018) and De Vargas et al. (2022a), serves as an additional measure to ensure the accuracy and reliability of the estimated results.

$$\mathbf{NSE} = \mathbf{1} - \left[\frac{\sum_{\mathbf{i=1}}^{\mathbf{n}}\left(\mathbf{V_i^{obs}} - \mathbf{V_i^{simu}}\right)^{\mathbf{2}}}{\sum_{\mathbf{i=1}}^{\mathbf{n}}\left(\mathbf{V_i^{obs}} - \mathbf{V^{medio}}\right)^{\mathbf{2}}}\right] \qquad \text{Equation 5.1}$$

$$\mathbf{RMSE} = \sqrt{\frac{\sum_{\mathbf{i=1}}^{\mathbf{n}}\left(\mathbf{V_i^{obs}} - \mathbf{V_i^{simu}}\right)^{\mathbf{2}}}{\mathbf{n}}} \qquad \text{Equation 5.2}$$

$$\mathbf{RSR} = \frac{\mathbf{RMSE}}{\mathbf{SD^{obs}}} \qquad \text{Equation 5.3}$$

Where the index i represents the sample, $V_i^{obs}$is the observed output at i; V_i^{simu} is the predicted output at i; V^{medio} is the mean of the observed output; n is the total number of samples ($n=50$) and the SD^{obs} is the standard deviation of the observed data.

5.2.3 Third step - Small watersheds

The municipality of Caxias do Sul, with a population of 523,716 inhabitants (IBGE, 2022), exhibits significant geomorphological diversity and a wide range of altitudes (Belladona et al., 2018). Due to this topographic variation, the municipality relies on the construction of small dams on creeks to meet its public water supply needs. For the purpose of testing the methodology, seven specific areas within Caxias do Sul were selected (Figure 5.1C), and the three socioeconomic indicators were estimated for each of these areas. Six of the selected areas were defined based on watershed boundaries, with five of them being subject to special land use legislation (Caxias do Sul, 2005). These five watersheds were chosen due to their importance in the city's water supply. Consequently, the management of water resources in these areas should not only focus on conservation but also on understanding the social and economic aspects that coexist within them. The seventh area, referred to as "Central," is not strictly a watershed but represents a part of the urban central area.

The watersheds of Faxinal (66.77 km²), Marrecas (53.14 km²), Maestra (15.28 km²), Dal Bó (6.31 km²), and Samuara (6.71 km²) creeks cover approximately 8.98% of the total area of the municipality of Caxias do Sul (Pozzebon et al., 2021). These watersheds play a crucial role in storing water for the supply of the city. The Maestra, Dal Bó, and Samuara watersheds are partially or entirely located within the urban perimeter, while the Faxinal watershed is predominantly situated in the rural area, with small portions in the west and northeast being included in the urban perimeter. The Marrecas watershed is mainly characterized by grasslands and forest formations, with only a tiny urban area present in the southwest portion.

On the other hand, the Belo creek watershed (16.27 km²) does not contribute to the public water supply. This particular watershed is situated in an area of the municipality that experiences significant real estate and economic pressures. It was selected as a test area for the methodology due to its size aligning with the research scope. However, the land use and occupation restrictions in this watershed are not as stringent as in the previously mentioned five watersheds. The Central area (7.11 km²) was chosen as a reference area to examine changes in the three indicators. This selection was made to analyze variations in the indicators within a location that does not experience a significant increase in radiance within the defined polygonal boundaries. Additionally, the Central area has minimal vacant land and is predominantly built-up.

The regression equations with the highest R^2 values, indicating the best fit to the data, were utilized to estimate the Y values (GDP, population, and the number of jobs) for the seven selected small areas. These equations, with R^2 values closest to 1, provide the most accurate estimation of the socioeconomic indicators for each of these areas.

5.3 Results and discussion

5.3.1 Relationship between radiance and GDP, population and jobs

Table 5.3 displays the evaluation results of the variable X (NTL) and the variables Y (GDP, population, and number of jobs) obtained through the application of the simple regression method to the 497 municipalities in Rio Grande do Sul (RS). The R^2 coefficient varied significantly depending on the type of regression fit employed. Exponential functions exhibited the poorest fit across all years, indicating that they are not the most suitable for estimating these socioeconomic indicators in RS. In general, only 21% to 38% of the variation in GDP, population, and number of jobs could be explained by the variation in NTL using exponential regression. Conversely, linear regression provided a better fit, explaining 84% to 96% of the variation in the socioeconomic indicators based on the variation in NTL.

The second-order polynomial functions demonstrated the best fit, with R^2 coefficients closer to 1 (ranging from 0.939 to 0.986). Therefore, these functions establish a stronger relationship between the X and Y variables (Figure 5.3). These findings suggest that the best estimation of the socioeconomic indicators for the years 2011, 2014, and 2018 in RS is achieved when applying second-order polynomial functions.

Table 5.3. Exponential, linear and polynomial regressions for the 497 municipalities in RS.

	Relationship	Exponential	Linear	Polynomial
2011	Rad X GDP	$GDP= 122375 \cdot e^{0.0003 \cdot Rad}$ ($R^2 = 0.226$)	$GDP = 1195.3 \cdot Rad + 57195$ ($R^2 = 0.939$)	$GDP=0.0006 \cdot Rad^2 + 1176.9 \cdot Rad + 62170$ ($R^2 = 0.939$)
	Rad X Population	$Pop = 6675.4 \cdot e^{0.0003 \cdot Rad}$ ($R^2 = 0.233$)	$Pop = 36.8 \cdot Rad + 6946.1$ ($R^2 = 0.963$)	$Pop = -0.0003 \cdot Rad^2 + 45.8 \cdot Rad + 4499.1$ ($R^2 = 0.973$)
	Rad X Jobs	$Job = 1001 \cdot e^{0.0003 \cdot Rad}$ ($R^2 = 0.21$)	$Job = 17 \cdot Rad – 913,13$ ($R^2 = 0.952$)	$Job = 0.0002 \cdot Rad^2 + 10.8 \cdot Rad + 776.1$ ($R^2 = 0.976$)
2014	Rad X GDP	$GDP= 160047 \cdot e^{0.0002 \cdot Rad}$ ($R^2 = 0.339$)	$GDP=1026.3 \cdot Rad - 160957$ ($R^2 = 0.940$)	$GDP=0.0092 \cdot Rad^2 + 664.6 \cdot Rad + 55057$ ($R^2 = 0.974$)
	Rad X Population	$Pop = 6351.7 \cdot e^{0.0002 \cdot Rad}$ ($R^2 = 0.340$)	$Pop = 25.3 \cdot Rad + 815.6$ ($R^2 = 0.965$)	$Pop = 0.00009 \cdot Rad^2 + 21.8 \cdot Rad + 2874.8$ ($R^2 = 0.970$)
	Rad X Job	$Job= 993.3 \cdot e^{0.0003 \cdot Rad}$ ($R^2 = 0.318$)	$Job = 11.3 \cdot Rad – 3477.7$ ($R^2 = 0.884$)	$Job = 0.0002 \cdot Rad^2 + 4.2 \cdot Rad + 746.7$ ($R^2 = 0.986$)
2018	Rad X GDP	$GDP= 198731 \cdot e^{0.0003 \cdot Rad}$ ($R^2 = 0.373$)	$GDP=1336.3 \cdot Rad - 270614$ ($R^2 = 0.921$)	$GDP= 0.0171 \cdot Rad^2 + 782.9 \cdot Rad + 70304$ ($R^2 = 0.974$)
	Rad X Population	$Pop = 5984.5 \cdot e^{0.0003 \cdot Rad}$ ($R^2 = 0.384$)	$Pop = 27.5 \cdot Rad – 1754.9$ ($R^2 = 0.956$)	$Pop = 0.0002 \cdot Rad^2 + 21.2 \cdot Rad + 2113.5$ ($R^2 = 0.973$)
	Rad X Job	$Job = 957.9 \cdot e^{0.0003 \cdot Rad}$ ($R^2 = 0.362$)	$Job = 10.6 \cdot Rad – 3649.7$ ($R^2 = 0.848$)	$Job = 0.0002 \cdot Rad^2 + 3.3 \cdot Rad + 870.3$ ($R^2 = 0.984$)

While the correlation results alone do not imply causation (Field, 2017), several previous studies point out to strong evidence that higher GDP is reflected in regions/cities with brighter results in terms of NTL (Chen and Nordhaus, 2011; Lu and Coops, 2018; Villa, 2016). Similarly, Dai et al. (2017) conducted a study in China using images from DMSP/OLS and NPP-VIIRS sensors and found that a polynomial fit better correlated GDP values with NTL. Levin and Zhang (2017) also observed a statistically significant correlation (ranging from 0.60 to 0.66) between GDP per capita and nighttime light radiance in the 200 largest urban areas worldwide.

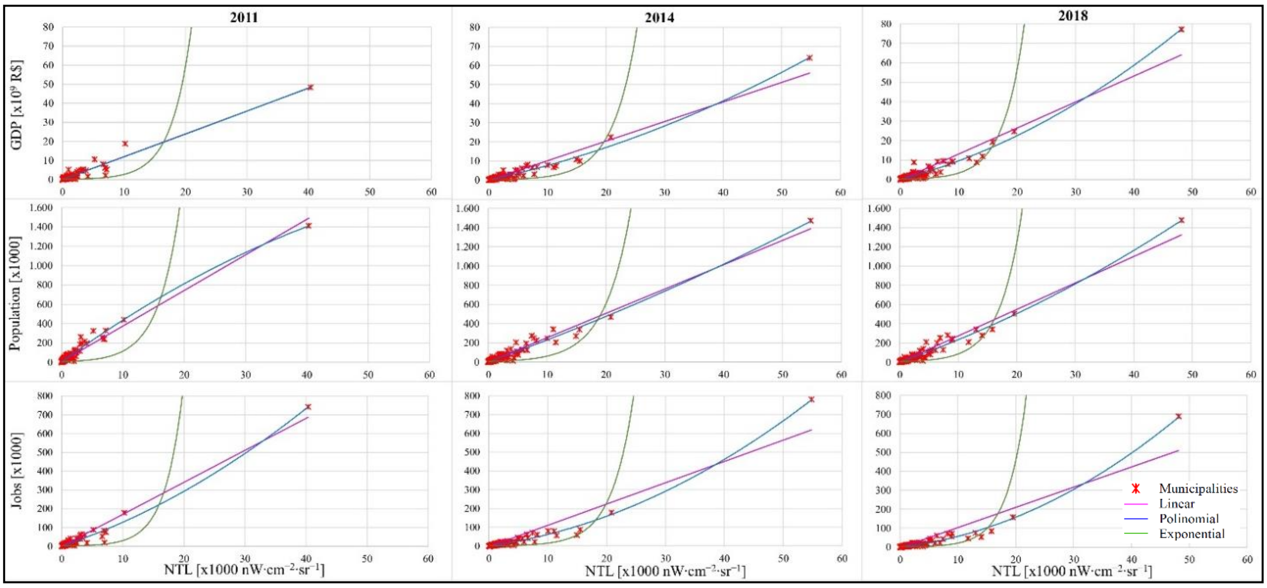

Figure 5.3. Regression for the three socioeconomic indicators for 2011, 2014 and 2018.
Radiance is represented in $nW \cdot cm^{-2} \cdot sr^{-1}$, GDP in BRL (R$), population in number of inhabitants and jobs represented in number of formal jobs.

5.3.2 Validation of the polynomial functions

The second-order polynomial regression functions, which exhibited the best fit, underwent validation to assess their performance. The validation results, represented by the NSE, RSR (Table 5.4), and RMSE performance metrics, confirm that all equations are suitable for estimating GDP, population, and the number of jobs for areas where data is lacking within the study region. The NSE and RSR metrics classified the performance of the equations as "Very Good," and all RMSE values are significantly below the threshold of 10% of the range (maximum and minimum values) of the observed data. These findings indicate that the equations provide accurate estimates for the socioeconomic indicators and demonstrate the reliability of the methodology.

Table 5.4. Validation of the polynomial regression equation to the municipalities in SC.

	Indicator	Equation	NSE	RSR	*Performance
2011	GDP	$GDP = 0.0006 \cdot Rad^2 + 1176.9 \cdot Rad + 62170$ ($R^2 = 0.939$)	0.983	0.128	Very good
	Population	$Pop = -0.0003 \cdot Rad^2 + 45.8 \cdot Rad + 4499.1$ ($R^2 = 0.973$)	0.968	0.177	Very good
	Job	$Job = 0.0002 \cdot Rad^2 + 10.8 \cdot Rad + 776.1$ ($R^2 = 0.976$)	0.983	0.130	Very good
2014	GDP	$GDP = 0.0092 \cdot Rad^2 + 664.6 \cdot Rad + 55057$ ($R^2 = 0.974$)	0.893	0.327	Very good
	Population	$Pop = 0.00009 \cdot Rad^2 + 21.8 \cdot Rad + 2874.8$ ($R^2 = 0.970$)	0.974	0.161	Very good
	Job	$Job = 0.0002 \cdot Rad^2 + 4.2 \cdot Rad + 746.7$ ($R^2 = 0.986$)	0.949	0.225	Very good
2018	GDP	$GDP = 0.0171 \cdot Rad^2 + 782.9 \cdot Rad + 70304$ ($R^2 = 0.974$)	0.963	0.190	Very good
	Population	$Pop = 0.0002 \cdot Rad^2 + 21.2 \cdot Rad + 2113.5$ ($R^2 = 0.973$)	0.986	0.115	Very good
	Job	$Job = 0.0002 \cdot Rad^2 + 3.3 \cdot Rad + 870.3$ ($R^2 = 0.984$)	0.953	0.216	Very good

* According to Moriasi *et al.* (2007).

5.3.3 Estimation of GDP, population and jobs for small watersheds

Figure 5.4 depicts the change in radiance values from 2011, 2014, and 2018 in the seven selected areas. The color ramp in the figure indicates an increase in radiance between 2011 and 2014, with a substantial 54.1% overall increase in NTL during that period. This increase in radiance may be associated with the economic growth experienced by Caxias do Sul and the surrounding region. Correspondingly, the municipal GDP increased by 36.1% during the same timeframe (IBGE, 2021b).

However, from 2014 to 2018, there was a reduction in the NTL values for the seven areas combined, amounting to a decrease of 8.7%. In this period, the increase in municipal GDP was limited to only 15.3% (IBGE, 2021b). It is worth noting that at the national level, the variation in Brazilian GDP for the four-year period from 2011 to 2014, compared to the previous four-year period (2007-2010), was 2.3%. Notwithstanding, for the period from 2015 to 2018, there was a decline of -1.1% in the national GDP (Balassiano and Pessôa, 2021). This data suggests that the reduction in radiance observed in six of the seven areas between 2014 and 2018 (as shown in Table 5.5) reflects the overall economic downturn at both the national and municipal levels, impacting the studied small areas during that period.

In the Faxinal watershed, a different pattern was observed between 2014 and 2018. The total radiance in this period increased from 162 to 192 $nW \cdot cm^{-2} \cdot sr^{-1}$, primarily concentrated in the northeast, south, and west limits of the area, as well as in a concave arc in the southwest direction (Figure 5.4). This distinct behavior of the Faxinal watershed can be attributed to population growth, as identified by Machado et al. (2022), who noted the expansion of irregular human settlements in these specific areas of the watershed. This relationship between radiance and anthropic occupation is also supported by Ge et al. (2018), who demonstrated the possibility of identifying "ghost" urban regions in cities using nighttime satellite imagery. These findings further validate the applicability of the method based on nighttime satellite imagery for studying small areas and understanding the relationship between radiance and human occupation.

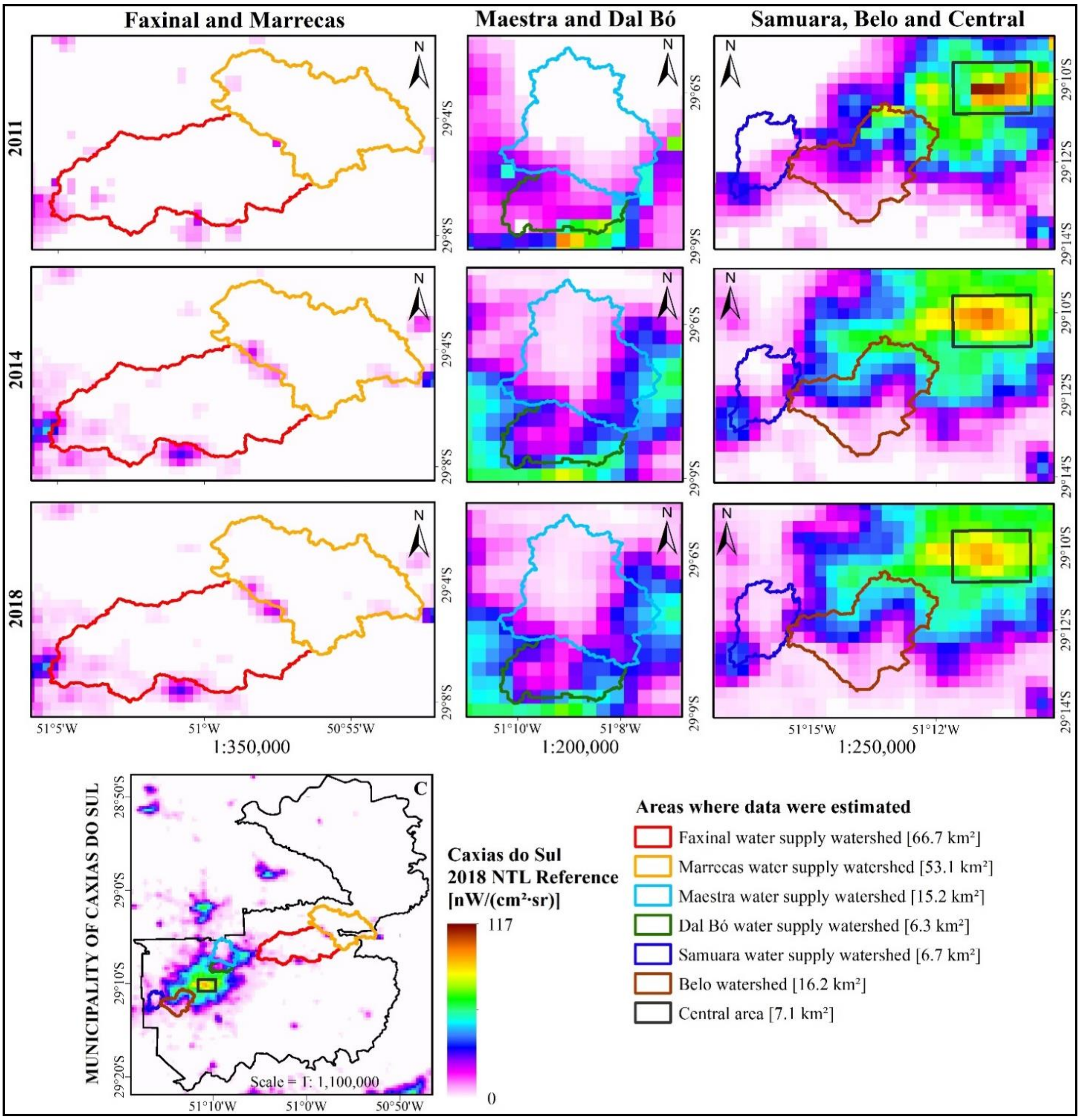

Figure 5.4. Radiance of the seven areas object of data estimation in Caxias do Sul.
Source of the nighttime light satellite image: Chen *et al.* (2020).

Table 5.5 provides the estimated socioeconomic indicators derived from the polynomial functions. Verifying the accuracy of these estimated results can be challenging due to the limited availability of information in the literature or from official sources for these specific areas. However, simple comparisons were conducted whenever possible. For example, the estimated GDP for the Marrecas watershed in 2018, as per the equation presented in Tables 5.4 and 5.5, was 20,751 BRL/ha. Information obtained at the municipal level indicated that the agricultural yield per hectare planted in Caxias do Sul for the same year was 17,657 BRL/ha (SEBRAE/RS, 2019). The GDP estimated by the method presented in this Chapter overestimates the value by 18.3%. This difference may be attributed to the fact that the former includes the GDP of agriculture and livestock, while the latter represents only agricultural income. These comparisons highlight the need for further validation and context-specific data, if available, to enhance the accuracy of the estimated results.

Table 5.5. Estimated GDP, population and jobs for the seven areas.

	AREA	*RADIANCE*	*GDP (x1,000 BRL)*	*POPULATION*	*JOBS*
2011	Faxinal	43	113,330	6,478	1,241
	Marrecas	5	68,380	4,728	831
	Maestra	83	159,330	8,268	1,674
	Dal Bó	330	450,985	19,596	4,362
	Samuara	72	146,698	7,777	1,555
	Belo	888	1,108,178	44,974	10,524
	Central	1,984	2,400,089	94,278	22,989
	**Caxias do Sul*	*10,179*	*12,104,002*	*439,613*	*131,431*
2014	Faxinal	162	113,225	6,413	1,442
	Marrecas	54	41,453	4,056	978
	Maestra	460	313,422	12,962	2,753
	Dal Bó	524	356,350	14,361	3,038
	Samuara	197	137,029	7,193	1,596
	Belo	1,162	790,101	28,407	5,974
	Central	2,688	1,858,712	62,325	13,664
	**Caxias do Sul*	*20,823*	*17,883,118*	*495,839*	*174,922*
2018	Faxinal	192	221,045	6,200	1,512
	Marrecas	51	110,273	3,199	1,039
	Maestra	427	407,547	11,228	2,319
	Dal Bó	472	443,736	12,201	2,477
	Samuara	188	217,982	6,117	1,499
	Belo	1,084	939,304	25,414	4,693
	Central	2,413	2,059,140	54,611	10,019
	**Caxias do Sul*	*19,497*	*21,834,779*	*491,476*	*141,237*

* GDP, population and jobs for Caxias do Sul are based on the equations of Table 5.4 and the radiance given.

The absence of other available values for GDP, population, and jobs, with a compatible scale, in the literature or official sources creates challenges in validating or conducting comparative analyses of the estimated values for areas with urban characteristics. The lack of such data hinders the ability to assess the accuracy and reliability of the estimated socioeconomic indicators in these areas. The study's contribution lies in filling these data gaps and providing a valuable resource for decision-making and planning purposes in small watershed.

The combined total area of the seven selected sites (171.3 km²) represents 10.5% of the total area of Caxias do Sul. However, the proportional contribution of these locales to the estimated indicators does not follow the same ratio. When considering the average estimated values for the three years, the GDP of these areas accounts for 23.4% of the municipal GDP, the population is equivalent to 30.4%, and jobs represent 21.5%. This disparity between the area size and the indicators highlights the significance of these small areas in the local socioeconomic scenario. Despite occupying a relatively smaller portion of the total area, they make substantial contributions to the overall economic activity, population, and employment in the municipality of Caxias do Sul.

5.4 Limitations of the method

The Maestra watershed, for instance, is situated in an urban area with a significant amount of remaining vegetation and rural uses. The presence of vegetation in this area was identified as a component negatively correlated with the NTL, which may totally or partially prevent the emission of light into the atmosphere (Levin and Zhang, 2017b), becoming a limitation of the method to be considered in the generation of estimated socioeconomic data.

Another limitation identified is the scale of the work in relation to the scale of the images used. While NTL has proven to be a reliable proxy for socioeconomic indicators at a macro level, the evidence is not as comprehensive at a local level, primarily due to data constraints at finer scales (Huang et al., 2021; Määttä et al., 2021). This limitation is particularly relevant for small watersheds, which often lack available socioeconomic data as they are not traditionally considered census tract units. Nonetheless, Liu et al. (2022) demonstrated a strong positive correlation between population density at fine scales and coarse NTL satellite images. They concluded that NTL emissions at microscales are not a reliable proxy for per capita income. Conversely, Mellander et al. (2015) found a moderate correlation (approximately 0.5) between NTL and economic activity at a micro-level. Smaller regions tend to exhibit stronger nonlinearity due to their homogeneity in terms of population density, economic activity, and economic structure (Bluhm and McCord, 2022), aligning with the findings of the polynomial correlation that resulted in the best fit in this study. Additionally, Määttä et al. (2021) confirmed that NTL data is an even stronger proxy for economic development at a local level than previous literature suggests.

The future availability of NTL data with finer spatial resolution and potentially treating some watersheds as census tract units could enhance the generation of more accurate socioeconomic indicators. These advancements could help address the limitations and improve the estimation of data in small watersheds.

5.5 Conclusions

This Chapter presented an approach to estimate socioeconomic indicators for small watersheds using nighttime light satellite images. The method was successfully applied to estimate GDP, population, and jobs in seven small areas within the municipality of Caxias do Sul, Brazil, for the years 2011, 2014, and 2018. The second-order polynomial regression analysis demonstrated the most suitable fit for estimating these indicators in the State of Rio Grande do Sul. The equations were further validated in 50 municipalities in the neighboring State of Santa Catarina, confirming their effectiveness in estimating socioeconomic indicators.

Two limitations were identified that can affect the accuracy of the estimated data. One is the presence of vegetation within the study areas can obstruct or limit the emission of light, impacting the reliability of nighttime light measurements. The second one is the scale of the study areas and the resolution of the NTL images that can pose challenges in accurately capturing localized socioeconomic dynamics. These limitations warrant further investigation and consideration in future research, particularly in evaluating other socioeconomic indicators specific to small watersheds.

Nevertheless, despite these limitations, the application of NTL images has shown promise as a tool for estimating socioeconomic data for small watersheds. This novel approach addresses the lack of information in these areas and provides valuable insights for hydroeconomic modeling and water resource management. Furthermore, the practical and accessible data generated through this methodology will be incorporated into the model to quantify the opportunity cost of conservation in Chapter 6.

Future studies can expand on these findings by conducting comprehensive analyses over longer time periods using the extensive time series available from NTL satellite images. Additionally, it is recommended to investigate the impact of vegetation and explore other socioeconomic indicators relevant to small watersheds. By addressing these aspects, this research contributes to the ongoing debate on hydroeconomic modeling and enhances the identification and implementation of effective water management policies in small watershed.

CHAPTER 6

6 THE OPPORTUNITY COST AND THE ECONOMIC BENEFIT OF WATERSHED CONSERVATION[3]

A B S T R A C T

Watershed conservation is a key element in ensuring water security for public water supply. However, those who undertake the responsibility of conservation often bear the burden without reaping the direct benefits, because they live away from the water distribution network. This Chapter introduces a novel methodological framework that addresses the relationship between the opportunity cost of watershed conservation and efficient water tariffs. The opportunity cost was determined using Data Envelopment Analysis, while efficient water tariffs were calculated based on the scarcity rent for water. The benefits of water conservation were also assessed by analyzing changes in consumer surplus. Results revealed that, despite the high average annual opportunity costs amounting to 2.52 million USD, the gains from conservation efforts can reach 4.66 million USD. These benefits are contingent upon the timing of implementing a backstop technology to address potential low water quality. It was observed that the economic benefits of watershed conservation outweigh its costs. However, the earlier a backstop technology needs to be implemented to support the existing treatment process, the greater the loss in consumer surplus. These findings unveil that, on an aggregate level, there is a positive economic balance when policies focus on preserving source water quality.

[3] This Chapter will be published as *Evaluating the cost and benefit of source water conservation based on the opportunity cost of watershed conservation and the scarcity rent for potable water*

6.1 Introduction

Watershed-scale management is a key component to urban water supply security. The impact of land use on stream integrity is contingent upon the scale at which it is examined, as identified by Allan et al. (1997). Local conditions, such as vegetation cover, primarily influence instream habitat structure and organic matter inputs. However, factors like nutrient supply, sediment loadings, and hydrology are influenced by broader regional elements such as landscape features and land use and cover. Such relationships place a significant importance on the land management at the watershed scale to conservation of key ecosystem services. In this context, the term conservation refers to policies that affect the types of agricultural or other land uses with the objective of reducing environmental degradation (Duke et al., 2012). Hence, watershed conservation here means actions aimed at maintaining or recovering physical and biological processes that are associated with raw water production (quantity and quality) necessary to maintain water supply sources. It may encompass, for example, maintaining original vegetated areas and implementing land management practices that reduce runoff. By reducing runoff, these practices contribute to increased infiltration and base flow recharge, as well as a reduction in the influx of contaminants into nearby surface water bodies.

The benefits of watershed-scale management in ensuring the quality and quantity of raw water are evident. However, it is often the landowners residing in rural areas who bear a substantial burden of conservation efforts, despite not usually benefiting from lower water tariffs given their distance from the distribution network. This indicates that conservation does have an opportunity cost, which refers to the foregone alternative use of resources that could have been pursued. Landowners recognize this opportunity cost when making decisions related to conservation, such as whether to participate in payment for environmental services (PES) programs. In such programs, landowners are compensated for their conservation efforts if the offered payment exceeds the opportunity cost of their participation (Wünscher et al., 2011).

Pulido-Velazquez et al. (2013) argue that the opportunity cost of conservation should be integrated into water tariff policies. Typically, water tariffs only cover the costs associated with conveyance, treatment, storage, and distribution services, while the costs of conservation are not fully internalized. According to Tietenberg and Lewis (2015) surface water tariffs are often set at low levels, either due to the use of historical average costs rather than marginal costs, or because the marginal scarcity rent is not factored into the price. Including the marginal scarcity rent in water tariffs serves an important purpose of safeguarding the interests of future consumers (Tietenberg and Lewis, 2015) by signaling to present users the future value associated with the use of a scarce resource (Moncur and Pollock, 1989). Future values can vary based on the needs to expand system capacity or improve the treatment technology of water supply systems.

Water can become scarce to public supply in the foreseeable future in several ways, such as when cities approach their legal entitlement to a given source (Ipe and Bhagwat, 2002), or when the quality of raw water exceeds the treatment capacity of existing systems (Jetoo et al., 2015). Failing to effectively signal to users the opportunity costs associated with watershed conservation and the future value of expansion costs water supply infrastructure can have detrimental consequences. The watershed's ability to produce higher quality raw water may decline, while water users may perceive reduced incentives to preserve and use available resources rationally. To avoid this pitfall, it is essential to implement watershed management policies that furnish urban planners and water users with accurate information and incentives to make informed decisions.

Several previous studies have focused on investigating opportunity costs and their application in the development of water management policies. These studies have explored various aspects, including the design of water pricing policies to enhance water use efficiency (Macian-Sorribes et al., 2015; Pulido-Velazquez et al., 2013; Shao, 2002), the examination of economic impacts resulting from shifts in water and land allocation between agricultural and urban demands (Jiang et al., 2014; Pearce and Markandya, 1987), the track and account for changes in water fluxes and their corresponding economic values in river basins (Tilmant et al., 2015) and the support for the implementation of improved payment for environmental services (PES) programs (Atisa et al., 2014; Leal et al., 2013; Ruijs et al., 2017).

Other methods have also been employed to quantify opportunity costs in relation to biodiversity conservation. Norton-Griffiths and Southey (1995) utilized the net benefits of biodiversity conservation as a basis for their assessment. Hedonic methods were employed by Chomitz et al. (2005), Machado et al. (2016) and Taylor et al. (2021) to quantify the opportunity cost of land and biodiversity conservation. Zhang et al. (2016) adopted a statistical approach to estimate the opportunity cost of water by comparing the water consumption of natural vegetation with that of afforested areas. Campos (2015) applied a slacks-based efficiency measure to estimate the opportunity cost of environmental regulation on farmlands.

Despite the existing body of research, there remains a gap in understanding the relationship between the opportunity cost of watershed conservation and efficient water tariffs specifically for urban water supply. Furthermore, there is a lack of comprehensive frameworks that support temporal decision-making for watershed-scale conservation. This Chapter aims to address this gap by introducing a novel methodology framework. The framework integrates the opportunity cost of watershed conservation, considering its impact on water treatment costs, with the concept of scarcity rent based on future expansion costs. This approach takes into account the use of backstop technologies to treat poorer quality raw water.

Unlike previous studies, such as Jiang et al. (2014) and Norton-Griffiths and Southey (1995), which primarily focused on determining opportunity costs and potential incentives for land use transformations, this Chapter takes a distinct approach. This study explicitly addresses the temporal aspects of these transformations by assessing the opportunity cost of watershed conservation under various future scenarios. Each scenario corresponds to a different trajectory of water tariffs based on how soon, or how later, new water treatment technologies need to be introduced.

The proposed methodology framework combines the multi-criteria Data Envelopment Analysis (DEA) method with microeconomic principles such as willingness-to-pay (WTP), scarcity rent, and consumer surplus. While Campos (2015) utilized a large set of municipalities as Decision-Making Units (DMUs) to evaluate performance, this Chapter introduces watersheds as distinct DMUs over different time intervals. This modification enables the application of DEA with smaller sample sizes, considering inputs and outputs specific to each watershed and year. The chosen approach ensures a reasonable level of discrimination, as it adheres to the requirement of having a minimum of three times as many DMUs as the sum of inputs and outputs (Avkiran, 2011).

The proposed work holds significant potential for enhancing water and land allocation strategies by achieving an optimal economic balance between efficient water tariffs and watershed-scale conservation efforts. By adopting this modified DEA methodology, we can effectively assess the efficiency and economic implications of watershed conservation at a finer spatial and temporal resolution.

6.2 Methods

To determine the opportunity cost of conservation at the watershed scale, the water supply system is discretized into smaller watersheds, each responsible for supplying a specific part of the city. Each watershed is considered a Decision-Making Unit (DMU) and consists of a combination of land uses, including preserved vegetated areas, agricultural areas, and urban areas. These land uses generate economic benefits and provide raw water. Expanding agricultural and urban areas can lead to increased economic benefits, but it also results in reduced watershed conservation and decreased raw water quality and quantity. This, in turn, increases the costs associated with water treatment and potable water tariffs, leading to a reduction in consumer surplus and overall economic welfare. On the other hand, increasing watershed conservation efforts has the opposite effect in the long-term, as it improves water quality, reduces tariffs, and increases consumer surplus. However, it also imposes constraints on land use and limits certain economic benefits that could be derived from alternative land uses.

In the proposed framework, each DMU decides to use its available land, which results in different outcomes. The land is considered as an input, and the decision to use it results in two distinct outputs. The first output represents a desirable outcome, which can be measured by various indicators such as GDP (Zhou et al., 2006), agricultural revenue (Campos and Bacha, 2016), total port operation revenue (Yang, 2013), or factors such as time and money (Mullainathan and Shafir, 2013; Thaler, 2015). The second output is related to land use changes that increase runoff. This increase in runoff has negative consequences, as it reduces the amount of water that can be effectively infiltrated into the soil, leading to a decrease in base flow recharge. Additionally, increased runoff can result in a higher influx of contaminants into surface water bodies from which the city draws its water supply. Environmental regulations place constraints on this second output, limiting the full utilization of the land and forfeiting the production of desirable outputs.

It is important to note that each watershed consists of similar-sized productive properties, and the framework assumes constant returns to scale (Charnes et al., 1978; Tone, 2001; Zhou et al., 2006). This assumption allows for consistent comparisons and assessments of the trade-offs between land use decisions and economic benefits across different watersheds.

The benefits of watershed conservation are quantified by comparing the total consumer surplus obtained with the current treatment technologies (lower tariffs) with alternative more complex (backstop) technologies that would be necessary at some time in the future under poorer raw water quality (higher tariffs).

In the absence of a well-defined market for water, the principle of willingness to pay (WTP) is employed as a means to measure its value (Young and Loomis, 2014). This is done through the estimation of marginal benefit (demand) functions for potable water, which enable the assessment of welfare losses experienced by water users when water becomes more expensive or scarce. Such situations may arise due to higher treatment costs or lower flows during dry periods when the watershed is not adequately preserved. The change in welfare is quantified using the concept of consumer surplus (Varian, 2014).

The calculation of consumer surplus involves integrating the potable water demand function, which is estimated using an approach that relies on price-demand elasticity data. This approach allows for the extrapolation of water demand values in the vicinity of an observed point where consumption and price (water tariff) are known (Griffin, 2006). To address the issue of observed water tariffs potentially underestimating the value of water (and consequently the consumer surplus), adjustments are made to the observed tariff values based on the concept of scarcity rent for water (Moncur and Pollock, 1988; Tietenberg and Lewis, 2015).

Finally, to quantify the benefits of watershed conservation, the scarcity rent is incorporated into the economic equilibrium point, resulting in an adjusted demand function. This adjusted demand function is then used to calculate changes in consumer surplus. The costs of watershed conservation are determined through the application of the DEA approach. Figure 6.1 provides an illustration of the general methodological framework employed in this Chapter.

6.2.1 Area of study

Data for this analysis were collected from various sources, including SAMAE, official entities, geoprocessing techniques, the estimate data presented in Chapter 5, and existing literature. The proposed methodology was evaluated using real-scale data from a utility located in the municipality of Caxias do Sul, which has been detailed in Chapter 3.

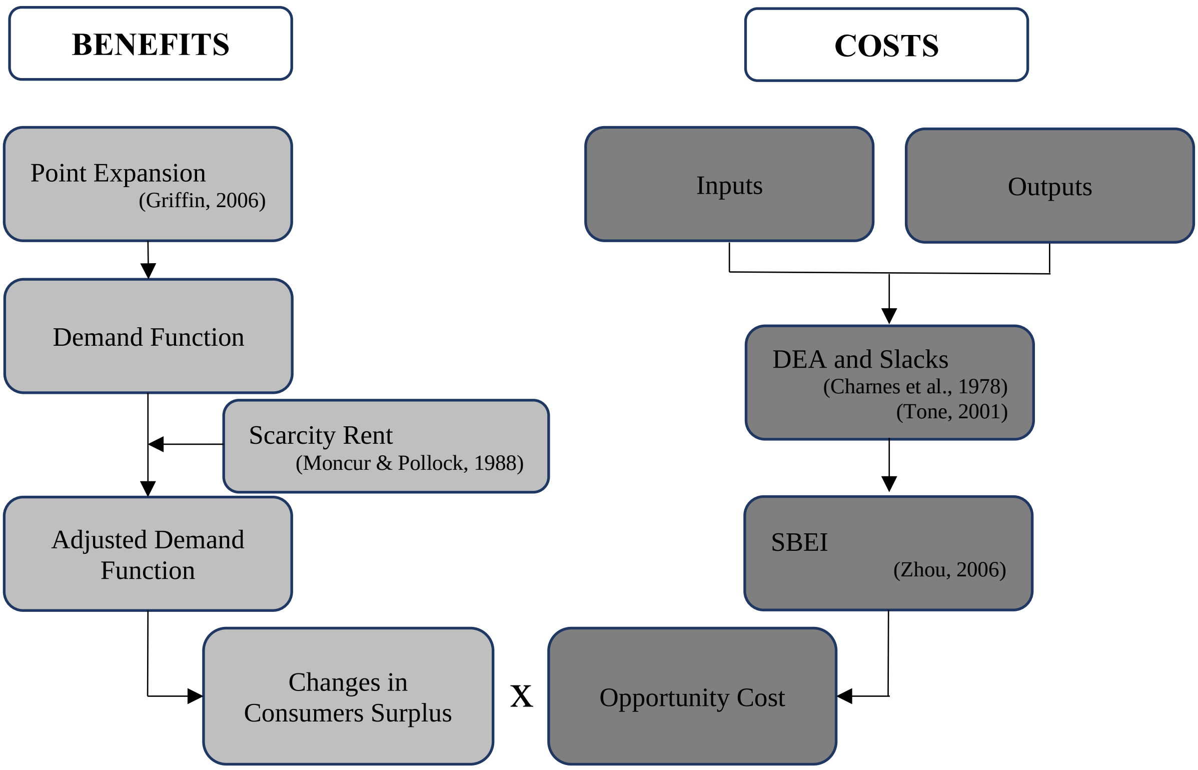

Figure 6.1. Methodological framework for Chapter 6.

6.2.2 Modeling Decision-Making Units (DMU)

Each sub-system in Caxias do Sul is modeled as a DMU that allocates different inputs and produces outputs according to the DEA, which is a non-parametric data-oriented approach used to evaluate relative efficiency among a set of DMUs. This approach is based on Charnes et al. (1978) to emphasize that the interest is centered on decision making by not-for-profit entities, such as schools, hospitals, cities and countries, rather than the customary firms. As it requires very few assumptions (Cooper et al., 2011), DEA allows the analysis of the complex relations between the multiple inputs and multiple outputs, also helping to identify sources of inefficiency (Cooper et al., 2007). However, some pitfalls in the empirical application of DEA should be avoided (Dyson et al., 2001), for instance: (1) the greater the number of inputs and outputs, the lower the discrimination, thus the number of efficient DMUs on the frontier tend to increase. A possibility to invert this effect is to select inputs and outputs with care or to choose categorical variables; (2) handling with undesirable outputs, such as pollution or environmental restrictions, presents problems in the application of DEA. One possibility to handle such variables is by moving it from the output to the input side.

The level of environmental efficiency at each DMU is assessed based on the undesirable output, which represents the land use change leading to increased runoff, such as urbanization and agricultural production. This measure allows the evaluation of how efficiently each DMU manages its environmental impact (undesirable output) relative to the production of the desirable output. A higher level of environmental efficiency indicates that a DMU can achieve a greater level of desirable output for a same level of runoff and land use change.

DEA constructs a production boundary combining observed input and output data, creating a frontier that “envelops” all DMUs. The efficiency of a DMU is quantified using a scalar value that ranges between zero, representing the worst efficiency, and one, representing the best efficiency (Tone, 2001). This scalar value can be determined through the application of linear programming techniques. Because its efficiencies (or inefficiencies) are obtained directly from the data, this method does not require preassigned input and output weights (Banker et al., 1984; Cooper et al., 2007). The opportunity cost of watershed conservation is obtained from the slacks-based efficiency index (SBEI) proposed by Zhou et al. (2006), whose foundation stems from the Charnes, Cooper and Rhodes (CCR) method (Charnes et al., 1978) and from the slacks-based measure of efficiency method (Tone, 2001).

The CCR method is utilized to evaluate the efficiency of not-for-profit entities that provide public services. This method involves transforming a fractional program into its equivalent linear form. The CCR model is built on the assumption of constant returns to scale (CRS), as depicted in Figure 6.2A. On the other hand, the Banker, Charnes, and Cooper (BCC) method (Banker et al., 1984) represents production frontiers with a piecewise linear and concave shape, indicating variable returns to scale (VRS), as shown in Figure 6.2B.

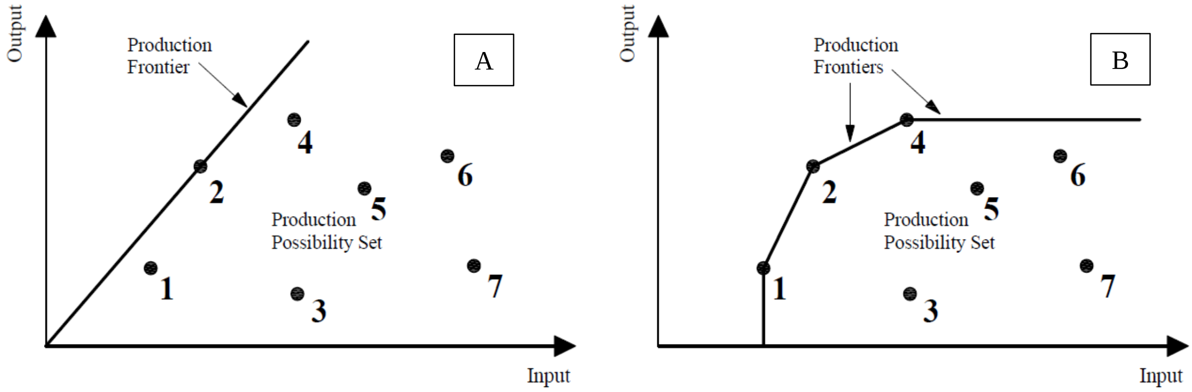

Figure 6.2. Production Frontiers.
Figure 6.2A shows the Production Frontier exhibiting CRS in the CCR model, Figure 6.2B depicts the Production Frontiers exhibiting VRS, in a piecewise and concave shape, in the BCC model. The Production Frontiers “envelop” all the DMUs (represented by the dots from 1 to 7) that fall on or below these frontiers. Source: adapted from Cooper et al. (2007).

Let a single input (x) produce a single output (y), the slope of the line connecting each point to the origin corresponds to each output (y) per each input (x) and the highest such slope is attained by the line from the origin through DMU-2 (Cooper et al., 2007), as shown in Figure 6.2A. This such line is called the “production frontier” (or efficiency frontier). Note that this frontier touches at least one point and all points are on or below this line. DEA identifies a DMU, like DMU-2 in Figure 6.2A, for future examination or to serve as a benchmark to use in seeking improvements for the other DMUs.

6.2.2.1 Measuring environmental efficiency at the watershed scale

The efficiency of a watershed (DMU) in terms of economic outputs determines the level of stringency in environmental regulations and the magnitude of opportunity costs. The slacks-based efficiency measure (SBM) is employed to assess the economic efficiency of a DMU by quantifying the input excess ($\mathbf{s}^-$) and the output shortfall ($\mathbf{s}^+$) in relation to the efficiency frontier. The economic efficiency score, denoted as θ_1, and the environmental efficiency score, denoted as θ_2, are subsequently defined (Zhou et al. 2006). The slacks-based efficiency index (SBEI) is calculated to evaluate the impact of environmental regulations on economic efficiency. θ_1 represents the technical efficiency, disregarding environmental regulations, while θ_2 reflects the environmental efficiency (Campos, 2015). The terminology “technical” suggests an efficiency assessment based on the relationship between the quantity of output obtained and the quantity of input used, while the term "environmental" indicates the inclusion of environmental regulations to constrain the undesirable output (**b**), which, in this study, refers to the land use change that interferes in runoff.

The economic efficiency (θ_1) can be obtained through an input-oriented linear programming problem (Tone, 2001) when the efficiency frontier exhibits constant returns to scale and when only the desirable products/outputs (**g**) are considered. When the undesirable outputs (**b**) are considered, the production process can be modeled by the environmental DEA technology. In this case, the economic efficiency of a DMU can be identified as its environmental efficiency (θ_2).

Finally, the SBEI for modelling environmental performance (Zhou et al., 2006) is defined as the quotient of θ_1 and θ_2, as shown in Equation 6.1. Chapter 11 contains the Appendix for Chapter 6 where further detail on the SBEI is provided.

$$\mathrm{SBEI} = \frac{\theta_1}{\theta_2} \qquad \text{Equation 6.1}$$

Table 6.1 displays the inputs (**x**) and outputs (**g**, **b**) employed in the slacks-based efficiency model. A group of 15 DMUs is assembled with data from 5 watersheds and 3 distinct time periods each: the years of 2011, 2014 and 2018. The selection of these years was based on the availability of data, particularly the availability of satellite images suitable for capturing land use and distinguishing between productive and unproductive land.

The model incorporates two inputs (**x**): the number of job opportunities within each DMU and the productive area of each DMU. Additionally, two outputs are included: a desirable output (**g**) represented by the Gross Domestic Product (GDP), and an undesirable output (**b**) represented by the impervious area of each DMU. The data for jobs and GDP were obtained from the results presented in Chapter 5. The productive area was obtained through geoprocessing techniques, while the runoff Curve Number (CN) (Mishra and Singh, 2003) was used as a proxy to quantify the impervious area of the watersheds, also generated through geoprocessing.

Job opportunities ($\mathbf{x_1}$) comprise the quantity of people working in $\mathrm{DMU_i}$. The productive area ($\mathbf{x_2}$) corresponds to the total area of the $\mathrm{DMU_i}$ minus the area restricted by national water resources conservation, minus the productive land that is not in use. The areas with no direct economical returns, such as abandoned quarries and roads, will also be subtracted. The mathematical representation of the productive area ($\mathbf{x_2}$) is demonstrated in Equation 6.2.

Table 6.1. Input and output data of the DMUs.

DMU	Name	Input		Output	
		Job opportunities [unit]	Productive area [m²]	GDP (g) [x1,000 BRL]	Impervious area (b) [CN]
1	*Faxinal-2018*	1,512	17,892,845	221,045	74.96
2	*Faxinal-2014*	1,442	19,516,560	113,225	74.42
3	*Faxinal-2011*	1,241	18,144,571	113,330	75.17
4	*Marrecas-2018*	1,039	20,406,175	110,273	73.95
5	*Marrecas-2014*	978	21,308,654	41,453	74.64
6	*Marrecas-2011*	831	21,038,865	68,380	74.16
7	*Maestra-2018*	2,319	4,713,506	407,547	77.29
8	*Maestra-2014*	2,753	5,306,266	313,422	78.38
9	*Maestra-2011*	1,674	5,346,954	159,330	78.24
10	*Dal Bó-2018*	2,477	2,279,022	443,736	79.24
11	*Dal Bó-2014*	3,038	2,111,645	356,350	78.62
12	*Dal Bó-2011*	4,362	2,261,263	450,985	79.37
13	*Samuara-2018*	1,499	1,368,592	217,982	66.47
14	*Samuara-2014*	1,596	1,439,249	137,029	68.92
15	*Samuara-2011*	1,555	1,351,893	146,698	66.42
	Mean	1,887.7	9,632,404	220,052	74.68
	Median	1,555	5,306,266	159,330	74.96
	St. Deviation	914.4	8,371,116	134,545	4.17

$$\boldsymbol{x_2} = ta_{DMUi} - \left(ra_{pres} + pa_{niu} + ia\right)_{DMUi} \qquad \text{Equation 6.2}$$

where, ta_{DMUi} is the total area of the DMU under analysis, ra_{pres} is the area restricted by national water resources conservation in DMU_i, pa_{niu} is the productive area of the DMU_i that is currently not being used and ia is the land with no direct economic return.

The desirable output (**g**) in the model represents the GDP, which quantifies the total value of goods and services produced within the boundaries of each DMU during a year. On the other hand, the undesirable output (**b**) corresponds to the land use change that leads to increased runoff, and is represented by the Curve Number (CN) of each DMU. The CN is a numerical parameter that ranges from 0 to 100 (Mishra and Singh, 2003), where higher values indicate greater runoff and reduced infiltration. The CN was chosen as a measure of the land use impact on surface runoff due to its availability and well-established use in assessing hydrological processes. In future improvements, methods such as those proposed by Gabriels et al. (2021) could be explored, which incorporate spatially distributed rainfall-runoff models to adjust CN values based on additional information.

6.2.3 The opportunity cost of watershed conservation

The SBEI (Equation 6.1) lies in the interval (0, 1]. When SBEI is equal to 1, $\theta_1 = \theta_2$, it implies that the transformation of the production process from the "technical" DEA to the "environmental" DEA has no effect on the economic efficiency of the DMU (Zhou et al., 2006). If SBEI < 1, it suggests that the environmental regulation restricts the full use of inputs and reduces the production of desirable outputs of the DMU (Campos, 2015). Therefore, there is an opportunity cost associated with that such regulation. The opportunity cost (OC) of an environmental regulation can be obtained by

$$OC = (1\text{-}SBEI)\times(\text{the desirable output}). \quad \text{Equation 6.3}$$

6.2.4 The benefit of watershed conservation

The benefit of watershed conservation can be quantified as the change in total consumer surplus when comparing the current treatment technology to the consumer surplus under a more advanced treatment system required for treating poorer quality raw water. Further analysis on the impact of poorer water quality on the necessity for improved treatment and its corresponding effect on tariffs can be found in Chapter 4 of this thesis.

The changes in consumer surplus are calculated by integrating a Marshallian demand function for potable water in the municipality. Prior to estimating the demand function, a source water value is determined by considering the scarcity rent, as proposed by Moncur and Pollock, (1988) and Tietenberg and Lewis (2015). This source water value is then added to the observed potable water tariff to obtain an adjusted tariff for the analysis.

6.2.4.1 Demand function estimation

The demand function is estimated using the method proposed by Griffin (2006), which extrapolates the demand curve in the vicinity of an observed point where the price paid for potable water (tariff p), the water quantity (w), and the price elasticity of demand (ε) are known. The price elasticity of demand is obtained from another method or source. In order to fully accomplish the demand function, the price elasticity of demand has to be given from another method. These inputs are customarily available. This equilibrium point indicates economic efficiency (James and Lee, 1971), as consumers are willing to consume the specific quantity of water at the given price. Deviations from this equilibrium point result in changes in water usage, either decreasing as prices increase or increasing as prices decrease, depicting movements along the Marshallian demand curve.

The demand function can be assumed to be either linear or exponential. Price elasticity of demand (ε) measures how much the quantity demanded responds to a change in price (Equation 6.4). Demand is said to be inelastic ($-1 < \varepsilon < 0$) if the quantity demanded responds only slightly to changes in the price (Mankiw, 2018), which is the case of urban water as there are no substitutes.

$$\varepsilon = \frac{\Delta w}{\Delta p} \cdot \frac{p}{w} \qquad \text{Equation 6.4}$$

For the linear demand $w = \alpha \cdot p + b$, since the point (p,w) is known and the elasticity (ε) is obtained exogenously, they can be substituted into Equation 6.4 to obtain the slope ($\alpha = \Delta w / \Delta p$) of the demand curve. Then, the substitution of the slope and the point allows the ordinate intercept b to be determined. Conversely, via integration and assuming ε constant, Equation 6.4 yields to the exponential demand function: $w = k \cdot p^{\varepsilon}$ (Griffin, 2006). Given the point and the elasticity are known, the constant k can be calculated to form the demand equation.

While easy to apply, the method oversimplifies the demand function (Rougé et al., 2017). It assumes a linear or exponential form, which can result in underestimating or overestimating demand as one moves away from the known point (Griffin, 2006). Additionally, the linear assumption may underestimate consumer surplus, while the exponential assumption leads to mathematically infinite consumer surplus, which is not practical for application.

To mitigate this effect, it is proposed in this Chapter a mean demand function (Figure 6.3A) where, for a subset of p values that is located close to the economic equilibrium point (known point), a mean value of w is calculated from the linear and from the exponential functions. In the case of this study, the subset is comprised of thirteen p values (five below and eight above the equilibrium point, ranging from 3 to 20 BRL.m^{-3}, considering that the p value at the known point is at 7.98 BRL.m^{-3}). For each of these p values (p_i) the linear function results in a w_{li}, which is the same for the case of the exponential function w_{ei}. Following the proposed method, it yields in a new w_i, given by the mean of w_{li} and w_{ei}, $(w_{li}+w_{ei})/2$, thus forming a scatter plot of (p_i,w_i) close to the known point. From the new mean demand points (p_i,w_i) an adjusted demand function is derived by regression. The regression equation that presents the best Coefficient of Determination (R^2), or the best fit, becomes the new demand function.

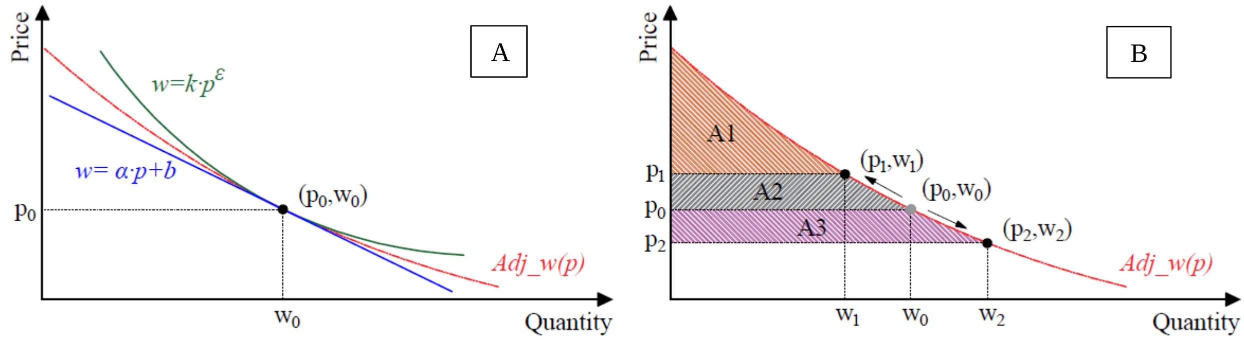

Figure 6.3. Water demand function.
Figure 6.3A depicts the linear, the exponential and the adjusted mean curves. Figure 6.3B demonstrates the fluctuation on consumer surplus when price changes from p_0 to p_1 or to p_2.

Price and volume were acquired internally with the technical departments of SAMAE. Unfortunately, the utility only provided data corresponding to the month of February of 2022. The tariff was established by local decree (Caxias do Sul, 2022) in 7.25 Brazilian Real per cubic meter ($BRL{\cdot}m^{-3}$), equivalent to 1.40 $USD{\cdot}m^{-3}$, for a residential demand with a monthly consumption of between 6 m^3 to 10 m^3. For a commercial use of an amount between 11 m^3 and 15 m^3 per month, the decree limited tariff to 8.71 $BRL{\cdot}m^{-3}$ (1.68 $USD{\cdot}m^{-3}$). As the utility does not aim at profit making, tariff must meet operating expenditures, depreciation quotas, the amortization and interest on credit operations (Caxias do Sul, 2016). So, during this period (February 2022) the average water price, considering domestic and commercial use, was 7.98 $BRL{\cdot}m^{-3}$ (equivalent to 1.54 $USD{\cdot}m^{-3}$) and the water consumption of the supplied inhabitants reached 1.83 hm^3 within this month. The price elasticity of demand ($\varepsilon = -0.27$) is based on Ghinis et al. (2020).

6.2.4.2 Scarcity rent of potable water and adjustment in the water tariff

Each new home connected to the water supply system contributes to increased water consumption, which entails costs associated with energy, chemical products, and other resources required for water production. Furthermore, it also consumes an additional fraction of the system treatment capacity, which will need to be expanded, or replaced by more complex (and expensive) systems in the future if the quality of raw water worsens.

Moncur and Pollock (1988) point out that in order to achieve more efficient water consumption, water prices should be adjusted to an efficient price (P_E) that incorporates the marginal cost of extraction (MC_E) and the scarcity rent (Φ), ($P_E = MC_E + \Phi$). The former is empirically determined by the utility, while the latter is a surrogate for the price for water, and it has to be indirectly estimated based on the cost of providing more water in the future, including the use of more complex water treatment technologies.

Consider a city supplied at time t_0 by a low-cost surface water source, where water quality allows conventional treatment. As land use changes in the upper watershed and raw water quality declines, the system eventually reaches a point, at time T, where a backstop technology has to be added, replacing the existing treatment system. Assuming also that marginal cost of extraction remains constant at C_1 from time t_0 until time T. From time T on, the marginal cost of extraction shifts to C_2 due to the additional costly treatment, thus $C_2 > C_1$ (Figure 6.4A). Letting C_t denote the constant extraction cost (MC_E) function of a given utility, then,

$$C_t = \begin{cases} C_1, t \leq T \\ C_2, t > T \end{cases}. \qquad \text{Equation 6.5}$$

Figure 6.4B demonstrates the function C_t as being the line C_1abd, and total cost is the area under C_t. The open-ended area *abdg* indicates the increase in total cost due to the necessity of switching to a more complex water treatment technology at time T. While *abdg* has infinite area, it has a finite present value. This present value will decrease if the utility can delay T to ($T+1$). The magnitude of this cost decrease is shown as *abef* (Moncur and Pollock, 1988).

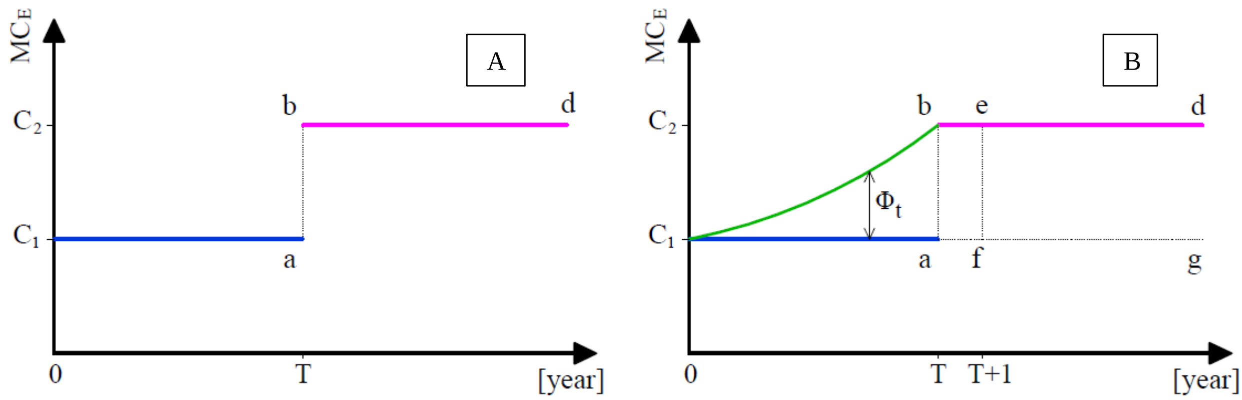

Figure 6.4. Marginal cost of extraction.
Figure 6.4A depicts the path of constant extraction costs. Figure 6.4B shows the efficient price path, C_1b. Adapted from Moncur and Pollock (1988).

If raw water pollution increases sooner, so should water tariffs to consumers. Conversely, if raw water quality is improved or maintained unaltered, T could be postponed. By postponing T to $T+t$, the present value of the investment on the backstop technology will decrease (Ipe and Bhagwat, 2002; Moncur and Pollock, 1988). The change in present value of additional costs per time at the end of the year T is the scarcity rent (the price for water at present time). The scarcity rent is then the present value of the magnitude $C_2 - C_1$ (Moncur and Pollock, 1988), expressed in Equation 6.6.

$$\Phi = (C_2 - C_1) \cdot e^{-r(T-t)}, \qquad \text{Equation 6.6}$$

where C_1 and C_2 are the marginal costs of extraction at the initial period (t until T) an after T, respectively (expressed in $/m³/s), r is the discount rate, t is the current time and T the time when backstop technology is operational. Adding Φ, for every t, to C_1, yields the efficient price path, P_E, represented by the line C_1b in Figure 6.4B.

After Φ is determined, it is added to adjust the potable water tariff according to Equation 6.7.

$$p_{\Phi} = p + \Phi. \tag{Equation 6.7}$$

Where p_{Φ} is an efficient tariff ($\$\cdot m^{-3}$) when source water price is considered, p denotes the actual price of tariff ($\$\cdot m^{-3}$) and Φ corresponds to the scarcity rent ($\$\cdot m^{-3}$). Hence, the new point used to extrapolate the water demand function is now given by (p_{Φ},w).

The marginal cost of extraction for the case of Caxias do Sul was obtained from the data presented in Chapter 4, where it is presented this cost as a composite of the energy required to convey and treat the water, the chemicals used in the treatment trains and the personnel directly involved from watershed control to overall operation and treatment. Chapter 4 presented the current costs (C_1) as 0.197 USD·m^{-3}, while, based on the estimate and on a scenario of watershed degradation where membrane filtration would have to be added to the treatment train to meet water quality standards, the future marginal cost of extraction (C_2) would raise to 0.420 USD·m^{-3}. The costs of *C1* and *C2* have been adjusted to October 2022 to account for inflation. By accounting for inflation, the costs were brought up to this Chapter analysis date.

6.2.4.3 Changes in consumer surplus

Nicholson and Snyder (2017) call the attention to an important problem in applied welfare economics, which is to devise a monetary measure of the utility gains and losses that individuals experience when prices change. One way to overcome this difficulty is to find out how much a person is willing to pay for a certain good, for example a cubic meter of potable water. If the price is higher than the price the person is willing to pay, there is a loss in welfare, while, on the contrary, there is a gain in this person's welfare. Individual consumer surplus is the difference between the maximum amount that a consumer is willing to pay for a good and the amount that the consumer actually pays (Pindyck and Rubinfeld, 2018). Any fluctuation on market price (p) or on quantity (w) will result in altering consumer surplus. A public policy that favors urban and agriculture expansion without conservation practices is likely to cause raw water quality degradation as well as its diminishment in the dry season due to reduced watershed infiltration and subsurface storage (Shuster et al., 2005).

The exponential demand function (shown in Figure 6.3A) will yield an infinite result when calculating the area under the curve. In contrast, the adjusted demand function can be easily calculated through integration. This adjusted function, unlike the others, helps mitigate the potential under or overestimation issues associated with the point expansion method. It is important to note that the focus of the analysis is not on absolute benefits, but rather on the relative changes driven by a policy.

Operationally, the demand functions must first be inverted in order to solve them for p before integrating (Griffin, 2006). The linear becomes $p = \frac{(w-b)}{\alpha}$, and the exponential demand function $p = \left(\frac{w}{k}\right)^{\frac{1}{\varepsilon}}$. This procedure will have to be performed on the adjusted demand curve after the function with the best fit has been determined. Suppose a quadratic function best fits the scatter plot of the mean points (p_i,w_i), which, as previously mentioned, are those closer to the equilibrium point (p,w), then its general form with respect to p yields $p = Aw^2 + Bw + C$, where $A \neq 0$ and A, B, $C \in \mathbb{R}$. Then, with some rearrangements, the total benefits (the areas beneath the linear, the exponential and the adjusted demand functions) are, respectively:

$$A_{lin} = \frac{1}{\alpha} \cdot \int_0^{w_0} (w - b)dw \qquad \text{Equation 6.8}$$

$$A_{exp} = \int_0^{w_0} \left(\frac{w}{k}\right)^{\frac{1}{\varepsilon}} dw \qquad \text{Equation 6.9}$$

$$A_{adj} = \int_0^{w_0} (Aw^2 + Bw + C)dw \qquad \text{Equation 6.10}$$

Following the steps above, the consumer surplus can be determined simply by subtracting the area below the market price or equilibrium point (p_0,w_0):

$$CS_{linr} = \left[\frac{1}{\alpha} \cdot \int_0^{w_0} (w - b)dw\right] - (p_0 \cdot w_0) \qquad \text{Equation 6.11}$$

$$CS_{exp} = \left[\int_0^{w_0} \left(\frac{w}{k}\right)^{\frac{1}{\varepsilon}} dw\right] - (p_0 \cdot w_0) \qquad \text{Equation 6.12}$$

$$CS_{adj} = \left[\int_0^{w_0} (Aw^2 + Bw + C)dw\right] - (p_0 \cdot w_0) \qquad \text{Equation 6.13}$$

A change in price (Figure 6.3B) yields a new point (p_1,w_1), (p_2,w_2) or (p_n,w_n) and a change in the consumer surplus. Consumer surplus measures how much better off people are, in the aggregate, because they can buy potable water. Because different individuals place different values on the consumption of a particular quantity of water, the maximum amount they are willing to pay for that water also differs. Figure 6.3B depicts the consumer surplus when price of potable water (tariff) is at p_0 and consumers are consuming w_0, which is the sum of areas A1 and A2. However, when price moves up to p_1, the amount of water consumed by individuals, in the aggregate, falls back to w_1 and the consumer surplus is reduced to A1. On the contrary, if tariff were to fall to p_2 people would be consuming more water and the total demand would reach w_2, thus consumer surplus would increase to the sum of A1, A2 and A3.

Even an efficient tariff (p_Φ) is likely to cause deviation along the demand curve. In any case, consumer surplus will inevitably vary. The difference in consumer surplus, due to the fluctuation on water tariff as a result of a policy action, is the benefit (gained or lost) that this Chapter is focused on measuring. Nonetheless, the challenge lies in the deep uncertainty associated with price and water quantity variations, as decision-making in urban water management relies on anticipating these changes. This anticipation is becoming increasingly difficult (Marchau et al., 2019), because climate change, future water demand, people's interests, financial crises and local land use policy are getting more and more unpredictable. The core of this approach is that it allows using scenarios that will produce unique efficient tariffs (p_Φ), and different consumer surplus that can generate distinct benefits.

6.3 Results and Discussion

6.3.1 Economic (θ_1) and environmental (θ_2) efficiencies

The calculated economic (θ_1) and the environmental efficiency (θ_2) of the 15 DMUs appear in Table 6.2. Results indicate that Dal Bó for the periods of 2011 (DMU12) and 2018 (DMU10) presented the highest economic efficiencies, suggesting that these DMUs were able to generate more GDP with the least productive area and quantity of jobs, while Samuara-2018 (DMU13) has the next highest economic efficiency (θ_1=0.815). The results demonstrate, by the DEA-SBEI methodology, that the highest values of economic inefficiency are assigned to the urban watersheds. As the urban watersheds are circumscribed, total or in part, within the urban perimeter with access to industry, commerce and service, they generate more revenue per input used in the economic process when compared to the rural DMUs. Despite this finding, DMU8 DMU9 and DMU14 (which are also urban) were not able to present an economic efficiency higher than 0.5, which demonstrates an excessive use of inputs to obtain a given GDP when compared to the benchmark DMUs (which have efficiency θ_1 =1). In general, the desirable product (GDP) of the rural watersheds is below the urban ones, despite the significantly larger area of the former.

The presence of the undesirable product (land use change that increases runoff) in the production process rearranged the inefficiency rank. The environmental efficiency of the DMUs posed a different behavior, with some of the rural watersheds presenting higher environmental than economic efficiency. Marrecas 2011 (DMU6), for instance, is efficient when the undesirable product was considered in the production possibility set. Given the higher the impervious area, the more undesirable product a given DMU is producing, rural DMUs were able to reduce their inefficiency (or improve their efficiency) significantly since their undesirable product is lower. Nonetheless, two rural DMUs still presented environmental inefficiency: DMU2 and DMU5. The case of DMU5 (Marrecas-2014), in special, can be explained by the fact that it presented one of the highest production areas (input), but with the lowest GDP (desirable output), even though the undesirable output (impervious area) did not vary significantly within 2011 and 2018.

Table 6.2. The economic and environmental efficiencies of the 15 DMUs.

DMU	Name	Zoning	θ_1	θ_2	SBEI	[1]OC	[1]OC/ha
1	*Faxinal-2018*	Rural	0.440	0.800	0.550	19,176	2.9
2	*Faxinal-2014*	Rural	0.234	0.395	0.593	888	1.3
3	*Faxinal-2011*	Rural	0.271	0.620	0.437	12,301	1.8
4	*Marrecas-2018*	Rural	0.310	0.882	0.352	13,774	2.6
5	*Marrecas-2014*	Rural	0.123	0.388	0.318	5,450	1
6	*Marrecas-2011*	Rural	0.238	1.000	0.238	10,040	1.9
7	*Maestra-2018*	Urban	0.713	0.752	0.948	4,115	2.7
8	*Maestra-2014*	Urban	0.469	0.469	1.000	0	0
9	*Maestra-2011*	Urban	0.342	0.567	0.603	12,181	8
10	*Dal Bó-2018*	Urban	1.000	1.000	1.000	0	0
11	*Dal Bó-2014*	Urban	0.761	0.787	0.967	2,269	3.6
12	*Dal Bó-2011*	Urban	1.000	1.000	1.000	0	0
13	*Samuara-2018*	Urban	0.815	1.000	0.815	7,774	11.6
14	*Samuara-2014*	Urban	0.484	0.588	0.824	4,647	6.9
15	*Samuara-2011*	Urban	0.542	1.000	0.542	12,946	19.3

[1]Values expresses in xUSD1,000 per year.
θ_1=economic efficiency. θ_2=environmental efficiency. SBEI=slack based efficiency index. OC=opportunity cost. OC/ha=opportunity cost per hectare.

The results indicate that the opportunity cost of conservation, when measured per unit area, is higher in urban watersheds compared to rural ones. This raises concerns about the feasibility of maintaining urban watersheds as a source of raw water. However, it is important to assess the risk associated with ensuring a reliable water supply, particularly during drier periods, before considering such a scenario in the decision-making process. It should be noted that the evaluation of supply guarantees is beyond the scope of this thesis.

6.3.2 The opportunity cost of conservation

In general, in absolute values the rural watersheds present opportunity costs of conservation about twice the cost of the urban watersheds (10.3 USD million/year vs. 4.9 USD million/year, respectively). However, in terms of unit area the result is the opposite: the opportunity cost of conservation is 1.9 USD thousands/hectare·year for rural watersheds and 5.8 USD thousands/ hectare·year for the urban watersheds. This presents a challenge for policies to constrain occupation and preserve some of the urban watersheds as water providers.

The ratio of both economic and environmental efficiencies is presented as the SBEI, which varied from 0.238 to 1.000. When SBEI = 1, it implies that the environmental regulation (restriction on land use) is not forfeiting economic production, while SBEI < 1 indicates that the environmental regulation constrains the undesirable product (impervious area) leading to losses in the production capacity of the desirable product (GDP) (Campos, 2015; Zhou et al., 2006). Hence, when SBEI < 1, there is opportunity cost. Table 6.2 shows that the effect of land use restriction is causing no efficiency change in DMU8, DMU10 and DMU12. Conversely, the rural watersheds had the lowest SBEI values, indicating that environmental regulation reduces the production capacity of the desirable output in these watersheds.

The environmental regulation imposes different annual OC on the DMUs studied, with values ranging from 0 to 19.3 thousand USD per hectare. Despite the three highest OC/ha take place on the urban watersheds, and its overall higher average OC/ha compared to the rural watersheds, the only two watersheds where the OC is equal to zero are also urban. The Dal Bó watershed, the one closest to the city center, had zero OC in 2011 and 2018. These results indicate that urban watersheds can be very efficient in dealing with undesirable products and that regulating its land use to promote watershed conservation and maintain local water supply sources may be possible without significantly impacting their economy. Another reason for the relative low OCs in the urban watersheds is that urban areas have to respect lower buffer zones to protect water resources or, not rarely, these resources no longer exist in urban sites because they have been filled in or channeled in the past to give away space to construction and urbanization. Hence, this reduction in conservation buffer zone is converted to productive area.

6.3.3 The benefits of watershed conservation

To assess the benefits of watershed conservation in the study area, the point expansion method was initially utilized to derive the Marshallian demand curve for treated water. The scarcity rent for the available water resources was then incorporated into the economic equilibrium price. Subsequently, changes in consumer surplus were quantified. There will be a benefit when consumer surplus is monetarily higher after a decision has been made, when compared to the preceding state Additionally, a lower monetary loss in consumer surplus under a particular watershed conservation scenario, compared to other scenarios, is also considered a benefit.

Provided with an average price (p = 7.98 BRL·m^{-3}), water quantity (w = 1.83 hm^3·month^{-1}), and price elasticity of demand (ε = -0.27), the linear demand function for the study area can be represented by Equation 6.14.

$$w = -61{,}008 \cdot p + 2{,}289{,}957 \qquad \text{Equation 6.14}$$

Solving it for p it yields Equation 6.15.

$$p = -1.64 \times 10^{-5} \cdot w + 37.54, \qquad \text{Equation 6.15}$$

where p is expressed in BRL·m^{-3} and w in m^3·month^{-1}. Whereas the exponential demand function was defined as Equation 6.16.

$$w = 3{,}159{,}108 \cdot p^{-0.27} \qquad \text{Equation 6.16}$$

Solving it for p it yields Equation 6.17.

$$p = 1.18 \times 10^{24} \cdot w^{-3.7037} \qquad \text{Equation 6.17}$$

Mathematically, the linear demand function results in a chocking price of 7.5 USD.m^{-3} when supply is null. This can be considered low in a scenario of severe scarcity, given the recent water crisis in São Paulo State where prices were observed from 11.6 USD.m^{-3} to 26.6 USD.m^{-3} (Carvalho, 2014), when water was also supplied by trucks. Moreover, on a more recent internet price survey, it was identified a mean cost of 9.90 USD.m^{-3} for supplying water by tank trucks from a diverse reliable source of potable water. The exponential demand function overestimates price when supply is drastically low, leading to an infinite price when water quantity tends to zero.

The adjusted demand function resulted in Equation 6.18, showing a very high fit, with a R^2 equals to 0.9994.

$$w = 588.7 \cdot p^2 - 67{,}378.6 \cdot p + 2{,}331{,}539, \qquad \text{Equation 6.18}$$

When solving it for p, it becomes Equation 6.19.

$$p = 6.66 \times 10^{-12} \cdot w^2 - 4.06 \times 10^{-5} \cdot w + 59.79 \qquad \text{Equation 6.19}$$

Where p is expressed in BRL·m^{-3} and w in m³·month^{-1}. This presented a chocking price of 12 USD.m^{-3} which better fits observation in real conditions. Figure 6.5 depicts the mean points, restricted to the interval p = [0,36] and w = [1000000,2400000], close to the economic equilibrium point, P_k (BRL7.98,1.83hm³). Note that the problem of infinity of the mean points, when p→0 or w→0, is resolved when an equation is fit to the mean points closer to the known point, P_k.

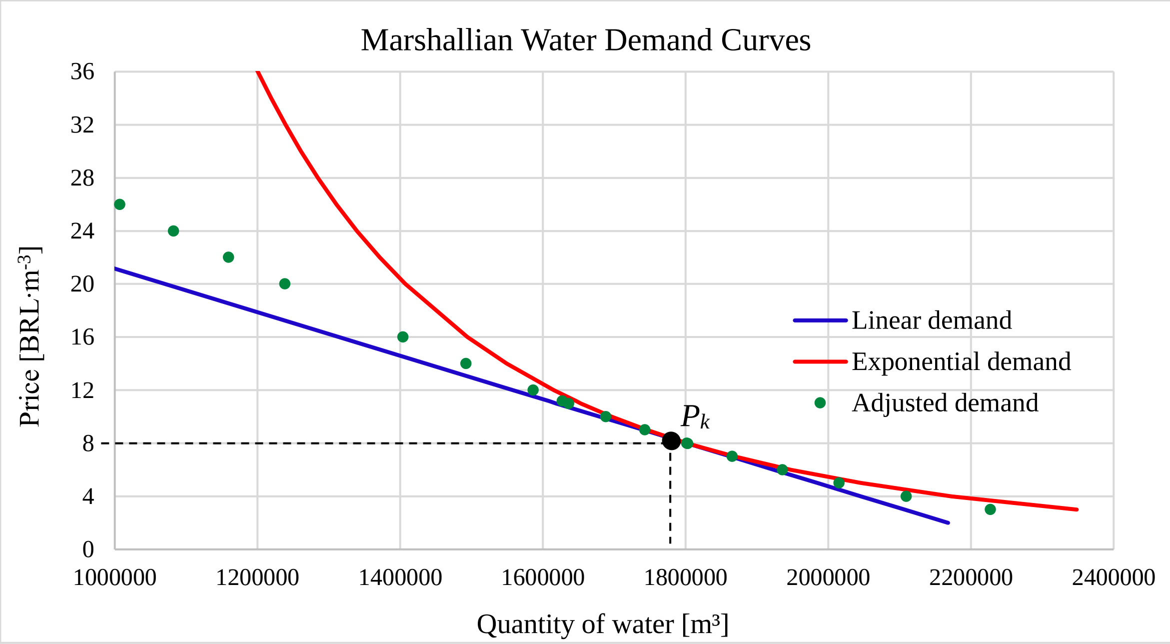

Figure 6.5. The demand curves for the city of Caxias do Sul.
The economic equilibrium point (P_k) corresponds to February of 2022.

6.3.3.1 Adding the water price to the economic equilibrium point (market price)

In scenarios in which the watersheds responsible for supplying water to the city of Caxias do Sul face quality degradation due to anthropogenic pressure, the scarcity rent (Φ) varies based upon the moment (T) the backstop technology is operational, which causes the efficient price (p) at current time (t_0) to also vary. T is scenario-based, as one cannot foresee precisely the moment an advanced treatment will have to be added to the existing train of the water treatment plant. Watershed conservation involves plenty of uncertainties. Thus, Φ is calculated in seven particular scenarios, each of which considers a variation of 5 years for the backstop technology to be operational. It is assumed a scenario in which, in 5 years from t_0, there will be a change in the marginal cost of extraction, then another scenario in which a change will only happen in 10 years and so on, until a last scenario in which watershed conservation postpones in 35 years a change in the marginal cost. The backstop technology considered in this study is membrane filtration.

To determine Φ a yearly discount rate of 2% (Arrow et al., 2013; Moncur and Pollock, 1988; Simpson, 2008; Weitzman, 1994) was applied to all seven scenarios. The current marginal cost of extraction (C_1 in 2022) was 0.197 USD·m^{-3}, and by adding membrane filtration to the treatment train, the marginal cost of extraction (C_2) increases to 0.420 USD·m^{-3}, as presented in Chapter 4. Table 6.3 shows the seven scenarios for scarcity rent for Caxias do Sul water.

Table 6.3. Efficient price (p) and in-situ (Φ) values for constant marginal cost of extraction.

		Scenario 1 T=5		Scenario 2 T=10		Scenario 3 T=15		Scenario 4 T=20		Scenario 5 T=25		Scenario 6 T=30		Scenario 7 T=35	
t	C_1	Φ	p	Φ	p	Φ	p	Φ	p	Φ	p	Φ	p	Φ	p
0	0.197	0.202	0.399	0.183	0.380	0.165	0.362	0.149	0.346	0.135	0.332	0.122	0.319	0.111	0.308
1	0.197	0.206	0.403	0.186	0.383	0.169	0.366	0.153	0.350	0.138	0.335	0.125	0.322	0.113	0.310
2	0.197	0.210	0.407	0.190	0.387	0.172	0.369	0.156	0.353	0.141	0.338	0.127	0.324	0.115	0.312
3	0.197	0.214	0.411	0.194	0.391	0.175	0.372	0.159	0.356	0.144	0.341	0.130	0.327	0.118	0.315
4	0.197	0.219	0.416	0.198	0.395	0.179	0.376	0.162	0.359	0.147	0.344	0.133	0.330	0.120	0.317
5	0.197	0.223	0.420	0.202	0.399	0.183	0.380	0.165	0.362	0.149	0.346	0.135	0.332	0.122	0.319
6	0.197	0.00	0.420	0.206	0.403	0.186	0.383	0.169	0.366	0.153	0.350	0.138	0.335	0.125	0.322
7	0.197			0.210	0.407	0.190	0.387	0.172	0.369	0.156	0.353	0.141	0.338	0.127	0.324
8	0.197			0.214	0.411	0.194	0.391	0.175	0.372	0.159	0.356	0.144	0.341	0.130	0.327
9	0.197			0.219	0.416	0.198	0.395	0.179	0.376	0.162	0.359	0.147	0.344	0.133	0.330
10	0.197			0.223	0.420	0.202	0.399	0.183	0.380	0.165	0.362	0.149	0.346	0.135	0.332
11	0.197			0.00	0.420	0.206	0.403	0.186	0.383	0.169	0.366	0.153	0.350	0.138	0.335
12	0.197					0.210	0.407	0.190	0.387	0.172	0.369	0.156	0.353	0.141	0.338
13	0.197					0.214	0.411	0.194	0.391	0.175	0.372	0.159	0.356	0.144	0.341
14	0.197					0.219	0.416	0.198	0.395	0.179	0.376	0.162	0.359	0.147	0.344
15	0.197					0.223	0.420	0.202	0.399	0.183	0.380	0.165	0.362	0.149	0.346
16	0.197					0.00	0.420	0.206	0.403	0.186	0.383	0.169	0.366	0.153	0.350
17	0.197							0.210	0.407	0.190	0.387	0.172	0.369	0.156	0.353
18	0.197							0.214	0.411	0.194	0.391	0.175	0.372	0.159	0.356
19	0.197							0.219	0.416	0.198	0.395	0.179	0.376	0.162	0.359
20	0.197							0.223	0.420	0.202	0.399	0.183	0.380	0.165	0.362
21	0.197							0.00	0.420	0.206	0.403	0.186	0.383	0.169	0.366
22	0.197									0.210	0.407	0.190	0.387	0.172	0.369
23	0.197									0.214	0.411	0.194	0.391	0.175	0.372
24	0.197									0.219	0.416	0.198	0.395	0.179	0.376
25	0.197									0.223	0.420	0.202	0.399	0.183	0.380
26	0.197									0.00	0.420	0.206	0.403	0.186	0.383
27	0.197											0.210	0.407	0.190	0.387
28	0.197											0.214	0.411	0.194	0.391
29	0.197											0.219	0.416	0.198	0.395
30	0.197											0.223	0.420	0.202	0.399
31	0.197											0.00	0.420	0.206	0.403
32	0.197													0.210	0.407
33	0.197													0.214	0.411
34	0.197													0.219	0.416
35	0.197													0.223	0.420
	0.197													0.00	0.420

Values expressed in USD.

The values presented in Table 6.3 were obtained assuming different time periods (5, 10, 15, 20, 25, 30, or 35 years) for the implementation of membrane filtration technology. These estimates indicate that, considering a constant marginal cost of extraction, the in-situ value of surface water ranges from 0.11 to 0.20 USD·m^{-3}, as of 2022 (t_0). This value is expected to increase as we approach the time when the backstop technology becomes a reality, which is the point at which the efficient price (*p*) equals the new marginal cost of extraction (*C2*).

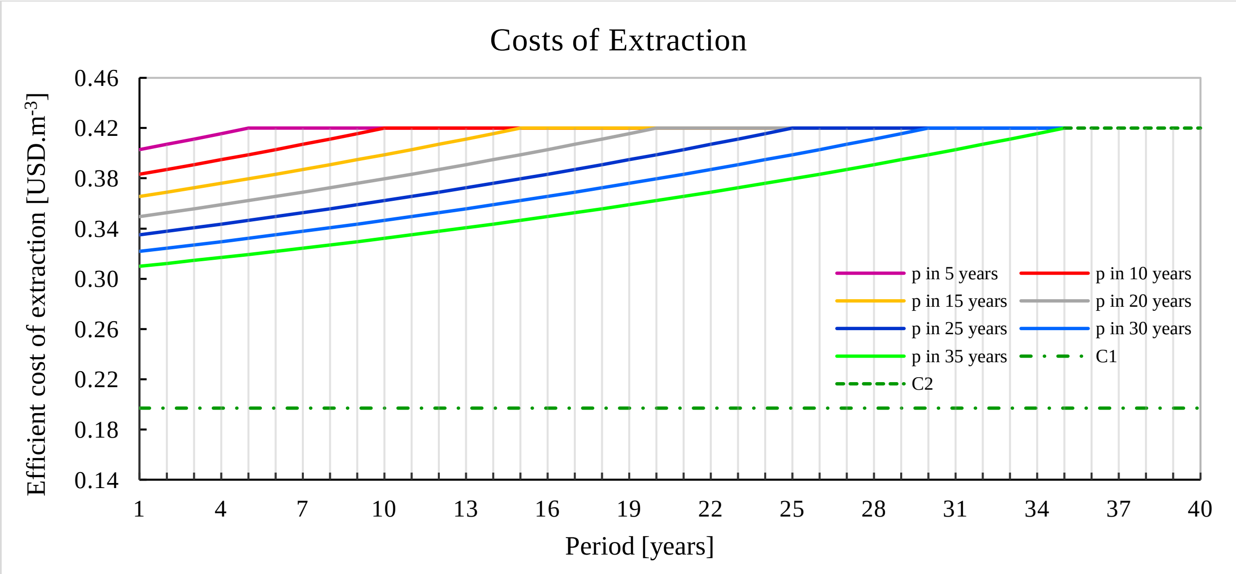

Figure 6.6. Evolution of incremental cost of extraction.
Evolution of costs of extraction towards the backstop technology in 35 years from 2015.

Table 6.3 and Figure 6.6 illustrate the relationship between the adoption of a backstop technology (e.g., membrane filtration) and its impact on the present value and potable water tariff. It demonstrates that postponing the implementation of the backstop technology leads to a decrease in the present value and reduces the impact on the water tariff. In cases where land use policies are implemented to prevent any future degradation of water quality, the scarcity rent associated with improving treatment efficiency becomes zero ($\Phi = 0$). This is similar to the concept described by Moncur and Pollock (1988), where areas with abundant water resources, such as the Amazon river basin, have a zero in-situ value for water because it is considered virtually unlimited in terms of quantity.

The scarcity rent for water is incorporated into the tariff to determine an efficient tariff (pΦ) for each of the seven scenarios, as presented in Table 6.4. In order to account for the water price (or in-situ value of water, source water value, or scarcity rent for water), the known equilibrium point, P_k (7.98 BRL, 1.83 hm^3), needs to be adjusted, hence resulting in P_e.

Table 6.4. New (adjusted) equilibrium point, P_e.

Scenario	Φ	P_e Price (p_Φ) at t_0	Quantity
No water quality degradation	0.00	1.54	1.830
T=5	0.20	1.74	1.771
T=10	0.18	1.72	1.777
T=15	0.16	1.71	1.780
T=20	0.15	1.69	1.786
T=25	0.13	1.68	1.789
T=30	0.12	1.66	1.795
T=35	0.11	1.65	1.798

Prices are given in USD·m^{-3} and quantity expressed in hm^3 for February/2022. An exchange rate of 1USD=5.19BRL was applied.

From Table 6.4, it becomes evident that the tariff increases as the water price is incorporated, according to each scenario, at the present time (t_0). However, as the operationalization of the backstop technology is postponed, the tariff becomes less expensive at t_0. The quantity demanded is determined based on the adjusted (mean) demand curve. This approach could potentially serve as a water policy instrument for promoting sustainable water use, as it demonstrates the impact on the present-time (t_0) water price and quantity demanded as the in-situ value of water adjusts to reflect future scenarios of water quality degradation. It is important to note that the in-depth exploration of this topic exceeds the scope of this thesis, and further research is recommended to address this gap.

6.3.3.2 Calculating changes in consumer surplus (CS)

The adjusted demand function (Equation 6.19) is integrated over the range between the quantity demanded at the observed value w_i (1.83 million of m^3·month^{-1}) and the new quantity demanded w_f under different (higher) prices corresponding to each distinct future scenario of water quality degradation. This integration results in Equation 6.20, where w represents the monthly consumed water in cubic meters and p_f denotes the new price in Brazilian Real (BRL) of the revised market price.

$$CS = \left(\int_{w_i}^{w_f} 6.66\times10^{-12}\cdot w^2 - 4{,}06\times10^{-5}\cdot w + 59.79\, dw\right) - \left(p_f \times w_f\right) \qquad \text{Equation 6.20}$$

Figure 6.7A visually represents the consumer surplus when the price of potable water was originally USD 1.54, represented by the area A1. However, when the tariff is adjusted to include the scarcity rent, present consumers are required to pay a higher price. This change in the potable water tariff, for example, from USD 1.54 to USD 1.74, results in a reduction of consumer surplus from the area A1 to the area A2, indicating a loss in consumer surplus equal to the area A3 (Figure 6.7B). This change in price corresponds to a shift of the equilibrium point, P_k, to a new equilibrium point P_e, where potable water consumption decreases from 1.83 to 1.77 hm^3. Consequently, the total benefit is reduced as the area under the marginal benefit curve is decreased. It is important to note that if the quality of raw water remains unchanged, P_e will coincide with P_k, and there will be no loss in consumer surplus, further highlighting the significance of watershed conservation.

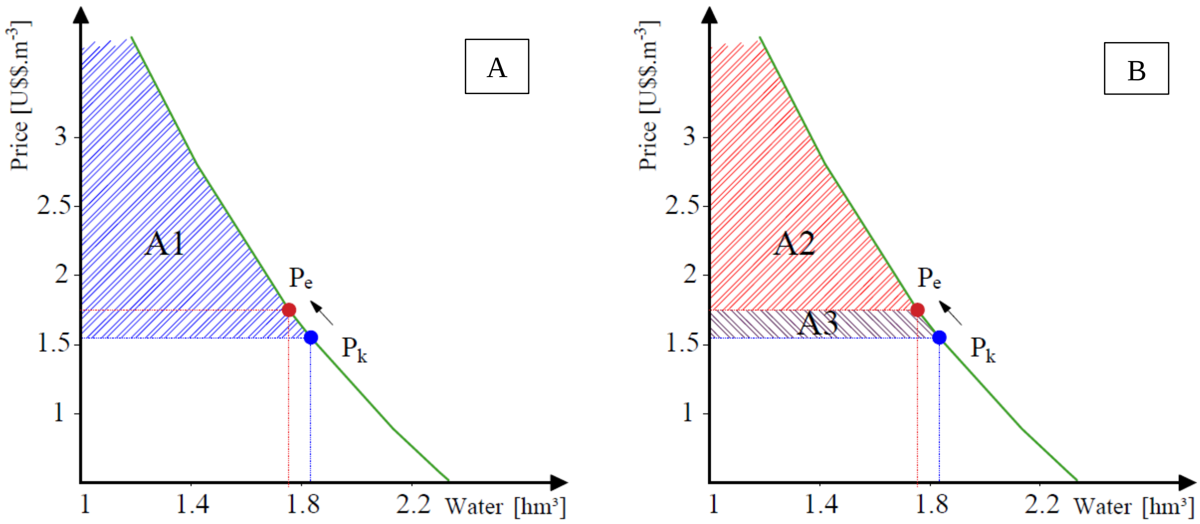

Figure 6.7. Changes in consumer surplus.

A scenario-based approach provides a comprehensive understanding of potential future outcomes and the effectiveness of present decisions. In this Chapter, seven scenarios were considered to forecast different present values based on the time when a backstop technology (membrane filtration) would need to be implemented to support the current water treatment process in addressing watershed degradation and ensuring potability standards. If, at current conservation status, the consumer surplus is the area A1, another conservation status could be represented by the area A2 (Figure 6.7), and so on for all other six scenarios. The resulting consumer surplus (CS) values for each scenario and their changes (ΔCS) are presented in Table 6.5.

Among the seven scenarios, the one in which water quality degradation occurs earlier, requiring the implementation of the backstop technology in five years, leads to the greatest loss in consumer surplus, amounting to USD 357,451, as of February 2022. This represents a decrease of 4.5% in economic welfare compared to the current observed conditions. On the other hand, the best alternative scenario considered is the one where water quality can be maintained through effective conservation policies, resulting in a delay of the backstop technology to T=35. In this scenario, the welfare loss is 2.5% (USD 198,141), relative to the month with available data, which is February 2022.

Table 6.5. Consumer surplus for the seven scenarios in February 2022.

n	Scenario	[1]CS	[2]ΔCS	% change
0	Current policy (t_0)	7,786,437	0	
1	T=5	7,428,986	-357,451	-4.6%
2	T=10	7,464,173	-322,264	-4.1%
3	T=15	7,481,813	-304,624	-3.9%
4	T=20	7,517,183	-269,254	-3.5%
5	T=25	7,534,915	-251,522	-3.2%
6	T=30	7,570,471	-215,966	-2.8%
7	T=35	7,588,296	-198,141	-2.5%

[1]Consumer Surplus for February of 2022. [2] Loss (–) or gain (+) in consumer surplus: $\Delta CS = CS_{(t0)} - CS_{(t0+n)}$. Data for the city of Caxias do Sul, Southern Brazil. Values have been converted to USD, following an exchange rate of 1USD=5.19BRL. T is expressed in years.

Conversely, the existing watershed conservation policy in the studied area leads to the highest consumer surplus. These findings confirm that by delaying additional future costs, the present value increases, and consumers are better off in the current time period compared to the near future. This serves as a valid approach for quantifying the economic benefits associated with watershed conservation. The results highlight the importance of proactive conservation measures in maintaining economic welfare and ensuring favorable outcomes for consumers.

6.3.4 Comparison between the cost and the benefit of watershed conservation

While fostering watershed conservation provides a clear economic benefit to consumers in the form of reducing the consumer surplus loss, it also entails an opportunity cost, particularly for landowners who face limitations on the complete use of their land.

Considering the current water tariff structure in Caxias do Sul, which does not differentiate the extraction costs among the five systems (Faxinal, Marrecas, Maestra, Dal Bó, and Samuara), but instead it is charged evenly based on local decree (Caxias do Sul, 2022), the same rationale will be considered to quantify the opportunity cost. The opportunity cost will be calculated based on a mean value for all five watersheds. In the case under study, the local conservation legislation leads to a mean annual foregone GDP, in present value (t_0), of 2.52 million USD for the five watersheds. Unfortunately, due to the application of the DEA-SBEI methodology and data limitations, it was not feasible to assess potential variations in the opportunity cost over time. Therefore, it is assumed to remain constant until the year 2057 (t_7), despite the understanding that more intensive land use would result in a lower opportunity cost.

At the current water tariff (t_0) and under the existing watershed conservation status, the annual consumer surplus amounts to 101.5 million USD. However, when the scarcity rent for water is added to the tariff, the consumer surplus decreases to 96.2 million USD in 2027 (t_1), which corresponds to the scenario where the backstop technology needs to be operational sooner, indicating the worst-case watershed conservation practice scenario. On the other hand, if the implementation of the backstop technology is postponed to 2057 (t_7), the consumer surplus increases to 98.9 million USD, as of 2022 (t_0), representing the scenario with the least loss in benefits.

The evaluation of the benefit of conservation is based on the magnitude of the aggregate consumer surplus loss associated with different decision outcomes. A careless decision regarding watershed conservation that would require the membrane technology to be functional in 2027 (t_1) would result in a loss of 4.66 million USD, which is approximately 85% higher than the opportunity cost of conservation. Even a slightly less restrictive land use policy in the watershed would lead to a loss of 2.58 million USD in consumer surplus by 2057 (t_7), which is about 2.5% higher than the opportunity cost of conservation.

Table 6.6 provides a summary of the costs and benefits associated with different scenarios of watershed conservation. The third column presents the annual consumer surplus for each scenario, representing the economic benefit derived from watershed conservation. It is evident that the longer the postponement of the backstop technology implementation (as indicated in the fourth column), the smaller the loss in consumer surplus.

Interestingly, maintaining the watershed with the same land use control and quality standards would result in zero loss in consumer surplus. These findings raise the possibility of future discussions regarding the implementation of payment for watershed services. Could a portion of the tariff be allocated to compensate for the opportunity cost of watershed conservation? This question highlights a knowledge gap that requires further research and investigation in the field.

Table 6.6. Annual economic costs and benefits of watershed conservation.

n	t	CS	ΔCS	OC
0	2022	101.502	0	2.520
1	2027	96.842	4.660	2.520
2	2032	97.301	4.201	2.520
3	2037	97.531	3.971	2.520
4	2042	97.992	3.510	2.520
5	2047	98.223	3.279	2.520
6	2052	98.686	2.815	2.520
7	2057	98.919	2.583	2.520

Consumer Surplus (CS) and the opportunity cost (OC) are expressed in x1000 USD. $\Delta CS = CS_{(t0)} - CS_{(t0+n)}$

6.4 Conclusions

This Chapter emphasizes the importance of watershed conservation in order to maintain optimal consumer surplus. Unrestricted land use in the watershed can jeopardize water security, leading to increased extraction costs and higher water tariffs as water quantity and quality decline. This Chapter contributes to the water management literature by providing an economic analysis that quantifies the cost and benefit of watershed conservation. The cost of watershed conservation is determined by the opportunity cost, which is assessed using the slacks-based efficiency index and DEA methodology. On the other hand, the economic benefit is measured by evaluating the loss in consumer surplus. The demand function is derived through the point expansion approach, incorporating the scarcity rent to estimate the economic value of in-situ water.

The research findings highlight that the economic benefit of watershed conservation outweighs its cost. It is evident that the sooner a backstop technology is implemented to support existing treatment processes, the greater the loss in consumer surplus. The case study of Caxias do Sul illustrates a significant opportunity cost, amounting to approximately 2.5 million USD per year. However, if a scenario arises where a membrane filtration system must be operational within a five-year timeframe, the municipality could potentially lose almost double that amount in consumer surplus.

These findings unveil the positive economic balance that can be achieved when policies prioritize source water quality. By investing in watershed conservation, the long-term economic benefits exceed the costs. It is crucial to consider the potential losses in consumer surplus when making decisions regarding the timing and implementation of backstop technologies.

The incorporation of an in-situ water price, known as the scarcity rent, could facilitate a reduction in the present value of the water tariff in the study area. By accounting for the future costs associated with implementing advanced treatment technologies, the need for immediate tariff adjustments may be delayed. Moreover, recognizing the opportunity cost, scarcity rent, and consumer surplus can support the implementation of payment for ecological service programs, which could be specifically referred to as payment for watershed services.

CHAPTER 7

7 ADAPTIVE DECISIONS PATHWAYS IN A WORLD UNDER DEEP UNCERTAINTIES[4]

A B S T R A C T

Water treatment costs represent the most important aspect of the economic dimension of urban water security. Water tariff affordability directly impacts people's access to this essential resource. However, the water supply sector is subject to a myriad of uncertainties. Land use, urbanization, water potability standards, water treatment technology and the economic development are external factors that can compromise decision-making. Robust decisions are important to urban water security, as the trade-offs between optimal land use at the watershed and optimal treatment cost are better evaluated and the outcomes are prone to success across various possible future scenarios. This Chapter addresses the need for robust decision-making in urban water supply by employing the Dynamic Adaptive Policy Pathways (DAPP) methodology, complemented by a heuristic mathematical approach. This novel application focuses specifically on the domains of watershed conservation and water treatment technology. The result is an adaptive decision map that provides insights into possible technological actions that can be taken to address different levels of watershed conservation or degradation. Rather than prescribing a single correct pathway, the decision map helps decision-makers understand the consequences and trade-offs associated with different alternatives. It highlights that there are multiple combinations of actions, each with its own benefits and costs.

[4] This Chapter will be published as *Adaptive decisions on water treatment technologies and watershed quality: shedding light on urban water supply in a deeply uncertain world*.

7.1 Introduction

Land use intensification and urbanization of watersheds have resulted in the introduction of elements that are not naturally found in surface water. Since the first outbreaks of waterborne diseases in the 19th century, regulatory authorities have demanded more stringent water standards, and the general population has become more knowledgeable and discerning about water quality (Crittenden et al., 2012). Furthermore, the number of regulated contaminants has increased throughout the 20th century, particularly after the 1980s (Howe et al., 2012). This situation has fostered the development of new technologies capable of ensuring the proper treatment of raw water and meeting drinking water standards. However, the utilization of these new technologies to achieve increasingly lower thresholds for contaminants entails additional capital, operational, and maintenance costs (McGivney and Kawamura, 2008). These high treatment costs translate into higher water tariffs, which in turn restrict access to potable water for a significant portion of the impoverished population, thereby compromising the economic dimension of urban water security (Aboelnga et al., 2019; ANA, 2019; Pozzebon et al., 2022).

The inclusion of these additional costs, as demonstrated in Chapter 4, can be mitigated through watershed conservation practices. Notwithstanding, Chapter 6 highlights that watershed conservation itself entails an opportunity cost that must be considered in decision-making processes aimed at ensuring long-term urban water security. The opportunity cost of watershed conservation can be understood as the cost associated with options that are forgone by not putting the watershed's resources to their best alternative use (adapted from Pindyck and Rubinfeld, 2018). Decision-making when faced with the trade-off between optimal land use within the watershed and optimal treatment costs is subject to uncertainties arising from various external factors, such as land use regulations, population growth, technological advancements, economic development, and climate change. Decision makers assume that the future can be predicted and shaped as a result of their beliefs (Loucks and van Beek, 2017). However, in a deeply uncertain world, the sequential actions taken over time, referred to as the plan, are likely to failure (Kwakkel et al., 2015). Hence, robust decisions are required to navigate this unpredictability. Robust decisions are those that yield favorable outcomes across a range of possible future scenarios (Marchau et al., 2019), and for every scenario there is a plausible action that can be taken (Kwakkel et al., 2015). Analyzing the plausibility of actions to balance watershed conservation and water treatment costs becomes complex because the plan must adapt over time, depending on transient scenarios.

Adaptive decision-making encompasses an initial sequence of actions aimed at achieving short-term targets, while also allowing for adjustments over time to align with long-term objectives in response to evolving circumstances. These evolving circumstances, referred to as revealing circumstances, can be seen as transient scenarios representing potential future states from the present to a defined point in time that correspond to relevant uncertainties and how they develop over time (Haasnoot et al., 2012). The Dynamic Adaptive Policy Pathways (DAPP) framework is a decision-making tool utilized primarily in the context of uncertainties arising from climate change. Haasnoot (2013) describes DAPP as an analytical planning process that enables the adoption of diverse actions based on the unfolding of external factors. This approach considers transient scenarios and facilitates the implementation of appropriate actions as the future unfolds, ensuring that the plan remains on track toward reaching success.

Previous studies have primarily examined the application of the DAPP framework in water resource planning projects focused on aspects such as river flow (Haasnoot et al., 2019, 2015), flood protection (Lawrence et al., 2019), water availability (Dias et al., 2020) and coastal erosion (Toimil et al., 2021), emphasizing the paradigm of climate change and its implications over a specific time period. However, this Chapter introduces a new perspective by addressing decisions related to watershed conservation and water treatment to ensure urban water security. It presents an innovative approach by showcasing a range of water treatment technologies that can be implemented based on changes in external factors, particularly land use modifications within the source water watersheds. Additionally, this Chapter introduces an innovative strategy for assessing trade-offs in this complex domain, particularly in terms of costs.

A trade-off is a measure of the relationship between changes in the values of different objective functions. Specifically, it quantifies how the increase in one objective function is accompanied by a decrease in the value of another objective function (Miettinen, 1999). This Chapter examines the trade-offs between two objective functions: minimizing the opportunity cost of watershed conservation and reducing the operational and maintenance costs associated with urban water purification. To analyze these trade-offs, it is employed the ε-Constraint multi-objective optimization method (Haimes et al., 1971; Lin, 1976; Miettinen, 1999). This methodological framework was applied in Caxias do Sul, southern Brazil, to develop an adaptive plan for long-term urban water management, taking into account the uncertainties inherent in decision-making under future scenarios. The case study focuses on the operations of a local public utility responsible for managing five distinct surface water treatment systems, comprising both the watershed and the treatment plants.

7.2 Dynamic Adaptive Policy Pathways (DAPP)

The DAPP framework, introduced by Haasnoot (2013), operates on the premise that policies have a limited lifetime and may become ineffective when operating conditions change. It emphasizes the importance of flexibility in planning and adaptability to plausible future scenarios (Haasnoot et al., 2019). If an action fails or as the future unfolds, this method allows the planner to identify additional actions that can be taken to achieve the originally defined objective (success), resulting in a range of potential pathways. A key concept within DAPP is the Adaptation Tipping Point (ATP) (Kwadijk et al., 2010), which acknowledges that policies and decisions have an expiry date and may become ineffective when operating conditions change.

In this thesis, success is defined as the provision of affordable drinking water to the municipality without compromising water security, while also avoiding excessive economic burdens on landowners in the form of opportunity cost. Once these objectives are established, an ATP can be identified as the point at which the magnitude of change resulting from external factors (e.g., economic development, changes in potability regulations, and political choices) is significant enough to render the current management strategy incapable of meeting success. At this stage, an alternative path (i.e., a different policy, decision, or technology) must be pursued. The choice between different paths is subjective and depends on the stakeholders involved, taking into consideration the trade-offs involved.

The DAPP framework consists of ten steps; however, due to the investigative nature of this study, only the initial five steps are applied. Consequently, steps 6 to 10, which entail the practical implementation of DAPP, have not been executed. For more detailed information on the complete DAPP framework, as well as steps 6 to 10, I recommend referring to Haasnoot (2013) and Haasnoot et al. (2019).

Step 1 of the DAPP framework involves providing a detailed description of the study area's context and system constraints. Additionally, this step focuses on defining the criteria for success within the given context. In Step 2, a comparison is made between the current situation and a range of plausible future scenarios to identify the point at which an action begins to fall short of achieving success, that is, when it is close to an ATP. Step 3 entails the identification of potential policies or measures that can be implemented to achieve success in the given context. In Step 4, the identified policies are evaluated and assessed to determine their effectiveness. Based on this evaluation, pathways can be designed if the set of policy actions is deemed appropriate. When an action reaches its ATP, an alternative pathway must be chosen to ensure success can still be achieved. The outcome of Step 5 is the creation of a pathways map (e.g., Figure 7.1), which resembles a metro map. This map provides a visual representation of all the policy actions and potential logical pathways that lead to success under changing conditions.

Based on the pathways map, Step 5 involves designing an adaptive plan that incorporates the best initial and long-term actions to achieve success. This plan takes into account the dynamic nature of the system and allows for adjustments and adaptations over time.

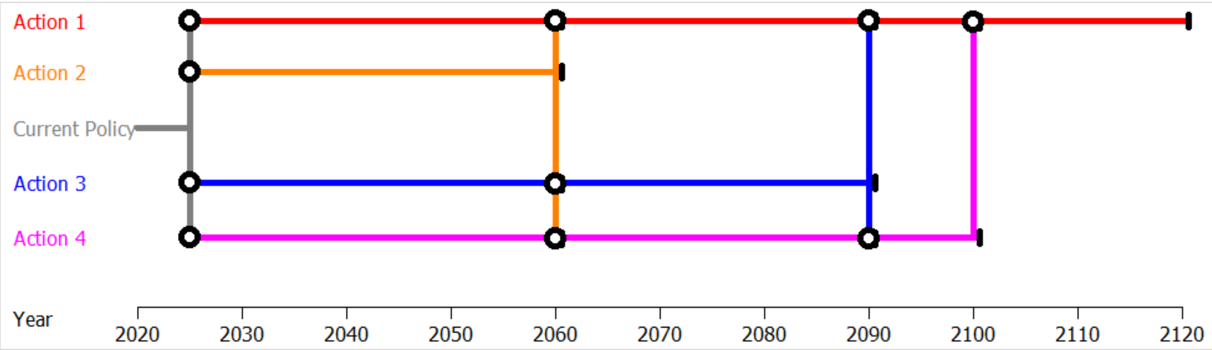

Figure 7.1. A generic example of a pathways map.

The map depicted in Figure 7.1 is time-based, as it illustrates the progression of actions over time in response to changing conditions. It showcases seven potential pathways that can lead to achieving the desired success or target by the year 2120. According to the map, the effectiveness of the current policy diminishes after five years, highlighting the need for alternative actions. In 2025, there are four different actions that can be taken to strive towards success. However, by 2060, an ATP is reached, indicating that Action 2 is no longer viable. At this point, the decision-maker must choose between Action 1, 3, or 4 as a replacement. Suppose Action 3 is selected instead of the other two alternatives. In that case, another ATP occurs in 2090, necessitating a decision between reverting to either Action 1 or Action 4. Opting for Action 1 in 2025, instead of the other three actions, guarantees the achievement of success by 2120. However, it is important to note that the present costs associated with Action 1 might be prohibitively high, potentially making investments in this option unattractive.

7.3 Application of DAPP

This section provides a comprehensive account of the application of the initial five steps of the DAPP approach in the context of the water supply systems in Caxias do Sul, southern Brazil. The primary objective of this application is to facilitate decision-making processes in the field. The implementation of the DAPP approach encompasses essential activities, including data collection and analysis, along with the utilization of a supporting approach that facilitates the identification and development of promising pathways within the urban water supply domain.

7.3.1 Steps 1 and 2: Current situation, success and uncertainties

Steps 1 and 2 entail describing the current situation, defining the criteria for success, and identifying the uncertainties associated with the urban water supply in the study area.

In this study, the public water supply system of Caxias do Sul is utilized as a testing ground to assess the applicability of the proposed method. Comprehensive information regarding the study area is provided in Chapter 3.

The capacity of each water supply system is influenced by factors such as the availability of water in the watershed and the ability to regulate flow through impoundments. Additionally, population growth has had a substantial impact on determining the capacity of these systems. Despite population growth over the years, which has had a significant impact on the capacity of the water supply systems, it is important to note that the mean annual water consumption remains closely linked to population growth. This means that as the population increases, so does the demand for water. However, it is worth highlighting that the city also faces challenges related to distribution water losses, which have reached a rate of 47.7% (MDR, 2022).

Equation 7.1 represents the total volume of water demanded in the city of Caxias do Sul, accounting for geometric growth projections (Tsutiya, 2005) and considering distribution water losses. This equation was established based on data from the 2000 and 2010 censuses, which reported populations of 360,223 and 435,564 inhabitants, respectively (IBGE, 2021d). As per the assumption based on ABNT (1993) standards, the average individual is expected to consume approximately 160 liters of water per day. Furthermore, it is important to highlight that surface water is accessible to 99% of the municipality, as reported by SAMAE (2022a). However, in sparsely populated rural areas, the remaining population relies on groundwater sources to meet their water supply needs.

$$V_t = 25{,}199{,}817 \times e^{0.0189 \times (t-2010)}, \qquad \text{Equation 7.1}$$

where t is the year being analyzed and V_t is given in m³ per year (t).

Table 7.1 provides a summary of key figures associated with each system. The water availability of the watersheds is reported to correspond to 50% of the long-term mean discharge. The treatment capacity represents the maximum capacity of each water treatment plant, based on the existing treatment train, while the population served refers specifically to the urban population, as the distribution systems are not operational in rural locations.

Table 7.1. Water treatment systems in Caxias do Sul.

System	[1] Construction Year	[1] Watershed Area [ha]	[2] Water Availability [$L.s^{-1}$]	Treatment Capacity [$L.s^{-1}$]	Population Served (%)
Faxinal (Fax)	1992	6,679	2,046	1,100	54
Marrecas (Mar)	2012	5,325	1,611	760	22
Maestra (Mae)	1971	1,526	489	325	19
Dal Bó (Dal)	1928	643	220	80	4
Samuara (Sam)	1957	687	234	38	1

Source: [1] (SAMAE, 2022b), [2] (Pozzebon et al., 2021).

The primary objective of any water utility is to provide an adequate amount of drinking water that meets potability standards and is affordable for consumers, while fostering access to services for the poor. In the case of Caxias do Sul, the success of the water supply system can be defined as follows: the plan will be considered successful if it is able to supply drinking water to meet demand at an affordable price for the next 35 years.

For the purpose of this Chapter, annual tariff adjustments due to inflation will not be taken into account. Instead, costs will be represented in present value, specifically as of the year 2022. One of the constraints considered is to minimize the opportunity cost, which means that landowners in the watershed should not face additional restrictions that could negatively impact source water quality and increase operational and maintenance (O&M) treatment costs. This constraint implies that an additional target must be added to the definition of success.

Consequently, the plan will be considered fully successful if drinking water can be supplied at an affordable tariff for a time horizon of 35 years, without imposing additional opportunity cost on landowners in the source watershed. Conversely, policies will be deemed to have failed if these targets are not achieved.

Population growth, land use regulation and future socio-economic development are major uncertainties that need to be accounted for when deciding what actions are applicable to reach this objective. Changes in the watershed, such as alterations in soil imperviousness or land use modifications, can give rise to a range of plausible future scenarios. In this study, a model is employed that integrates all these uncertainties into a single variable known as the Curve Number (CN).

The CN is an empirical hydrological parameter commonly utilized to estimate the volume of runoff generated from a rainfall event (Mishra and Singh, 2003). It serves as a surrogate for uncertainty because it is influenced by factors that represent changes in the watershed. The value of

CN is influenced by key watershed characteristics, including vegetation cover and land use, which are closely tied to the aforementioned uncertainties. The CN parameter ranges from 0 to 100, where 0 represents no anthropogenic use and complete permeability, while 100 represents a scenario of full watershed utilization. Each value of CN is a possible future, which may necessitate adjustments to the water supply management system and the implementation of specific actions to address vulnerabilities.

Monitoring land use changes in the watersheds is essential and can be achieved through methods such as remote sensing or on-site control and surveillance. This monitoring serves as a signpost to help identify ATPs, indicating when current actions may no longer lead to success.

The factors that influence the value of CN can also have an impact on water quality within the watershed. For example, the expansion of urban areas and agricultural land can lead to increased concentrations of total phosphorus, while grassland and forest areas tend to reduce nitrogen concentrations (Lu et al., 2021). General change in land use at the watershed can explain shifts in water quality indexes (Abbasi and Abbasi, 2012).

One of the challenges lies in determining the extent to which a unit change in CN affects water quality. This is because different types of settlements produce varying pollutants, and comprehensive and reliable data on water quality is not widely available. To address this knowledge gap, a data-driven approach was developed, utilizing available data from the study area for the periods of 2011, 2014, and 2018. The approach involved correlating changes in the watershed, such as land use changes, with water quality parameters represented in an index form.

By establishing correlations between these watershed changes and water quality indexes, a better understanding of the relationship between CN and water quality can be obtained. This approach helps bridge the knowledge gap and provides valuable insights into the potential impacts of land use changes on water quality within the study area.

A water quality index serves as a useful tool to simplify complex water quality data and enhance the understanding of raw water quality for both experts and the general public. It also acts as a proxy for assessing watershed quality. In this analysis, the water quality index proposed by CCME (2001), also referred to as the water compliance index (WCI) (Amaro and Porto, 2009; Oliveira et al., 2018), is applicable. The WCI integrates parameters typically monitored by water utilities, making it suitable for this study.

It this study the WCI is composed based on six parameters, as detailed in Chapter 4, which are universally significant in water treatment plants and represent a combination of biological, chemical, and physical pollutants. These parameters include dissolved oxygen (DO), cyanobacteria, turbidity, manganese, nitrite, and pH. By considering these parameters, the WCI is capable of providing a comprehensive assessment of water quality. The WCI is measured on a scale ranging from 0 to 100, where 0 indicates the poorest possible water quality and 100 represents the highest level of water quality.

7.3.1.1 Correlation between land use (CN), watershed quality (WCI) and opportunity cost (OC)

In the case of Caxias do Sul water systems, both watershed quality and opportunity cost are considered as functions of land use, *WCI=f(CN)* and *OC=f(CN)*. However, the limited availability of data for small localities, such as the watersheds examined in this study (ranging from 5 to 100 km²), poses a challenge for the application of dynamic adaptive planning methodologies. Consequently, the dearth of data for the study area made it difficult to establish a robust correlation between these variables.

To address this challenge, the OC associated with watershed conservation was obtained from the results presented in Chapter 6, which provides insights into the opportunity cost of implementing conservation measures. Additionally, changes in the WCI from 2011 to 2019 in the source water watersheds were identified in Chapter 4, offering information on variations in water quality over time. Figure 7.2 compiles these available data and presents their linear correlation.

System	WCI	CN
Fax2011	57.07	75.17
Fax2014	69.13	74.42
Fax2018	68.87	74.96
Mar2018	78.16	73.95
Mae2011	53.29	78.24
Mae2018	66.31	77.29
Dal2014	80.49	78.62
Sam2011	88.46	66.42
Sam2014	80.07	68.92
Sam2018	75.94	66.47

System	OC	CN
Fax2011	1.8	75.17
Fax2014	1.3	74.42
Fax2018	2.9	74.96
Mar2018	2.6	73.95
Mae2011	8	78.24
Mae2018	2.7	77.29
Dal2014	3.6	78.62
Sam2011	19.3	66.42
Sam2014	6.9	68.92
Sam2018	11.6	66.47

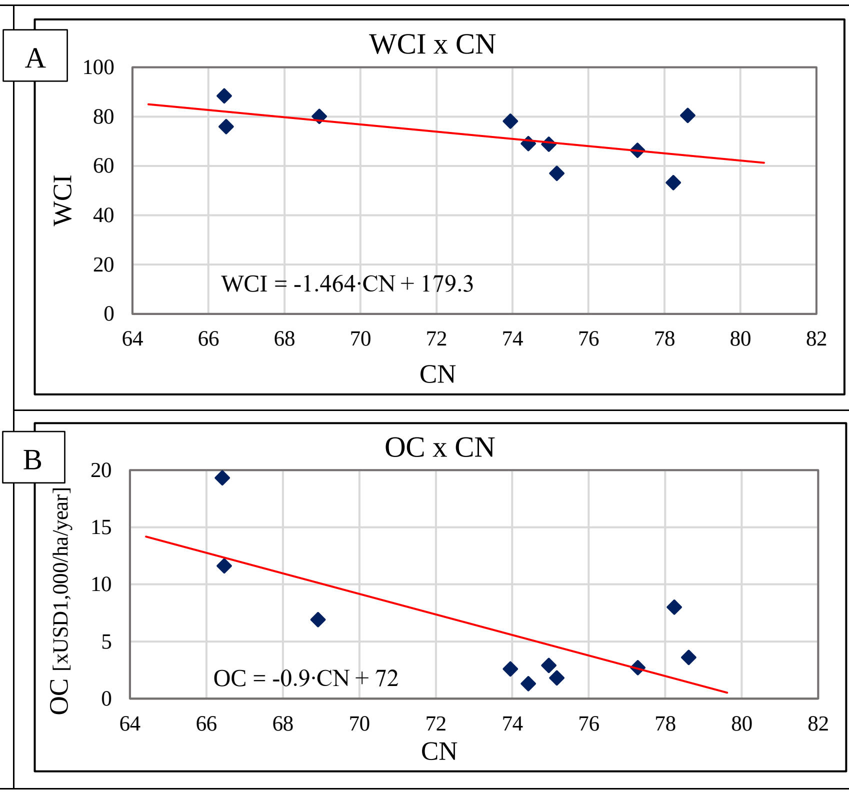

Figure 7.2. Correlation between CN, WCI and OC in the study area.

From Figure 7.2A, the linear regression analysis suggests a negative correlation between land use intensity (represented by the CN value) and water quality (represented by the WCI). As the CN value increases, indicating higher land use and anthropogenic impact on the watershed, the WCI value decreases. For instance, when the CN value increases from 66 to 80, the WCI value drops from 82 to 60. This indicates that as the watershed's resources are more extensively used, the water quality deteriorates.

Similarly, Figure 7.2B displays a linear fit that demonstrates the relationship between land use intensity (CN) and opportunity cost (OC). The linear fit suggests that as the CN value increases, representing a more intensive anthropogenic use of the land, the OC decreases. In fact, the linear fit indicates that there is no opportunity cost associated with conservation when the CN value reaches 80, which corresponds to a scenario of intense land use. This suggests that at such high levels of land use intensity, the opportunity cost of preserving the watershed becomes negligible.

These data-driven results from the study area indicate that as the land use increases (higher CN values), the quality of the watershed significantly deviates from its pristine condition, leading to a decline in water quality. Simultaneously, the opportunity cost of conservation decreases and becomes negligible at high levels of land use intensity. This emphasizes the trade-off between land use and water quality, where increasing land use for development purposes comes at the cost of degrading the natural condition of the watershed.

7.3.2 Step 3: Defining the technology actions

The evolution of water treatment has benefited the development of cities, transitioning from early simple batch operations that physically eliminate organic and inorganic particles and microorganisms, to contemporary advanced membrane filtration technologies that enhance the safety of potable water. However, this progression incurs an increase in treatment costs that are not solely linked to the employed technology, but also contingent upon energy requirements and the volume of water to be treated (McGivney and Kawamura, 2008). Additionally, the growing number of regulated contaminants and more stringent maximum contaminant levels, prompted by subpar raw water quality, require the implementation of advanced technologies to comply with rigorous drinking water standards.

Chapter 4 elucidates that water availability in the studied watersheds does not appear to pose a significant quantity threat to urban water security, at least within the next 35 years. Therefore, for the purpose of achieving success, only raw water quality and O&M costs will be considered as relevant factors. Based on the case of Caxias do Sul, it is evident that there is an opportunity cost associated with preserving the source watersheds. Additionally, water treatment costs play a crucial role in determining the tariff structure of any water utility. Imposing higher water tariffs due to increased treatment costs is not administratively favorable, and adding further land use restrictions on landowners in the watersheds is not politically appropriate.

Nevertheless, it is imperative to implement actions that guarantee the supply of safe drinking water to the city and uphold urban water security. A range of policies and combinations of actions can be explored to achieve the desired success within the next 35 years. One significant aspect of these alternative actions will revolve around the utilization of different water treatment technologies. However, it is worth noting that the selection of specific actions may be subject to discretion and depend upon the availability of data.

Table 7.2 presents a comprehensive overview of six alternative technological actions that can be implemented as measures towards achieving success. The Table also provides information on the intrinsic costs associated with each technology. It is important to note that all equations presented in the Table are based on the volume being treated at the water treatment plant, denoted as Cap. At this investigative stage, capital costs are not taken into account as they are subject to significant variation depending on the specific region. Factors such as sitework, pipeline installation, electrical work, and construction expenses are influenced by external factors rather than being directly linked to the land use and quality of the watershed.

The use of raw water is constrained to purification technology or adherence to regulated standards. In the study area, the regulations set by Brasil (2005) prohibit the utilization of low-quality raw water for public water supply. Consequently, it is crucial for a utility that operates diverse water treatment systems to ascertain the extent of raw water quality deterioration that each treatment technology can tolerate. This knowledge allows for an understanding of the minimum quality level that each of the six technology actions can guarantee for drinking water. To establish these thresholds, certain assumptions were made at the outset of the analysis.

According to Brasil (2005), surface fresh water classified as Level 4 is deemed unsuitable for public consumption, indicating that no treatment method can render such water suitable for distribution in the city. The WCI methodology, utilizing the six parameters, yields a WCI value of 45 when the raw water quality corresponds to Level 4. Conversely, Level 1 surface fresh water is characterized as high quality and requiring simplified treatment, with a corresponding minimum WCI of 91. Level 2 surface fresh water requires at least conventional treatment (Brasil, 2005), resulting in a minimum corresponding WCI of 86.

In the case of Caxias do Sul's treatment train, it operates with conventional treatment followed by powdered activated carbon (PAC) and iron and manganese removal. In a previous occurrence, one of the treatment plants had to be shut down due to low raw water quality. The evaluation of the corresponding index during that event yielded a WCI of 53, which serves as the limit for this specific technology.

For the remaining two technologies, the lower bounds were determined through interpolation of the other four technology thresholds. This resulted in a WCI of 76 for conventional treatment with iron and manganese removal, and a WCI of 64 for conventional treatment followed by PAC. These lower bounds for each technology, referred to as Water Compliance Index Lower Bound (WCIL), are also presented in Table 7.2.

Table 7.2. Overview of the technology actions.

Technology actions	**Description**	**Cost [USD.m^{-3}]**	***WCIL*, ATP**
Simplified	Treatment that relies on clarification by means of filtration and disinfection	[1]$C_1 = 7{,}696 \times Cap^{-0{,}769}$	91
Conventional	Treatment that involves coagulation followed by flocculation, sedimentation, filtration with final disinfection by chlorine	[1]$C_2 = 6{,}359 \times Cap^{-0{,}705}$	86
Conv. with Fe and Mn removal	Conventional treatment with additional iron and manganese removal through chemical oxidation	[1]$C_3 = 19{,}65 \times Cap^{-0{,}775}$	76
Conv. + PAC	PAC is applied to the conventional treatment as adsorbent for the removal of synthetic organic contaminants, pesticides, herbicides, taste and odor.	[2]$C_4 = 0{,}641 \times Cap^{-0{,}214}$	64
Conv. + PAC with Fe and Mn removal	Conventional treatment with additional iron and manganese chemical removal and PAC adsorption of synthetic organic contaminants, pesticides, herbicides, taste and odor.	[3]$C_5 = 7{,}392 \times Cap^{-0{,}526}$	53
Conv. + MB	Conventional treatment followed by low pressure membrane filtration, such, as microfiltration (MF) and ultrafiltration (UF)	[3]$C_6 = 3{,}890 \times Cap^{-0{,}360}$	45
External source	When raw water is no longer adequate to public supply, tanker trucks import drinking water from a reliable external source (meet potability standards).	C = 9,90 USD.m^{-3}	N.a.

Source: [1] (McGivney and Kawamura, 2008). [2] From Chapter 4. [3] Based on (McGivney and Kawamura, 2008) and on Chapter 4. All costs have been adjusted to May 2022 according to the Consumer Price Index (CPI) (BLS, 2022). *Cap* is expressed in L.s^{-1}. N.a. stands for “not applicable”.

Note that water from a watershed with a water quality below a WCI value of 45 cannot be utilized for public water supply. At such a degraded level, the watershed is deemed inadequate for providing water to the municipality, consequently jeopardizing urban water security. In such a scenario, the municipality must make a decision to either invest in improving the quality of the watershed or seek alternative water sources, if available. Table 7.2 includes the cost of importing water from an external source, which is presented in the last line. The cost of 9.90 USD.m^{-3} accounts for the expenses associated with importing water via tank trucks from a dependable potable water source, in the event that an existing watershed becomes compromised. Considering the substantial quantity of water required to meet the public demand, this cost was estimated based on 50% of the mean values obtained from an internet market survey conducted in Brazil in 2022.

The lower bounds (WCIL) identified in this study serve as the ATPs for ensuring urban water security in the case study. These points indicate the threshold at which the degradation of watershed quality, resulting from anthropogenic activities, reaches a level where the existing treatment train is no longer capable of delivering drinking water that meets potability standards. However, reaching an ATP does not imply an imminent water shortage or a complete absence of water security for the residents. Rather, it suggests that an alternative plan is required to manage the system effectively (Kwadijk et al., 2010).

One of the challenges faced in the implementation of DAPP is determining the rule for triggering new actions (Kwakkel et al., 2015). In the context of urban water, monitoring the raw water quality using the WCI resolves this challenge. Whenever the water quality approaches a lower bound (WCIL), indicating a potential ATP, it serves as a signal to assess alternative candidate policy actions to ensure continued water security. This approach provides a clear rule for decision-making within the DAPP framework.

7.3.3 Step 4: Evaluating the technology actions

In the fourth step of the DAPP methodology, the actions and their associated opportunity cost of conservation will be jointly assessed following a multiobjective optimization mathematical model that employs heuristics.

The rationale behind running the optimization algorithm is to minimize the total water treatment cost while also minimizing the total opportunity cost of watershed conservation. The objective is to find a solution that achieves the lowest treatment cost by striving for a pristine watershed. At the same time, the model aims to optimize the economic use of the land, seeking a scenario where the watershed is utilized to its fullest potential and the opportunity cost of conservation is optimized. By minimizing the water treatment cost, the model acknowledges the significance of accessing high-quality raw water, which leads to more cost-efficient treatment operations. Simultaneously, it recognizes the economic value associated with the land.

There are several methods available to solve multiobjective optimization problems. The selected method for this study is the ε-Constraint approach, which involves selecting one objective function to be optimized while converting the other objective into a constraint by setting an upper bound. This effectively transforms the problem into a scalar or mono-objective optimization (Miettinen, 1999). In ε-Constraint problems, the primary objective is typically minimized, and the other objective is converted into either an inequality constraint (Haimes et al., 1971) or an equality constraint (Lin, 1976). In this study, the latter approach is followed.

To implement the ε-Constraint method, an evolutionary algorithm is employed by using an electronic spreadsheet. The choice of using a spreadsheet is due to its user-friendly nature and wide accessibility for communication and analysis purposes (Baker, 2015). The evolutionary algorithm is a heuristic computational approach that is capable of gradually improving and searching for the global optimum or near-optimum solution (Reddy and Kumar, 2020; Yu and Gen, 2010). Given the popularity of optimization as a decision-making support tool, the evolutionary algorithm serves as an effective approach to assist DAPP.

In the ε-Constraint method implemented in this Chapter, the opportunity cost (OC) is considered as a constraint, while the decision variables are adjusted to optimize the other objective. The objective function (*Of*) aims to minimize the annual sum of the treatment cost (TC) for each surface water treatment system (*i*) and for each type of treatment technology train (*p*) used in the process. This objective function is represented by Equation 7.2.

$$Of = min \sum_{i} \sum_{p} TC_{i,p} \qquad \text{Equation 7.2}$$

subject to

$$TC_{i,p} = a_p * Cap_{i,p}^{b_p}$$

$$\sum_{p} Cap_{i,p} = D_i \qquad \forall i$$

$$Cap_{i,p} \leq Wa_i \quad \forall i$$

$$Cap_{i,p} \leq \frac{\left|WCI_i - WCIL_p\right| + \left(WCI_i - WCIL_p\right)}{2} * 1E6$$

$$WCI_i = a_{WCI} * CN_i + b_{WCI}$$

$$\sum_{i} OC_i = \varepsilon \quad \{\varepsilon \in \mathbb{R}^+ \mid \varepsilon \geq 0\}$$

$$OC_i = a_{OC} * CN_i + b_{OC} \quad \forall i,$$

where *Cap* is the volume of treated water [L.s^{-1}] that needs to be supplied to the city; D is the demand for water [L.s^{-1}], intrinsically related to the population served in the analyzed years (2015 and 2050 in this study); a_p and b_p are coefficients derived from the cost function of the treatment technology (*p*), as shown in Table 7.2; *Wa* represents the water availability of the watershed [L.s^{-1}]; WCI is the water compliance index, whose quality objective considered meeting (being in compliance with) is the fresh water Level 2 of the Brazilian water quality guidelines regulation (Brasil, 2005); WCIL denotes the lowest raw water quality (lowest WCI) that the treatment technology (*p*) can ensure for drinking water; CN is the curve number; a_{WCI}, b_{WCI}, a_{OC} and b_{OC} are data-driven coefficients derived from linear fits based on the results presented in Chapter 4 and Chapter 6; the OC is given in x10^{3}USD/(hectare·year). According to the data-driven results for the study case, the coefficients a_{WCI}, b_{WCI}, a_{OC} and b_{OC} are, respectively, -1.464, 179.3, 0.9 and 72 (obtained from Figure 7.2).

The 35-year plan considered in this study spans from 2015 to 2050. The starting point of 2015 was chosen as it represents the mean period (2011 to 2018) for which data was available to conduct the analysis in this Chapter. By considering these two extremes, the optimization process encompasses the beginning and end of the plan.

Although this thesis does not delve into the identification of tariff affordability or quantifying the extent to which landowners are willing to bear opportunity cost, the use of a Pareto front can aid in identifying non-dominated solutions. In the context of this thesis, non-dominated solutions refer to those that strike a balance between low treatment cost and low opportunity cost, with equal importance placed on both objectives. Solutions located on the Pareto front are considered efficient as they cannot be improved without compromising one objective for the other.

In multi-objective problems, the concept of "optimum or efficient solutions" is different compared to single-objective problems because the focus is on trade-offs rather than finding a single solution that optimizes all objectives simultaneously (Coello et al., 2002). Figure 7.3 illustrates the pointwise projection of the Pareto front for both 2015 and 2050. It is noticeable that the treatment cost is higher in 2050, which can be attributed to population growth and the subsequent increase in water demand, leading to a larger volume of treated water (*Cap*). The annual volume expected for 2015 was 2.77x10^7 m^3, whereas for 2050, it increased to 5.37x10^7 m^3.

In the face of numerous trade-offs, decision-making becomes crucial to ensure urban water security, as the best choice may not necessarily lead to the most favorable outcome. Furthermore, different optimal solutions yield different utility, and some optimal solutions may result in adverse economic consequences. From Figure 7.3, it is evident that beyond an opportunity cost of 40,000 USD/hectare·year, there is no significant reduction in treatment cost that would justify additional land use restrictions at the watershed. This does not imply that decision-making will prioritize sacrificing the watershed; rather, it suggests that further conservation measures become economically irrational. On the other hand, opportunity costs lower than 5,000 USD/hectare·year marginally increase treatment costs to levels that could potentially impact tariffs. If a convex curve were to be fitted to the pointwise projection, its two extremes would strongly favor opposite single-objective optima. The extreme optimal solutions show virtually no trade-off, indicating that decisions made at these points are likely to result in social discontent. In summary, careful consideration is required when navigating these trade-offs to ensure a balance between watershed conservation and the economic viability of the water supply system, while also taking into account public acceptability and social concerns.

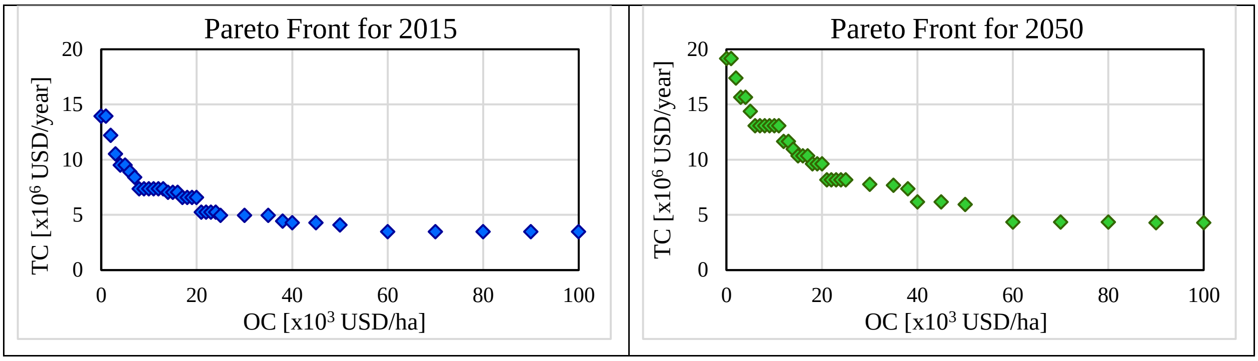

Figure 7.3. Pareto front for 2015 and 2050.

Figure 7.3 also reveals clusters of points where the treatment cost remains constant even with increasing land use restrictions at the watershed (higher opportunity cost). These clusters, referred to as economic flat zones of performance by Null et al. (2021), indicate that certain thresholds exist for one objective but not the other. In the current case, these zones highlight the range within which a specific water treatment technology can handle a certain level of watershed degradation. When the watershed quality reaches a certain threshold, the water utility company must modify the treatment train. Such thresholds in trade-off can make negotiation of solutions more difficult (Null et al., 2021). In the flat zones of performance, there is no improvement in one objective while the other is enhanced. For example, although there is no reduction in treatment cost, the opportunity cost is optimized.

Table 7.3 provides the results obtained from the optimization model, indicating the recommended technological action for six examples of conservation strategies based on the associated opportunity cost. In general, for both 2015 and 2050, lower watershed quality corresponds to lower opportunity cost, indicating that the watershed resources are being utilized optimally or near optimally, while treatment approaches the threshold of requiring advanced technology due to the system's proximity to an ATP with a WCI of 53.

On the 2015 Pareto front, at the efficient point where the opportunity cost exceeds 30, the optimization model suggests a high level of conservation, almost pristine, for the Faxinal watershed, but not necessarily for the other systems (Table 7.3). This behavior is mainly attributed to the volume of treated water supplied by the Faxinal system, which accounts for more than 50% of the city's water demand. By reducing the treatment cost at the Faxinal system, the local utility becomes globally efficient, even though the other systems operate with more costly treatment trains. A similar behavior is observed in the Marrecas system for 2050, which can also be attributed to the volume of treated water it provides.

Table 7.3. Changes in watershed quality (WCI) at the Pareto Front.

	Faxinal		Marrecas		Maestra		Dal Bó		Samuara	
OC	**2015**	**2050**	**2015**	**2050**	**2015**	**2050**	**2015**	**2050**	**2015**	**2050**
0	59.24	61.71	58.16	60.07	60.42	61.43	58.82	58.21	55.60	57.32
5	64.31	65.06	59.05	64.05	65.63	65.56	64.74	57.79	51.75	61.96
10	67.00	70.60	64.96	64.46	64.32	64.87	65.71	64.17	65.19	63.08
20	78.37	77.36	64.85	66.10	67.17	67.89	66.17	67.15	66.84	64.94
30	92.78	92.78	67.81	64.85	65.47	70.20	68.97	67.18	64.65	64.73
40	91.63	91.12	64.17	88.06	91.27	67.30	64.41	64.66	64.48	64.76

Opportunity cost (*OC*) given in x10^3 USD/ha·year.

When evaluating the systems based on the land use in their respective watersheds, it can be observed that the more land use restrictions there are (lower CN values), the higher the opportunity cost (Table 7.4), which limits landowners from making optimal use of their land in the watersheds. It is worth noting that by prioritizing the conservation of the Faxinal system over the other systems, the water system of Caxias do Sul reaches its efficient frontier. This effect can be attributed to the higher volume of treated water supplied by the Faxinal system (*Cap*).

Table 7.4. Land use at the watershed (*CN*) according to the *WCI* at the Pareto Front.

	Faxinal		Marrecas		Maestra		Dal Bó		Samuara	
OC	**2015**	**2050**	**2015**	**2050**	**2015**	**2050**	**2015**	**2050**	**2015**	**2050**
0	80.41	80.32	80.32	81.44	81.36	80.51	84.48	82.71	80.02	83.32
5	78.54	78.03	82.14	78.72	77.64	77.69	78.25	83.00	87.12	80.15
10	76.71	74.25	78.10	78.45	78.54	78.16	77.59	78.64	77.94	79.38
20	68.94	69.63	78.17	77.32	76.59	76.10	77.27	76.61	76.82	78.11
30	59.10	59.10	76.16	78.18	77.75	74.52	75.36	76.59	78.31	78.26
40	59.90	60.23	78.61	62.32	60.16	76.50	78.45	78.31	78.42	78.24

Opportunity cost (*OC*) given in x10^3 USD/ha·year.

Higher opportunity cost implies a higher level of conservation, as it indicates that the resources in the watershed are moving further away from their optimal economic use. As a result, the cost of water treatment decreases when the quality of the watershed (WCI) is higher (Table 7.5).

Table 7.5. Water treatment cost according to the *WCI* at the Pareto Front.

	Faxinal		Marrecas		Maestra		Dal Bó		Samuara	
OC	**2015**	**2050**	**2015**	**2050**	**2015**	**2050**	**2015**	**2050**	**2015**	**2050**
0	5,205,218	7,121,362	3,397,544	4,651,792	3,174,731	4,341,276	1,516,891	2,077,863	633,403	962,583
5	3,486,575	5,863,054	3,397,544	2,893,654	1,535,764	2,580,448	451,278	2,077,863	633,403	962,583
10	3,486,575	5,863,054	1,718,592	2,893,654	1,535,764	2,580,448	451,278	760,434	151,784	962,583
20	2,706,798	3,141,027	1,718,592	2,893,654	1,535,764	2,580,448	451,278	760,434	151,784	253,774
30	1,102,637	1,284,611	1,718,592	2,893,654	1,535,764	2,580,448	451,278	760,434	151,784	253,774
40	1,102,637	1,284,611	1,718,592	1,292,110	866,533	2,580,448	451,278	760,434	151,784	253,774

Opportunity cost (*OC*) given in x10^3 USD/ha·year, while water treatment cost (TC) in USD/year.

From Table 7.6, it is evident that the marginal cost of water treatment increases when the initial policies for watershed conservation are implemented. As the opportunity cost increases from zero to one unit (x10^3 USD/ha·year), the total reduction in treatment costs exceeds USD 850,000 in 2050 and even higher in 2015. However, as the policies for watershed conservation become more stringent, such as when the opportunity cost exceeds 20 (x10^3 USD/ha·year), the marginal reduction in treatment costs decreases to values lower than USD 60,000 in 2015 and USD 200,000 in 2050. However, as the opportunity cost further increases, particularly values exceeding 30 (x10^3 USD/ha·year), there is a noticeable increase in the reduction of the marginal cost. This is primarily attributed to the significant reduction in treatment costs achieved in the Marrecas and Maestra systems, because the optimization model indicates a simpler technology to be operational as watershed quality augmented.

Table 7.6. Marginal changes in total water treatment cost in the study area.

	2015		2050	
OC	*TC*	Marginal change	*TC*	Marginal change
0	13,927,787	-912,482	7,121,362	-851,708
5	9,504,564	-596,276	5,863,054	-621,934
10	7,343,993	-348,162	5,863,054	-436,667
20	6,564,216	-56,206	3,141,027	-199,652
30	4,960,055	-36,614	1,284,611	-140,663
40	4,290,824	-289,387	1,284,611	-259,700

Opportunity cost (*OC*) given in x10^3 USD/ha·year, while water treatment cost (*TC*) in USD/year.

7.3.4 Step 5: The adaptive plan map

The adaptation pathways map illustrates potential technological actions in conjunction with varying levels of watershed conservation (WCI) for the designated analysis period to achieve success. In this particular instance, the alternative pathways and thresholds (ATP) were determined through mathematical optimization utilizing the ε-Constraint method, employing an evolutionary algorithm model as outlined in Section 7.3.3.

Figure 7.4 depicts the adaptation pathways map based on the WCI. The tipping point occurs when a technology becomes inefficient in treating a specific raw water quality, and the existing infrastructure no longer meets potability standards. Pursuing an alternative pathway entails cost implications. This map serves as a guideline for local policymakers, empowering them to make well-informed decisions within an uncertain environment. In this innovative application, the key to achieving a successful urban water supply lies in effectively assessing the trade-offs between watershed conservation and treatment costs. The preferred pathway adheres to an adaptive plan that ensures water quality within potability standards while safeguarding urban water security, with cost being a crucial aspect to consider.

The plan should prioritize affordable tariffs, which are notably attainable when the raw water quality is high and the watershed is preserved. However, maintaining the watershed in close proximity to its pristine environmental state, particularly near metropolitan areas, has social and economic implications that must be weighed against the cost of water tariffs. Figure 7.4 presents a range of alternative treatment technology pathways that decision-makers can pursue when the quality of raw water fluctuates within a selected time period.

To commence the implementation of the DAPP, a set of seven scenarios is introduced for the evaluation of candidate pathways. Among these scenarios, six align with the proposed success period outlined in this Chapter, while the last scenario extends the analysis to a timeframe of 100 years from 2015. These scenarios were formulated based on the seven states for scarcity rent for water in Caxias do Sul, as proposed in Chapter 6. Each scenario represents a potential future wherein the quality of the watershed has deteriorated to a degree that necessitates the deployment of a backstop technology.

In this case, the backstop technology is the conventional treatment train followed by membrane filtration, as it represents the utmost resource of technology evaluated in this thesis. The further into the future the backstop technology is implemented, the longer the existing technology will continue to meet water standards. For instance, if the implementation of the backstop technology is planned for 15 years from now (in 2030), the decision-maker would need to transition from, for example, a conventional treatment process with iron and manganese removal in 2027 to the backstop technology. However, in the scenario spanning 100 years, this transition in the candidate pathway would occur in 2100 instead.

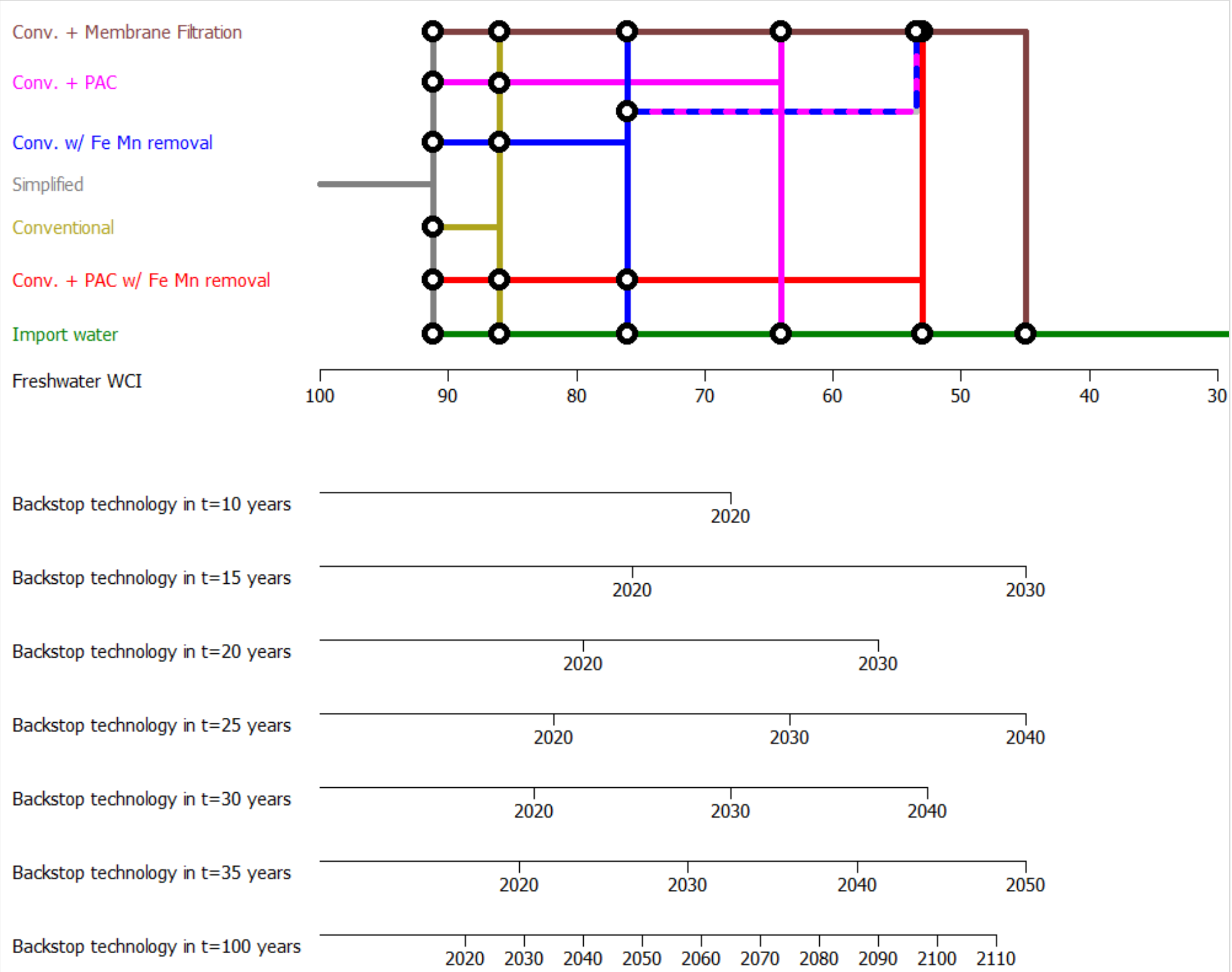

Figure 7.4. Adaptation pathways map for urban water security in Caxias do Sul, Brazil.

The adaptation pathways map (Figure 7.4) purposefully does not present the optimal solution. Rather, it unravels some alternatives of robust candidate technologies that can be operated in a wide range of watershed quality conditions. The objective of this map is not to prescribe a specific path for the municipality to follow, but rather to illustrate that there exist multiple alternative combinations, each with its own benefits and costs.

Table 7.7 exhibits the scorecard for the selected candidate pathways, including their respective costs. This scorecard is a valuable resource for decision-makers and stakeholders as it promotes awareness of the trade-offs between watershed conservation and cost-effective water treatment. The strategy employed in pointing out these trade-offs is expressed through the lens of cost, bearing in mind that the objective is minimizing both treatment cost and opportunity cost of watershed conservation. Taking Pathway 1 as an example, it is characterized by low treatment costs (TC) but high opportunity costs (OC). In this case, the negative economic side effects are high in the watershed. Conversely, selecting Pathway 6 results in minimal or no economic side effects in the watershed, but it involves the highest treatment costs. Consequently, consumers will bear the additional costs associated with treating a poorer raw water quality, which will be reflected in the tariffs they pay. By carefully analyzing the consequences of each action plan, decision-makers can gain a clear understanding of the implications of their choices, enabling them to make more informed decisions.

Table 7.7. Scorecard for some candidate pathways in Caxias do Sul.

	Pathway	TC	OC	WCIL	CN
1	Simplified	7.74	10.32	91	68.53
2	Conventional	8.14	9.17	86	69.81
3	Conv+Fe-Mn rem	9.07	6.87	76	72.37
4	Conv+PAC	10.48	4.10	64	75.44
5	Conv+PAC+Fe-Mn rem	12.11	1.57	53	78.25
6	Conv+Memb	13.33	0.00	45	80.00
7	*Import water	274.20	0.00	0	100

Costs at the Pareto front as for 2015. Opportunity cost (*OC*) given in x10^3 USD/ha·year. Water treatment cost (*TC*) in x10^6 USD/year. *Value corresponds to importing water, not treating it, as it is assumed to be already treated.

7.4 Discussions

The concept of avoided costs during treatment is intricately linked to preservationist measures and the management of land use within the watershed (Dearmont et al., 1998; Heberling et al., 2015; Warziniack et al., 2017). The relationship between these factors has been a subject of discussion and extensive debate, as highlighted in Chapter 4. Previous studies have demonstrated the correlation between these factors based on existing financial data. However, in this Chapter, the correlation is presented from an economic standpoint, providing a deeper understanding of the economic implications involved.

While city dwellers benefit from affordable drinking water, the burden of preserving the watershed falls solely on the landowners within that watershed. Chapter 6 discusses the discrepancy that those who contribute to watershed conservation rarely reap the benefits, particularly because they usually live away from the distribution system. This asymmetry in urban water management highlights the need for resolution.

It has been emphasized that the responsibility for conservation should be assessed, and the concept of OC is identified as a valuable metric for monitoring this effect. Paradoxically, low treatment costs are accompanied by high economic costs, represented by the opportunity cost of watershed conservation. This shadow price of urban water needs to be acknowledged, and appropriate actions must be taken to compensate landowners. One potential approach for future decision-making, particularly within frameworks, like the DAPP, which deal with deep uncertainties, is to consider a payment for watershed services. This concept draws inspiration from the payment for environmental services (Martínez-Jauregui et al., 2023; Peterson et al., 2015; Wunder, 2015), but with a specific focus on compensating landowners for their contributions to preserving the watershed.

The technologies handled in this thesis are used to physically and chemically remove the undesirable constituents in surface water, hence selecting treatment technology trains is profoundly connected to the environmental state of the studied watersheds. However, uncertainties, such as water committee guidance and future health, hinder technology selection and cost prediction. Although treating water is relatively inexpensive on a volumetric basis (Crittenden et al., 2012), this research demonstrated that the process interferes significantly on final annual costs. In the case of Caxias do Sul, where the geology contributes to the presence of iron and manganese in the natural water sources, the removal of these elements becomes a necessary step in the treatment process. Notwithstanding, the DAPP approach can explore alternative combinations of treatment technology trains in areas where the water supply relies on different sources, such as groundwater, lotic aquatic systems, or brackish and saltwater sources.

The ε-constraint method followed by evolutionary optimization has proven to be a versatile and effective approach for DAPP, particularly in scenarios where data availability is limited. This approach is necessary due to the lack of accessible open-source data sets pertaining to water treatment costs in Brazil. Another constraint is the dearth of socioeconomic data specific to small watersheds, which has been addressed in Chapter 5. The innovative application of a water quality index (the WCI) as an adaptation tipping point serves as a valuable tool for decision-makers, stakeholders, and the general public, enabling them to gain a deeper understanding of the rationale behind changes in decision-making processes. When a city identifies that urban water security is approaching or has exceeded a predetermined threshold (ATP), the current course of action is deemed inadequate, and an alternative pathway must be pursued to ensure the continued success of the urban water management plan.

Water resources management is a complex field (Loucks and van Beek, 2017), characterized by intricate trade-offs, particularly in urban water supply scenarios where significant financial and political factors come into play, both at the local and regional levels. The paradigm of balancing conservation and economics has gained further understanding through the research presented in this thesis. The Pareto front, depicted in Figure 7.3, offers a means to explore alternatives that may transcend this paradigm. When evaluating the extremes of the Pareto front, it becomes evident that marginal changes in either the opportunity cost or treatment cost may not appear appealing for decision-making. For instance, when the opportunity cost is already high (indicating a strong emphasis on watershed conservation) and is reduced from 40 to 39, the marginal change in treatment cost is negligible. Conversely, the other end of the Pareto front is similarly unattractive, as a reduction of one unit in opportunity cost can lead to a significant increase in treatment cost and, consequently, water tariffs. However, within the economic flat zones of performance along the Pareto front, negotiations can become exhaustive, because within the threshold the opportunity cost can increase without the benefits of reducing treatment costs.

The identification of thresholds in urban water supply, as revealed in this thesis, represents a significant and noteworthy finding. Further dedicated studies in the future should be conducted to deepen the understanding of the economic implications for municipalities situated at the edges of these flat zones.

7.5 Conclusions

In this Chapter, an ensemble of methods was used alongside the DAPP approach to support decision-making. Specifically, the ε-constraint method was utilized in conjunction with an evolutionary algorithm to support decision-making processes. Additionally, a water quality index was computed using six analytical water parameters and treated as a compliance index (WCI). Lastly, the curve number, a widely used parameter in hydrology, was employed as a surrogate for assessing land use modifications.

Effectively managing urban water systems necessitates a thorough understanding of the inherent uncertainties involved. This study acknowledged some inherent uncertainties that permeate the economic dimension of urban water security. Water treatment cost and tariff represent the most important aspect of this dimension, since in order for people to have the benefit of this economic good, they have to be able to afford the served water. The uncertainties acknowledged in this thesis include land use changes in the watershed, the source water quality, and the technologies employed in the treatment process. Furthermore, conditions can be unpredictably altered by socio-economic factors, potability standards, land use regulations, and the impacts of climate change on local hydrology.

Water treatment technology is an ongoing uncertainty that warrants continuous investigation, as advancements in scientific fields, including urban water supply, are constantly evolving. These advancements may necessitate adaptive planning, and decision-makers and stakeholders must be cognizant of how their decisions should be guided as the future unfolds.

Although the methodology employed in this Chapter provides a broad overview of the uncertainties associated with technology, further in-depth investigation into this matter would be beneficial in refining the adaptation pathways maps and enhancing our understanding of the implications of technological uncertainties.

The main objective of this Chapter is not to tell a decision maker or stakeholders what the best option is, but rather to provide them with the means to comprehend the implications of various alternatives. Instead of prescribing a specific pathway, the focus is on offering a comprehensive understanding of the trade-offs involved, primarily in terms of costs. Through this research, it becomes clear that land use changes in source water watersheds can significantly impact the initial plan, emphasizing the need to account for uncertainties in the complex task of decision-making. By acknowledging and taking into account these uncertainties, decision-makers and stakeholders can navigate the intricate terrain of decision-making with greater awareness and a comprehensive perspective. This recognition allows for a more informed understanding of the potential trade-offs associated with different courses of action.

CHAPTER 8

8 DISCUSSIONS

8.1 Overview of the presented research

This thesis has unveiled a cost that has frequently been referenced in the technical realm of urban water treatment but has not been adequately quantified until now: the opportunity cost of watershed conservation. However, the scope of this study extends beyond merely quantifying this economic cost. It also emphasizes the decision-making process related to treatment technology and the associated expenses, with the ultimate goal of safeguarding urban water security in a deeply uncertain world.

To fulfill these requirements, it was essential to select a study area or multiple areas that encompassed a comprehensive water treatment system, consisting of a watershed, pipeline, and water treatment plant. The chosen watershed had to be representative in size to the volume of water withdrawal, and its land use had to be locally regulated to preserve the quality of raw water. For example, cities reliant on rivers as their water source were excluded from the study area, as the upstream basin would be extensive, making it challenging to pinpoint significant land use changes that impact water quality. Furthermore, the applicable land use regulations and restrictions had to be within the bounds of local legislation. Areas that spanned multiple municipalities, typically regulated by the state or federal government, were deemed unsuitable for inclusion in this thesis. Lastly, and arguably the most crucial condition, the water utility responsible for operating in the selected area needed to be willing to provide local data for this research. Obtaining treatment information and data on the local water source is often challenging, particularly with private water companies, and even with public utilities, as raw water quality information is not readily accessible to the public or academia. Additionally, the costs associated with the treatment aspect of the system are not readily available, and gathering historical records diverts utility personnel from their routine tasks.

The case of Caxias do Sul demonstrates several successful aspects. Firstly, the city is served by five distinct water supply systems. Secondly, the water system in the city is managed by a single company. These first two attributes make it easier to request data from within the company. Thirdly, the geographic and geomorphological location of Caxias do Sul is noteworthy. The municipality was established in the late nineteenth century in a region of Rio Grande do Sul State that lacks major rivers but possesses a heterogeneous and anisotropic fractured aquifer, characterized by acidic volcanic flow outcrops (De Vargas et al., 2022b). Due to limited groundwater availability, the city relies on damming small creeks as a solution to provide water. Consequently, these dams create small watersheds upstream. Fourthly, the municipality has experienced significant population growth, with over half a million inhabitants, and is now the second-largest economy in the state. This growth has intensified human activities in both urban and rural areas, including within the five watersheds. Fifthly, since 1978, Caxias do Sul has implemented local regulations for land use in the five watersheds (Caxias do Sul, 2005, 1978). Lastly, since the water systems are individualized, it was possible to obtain treatment cost data for each system separately. SAMAE data for the period from 2011 to 2019, this last year coincides with the start of this doctoral study.

Despite the broad and ambitious nature of this research topic, before delving into the research question of this thesis, it was necessary to gather socioeconomic data from the watersheds. A particular challenge was faced in obtaining socioeconomic data from areas that were not officially recognized as census tract units. Chapter 5 provides a comprehensive explanation of the approach used to overcome this obstacle. The solution emerged through establishing a correlation between the radiance observed in satellite images capturing nighttime light and data obtained from established census tract units such as cities.

8.2 Revisiting the research questions

This chapter examines the principal findings of this thesis by addressing the central research questions outlined in Section 1.2: " Which decisions regarding water treatment technologies, aimed at maintaining low tariffs, have the greatest impact on the opportunity cost of watershed conservation?”. While the central question will be discussed in due course, it is important to revisit the four subquestions first.

> *What are the long-term economic effects on water treatment costs when current decisions are being blurred by modest short-term gains in avoided treatment costs?*

This thesis has demonstrated that decision makers in the urban water supply sector often prioritize short-term economic data, which can blur the negative long-term consequences. While existing literature has argued that marginal improvements in water quality within a watershed result in modest savings in immediate treatment costs, this thesis goes beyond by illustrating that these seemingly negligible short-term gains can have significant long-term implications. The objective was to showcase that even in a scenario where total water withdrawals do not exceed the watershed's capacity and the immediate cost savings do not motivate private or public investments in source water protection, the long-term economic effects are detrimental to tariff rates and, consequently, to the consumers who bear the cost.

In this study, I have utilized data obtained from a municipal water company to demonstrate a negative correlation between watershed quality, represented in the form of a water quality compliance index (WCI), and the cost of chemicals utilized in the water treatment processes. It has been observed that lower water quality corresponds to higher treatment costs. In the case of Caxias do Sul, where decision makers are influenced by short-term cost savings, the anticipated increase in treatment costs could reach as high as 242% in the long-term if a backstop technology such as membrane filtration were to be added to the existing conventional treatment train.

Nonetheless, these costs could escalate further if water quality were to deteriorate to an extent where treatment becomes impractical, necessitating long-distance transwatershed diversion or the construction of entirely new water systems. Elaborating on these costs falls outside the scope of this study. Therefore, I recommend that future researchers address this knowledge gap by exploring the detailed costs associated with such scenarios.

The data from Caxias do Sul raises concerns about the city's water supply system. Over the nine-year period of available data, the costs associated with chemicals used in water treatment have increased by a staggering 233%. This astonishing increase cannot be solely attributed to land use practices within the watershed. External factors such as climate change, population growth, economic developments, regulations, and technological advancements also contribute to these cost escalations. However, regardless of the underlying factors, the nominal value of this component in water costs can easily cause a blurring effect on decision-making when it comes to prioritizing watershed conservation efforts.

In nominal terms, the 233% increase in chemical costs only resulted in a change in overall treatment costs from 0.012 USD·m^{-3} in 2011 to 0.028 USD·m^{-3} in 2019 (adjusted to December 2019). This thesis demonstrates that there has been a marginal change of just 0.0002 USD.m^{-3} in treatment costs for every one-unit change in the water quality compliance index (WCI). This finding further confirms the existence of the blurring effect, where the remarkably low treatment costs, even after doubling their nominal value from 2011 to 2019, can blur the importance of preserving watershed quality.

To address the blurring effect and shed light on the potential consequences of watershed conservation, Chapter 4 of this thesis proposed a scenario of significant watershed degradation. In this scenario, the addition of membrane filtration to the existing conventional treatment train was considered. However, it was challenging to find comprehensive references in the literature that specifically explored membrane filtration applied to surface water.

Despite this limited availability of references, the results of the analysis indicate that in the long-term, treatment costs could increase by as much as 242% for the case of Caxias do Sul if membrane filtration were to be implemented. This finding challenges the myopic conclusion of solely focusing on preserving the watershed. Instead, this research ventured in the opposite direction by exploring the concept of the opportunity cost associated with conservation efforts.

What is the opportunity cost of watershed conservation to keep affordable water tariff to city dwellers?

The utilization of Data Envelopment Analysis (DEA) proved to be effective in assessing the relative economic and environmental efficiency of the watersheds in Caxias do Sul. DEA allowed for the evaluation of efficiency without the need to search for data from regions outside the study area and without the requirement of preassigned relative weights for each location. This approach facilitated the analysis of the intricate relationships between multiple inputs and outputs within the watersheds. Among the fifteen Decision-Making Units (DMUs) studied, three exhibited no opportunity cost in terms of watershed conservation within the limits required by legislation. This indicates that these three watersheds are both technically and environmentally efficient. However, for the remaining watersheds, the annual opportunity cost of conservation ranged from 1,000 to 19,300 USD per hectare.

This thesis has provided an indirect estimation of the upper bound economic cost associated with the conservation of each watershed examined. In the aggregate, the annual opportunity cost amounted to USD 2,520,000 for the study area. One particularly noteworthy finding is that the degree of attention given by legislation to the permeability of watersheds has a limited impact or a lesser impact on the economy of urban watersheds compared to rural ones. This observation is supported by the fact that urban watersheds generally exhibited lower or no opportunity costs. However, when analyzed on a per-unit area basis (hectare), the opportunity cost of conservation was found to be higher in urban watersheds.

How can the opportunity cost of watershed conservation be compared to the economic benefits of such preservation?

Addressing this research subquestion posed its challenges, primarily due to the diverse economic implications spread across different areas of the municipality. The cost of watershed conservation was found to predominantly fall on landowners, who have their option to select the best economic alternative for their land curtailed. On the other hand, the urban residents (consumers) reap the benefits of conservation, which manifest as changes in consumer surplus. In economics, the focus is typically on the change in consumer surplus resulting from policy shifts, rather than its absolute level (Varian, 2014). Consumer surplus measures the benefits consumers receive when participating in the market and serves as a reasonable gauge of welfare in the context of urban water. Therefore, when tariffs are lower, consumers are more willing to participate in the market, leading to an increase in consumer surplus. Conversely, when policies neglect watershed conservation, the number of regulated contaminants increases, and stricter maximum contaminant levels are imposed. This necessitates the adoption of additional purification technologies in water treatment, resulting in higher treatment costs and subsequently elevated tariffs. As a consequence, consumers, as a whole, reduce their consumption quantity, leading to a loss in consumer surplus.

In this thesis, scarcity rent was factored into the known tariff, and the Marshallian demand function was derived. In Brazil, a tariff structure incorporating the price of water specifically is not commonly observed. Instead, tariffs typically account for the costs associated merely with the conveyance, treatment, storage, and distribution of water.

The research conducted using the case study of Caxias do Sul has demonstrated that in a scenario where a backstop technology, such as additional membrane filtration, is required within five years to compensate for inefficiencies in watershed conservation, the price of the tariff could increase by 13%. As a result, water consumption is estimated to decrease by 3.4%. These changes in price and quantity lead to a movement along the demand curve, altering the market equilibrium point. The consequence of this shift is a reduction in the total benefit derived from water consumption, as the area under the marginal benefit curve decreases. This reduction signifies a loss in consumer surplus, reflecting the decreased welfare experienced by consumers due to the increase in price and decrease in quantity consumed.

Returning to the research subquestion, the answer becomes evident. Albeit cost and benefits are being attributed to different parties, the opportunity cost can be compared to the benefits of conservation by quantifying the loss or gain in consumer surplus. In the case of Caxias do Sul, the scenario in which policies neglect conservation efforts results in an aggregate benefit loss that is 185% higher than the opportunity cost associated with conservation. For the scenario where watershed quality deteriorates more rapidly, necessitating the addition of a backstop technology within the first five years, the annual loss in consumer surplus amounts to USD 4,600,000. In the alternative scenario where the backstop technology is required after 35 years, this loss is reduced to USD 2,583,000. In either case, the loss in benefits exceeds the actual annual opportunity cost.

Maintaining the current land use control policy may prevent a loss in consumer surplus, thereby benefiting consumers. However, it is important to recognize that landowners are still incurring the opportunity cost of conserving the watershed, without directly benefiting from this policy. These findings can serve as a foundation for considering economic compensation to address this restraint. One possible approach is the implementation of a payment for watershed services, similar to the concept of payment for environmental services. Overall, the idea of implementing a payment for watershed services program can help bridge the gap between the costs borne by landowners and the benefits realized by consumers.

What decisions can be made today and during the course of time to ensure urban water security in the future in a world of deep uncertainties?

Traditionally, water planners have relied on historical data, computer models, extrapolations of social, economic, and environmental interactions, and personal beliefs to make estimates and predictions about the future. However, this approach falsely assumes that we can accurately predict the future and overlooks the dynamic nature of water resources systems, which are characterized by deep uncertainties (Haasnoot, 2013). In this thesis, an innovative approach called Dynamic Adaptive Policy Pathways (DAPP) is applied to help decision makers adapt to plausible futures and develop alternative actions to achieve success. Rather than telling what decisions are best, the underlying concept of this thesis is enabling planners to understand the potential consequences of different alternatives.

In Chapter 7, a multi-objective problem was formulated, aiming to optimize both water treatment costs and opportunity costs simultaneously. However, the mathematical method employed to solve this problem resulted in a Pareto front rather than a single solution. The Pareto front represents a set of non-dominated solutions, where improving one objective would require sacrificing the other. The presence of a Pareto front leaves the responsibility of decision-making with the planner. Each point on the Pareto front represents a trade-off between water treatment costs and opportunity costs, offering different sets of consequences.

The complex landscape of trade-offs necessitates a thoughtful approach to decision-making in order to ensure long-term urban water security. It is crucial to recognize that the optimal choice may not always align with the option that produces the most favorable immediate consequences.

Therefore, the central research question emerges from this context of competing objectives and uncertain outcomes.

Decision-making under this myriad of trade-offs requires identifying the pathways to ensure urban water security in the long-term. It is crucial to recognize that the optimal choice may not always align with the option that produces the most favorable immediate consequences. Therefore, the central research question emerges from this context of competing objectives and uncertain outcomes.

8.3 The central research question

The content of this thesis unequivocally establishes that human interventions in the watershed of the water source can lead to modifications in the initial water treatment technology. Furthermore, the profound uncertainties inherent in urban water management, such as land use regulations, population growth, technological advancements, economic development, climate change, the number of regulated contaminants, and maximum contaminant levels, also contribute to the alteration of the initial plans. Throughout the course of this thesis, the concern regarding the escalation of water tariffs has been consistently highlighted, as urban water security is contingent on consumers' ability to afford the cost of potable water supplied to their taps. However, this thesis goes beyond the realm of conventional urban water management by striving for not only low treatment costs but also low opportunity costs associated with watershed conservation. The fundamental challenge lies in identifying actions that can simultaneously satisfy both objectives.

> *Which decisions regarding water treatment technologies, aimed at maintaining low tariffs, have the greatest impact on the opportunity cost of watershed conservation?*

The response to the central research question offers a resolution to this problem, and the DAPP approach has proven to be a valuable tool in addressing this challenge. By adjusting the level of watershed conservation, represented by a water quality index (WCI), and understanding the adaptation tipping point (ATP) for each treatment technology, this thesis has generated an adaptation pathways map. Each pathway within the map carries socioeconomic and technical implications that are likely to arise as a consequence of the corresponding decision.

In Section 7.3.4, it has been clearly shown that there is always a technological solution available to supply water to the city, even if it involves the often stigmatized and expensive option of importing water. However, such a solution is found to be 20 times costlier than the most advanced technological train evaluated in this research. This highlights the precarious state of the economic dimension of urban water security. Decisions must therefore prioritize affordable tariffs while still enabling social and economic activities to take place within the watershed. Similarly, land use policies cannot be excessively permissive as they can lead to a decline in watershed quality, ultimately compromising the existing water treatment technology and increasing costs.

The purpose of the adaptation pathways map is not to identify the single best solution, but rather to provide guidance on alternative decisions that can be made as the future unfolds. Each pathway outlined in the map comes with its own predicted benefits and costs, allowing decision makers to consider the trade-offs associated with each option.

The decision to maintain the watershed in a near-pristine condition can indeed reduce treatment costs, but it may also impede socioeconomic development within the watershed, leading to higher opportunity costs for landowners. Conversely, promoting local growth within the watershed poses a threat to urban water security, as treatment costs can increase by up to 242%, resulting in higher tariffs. In this context, the optimal decisions to be made are primarily linked to carefully weighing trade-offs rather than searching for an ideal solution, as there exists a range of non-dominated solutions that can fulfill the objectives. Successful decision-making will ultimately be achieved by considering and balancing these trade-offs effectively.

CHAPTER 9

9 CONCLUSIONS

In contrast to top-down approaches, this thesis adopts a bottom-up approach, which requires the use of local and detailed data as well as specific knowledge to generate its findings. However, acquiring such data at the small watershed scale poses significant challenges as much of it is either unavailable or non-existent. To overcome this barrier, substantial effort was devoted to working closely with the Public Water and Wastewater Service of Caxias do Sul (SAMAE). SAMAE was instrumental in halting their routine activities to facilitate the selection and extraction of historical data, enabling the study to proceed.

Additionally, the utilization of nighttime light satellite images in Chapter 5 offered a means to estimate the Gross Domestic Product and employment within each Decision-Making Unit (watershed). These estimates proved invaluable in studying the opportunity cost of watershed conservation in Chapter 6, thereby affirming the robustness of the approach for future hydroeconomic modeling as demonstrated in Chapter 7.

One of the main driving forces behind undertaking this thesis was the awareness of the modest gains in avoided treatment costs associated with marginal changes in water quality at the watershed. Chapter 4 of this research sheds light on fundamental references that highlight concerning results in this regard. However, this study delves even deeper into this issue and demonstrates that decisions influenced by short-term data can easily transform these seemingly modest short-term gains into a significant long-term threat to water tariffs. To describe this phenomenon, I propose the term "short-term blurring effect" to refer to data and decisions that mislead short-term choices and contribute to unsatisfactory outcomes in the long-term.

This blurring effect, as reported in the existing literature, is no different within the study area. While the water systems in Caxias do Sul may only save 0.0002 $USD \cdot m^{-3}$ in treatment costs for each one-unit change in the WCI, which serves as a surrogate for watershed quality, these costs can escalate by up to 242% in the long-term. Even though, considering only one scenario of backstop technology to the existing treatment of Caxias do Sul, these results allow the conclusion that the short-term blurring effect is a potential risk to urban water security, particularly in its economic dimension. This is because the additional treatment costs will inevitably be passed on to the water tariffs.

As discussed extensively in Chapter 6, which forms the core of this thesis, watershed conservation plays a crucial role in ensuring the quality and quantity of raw water. However, it is the landowners who predominantly bear the burden of this conservation. These landowners, often residing in rural areas that are geographically distant from the water distribution network, rarely reap the benefits of lower water tariffs and access to high-quality drinking water. Moreover, the imposition of land use restrictions adds an opportunity cost for these individuals, which is referred to as the opportunity cost of watershed conservation within this thesis. Because the concept of opportunity cost is abstract and less tangible compared to other financial costs, poorer landowners are the ones to come the closest to feeling the effects of this economic cost.

The results of the case study revealed an annual opportunity cost of USD 2,520,000 for all watersheds, which encompasses a total area of 14,860 hectares. When examining the opportunity cost on a per-unit area basis, it was found that the urban watersheds had higher costs compared to the rural watersheds. However, a different result was obtained when considering the total annual opportunity cost: the rural watersheds exhibited the lowest overall costs. This suggests that urban watersheds, despite having smaller areas, are generating a higher ratio of desirable/good product (GDP) and a lower ratio of undesirable/bad product (impervious area) per unit area when compared to other watersheds. However, due to the larger combined area of the rural watersheds in comparison to the urban ones, the rural watersheds ultimately incur a higher total opportunity cost.

In addition to determining the opportunity cost of conservation, this thesis recognized the importance of assessing the benefits derived from watershed conservation. It was observed that environmental conservation not only has ecological value but also economic benefits. With a comprehensive understanding of the importance of watershed conservation, this research dedicated itself to quantifying the collective benefit derived from such efforts. To accomplish this, the consumer surplus approach was employed as a means of assessment.

The consumers surplus approach, despite its simplicity, benefits from easily available information and provides a direct approach to quantify the overall welfare of consumers in different scenarios, considering their ability to participate in the urban water market. The analysis revealed that the benefits of conservation outweighed the costs, ranging from 3% to 85%. This reaffirms the notion that conservation is a worthwhile investment. However, for the case study, the ones who benefit are not the same ones who pay the price. This discrepancy highlights an important knowledge gap.

Paradoxically, while low treatment costs are associated with conservation, they also result in high economic costs for certain stakeholders. This shadow price of urban water must be acknowledged, and actions need to be taken to address and compensate landowners who shoulder the burden. Resolving this incongruity calls for further investigation and analysis. Therefore, I recommend that future studies focus on exploring ways to reconcile this disparity.

Throughout the course of this thesis, there was an initial expectation that simple decision alternatives leading to success in achieving long-term urban water security would emerge. However, the paradigm of balancing conservation and economics took an unexpected turn. Instead of providing a single straightforward solution, the analysis revealed a diverse set of possible alternatives represented by the Pareto front, as presented in Chapter 7. These alternatives encompass various trade-offs that need to be carefully considered. To navigate this array of alternatives and trade-offs, the DAPP approach indicates to be a suitable framework for application in urban water management, particularly in situations involving trade-offs and uncertainties.

This study acknowledges the presence of inherent uncertainties in relation to urban water security. Beyond the external uncertainties that are widely recognized, this research recognizes three additional sources of uncertainty: land use changes within the watershed, the quality of the source water watershed, and the selection of treatment technology. Hence, it emphasizes the importance for decision makers and stakeholders to be prepared to adapt their planning strategies in response to evolving future conditions.

In essence, the key message of this work is to prioritize the identification of multiple alternatives for achieving water security, rather than presenting a single "correct" pathway to decision-makers. By understanding the trade-offs and crucial tipping points of these alternatives, we can create a pragmatic pathways map of solutions to adapt and prepare our systems against future challenges. This approach empowers decision-makers to make well-informed choices and ensures a resilient water management strategy to safeguard our water resources.

CHAPTER 10

10 REFERENCES

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CHAPTER 11

11 APPENDIX OF CHAPTER 6

11.1 Slacks-based efficiency index (SBEI)

Consider a production possibility set P using ***x*** inputs to produce ***g*** desirable ("*good*") products to be defined as

$$P = \{(\mathbf{x}, \mathbf{g}) |\ \mathbf{x} \geq X\boldsymbol{\lambda},\ \mathbf{g} \geq G\boldsymbol{\lambda},\ \boldsymbol{\lambda} \geq 0\},$$ Equation 11.1

where $\boldsymbol{\lambda}$ is a nonnegative vector in $\mathbb{R}^n$ that represents the slope of the line connecting each point to the origin, $X = (x_{ij}) \in \mathbb{R}^{m \times n}$ is the input matrix and $G = (g_{ij}) \in \mathbb{R}^{s \times n}$ the output matrix, assuming X and $G > 0$.

Tone (2001) defines slacks-based measure (SBM) of efficiency as the input excess ($\mathbf{s}^-$) and the output shortfall ($\mathbf{s}^+$) in relation to the efficiency frontier. Let a DMU ($\mathbf{x}_o$, $\mathbf{g}_o$) be represented by the production of $\mathbf{g}_o$ desirable products using $\mathbf{x}_o$ amount of inputs, which can be described as

$$\mathbf{x}_o = X\boldsymbol{\lambda} + \mathbf{s}^-,$$ Equation 11.2

$$\mathbf{g}_o = G\boldsymbol{\lambda} - \mathbf{s}^+,$$ Equation 11.3

being the vectors $\boldsymbol{\lambda} \in \mathbb{R}^n$, $\mathbf{s}^-$ and $\mathbf{s}^+ \geq 0$. The vectors $\mathbf{s}^- \in \mathbb{R}^m$ and $\mathbf{s}^+ \in \mathbb{R}^s$ are called slacks. Figure 11.1 depicts four DMUs (A, B, C and D), each with one input ($\mathbf{x}_o$) and one output ($\mathbf{g}_o$). The efficiency frontier, on the assumption of VRS, is represented by the continuous line ABC. As the efficiency frontier envelops all the DMUs, it is clear that D is not efficient when compared to A, B and C, which becomes the benchmarks. DMU-D, nonetheless, can become efficient if all the inefficiencies are removed. There are three possible ways to do so: either by reducing the amount of input ($\mathbf{x}_o$) and, thus, eliminating the input excess ($\mathbf{s}^-$), by optimizing the process to produce more ($\mathbf{g}_o$) with the same quantity of resources ($\mathbf{x}_o$), eliminating the output shortfall ($\mathbf{s}^+$), or by a combination of both.

The economic efficiency of a DMU can be evaluated by a slacks-based efficiency measure ρ, for ρ (0,1], as defined below (Tone, 2001).

$$\rho\ = \frac{1-\frac{1}{N}\sum_{n=1}^{N}\frac{s_n^-}{x_{no}}}{1+\frac{1}{M}\sum_{m=1}^{M} s_m^+/g_{mo}},$$ Equation 11.4

Note that by Equation 11.4 the more slack there is in output ($\mathbf{s}^+$) the less efficient the production ρ, the same is true for the input excess ($\mathbf{s}^-$), remaining all other variables constant.

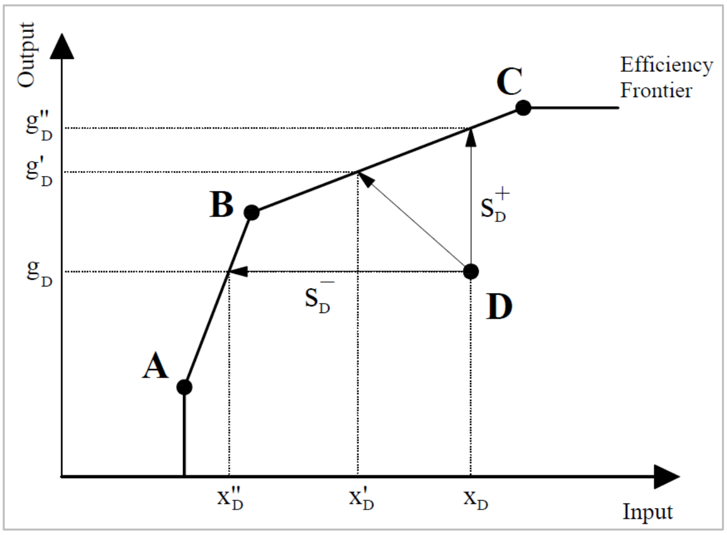

Figure 11.1. Schematic representation of slacks.
Representation of slacks from DMU-D and the remainder efficient DMUs with an efficiency frontier exhibiting variable returns to scale (VRS). Adapted from Cooper et al. (2007).

Now consider another production possibility set (F) in which the amount of $\boldsymbol{x}$ input will jointly produce $\boldsymbol{g}$ desirable ("*good*") product and $\boldsymbol{b}$ undesirable ("*bad*") product. The production technology (Färe et al., 1989; Zhou et al., 2006) can be described as

$$F = \{(\mathbf{x}, \mathbf{g}, \mathbf{b}): \mathbf{x} \text{ can produce } (\mathbf{g}, \mathbf{b})\}, \qquad \text{Equation 11.5}$$

This production possibility set "F" is the case when the traditional assumption of maximizing all outputs will no longer satisfice the production technology, as it is recognized that maximizing undesirable outputs is irrational. Färe et al. (1989) explain that it is unreasonable to assume strong disposability of outputs, since that implies that the undesirable, as well as the desirable ones, can be freely disposed. Therefore, it is assumed that outputs are only weakly disposable, even though the bad products are not treated symmetrically with the good ones. According to Kuosmanen and Kazemi Matin (2011), the assumption of weak disposability enables undesirable outputs to be treated as outputs, considering the tradeoffs between good and bad outputs. And the following two assumptions are imposed on F (Färe et al., 1989; Zhou et al., 2006):

1. Outputs are weakly disposable, if $(\mathbf{x}, \mathbf{g}, \mathbf{b}) \in F$ and $0 \leq \beta \leq 1$, then $(\mathbf{x}, \beta\mathbf{g}, \beta\mathbf{b}) \in F$.
2. Desirable and undesirable outputs are null-joint, if $(\mathbf{x}, \mathbf{g}, \mathbf{b}) \in F$ and $\mathbf{g} = 0$, then $\mathbf{b} = 0$.

To calculate the economic efficiency (θ_1), specifically when the efficiency frontier exhibits constant returns to scale (CRS) and undesirable products are not considered, the input-oriented linear programming problem can be employed. The input-oriented linear programming problem, as described by Tone (2001), allows for the transformation of the production system and provides a mathematical proof. For more detailed information on the transformation process and mathematical proof, please refer to Tone's work on page 500.

$$\theta_1 = \min\left\{t - \frac{1}{N}\sum_{n=1}^{N}\frac{s_n^-}{x_{no}}\right\}, \quad \text{Equation 11.6}$$

$$\begin{aligned}
\text{Subject to} \quad & \sum_{k=1}^{K}\lambda_k x_{nk} + s_n^- = tx_{no} && n = 1, 2, \ldots, N \\
& \sum_{k=1}^{K}\lambda_k g_{mk} - s_m^+ = tg_{mo} && m = 1, 2, \ldots, M \\
& t + \frac{1}{M}\sum_{m=1}^{M} s_m^+/g_{mo} = 1 \\
& \lambda_k \geq 0; \quad k = 1, 2, \ldots, K; \text{ and } s_n^-, s_m^+ \geq 0,
\end{aligned}$$

where λ_k is the slope of the DMU-k (input, output), x_{no} and g_{mo} are, respectively, that input (o) and output (o) being analyzed among all the other Ns and Ms. And t (t > 0) is a scalar variable from the Charnes-Cooper (Charnes and Cooper, 1962) linear transformation.

However, when the undesirable products (**b**) are considered, the production process can be modeled by the environmental DEA technology. Then the economic efficiency of DMU_o can be identified as its environmental efficiency (θ_2) (Zhou et al., 2006) as

$$\theta_2 = \min\left\{t - \frac{1}{N}\sum_{n=1}^{N}\frac{s_n^-}{x_{no}}\right\}, \quad \text{Equation 11.7}$$

$$\begin{aligned}
\text{Subject to} \quad & \sum_{k=1}^{K}\lambda_k x_{nk} + s_n^- = tx_{no} && n = 1, 2, \ldots, N \\
& \sum_{k=1}^{K}\lambda_k g_{mk} - s_m^+ = tg_{mo} && m = 1, 2, \ldots, M \\
& \sum_{k=1}^{K}\lambda_k b_{jk} = tb_{jo} && j = 1, 2, \ldots, J \\
& t + \frac{1}{M}\sum_{m=1}^{M} s_m^+/g_{mo} = 1 \\
& \lambda_k \geq 0; \quad k = 1, 2, \ldots, K; \text{ and } s_n^-, s_m^+ \geq 0,
\end{aligned}$$

where b_{jo} is that undesirable output (o) being analyzed among all the other Js.

Finally, the SBEI for modelling environmental performance (Zhou et al., 2006) is defined as the quotient of θ_1 and θ_2, as shown in Equation 11.8,

$$\text{SBEI} = \frac{\theta_1}{\theta_2}. \quad \text{Equation 11.8}$$

Zhou et al. (2006) explain that since θ_2 and θ_1 are, respectively, the economic efficiency scores when undesirable outputs are and are not considered, the proposed SBEI can be used to model the impacts of environmental regulations on economic efficiency. Campos (2015) highlights that, on the one hand, θ_1 measures the technical efficiency, ignoring environmental restriction regulation, but on the other, θ_2 measures the technical and environmental efficiency. The terminology "technical" suggests an efficiency between the amount of product obtained in the process and the quantity of input used, while the "environmental" term indicates that environmental regulation was added to the analysis as an undesirable output (**b**).

The CCR model (employing constant returns to scale) differs from the BCC model (employing variable returns to scale) only in that the BCC model incorporates the convexity condition $\sum_{k=1}^{K} \lambda_k = 1$, $\lambda_k \geq 0$, ∀k in its constraints, while the CCR model does not. Likewise, to estimate efficiency under VRS, following the BCC model, one should add $\sum_{k=1}^{K} \lambda_k = 1$ as a constraint to Equations 11.6 and 11.7.

Figure 11.2 depicts two production frontiers (PF) that exhibit constant returns to scale and illustrates the positions of two Decision-Making Units (DMU-1 and DMU-2) in terms of their efficiency or inefficiency regarding inputs and outputs. PF1 represents the efficiency frontier when undesirable products are not considered or are assumed to be strongly disposable. On the other hand, PF2 represents the efficiency frontier when undesirable products are considered or are assumed to be weakly disposable.

It is important to note that neither DMU-1 nor DMU-2 is efficient in either case, as they do not lie on the efficiency frontier. However, both DMUs are closer to PF2, indicating that they are less inefficient when the undesirable products are taken into account in the production function. Nevertheless, there is a loss in the production potential that could be achieved if the undesirable products were assumed to be strongly disposable, and this potential loss is represented by the distance D in Figure 11.2A.

In order for DMU-1 and DMU-2 to reach efficiency, they can either reduce their inputs while maintaining the same level of outputs or increase their production without increasing inputs, or pursue a combination of both strategies. Figure 11.2A illustrates one of the many possible combinations (d' and d") that demonstrate how far DMU-1 and DMU-2 are from efficiency. The distance D represents the cost that the presence of undesirable products imposes on the production of desirable ones. The value of D depends on the excess of inputs ($\mathbf{s}^-$) and/or the shortfall of outputs ($\mathbf{s}^+$), and its orientation can vary in practice and is not limited to the direction shown in Figure 11.2. For example, D could be horizontal if the focus is solely on reducing ($\mathbf{s}^-$) while keeping output constant, or vertical if the intention is to eliminate ($\mathbf{s}^+$) while maintaining the same level of inputs in the production process.

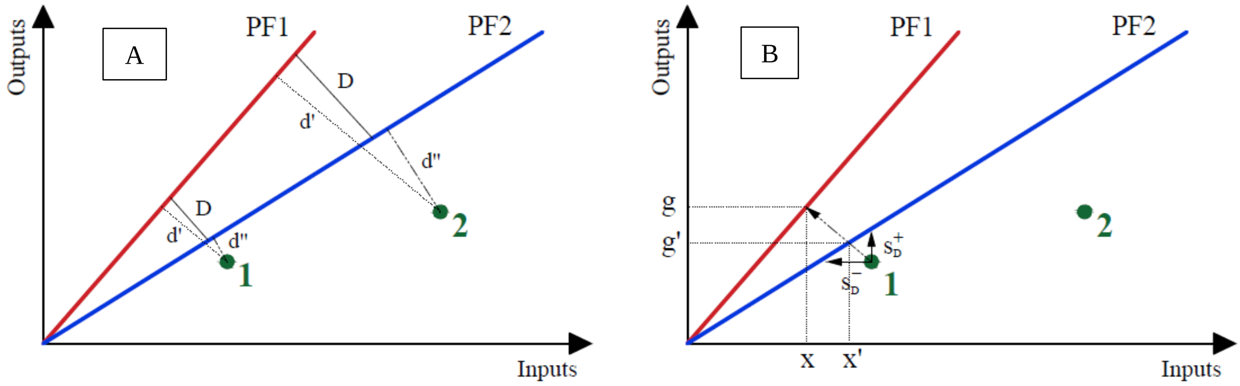

Figure 11.2. The effect of the undesirable product over the efficiency frontier (PF).
Schematic representation exhibiting constant returns to scale (CRS).

Let's focus on DMU-1 in Figure 11.2B. Suppose that in order for DMU-1 to reach efficiency on PF1, it would require a certain amount **x** of inputs to produce **g** outputs. However, when guided by the efficiency frontier PF2, DMU-1 can only produce up to **g**' outputs using inputs equivalent to **x**'. Although DMU-1 is less inefficient in relation to PF2 compared to PF1, the adoption of the assumptions of weak disposability and the null-joint in the production possibility set results in a reduction of DMU-1's production potential.