From the Institute of Landscape Ecology and Resources Management

A dissertation to obtain the degree of Doctor of Natural Sciences (Dr. rer. nat.) in the
Faculty of Agricultural Science, Nutritional Science and Environmental Management of the
Justus Liebig University Gießen

WATER QUALITY IMPROVEMENT USING NATURE-BASED SOLUTIONS AND ITS INTERLINKAGES WITH SUSTAINABLE DEVELOPMENT GOALS

KALINA MARCELA FONSECA LARGO

Referees from the Justus Liebig University Gießen:

1st Supervisor: Prof. Dr. Lutz Breuer
2nd Supervisor: Prof. Dr. Ramona Teuber
Examiner: Prof. Dr. Till Kleinebecker
Examiner: Prof. Dr. Suzanne Jacobs

Submitted: January 2025

Abstract

Nature-based solutions (NbS) in aquatic ecosystems catch the attention of policymakers as strategies to directly contribute to Sustainable Development Goal (SDG) 6 and indirectly support other SDGs. However, evidence of their effectiveness is limited. NbS in aquatic ecosystems involve more than addressing water quality issues; they require a clear understanding of their macro-environment. Thus, political, economic, social, technological, environmental, and legal (PESTEL) dimensions significantly influence the success or failure of NbS. To bridge this gap, I developed a multi-method framework that guides the alignment of NbS in crucial aquatic ecosystems within the PESTEL dimensions of developing-country communities. Focusing on Ecuador, rivers, estuaries, lakes, ponds, reservoirs, and riparian zones, which play vital roles as water sources for drinking, irrigation, and fishing in the selected communities, were used to propose NbS. A three-stage framework that integrates a mixed methods approach and involves multi-sectoral stakeholders was employed to shape NbS according to the PESTEL context of the local communities. Artificial floating islands (AFIs), a cost-effective macrophyte-based technology, and passive ecological restoration, a strategy to promote the natural regeneration of headwater vegetation, were evaluated as NbS at different stages. The stages included i) adapting AFIs in local communities by fuzzy cognitive maps, ii) assessing the environmental impacts of passive ecological restoration on páramo communities by remote sensing techniques, water quality assessment, and correlation analysis, and iii) merging AFIs with passive ecological restoration in páramo communities via bibliometric analysis. As a result, in the first stage, multiple PESTEL factors influencing water quality deterioration were identified, along with the potential of AFIs to respond to these challenges. The most influential PESTEL factors recognised were natural water pollutants, human exposure to environmental pollutants, and the violation of environmental legislation in the páramo, mangrove, and rainforest communities, respectively. Moreover, model simulations combining AFIs with strengthening training and education programs, demanding corporate environmental responsibility, and paying for ecosystem services showed an improvement in sustainable water management. In the second and third stages, páramo communities were the focus, including smallholder indigenous and peasants who depend on the páramo for water ecosystem services. Hence, I demonstrated that several PESTEL factors guided by bottom-up (local level) management successfully facilitate passive ecological restoration in páramo communities, maintaining excellent *w*ater quality. In contrast, top-down (national level) management overlooks local environmental problems, obstructing passive ecological restoration in páramo to improve water quality. Finally, AFIs and passive ecological restoration were integrated into páramo communities to address several PESTEL challenges related to insufficient funding, tech innovation, community commitment, and land use regulations and to contribute to achieving multiple SDGs. This novel and inclusive framework synthesis of diverse data sources and methodologies provided valuable insights tailored to local realities and adaptable to similar contexts in developing countries.

Abbreviations

AFIs	Artificial Floating Islands
CWQI	Canadian Water Quality Index
D_{AFIs}	Policies with Artificial Floating Islands
DEM	Digital Elevation Model
FCMs	Fuzzy Cognitive Maps
GARI	Green Atmospherically Resistant Vegetation Index
LULC	Land Use and Land Cover
NbS	Nature-Based Solutions
NDBI	Normalised Difference Built-up Index
NDVI	Normalised Difference Vegetation Index
NDWI	Normalised Difference Water Index
NGO	Non-Governmental Organisation
PESTEL	Political, Economic, Social, Technological, Environmental, and Legal
PCC	Pearson Correlation Coefficient
PVR	Photosynthetic Vigour Ratio
SDGs	Sustainable Development Goals
SIPI	Structure Intensive Pigment Index
UCutuchiRB	Upper Cutuchi River Basin
UPitaRB	Upper Pita River Basin
VARIGreen	Visible Atmospherically Resistant Index Green

Table of contents

Chapter I: Extended Summary

Chapter II: Using fuzzy cognitive maps to promote nature-based solutions for water quality improvement in developing-country communities

Chapter III: Assessing the potential of nature-based solutions as sustainable land and water management strategies in the high tropical Andean páramo ecosystem

Chapter IV: Integrating community knowledge into nature-based solutions for the sustainability of water ecosystem services: Insight from local communities in Ecuador

Chapter I: Extended Summary

1.1 Introduction

Healthy aquatic ecosystems, including rivers, estuaries, lakes, ponds, and reservoirs, and their diverse habitats, such as riparian zones, transfer social, ecological, and economic benefits from nature to humanity (Blahna et al., 2017; Culhane et al., 2019; Grizzetti et al., 2016). Conversely, poor water quality is a crucial factor leading to water scarcity, jeopardising livelihoods and the socio-economic development of communities worldwide (Alcamo, 2019; Carroll et al., 2019).

In developing countries, where ecological degradation of aquatic ecosystems is widespread, and local communities heavily depend on water ecosystem services, nature-based solutions (NbS) play a pivotal role in supporting sustainable livelihoods (Kenter et al., 2011; Telwala, 2023; UNCTAD, 2022). Cost-effective alternatives (Colares et al., 2020), promoting efficient and sustainable use of water resources (Souliotis & Voulvoulis, 2022), stakeholder engagement (Pagano et al., 2019), incorporation of community knowledge (Baustian et al., 2020), and improving the delivery of a range of ecosystem services (Liquete et al., 2016) by improving the water quality of aquatic ecosystems (Echavarria et al., 2021; Sowińska-Świerkosz & García, 2022) are some of the benefits highlighted by NbS. These efforts directly support the achievement of Sustainable Development Goal SDG 6 (clean water and sanitation), targets 6.3 (water quality improvement), and 6.6 (protect and restore water-related ecosystems) (UN, 2015).

Artificial floating islands (AFIs), a phytotechnology innovation for remediating degraded water bodies, stand out among NbS due to their low operational costs and maintenance, lack of land requirements, and ease of implementation (Afzal et al., 2019). The low operating cost of the AFIs is attributed to the inexpensive materials, offering a low-cost solution with high removal efficiency (Cui et al., 2022; Souliotis & Voulvoulis, 2022; Yeh et al., 2015). The main component of the AFIs is macrophytes with their associated microbial communities of biofilms and zooplankton, which together play a dual role by not only directly assimilating pollutants into their tissues but also acting as catalysts for purification reactions (Benvenuti et al., 2018; Cui et al., 2022). AFIs in estuaries, lakes, ponds, and reservoirs contribute directly to vital water ecosystem services such as irrigation water supply, fisheries, and aquaculture and indirectly to water purification and maintaining populations and habitats when improving water quality (De Stefani et al., 2011; Huang et al., 2017; Lu et al., 2015; Prashant & Billore, 2020).

Moreover, passive ecological restoration, also known as passive river restoration, is especially suitable for headwater streams where water quality conditions can be restored by eliminating disturbance factors and allowing for natural vegetation recovery (Muller et al., 2016; Taniwaki et al., 2019; Trujillo-Miranda et al., 2018). Headwater streams are characterised by controlling both the structure and operation of superior-order rivers (Forget et al., 2013), often located outside the economic centre of a country, isolated and commonly neglected (Haigh & Křeček, 1991). Therefore, by fostering natural vegetation regeneration, passive ecological restoration helps maintain drinking water sources and indirectly supports soil formation, composition and carbon sequestration (Brauman et al., 2019; Jähnig et al., 2010; Muller et al., 2016).

To fully realise their potential in aquatic ecosystems of local communities, AFIs and passive ecological restoration should be tailored to address the environmental issues within their political, economic, social, technological, environmental, and legal (PESTEL) dimensions (Fonseca et al., 2022). Due to the complex adaptive dynamics of NbS, PESTEL analysis has become a crucial tool for decision-makers to identify factors influencing their performance in improving environmental conditions and to guide adaptive responses (Den Heijer & Coppens, 2023). In addition, involving local communities and other key stakeholders, from problem identification to decision-making, contributes to more scientifically legitimate and publicly accountable decisions (Corburn, 2007; Nare et al., 2011).

Nonetheless, most studies about NbS do not include such analysis, limiting policymakers' understanding of their potential to support SDG 6 and contribute to several other SDGs. To address this gap, I developed a multi-method framework to guide the alignment of NbS within the PESTEL dimensions. The framework was tested on several Ecuadorian communities and two types of NbS, i.e. artificial floating islands and passive ecological restoration.

1.2 Research objectives and structure

The research was organised into three main objectives, aligning with the stages of the framework's development. Each objective was guided by research questions and discussed in a separate chapter of the thesis:

Objective 1: Identify the principal PESTEL factors that influence water quality at the community level and explore the implementation of artificial floating islands and policy interventions to improve water quality.

Based on a fuzzy cognitive maps approach, I answered the following questions: (1) What are the PESTEL factors related to water quality deterioration at the community level considering multi-sectoral stakeholder perceptions? (2) How can local water quality be improved if communities implement artificial floating islands? (3) Which policies of the PESTEL concepts can be combined with artificial floating islands to strengthen their effectiveness in improving water quality?

Objective 2: Capturing PESTEL factors that favour (or constrain) the implementation of passive ecological restoration and quantify their potential environmental impacts in páramo areas.

Combining scoping review with remote sensing techniques, water quality assessment, and correlation analysis, I addressed the following research questions: (1) What PESTEL factors favour (or constrain) the implementation of passive ecological restoration in the páramo ecosystems? (2) How do spatiotemporal changes in land use/land cover (LULC) and water quality differ between páramo areas with and without passive ecological restoration? (3) How does combining PESTEL analysis with environmental impact assessment of passive ecological restoration contribute to decision-making for sustainable land and water management in páramo areas?

Objective 3: Combining a multi-stakeholder workshop with a bibliometric analysis, I investigated: (1) How do PESTEL dimensions of local communities influence the water ecosystem services' sustainability? (2) How does integrating artificial floating islands with passive river restoration within the PESTEL dimensions of local communities contribute to the sustainability of water ecosystem services?

This framework, focused on Ecuador, provides strategies to sustain aquatic ecosystem health through a critical assessment of NbS in a developing country while remaining adaptable to countries with similar contexts. Furthermore, as a policy tool, it highlights the benefits of artificial floating islands and passive ecological restoration for achieving multiple SDGs and advocates their integration into conservation and restoration projects. The three objectives of this dissertation were tackled in three respective publications.

1.3 Materials and Methods

1.3.1 Study area

The study focused on local communities in three water-rich ecosystems in Ecuador, i.e., páramo, mangrove and tropical rainforest (Fig. 1-1).

Páramo is a unique tropical montane tundra ecosystem in the high tropical Andes at elevations of 3,000 to 5,000 meters above sea level (m.a.s.l.) (Buytaert et al., 2006; Christmann & Oliveras, 2020; Patiño et al., 2021). It is characterised by high water retention capacity and provides drinking and irrigation water to almost all Andean communities (Buytaert et al., 2006; Mosquera et al., 2023).

Mangroves consist of trees and shrubs adapted to grow in intertidal environments along tropical coasts (Feller et al., 2010). They offer ecological and economic benefits to coastal fisheries communities by providing nursery habitats and feeding grounds for numerous fish species, crustaceans, and molluscs (Calle et al., 2018).

Tropical rainforests are some of the world's most complex and abundant ecosystems, where rivers provide drinking water and support aquaculture for local communities (Delgado-Aguilar et al., 2017).

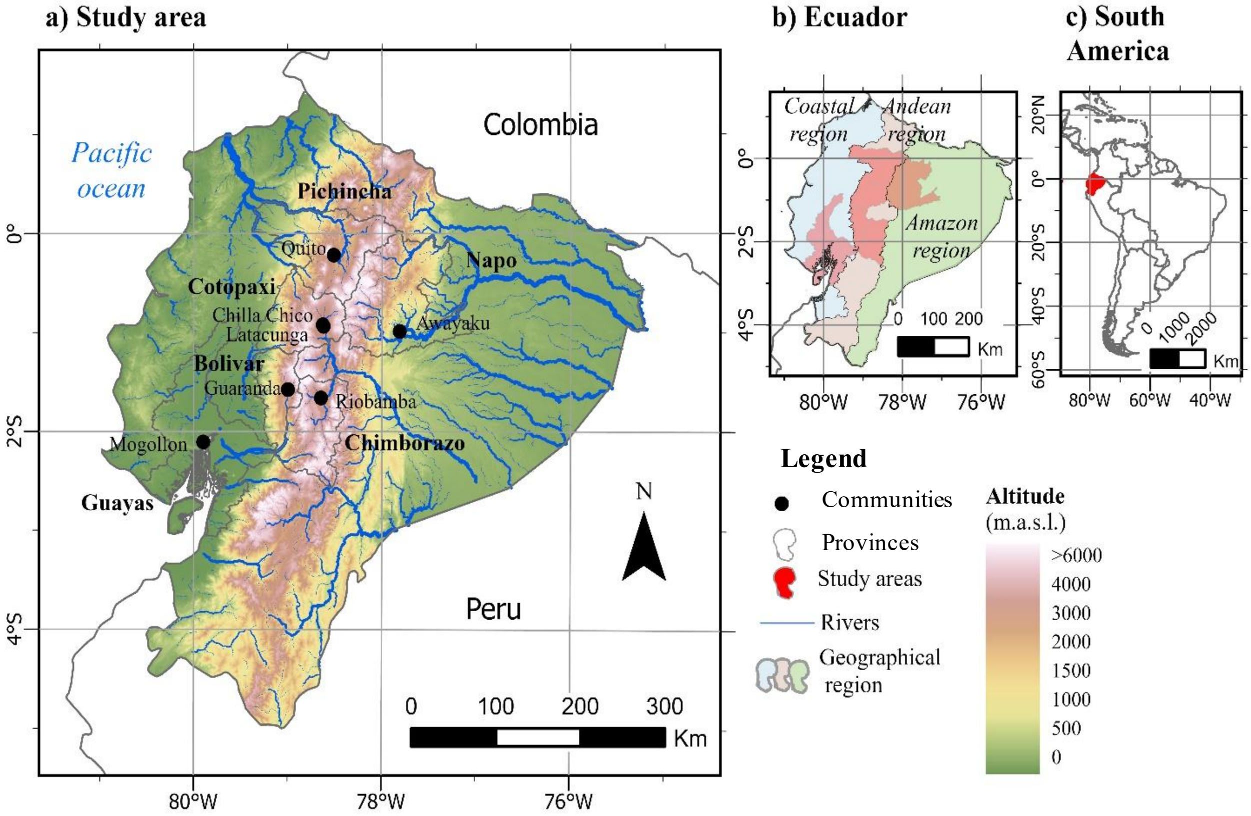


Fig. 1-1. Study area, (a) local communities (black dots) and (b) regions home to páramo, mangrove, and rainforest communities in (c) Ecuador located in South America.

Parámo is home to smallholder indigenous and peasant communities heavily reliant on water ecosystem services from headwater streams, lakes, ponds, and reservoirs. The páramo communities for this study are distributed in the provinces of Pichincha, Cotopaxi, Bolivar and Chimborazo, covering a landscape of 26,424 km^2 in the central Andean region of Ecuador (Fig. 1-1). These provinces are the source of three major rivers, the Guayas, Esmeraldas, and Pastaza, and are home to 68% of Ecuador's Andean population (INEC, 2022).

Mangrove and rainforest communities include indigenous and non-indigenous populations who depend on the ecosystem services of estuaries and rivers within mangrove and rainforest ecosystems, respectively. The mangrove-covered Mogollon estuary bank is one of the main tributaries of the Salado estuary in the province of Guayas (Hechavarría Hernández et al., 2018). The upper tributaries of the Napo River in the rainforest province of Napo are characterised by elevations ranging from 520 to 1,240 m a.s.l (GADR San Pablo, 2015; Pueblo Kichwa de Rukullakta, 2018).

1.3.2 Water quality deterioration in aquatic ecosystems in the study area

Problems related to poor water quality are common in developing countries (UNICEF, 2020), which often face economic water scarcity due to several PESTEL challenges, making it extremely difficult to achieve SDG 6 (Gude, 2017; Vinueza et al., 2021). The local communities I investigated rely heavily on water ecosystem services for their livelihoods, but agricultural and forestry activities, grazing, mining, and urban development threaten the maintenance of their aquatic ecosystems (Fonseca et al., 2024; Vinueza et al., 2021).

Aquatic ecosystems used for agriculture, livestock, fish farming, and drinking water often contain high levels of total coliforms and agrochemicals, as well as metal concentrations that exceed acceptable water quality standards (Capparelli et al., 2020; Vinueza et al., 2021; Zapata et al., 2021). It is widely recognised that contaminated water exposes the population to waterborne diseases related to faecal contamination and heavy metals, leading to significant health risks, including high morbidity and mortality (Mitra et al., 2022; Some et al., 2021).

The local communities, situated in rural areas, are especially vulnerable, with only 48.5% of their population having access to drinking water that meets the national regulations (INEC, 2022). Apart from the impacts on providing and regulating water ecosystem services, impacts on cultural ecosystem services affect indigenous populations due to their cosmovision regarding the meaning of water (Kleemann et al., 2022).

1.3.3 Multi-sector stakeholder participation

Representatives of páramo, mangrove and rainforest communities, alongside decentralised autonomous governments at provincial, municipal, and parish levels in each province, were the primary stakeholders involved in building community knowledge through participatory workshops. Additionally, participants from Non-Governmental Organizations (NGOs) were also included, such as representatives from CARE (Cooperative for Assistance and Relief Everywhere), Plan Internacional, Misión Asistencia, 180 GRADOS, and the Environmental Fund for Water Protection of Quito (FONAG). Professors and researchers from the Technical University of Cotopaxi, University of Guayaquil, Regional Amazonian University Ikiam, Justus Liebig University Giessen, and consultants were also invited to participate in the workshops.

The selection of participation sectors was guided by Ecuadorian water law (Asamblea Nacional., 2014), emphasising collective responsibility for sustainable water resource management shared among the national government, various governmental levels, and local communities, as outlined in articles 12 and 19.

1.4 Multi-method framework for NbS adaptation

The framework was organised in three main stages that integrate NbS with the PESTEL dimensions of local communities, as illustrated in Fig. 1-2.

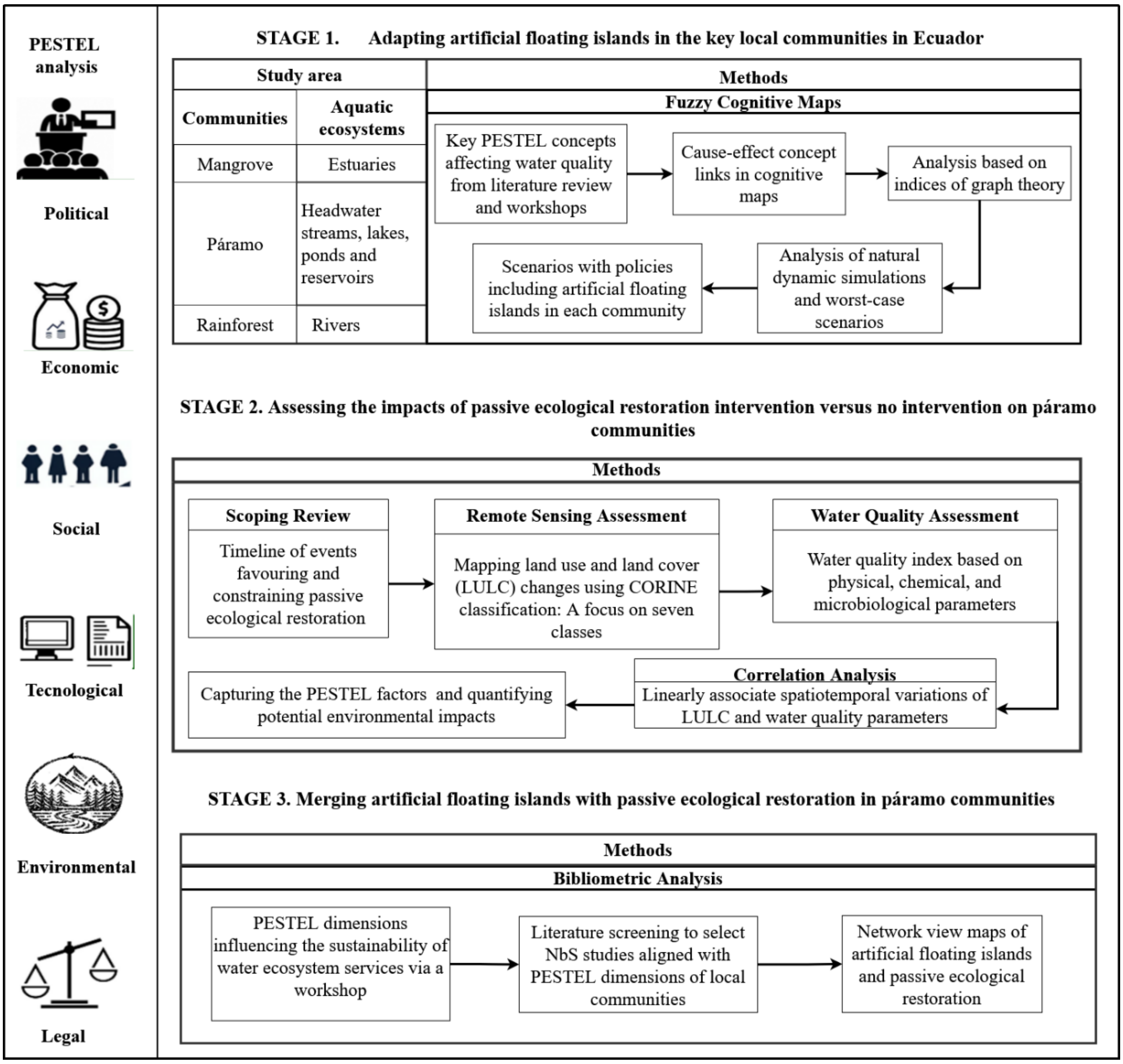


Fig. 1-2. The multi-method framework integrating artificial floating islands (AFIs) and passive ecological restoration within local communities' PESTEL dimensions.

Stage 1. Adapting artificial floating islands in the key local communities in Ecuador[1]

At this first stage of the framework development, I aimed at a) identifying the principal concepts that affect water quality from multi-stakeholder perspectives and b) exploring theoretically the use of artificial floating islands combined with different policies

---

[1] Published as: Fonseca, K., Espitia, E., Breuer, L., & Correa, A. (2022). Using fuzzy cognitive maps to promote nature-based solutions for water quality improvement in developing-country communities. *Journal of Cleaner Production*, *377*, 134246. https://doi.org/10.1016/j.jclepro.2022.134246

for improving water quality given local conditions. Three local communities located in páramo, mangrove, and tropical rainforest ecosystems, respectively, participated in extracting qualitative crucial information based on fuzzy cognitive maps and supporting decision-making in ecosystem management (Adriaenssens et al., 2004; Vergini & Groumpos, 2021). The fuzzy cognitive maps are a semi-quantitative method that depicts expert knowledge, causal reasoning, and stakeholder perceptions by signed directed graphs (Gray et al., 2015; Moosavi et al., 2021; Santoro et al., 2019). By applying the principles of fuzzy logic and cognitive maps, different sources of knowledge facilitate the decision-making processes (Shahvi et al., 2021).

This stage includes three main steps in forming the fuzzy cognitive maps approach: listing concepts, drawing causal relationships among these concepts by cognitive maps, and estimating the fuzzy inference (Fig. 1-2). In the initial step, I identified vital PESTEL concepts from literature and meetings with multi-sectoral stakeholders, including local communities. These concepts were then pre-selected and validated through a closed survey before being included in the social cognitive maps. After that, the local communities participated in the map's construction, which was structured by nodes and bidirectional connections between nodes. Nodes represent physical quantities or abstract ideas, while connections represent the influencing degree (C_{ij}) of the cause of concept i on the effect of concept j (Kosko, 1986). The cognitive maps provided by the local communities were converted into adjacency matrices according to Özesmi & Özesm (2004) for analysis based on indices of graph theory (Mourhir, 2021). In the final step, a fuzzy inference was conducted, including a natural dynamic simulation, a worst-case scenario, and a scenario influenced by policy drivers. The natural dynamic simulation determined a scenario without external influences known as the steady-state (Adriaenssens et al., 2004), then a worst-case scenario showed the maximum influence of the concepts affecting water quality (Ghaboulian Zare et al., 2022; Gray et al., 2015), and finally, potential outcomes in the three communities were assessed by analysing scenarios involving policies that incorporate artificial floating islands.

Stage 2. Assessing the environmental impacts of passive ecological restoration interventions versus no intervention on páramo communities[2]

In the second stage, the Upper Pita and Cutuchi River basins in Ecuador served as natural laboratories to illustrate the role of the páramos in the northern Andes of South America for drinking and irrigation water supply. This stage is framed in four main steps (Fig. 1-2). First, a scoping review was conducted from August 2022 to April 2023 following the methodology by Levac et al. (2010) to identify, select, and create a timeline of relevant studies on passive ecological restoration in the páramo areas based on the analysis of PESTEL factors. Second, using Google Earth Engine, as suggested by Floreano and De Moraes (2021) and Gorelick et al. (2017), it was performed image collection, estimated spectral indices, and conducted supervised classification using a random forest classifier to create LULC maps and changes for 1999, 2010, 2017, and 2022 corresponding to the crucial years of passive ecological restoration initiatives in both upper river basins. Third, it was calculated the Canadian Water Quality Index to compare water quality for drinking and irrigation sources between the study basins, chosen for its flexibility in the type and number of water quality parameters, the period of application, and the waterbodies (CCME, 2017; Hurley et al., 2012). Finally, Pearson's correlation coefficient was applied to linearly associate spatiotemporal variations of LULC with water quality parameters (Li et al., 2008) and to evaluate the possible impacts of LULC changes on water quality in the Upper Pita River Basin and Upper Cutuchi River Basin.

---

[2] Published as: Fonseca, K., Acero Triana, J. S., Ramírez, M., Martínez, W., Ilbay, M., Espitia-Sarmiento, E., & Breuer, L. (2024). Assessing the potential of nature-based solutions as sustainable land and water management strategies in the high tropical Andean páramo ecosystem. *Journal of Environmental Management*, *372*, 123350. https://doi.org/10.1016/j.jenvman.2024.123350

Stage 3. Merging artificial floating islands with passive ecological restoration in páramo communities[3]

Albert et al.'s (2021) NbS adaptation methodology was applied and divided into two steps (Fig. 1-2). In the first step, multi-sectoral stakeholders were engaged in a workshop to devise NbS that sustain water ecosystem services for local communities and to assess challenges through multidimensional assessment by the PESTEL analysis (Balzan et al., 2022; Den Heijer & Coppens, 2023). Subsequently, the information gathered from the workshop was consolidated and combined with studies to extend its applicability to countries experiencing similar contexts.

In the second step, I used the Web of Science platform for bibliometric analysis of scientific data on artificial floating islands and passive ecological restoration, chosen for its high-quality journal access and bibliometric tool compatibility. After identifying relevant studies in line with the selected keywords, it was conducted an abstract screening to select studies aligned with the PESTEL dimensions of local communities. The literature screening concluded once no new information was obtained. Data was extracted from 62 peer-reviewed articles, academic books, and book chapters published between 1991 and 2023. Subsequently, I utilised the VOSviewer software to create network view maps of the collected scientific knowledge based on Colares et al. (2020) and Donthu et al. (2021). The network view maps of artificial floating islands and passive ecological restoration were created using terms extracted from text data, including titles and abstracts. These maps

---

[3] Published as: Fonseca, K., Espitia-Sarmiento, E. F., Ilbay-Yupa, M., & Breuer, L. (2024). Integrating community knowledge into nature-based solutions for the sustainability of water ecosystem services: Insight from local communities in Ecuador. *Frontiers in Sustainability*, *5*, 1491776. https://doi.org/10.3389/frsus.2024.1491776

allowed analysis from cluster and term perspectives (Colares et al., 2020) to explore the existing or future relationships among topics in a research field (Emich et al., 2020).

1.5 Main results

Stage 1: Adapting artificial floating islands in the key local communities in Ecuador

From literature-based research, 40 PESTEL concepts associated with water quality deterioration were identified for the páramo, mangrove, and rainforest communities. Afterwards, local experts in each community recognised vital concepts relevant to local conditions, added further insights, and communities then developed fully democratic social cognitive maps. Subsequently, the cognitive map from the páramo community was constructed with 17 concepts mainly driven by environmental (23%) and economic (23%) aspects. The major problem identified was natural water pollutants (EV_1), with the highest centrality value (c_i = 12.22). Moreover, the mangrove community used 19 concepts influenced by political aspects (48%), and the primary issue was human exposure to environmental pollutants (EV_7) and a c_i = 16.27. The rainforest community used 15 concepts related to the economic aspect (40%), and the major problem was the violation of environmental legislation (L_2) with a c_i = 15.96, depicted in Fig. 1-3.

Additionally, a worst-case scenario was simulated to illustrate the maximum impact of the main concepts affecting water quality on the other PESTEL concepts at the community level, assigning them a high influence value (=1). It implied focusing on concepts with the highest centrality in each community, thus EV_1 in the páramo, EV_7 in the mangrove, and L_2 in the rainforest. The remaining concepts were affected by 89%, 90% and 87.5%, respectively (Fig. 1-3). Besides, in the communities, different levels of alteration were identified in population growth (S_2), anthropogenic water pollutants (EV_2), and social well-being (S_7) (Fig. 1-3).

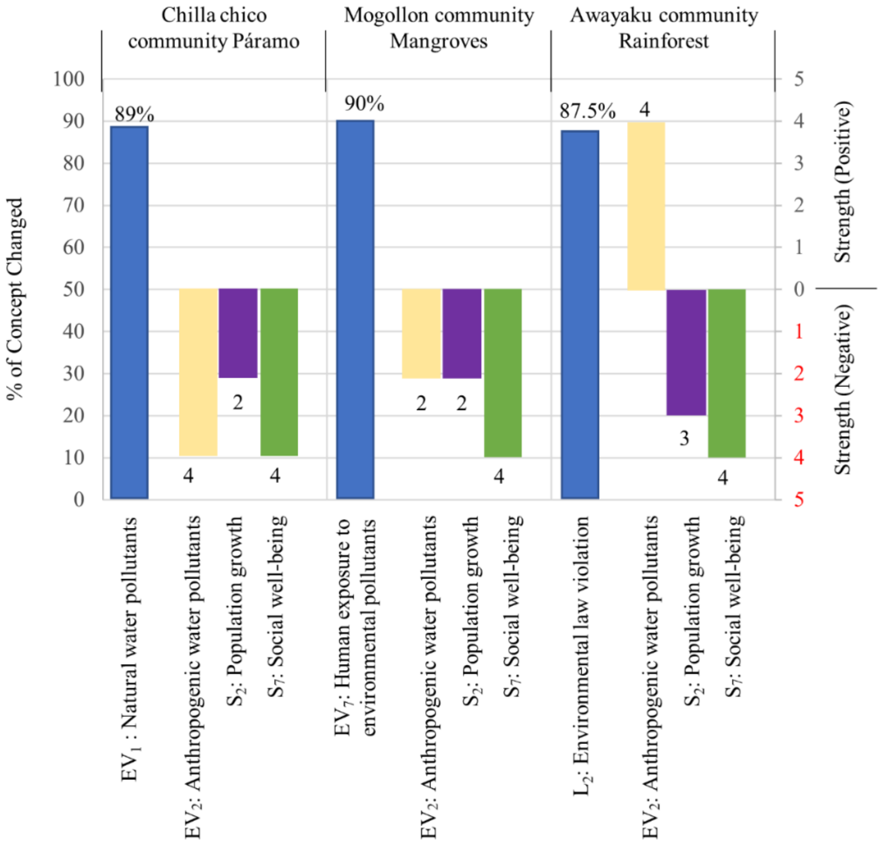
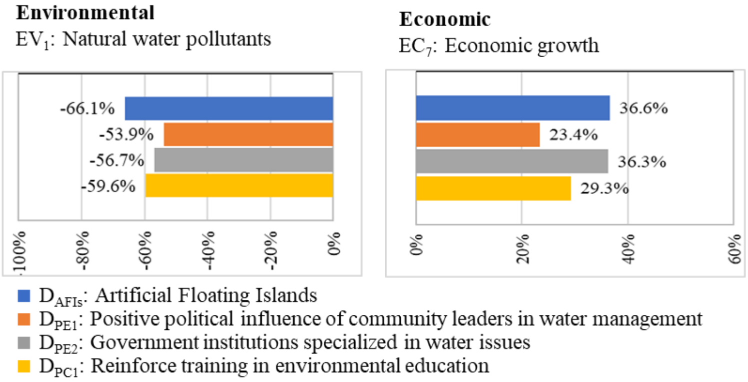


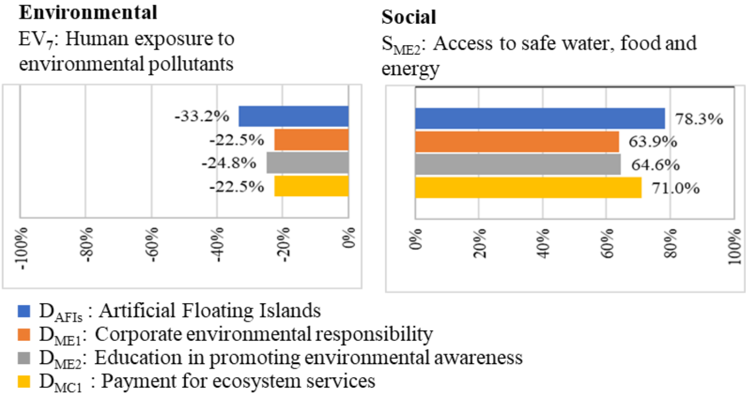
Fig. 1-3. Results of the worst-case scenario for the páramo, mangrove, and rainforest communities. Results are only shown for concepts with moderate to strong changes that have been proposed for all three communities. Numbers at bars indicate the absolute number of concepts identified.

Finally, I explored potential outcomes in the three communities by analysing scenarios that combined policies with artificial floating islands, referred to as D_{AFIs} in Fig. 1-4. For instance, the local experts suggested government institutions specialised in water issues for the páramo community (D_{PE2}), corporate environmental responsibility for the mangrove community (D_{ME1}), and better educational levels of community members in the rainforest community (D_{RE1}). In addition, the local communities suggested reinforcing training in environmental education (D_{PC1}) in the páramo, payment for ecosystem services (D_{MC1}) in the mangrove, and a community committee to denounce non-compliance with the law to the government (D_{RC1}) in the rainforest, all detailed in Fig. 1-4.

(a) Chilla chico community Páramo



(b) Mogollón community Mangroves



(c) Awayaku community Rainforest

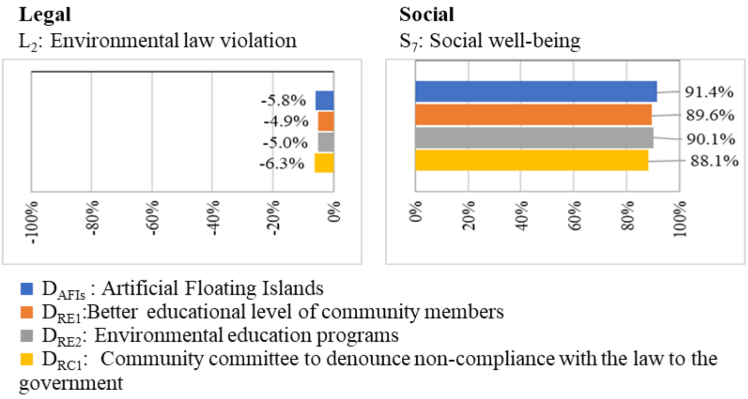


Fig. 1-4. Comparison between steady-state and change rate concepts by clamping each policy to 1 in the (a) páramo community, (b) mangrove communities and (c) rainforest community.

The results in Fig. 1-4a showed that the policies identified in the páramo community as most crucial to reducing the natural water pollutants (EV_1) are D_{AFIs}, D_{PC1}, D_{PE2}, and D_{PE1}, in decreasing order. In the mangrove community (Fig. 1-4b), the implementation of D_{AFIs} could decrease human exposure to environmental pollutants (EV_7) and increase access to safe water, food and energy (S_{ME2}). In the rainforest community (Fig. 1-4c), the DAFIs have a positive impact on social well-being (S_7), and environmental law violation (L_2) can be faced with any of the four policies.

The results revealed how combining locally adapted policies with artificial floating islands can directly contribute to achieving SDG target 6.3 (improving water quality) through fuzzy cognitive maps. Additionally, as pioneering work in the region, These results have the potential to be replicated in different communities in developing countries, where access to data is limited, but the motivation and knowledge of stakeholders exist.

Stage 2: Assessing the environmental impacts of passive ecological restoration interventions versus no intervention on páramo communities

We provided vital insights into the challenges and opportunities posed by the PESTEL framework in integrating passive ecological restoration at the headwaters of the important Andean river basins within páramo communities to address widespread land use changes and their impacts on water quality. According to the PESTEL analysis from the literature review, páramo transformations in the 20th century were mainly due to the expansion of large farms (haciendas) for camelid grazing and the development of water and electricity infrastructure in the Upper Pita River basin (UPitaRB). Meanwhile, in the Upper Cutuchi River basin (UCutuchiRB), the focus was on afforestation with *Pinus* species for timber production and agricultural and mining development.

Several PESTEL factors guided by bottom-up management promoted policies, regulations, social agreements, and financial support in 1999 to achieve passive ecological restoration in UPitaRB between 2010 and 2017. Utilising remote sensing techniques and water quality assessment in UPitaRB, we found *excellent* drinking water quality and 73.4% of natural páramo conserved by 2022. Mainly, LULC changes from 2010 to 2017 showed an increase to 81.8% of natural páramo and a decrease to 3.0 % of open spaces with little or no vegetation, 1.6% of temporary crops, and 0.7% of forests (Fig 1-5b, c, and e). In addition, for our study period (1999-2022), the correlation analyses showed that temporary crops could be the main land use class to predict high levels of faecal coliforms, total phosphorus, turbidity, and low dissolved oxygen levels in UPitaRB.

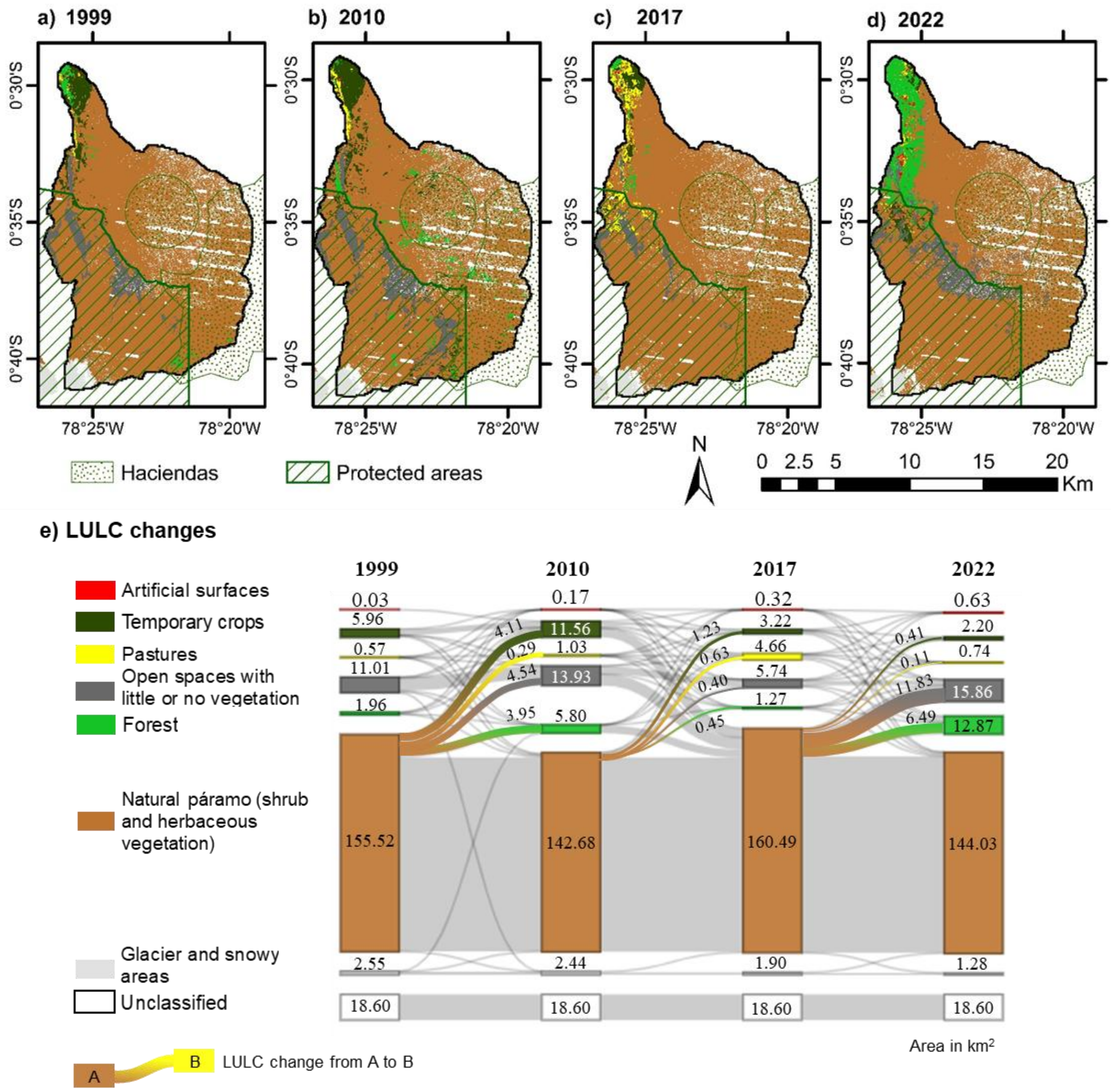


Fig. 1-5. LULC changes in UPitaRB between 1999 and 2022. Unclassified = Areas of persistent cloud cover located near snow-capped mountains.

Conversely, top-down management, characterised by national-level strategies, lacks the legal, economic, and political interconnections necessary for successful passive ecological restoration in páramo ecosystems. Without intervention, there is an expected decrease of approximately 30% in natural páramo, accompanied by an expansion of pine plantations linked to elevated total phosphorus levels in the basin. Hence, in UCutuchiRB, by 2022, its water quality was *marginal*, and only 31.6% of the natural páramo remained (Fig. 1-6), along with non-native pine forests linked in the correlation to high levels of total phosphorus in the basin. Agricultural and mining activities also contribute to the deterioration of water source quality in this basin, making it challenging to rely solely on passive ecological restoration.

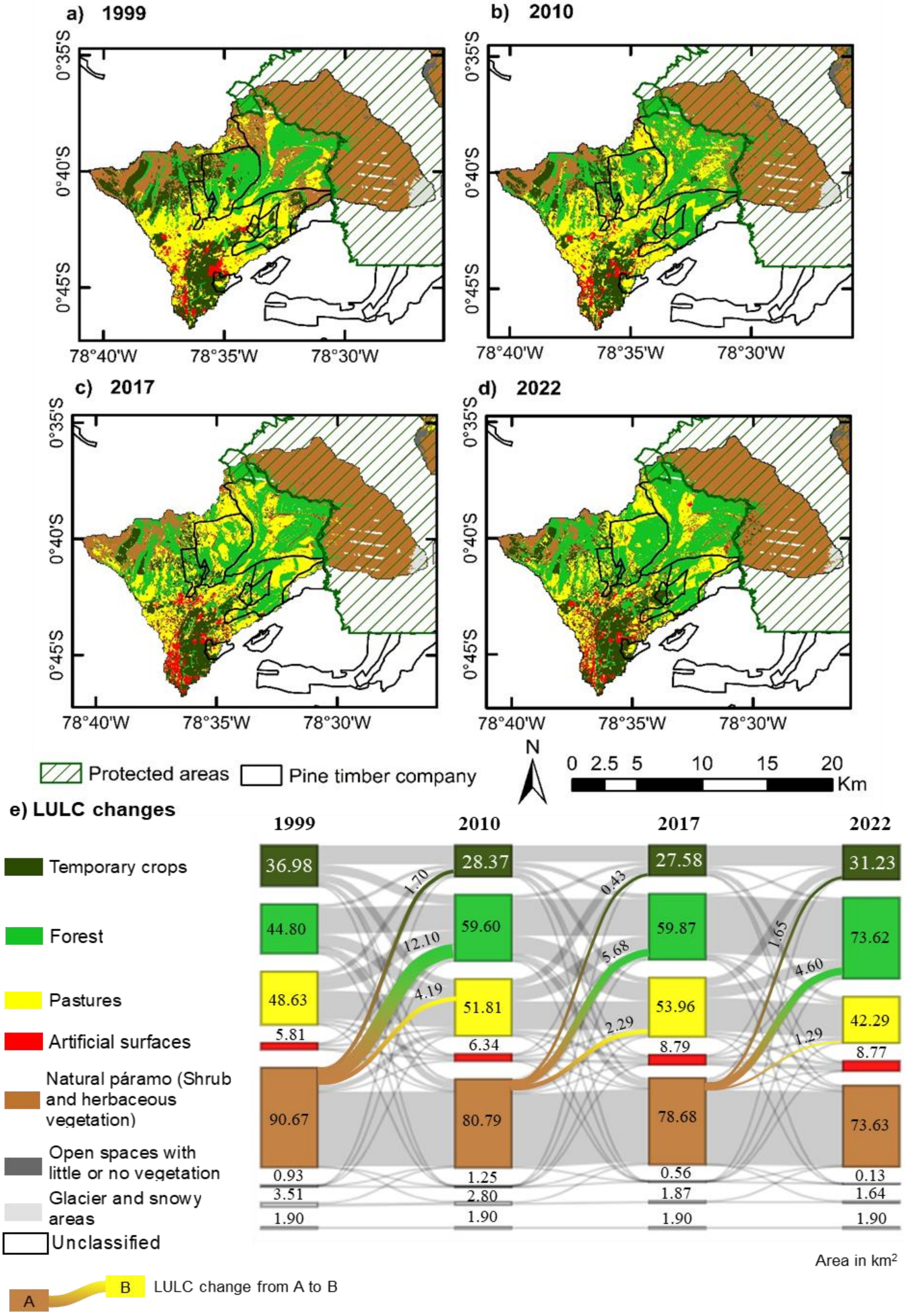


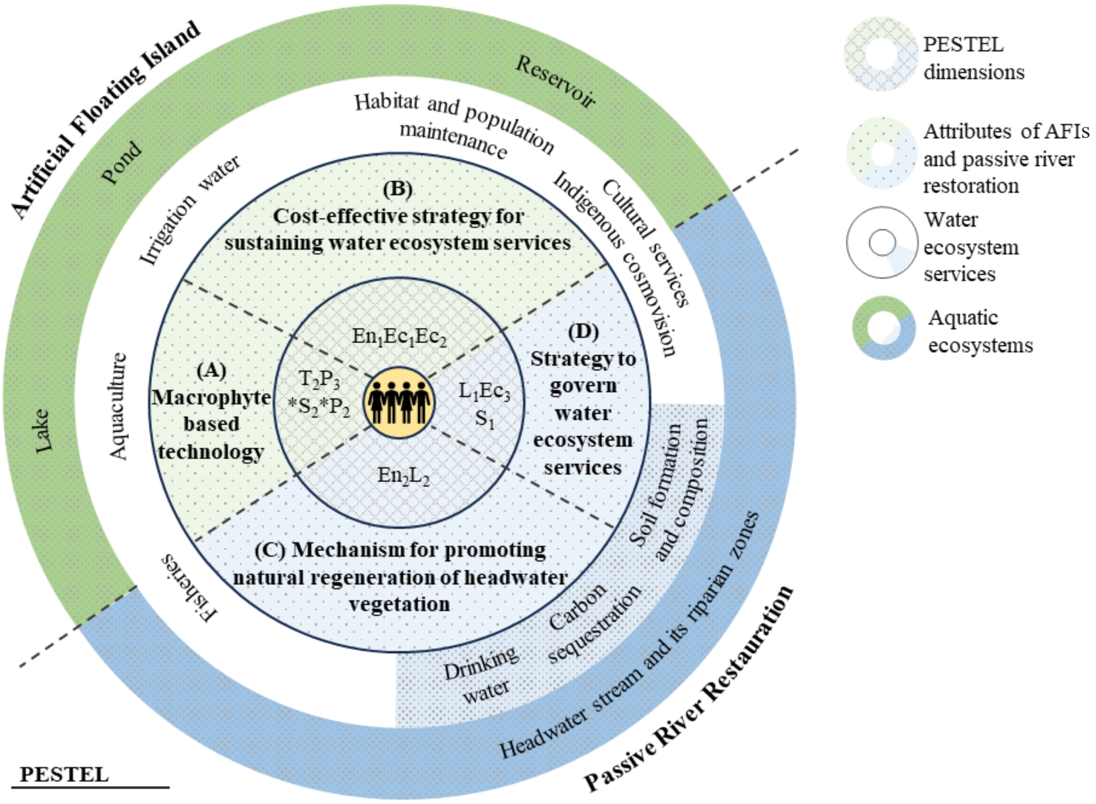
Fig. 1-6. LULC changes in UCutuchiRB between 1999 and 2022. Unclassified = Areas of persistent cloud cover located near snow-capped mountains.

This multi-method framework accurately identifies the PESTEL factors that successfully favour the implementation of passive ecological restoration, potentially contributing to multiple SDGs through sustainable and efficient use of natural resources (target 12.2; SDG 12) to reduce land degradation (target 15.3; SDG 15), restore aquatic ecosystems (target 6.6; SDG 6), and promote the conservation of mountain ecosystems (target 15.4; SDG15).

Stage 3: Merging artificial floating islands with passive ecological restoration in páramo communities

A novel framework was developed to show how combining AFIs and passive ecological restoration addresses the complexity of local community PESTEL factors influencing the sustainability of water ecosystem services.

Workshop participants reported the main PESTEL challenges influencing the sustainability of water ecosystem services in the páramo communities. Thus, the sustainability of water ecosystem services is primarily affected by the lack of political will for strategies that sustain healthy aquatic ecosystems, insufficient financial support, urban-centred environmental investment, and agricultural expansion regarding the political, economic, and environmental dimensions. Social, legal, and technological dimensions encompass community dissatisfaction, resistance to conservation, neglect of clean water and land use regulations, and limited innovation investment (Fig. 1-7).



PESTEL

P: Political

*P_1 Lack of political will to sustain healthy aquatic ecosystems.

*P_2 Limited local community participation in ecosystem management.

P_3 Absence of long-term strategies.

S: Social

S_1 Local community growth causes conflicts between water users in the upper, middle and lower basins.

* S_2 Dissatisfaction and resistance to conservation strategies.

En: Environmental

En_1 Local communities heavily rely on water ecosystem services for their livelihoods.

En_2 Agricultural expansion, grazing and urban development in páramo ecosystems degrade essential water ecosystem services.

Ec: Economical

Ec_1 Limited governmental financial support.

Ec_2 Focus on environmental investment in large urban centres.

Ec_3 Absence of investment returns.

T: Technological

*T_1 Mismatches between data and technology.

T_2 Scarce investment in technology innovation.

L: Legal

L_1 Laws to protect aquatic ecosystems are neglected.

L_2 Lack of strict sanctions against environmental violations.

* L_3 Regulations historically favoured large-scale agriculture in crucial aquatic ecosystems.

* Future research directions

Fig. 1-7. NbS to sustain water ecosystem services embedded in the local communities' PESTEL dimensions.

I examined the network view maps of two specific NbS for the local communities: AFIs and passive river restoration. The terms macrophytes, water quality, restoration, and governance and their associated items caught the attention in the network view maps, respectively. By associating these terms and items, and supported by the literature review, the artificial floating islands were characterised as (A) macrophyte-based technology and (B) a cost-effective strategy for sustaining water ecosystem services, aligning with multiple PESTEL dimensions of the local communities (Fig. 1-7). In turn, passive ecological restoration complemented the PESTEL dimension that the AFI system did not accommodate. Hence, it serves as a (C) mechanism to promote the natural regeneration of headwater vegetation and as a (D) strategy to govern water ecosystem services (Fig. 1-7).

Artificial floating islands are a cost-effective macrophyte-based technology, adaptable to limited financial support, inclusive of community plant knowledge, and beneficial for the health of aquatic ecosystems in both rural and urban landscapes. They contribute directly to vital water ecosystem services such as irrigation water supply, fisheries, and aquaculture, as well as indirectly to water purification and maintaining populations and habitats. Meanwhile, passive ecological restoration complements them by promoting the natural regeneration of headwater vegetation and serving as a strategy for governing water ecosystem services through government-led land initiatives. Primarily, it contributes to sustaining drinking water sources and indirectly soil formation and composition and carbon sequestration. Additionally, enhancing aquatic ecosystems has an impact on indigenous cosmovision.

Integrating AFIs with passive river restoration can also achieve several SDGs, including promoting sustainable and efficient natural resource use (Target 12.2, SDG 12), reducing water pollution (Target 6.3, SDG 6), restoring aquatic ecosystems (Target 6.6, SDG 6), mitigating land degradation (Target 15.3, SDG 15), and conserving mountain ecosystems (Target 15.4, SDG 15). This integration also supports sustaining drinking water sources and ensuring access to adequate, safe, and affordable basic services (Target 11.1, SDG 11), improving local community livelihoods (Target 1.4, SDG 1), and mobilising and sharing knowledge, expertise, technology, and financial resources to further the SDGs (Target 17.16, SDG 17)

Moreover, the lack of political will to maintain healthy aquatic ecosystems, the mismatches between data and technology, and regulations historically favouring large-scale agriculture in crucial aquatic ecosystems of the PESTEL dimensions of local communities extend beyond the artificial floating islands and passive ecological restoration (Fig.1-7). Since these challenges cannot be addressed by local community solutions alone, there is potential for future research to explore them in regional or national frameworks.

1.6 Implications and Outlook

I developed a novel framework contributing to more scientifically legitimate and publicly accountable decisions when designing NbS to achieve healthy aquatic ecosystems. This multi-method framework rigorously identifies the PESTEL factors influencing the implementation of artificial floating islands and passive ecological restoration as NbS in several local communities by synthesising diverse data sources and methodologies.

I unveiled that artificial floating islands and passive ecological restoration address the complexity of local community PESTEL factors to contribute directly to SDG 6 and indirectly to several other SDGs. However, some PESTEL factors pose significant challenges for those NbS. For instance, a lack of political will to maintain healthy aquatic ecosystems, the mismatches between data and technology, and regulations historically favouring large-scale agriculture in crucial aquatic ecosystems extend beyond the scope of these solutions. Given that local community solutions alone cannot address these challenges, future research could explore alternative strategies within regional or national frameworks. Additionally, the roles of climate change and local community involvement as catalysts for these NbS should be examined, specifically in developing-country communities, where these aspects have yet to influence these NbS initiatives.

In particular, artificial floating islands require thoroughly exploring their combination with policies like reinforcing training and educational programs, demanding corporate environmental responsibility, implementing payment for ecosystem services, and organising community committees to denounce non-compliance with the law. For passive ecological restoration, research should focus on high-elevation ecosystems across continents, often called water towers, to better understand its benefits and compare with results in páramo ecosystems. In addition, comparing the outcomes of bottom-up versus top-down management approaches in artificial floating islands and passive ecological initiatives will be crucial to determining the most effective management model. Recognising these aspects could enhance the contribution of these NbS to achieving more SDGs. Also, engaging local communities, such as indigenous people across several countries, could lead to understanding the impacts of aquatic ecosystem enhancement on indigenous cosmovision.

Lastly, challenges related to limited data availability underscore the need for future research to strengthen these methods. Grey literature was incorporated into the PESTEL analysis due to the scarcity of peer-reviewed research on sustainable land and water management strategies. Similarly, limited LULC classifications have been performed in páramo ecosystems because they are typical cloud-prone regions that restrict data availability. Furthermore, given the absence of up-to-date official water quality monitoring stations, data collection relied on sources such as universities, local government agencies, and NGOs. While the results provide a valuable approximation of the environmental impacts of NbS, ongoing efforts to secure continuous, representative, and accurate datasets would enhance the validity of these findings.

Chapter II: Using fuzzy cognitive maps to promote nature-based solutions for water quality improvement in developing-country communities

This chapter is published as:

Fonseca, K., Espitia, E., Breuer, L., & Correa, A. (2022). Using fuzzy cognitive maps to promote nature-based solutions for water quality improvement in developing-country communities. *Journal of Cleaner Production*, *377*, 134246. https://doi.org/10.1016/j.jclepro.2022.134246.

Abstract

An adequate strategy for water quality improvement must consider a range of political, economic, social, technological, environmental, and legal (PESTEL) concepts. Nature-based solutions have emerged as promising tools to improve water quality while considering these factors. In this context, fuzzy cognitive maps and the PESTEL approach have been merged to i) identify the principal concepts that affect water quality from different perspectives, and ii) theoretically explore the use of artificial floating islands as a measure of nature-based solution combined with different policies, to find strategies to improve water quality given local conditions. For this purpose, three Ecuadorian communities are used as scenarios. The communities are located in different geographical regions, i.e., páramo (an alpine tundra ecosystem), coastal mangrove, and tropical rainforest. From literature-based research, 40 PESTEL concepts are identified, then local experts recognize relevant concepts related to water quality deterioration regarding local conditions, and the communities develop fully democratic social cognitive maps. The cognitive map from the páramo community is constructed with 17 concepts mainly driven by environment (23%) and economy (23%). The major problem identified is natural water pollutants with the highest centrality value ($c_i = 12.22$). The mangrove community uses 19 concepts influenced by policy (48%), and the major issue is human exposure to environmental pollutants ($c_i = 16.27$). The rainforest community uses 15 concepts related to the economy (40%), and the major problem is the violation of environmental legislation ($c_i = 15.96$). Our pioneer work to predict the future of

water management shows that in the worst-case scenario, more than 85% of concepts are affected in all communities. However, the implementation of policy strategies in combination with artificial floating islands demonstrates a large potential for improving water quality. With this study, we provide a novel, inclusive, and locally adapted framework to guide future water management and contribute to achieving the Sustainable Development Goal SDG 6.

2.1 Introduction

Healthy aquatic ecosystems (e.g., rivers, lakes, coastal waters, mangroves) provide local communities with valuable ecosystem services ranging from food, biodiversity, energy, and water supply for consumption and recreation (Alam et al., 2017; Brown et al., 2021; Grizzetti et al., 2016). However, impaired water quality contributes to the general problem of water availability, threatening livelihoods and socio-economic growth (Beitl et al., 2019; Grigg, 2016). Absent or inadequate water management is linked to the presence of water-related diseases such as cholera, diarrhea, dysentery, hepatitis A, typhoid, or polio (WHO, 2019). Problems related to poor water quality are even more common in developing countries (UNICEF, 2020), which often face economic water scarcity due to political, social, institutional, or financial conditions (Gude, 2017). Under these circumstances, achieving target 6.3 of the Sustainable Development Goals ("*By 2030, improve water quality by reducing pollution, eliminating dumping and minimizing release of hazardous chemicals and materials, halving the proportion of untreated wastewater and substantially increasing recycling and safe reuse globally*") represents a significant challenge for communities in developing countries which face water quality issues.

A holistic strategy for water quality improvement must consider the macro-environmental factors affecting its quality, and the engagement of multisectoral stakeholders involved in water management decisions (Ghaboulian Zare et al., 2022; Giordano et al., 2020; Grigg, 2016; Özesmi & Özesmi, 2004; Sansa et al., 2021; Wilson et al., 2018). The exploration of macro-environmental factors based on the political, economic, social, technological, ecological, and legal (PESTEL) concepts has gained increased attention in recent years (Iacovidou & Zorpas, 2022). With an in-depth analysis, decision-makers can identify PESTEL concepts to be monitored as opportunities and threats in the implementation of new strategies (De Sousa & Castañeda-Ayarza, 2022) to solve water quality issues.

Strengthening the links between humans and nature increase the affordability of clean water by implementing nature-based solutions, i.e., remediation methods that emulate natural processes. Among nature-based solution strategies, Artificial Floating Islands (AFIs) have proved to be efficient in improving water quality at low operational costs, low energy consumption, and with negligible environmental impact (Benvenuti et al., 2018; Fonseca, 2021; Martelo & Borrero, 2012; Stewart et al., 2008). AFIs are composed of floating platforms that support the growth of sediment-rooted emergent wetland plants, macrophytes, microbes, and related ecological communities such as algae, biofilms, zooplankton, and small invertebrates (Colares et al., 2020; Yeh et al., 2015). By using AFIs, polluted aquatic ecosystems are restored as water passes beneath the floating mat through the following mechanisms: plant root uptake of metals and nutrients, biofilm development, extracellular enzyme release, contaminant settling, and binding, as well as suspended matter flocculation enhancement (Yeh et al., 2015). AFIs as a potential strategy in combination with water management policies are more feasible to implement in small, low-income communities, especially in developing countries where treatment infrastructure is inadequate or limited to large cities (ONU, 2018). Several studies (Kusin et al., 2019; Lu et al., 2015; Nuruzzaman et al., 2021; Stewart et al., 2008; Yeh et al., 2015) have demonstrated the potential of AFIs to clean polluted water from natural and anthropogenic sources in the communities within the scope of the aforementioned. However, there is a lack of experimental data (Colares et al., 2020) and multidisciplinary studies that engage stakeholders and consider PESTEL concepts to design AFIs implementation for the improvement of water quality (Benvenuti et al., 2018; Fonseca et al., 2020; Negrete et al., 2019.; Yeh et al., 2015). Moreover, several AFIs projects face sustainability issues due to insufficient community engagement and multisectoral planning, limited funds and technical support (Watkins et al., 2017).

Fuzzy Cognitive Maps (FCMs) are tools to analyze complex systems from the perception of stakeholders. They can extract qualitative key information to support decision-making in ecosystem management (Adriaenssens et al., 2004; Vergini & Groumpos, 2021). FCMs are based on human reasoning and linguistic approach to deal with vague, and uncertain data that are interpreted in fuzzy rule-based models.

This study aims to develop a decision-making framework that identifies the principal concepts that affect water quality at the community level and explores the implementation of AFIs together with policy interventions to improve water quality. We implemented the framework in three Ecuadorian communities with diverse socio-cultural backgrounds and located at different geographical regions with different hydroclimatic and physiographic conditions. Yet they all share the common ground of water quality issues and that water and its associated ecosystem services provide livelihood, and food and also play a crucial role in family income generation (Calle et al., 2018; Delgado-Aguilar et al., 2017; García et al., 2019a). Based on a FCM approach, we will answer the three following questions: (1) What are the PESTEL concepts related to water quality deterioration at the community level considering multi-sectoral stakeholder perceptions? (2) How can local water quality be improved if communities implement AFIs? (3) Which policies of the PESTEL concepts can be combined with AFIs to strengthen their effectiveness to improve water quality?

2.2 Materials and Methods

2.2.1 Study area

This study analyses three communities in Ecuador (Fig. 2-1): Chilla Chico in the Cotopaxi province, located in the páramo of the Andes (Fig. 2-1a), Mogollon in the Guayas province, a community characterized by mangroves at the Pacific Coast (Fig. 2-1b), and Awayaku in the Napo province, situated in the Amazon rainforest region (Fig. 2-1c). Hereinafter referred to as the Páramo, Mangrove, and Rainforest community, respectively.

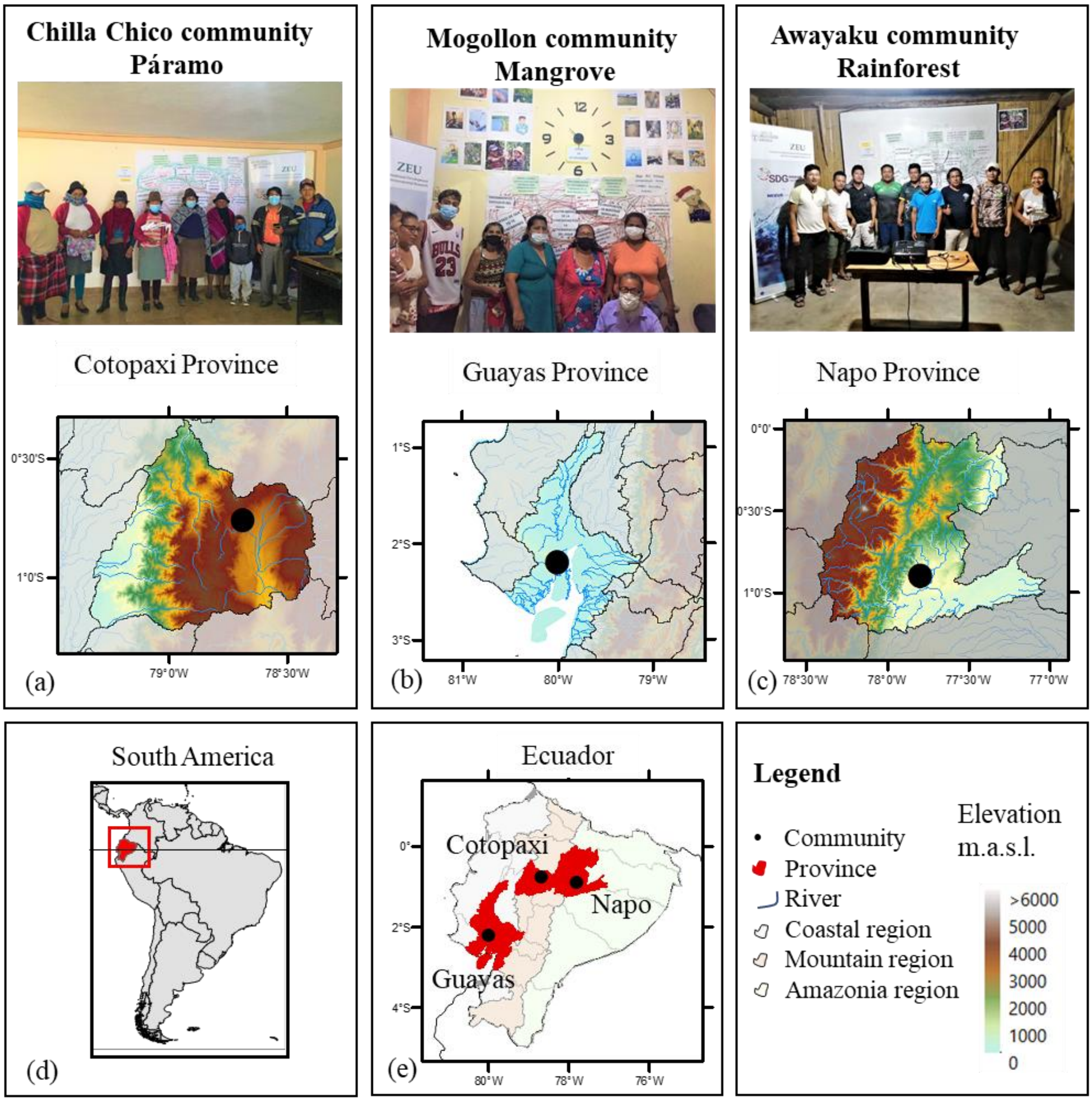


Fig. 2-1. Study area, (a) Chilla Chico community, (b) Mogollon community, (c) Awayaku community, (d) location of Ecuador in South America, and (e) location of the study communities (black dots) and respective provinces (red surfaces) in Ecuador.

The Parámo community is home to approximately 1,900 low-income indigenous people who subsist on the farming of potatoes, corn, and beans at 3,000 m above sea level (m.a.s.l.) (Oña et al., 2021; Terán Cazar, 2017). In this area, the páramo ecosystem provides water mainly for human consumption and irrigation. However, pollutants have impaired the ecosystem, including the community irrigation ponds. These contaminants mainly include arsenic and other heavy metals from the Iliniza Volcanic Complex (Fonseca et al., 2020); nitrogen and phosphorus from livestock and fertilizer application; and fecal coliforms from the feces of all warm-blooded animals and humans (Fonseca & Clairand, 2018).

In the Mangrove community, 1,750 inhabitants largely depend on fishing along the Mogollon estuary bank, one of the most important tributaries of the Salado estuary (Hechavarría Hernández et al., 2018). The Salado estuary is characterized by extensive tidal bays within river mangroves that provide nursery habitat and feeding grounds for numerous species of fish, crustaceans, mollusks, and wading birds (Calle et al., 2018). High concentrations of heavy metals such as cadmium, copper, chromium, and lead (Fernández-Cadena et al., 2014), fecal coliforms, and low dissolved oxygen content (Negrete et al., 2019) are present in these coastal estuarine systems due to the accelerated urban development in the last eight decades (Calle et al., 2018).

The Rainforest community, is a small community of 700 indigenous people from the Kichwa Rukullacta group, located in the upper tributaries of the Napo River between 520 and 1,240 m a.s.l.. The community integrates subsistence agriculture with tilapia and carachama fishing as the primary source of economic income (GADR San Pablo, 2015; Pueblo Kichwa de Rukullakta, 2018). In the Napo River, concentrations of cadmium, lead, copper, zinc, and mercury have increased, mainly due to legal and illegal small-scale mining, sewage discharges, fish farming, and non-functional landfills (Capparelli et al., 2020).

All three communities are exposed to impaired water quality along the water-soil-crop-food chain, but its potential impact on public health is unknown. Nevertheless, several diseases are linked to water contaminated with heavy metals and feces from untreated or inadequately treated sources (Qian et al., 2022). The main toxic effects of heavy metals on humans include nephrotoxicity, neurotoxicity, hepatotoxicity, skin toxicity, and cardiovascular toxicity (Mazumder, 2008; Mitra et al., 2022; Srivastava et al., 2012). Moreover, gastro-enteric epidemics caused by water contaminated with human and animal feces are a major health concern, not only due to considerable human morbidity and mortality but also because of alarming rate in the spread of drug-resistant bacteria (Some et al., 2021). In the three communities, interviews and workshops are held to build FCMs based on the perception of multi-sectorial stakeholders regarding water quality improvement.

2.2.2 Fuzzy cognitive maps

The FCM approach is a semi-quantitative method that depicts expert knowledge, causal reasoning, and stakeholder perceptions by signed directed graphs (Gray et al., 2015; Moosavi et al., 2021; Santoro et al., 2019). By applying the principles of fuzzy logic and cognitive maps, different sources of knowledge can facilitate the decision-making processes (Shahvi et al., 2021). This knowledge is framed in three main steps forming the FCM approach: listing of concepts, drawing causal relationships among these concepts by cognitive maps, and estimating the fuzzy inference.

- ***List of concepts***

The procedure first involves identifying concepts and then selecting the most important ones to include in the map. Different approaches can be used, such as selecting a list of preliminary concepts from the literature (Morone et al., 2021) or from meetings with experts (Özesmi & Özesmi, 2004). Subsequently, a close-ended survey is conducted to validate the concepts based on the opinion of the experts, who rate the level of connection between the concepts and the aim of the study using a unipolar Likert scale (Ghaboulian Zare et al., 2022; Morone et al., 2021).

- ***Drawing of social cognitive maps***

The principal output of the FCM approach is a mental map, structured by nodes and bidirectional connections between nodes. Nodes are concepts, representing physical quantities or abstract ideas, while connections represent the influencing degree (C_{ij}) of the cause of concept i on the effect concept j (Kosko, 1986). The stakeholders participating in the map construction decide when a connection exists by assigning a status value of concept represented as a number within [0, 1], and subsequently weights (w_{ij}) to concept pairs (between −1 and 1). When w_{ij} is positive, there is a positive influence of concept C_i on concept C_j. On the contrary, when w_{ij} is negative, there is a negative influence of concept C_i on concept C_j. When w_{ij} is equal to zero, it is assumed that there is no relationship between concepts (Kosko, 1986). The degree of influence among the concepts is coded into an adjacency matrix (a_{ij}) (Chen & Chiu, 2021). This matrix is structured by three concept types (i.e., receiver, driver, and ordinary) that interact with each other. The receiver concepts accept input from others, being most influenced by others (Özesmi & Özesmi, 2004). The policy strategies or policy drivers are driver concepts in the FCM, ideal candidates to manipulate the system due to their nature of sending stimuli and not receiving incoming connections (Ghaboulian Zare et al., 2022; Gray et al., 2014; Morone et al., 2021; Solana-Gutiérrez et al., 2017). Concepts with both driving and receiving features are called ordinary concepts (Gray et al., 2015; Özesmi & Özesmi, 2004). A large number of concepts and connections indicate a major degree of interaction between concepts in the cognitive map (Gray et al., 2014). Following the above guidelines, the stakeholders can build two types of cognitive maps: individual or social. According to Mourhir (2004) and Özesmi & Özesmi (2004) to build social cognitive map compared to the individual one, it consumes less time and resources, as it can be produced collaboratively by stakeholders in a workshop and facilitates social learning.

One way to evaluate the complexity of a FCM model is by using graph theory indices (e.g., density, hierarchy index, outdegree, indegree, centrality) that are used to analyze the concept contribution in the cognitive map.The number of concepts (N) and connections (C) represent the density of the map (D) (Nikas et al., 2019). When the stakeholders identify a large number of causal relationships among concepts, the density of the map (Eq. 1) is high (Blacketer et al., 2021; Özesmi & Özesmi, 2004; Shahvi et al., 2021).

$$D = \frac{C}{N(N-1)} \tag{1}$$

The hierarchy index (h) represents a democratic or hierarchical map. When the hierarchy index (Eq. 2) is equal to 1, the map is fully hierarchical. On the contrary, when h is equal to 0, the system is fully democratic (Mourhir, 2021; Özesmi & Özesmi, 2004).

$$h = \frac{12}{(N-1)N(N+1)} \sum_i \left[\frac{od(v_i) - (\sum od(v_i))}{N} \right]^2 \tag{2}$$

The outdegree indice $od(v_i)$ (Eq. 3) corresponds to the row sum of absolute values v of a concept i. It describes the cumulative strengths of connections (a_{ij}) leaving the concepts.

$$od(v_i) = \sum_{k=1}^{N} |a_{ik}| \tag{3}$$

The indegree indice $id(v_i)$ (Eq. 4) is calculated as the sum of the absolute weights of the incoming FCM graph edges.

$$id(v_i) = \sum_{k=1}^{N} |a_{ki}| \tag{4}$$

A high value of $od(v_i)$ and zero of $id(v_i)$ classifies the concept as a driver, while a high value of $id(v_i)$ and zero of $od(v_i)$ classifies the concept as a receiver (Özesmi & Özesmi, 2004; Solana-Gutiérrez et al., 2017). The outdegree and indegree together describe the centrality (c_i) of the system (Eq. 5), which represents the overall importance of a given concept in the causal flow of the cognitive map (Ghaboulian Zare et al., 2022; Nikas et al., 2019; Özesmi & Özesmi, 2004; Schiavon et al., 2021).

$$c_i = td(v_i) = od(v_i) + id(v_i) \tag{5}$$

- ***Fuzzy inference***

The fuzzy inference system involves three stages i.e., the natural dynamic simulation, worst-case scenario and scenarios with policy drivers (Gray et al., 2015). In the natural dynamic simulation, the receiver and ordinal concepts are activated (0 means no-activate and 1 means activate) to predict a scenario without external influences, known as the steady-state (Eq. 6) (Adriaenssens et al., 2004). Based on the collective stakeholder knowledge, each activated Concept contributes its weight to activate its descendent, interacting with each other

(Solana-Gutiérrez et al., 2017). That means a steady-state snapshot of how the concepts and linkages of the system, given the current structure, would be resolved in the absence of change or intervention, at different periods (Chen & Chiu, 2021; Kosko, 1986; Lopolito et al., 2020).

$$A_i^{t+1} = f\left(\sum_{j=1}^{n} A_j^t \, . \, W_{ji}\right) \tag{6}$$

That is, A_i^t is the status value of concept C_i at period t; A_j^t is the status value of concept C_j at period t; A_i^{t+1} is the status value of concept C_i at period t+1; W_{ji} is the corresponding fuzzy relation degree between C_j and C_i; and f is a sigmoid threshold function (Eq. 7).

$$f(x) = \frac{1}{1 + e^{-\lambda x}} \tag{7}$$

Specifically, $\lambda > 0$ determines its steepness (Vergini & Groumpos, 2021; Xiao et al., 2012). The sigmoid function transforms the status value of each node into the interval [0, 1] at each iteration (Chen & Chiu, 2021), providing non-negative values that are easy to compare and allow to reach steady-state scenario (Lopolito et al., 2020).

In the second stage, a worst-case scenario is estimated in which the concept with the highest centrality value (Eq. 5) is clamped to a maximum value of 1 (Eq. (6) and Eq. (7)) to understand its influence on the others. The outcome of this scenario is compared with the steady-state to understand the relative changes (Gray et al., 2015).

Finally, the third stage determines what may result if drivers concepts or policy drivers are implemented. A similar procedure to the one described in the previous step is followed. The only difference is that the policy drivers are now set at their maximum value (1) (Morone et al., 2021). The policy effect is assessed by calculating the difference of the policy intervention and the steady-state scenarios.

2.2.3 Implementation of the FCM approach in the communities

Following the steps described in section 2.2, the FCMs were co-created with the communities. A flowchart for its implementation is presented in Fig. 2-2.

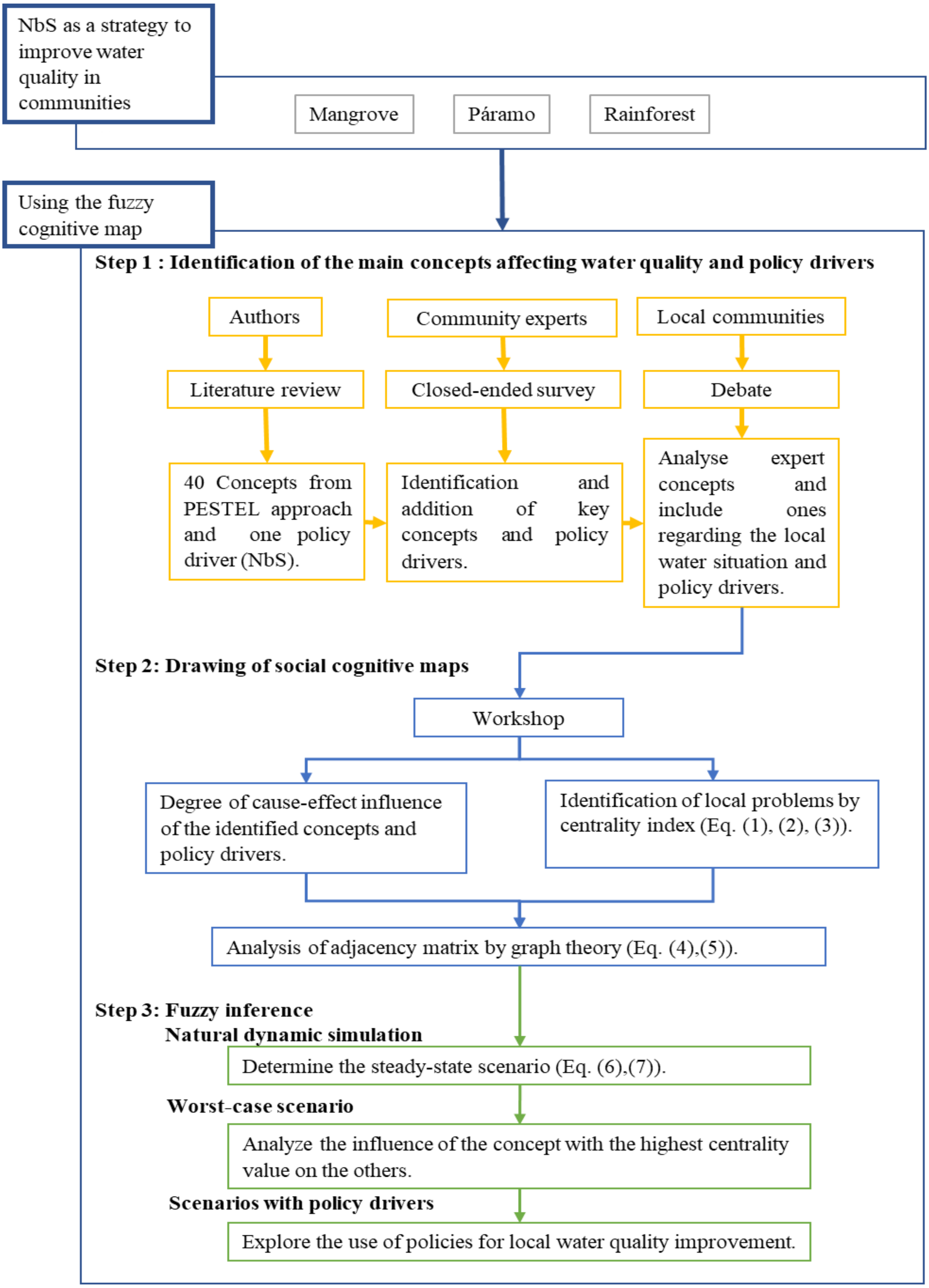


Fig. 2-2. Flowchart to build FCMs as a decision-making framework to improve water quality at community level. NbS = nature-based solutions.

Step 1: identification of the main concepts affecting water quality and policy drivers

A literature review was conducted (Morone et al., 2021). We selected receiver and ordinary concepts related to water quality issues under the PESTEL approach, and a driver concept based on nature-based solutions to improve the water quality. We used various databases (Scopus, ScienceDirect, SpringerLink, Web of Science) to search for terms that include a combination of keywords related to water pollution, water management-governance, water and SDGs, water treatment technology, water economy, and water users.

Studies were included in the review if they fullfiled the following criteria: are written in English to focus on the literature of international impact; are studies from 2000 to 2022, i.e., since the term water governance first appeared concerning the range of political, social, economic, and administrative systems (Baumgartner & Pahl-Wostl, 2013); and investigate the link between water deterioration and PESTEL approach. Initially, 50 research papers were identified with these criteria, based on the title and abstract. Subsequently, the papers were examined to extract concepts related to our study goal. Tweenty two papers met this purpose (Table 2-1).

Table 2-1. PESTEL concepts and policy drivers, extracted from literature review.

Political (P)		
Concepts	**Id**	**Studies**
Environmental education focused on key ecosystems	P_1	(Sarkar et al., 2007)
Land use planning	P_2	(Gyawali et al., 2013)
Decision-makers without adequate training	P_3	(Apostolaki et al., 2019)
Short-term environmental projects without results	P_4	(San Llorente Capdevila et al., 2020)
Few legitimate conservation policies for key ecosystems	P_5	(Rahm et al., 2013)
Experts in decision making	P_6	(San Llorente Capdevila et al., 2020)
Initiatives to accelerate progress on the SDGs	P_7	(Alcamo, 2019)
Strong institutions	P_8	(Davies & Mazumder, 2003)

Economic (EC)		
Concepts	**Id**	**Studies**
Fish production	EC_1	(Pulford et al., 2017)
Food Security	EC_2	(Grigg, 2016)
Costly treatments of water	EC_3	(Pilon-Smits, 2005)
Inhabitants incur medical costs	EC_4	(Grigg, 2016)
Water demand for economic activities: crop and fishing	EC_5	(Gyawali et al., 2013)
Support to sustainable productive projects	EC_6	(D. Li et al., 2021)
Economic growth	EC_7	(Brockwell et al., 2021)
Water quality for food production	EC_8	(Pulford et al., 2017)
Social (S)		
Concepts	**Id**	**Studies**
Stakeholders involvement	S_1	(Wilson et al., 2018)
Population growth	S_2	(Brockwell et al., 2021)
Lack of community cohesion	S_3	(Wilson et al., 2018)
Persistent equity issue	S_4	(Grigg, 2016)
Poverty rate	S_5	(Grigg, 2016)
Sustainable communities	S_6	(Davies & Mazumder, 2003)
Social well-being	S_7	(Brockwell et al., 2021)
Achieve gender equality	S_8	(Anderson et al., 2021)
Technological (T)		
Concepts	**Id**	**Studies**
Alternative water technology	T_1	(Sarkar et al., 2007)
Local development technology	T_2	(Rolfe & Harvey, 2017)
Innovation in the water sector	T_3	(Nyiwul, 2021)
Environmental (EV)		
Concepts	**Id**	**Studies**
Natural water pollutants	EV_1	(Samal et al., 2011)
Anthropogenic water pollutants	EV_2	(Oladipo et al., 2021)
Climate change	EV_3	(Moosavi et al., 2021)
Waterbody restoration	EV_4	(Moosavi et al., 2021)
Conventional treatments	EV_5	(Sarkar et al., 2007)
Ecosystem conservation	EV_6	(Brockwell et al., 2021)
Human exposure to environmental pollutants	EV_7	(Hartmann et al., 2018)
Safe water	EV_8	(Maurice et al., 2019)
Legal (L)		
Concepts	**Id**	**Studies**
Right to live in a healthy environment	L_1	
Environmental law violation	L_2	(Wilson et al., 2018)

Legal actions by users	L_3	(Maurice et al., 2019)
Water governance crisis	L_4	(Rolfe & Harvey, 2017)
Dissemination of water laws to communities	L_5	(Wilson et al. 2018)
Policy driver (D)		
Concepts	**Id**	**Studies**
Artificial Floating Island	D_{AFIs}	(Yeh et al. 2015)

We identified 40 initial concepts and the community experts validate their PESTEL linkage according to the local situation of each community. The group of local experts (Appendix A) was chosen based on their knowledge and influence in water management and decision making. They invited other specialists following the snowball sampling approach. Thus, the final number of experts was 10 in páramo and mangrove, respectively, and 8 in the rainforest community. Socialization meetings were held face-to-face or in remote mode to explain the objective and steps of the methodology (Fig. 2-2). After that, closed-ended surveys were sent to the 28 experts to answer the question: Which of the 40 concepts do you think is definitely connected when considering the deterioration of water quality according to water use (agriculture and aquaculture, respectively)? Using a score unipolar Likert scale 5-point (i.e. from 5 = definitely connected to 1 = definitely NOT connected). In addition, experts were asked to add missing concepts and policy drivers to improve water quality. Concepts and policy drivers added by experts were coded according to the community: páramo (P), mangrove (M), and rainforest (R), plus (E) for experts and the number to which it belongs in the PESTEL table. Finally, to identify the representative concepts based on the experts' opinions, we conducted an exploratory analysis of the survey responses by assessing the frequency and standard deviation of responses. The concepts with the highest averages and the lowest standard deviations were selected, to reflect the highest degree of connection among the opinions of the experts.

To include the perception of the local communities, three workshops were held in December 2021, one in each community with the participation of 10 people. In the workshops, the concepts and policy drivers defined by the experts were debated in a fully democratic and participative atmosphere. In the first part of the workshop, each concept and driver policy (D_{AFIs}) were explained and posted on the wall for the community to remember as the build the FCMs. Besides, local communities added the missing concepts and policy drivers according to their perception, which were coded in the same way as for experts, with the only difference being the letter (C) for communities.

Step 2: Drawing of Social Cognitive Maps

The instructions to draw the social cognitive maps were explained to the local communities by video. The community received a sheet of paper with the predefined concepts (ordinary and receiver) printed at random and asked to rate the degree of cause-effect influence of the concepts. The rating was performed using arrows and rating the degree of influence between these connections using expressions via a 6-degree scale, ranging from -3 (strong negative) to +3 (strong positive) (Nyaki et al., 2014). To end, the group average perception was assessed (Gray et al., 2014). Finally, the cognitive maps provided by the communities were converted into adjacency matrices (Özesmi & Özesmi, 2004) to analyses indices of graph theory (Eq. 1-5) (Mourhir, 2021) using the FCMapper software (available: http://www.fcmappers.net/joomla/).

Step 3: Fuzzy inference

Using the calculation algorithm presented in Eq. (6) and (7) (FCMapper), the model was run to get the steady-state and make a comparison with the worst-case and policy intervention scenarios. In the first stage, the worst case scenario represents the maximum influence of the main concept that affect water quality at the community level on the other PESTEL concepts. For the second stage, the communities were asked to score policy drivers (via a 6-degree scale as step 2) to understand how they might mitigate unwanted outcomes. In this policy intervention scenario, the drivers were introduced individually i.e., the drivers were set to 0 except the selected one, to isolate and assess the role of each policy driver on the community model. The value for the simulated scenario was set to 1, urging the system to fully adopt the policy measure implemented (Morone et al., 2021). Finally, the policy effect was calculated using the difference between the steady-state and each policy intervention.

2.3 Results

2.3.1 Identification of the main concepts affecting water quality

- ***Páramo community***

The community experts chose 14 out of 40 initial concepts PESTEL, then added two new ones, and the community added one concept (Table 2-2a) . In total 17 concepts related to political 18%, environmental 23%, social 18%, technological 0%, economic 23%, and legal 18% factors (Fig. 2-6a). The community identified as the most influential concept, i.e., with the highest centrality value EV_1 ($c_i = 12.22$) natural water pollutants (Table 2-2a). The community linked the presence of EV1 mainly as a cause of the decline of S7 (-1), and increase of EC3 (+1) (Fig. 2-3). The impact of EV_1 on the other concepts is relatively minor (Fig. 2-3). Concerning the highest outdegree of the model, P_3: Decision-makers without adequate training (*od* = 10.25), reveal a strong influence on other concepts; however, our focus is on the centrality value as it controls the dynamics of the whole community model. In addition, table 2-2a shows that EC_3: Costly treatments of water, EC_5:Water demand for economic activities: crop, L_3: Legal actions by users, and L_4: Water governance crisis have the highest steady-state value; nonetheless, their outdegrees are lower as well as influence on the model. The most influenced concepts, determined via indegree scores, are L_3: Legal actions by users and EC_7: Economic growth, in decreasing order.

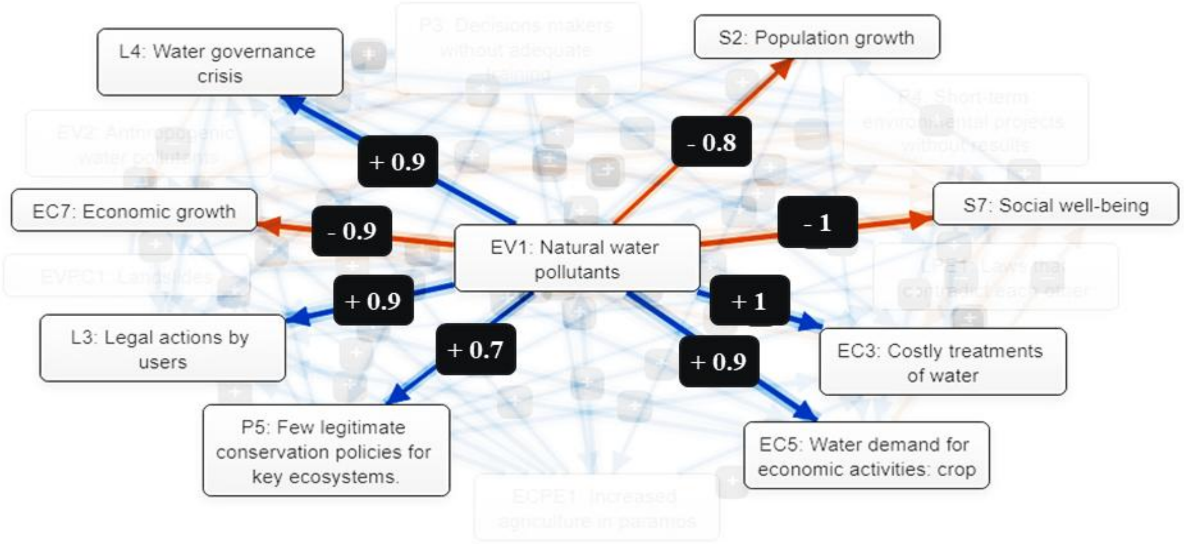


Fig. 2-3. The main problem and its impact on PESTEL concepts identifed in the social cognitive map by the páramo community (full map in appendix B(1)) .Blue lines represent positive influence of concept *Ci* on concept *Cj*, and orange lines represent a negative influence of concept *Ci* on concept *Cj*. Line thickness indicates the degree of influence between components**.**

- ***Mangrove community***

The experts chose 11 out of 40 PESTEL and added seven new concepts, and the community one (Table 2-2b). Hence, the model is composed of overall 19 concepts which can be grouped as political 48%, environmental 21%, social 21%, technological 0%, economic 21%, and legal 5% (Fig. 2-6b). The community identifyed the major water quality issue in view of human exposure to environmental pollutants EV$_7$ (c_i=16.27) which belongs to the environmental concept group (Table 2-2b). The community connected the EV$_7$ concept primarily as a cause of the decline of EC$_8$ (-1), and increase EC$_4$ (+1) (Fig. 2-4). The impact of EV$_7$ on the other concepts is relatively minor (Fig. 2-4). Moreover, in table 2-2a, EV$_5$: Conventional treatments has the highest outdegree value (*od* = 9.93); however, the concept is not perceived as the most important in the model i.e., it has a lower centrality value than EV$_7$. Also, EC$_3$: Costly treatments of water and EC$_4$: Inhabitants incur medical costs have the highest steady-state value, but the low outdegrees indicate a slight influence on the model. Finally, the community perceived as the most affected concepts, regarding indegree score,

S_7: Social well-being, EC_4: Inhabitants incur medical cost, and EV_{E9}: Waste disposal problems.

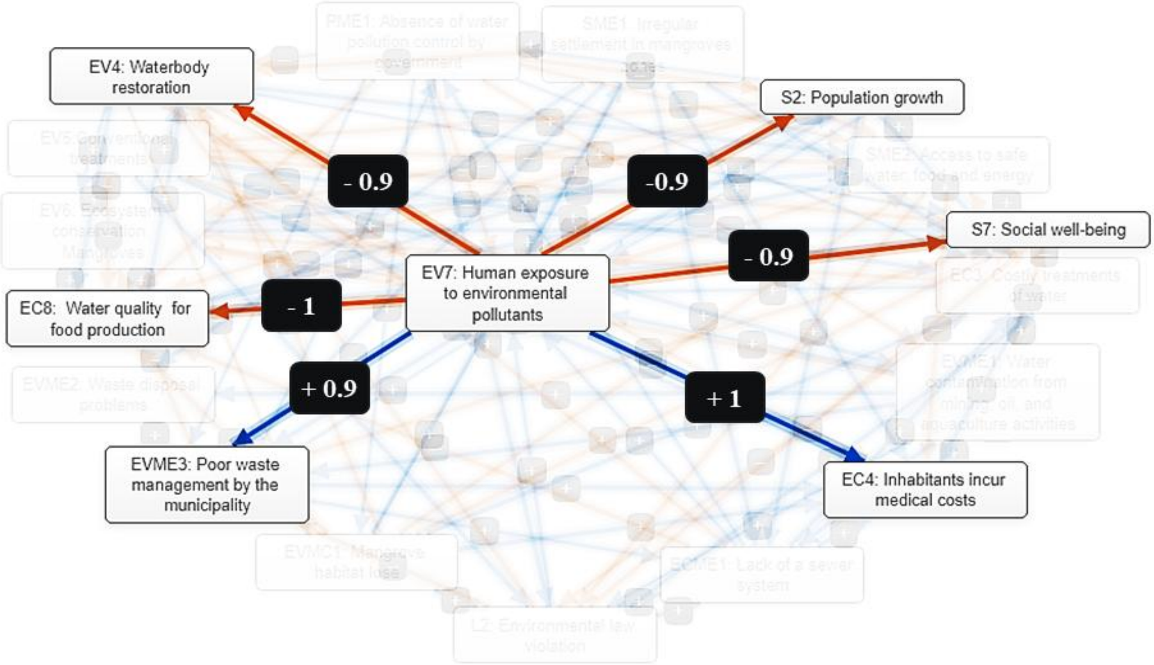


Fig. 2-4. The main problem and its impact on PESTEL concepts identifed in the social cognitive map by the Mangrove community (full map in appendix B (2)). Blue lines represent positive influence of concept C_i on concept C_j, and orange lines represent a negative influence of concept C_i on concept C_j. Line thickness indicates the degree of influence between components**.**

- ***Rainforest community***

The experts selected 13 out of 40 PESTEL concepts and add one more. The community proposed an additional one (Table 2-2c). The model consists of 15 concepts, split into political 0%, environmental 26%, social 20%, technological 7%, economic 40%, and legal 7% concepts (Fig. 2-6c). The community connected L_2 with a high impact on EV_7 (+1), and a decrease in EV_6 (-1) (Fig. 2-5). The impact of L_2 on the other concepts is relatively minor (Fig. 2-5). The community identified (L_2) as being most relevant from the perspective of centrality, outdegree, and steady-state value (c_i = 15.96, od = 9.3, and steady-state = 1), and S_7: Social well-being is perceived, regarding indegree score, as the concept most affected (Table 2-2c).

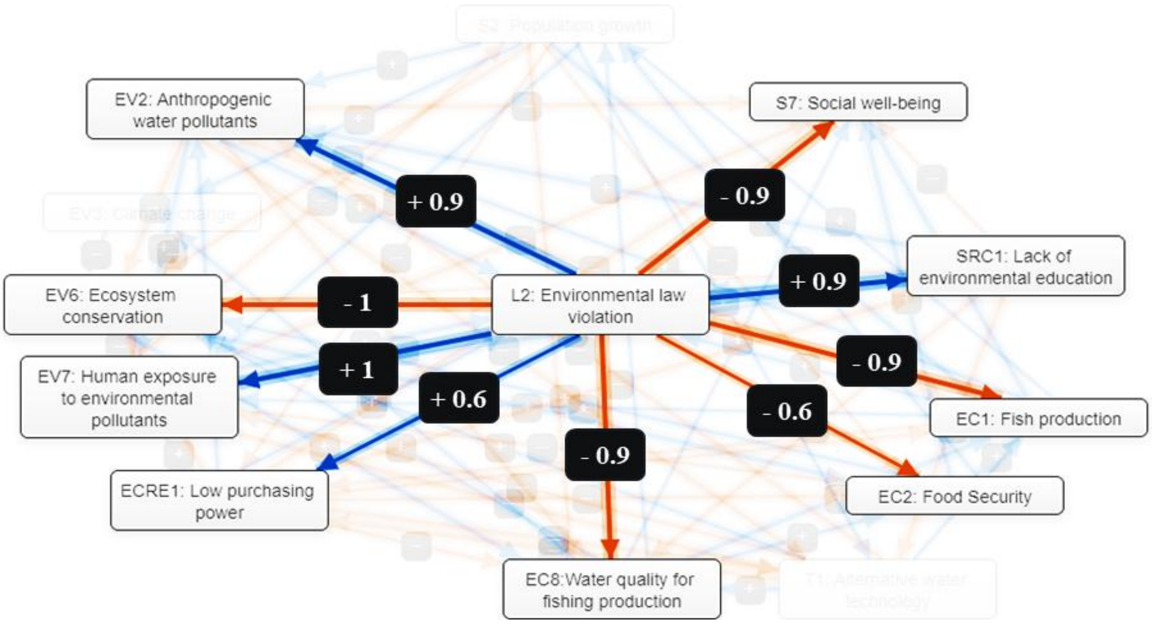


Fig. 2-5. The main problem and its impact on PESTEL concepts identifed in the social cognitive map by the Rainforest community (full map in appendix B (3)). Blue lines represent positive influence of concept C_i on concept C_j, and orange lines represent a negative influence of concept C_i on concept C_j. Line thickness indicates the degree of influence between components.

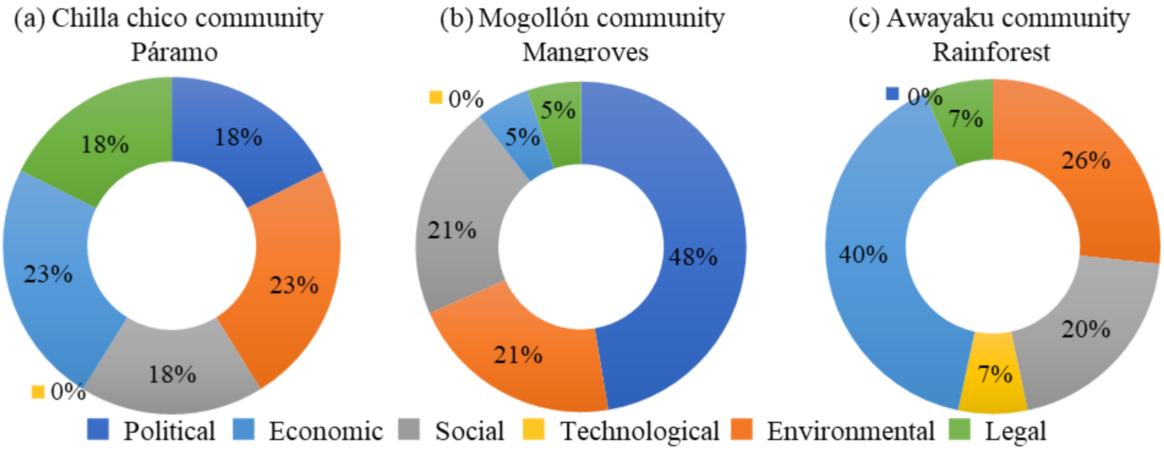


Fig. 2-6. Concepts from PESTEL perspective in the (a) páramo community (b) mangrove communities and (c) rainforest community.

Table 2-2. PESTEL concepts and policy drivers with their outdegree, indegree, centrality, steady-state value and type identified for the (a) páramo, (b) mangrove, and (c) rainforest communities. In addition to the existing 40 PESTEL concepts, further concepts for the six PESTEL factors P, EV, S, EC L and D are added for each community páramo (P), mangrove (M) and rainforst (R) by either experts (E) or communities (C). For example: EV_{PC1} is an additional concept for the environmental factor in the páramos identified by the community. D_{AFIs} is the policy driver

(a) Páramo community						
PESTEL	**Concepts**	**Outdegree**	**Indegree**	**Centrality**	**Steady-state value**	**Type**
Political	P_3: Decision-makers without adequate training	10.25	1.00	11.25	0.87	Ordinary
	P_4: Short-term environmental projects without results	9.33	2.66	11.99	0.97	Ordinary
	P_5: Few legitimate conservation policies for key ecosystems	4.76	1.99	6.75	0.92	Ordinary
Environmental	EV_1: Natural water pollutants	8.00	4.22	12.22	0.99	Ordinary
	EV_2: Anthropogenic water pollutants	5.27	4.48	9.75	0.99	Ordinary
	EV_3: Climate change	3.78	2.82	6.60	0.96	Ordinary
	EV_{PC1}: Landslides	2.21	3.62	5.83	0.98	Ordinary
Social	S_2: Population growth	3.40	1.13	4.53	0.40	Ordinary
	S_3: Lack of community cohesion	4.28	1.25	5.53	0.89	Ordinary
	S_7: Social well-being	1.33	8.49	9.82	0.00	Ordinary
Economic	EC_3: Costly treatments of water	1.33	9.02	10.35	1.00	Ordinary
	EC_5: Water demand for economic activities: crop	3.90	4.85	8.75	1.00	Ordinary
	EC_7: Economic growth	0.00	6.31	6.31	0.00	Receiver
	EC_{PE1}: Increased agriculture in páramo	6.46	1.98	8.44	0.93	Ordinary
Legal	L_3: Legal actions by users	0.00	8.35	8.35	1.00	Receiver
	L_4: Water governance crisis	3.69	7.10	10.79	1.00	Ordinary
	L_{PE1}: Laws that contradict each other	2.99	1.71	4.70	0.92	Ordinary
Policy drivers	D_{AFIs}: Artificial floating islands	12.18	0	12.18	0	Driver
	D_{PE1}: Positive political influence of community leaders	8.20	0	8.20		
	D_{PE2}: Government institutions specialized in water issues	11.38	0	11.38		
	D_{PC1}: Reinforce training in environmental education	9.07	0	9.07	0	Driver

Table 2-2 Continued

(b) Mangrove community

PESTEL	Concepts	Outdegree	Indegree	Centrality	Steady-state value	Type
Political	P_{ME1}: Absence of water pollution control by government	6.63	1.98	8.61	0.92	Ordinary
Environmental	EV_{2}: Anthropogenic water pollutants	8.63	6.00	14.63	0.66	Ordinary
	EV_{4}: Waterbody restoration	4.31	8.66	12.97	0.00	Ordinary
	EV_{5}:Conventional treatments	9.93	1.00	10.93	0.64	Ordinary
	EV_{6}: Ecosystem conservation: Mangrove	4.33	7.13	11.46	0.02	Ordinary
	EV_{7}: Human exposure to environmental pollutants	7.00	9.27	16.27	0.98	Ordinary
	EV_{ME1}: Water contamination from mining, oil, and aquaculture activities	9.26	1.66	10.92	0.67	Ordinary
	EV_{ME2}: Waste disposal problems	0.00	3.00	3.00	0.97	Receiver
	EV_{ME3}: Poor waste management by the municipality	6.77	5.20	11.97	0.94	Ordinary
	EV_{MC1}: Mangrove habitat lose	8.26	0.66	8.92	0.74	Ordinary
Social	S_{2}: Population growth	6.66	4.77	11.43	0.08	Ordinary
	S_{7}: Social well-being	0.00	11.33	11.33	0.00	Receiver
	S_{ME1}: Irregular settlement in mangrove zones	7.96	1.00	8.96	0.83	Ordinary
	S_{ME2}: Access to safe water, food and energy	4.87	8.66	13.53	0.01	Ordinary
Economic	EC_{3}: Costly treatments of water	5.32	8.04	13.36	1.00	Ordinary
	EC_{4}: Inhabitants incur medical costs	0.00	10.56	10.56	1.00	Receiver
	EC_{8}: Water quality for food production	4.93	8.66	13.59	0.01	Ordinary
	EC_{ME1}: Lack of a sewer system	7.09	4.78	11.87	0.99	Ordinary
Legal	L_{2}: Environmental law violation	6.56	6.15	12.71	0.98	Ordinary
Policy drivers	D_{AFIs}: Artificial floating islands	10.29	0	10.29	0	Driver
	D_{ME1}: Corporate environmental responsibility	8.13	0	8.13	0	Driver
	D_{ME2}: Education in promoting environmental awareness	6.91	0	6.91	0	Driver
	D_{MC1}: Payment for ecosystem services	7.74	0	7.74	0	Driver

Table 2-2 Continued

(c) Rainforest community

PESTEL	Concepts	Outdegree	Indegree	Centrality	Steady-state value	Type
Environmental	EV_2: Anthropogenic water pollutants	8.66	6.12	14.78	0.99	Ordinary
	EV_3: Climate change	0.33	2.16	2.49	0.73	Ordinary
	EV_6: Ecosystem conservation	5.26	6.77	12.03	0.02	Ordinary
	EV_7: Human exposure to environmental pollutants	5.99	6.39	12.38	0.97	Ordinary
Social	S_2: Population growth	6.13	2.31	8.44	0.40	Ordinary
	S_7: Social well-being	0.00	7.01	7.01	0.01	Receiver
	$S_{\mathbf{RC1}}$: Lack of environmental education	6.32	1.95	8.27	0.69	Ordinary
Technological	T_1: Alternative water technology	5.56	5.49	11.05	0.01	Ordinary
Economic	EC_1: Fish production	5.25	5.26	10.51	0.08	Ordinary
	EC_2: Food Security	2.39	5.02	7.41	0.14	Ordinary
	EC_3: Costly treatments of water	5.14	8.00	13.14	0.94	Ordinary
	EC_4: Inhabitants incur medical costs	2.82	2.72	5.54	0.90	Ordinary
	EC_8: Water quality for fishing production	3.75	6.27	10.02	0.02	Ordinary
	$EC_{\mathbf{RE1}}$: Low purchasing power	8.82	3.61	12.43	0.93	Ordinary
Legal	L_2: Environmental law violation	9.31	6.65	15.96	1.00	Ordinary
Policy drivers	$D_{\mathbf{AFIs}}$: Artificial floating islands	10.48	0	10.48	0	Driver
	$D_{\mathbf{RE1}}$: Better educational level of community members	7.62	0	7.62	0	Driver
	$D_{\mathbf{RE2}}$: Environmental education programs	7.61	0	7.61	0	Driver
	$D_{\mathbf{RC1}}$: Community committee to denounce non-compliance with the law to the government.	7.52	0	7.52	0	Driver

2.3.2 Identification of the main policy drivers to improve water quality

We proposed D_{AFIs} as a nature-based solution to remove contaminants and therefore improve the water quality in the three communities. Local experts suggested two additional policy drivers and the community one, this is described in Table 2-2 (a), (b), and (c). Thus, the most influential drivers according to outdegree scores were in three communities the D_{AFIs}, followed by D_{PE2}: Government institutions specialized in water issues for the páramo community (*od* = 11.38); D_{ME1}: Corporate environmental responsibility for the mangrove community (*od* = 8.13), and D_{RE1}: Better educational level of community members in the rainforest community (*od* = 7.62).

2.3.3 Analysis of drawn social cognitive maps

In all communities, the majority of identified concepts are ordinary, and only four are policy drivers. The mangrove community listed a slightly higher number of concepts in relation to the other communities, which indicates a larger amount of problems. Meanwhile, the rainforest community identified a higher number of connections that show a better interaction between concepts. A hierarchy index *h* of 0.35, 0.40, and 0.40 for the Páramo, Mangrove, and Rainforest community, respectively, depict that the developed maps are fully democratic. According to the density (*D*) ranging between 0.3 and 0.5, participants did not identify a large number of causal relationships among the concepts (Table 2-3).

Table 2-3. Analysis of social cognitive maps from graph theory.

	Páramo community	**Mangrove community**	**Rainforest community**
No. of concepts (*N*)	17	19	15
No. of connections (C)	91	123	103
No. of receiver concepts	2	3	1
No. of ordinary concepts	15	16	14
Number of policy drivers (driver concepts)	4	4	4
Hierarchy index (*h*)	0.35	0.40	0.40
Density (*D*)	0.31	0.35	0.46

2.3.4 Fuzzy inference: Comparison between steady-state and simulated worst-case scenario

To better understand how PESTEL concepts might respond to a change without policies, the most influential concepts, i.e. the concepts with high centrality value were set as high (=1) (Appendix C). At the moment that the concepts EV_1 in Páramo, EV_7 in Mangrove, and L_2 in Rainforest were set to 1, the remaining concepts were affected by 89%, 90% and 87.5%, respectively. Besides, in the communities, different levels of alteration were identified in population growth (S_2), anthropogenic water pollutants (EV_2), and social well-being (S_7). In the Páramo and Mangrove communities, S_2 shows a moderate decrease, where the zones are most populated, while for Rainforest community shows a weak decrease. In relation to EV_2 the decrease is very weak in the Páramo community because the main problem in the area is natural contamination, and a very weak increase in the Rainforest if the violation of the law persists. In the Mangrove, there is a decrease, as it is taken into account only from the perspective of contamination towards humans and not from the environmental factor. Finally, S_7 shows a weak decrease in the three communities (Fig. 2-7).

2.3.5 Fuzzy inference: Comparison between steady-state and D_{AFIs} with additional policies

The impact of the D_{AFIs} and the other drivers policy in the communities were examined in isolation in the fuzzy inference estimation of step 2 of the FCM approach (Appendix D).

The results depicted in Fig. 2-8a. show that the policies identified in páramo community as most important to reduce the natural water pollutants (EV_1) are D_{AFIs}, D_{PC1}, D_{PE2}, and D_{PE1}, in decreasing order. In addition, considering that water plays a rol in the economic income of farming families, the policies D_{AFIs} and D_{PE2} could improve the water quality as well as crops quality and contribute the economic growth (EC_7). In the Mangrove community (Fig. 2-8b), the implementation of D_{AFIs} could decrease the human exposure to environmental pollutants (EV_7), and increase the access to safe water, food and energy

(S_{ME2}). In the Rainforest community (Fig. 2-8c), the D_{AFIs} has positive impact on social well-being (S_7), and environmental law violation (L_2) can be faced with any of the 4 policies.

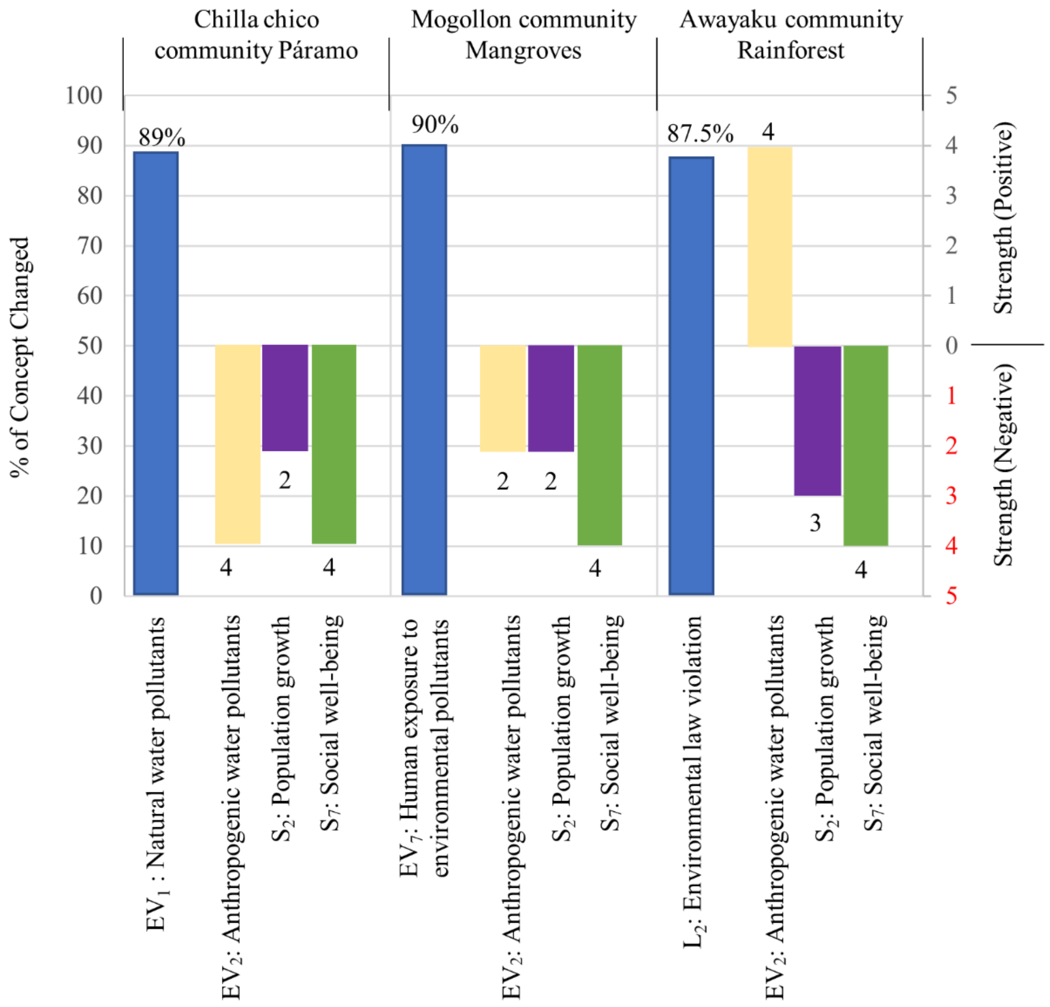
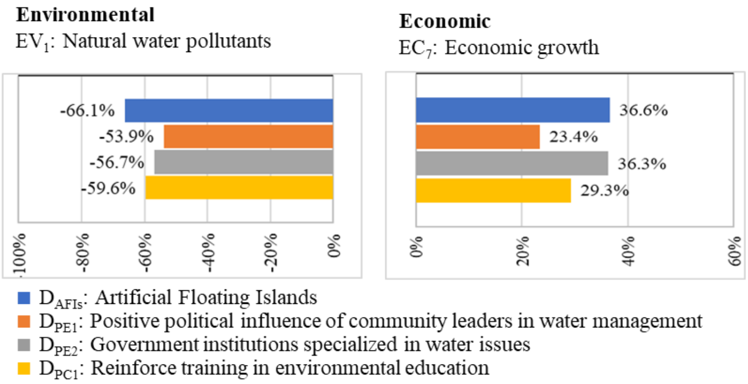
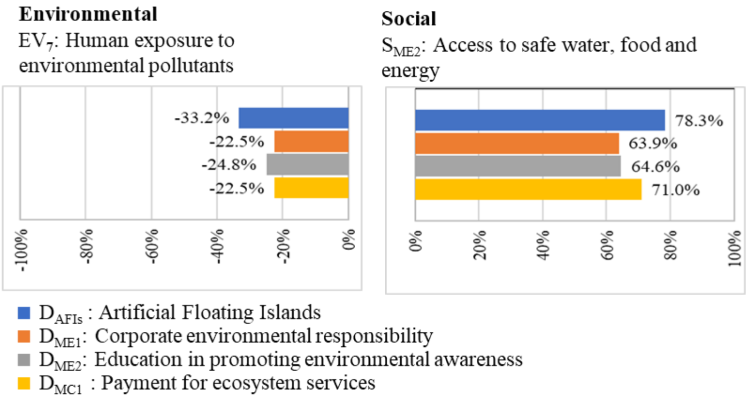


Fig. 2-7. Results of the worst-case scenario for the páramo community, mangrove communities and rainforest communities. Results are only shown for concepts with moderate to strong changes and which have been proposed for all three communities. Information on all concepts is given in appendices C and D. Numbers at bars indicate the absolute number of concepts identified.

(a) Chilla chico community Páramo



(b) Mogollón community Mangroves



(c) Awayaku community Rainforest

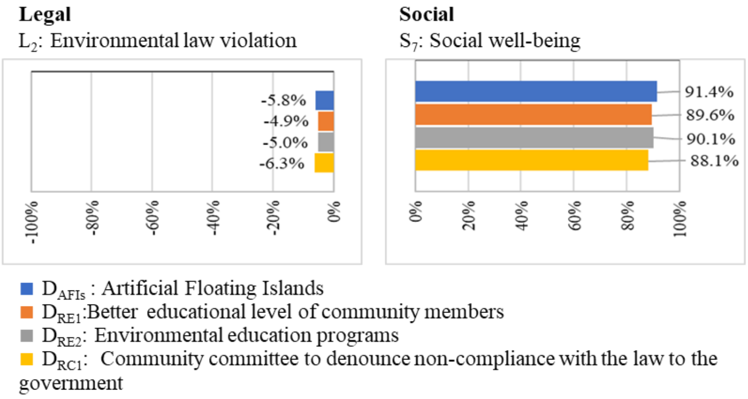


Fig. 2-8. Comparison between steady-state and change rate concepts by clamping each policy to 1 in the (a) páramo community (b) mangrove communities and (c) rainforest community.

2.4 Discussion

We co-designed together with local experts and community members from three communities in Ecuador, a decision-making framework to identify what affect water quality, and potential strategies and policies to improve its condition. Furthermore, we developed an explanatory model that analyses the current status and simulates future water quality scenarios.

Common issues in developing countries are the limited number of scientific studies that identify strategies to improve water quality. In addition, water quality data is often not available because it is either limited in public access or expensive to measure (Özesmi & Özesmi, 2004). FCMs offer an alternative approach to develop strategies to improve the water management, through a formal assessment of linguistic data (Adriaenssens et al., 2004).

To obtain reliable FCMs the participation of local experts that share experience and knowledge of the system under consideration is essential. These can include specialists from governmental organizations, non-governmental organizations (NGOs), universities, consultants, and communities (Blacketer et al., 2021; Ghaboulian Zare et al., 2022; Giordano et al., 2020; Gray et al., 2015; Morone et al., 2021; Mourhir, 2021; Santoro et al., 2019). For example, the study of Giordano et al. (2020) considers several stakeholders in the process of designing and implementing nature-based solutions as a key driver for enhancing community involvement and institutional cooperation. Regarding the number of experts to be consulted, we involved more than eight per community, satisfactorily meeting recommendations of seven to 15 participants reported elsewhere (Gómez Martín et al., 2020; Videira et al., 2014). Building democratic FCMs requires recognition of local problems and solutions to improve water quality from the perspectives of the local experts and community members. For the Páramo community, where economic and environmental concepts were identified as critical influencers, the water is a strategic, economic, and political resource, being therefore its access and management a source of conflict, and power (Pila, 2018). Many rural and indigenous communities in Cotopaxi face socio-economic problems such as high poverty rates, water scarcity, lack of access to irrigation, and unequal distribution of land and water (Partridge, 2016). These factors have determined, as in other Andean communities, to use the ecosystem as a natural resource and capital for family sustenance (United Nations Development Programme (UNDP), 2021). The páramo ecosystem itself has been used for thousands of years for different human activities, facing unprecedented anthropogenic pressures (Correa et al., 2020). The need to generate income through activities such as cultivation, intensification of livestock grazing, *Pinus* plantations, clearing activities, and tourism, may significantly alter the hydrological processes of the páramo ecosystem, affecting directly the water supply function (Buytaert et al., 2006; García et al., 2020a). Regarding the main problem recognized by locals (centrality index), the people from the Páramo community identified natural water pollutants. The presence of As with Fe is widespread from natural geological origins in multiple zones of Cotopaxi (Bundschuh et al., 2021; Morales-Simfors et al., 2019), confirming what the locals perceive. In addition, the community linked the presence of natural pollutants mainly as a cause of declining social well being (-1), similar to finding in Bundschuh et al. (2021), and increase water treatment costs (+1) as stated in Joseph et al. (2019) (Fig. 2-3).

In our findings from the Mangrove comunity, the political influence is critical on the water quality. In the same line, Beitl et al. (2019) highlight that the mangrove ecosystem in Ecuador has been undervalued and converted to other uses by ineffective policies. The results obtained in the study of Pazmiño Manrique et al. (2018) indicate that the management of marine areas, including mangrove, was not well-positioned within the Ecuadorian political agenda. Its management model is confusing and its implementation is slow and yields poor results In the Mangrove community human exposure to environmental pollutants was the key identified problem which has a major impact on the medical expenses of the inhabitants (+1), and decrease the water quality for food production (-1) (Fig. 2-4). In water, exposure to heavy metal(loid)s through ingestion of tap water and incidental ingestion of surface water, results in unacceptable risk levels (carcinogenic and non-carcinogenic) for human health in adults and children in contaminated areas on the Ecuadorian coast (Jiménez-Oyola et al., 2021). In food, concentrations of cadmium and mercury above the limits considered safe for human consumption established by the European Union have been found in yellowfin tuna caught off the Ecuadorian coast (Araújo & Cedeño-Macias, 2016). In the same line, Fernández-Cadena et al., (2014) reported the presence of several metals (Cd, Cu, Cr, and Pb), and Calle (2018) concentrations of total mercury (THg), which exceeded the permissible level by Ecuadorian legislation (Asamblea Nacional, 2014) in the Salado estuary. This, due to urban sprawl, industrial growing of Guayaquil city and inefficient water management practices (Calle et al., 2018; Vinueza et al., 2021). Some of these elements can bioaccumulate and possibly biomagnify throughout the food web, being potential risks to human health (Calle et al., 2018).

The Rainforest community perceives the economic concept as an important concept influencing water quality, which is confirmed by different studies (Cammelli et al., 2020; Garrett et al., 2017) that state that the majority of the people residing in forested regions remain impoverished due to environmental degradation and low-income. Therefore, the increasing pressure on land use over the last decades can be responsible for leaving water reservoirs unprotected, resulting in a runoff of pollutants and nutrients into such reservoirs and increasing the water treatment costs, as shown by Danelon et al. (2021) for the Brazilian case, including the Amazon forest. Finally, in the Rainforest community, the violation of environmental legislation (Buccina et al., 2013) was the most influential problem, and has a high impact on human exposure to environmental pollutants (+1) and a decrese in ecosystem conservation (-1) (Fig. 2-5). A large majority of people living in the Amazon region have no access to drinking water distribution systems and collects water from rain, wells, or small stream (Maurice et al., 2019). The lack of access to good quality water violates the inhabitant's right to live in a healthy environment according to the Ecuadorian constitution (Asamblea Constituyente, 2008). In addition, in the same Ecuadorian constitution, the nature has the right to integral respect for its existence and for the maintenance and regeneration of its life cycles, structure, functions and evolutionary processes (Asamblea Constituyente, 2008). However, Capparelli et al. (2020) found that in the zone of the Rainforest community, anthropogenic activities are introducing metals to the aquatic ecosystem, as some metals were up to 500 times above the maximum permissible limits for the preservation of aquatic life established by Ecuadorian and North American guidelines.

Our exploration to anticipate the future of water management at the community level with D_{AFIs} and complementary policy drivers (Ghaboulian Zare et al., 2022), leads us to suggest the use of two scenarios: the worst-case scenario and a managed policy scenario for the three communities. In the worst-case scenario of the three communities, more than 85% of the concepts are changing. However, these changes range from very weak to medium, and no concept experiences a strong change (Appendix C). But this is not necessarily always the case. Gray et al. (2015) use the worst-case scenario idea in the community model of the bushmeat trade in Tanzania and create a scenario in which the concept of the increased human population is characterized "as high".

Concerning the reaction of the system to the proposed policies, the implementation of AFIs as nature-based solutions leads to a decrease in natural and anthropogenic water pollutants in the Páramo community. Our findings are in line with the study of Ladislas et al. (2015) who show that AFIs can effectively remove some heavy metals, such as cadmium, nickel, and zinc from stormwater ponds. Also, the results of Wang et al. (2020) reveal the purification abilities of AFIs for carbon, nitrogen, and phosphorus. In the Mangrove community, the AFIs could increase access to safe water, food and energy, and improve the water quality for food production. In accordance with Karstens et al. (2018), the goal of AFIs in coastal habitats is the restoration and rehabilitation as well as the local enhancement of water quality by nutrient absorption and removal. Due to their generally low cost and simple construction, AFIs are used to treat wastewater from secondary effluents, stormwater, and agricultural runoff (Benvenuti et al., 2018; Colares et al., 2020). Regarding food production and energy, the plants harvested from very large AFIs can be processed into biogas, bio-fertilizer, biomaterials, or even food for animals and humans (Yeh et al., 2015). In the Rainforest community, the AFIs have a positive impact on ecosystem conservation and as an alternative to common water treatment technologies. Sharma et al. (2021) and Nuruzzaman et al. (2021) state that the AFIs are a cost-effective and eco-friendly phyto-technology which potentially benefit ecosystem quality preservation and landscape conservation. As reported by UNESCO (2019), nature-based solutions offer some of the most effective and sustainable ways to improve water security, supporting the achievement of SDG target 6.3. They also offer additional benefits to the communities in which they are applied, such as improved agriculture, job creation, climate resilience, and the achievement of several other SDGs.

In our communities, various water quality problems can be addressed with technical solution such as AFIs. However, community members outlined complementary policy drivers or combinations with AFIs which are needed to demonstrate significant changes in water quality. Strategies such as the positive political influence of community leaders (Pila, 2018), the government institutions specializing in water (Davies & Mazumder, 2003), and reinforcing training in environmental education (Suárez-Perales et al., 2021) can be applied to the problem of a few legitimate conservation policies in the Páramo community. For the environmental law violation in the Mangrove community, corporate environmental responsibility (Hambira & Kolawole, 2021), education in promoting environmental awareness (Suárez-Perales et al., 2021), and payment for ecosystem services (Fu et al., 2018) are key aspects to improve the water quality. Moreover, to achieve the social well-being in the Rainforest community, a better education level of communities members (Al Amin et al., 2021), environmental education programs (Suárez-Perales et al., 2021), and community committees to denounce non-compliance with the law to the government (Buccina et al., 2013) are presented as potentially promising concepts.

2.5 Conclusions

In our study, the FCM method proves to be a useful tool to identify concepts that influence water quality based solely on a linguistic description from local participants. Using diverse sources of knowledge from people in three communities located in key ecosystems in Ecuador, we identify issues that locals relate to poor water quality such as natural pollutants, human exposure to environmental pollutants, and violation of environmental legislation.

The application of FCMs offers a promising method to analyse and identify AFIs as nature-based solutions for water quality improvement. However, not all the problems recognized by the communities are solved by this option. Therefore, model simulations combining AFIs with different policies show better and more comprehensive options for improving water quality and ensuring future sustainable water governance. Some of the main policies suggest reinforcing training and educational programs to promote environmental awareness, demand corporate environmental responsibility, implement payment for ecosystem services, and organize community committees to denounce non-compliance with the law.

We suggest that future studies should consider three aspects of the community FCMs: the criteria that determine the necessary participation of stakeholders, the policy driver associated with the nature-based solution, and the use of understandable rating scales for the degree of influence between concepts. Related to criteria of participation, FCMs encourage stakeholders to take part in water management decisions. Nevertheless, participants should fulfill the following criteria: In the communities, the water must play a crucial role in family income generation and, in the case of experts, their knowledge, experience and influence on water decisions making. In this way, it is possible to identify the real problems and propose adequate strategies based on internal mental models of stakeholders. Regarding the policy driver, we propose AFIs as a nature-based solution taking into account the economic water scarcity of communities, although for future studies we recommend exploring theoretically other nature-based solutions that also cover the problem of physical water scarcity. This approach suggests that for scoring the degree of cause-effect influence of concepts, simple scales may adequately capture our objective, otherwise the community may perceive long scales as a complex methodology.

Our results show that the simple nature of the FCMs provides a novel, inclusive, and locally adapted framework to understand how water quality could be improved, guiding the future water management and contributing to achieving the SDGs, particularly target 6.3 in developing countries. Furthermore, as pioneering work in the region, our results have the potential to be replicated in different communities in developing countries, where access to data is limited, but the motivation and knowledge of stakeholders exist.

Appendix A

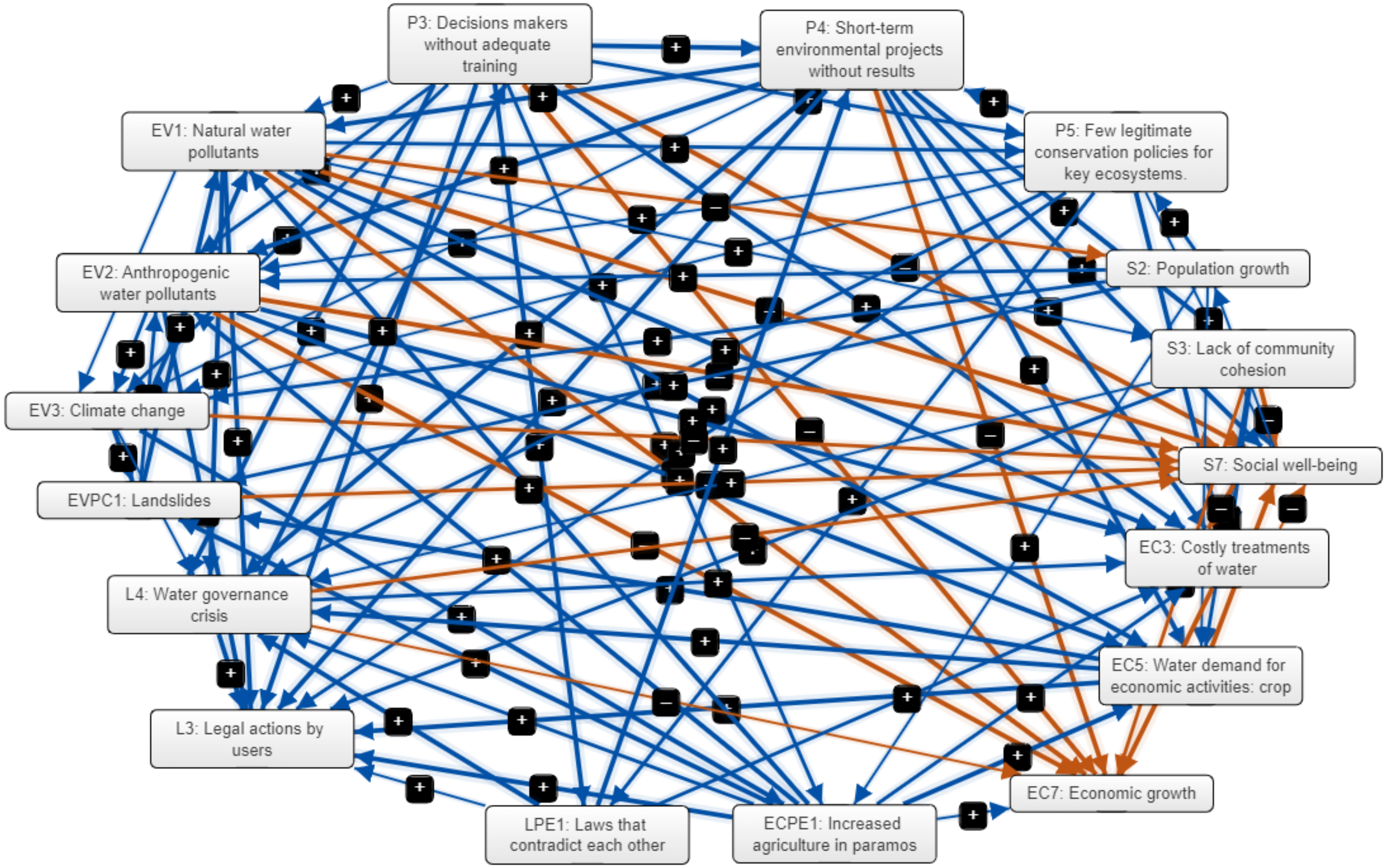
Participation of multisectoral stakeholders in water management decisions

Stakeholders	Category	Function	Expert' role	Number experts	Invited participants	Total of experts	Participants of community
Páramo	Decentralized Autonomous Government (GAD) of the province of Cotopaxi/ YakupakWasi	Irrigation water management in the province of Cotopaxi, including páramo areas	Director of the irrigation and drainage	1	2	10	10
	Cotopaxi Technical University - Environmental Engineering Department	Training of professionals in the management and protection of water, soil and air	Director of the project: nature-based solutions in the páramo of Cotopaxi	1	2		
	Independent professional	Consultant	Specialist in conventional water treatment plants and irrigation reservoirs	1	2		
	NGO "Plan Internacional"	Promote associativity, sustainable production, and fair trade	General manager in Cotopaxi	1	0		
Mangrove	Ministry of the Environment, Water and Ecological Transition of Ecuador (MAAE).	Salado Estuary and Santay Island recovery Project (PRESIS)	Manager of project PRESIS	1	2	10	10
	University of Guayaquil	Training of professionals in biotechnology, biodiversity, and conservation of natural resources	Director of biology department	1	0		
	Municipal Water and Sewerage Company of Guayaquil (EP EMAPAG)	Consultant	Specialist in conventional water treatment plants	1	2		

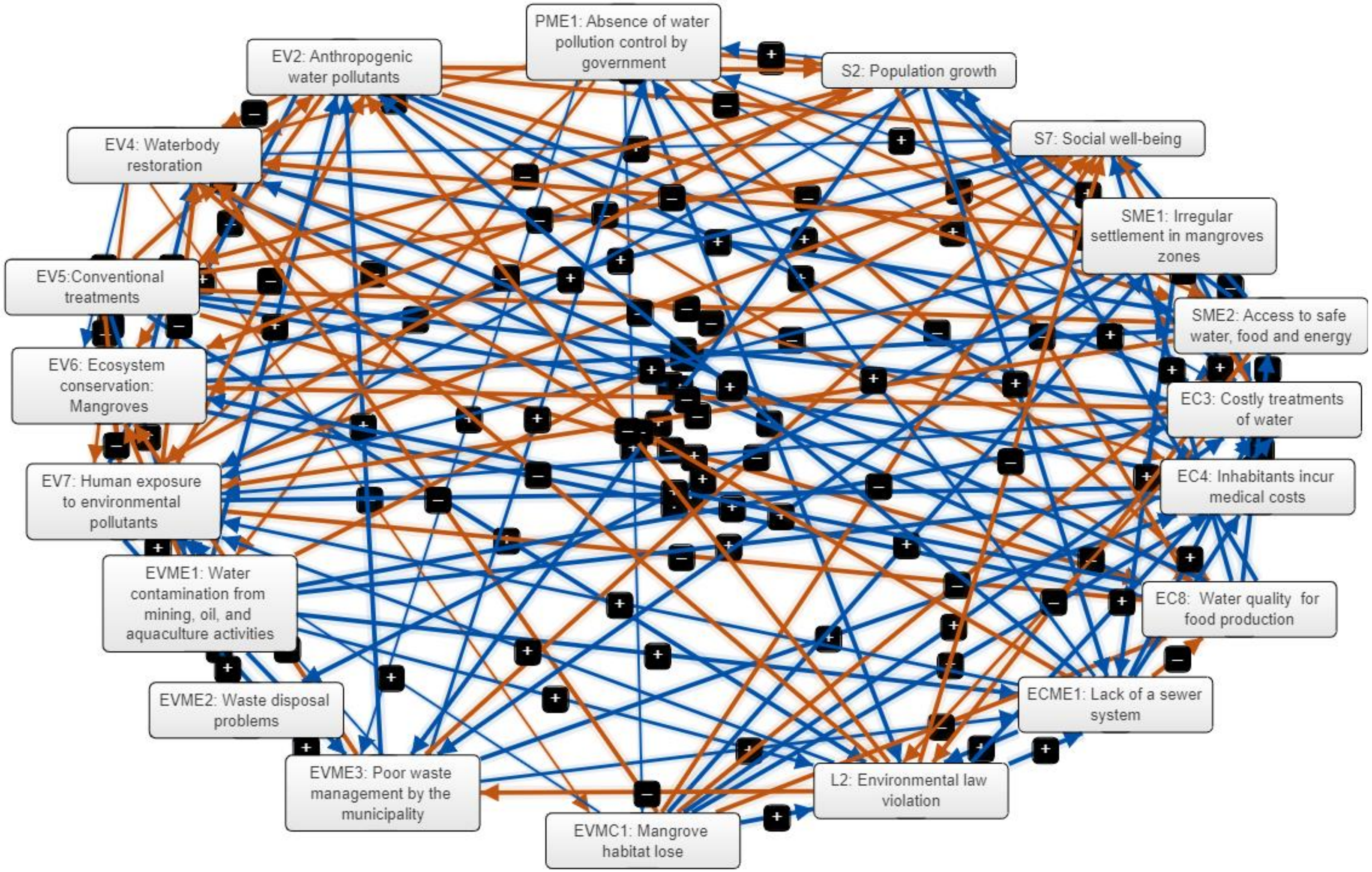
Stakeholders	**Category**	**Function**	**Expert' role**	**Number experts**	**Invited participants**	**Total of experts**	**Participants of community**
	ONG "Misión Asistencia"	Promotes the protection of people and communities against the harmful effects of natural or anthropogenic phenomena	Executive director	1	2		
Rainforest	Decentralized Autonomous Government (GAD) of the province of Archidona	Application of environmental management policies in the territory of Archidona	Coordinator of the environmental management and control department	1	1	8	8
	Amazon Regional University Ikiam- Department of Hydrology	Training of professionals in management and protection of water	Coordinator of department of hydrology	1	2		
	Independent professional	Consultant	Specialist in conventional water treatment plants	1	0		
	NGO "180 GRADOS"	Development of social projects with a gender approach	Executive director	1	1		

Appendix B

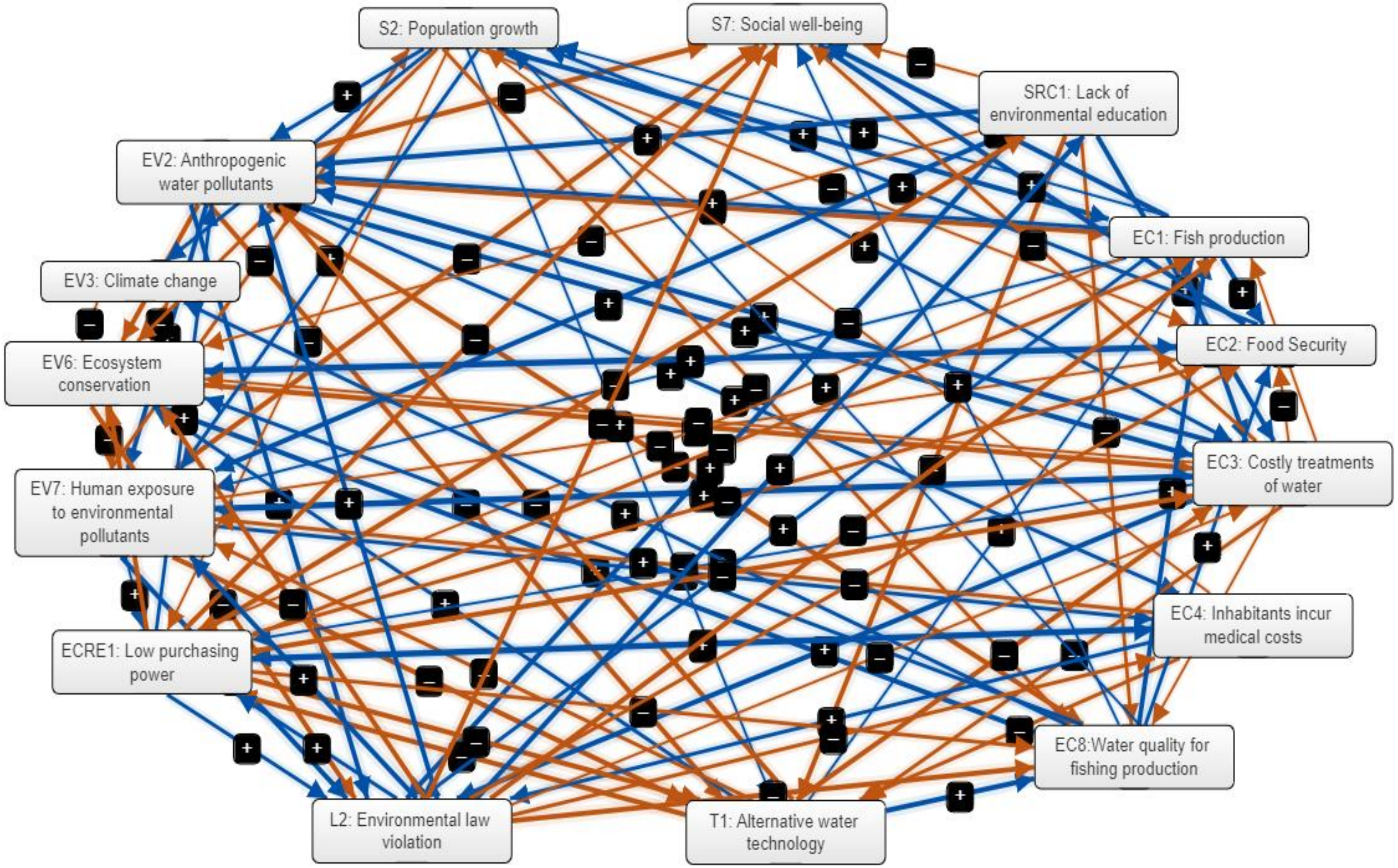
(1) Social cognitive map constructed in the workshop by the páramo community. Blue lines represent positive influence of concept C_i on concept C_j, and orange lines represent a negative influence of concept C_i on concept C_j. Line thickness indicates the degree of influence between components.



(2) Cognitive map constructed in the workshop by the mangrove community. Blue lines represent positive influence of concept C_i on concept C_j, and orange lines represent a negative influence of concept C_i on concept C_j. Line thickness indicates the degree of influence between components.



(3) Social cognitive map constructed in the workshop by the rainforest community. Blue lines represent positive influence of concept C_i on concept C_j, and orange lines represent a negative influence of concept C_i on concept C_j. Line thickness indicates the degree of influence between components.



Appendix C

(1) Results of worst-case scenario in the páramo community. Factors with an increasing trend are listed in the left column, factors with a decreasing trend in the right column. 1 = strong change (red color); 2 = medium change (blue color); 3 = weak change (purple color); 4 = very weak change (grey color).

Concept		**EV_1: Natural water pollutants**	
% of Concept Changed		89	
Concept	**Strength (Positive)**	**Concept**	**Strength (Negative)**
P_3: Decision-makers without adequate training	4	EV_2: Anthropogenic water pollutants	4
P_4: Short-term environmental projects without results	4	AC_1: Landslides	4
P_5: Few legitimate conservation policies for key ecosystems	3	$S_{2:}$ Population growth	2
EV_3: Climate change	4	S_7: Social well-being	4
S_3: Lack of community cohesion	3	EC_7: Economic growth	4
EC_3: Costly treatments of water	4	EC_{PE1}: Increased agriculture in páramo	4
EC_5: Water demand for economic activities: crop	4		
L_3: Legal actions by users	4		
L_4: Water governance crisis	4		
L_{PE1}: Laws that contradict each other	4		

(2) Results of worst-case scenario in the mangrove community. Factors with an increasing trend are listed in the left column, factors with a decreasing trend in the right column. 1 = strong change (red color); 2 = medium change (blue color); 3 = weak change (purple color); 4 = very weak change (grey color).

Concept		**EV_7: Human exposure to environmental pollutants**	
% of Concept Changed		90	
Concept	**Strength (Positive)**	**Concept**	**Strength (Negative)**
P_{ME1}: Absence of water pollution control by government	4	EV_2: Anthropogenic water pollutants	2
EV_5:Conventional treatments	3	EV_4: Waterbody restoration	4
EV_6: Ecosystem conservation: Mangrove	4	S_2: Population growth	2
EV_{ME1}: Water contamination from mining, oil, and aquaculture activities	4	S_7: Social well-being	4
EV_{ME2}: Waste disposal problems	4	EC_3: Costly treatments of water	4
EV_{ME3}: Poor waste management by the municipality	2	EC_8: Water quality for food production	3
EC_{ME1}: Lack of a sewer system	4	EC_{ME1}: Lack of a sewer system	4
S_{ME1}: Irregular settlement in mangrove zones	4		
S_{ME2}: Access to safe water, food and energy	4		
EC_4: Inhabitants incur medical costs	4		
L_2: Environmental law violation	4		

(3) Results of worst-case scenario in the rainforest community. Factors with an increasing trend are listed in the left column, factors with a decreasing trend in the right column. 1 = strong change (red color); 2 = medium change (blue color); 3 = weak change (purple color); 4 = very weak change (grey color).

Concept	**Scenario: L_2: Environmental law violation**		
% of Concept Changed	87.5		
Concept	**Strength (Positive)**	**Concept**	**Strength (Negative)**
EV_2: Anthropogenic water pollutants	4	EV_3: Climate change	4
EV_7: Human exposure to environmental pollutants	3	EV_6: Ecosystem conservation	3
S_{RC1}: Lack of environmental education	2	S_2: Population growth	3
EC_{RE1}: Low purchasing power	3	S_7: Social well-being	4
		T_1: Alternative water technology	4
		EC_1: Fish production	3
		EC_2: Food Security	3
		EC_3: Costly treatments of water	4
		EC_4: Inhabitants incur medical costs	3
		EC_8:Water quality for fishing production	3

Appendix D

(1) Comparison between steady-state and change rate concepts by clamping each policy to 1 in the páramo community

PESTEL	Concepts	D_{AFIs}: Artificial Floating Islands	D_{PE1}: Positive political influence of community leaders in water management.	D_{PE2}: Government Institutions Specialized in Water Issues	D_{P1}: Reinforce training in environmental education
Political	P_3: Decision-makers without adequate training	-53.2%	-49.9%	-53.2%	-49.9%
	P_4: Short-term environmental projects without results	-73.2%	-69.3%	-75.4%	-69.2%
	P_5: Few legitimate conservation policies for key ecosystems	-80.2%	-78.9%	-79.4%	-78.1%
Environmental	EV_1: Natural water pollutants	-66.1%	-53.9%	-56.7%	-59.6%
	EV_2: Anthropogenic water pollutants	-67.9%	-56.4%	-62.7%	-66.4%
	EV_3: Climate change	-59.5%	-49.5%	-48.0%	-55.0%
	EV_{PC1}: Landslides	-53.6%	-36.1%	-45.7%	-48.6%
Social	S_2: Population growth	2.7%	1.9%	3.3%	0.4%
	S_3: Lack of community cohesion	-21.0%	-21.0%	-28.1%	-24.6%
	S_7: Social well-being	31.0%	19.6%	25.6%	25.5%
Economic	EC_3: Costly treatments of water	-42.6%	-26.1%	-35.2%	-32.8%
	EC_5: Water demand for economic activities: crop	-51.5%	-36.5%	-44.4%	-46.0%
	EC_7: Economic growth	36.6%	23.4%	36.3%	29.3%
	EC_{PE1}: Increased agriculture in páramo	-65.5%	-68.2%	-62.7%	-69.3%

PESTEL	Concepts	D_{AFIs}: Artificial Floating Islands	D_{PE1}: Positive political influence of community leaders in water management.	D_{PE2}: Government Institutions Specialized in Water Issues	D_{P1}: Reinforce training in environmental education
Legal	L_3: Legal actions by users	-20.4%	-12.6%	-15.6%	-13.4%
	L_4: Water governance crisis	-21.1%	-16.1%	-22.3%	-17.3%
	L_{PE1}: Laws that contradict each other	-80.7%	-79.0%	-80.6%	-79.0%

(2) Comparison between steady-state and change rate concepts by clamping each policy to 1 in the mangrove community

PESTEL	Concepts	D_{AFIs}: Artificial Floating Islands	D_{EM1}: Corporate Environmental Responsibility	D_{ME2}: Education in promoting environmental awareness	D_{MC1}: Payment for ecosystem services
Political	P_{ME1}: Absence of water pollution control by government	-28.2%	-32.5%	-29.9%	-37.4%
Environmental	EV_2: Anthropogenic water pollutants	-49.4%	-51.0%	-50.0%	-46.6%
	EV_4: Waterbody restoration	50.5%	40.2%	41.3%	42.4%
	EV_5:Conventional treatments	-40.7%	-32.9%	-35.8%	-33.8%
	EV_6: Ecosystem conservation: Mangrove	39.5%	31.3%	33.2%	31.7%
	EV_7: Human exposure to environmental pollutants	-33.2%	-22.5%	-24.8%	-22.5%
	EV_{ME1}: Water contamination from mining, oil, and aquaculture activities	-57.5%	-57.6%	-56.9%	-56.2%
	EV_{ME2}: Waste disposal problems	-31.5%	-41.8%	-40.5%	-40.8%
	EV_{ME3}: Poor waste management by the municipality	-9.4%	-8.2%	-10.3%	-12.9%
	EV_{MC1}: Mangrove habitat lose	-49.2%	-48.8%	-50.0%	-52.7%
Social	S_2: Population growth	50.7%	41.0%	41.9%	44.5%
	S_7: Social well-being	58.9%	38.8%	43.9%	45.9%
	S_{ME1}: Irregular settlement in mangrove zones	-26.1%	-26.0%	-32.0%	-31.0%
	S_{ME2}: Access to safe water, food and energy	78.3%	63.9%	64.6%	71.0%
Economic	EC_3: Costly treatments of water	-5.1%	-4.5%	-4.4%	-4.4%
	EC_4: Inhabitants incur medical costs	-5.2%	-3.4%	-3.6%	-3.7%
	EC_8: Water quality for food production	55.3%	42.4%	45.9%	46.2%
	EC_{ME1}: Lack of a sewer system	-1.3%	-1.7%	-1.6%	-1.7%
Legal	L_2: Environmental law violation	-24.7%	-32.2%	-23.7%	-27.2%

(3) Comparison between steady-state and change rate concepts by clamping each policy to 1 in the rainforest community

PESTEL	Concept	D_{AFIs}: Artificial Floating Islands	D_{RE1}:Better educational level of community members	D_{RE2}: Environmental education programs	D_{RC1}: Community committee to denounce non-compliance with the law to the government
Environmental	EV_2: Anthropogenic water pollutants	-14.7%	-9.2%	-9.5%	-10.0%
	EV_3: Climate change	4.3%	3.8%	3.8%	4.1%
	EV_6: Ecosystem conservation	80.0%	71.9%	73.4%	67.8%
	EV_7: Human exposure to environmental pollutants	-44.9%	-35.5%	-39.1%	-34.8%
Social	S_2: Population growth	44.1%	42.1%	42.6%	41.8%
	S_7: Social well-being	91.4%	89.6%	90.1%	88.1%
	S_{RC1}: Lack of environmental education	-35.4%	-38.1%	-38.7%	-29.6%
Tecnological	T_1: Alternative water technology	35.5%	21.1%	21.9%	18.8%
Economic	EC_1: Fish production	81.8%	75.3%	75.7%	75.0%
	EC_2: Food Security	83.9%	83.6%	83.7%	83.3%
	EC_3: Costly treatments of water	-28.7%	-16.7%	-18.2%	-13.8%
	EC_4: Inhabitants incur medical costs	-77.0%	-76.1%	-74.9%	-74.4%
	EC_8:Water quality for fishing production	76.3%	66.0%	67.2%	63.8%
	EC_{RE1}: Low purchasing power	-85.3%	-82.7%	-82.2%	-82.4%
Legal	L_2: Environmental law violation	-5.8%	-4.9%	-5.0%	-6.3%

Chapter III: Assessing the potential of nature-based solutions as sustainable land and water management strategies in the high tropical Andean páramo ecosystem

This chapter is published as:

Fonseca, K., Acero Triana, J. S., Ramírez, M., Martínez, W., Ilbay, M., Espitia-Sarmiento, E., & Breuer, L. (2024). Assessing the potential of nature-based solutions as sustainable land and water management strategies in the high tropical Andean páramo ecosystem. *Journal of Environmental Management*, *372*, 123350. https://doi.org/10.1016/j.jenvman.2024.123350

Abstract

Nature-based solutions (NbS) are gaining attention as viable strategies for restoring water-rich ecosystems such as the páramo. However, the lack of evidence on their effectiveness, coupled with a limited understanding of their political, economic, social, technological, environmental, and legal (PESTEL) context, hinders their widespread adoption. To address this gap, we propose a multi-method framework that (i) captures PESTEL factors influencing the implementation of passive ecological restoration in páramo ecosystems as a measure of NbS and (ii) assesses its associated environmental impacts. Our approach integrates a scoping review with remote sensing techniques, water quality assessment, and correlation analyses. Focusing on the Upper Pita and Upper Cutuchi River Basins-key water sources for Andean communities in Ecuador, we reveal that the deterioration of their páramo ecosystems in the 20th century was mainly due to camelid grazing and pine timber production. In response to these threats, bottom-up management efforts starting in 1999, guided by various PESTEL factors, promoted policies, regulations, social agreements, and financial support for achieving passive ecological restoration between 2010 and 2017 in the Upper Pita River Basin. As a result, the conservation of 73.4% of natural páramo with *excellent* water quality in 2022. By contrast, top-down management in the Upper Cutuchi River Basin was

ineffective because national strategies failed to tackle the local environmental challenges posed by the PESTEL factors. Hence, only 31.6% of the natural páramo remained with poor water quality by 2022. Our study is the first to demonstrate that passive ecological restoration benefits these ecosystems, while its absence results in significant changes that require additional restoration strategies.

3.1 Introduction

The high mountain landscape of the northern Andes region in South America is dominated by a unique tropical montane tundra ecosystem known as páramo, characterized by a high water retention capacity (Buytaert et al., 2006; Castanier, 2015a; Christmann & Oliveras, 2020; Patiño et al., 2021). The páramo encompasses the high tropical Andes at elevations ranging from 3,000 to 5,000 meters above sea level (m.a.s.l.) and plays a vital role in water supply for Andean communities, including those in large cities such as Quito (Ecuador) and Bogotá (Colombia) (Buytaert et al., 2006; Mosquera et al., 2023). The main river basins supplying these local communities have headwaters in this ecosystem, which provide high-quality drinking and irrigation water for the downstream population (Ávila & Gallo, 2021; Mulligan et al., 2010).

Despite the importance of páramo ecosystems in providing water ecosystem services to Andean communities, their intensive exploitation has become widespread in the northern Andes, posing a growing threat to water security (Buytaert et al., 2006; Mulligan et al., 2010). The natural páramo vegetation (i.e., tussock-forming grasses, xerophytic shrubs, cushion-forming plants, and giant caulescent rosettes), which acts as giant sponges steadily supplying water, has been converted, often by burning into agricultural, forestry, grazing, mining, and urban systems (Castelo-Cabay et al., 2022; Giles et al., 2018). Transforming the vegetation cover and altering the hydro-physical properties of soil can significantly impact the hydrological performance of páramo ecosystems, affecting both the quality and quantity of water they provide (Patiño et al., 2021).

A crucial measure to reduce the impact of land use and land cover (LULC) changes on the water resources and to conserve the remnants of natural páramo vegetation is implementing sustainable management strategies (Buytaert et al., 2006; García et al., 2019; Mosquera et al., 2023; Rey-Romero et al., 2022). Many sustainable development agendas highlight nature-based solutions (NbS) among the available strategies because of the benefits of working with nature (Hanson et al., 2020; Possantti & Marques, 2022; Sowińska-Świerkosz & García, 2022). Cost-effectiveness, adaptation to local environmental issues, and contribution to achieving multiple Sustainable Development Goals (SDGs) are among the main advantages of NbS (Keesstra et al., 2018; Sowińska-Świerkosz & García, 2022).

In particular, in the headwaters of crucial Andean river basins, NbS initiatives, such as passive ecological restoration, are drawing the attention of the decision-makers to sustain drinking water sources (Brauman et al., 2019; Castanier, 2015). Passive ecological restoration is particularly well-suited for headwater streams, as it can restore their water quality by eliminating disturbances and favoring natural vegetation recovery (Buytaert et al., 2007; Taniwaki et al., 2019; Trujillo-Miranda et al., 2018). Nonetheless, conclusive results demonstrating that passive ecological restoration impacts on páramo ecosystems' environmental health are lacking. Similarly, the macro-environment surrounding the management of passive ecological restoration, including the political, economic, social, technological, environmental, and legal (PESTEL) context (Den Heijer & Coppens, 2023; Fonseca et al., 2022), remains largely unknown.

Recognizing the benefits of passive ecological restoration to reduce land degradation and improve water quality in the PESTEL context of páramo ecosystems requires an integrated assessment from different perspectives using multiple data sources. In this study, we propose a novel multi-method framework to i) capture PESTEL factors favoring (or constraining) the implementation of passive ecological restoration and ii) quantify its potential environmental impacts in páramo areas. It combines scoping review with remote sensing techniques, water quality assessment, and correlation analysis to address the following research questions: (1) What PESTEL factors favor (or constrain) the implementation of passive ecological restoration in the páramo ecosystems? (2) How do spatiotemporal changes in LULC and water quality differ between páramo areas with and without passive ecological restoration? (3) How does combining PESTEL analysis with environmental impact assessment of passive ecological restoration contribute to decision-making for sustainable land and water management in páramo areas? In this study, we selected the Upper Pita River (UPitaRB) and the Upper Cutuchi River Basins (UCutuchiRB) in Ecuador to illustrate the role of the páramos in the northern Andes of South America for drinking and irrigation water supply. By analyzing local passive ecological restoration efforts to maintain water ecosystem services, we provide decision-makers with strategies to improve páramo management.

3.2 Materials and Methods

3.2.1 Study basins

The UPitaRB and UCutuchiRB are located in the Andean range within Ecuador's páramo ecosystems and along the protected areas of the Cotopaxi National Park and the El Boliche National Recreation Area, respectively, shown in Fig. 3-1. The UPitaRB covers 197 km^2 and is located ~20 km from Quito between the northeast side of the Cotopaxi volcano and the southwest side of the Sincholagua volcano (Fig. 3-1a). UPitaRB's elevations range from 3,280 to 5,800 m.a.s.l. with a mean slope of 16%. The UCutuchiRB covers 233 km^2, and it is situated ~23 km from Latacunga across the inter-Andean valley (2,980-5,800 m.a.s.l.; mean slope of 14%) to the west of the Cotopaxi volcano (Fig. 3-1b). According to the Köppen-Geiger classification, the climate in the basins is warm temperate (Cfb) (Kottek et al., 2006), with temperatures ranging from 6 to 12°C (INAMHI, 2017). The mean annual precipitation in the UPitaRB varies between 1,064 and 1,406 mm, while in UCutuchiRB, between 662 and 1,406 mm (1968-2014) (Ilbay-Yupa, Lavado-Casimiro, et al., 2021). A sandy-loamy soil is predominant in both basins. Specifically, 80% of UPitaRB's soils are classified as andisols (middle and upper parts), 10% entisols, and 10% mollisols (lower part).

In the UCutuchiRB, andisols represent 40% of the basin's area, while the remaining 60% are equally covered by mollisols and inceptisols (MAG, 2019).

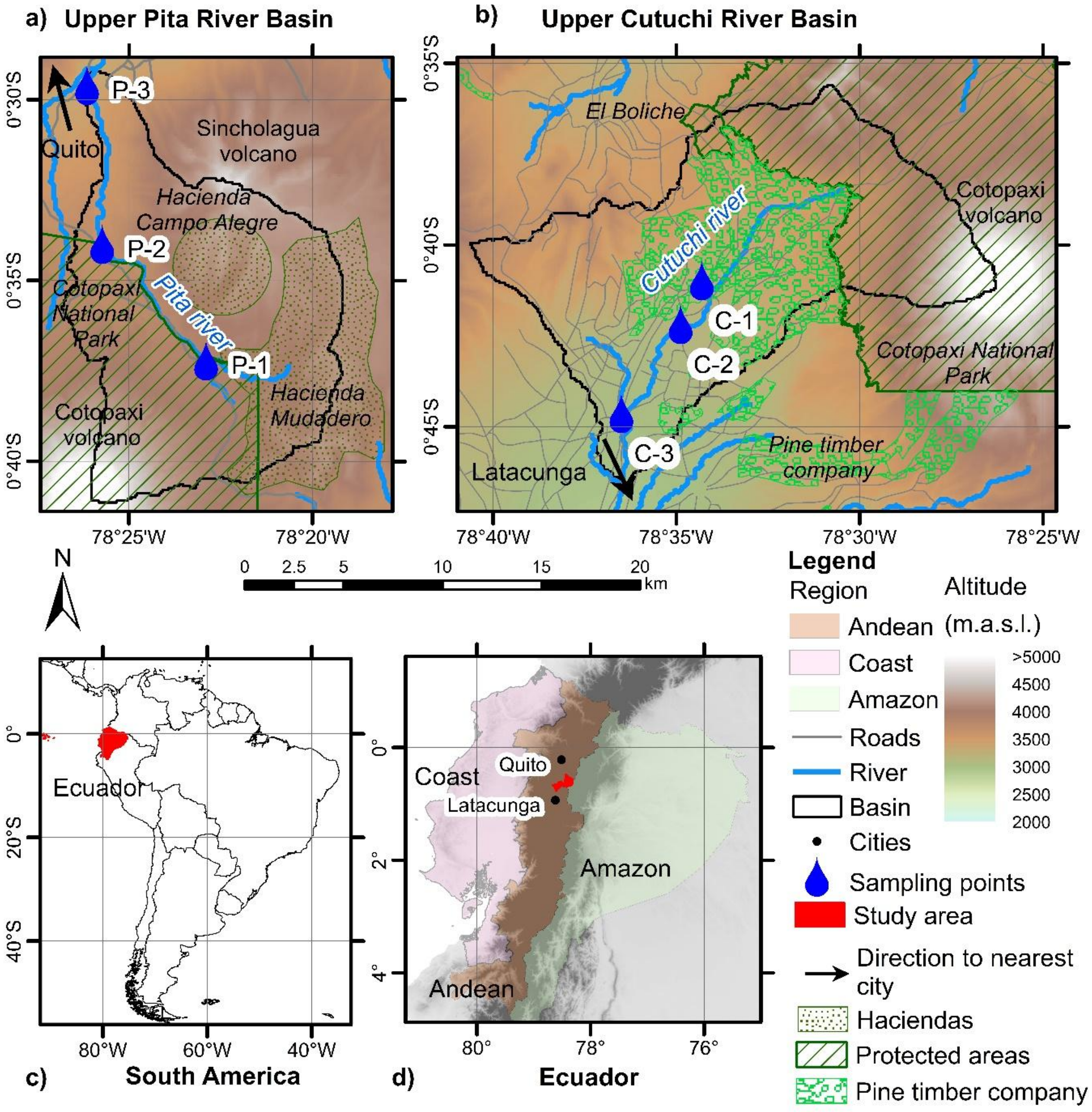


Fig. 3-1. Topography, hydrography, and protected areas of the (a) Upper Pita and (b) Upper Cutuchi River Basins, located in (c) South America and (d) Ecuador.

The Pita and Cutuchi Rivers are a crucial source of drinking and irrigation water for Quito, the country's second-largest city (~2.8 million inhabitants), and for Latacunga, a relevant agroindustrial city (Aguilera et al., 2004; Castanier, 2015; SNAP, 2015; Zapata et al., 2021). In the UPitaRB, water is allocated to irrigation (53.8%), human consumption (25%), pisciculture (21%), and ponds (0.2%). Similarly, in the UCutuchiRB, water rights are predominantly concessioned to irrigation (87.5%), industrial supply (10.8%), human consumption (1.2%), and recreational use, pisciculture, and bottled water companies (0.5%) (SENAGUA, 2021). UPitaRB and UCutuchiRB are part of the fourth (9.65%) and sixth (7.30%) Ecuadorian provinces (i.e., Pichincha and Cotopaxi), which have the largest páramo areas (mainly shrubs and herbaceous vegetation) of all provinces (García et al., 2019). Despite their importance to the national territory, the limited spatial scope of passive ecological restoration has led to uneven protection of these ecosystems, with páramo areas continuing to be transformed into other LULC, jeopardizing water security in the Andean cities (Castanier, 2015; Zapata et al., 2021).

3.2.2 Multi-method framework

The designed framework consists of (a) a PESTEL analysis to create a timeline of PESTEL factors influencing passive ecological restoration through a scoping review and b) an integration of remote sensing techniques, water quality assessment, and correlation analysis to characterize spatiotemporal changes of LULC and water quality in the study area (Fig. 3-2).

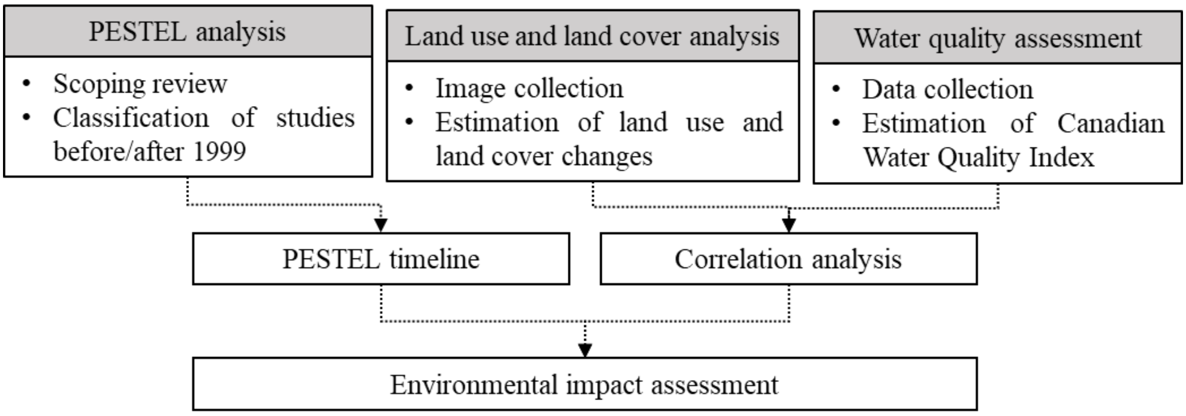


Fig. 3-2. Multi-method framework to capture the PESTEL factors involved in passive ecological restoration and quantify its potential environmental impacts.

3.2.3 PESTEL analysis

A scoping review was conducted from August 2022 to April 2023 following the methodology by Levac et al. (2010) to identify, select, and summarize relevant studies on passive ecological restoration in the páramo areas based on the analysis of PESTEL factors (Den Heijer & Coppens, 2023; Fonseca et al., 2022). This approach allows the incorporation of formal and grey literature to assess the size and scope of available literature on a specific topic and identify key trends and research gaps (O'Brien et al., 2016). We utilized widely recognized scientific databases, including ScienceDirect, Web of Science, Google Scholar, SpringerLink, and the Integrated System of Ecuadorian University Libraries. The studies on PESTEL factors were identified to address the research question: (1) What PESTEL factors favor (or constrain) the implementation of passive ecological restoration in the páramo ecosystems? The literature search consisted of keyword combination searches that associate PESTEL factors in each of the study basins with keywords such as LULC changes, NbS, basin management, sustainable land and water management strategies, sustainable land and water management initiatives, páramo transformation, passive ecological restoration, water quality, and land degradation. The selected studies met the following inclusion criteria proposed by the authors: a) peer-reviewed textbooks and book chapters, b) grey literature from non-governmental organizations (NGOs), consulting companies, and government agencies, and c) national and local laws, regulations, and policies.

Since passive ecological restoration plans began in the study area in 1999, studies conducted before this year were classified as development pathways of land and water management. While studies after 1999 were viewed as an adaptation of passive ecological restoration to páramo ecosystems, with significant implementation years being 2010, 2017, and 2022. Overall, data from 48 studies were extracted and presented in a timeline view to capture the PESTEL factors favoring and constraining the implementation of passive ecological restoration in páramo ecosystems.

3.2.4 Remote sensing assessment

The compilation of primary data was necessary to assess the spatiotemporal LULC changes across the UPitaRB and UCutuchiRB due to the lack of national high-resolution LULC datasets. Using Google Earth Engine, as suggested by Floreano and De Moraes (2021) and Gorelick et al. (2017), we performed image collection, estimated spectral indices, and conducted supervised classification using a random forest classifier. For the image collection, we combined annual medians of Level-2 Landsat 30-m multispectral imagery (TM5 for 1999, ETM+7 for 2001-2013, and OLI8 for 2013-2022) and a 30-m Digital Elevation Model (DEM) from February 2000 of the Shuttle Radar Topography Mission (SRTM) (L. Yang et al., 2011). The Level 2 Landsat imagery passed through a pre-processing step to eliminate atmospheric effects and, thus, reduce the probability of LULC misclassification in cloud-prone areas (Kuhn et al., 2019). Subsequently, these data layers were used to estimate seven spectral indices that served as predictor variables of LULC and discriminate between vegetation and developed areas: Normalized Difference Vegetation Index (NDVI) (Maxwell & Sylvester, 2012), Structure Intensive Pigment Index (SIPI) (Kobayashi et al., 2020), Photosynthetic Vigour Ratio (PVR) (Warren & Metternicht, 2005), Visible Atmospherically Resistant Index Green (VARIGreen) (Kobayashi et al., 2020), Green Atmospherically Resistant Vegetation Index (GARI) (Sonobe et al., 2018), Normalized Difference Built-up Index (NDBI), and Normalized Difference Water Index (NDWI) (Ashok et al., 2021).

Using a random forest classifier trained on labeled data from approximately 100 sampling points per land cover type, we generated LULC maps for 1999, 2010, 2017, and 2022 (key years related to passive ecological efforts). Random Forest algorithm was used to classify LULC due to the effective handling of large multi-temporal remote sensing datasets and high resistance to noise and overfitting (Acharki, 2022; Andrade et al., 2021; Hasan et al., 2023). The labeled data was identified by visual interpretation of medium-resolution Landsat and ESRI mosaics following the CORINE land cover classification (IDEAM, 2010) considering seven classes: (1) artificial surfaces, (2) forests (e.g., pine), (3) glacier and perpetual snow, (4) open spaces with little or no vegetation (including bare rocks, sparse vegetation and burned areas), (5) pastures, (6) shrub and herbaceous vegetation (categorized as natural páramo), and (7) temporary crops (e.g., potato, corn, beans). The accuracy of the LULC classification used the precision and kappa metrics, according to Foody (2020), at 30% of the sampling points for each class (Hastie et al., 2009). The LULC classification performed satisfactorily during calibration (precision = 87% and Kappa index ≥ 0.85) and validation (precision= 69% and Kappa index ≥ 0.63) according to the Random Forest algorithm. The resulting LULC maps were further refined using a majority filter (Kim, 1996) to remove isolated pixels. To identify spatiotemporal LULC changes, we used a change matrix upon four standardized categories (i.e., deforestation, pine plantation, crops, and converted) following the guidelines of IDEAM et al. (2010) and considering changes in spatial units of at least 4,000 m^2.

3.2.5 Water quality assessment

We calculated the Canadian Water Quality Index (CWQI) to compare water quality for drinking and irrigation sources between the study basins, chosen for its flexibility in the type and number of water quality parameters, the period of application, and the waterbodies (CCME, 2017; Hurley et al., 2012). The CWQI (Eq 1) is calculated as the complement of the square root of the sum of squares of the scope (F_1), frequency (F_2), and amplitude (F_3) of the water samples' exceedance from given standards (Hurley et al., 2012; Kaur et al., 2023).

$$\mathrm{CWQI} = 100 - \left(\frac{\sqrt{F_1^2 + F_2^2 + F_3^2}}{1.732}\right) \qquad \text{(Eq 1)}$$

F_1 represents the number of parameters not meeting the water quality standards, while F_2 represents the number of times the water quality standards are not met. On the other hand, F_3 represents the deviation of the non-compliant parameters from their corresponding water quality standards (Ahmed et al., 2020; Kaur et al., 2023). CWQI varies between 0 and 100, with the latter indicating the best water quality (Marselina et al., 2022).

The acceptable standards for the quality of drinking and irrigation water sources used for calculating the CWQI are derived from national regulations (MAE, 2002, 2015), except total phosphorus (for both uses), nitrate-nitrogen, and turbidity for irrigation that followed international regulations (FAO, 1994; U.S. EPA, 1988, 2012) (Table 3-1).

Table 3-1. Acceptable drinking and irrigation water quality standards

Parameter	Unit	Drinking Water	Irrigation water
Dissolved oxygen (DO)	mg/L	≥6	≥3
Fecal coliform (FC)	NMP/100mL	≤1,000	≤1,000
Nitrate-Nitrogen ($\mathrm{NO_3-N}$)	mg/L	≤10	≤10
pH	-	6-9	6-9
Total dissolved solids (TDS)	mg/L	≤500	≤2,000
Total phosphorus (TP)	mg/L	≤0.1	≤2
Turbidity (TURB)	NTU	≤100	≤2

The CWQI model requires a minimum of four water quality parameters but does not specify which ones. Additionally, the parameters used can vary by region, depending on local conditions, water use purposes, and quality issues (CCME, 2017; Uddin et al., 2021). Our water quality database incorporated seven physical, chemical, and microbiological parameters (Table 3-1) based on data availability in the study area, their relevance to drinking and irrigation water sources, and frequent parameters incorporated in water quality indices, as recommended by Uddin et al. (2021). These parameters include dissolved oxygen (DO), fecal coliform (FC), nitrate-nitrogen ($\mathrm{NO_3-N}$), pH, total dissolved solids (TDS), total phosphorus (TP), and turbidity (TURB) (Table 3-1). The water quality database includes data collected between 3,000 and 5,000 m.a.s.l. in the high tropical Andes from studies funded by governmental agencies, universities, and NGOs, was considered for the water quality database (Acosta, 2018; Amores, 2019; CODERECO, 2002; Quishpe, 2018; Rojas, 2020). In addition, a fieldwork campaign took place in 2022, in which the collection and preservation of additional samples complied with Ecuadorian technical standards for drinking water (INEN, 2013a, 2013b), Unified Text of the Secondary Legislation of the Ecuadorian Ministry of Environment (MAE, 2015), and section 1060 of the Standard Methods for the Examination of Water and Wastewater (APHA, AWWA, WEF, 2005, 2017).

The water quality database for UPitaRB and UCutuchiRB included 441 and 320 water samples, respectively, representing seven selected quality parameters at three sampling points in 2001, 2010, 2017, and 2022 (Fig. 3-1a and b). In UCutuchiRB, water quality data is scarcer than in UPitaRB, with only one data point available from 2001. This year was the starting point for analyzing passive ecological restoration initiatives in both basins from a water quality perspective.

3.2.6 Correlation analysis

Pearson's correlation coefficient was applied to linearly associate spatiotemporal variations of LULC with water quality parameters (Li et al., 2008) and to evaluate the possible impacts of LULC changes on water quality in the UPitaRB and UCutuchiRB. Correlation analyses (Eq 2) were conducted using average values of each water quality parameter (441 and 320 water samples for UPitaRB and UCutuchiRB, respectively) for the period 2001 to 2022, and each LULC class between 1999 and 2022 as follows.

$$\mathrm{PCC} = \frac{\sum(\mathrm{LULC_i} - \overline{\mathrm{LULC}})(\mathrm{WQP_i} - \overline{\mathrm{WQP}})}{\sigma_{\mathrm{LULC}}\ \sigma_{\mathrm{WQP}}} \qquad \text{(Eq 2)}$$

With Pearson's correlation coefficient (PCC), $\mathrm{LULC_i}$ and $\mathrm{WQP_i}$ the *i*th observation of LULC and water quality parameters, respectively. $\overline{\mathrm{LULC}}$ and $\overline{\mathrm{WQP}}$ are the mean observations, and σ represents the standard deviation (Herzog et al., 2019; Prion & Haerling, 2014). Correlations were considered statistically significant with $\mathrm{PCC} \geq 0.8$ and $p < 0.05$. The statistical analysis was performed using the R programming language.

3.3 Results

3.3.1 Development pathways of land and water management in the 20th century

Exploration of PESTEL factors over the period 1900-1999 revealed their influence on both land use changes in the páramo areas of UPitaRB and UCutuchiRB and the introduction of sustainable land and water management strategies, including passive ecological restoration as shown in Table 3-2. Predominantly political, economic, legal, and social factors led to the conversion of substantial páramo areas into large farms (estates known as haciendas) and later into a combination of large and small farms due to the 1964 agrarian reform in Ecuador (Coral et al., 2021; Goodwin, 2017; Solo De Zaldívar, 2015). Although a more equitable land distribution was expected with the agrarian reform, disparities between the landowners and peasant communities prevailed due to the landowners' strategies to sidestep the reform by dividing and transferring their lands to fictitious buyers (i.e., frontmen, usually relatives) (Coral et al., 2021; Solo De Zaldívar, 2015).

Table 3-2. Relevant historical events associated with PESTEL factors affecting páramo areas over the 20th century. P: Political, Ec: Economic, S: Social, T: Technological, En: Environmental, L: Legal; blue = UPitaRB, grey = UCutuchiRB; white = both basins.

Year	PESTEL factor	Relevant historical events
1900	P	Large farms known as haciendas have been expanding in the páramo since colonial times
	S	Hacienda peasants work in precarious conditions
1922	T	Quito's hydropower plant in development
1928	T	California pine trees are introduced in high-altitude ecosystems
1964	L	Agrarian reform is enacted to redistribute land among peasant communities
	S	Landowners divide lands between relatives to sidestep the agrarian reform
1970	Ec	Peasants cannot afford the increasing land prices after the agrarian reform
	P	Area disparities in land redistribution
	Ec	Haciendas encompass páramo areas despite the agrarian reform
	Ec	Some new landowners intensify agriculture and livestock farming
	Ec	Peasant landowners cannot afford large-scale farming infrastructure
	T	Road network upgrading project begins
	S	Andean urbanization peaks, demanding more water from páramo
	T	Latacunga's utility infrastructure is modernized, excluding the basin
	T	Modernize and intensify agriculture and livestock farming
1973	L	Forestry haciendas (pine timber company) excluded from protected areas
1975	En	Cotopaxi National Park (334 km^2) is founded as a protected area
1977	T	Quito's water infrastructure in development
1978	T	A large pine tree lumber company begins activities on 16 km^2
1979	En	Boliche Recreation Area (3.9 km^2) is founded as a protected area
1983	T	Some gravel and rock deposits are unexpectedly discovered
1985	En	Alpaca population increases in the highlands
	Ec	Development of the wool industry
1986	T	Salcedo-Ambato canal is being constructed to irrigate 70 km^2 in Latacunga
	En	Water impairment by fecal coliform is reported
1990	P	Management strategies of the Ecuadorian Institute of Water Resources fail due to water disputes among stakeholders
1997	L	National Ecuadorian Institute of Water Resources is dissolved, and some functions are replaced by the National Water Resources Council creation
1998	L	The new constitution considers the protection of natural resources
1999	L	The newly created Environment Ministry approves the environmental management law
	Ec	Environmental investment is significantly constrained by a deep economic crisis
	En	Páramo areas are threatened by degradation agricultural expansion, livestock, and tourism
	En	Water impairment by pathogenic bacteria and total suspended solids is reported

On the other hand, economic, social, and technological factors drove the introduction in the haciendas of non-native species, such as California pine trees in UCutuchiRB (1928; Table 3-2) and alpacas in UPitaRB (1985; Table 3-2), to develop the lumber and wool industry, respectively (Buytaert et al., 2007; MAE, 2007; Metcalf et al., 2014). Thus, the Mudadero and Campo Alegre haciendas covered UPitaRB, while forestry haciendas (pine timber company) occupied UCutuchiRB (Fig. 3-1a and b) (Coronel, 2019; MAE, 2007). In addition, peasant communities established small farms in these páramo areas, lacking the resources for the large-scale infrastructure (Keese, 1998; Partridge, 2016). Simultaneously, water and electricity infrastructures for Quito were developed across UPitaRB between the 1920s and 1970s (Carrion et al., 1997; González-Zeas et al., 2022) (Table 3-2). UCutuchiRB, meanwhile, focused on the development of irrigation and transport infrastructure (1970-1986) for the nascent timber industry in Latacunga, which began in 1978 on 16 km^2 of páramo land and the mining industry that took off after the discovery of registered gravel and stone deposits as early as 1983 (Table 3-2) (Aglomerados Cotopaxi, 2021; Allou et al., 1987; Martínez, 2006). At the same time, drinking water and electricity infrastructures for Latacunga were developed in other páramo areas in the southwest of the city (Ibarra, 2018).

During 1960-1980, the urbanization of Andean cities reached its peak, as well as the demand for food and irrigation water, which led to the intensification and modernization of agriculture and livestock farming in both basins (Alvarez & Sanchez, 2018; Cornejo & Wilkie, 2010; Rudel & Richards, 1990; Solo De Zaldívar, 2015). To counteract these anthropogenic pressures in the páramo areas associated with economic, social, and technological development and to promote tourism, the creation of protected areas was proposed as one of the early sustainable management strategies. With the prior agreement to exclude UCutuchiRB forestry haciendas from the protected areas, Cotopaxi National Park and the El Boliche National Recreation Area were founded in 1975 and 1979, respectively, to achieve ecological restoration objectives at the national level guided by a top-down management approach (from the National government) (MAE, 2007; SNAP, 2015).

Moreover, the National Ecuadorian Institute of Water Resources (INERHI) strategies failed to resolve water user conflicts and disputes within public institutions. Consequently, INERHI was dissolved, irrigation development competencies were decentralized to the Regional Development Corporations in 1994, and water resources management was transferred to the National Water Resources Council (CNRH) (Hoogesteger et al., 2016; Warner et al., 2014).

Despite the establishment of protected areas for sustainable land and water management in UPitaRB and UCutuchiRB, the continued degradation of páramo areas and their water quality (Allou et al., 1987; Brauman et al., 2019) prompted a reconsideration of new strategies for the páramo. Thus, several legislative initiatives were launched at the national level to safeguard ecosystems and promote the sustainable use of land and water resources (Asamblea Nacional Constituyente, 1998; Congreso Nacional, 1999; Warner et al., 2014). These initiatives could have been more effective if the environmental investment had not been significantly constrained by a deep economic crisis that led to the dollarization of the country's economy in 1999 (Joslin & Jepson, 2018) (Table 3-2).

3.3.2 Conditions favoring passive ecological restoration in Upper Pita River Basin in the 21st century

- **PESTEL factor analysis in UPitaRB**

Since 1999, passive ecological restoration of páramo areas in the UPitaRB has been promoted by several PESTEL factors to protect Quito's drinking water resources (Kauffman, 2014; Wiegant, 2022) (Table 3-3). The leading promoters at the political and economic levels were Quito's water and electricity utility companies and the Quito Water Fund, which had legal support at the local governance scale (bottom-up management) (Coronel, 2019; Kang et al., 2023).

In particular, the Water Fund created as an endowment fund in 2000, has been supported by donations from public, private, and non-profit organizations, including The Nature Conservancy, to invest in restoration projects in basins supplying water to Quito

(Brauman et al., 2019; Coronel, 2019). However, the implementation of sustainable strategies did not occur until 2010, not meeting the expectations of the Water Utility Company, the largest donor of the Water Fund, which aimed to increase safe drinking water access by that time (Table 3-3) (Joslin & Jepson, 2018; Molina-Vera et al., 2018).

Table 3-3. Historical events associated with passive ecological restoration in páramo areas across UPitaRB in the 21st century. P: Political, Ec: Economic, S: Social, T: Technological, En: Environmental, L: Legal.

Year	PESTEL factor	UPitaRB
1999	P	Policymakers propose a water fund financed by the public, private sectors, and NGOs
	L	Public institutions can donate to independent water funds
2000	P	Water Fund is created by a municipal ordinance
	Ec	Water Utility Company and The Nature Conservancy become the first Water Fund donors
	P	Water Fund promotes management strategies in Quito's water-supplying basins
2001	T	Water Utility Company aims to increase high-quality water coverage by 2010
2002	Ec	Water Fund can only invest using its profits, which are limited
2003	Ec	Several beverage companies utilizing water from the páramo of the UPitaRB contribute to the Water Fund
2005 – 2009	P	Water Fund management strategies do not fulfill Water Utility Company expectations
2010	P	Water Utility Company purchases 74 km^2 of the UPitaRB without financial support from the Water Fund
2011 – 2013	S	Landowners agree to relocate 10,000 alpacas
	Ec	Water Fund is allowed to invest up to 30% of its annual income in contributions
	En	Alpaca grazing is reduced by 37% of the basin
	P	Water Fund restoration priorities deep the differences with Water Utility Company
2017 – 2020	P	Water Fund purchases 27 km^2 of the UPitaRB with Electricity Utility Company support
	En	Alpaca grazing is reduced by 13% of the basin
	S	Landowners agree to relocate 900 alpacas
	Ec	Water Utility Company reduces operative costs by improving water quality upstream
2022	En	50% of the upper basin undergoes passive ecological restoration

In 2010, passive ecological restoration efforts began in the páramo areas of UPitaRB, focusing on reducing alpaca grazing. The Water Utility Company started with the Mudadero hacienda acquisition (73.89 km^2), which occupied 37% of the páramo area. Then, agreements were reached with landowners to reduce the camelid population by approximately 10,000 alpacas and other anthropogenic pressures (Table 3-3) (Castanier, 2015; FONAG, 2019).

Since 2011, the Water Fund has been allowed to allocate up to 30% of its annual income to contributions (Kang et al., 2023), yet passive ecological restoration projects did not expand until 2017 (Table 3-3). Nonetheless, this year, political, economic, and social factors enabled the Water Fund to purchase 27.2 km^2 of the Campo Alegre hacienda with the Electricity Utility Company support, turning an additional 13% of UPitaRB's páramo into passive ecological restoration after the relocation of 900 alpacas (Coronel, 2019; FONAG, 2019). In 2022, 50% of the 196 km^2 of páramo in the UPitaRB were subject to passive ecological restoration by eliminating disturbance factors and allowing for páramo natural vegetation recovery. The passive ecological restoration of páramo in UPitaRB by bottom-up management has effectively mitigated widespread land and water degradation, influenced by a complex interplay of PESTEL factors since the 20th century.

- **LULC changes in UPitaRB**

The LULC maps and changes for 1999, 2010, 2017, and 2022, corresponding to the key years associated with passive ecological restoration of páramo areas in the UPitaRB, are shown in Fig. 3-3. In the UPitaRB, the main LULC class in 1999 identified was natural páramo, covering 79.3% (Fig. 3-3e), despite a few patches of temporary crops, forests, and pastures due to anthropogenic activities (Fig. 3-3a). In addition, open areas with little or no vegetation were observed, including bare rocks, sparse vegetation and burned areas associated with natural and human factors, respectively (Fig. 3-3a). Between 1999 and 2010, significant changes occurred in the LULC of the basin. The natural páramo decreased to 72.7% and increased open spaces with little or no vegetation to 7.1%, temporary crops to 5.9%, forests to 3.0%, and pastures to 0.5% (Fig. 3-3b and e). Natural páramo areas in Cotopaxi National Park, haciendas, and the lower part of the basin, as depicted in Fig 3-3b and e, were primarily transformed into these four specified LULC.

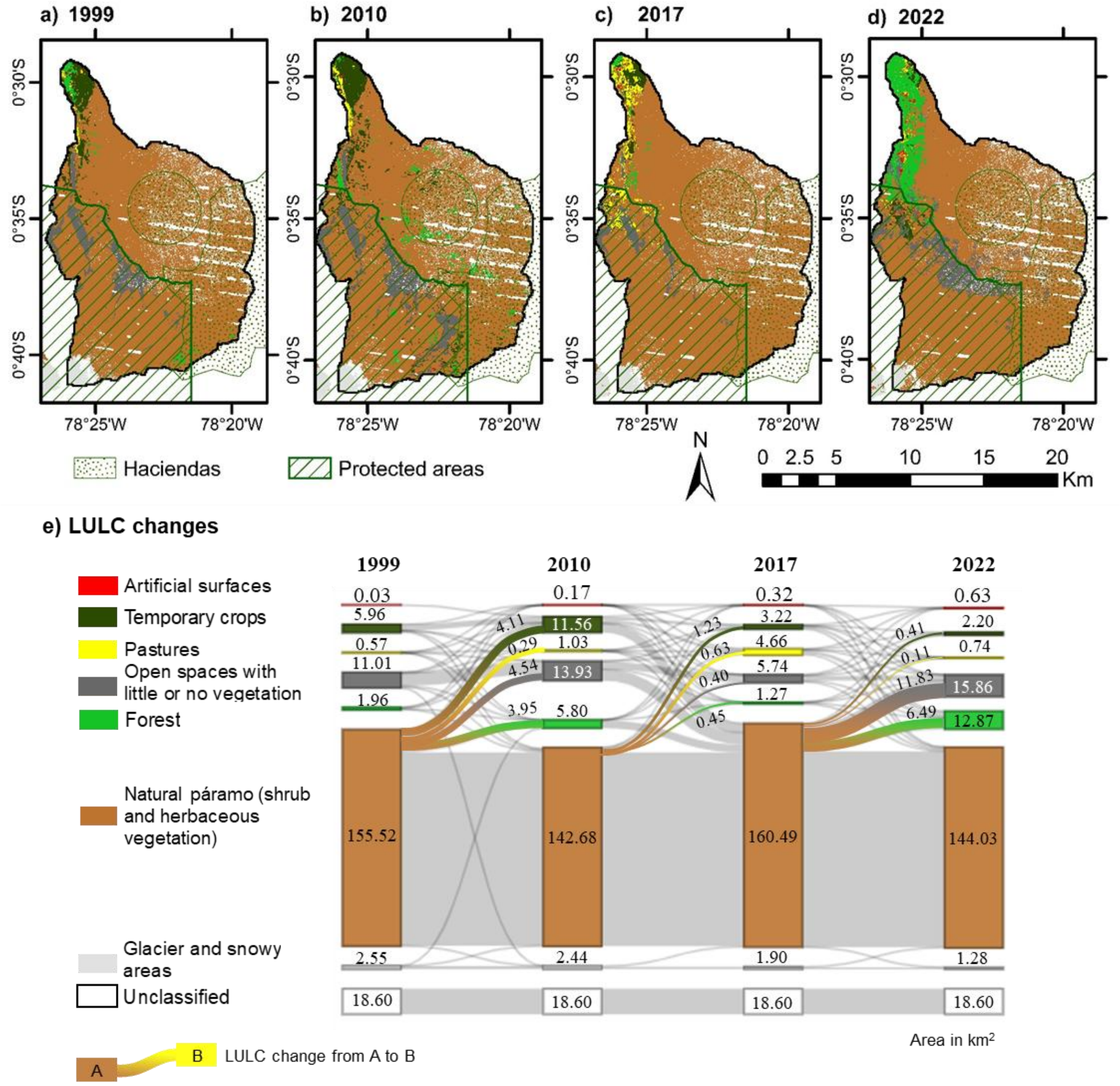
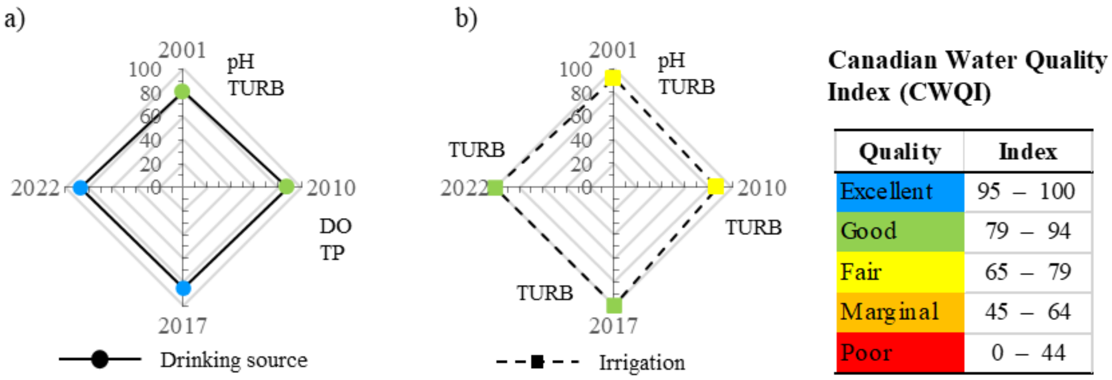


Fig. 3-3. LULC changes in UPitaRB between 1999 and 2022. Unclassified = Areas of persistent cloud cover located near snow-capped mountains.

LULC changes from 2010 to 2017 showed an increase to 81.8% of natural páramo and a decrease to 3.0 % of open spaces with little or no vegetation, 1.6% of temporary crops, and 0.7% of forests (Fig 3-3b, c, and e). Comparing natural páramo between 2010 and 2017, it increased by 9.1%, mainly in the areas under passive ecological restoration (establishment of haciendas) (Fig 3-3c). Moreover, pastures increased by 1.8% in the lower part of the basin, which did not evidence this strategy (Fig. 3-3c). In the last analysis period (2017-2022), the natural páramo remained stable at 73.4% by 2022; however, open spaces with little or no vegetation increased to 8.1% and forest to 6.6 % in the areas outside this strategy (Fig. 3-3d and e).

- **Water quality changes in UPitaRB**

The spatial and temporal variations of physical, chemical, and microbiological water quality parameters and the CWQI for 2001, 2010, 2017, and 2022 are detailed in Fig.3-4. Between 2001 and 2010, the lowest water quality in UPitaRB was observed at the low and medium sampling points (Fig 3-1a) due to elevated values of pH, TURB, TP, and low DO exceeded the water quality standards specified in Table 3-1. The CWQI categorizes these ranges as *good* for drinking water sources and *fair* for irrigation (Fig 3-4).



Canadian Water Quality Index (CWQI)

Quality	Index
🟦 Excellent	95 – 100
🟩 Good	79 – 94
🟨 Fair	65 – 79
🟧 Marginal	45 – 64
🟥 Poor	0 – 44

Fig. 3-4. The lowest CWQI values in 2001, 2010, 2017, and 2022 and associated water quality parameters (see Table 3-1 for acronyms) that do not comply with water quality standards for a) drinking water and b) irrigation water in UPitaRB.

In 2017-2022, with 50% under passive ecological restoration, the CWQI for drinking water indicated *excellent* quality for both years in all sampling points. Nonetheless, high levels of turbidity, caused the irrigation water to fall short of the highest water quality standards in all sampling points (Fig 3-4).

- **Impacts of LULC changes on water quality in UpitaRB**

For our study period (1999-2022), the correlation analyses by applying the average values of each water quality parameter and each individual LULC class in Table 3-4 indicates that temporary crops could be the main land use class to predict high levels of fecal coliform (FC) (PCC = 0.97; $p<0.05$), total phosphorus (TP) (PCC = 0.96; $p<0.05$), turbidity (TURB) (PCC=0.98; $p<0.05$) and low level of dissolved oxygen (DO) (PCC = -0.99; $p<0.05$) in UPitaRB.

Table 3-4. Correlation analysis using Pearson's correlation coefficient (PCC) between water quality parameters and percent changes in LULC in UPitaRB; significant levels in grey with *$p \leq 0.05$, **$p \leq 0.01$. For abbreviations of water quality parameters see Table 3-1.

LULC	Water quality parameter						
	DO	**FC**	**NO_3-N**	**pH**	**TDS**	**TP**	**TURB**
Artificial surfaces	0.68	-0.69	-0.31	-0.91	-0.067	-0.77	-0.58
Forests	0.17	-0.40	-0.15	-0.50	0.24	-0.25	-0.07
Glacier and snowy areas	-0.82	0.80	0.45	0.98	0.26	0.89	0.75
Open spaces with little or no vegetation	-0.28	-0.03	0.02	0.01	0.39	0.25	0.36
Pastures	0.44	-0.09	0.10	-0.35	-0.17	-0.48	-0.46
Natural páramo (Shrub and/or herbaceous vegetation)	0.42	-0.21	-0.39	-0.05	-0.72	-0.32	-0.52
Temporary crops	-0.99*	0.97*	0.85	0.90	0.79	0.96*	0.98*

Moreover, there is a lack of strong correlation between the natural páramo class and impacts on water quality parameters, as shown in Table 3-4.

3.3.3. Conditions constraining passive ecological restoration in Upper Cutuchi River Basin in the 21st century

- **PESTEL factor analysis in UCutuchiRB**

Between 2001 and 2022, the PESTEL analysis revealed that the top-down approach to sustainable land and water management in UCutuchiRB did not succeed in the ecological restoration of páramo areas (Table 3-5). Instead, there was a páramo loss despite the protected areas established, such as the Cotopaxi National Park and the El Boliche National Recreation Area. At the political level, the Cotopaxi Regional Development Corporation launched an integrated basin management plan in 2001 that combined passive ecological restoration of páramo areas and sustainable farming with conventional methods to improve the water quality of the Cutuchi River (CODERECO, 2002; Zapata et al., 2021). This initiative had to be managed by the National Water Resources Council (CNRH) due to its role in enforcing legal standards for water use (Hoogesteger et al., 2016). Although the CNRH was dissolved and replaced by a single governmental water authority in 2007 (Warner et al., 2014), economic resources have not yet been allocated, and environmental improvements have yet to be observed (Zapata et al., 2021). Then, in 2014, another strategy was proposed, according to Castillo (2022), a creation of the Cutuchi sub-river basin council to facilitate the formulation and implementation of water management plans and water fund mechanisms by the Organizational Code of Territorial Organization (COOTAD) and the Ecuadorian water law (Asamblea Nacional, 2010, 2014a). Nevertheless, a legal loophole impeded the council from being recognized, restricting its access to state funds (Asamblea Nacional, 2014a).

Table 3-5. Historical events associated with passive ecological restoration initiatives in páramo areas across UCutuchiRB in the 21st century. P: Political, Ec: Economic, S: Social, T: Technological, En: Environmental, L: Legal.

Year	PESTEL factor	UCutuchiRB
2001	P	Regional Development Corporation proposes an integrated basin management plan managed by the National Water Resources Council
2002	Ec	No evidence of an executed budget for any basin management plan
2004	T	The largest pine tree lumber company owns 33% (77 km^2) of the basin area
2014	L	A subbasin council is created by Ecuadorian law
	P	Subbasin council creation promotes management plans and water fund mechanisms
2015	L	Non-approval of the state budget to the subbasin Council as the government prioritizes larger demarcation
2021	Ec	Ecuador's largest pine timber company owns ~40% (95 km^2) of the basin
	En	Alarming levels of contamination of the Cutuchi River and lack of management strategies
2022	P	Subbasin council lacks recognition due to a legal loophole in Ecuadorian law

In 2022, as a consequence of poor land and water management, 233 km^2 of páramo in the UCutuchiRB continues to be used for pine timber production (40% of páramo areas by 2021) (Aglomerados Cotopaxi, 2021), stone and gravel mining (La Hora, 2018), agricultural activities (Solo De Zaldívar, 2015), and high levels of water pollution (Zapata et al., 2021).

- **LULC changes in CutuchiRB**

From 1999 to 2022, natural páramo coverage of UCutuchiRB was substantially lower than in UPitaRB. It started at 38.9% in 1999, then decreased to 31.6% in 2022 (Fig. 3-5e). During this period, there has been a total decrease of 7.3% (17.0 km^2) of natural páramo and an increase of 12.4% (28.8 km^2) of non-native pine forest (Fig. 3-5a, d, and e). Thus, the pine forests covered 19.2% of the basin in 1999 and reached 31.6% in 2022, covering a similar area as the páramo (Fig. 3-5a, d, and e).

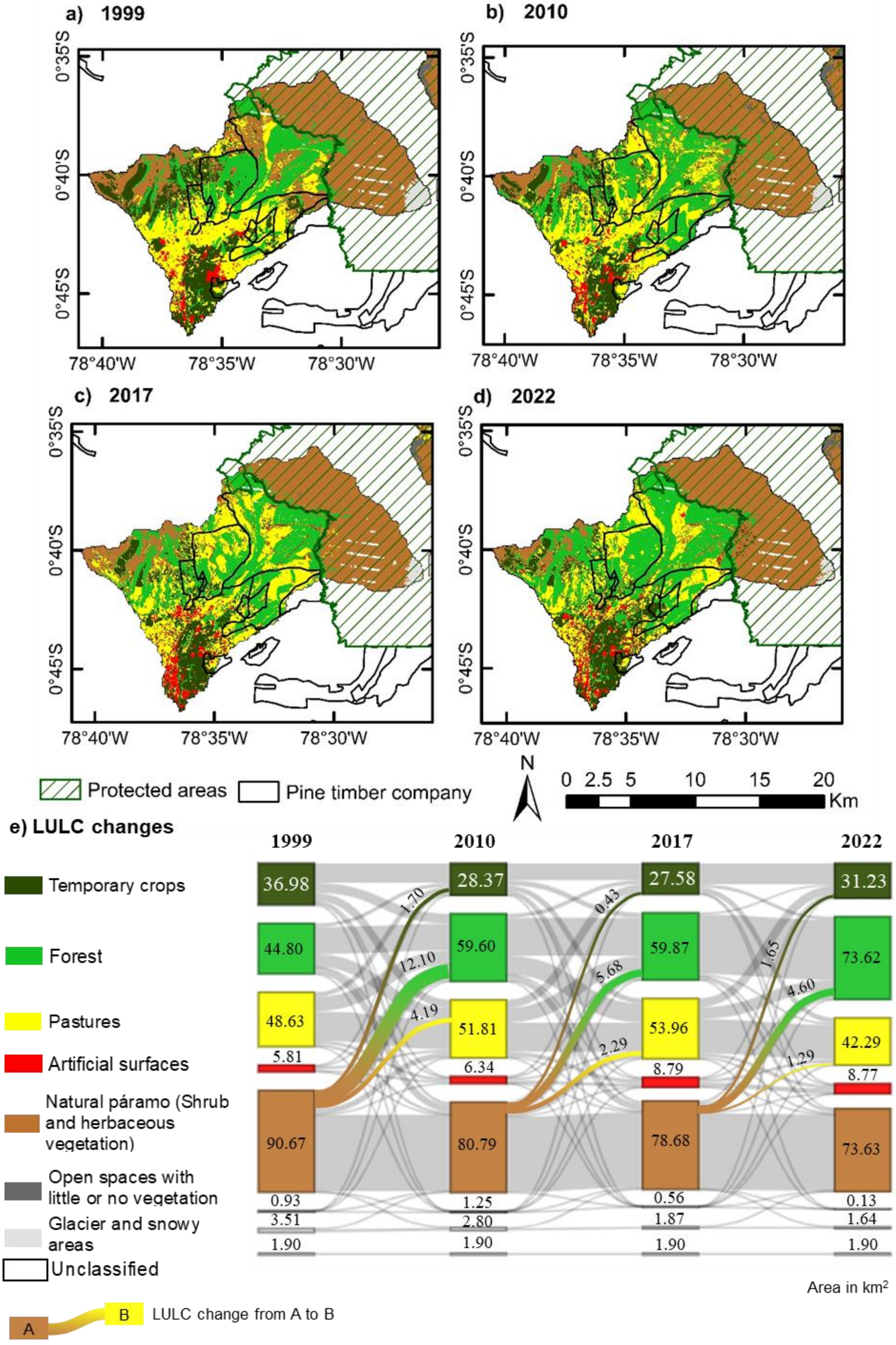
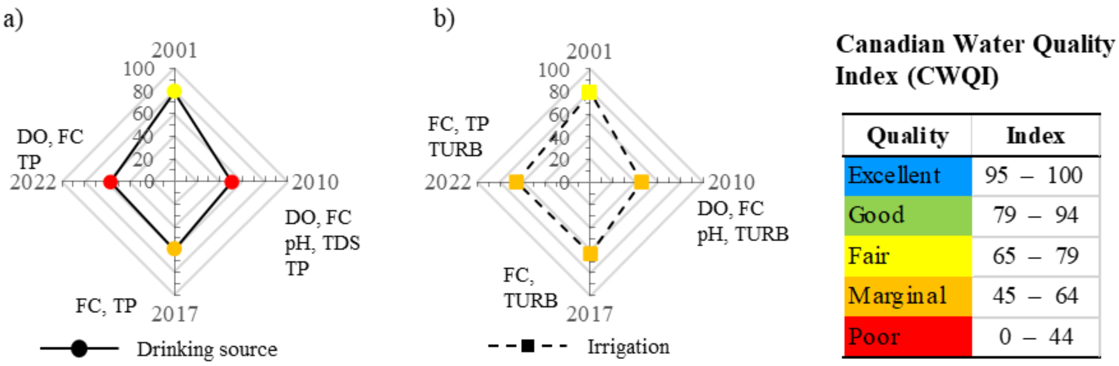


Fig. 3-5 LULC changes in UCutuchiRB between 1999 and 2022. Unclassified = Areas of persistent cloud cover located near snow-capped mountains.

Páramo losses occurred in the upper UCutuchiRB, with increased pine forests and pastures in the upper and middle basin areas, including Cotopaxi National Park, a pine timber company, and El Boliche National Recreational Area (Fig. 3-5a and d). Fig 3-5e shows that the natural páramo was converted primarily to pine forest, followed by temporary crops and pastures (major changes between 1999-2010).

- **Water quality changes in UCutuchiRB**

UCutuchiRB's sampling points (Fig. 3-1b) showed low DO and pH, along with high levels of FC, TDS, TP, and TURB that do not meet drinking water or irrigation water quality standards of Table 3-1. Consequently, between 2001 and 2022, UCutuchiRB experienced a decrease in CWQI from *fair* to *poor* for drinking water and from *fair* to *marginal* for irrigation water (Fig. 3-6).



Quality	Index
Excellent	95 – 100
Good	79 – 94
Fair	65 – 79
Marginal	45 – 64
Poor	0 – 44

Fig. 3-6. The lowest CWQI values in the years 2001, 2010, 2017, and 2022 and associated water quality parameters (see Table 3-1 for acronyms) that do not comply with water quality standards for a) drinking water and b) irrigation water in UCutuchiRB.

- **Impacts of LULC changes on water quality in UCutuchiRB**

Correlation analyses by applying the average values of each water quality parameter and each individual LULC class for the study period shows in Table 3-6 that the forests class (pine forest) was significantly correlated to total phosphorus TP (PCC = 0.99; $p < 0.01$) and open spaces with little or no vegetation was positively related to TDS (PCC = 0.99; $p < 0.01$).

Table 3-6. Correlation analysis using Pearson's correlation coefficient (PCC) between water quality parameters and percent changes in LULC in UCutuchiRB; *p≤0.05; **p≤0.01. For abbreviations of water quality parameters see Table 3-1.

LULC	**Water quality parameters**						
	DO	**FC**	**NO_3-N**	**pH**	**TDS**	**TP**	**TURB**
Artificial surfaces	0.99	0.90	0.12	0.94	-0.68	0.32	-0.54
Forests (pine)	0.39	0.70	0.98	-0.02	-0.91	0.99**	0.63
Glacier and snowy areas	-0.95	-0.99	-0.49	-0.74	0.91	-0.66	0.18
Open spaces with little or no vegetation	-0.74	-0.93	-0.81	-0.40	0.99**	-0.91	-0.24
Pastures	-0.28	-0.61	-0.99	0.13	0.86	-0.99	-0.71
Natural páramo (Shrub and/or herbaceous vegetation)	-0.52	-0.80	-0.94	-0.13	0.96	-0.99	-0.50
Temporary crops	0.30	0.62	0.99	-0.12	-0.87	0.99	0.70

3.4 Discussion

Passive ecological restoration as a NbS measure in water-rich ecosystems affected by anthropogenic pressures, such as páramo, requires a clear understanding of these ecosystems as complex biophysical arrangements that interact with PESTEL contexts to influence NbS performance.

Several studies indicate that since the 20th century, initiatives for sustainable land and water management, including passive ecological restoration, have been introduced at both national and local levels as strategies to reduce land degradation in páramo ecosystems and ensure water supply for Andean communities.

3.4.1 National-level strategies failing passive ecological restoration

The creation of protected areas (Echavarria, 2002; Joslin & Jepson, 2018), river basin management approach (Hoogesteger et al., 2016; Warner et al., 2014), and NbS such as passive ecological restoration (Brauman et al., 2019; Castanier, 2015; Wiegant, 2022), have been launched as this level across several páramos. However, our PESTEL analysis and corroborating studies demonstrate that many of these efforts have not been achieved due to their linkage to top-down management (national-level strategies) that do not fit local needs. For instance, protected areas managed face difficulties in addressing the complex transformations in páramo areas due to the national government's lack of active management and control over the territory, as well as limited financial resources (Echavarria, 2002; Joslin & Jepson, 2018). Similarly, river basin management initiatives, like those experienced in UCutuchiRB, as noted by Warner et al. (2014), have been largely ineffective in Ecuador because without the government's attaching political significance to this management scale, it is almost doomed to fail from the beginning. Regarding water-related ecological restoration efforts in Ecuador, research by Wiegant (2022) and Wiegant et al. (2020) shows that short-term electoral cycles and the desire to meet short-term political interests conflict with long-term restoration timelines.

Due to the ineffective implementation of land and water management strategies identified, large areas of Ecuador's páramo continue to be used and expanded primarily for extensive cattle ranching, large-scale agriculture, mineral extraction, and afforestation (Brück et al., 2023; Mosquera et al., 2023; Thompson et al., 2021). We observed that the natural páramo areas in the UCutuchiRB in 2022 covered only 31.6%, in contrast to the UPitaRB, which maintained 73.4% natural páramo. Similar results for natural páramo remnants (~30%) in central and southern Ecuador were obtained by García et al. (2019, 2020), where large-scale agriculture, cattle ranching, and forestry are present despite the establishment of protected areas.

The intensive páramo exploitation poses a significant threat to water security, given that even minor changes cause impacts on water quality. Rey-Romero et al. (2022) found that a small change of 15% from natural páramo to agricultural land use impairs water quality for drinking water sources through increased turbidity and fecal coliform levels. These findings align with the ecological conditions in the UCutuchiRB, where the water quality for drinking and irrigation is *poor* and *marginal*, respectively, both inside and outside protected areas (Acosta, 2018; Quishpe, 2018; Rojas, 2020; Zapata et al., 2021). In UCutuchiRB, the pine plantations cover significant areas of páramo areas, which our study identifies as the main class to predict high levels of total phosphorus and total dissolved solids in the basin. The presence of high concentrations of phosphates in the basins can be related to fertilizer management practices such as urea or superphosphate application (Lebron et al., 2012; Shah et al., 2022). In addition, livestock, mining, and urban systems adjacent to water courses have an impact on the input of materials to rivers and, therefore, also influence water quality (Cheng et al., 2022; Stenfert Kroese et al., 2020). Thus, Zapata et al. (2021) and Rojas (2020), based on a review of technical documents, found low concentrations of dissolved oxygen and high levels of fecal coliform, total dissolved solids, and turbidity in UCutuchiRB for drinking and irrigation water uses, respectively, in line with our results.

3.4.2 Local-level strategies promoting passive ecological restoration

In contrast, páramos like the UPitaRB implemented partially passive ecological restoration through bottom-up management (local-level strategies) to sustain mainly Quito's drinking water (Brauman et al., 2019; Castanier, 2015a; Coronel, 2019; Kauffman, 2014). Thus, at the political, social, and economic level, páramo headwater management and financial support directly relied on local utility companies, private and public water users, NGOs, and a Water Fund (Bremer et al., 2016; Coronel, 2019). At the economic and environmental level, given the limited financial resources and the fragility of the páramos, choosing passive ecological restoration was a key strategy. This strategy is much more cost-effective and requires less time than active restoration (Trujillo-Miranda et al., 2018).

Passive ecological restoration is especially effective for headwater streams to support high water quality conditions often found in these streams, typically situated in isolated areas away from populated areas (Haigh & Křeček, 1991; Taniwaki et al., 2019; Trujillo-Miranda et al., 2018). Furthermore, passive ecological restoration supports the achievement of several SDGs by promoting sustainable and efficient use of natural resources (target 12.2; SDG 12) (Groll, 2017), restoring aquatic ecosystems (target 6.6; SDG 6) (Kauffman, 2014), promoting the conservation of mountain ecosystems (target 15.4; SDGs 15) (Brauman et al., 2019), and reducing land degradation (target 15.3; SDG 15) (Muller et al., 2016). Additionally, it supports access to adequate, safe, and affordable basic services, focuses on impoverished neighborhoods (target 11.1; SDGs 11) (Castanier, 2015), and enhances local community livelihoods (target 1.4 SDG 1) (Coronel, 2019).

As a result of implementing passive ecological restoration in 50% of the basin, the UPitaRB maintained 73.4% natural páramo, *excellent* water quality for drinking water sources, and *good* water quality for irrigation sources until 2022. Equivalent results were obtained by García et al. (2020), who identified a páramo coverage of 74% in Cajas National Park and excellent water quality, where the implementation of strategies follows the UPitaRB management approach (Ministerio del Ambiente, 2018). In terms of investing in passive ecological restoration, Brauman et al. (2019) anticipate a positive financial return over the next twenty years for Water Utility Company in Quito. This projection is based on savings in water treatment costs resulting from reduced levels of nutrients, bacteria, turbidity, and sediments in the source water. Although there is no strong correlation between the impact of natural páramo on water quality parameters in UPitaRB in our study, this does not necessarily mean that the restoration has failed. Muller et al. (2016) recommend understanding the effects of passive ecological restoration on water quality by recognizing the complex nature of water quality as a dynamic watershed-scale process. Furthermore, passive ecological recovery takes time, even under optimal conditions, so relying solely on natural recovery processes is often seen as too slow and uncertain to meet the timelines and performance expectations of decision-makers (Zahawi et al., 2014).

Ongoing efforts to sustain aquatic ecosystem health are vital for long-term sustainability, and any interruption in these endeavors jeopardizes the progress made so far (Wang et al., 2016). Castillejo et al. (2018) found that the areas of the Pita River Basin with passive ecological restoration exhibited higher water quality compared to those without such interventions. The latter areas displayed lower water quality characterized by high levels of total phosphorus, high turbidity, and low dissolved oxygen levels associated with more populated zones. Although our findings suggest that temporary crops primarily negatively affect water quality parameters in UPitaRB forests, and open spaces with little or no vegetation could also contribute to this impact. For instance, open spaces with little or no vegetation (including bare rocks, sparse vegetation, and burned areas) could affect turbidity levels identified in UPitaRB and have been reported in similar studies by Carrillo & Díaz-

Villanueva (2021) and Chen & Chang (2023). Moreover, with the surge in afforestation in the UPitaRB, future research should prioritize the identification of tree species driving forest change. *Pinus radiata* and *Pinus patula* plantations are prevalent in páramo ecosystems. However, Mosquera et al. (2023) recommend investigating the impacts of other exotic (*Cupressus* and *Eucalyptus*) and native (*Polylepis*) plantations on water resources. This information can be valuable in understanding the direct impact of plantations on ecosystem water quality and can compel the adoption of sustainable management strategies.

3.5 Conclusions

Passive ecological restoration through bottom-up management in páramo ecosystems emerges as an effective NbS to mitigate widespread land and water degradation, shaped by a complex interplay of PESTEL factors since the 20th century. Over seven years (2010-2017), bottom-up management facilitated the recovery of up to 9.1% of the natural páramo while sustaining the capacity of headwater to provide safe drinking and irrigation water. Although our study did not reveal a strong correlation between restoration and water quality parameters, this does not necessarily indicate that restoration efforts have failed. Rather, it suggests the need for further investigation into potential influencing factors.

Conversely, top-down management, characterized by national-level strategies, lacks the legal, economic, and political interconnections necessary for successful passive ecological restoration in páramo ecosystems. Based on this management, several Ecuadorian páramos have lost up to 30% of their natural páramo vegetation, coinciding with an expansion of pine plantations and elevated total phosphorus levels in the basins. Additionally, agricultural and mining activities also contribute to the deterioration of water source quality. Given the severe degradation of páramo ecosystems, passive ecological restoration alone may not suffice to mitigate environmental impacts. Therefore, exploring alternative NbS or conventional methods, or their combination, is advisable to enhance the restoration process.

By integrating diverse data sources and methodologies, our findings provide valuable insights for decision-makers. However, challenges related to limited data availability underscore the need for future research to obtain continuous, representative, and accurate datasets to ensure the validity of our results. In particular, long-term monitoring programs are needed to assess surface water quality and conduct spatial analyses of LULC in páramos.

Chapter IV: Integrating community knowledge into nature-based solutions for the sustainability of water ecosystem services: Insight from local communities in Ecuador

This chapter is published as:

Fonseca, K., Espitia-Sarmiento, E. F., Ilbay-Yupa, M., & Breuer, L. (2024). Integrating community knowledge into nature-based solutions for the sustainability of water ecosystem services: Insight from local communities in Ecuador. *Frontiers in Sustainability*, *5*, 1491776. https://doi.org/10.3389/frsus.2024.1491776

Abstract

The political, economic, social, technological, environmental, and legal (PESTEL) dimensions in a local community shape the adoption of specific nature-based solutions (NbS). This study provides crucial insights on NbS tailored to smallholder indigenous and peasant communities heavily reliant on water ecosystem services from headwater streams, lakes, ponds, and reservoirs in the páramo ecosystems of the central Ecuadorian Andes. Combining a multi-stakeholder workshop with bibliometric analysis, we developed a framework that integrates NbS with local communities' PESTEL dimensions to sustain water ecosystem services. As a result, the lack of political will to sustain healthy aquatic ecosystems, urban- centered environmental investment, and agricultural expansion mainly influence the sustainability of water ecosystem services in the political, economic and environmental dimensions. Social, legal, and technological dimensions encompass community dissatisfaction, resistance to conservation, neglect of clean water and land use regulations, and limited innovation investment. Artificial floating islands and passive river restoration were the NbS adaptable to these PESTEL dimensions in our local communities. Artificial floating islands, a macrophyte- based technology that integrates community plant knowledge, are feasible even with limited financial resources. Passive river restoration complements them to promote headwater vegetation recovery and governance of water ecosystem services. Their integration supports drinking water supply, irrigation, fisheries, water purification, habitat conservation, soil formation carbon sequestration, and the

achievement of several Sustainable Development Goals (SDGs). We provide decision-makers with a rigorous assessment of NbS for local communities, with the potential to scale to countries with similar contexts and highlight the need for future research to explore NbS in regional or national frameworks.

4.1 Introduction

Relying on nature to improve human well-being and the environment is a shared goal of ecosystem services and nature-based solutions (Grizzetti et al., 2016; Jarosiewicz et al., 2022). Thus, ecosystem services are crucial for delivering social, ecological, and economic benefits to humanity. Moreover, nature-based solutions (NbS) maintain these benefits by conserving or restoring terrestrial and aquatic ecosystems (Blahna et al., 2017; Keesstra et al., 2018; Oral et al., 2020). In aquatic ecosystems, including rivers, lakes, ponds, and reservoirs and their diverse habitats, like riparian zones (Culhane et al., 2019; Grizzetti et al., 2016), NbS are crucial for sustaining several water ecosystem services. These include maintaining water quality and quantity, supporting nutrient cycling, maintaining populations and habitats, and promoting recreational and cultural services (Jarosiewicz et al., 2022; Possantti & Marques, 2022; Souliotis & Voulvoulis, 2022).

In developing countries, where ecological degradation of aquatic ecosystems is widespread, and local communities rely heavily on water ecosystem services for survival and livelihoods, NbS are crucial for achieving sustainability (Kenter et al., 2011; Telwala, 2023; UNCTAD, 2022). Cost-effective alternatives (Colares et al., 2020), promoting efficient and sustainable use of water resources (Souliotis & Voulvoulis, 2022), stakeholder engagement (Pagano et al., 2019), incorporation of community knowledge (Baustian et al., 2020), developing a healthier relationship between humans and the nonhuman world (Hoffman, 2023), improving the delivery of a range of ecosystem services (Liquete et al., 2016), and contribution to achieving multiple Sustainable Development Goals (SDGs) (Sowińska-Świerkosz & García, 2022) are some of the benefits highlighted by NbS.

Artificial floating islands (AFIs), one of the latest phytotechnology innovations in the remediation of degraded water bodies, stand out among NbS due to their low operational costs and maintenance, lack of land requirements, and ease of implementation (Afzal et al., 2019; Fonseca et al., 2021). The main component of AFIs is macrophytes with their associated microbial communities of biofilms and zooplankton, which play a dual role in directly assimilating pollutants into their tissues and acting as catalysts for purification reactions (Benvenuti et al., 2018; Cui et al., 2022). Moreover, passive ecological restoration, also known as passive river restoration, is especially suitable for headwater streams where water quality conditions are restored by eliminating disturbance factors and allowing for natural vegetation recovery (Muller et al., 2016; Taniwaki et al., 2019; Trujillo-Miranda et al., 2018).

To maximise their potential in local communities' aquatic ecosystems, AFIs and passive river restoration should be tailored to the broader macro-environment in which they operate. This macro-environment includes political, economic, social, technological, environmental, and legal factors, commonly known as PESTEL (Den Heijer & Coppens, 2023; Kansongue et al., 2023). However, the impact of PESTEL factors on water ecosystem services initiatives remains largely unknown.

Due to these complex adaptive dynamics of those NbS, PESTEL analysis has become a crucial tool for decision-makers to identify factors influencing their performance in improving environmental conditions and to guide adaptive responses (Den Heijer & Coppens, 2023). Similarly, the involvement of local communities is a vital aspect of NbS inclusion, as the degradation of aquatic ecosystems is not only critical on a global scale but also profoundly impacts locally(Baustian et al., 2020; Carroll et al., 2019; Giordano et al., 2020). Local communities, deeply rooted in agriculture and ancestral traditions (Conable, 2015), stand out as primary stakeholders in NbS design due to their experience gained through their long interaction with the environment and heavy reliance on ecosystem services (Balzan et al., 2022; Gbedemah, 2023).

Involving local communities and other key stakeholders, from problem identification to decision-making, contributes to more scientifically legitimate and publicly accountable decisions (Corburn, 2007; Khatibi et al., 2021). Therefore, community knowledge is growing in significance as it extends beyond individual ideas, is accessible to multi-stakeholders and the public, and offers opportunities for networked participation (Corburn, 2007; Hong & Scardamalia, 2014). Also, positive changes in resource management are more likely to be initiated when the attitudes, beliefs, or preferences of multi-stakeholders are considered in problem identification and the development of solutions (Lynam et al., 2007). Furthermore, for a practical application, Corburn (2007) recommends critical and sustained analysis of community knowledge. Thus, incorporating policy documents, grey literature, academic articles, and fieldwork strengthens community knowledge integration in research (Bisaga et al., 2021; Skrydstrup et al., 2020).

In developing countries, ensuring the health of aquatic ecosystems to sustain their services is a critical challenge; however, it also offers an opportunity for community collaboration, fostering community learning and innovation (Carroll et al., 2019; Vollmer et al., 2022). In this context, we developed a framework that integrates artificial floating islands and passive river restoration aligned with the PESTEL dimensions of local communities to sustain water ecosystem services in the central Ecuadorian Andes. Combining a multi-stakeholder workshop with a bibliometric analysis, we answered our research questions: (i) How do PESTEL dimensions of local communities influence the water ecosystem services' sustainability? (ii) How does integrating artificial floating islands with passive river restoration within the PESTEL dimensions of local communities contribute to the sustainability of water ecosystem services? In this study, we selected several Andean communities in Ecuador to illustrate the importance of healthy aquatic ecosystems in sustaining the water ecosystem services providing livelihoods for millions of people (Mosquera et al., 2023; Thompson et al., 2021).

4.2 Materials and methods

4.2.1 Study area

The study focused on local communities situated in the provinces of Pichincha, Cotopaxi, Bolivar and Chimborazo, encompassing 26,424 km2 in the central Andean region of Ecuador (Fig. 4-1). In these provinces, three major river basins (Guayas, Esmeraldas, and Pastaza) emerge, housing 68% of the Ecuadorian Andean population (INEC, 2022).

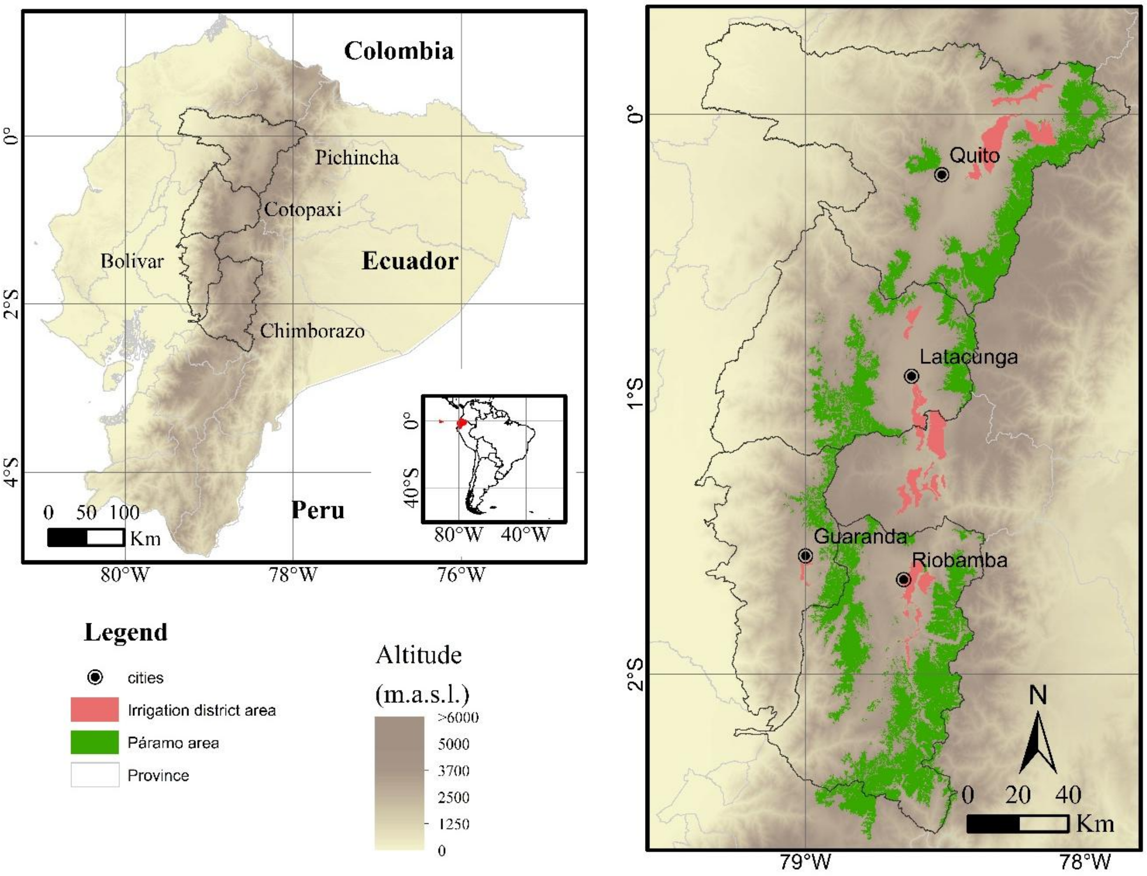


Fig. 4-1. Map depicting the central Ecuadorian Andes, which is home to the local communities

Agriculture, tourism, and mining are essential pillars of the economy of these provinces, collectively contributing 32% to Ecuador's gross domestic product (BCE, 2021). Despite this, the population grapples with significant levels of extreme poverty, with Chimborazo at 38.9%, Bolívar at 34.1%, and Cotopaxi at 26.3% (InfoMIES, 2022), rendering them particularly vulnerable. In addition, Chimborazo, Bolívar and Cotopaxi rank as

Ecuador's top provinces for chronic malnutrition in children under five, with rates of 35.2%, 35.1%, and 31.8%, respectively (INEC, 2019).

4.2.2 Water ecosystem services delivered to our local communities

High-elevation tropical ecosystems, known as páramo ecosystems, are home to local communities in our study area (Fig. 4-1). Local communities, including smallholder indigenous and peasant communities, in Ecuador's páramo ecosystems are authorised by articles 32, 45 and 73 of the Ecuadorian water law (Asamblea Nacional., 2014) to establish drinking water and irrigation boards and participate in the use and the community management of the water resources that flow through their territory.

The páramos encompass crucial aquatic ecosystems, i.e., the headwaters of many Andean river basins, along with lakes, ponds and reservoirs, which serve as primary drinking and irrigation water sources for the local communities (Mosquera et al., 2023; SENAGUA, 2021). Despite the benefits that urban and rural areas in Pichincha, Cotopaxi, Bolívar, and Chimborazo gain from the páramo ecosystems (Hofstede et al., 2014; SENAGUA, 2021), several studies indicate that their degradation threatens their sustainability (García et al., 2019; Mosquera et al., 2023; Thompson et al., 2021; Vinueza et al., 2021; Zapata et al., 2021). Hence, the provision of water ecosystem services, like drinking water, irrigation water, fisheries, and aquaculture, is directly compromised. Indirectly, the regulation and maintenance of further ecosystem services, including water purification, habitat and population maintenance, soil formation and composition, and carbon sequestration, are also affected. Moreover, there is a significant cultural impact concerning the indigenous cosmovision, where aquatic ecosystems hold a vital place in Mother Earth "Pachamama" alongside people, animals, ancestors, and land to achieve a holistic and well-balanced way of living, known in the indigenous language as "Sumak kawsay" (Kleemann et al., 2022; Salmoral et al., 2018).

4.2.3 Multi-stakeholder participation

Representatives of local communities, alongside decentralised autonomous governments at provincial, municipal, and parish levels in each province, were the primary stakeholders in building community knowledge. Additionally, participants from a non-governmental organisation (NGO) CARE (Cooperative for Assistance and Relief Everywhere), the Environmental Fund for Water Protection of Quito (FONAG), professors and researchers from the Technical University of Cotopaxi and the Justus Liebig University Giessen, were involved.

The selection of participation sectors was guided by Ecuadorian water law (Asamblea Nacional., 2014), which emphasises collective responsibility for sustainable water resource management shared among the national government, various governmental levels, and local communities, as outlined in articles 12 and 19. Furthermore, universities were invited to participate according to this law (Asamblea Nacional., 2014).

4.2.4 Developing a framework for NbS adaptation

We applied Albert et al.'s (2021) six-step NbS adaptation, divided into two stages. In the first stage, a participatory workshop was conducted to (1) co-defining the setting, which involved the workshop kick-off and clarifying the context, overarching societal challenges, aims, and processes, and (2) assessing challenges through multidimensional assessment, choosing the PESTEL dimension for our study (Den Heijer & Coppens, 2023; Kansongue et al., 2023). Subsequently, through bibliometric analysis of scientific data, we (3) identified that AFIs and passive river restoration have the potential to sustain water ecosystem services in our local communities. Hence, those NbS were (4) aligned with most of their PESTEL dimensions and related to several SDGs. Additionally, we (5) identified opportunities and obstacles and suggested future research directions for their implementation. The first NbS actions, outlined by Albert et al. (2021) as a six-step, do not align with the scope of this study.

4.2.5 Local communities' PESTEL dimensions captured by the workshop

In the first stage, multi-stakeholders were engaged in a workshop to analyse the multidimensional challenges in their local communities influencing the sustainability of water ecosystem services. Four participant groups were established based on their respective provinces to answer the research question: (i) How do PESTEL dimensions of local communities influence the water ecosystem services' sustainability? Subsequently, the information from these groups was consolidated and combined with studies from other developing countries to extend its applicability to countries experiencing similar contexts. A total of 39 participants attended the workshop held on 29 and 30 November 2022 in Latacunga, Ecuador.

4.2.6 Exploring and analysing scientific data for NbS adaptation

We began by selecting the Web of Science platform for scientific literature and bibliometric analysis because of its access to high-quality research published in scientific journals and alignment with bibliometric analysis tools. In the scientific literature search, we combined artificial floating islands and passive river restoration with aquatic ecosystems and water ecosystem services specific to our local communities using the following keywords: "artificial floating islands" "constructed wetlands", "passive river restoration", "passive ecological restoration" with "páramo", "headwater stream", "lake", "pond", and "reservoir" as well as "drinking water", "irrigation water", "fisheries", "aquaculture", "water purification", "habitat and population maintenance", "soil formation and composition", "carbon sequestration", "climate regulation", "cultural impact" and "SDGs", respectively. After identifying relevant studies, we conducted an abstract screening to select studies aligned with PESTEL dimensions of local communities. The literature screening concluded once no new information was obtained. In total, data was extracted from 62 peer-reviewed articles, academic books, and book chapters published between 1991 and 2023.

Subsequently, we utilised the VOSviewer software to create network view maps of the collected scientific knowledge based on Colares et al. (2020) and Donthu et al. (2021). Following data cleaning, the extracted terms were visualised as network view maps, showcasing clusters, core terms (large circles), and items (small circles). These clusters encompassed main core terms and items, with connections denoting links between them, and distances reflect the strength of relationships. The network view maps of artificial floating islands and passive river restoration were created using terms extracted from text data, including titles and abstracts. These maps allowed analysis from cluster and term perspectives (Colares et al., 2020) to explore the existing or future relationships among topics in a research field (Emich et al., 2020).

4.3. Results and discussion

This section presents the PESTEL dimensions influencing the sustainability of water ecosystem services in the local communities, identified by a multi-stakeholder workshop (Fig. 4-2), along with NbS adapted to these dimensions through a bibliometric analysis. Combining these results with discussion extends the applicability to countries facing similar contexts.

4.3.1. The PESTEL dimensions in the water ecosystem services' sustainability

- Political (P) dimension

Workshop participants reported that the main challenge faced by local communities is the lack of political will to support strategies that sustain healthy aquatic ecosystems (P1), a frequent obstacle in water resources management in several developing countries. Warner et al. (2014) partly attributed these political challenges in developing countries to the complexity of managing aquatic ecosystems and conflicting interests. For example, in recent decades, the river basin scale has been deemed the "natural" scale for water planning and management (Molle, 2008), but it is not a priority in countries like Ecuador. In contrast, politicians often use more relevant issues, such as land tenure conflicts, as a political platform to gain political support, as reported by Coral et al. (2021) in their study on lessons from Ecuador's history.

The undervaluation of water ecosystem services due to political priorities in developing countries hinders the implementation of sustainable practices in key sectors, like agriculture, that impact the sustainability of water ecosystem services (Nahuelhual et al., 2018; Salmoral et al., 2018). Similarly, the limited local community participation in ecosystem management (P3) exacerbates these challenges from a multi-stakeholder perspective. Inadequacies of poor communities' political socioeconomic infrastructure and knowledge base and distrust are the main issues for the governance of sustained participation. Deficiencies in the socio-political infrastructure of local communities and mistrust in these systems are the main problems, as per Khatibi et al. (2021), for sustained participation in governance. Carroll et al. (2019) observed that lack of community engagement can increase the risks of water catastrophes. These findings align with our local communities, where unsustainable management in agriculture, livestock grazing, and urban growth contribute to the degradation of aquatic ecosystems (Castelo-Cabay et al., 2022; Hofstede et al., 2023).

Concerning policies aimed at sustaining water ecosystem services in our local communities, a lack of long-term strategies is common (P2). Research by Wiegant (2022) and Wiegant et al. (2020) on strategies related to water ecosystem services in Ecuador shows that short-term election cycles and the desire to meet political interests at the governance scale conflict with long-term restoration timelines.

- Economic (Ec) dimension

The limited governmental financial support (Ec1), as recognised by participants in the workshop and noted by Saud et al. (2019), is usual in most countries' economies because environmental protection strategies are not a priority and are often neglected in government expenditure. This situation is particularly acute in developing countries, where economic and financial crises frequently result in significant cuts to public spending on the environment (Coral et al., 2021; Sarkar et al., 2007). In addition, international and private funding is insufficient to protect and restore water-related ecosystems (UNEP, 2023; Wiegant et al., 2020). While developing countries urgently need increased efforts to address the escalating crisis of freshwater resource scarcity, current initiatives and research predominantly concentrate on developed nations (Chen et al., 2019; UN-Water, 2016).

Furthermore, workshop participants considered that the focus on environmental investment in large urban centres (Ec2) impacts local communities. Policies at the national government level in environmental protection, as described by Chen et al. (2019), frequently overlook the surrounding areas of large cities. For instance, in Quito, the capital of Ecuador, around 98% of the population has access to safe drinking water, in contrast to rural areas where access is only 54% (EPMAPS, 2021; Vinueza et al., 2021). Similarly, strategies for agricultural water management are absent in many developing nations across Eastern Europe, South Asia, and South America (Chen et al., 2019).

Meanwhile, investment in environmental protection and green innovation encounters the absence of investment returns (Ec3) in our local communities. The unavailability of immediate revenue streams to offset considerable up-front costs, the lack of public accounting practices to assess the economic value of water ecosystem services (Den Heijer & Coppens, 2023) and the extended waiting period (five to ten years or more) before reaping the benefits (Hudson et al., 2023), hinder environmental investment.

- Social (S) dimension

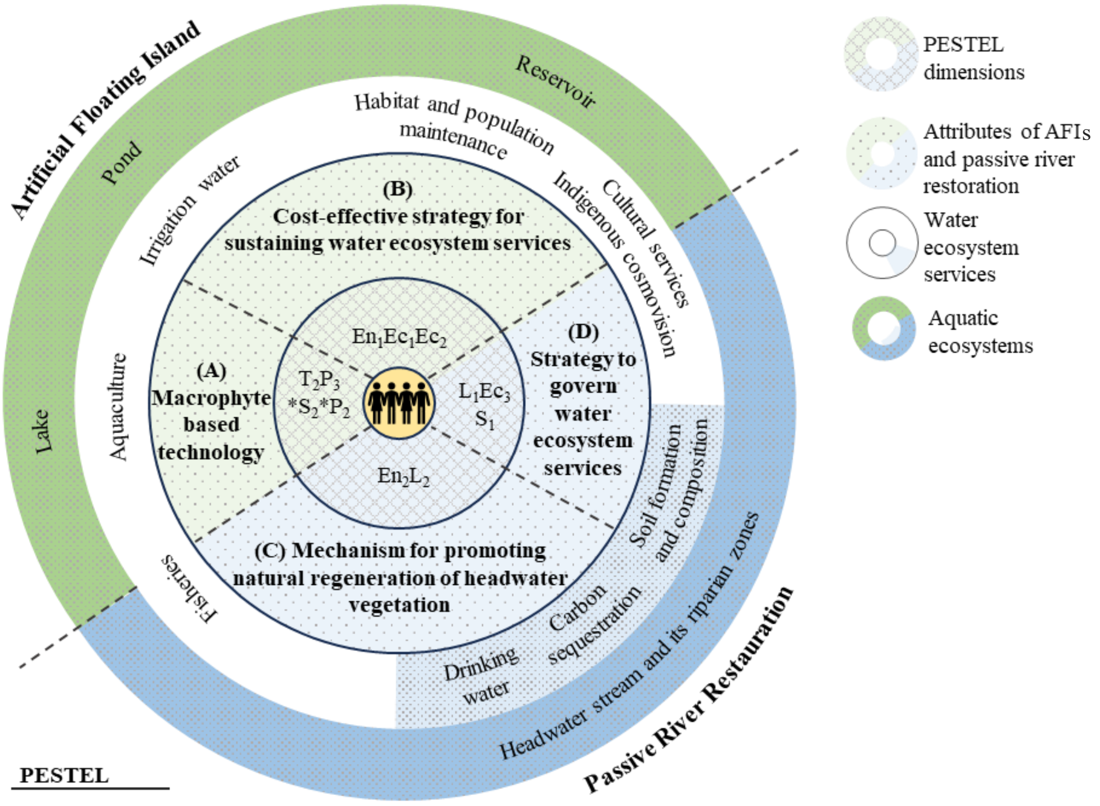
Participants acknowledged that local community growth causes conflicts between water users in the upper, middle and lower basins (S1). Local community growth, inadequate land use planning and migration to urban centres exacerbate competition for water resources in the basins (Mulligan et al., 2010). It is worth noting that population migration to large cities intensifies the demand for various ecosystem services such as water, food, and others provided by rural areas to meet the needs of the urban population (Mulligan et al., 2010). As urbanisation accelerates, competition for freshwater between cities and agriculture is expected to escalate, with urban water demand projected to rise by 80% by 2050 (UN-Water, 2023).

Additionally, dissatisfaction and resistance to conservation strategies (S2) arise from factors like perception outcomes and familiarity with specific practices within our local communities. Perception of ecosystem recovery from restoration projects often hinges more on aesthetic criteria than recognition of ecological quality (Junker & Buchecker, 2008). As indicated by Jourdain et al. (2023), recognising the environmental effects of protecting aquatic ecosystems is challenging, given that the outcomes heavily rely on the diverse perspectives of multi-stakeholders with varying interests in water ecosystem services and different opinions on problem definitions and potential solutions. Similarly, different backgrounds, educational levels, and traditional beliefs can lead to varying perceptions, often resulting in misunderstandings about environmental impacts (Arsénio et al., 2020; Porras et al., 2018). Additionally, people frequently resist transitioning from conventional to environmentally friendly strategies due to economic uncertainty, financial investment, and the need to develop new skills (Rode, 2021).

- Technological (T) dimension

Improving the health of aquatic ecosystems is a significant challenge because of the mismatches between data and technology in our local communities (T1). Scientific data regarding the effects of implemented technologies on natural resources and their benefits for humans and ecosystems are not easily accessible (Apostolaki et al., 2019; Grigg, 2016; Jourdain et al., 2023). Strategies to address this data problem, such as governments embracing mandatory environmental information disclosure, have emerged worldwide (Chen & Cho, 2019). However, the mechanisms by which this disclosure can effectively garner public support for environmental initiatives remain unclear, while decision-making processes continue to be influenced by the information available (Glaser et al., 2021; Grigg, 2016). Hence, decision-makers are either economically rational, making consistent decisions when they gather sufficient information, or economically irrational, varying their choices based on available options (Glaser et al., 2021; Grigg, 2016).

In addition, according to UN-Water (2023), the innovation with partners from the North and South could go a long way towards developing technically feasible, economically viable, socially acceptable and locally adaptable solutions to core water-related challenges. Considering patenting activity for water-related innovation has more than doubled since 1990 and is monopolised in developing countries (OECD, 2018), and the scarce investment in technology innovation (T2) within our local communities, achieving collaborative innovation remains a far-off objective.



PESTEL

P: Political

$*P_1$ Lack of political will to sustain healthy aquatic ecosystems.
$*P_2$ Limited local community participation in ecosystem management.
P_3 Absence of long-term strategies.

Ec: Economical

Ec_1 Limited governmental financial support.
Ec_2 Focus on environmental investment in large urban centres.
Ec_3 Absence of investment returns.

S: Social

S_1 Local community growth causes conflicts between water users in the upper, middle and lower basins.
$* S_2$ Dissatisfaction and resistance to conservation strategies.

T: Technological

$*T_1$ Mismatches between data and technology.
T_2 Scarce investment in technology innovation.

En: Environmental

En_1 Local communities heavily rely on water ecosystem services for their livelihoods.
En_2 Agricultural expansion, grazing and urban development in páramo ecosystems degrade essential water ecosystem services.

L: Legal

L_1 Laws to protect aquatic ecosystems are neglected.
L_2 Lack of strict sanctions against environmental violations.
$* L_3$ Regulations historically favoured large-scale agriculture in crucial aquatic ecosystems.

* Future research directions

Fig. 4-2. NbS to sustain water ecosystem services embedded in the local communities' PESTEL dimensions

- Environmental (En) dimension

Our local communities heavily rely on water ecosystem services for their livelihoods (En1); nevertheless, agricultural expansion, grazing and urban development in páramo ecosystems degrade essential water ecosystem services (En2). Páramo ecosystems, along with several high-elevation ecosystems across continents, act as water towers providing numerous ecosystem services while remaining vulnerable to anthropogenic pressures and highly sensitive to climate change (Buytaert et al., 2006; Shahgedanova et al., 2021). For instance, agricultural development in páramo ecosystems is attributed to páramo soils, which have a high carbon content, a high soil moisture content, a high water retention capacity, and a high hydraulic conductivity (Buytaert et al., 2005; Calispa et al., 2021; García et al., 2020b). Besides, the páramo ecosystems experience high frequencies of precipitation, which are key for agriculture, primarily attributable to orographic effects, resulting in annual sums of precipitation exceeding 1,500 mm (Crespo et al., 2019; Ilbay-Yupa, Ilbay, et al., 2021).

In local communities, agriculture is crucial in providing ecosystem services, such as food (Mosquera et al., 2023). However, the long-term impact of agriculture significantly undermines other ecosystem services (Gordon et al., 2010). For example, water quality is affected by unsustainable agriculture, even in regions with abundant water availability (Mulligan et al., 2010). Agricultural land management, plant protection and fertiliser application contribute significantly to water pollution, as heavy rainfall leads to surface runoff, erosion and subsequent sediment transport into rivers, or nutrients, pollutants (agrochemicals) and faecal coliform bacteria are washed out and enter surface waters (Clausen & Meals, 1989; Rey-Romero et al., 2022). New data from Lu and Tian (2017) show that the use of N and P fertilisers per agricultural area has increased by a factor of eight for nitrogen and three for phosphorus since the 1960s, contributing to widespread eutrophication and the development of hypoxic zones in coastal zones (Maúre et al., 2021), but also to an immense spread of eutrophic conditions in inland waters (Wang et al., 2018).

At the same time, livestock grazing intensifies nutrient inputs (from manure), alters plant diversity through trampling and consumption, compacts the soil, and induces stream bank collapse, thus affecting stream morphology and aquatic and riparian zones (Muller et al., 2016). Additionally, the introduction of non-native species in the páramo, despite habitat alterations, is usual to boost economic returns in less viable agricultural areas (Buytaert et al., 2007). In the same way, urbanisation significantly modifies the hydrology of basins and the transport of sediment, nutrients, and pollutants, thereby impacting water quality (Gyawali et al., 2013; Sheldon et al., 2019).

As a result, in Ecuador, aquatic ecosystems used for agriculture, livestock, fish farming, and drinking contain high levels of total coliforms, and agrochemical and metal concentrations that exceed acceptable water quality standards (Capparelli et al., 2020; Vinueza et al., 2021). It exposes the population to waterborne diseases related to faecal contamination and heavy metals, leading to significant health risks, including high morbidity and mortality (Mitra et al., 2022; Some et al., 2021). Specifically, the Ecuadorian rural sector is vulnerable, as only 48.5% have access to drinking water that meets the national regulations (INEC, 2022). Apart from the impacts on providing and regulating water ecosystem services, impacts on cultural ecosystem services affect indigenous populations due to their cosmovision regarding the meaning of water (Kleemann et al., 2022).

- Legal (L) dimension

The right to use water must be addressed by water law so that people and organisations can have security of their access to water to meet their needs (Grigg, 2016). Nonetheless, participants reported that laws to protect aquatic ecosystems are neglected in the local communities (L1), resulting in deteriorating water quality (Vinueza et al., 2021; Zapata et al., 2021). This global concern impacts the sustainability of water ecosystem services despite efforts with modern paradigms such as stringent regulations, integrated water resources management, and sustainable sanitation (Jourdain et al., 2023; UN-Water, 2016). A significant challenge in regulation is that it aims to enforce laws to control behaviour in the public interest, yet defining the public interest remains a complex and elusive objective (Grigg, 2016).

From the point of view of participants, in the local communities, there is a lack of strict sanctions against environmental violations (L2). According to Grigg (2016), water users should not expect to self-regulate, and oversight is essential to ensure that the bodies responsible for creating and enforcing rules are held accountable. In addition, specialised institutions must be vigilant to ensure that organisations that have violated the social contract, like environmental damage, are liable to bear the cost of the damage caused to society as per legitimacy theory (Buccina et al., 2013). An aspect highlighted in Wiegant (2022) is that limitations on water ecosystem services for local actors, without fair compensation, are ineffective. Instead of complying, local actors may circumvent regulations, like cutting fences around protected areas and enabling illegal grazing by livestock again.

A final point derived from participants was that regulations historically favoured large-scale agriculture in crucial aquatic ecosystems (L3), thereby disadvantaging peasant farmers (Solo De Zaldívar, 2015). These regulations limit restoration strategies to cover private areas or pose problems in addressing widespread ecosystem degradation (Coronel, 2019; Partridge, 2016; Wiegant, 2022).

4.3.2. NbS embedded in the local communities' PESTEL dimensions

We examined the network view maps of two specific NbS for our local communities: artificial floating islands (AFIs) and passive river restoration. The terms macrophytes in Fig. 4-2, water quality in Fig. 4-3, restoration in Fig. 4-4, and governance in Fig. 4-5, as well as their associated items, caught our attention in the network view maps, respectively. By associating these terms and items, and supported by the literature review, AFIs were characterised as (A) macrophyte-based technology and (B) a cost-effective strategy for sustaining water ecosystem services, aligning with multiple PESTEL dimensions of our local communities (Fig. 4-2). In turn, passive river restoration complemented the PESTEL dimension that AFIs did not accommodate. Hence, it serves as a (C) mechanism to promote the natural regeneration of headwater vegetation and as a (D) strategy to govern water ecosystem services (Fig. 4-2).

- Macrophyte-based technology

The term macrophyte in Fig. 4-3 served as a crucial term for bibliometric analysis, as highlighted by Colares et al. (2020), and its items such as phytotechnology, root, shoot, mechanism, removal, system, performance, water body, time, total phosphorus, total nitrogen, chemical oxygen demand, biochemical oxygen demand, turbidity, heavy metals, and macrophytes (Canna and Juncus), lead to characterise AFIs. AFIs are one of the latest phytotechnology innovations in the remediation of degraded water bodies. Its natural mechanism for removing pollutants, low maintenance, lack of land requirements, and ease of implementation (Afzal et al., 2019; Colares et al., 2020; Prashant & Billore, 2020) make it ideal for local communities like ours, considering the scarce investment in technology innovation (T2) and the absence of long-term strategies (P3) (Fig. 4-2).

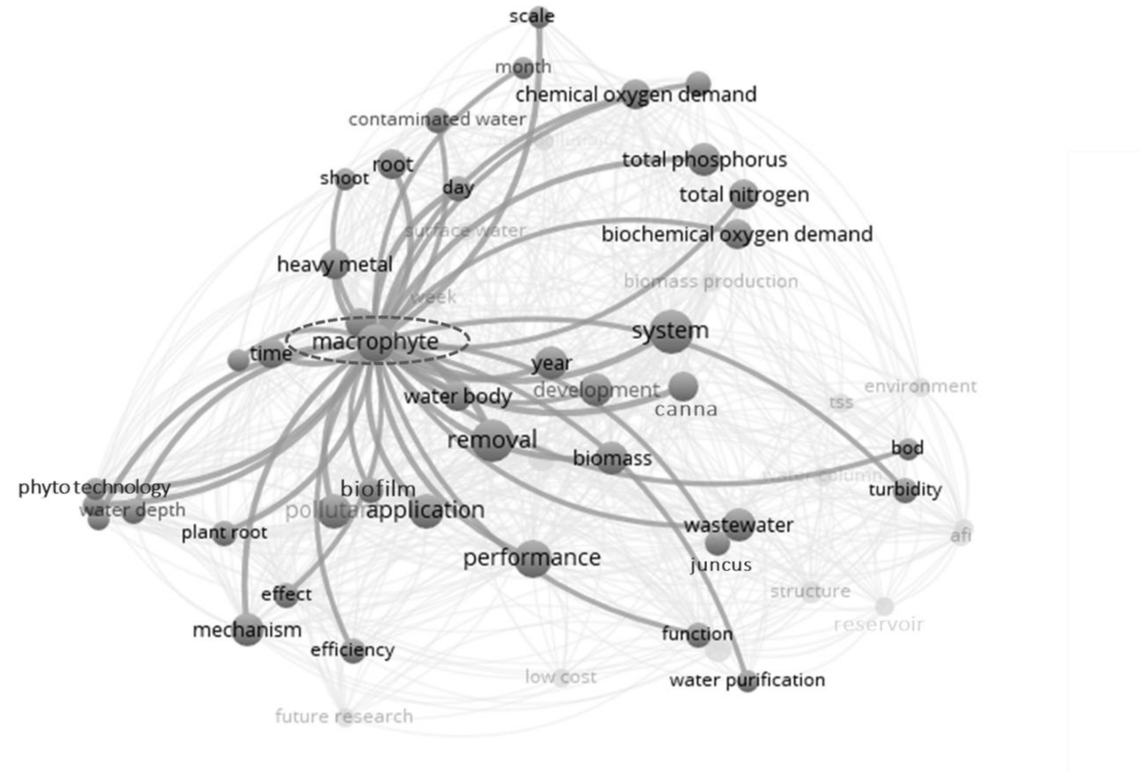


Fig. 4-3. Choosing "macrophyte" in the AFIs network view map highlights connected items.

The low operating cost of AFIs is due to the high removal efficiency, mainly dependent on macrophytes (Cui et al., 2022; Samal, 2019; Souliotis & Voulvoulis, 2022; Yeh et al., 2015). The main component of AFIs is macrophytes with their associated microbial communities of biofilms and zooplankton, which play a dual role in directly assimilating pollutants into their tissues and acting as catalysts for purification reactions (Benvenuti et al., 2018; Cui et al., 2022).

Floating, floating-leaf, emergent and submersed macrophytes are used to improve the polluted water bodies (De Stefani et al., 2011; Zhao et al., 2012). A broad range of plants are suitable for these purposes, with the most common genera *Canna, Carex, Cyperus, Juncus,* and *Typha* (Colares et al., 2020). Also, *Phragmites australis* (common reed), *Oenanthe javanica* (water celery), *Iris pseudacorus* (yellow flag), *Glyceria maxima* (sweet manna grass), *Chrysopogon zizanioides* (vetiver grass), and *Ipomea aquatica* (water spinach) are commonly used (Chance et al., 2022; De Stefani et al., 2011; Samal, 2019). Using different types of plants leads to diversity within the island, which results in more biodiversity, better functioning, and more stable islands (Idris et al., 2010).

Hence, AFIs restore aquatic ecosystems through mechanisms, including water self-purification, bioaccumulation, plant uptake, microbial assimilation, adsorption-sedimentation, and other pathways (Hwang et al., 2020; Tanner & Headley, 2011; Yeh et al., 2015). Consequently, a reduction in pollutants, suspended solids, oxygen demand and excessive nutrients is achieved in the water (Afzal et al., 2019; Benvenuti et al., 2018; Prashant & Billore, 2020). Similarly, concentrations of metals like arsenic, iron, lead, copper, chromium, cadmium, and nickel (Afzal et al., 2019; Fonseca et al., 2020; Gaballah et al., 2021). In addition, AFIs effectively remove bacteria such as faecal coliforms from water, reduce turbidity, and increase dissolved oxygen (DO) (Olguín et al., 2017; Prashant & Billore, 2020).

This water improvement is analysed over short-term periods such as seven days (Gaballah et al., 2021), 20 days (Guo et al., 2014), and 120 days (Fonseca et al., 2020), as well as over long-term periods including 12 months (Benvenuti et al., 2018), 18 months (Afzal et al., 2019), and 24 months (Olguín et al., 2017).

- A cost-effective strategy for the water ecosystem services' sustainability

The term water quality includes items such as low cost, efficiency, system, performance, water purification, wastewater, lake, reservoir, biomass, water body, application, and development, as depicted in Fig. 4-4. This guided the alignment of AFIs for local communities heavily reliant on water ecosystem services for their livelihoods (En1), despite facing challenges of limited governmental financial support (Ec1), and focus on environmental investment in large urban centres (Ec2), as shown in Fig.4-6.

AFIs contribute directly to vital water ecosystem services such as irrigation water supply, fisheries, and aquaculture and indirectly to water purification and maintaining populations and habitats by reducing pollution, aligning with Target 6.3 of Sustainable Development Goal 6 in various water bodies (De Moraes et al., 2023; Fletcher et al., 2024; Masi et al., 2018).

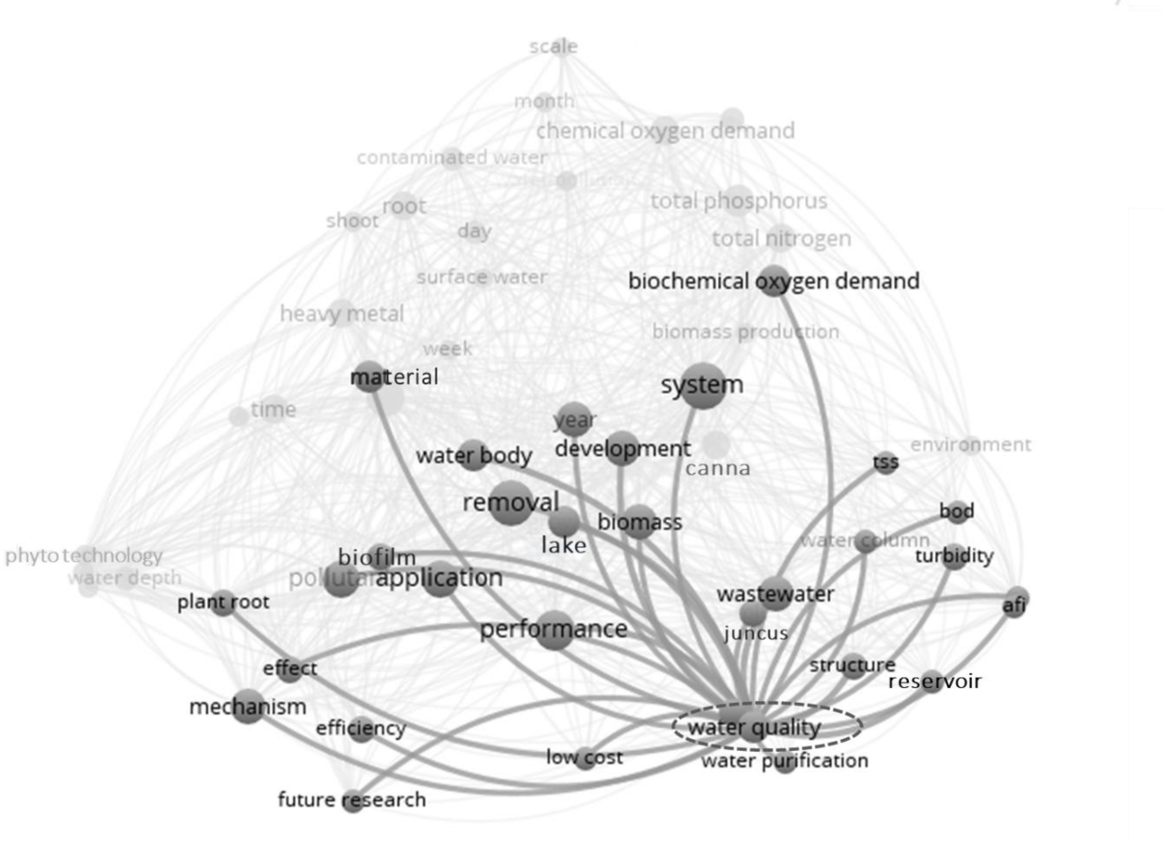


Fig. 4-4. Choosing “water quality” in the AFIs network view map highlights connected items.

In particular, in developing countries, AFIs have shown success in regulating severe eutrophic river conditions (Guo et al., 2014), treating oil-polluted water in pits (Afzal et al., 2019), reducing mainly pollutants in wastewater treatment plants (Benvenuti et al., 2018; Prashant & Billore, 2020), providing water for agriculture (Fonseca et al., 2020) and fisheries from reservoirs and lakes (De Moraes et al., 2023), and enhancing the aesthetic appreciation of ponds (Olguín et al., 2017).

To enhance pollutant removal, key attributes of the materials used for AFIs construction, such as buoyancy, durability, anchoring, flexibility, easy installation, affordability, and hydrophobic materials, must be considered (Samal, 2019; Sharma et al., 2021). Bamboo, PVC pipes, polypropylene pipes, polystyrene sheets, and inflatable vinyl are used for the main features of the system's buoyancy (Samal, 2019). To fix plants and facilitate biofilm attachment, plastic nets and mediums such as rice straw, bristle coir fibre, and volcanic gravel, among others, are used (Samal, 2019; Wang et al., 2020). The selection of materials depends to a large extent on their mechanical resistance associated with biological, chemical, and weather resistance to prevent the potential degradation of materials such as plastic (De Stefani et al., 2011; Ware & Callaway, 2019).

In addition, macrophyte type, environmental conditions, water depth, buoyancy, coverage rate, macrophyte configuration mode, and periodic removal are crucial factors for ensuring optimal performance of AFIs. The selection of macrophytes is based mainly on the ability of the roots to distribute throughout the water column, on the activity of the leaves to transpire water, nutrients and heavy metal storage in plant material, and root filtration (Barco & Borin, 2020; Hwang et al., 2020). A water depth consideration is crucial depending on the type of macrophyte species planted in the water to ensure that the roots float in the aquatic ecosystem and prevent them from adhering to the sediment (Samal, 2019). Furthermore, flotation enables plants to remain unaffected by water level fluctuations (De Stefani et al., 2011). Moreover, vegetation coverage ratio and different configurations of macrophytes can improve pollutant removal. A high vegetation cover (more than 50%) may create anoxic conditions in the water as it prevents diffusion of oxygen from the air to water due to wind activity, whereas low coverage (9–18%) may add an insignificant amount of treatment effect (Samal, 2019). The diversity of macrophytes composing the AFIs is also relevant, with single-species AFIs showing lower total biomass production (Liu et al., 2014). Finally, periodic removal of plant biomass (harvest) from water bodies is essential to maintain purification efficiency and prevent pollutants from returning to the water during decomposition (Zhao et al., 2012). Harvesting above-mat vegetation is a common practice to enhance nutrient removal efficiency, considering seasonal variations and the type of macrophyte involved (Barco & Borin, 2020; Samal, 2019).

Plants on floating islands can be harvested for various purposes, including producing saleable plants that generate profits over the lifetime of AFIs, serving as animal feeds, or processing into biogas, bio-fertiliser, and bio-materials (Yeh et al., 2015; Zhao et al., 2012). Yet, questions remain about AFIs use for human food consumption due to health concerns, lack of definitive safety tests, and economic potential for reselling the whole plant post-harvest from a treatment system (Chance et al., 2022).

- A mechanism for promoting natural regeneration of headwater vegetation

In Fig. 4-5, the term restoration and items like mechanism, land, time, headwater, river, riparian ecosystem, vegetation recovery, landowner, stakeholder, perception, restoration effort, potential, effect, limitation, disturbance, natural succession, and rural landscape, are highlighted to align with two crucial PESTEL dimension. Agricultural expansion, grazing and urban development in páramo ecosystems degrade essential water ecosystem services (En2) due to the lack of strict sanctions against environmental violations (L2), which could be mitigated through passive river restoration (Fig. 4-2).

Passive river restoration is particularly well-suited for headwater streams (first and second-order streams), where water quality conditions are restored by eliminating disturbance factors and allowing for natural vegetation recovery (Buytaert et al., 2007; Taniwaki et al., 2019; Trujillo-Miranda et al., 2018). Headwater streams are characterised by controlling both the structure and operation of superior-order rivers (Forget et al., 2013), often located outside the economic centre of a country, isolated and commonly neglected (Haigh & Křeček, 1991).

Passive river restoration, such as livestock enclosure, is promoted because it saves time, effort, and money and can be implemented along an entire stream, whereas active restoration often occurs only on a short river section (Jähnig et al., 2010; McIver & Starr, 2001; Prach & Hobbs, 2008). By limiting or removing sources of disturbance from streams with some existing level of bank stability and riparian vegetation, autogenic primary processes may allow some level of recovery to in-stream habitats (Hough-Snee et al., 2013). Mechanisms like fencing out livestock, fallowing cropland, removing over-abundant lianas or thinning and controlled burns in fire-suppressed forests promoting natural succession through minimal management intervention in an ecosystem offer legal, economic and environmental advantages (Arsénio et al., 2020; Brauman et al., 2019).

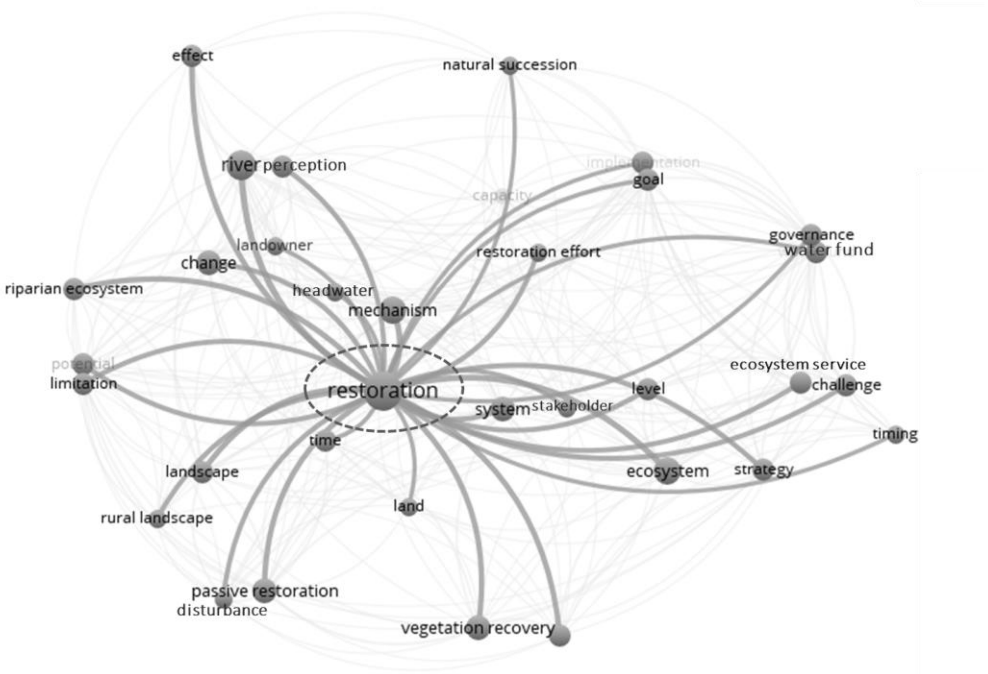


Fig. 4-5. Choosing "restoration" in the passive river network view map highlights connected items

Land ownership, water users, and management rights are factors that can either limit or drive restoration efforts. Some actors may perceive restoration efforts as a means to assist ecosystem recovery and its biotic community, while others may prioritise more highly managed or commercially focused interventions (Chazdon et al., 2021). Also, gaining the support of mainly landowners is a challenge, as it often entails the loss of grazing land. For instance, in the Oir River basin in France, according to Muller et al. (2016), the maximum distance a river manager can handle is one meter away from the river bank. Therefore, it is recommended that restoration efforts be integrated within a restoration governance framework to enhance the likelihood of success and sustainability of restoration programs (Sapkota et al., 2018).

- A strategy for governing water ecosystem services

In Fig. 4-6, the term governance and associated items such as goal, water fund, ecosystem service, restoration effort, challenge, capacity, mechanism, restoration, and system are highlighted to align several PESTEL dimensions. Thus, neglect of laws protecting aquatic ecosystems in local communities (L1), the absence of investment returns (Ec3), and conflicts between water users in the upper, middle and lower basins caused by local community growth (S1) are depicted in Fig. 4-2.

Along a headwater stream, interactions between the stream and riparian area are tightly coupled (Richardson & Danehy, 2007), playing a crucial role in restoring aquatic ecosystems (target 6.6; SDG 6). Riparian vegetation contributes to water purification by filtering sediments, pesticides, and particulate organic matter and reduces nutrients such as nitrate and phosphorous in groundwater (Muller et al., 2016; Yang et al., 2021). It creates deep, dense root networks to protect the soil (Forget et al., 2013). In particular, in the headwaters of the important Andean river basins, natural páramo vegetation contributes to soil carbon accumulation (Brauman et al., 2019; Mosquera et al., 2023). Thus, water, soil and vegetation, the three elements of the riparian area (Yang et al., 2021), are the focus of passive river restoration to directly support the sustainability of drinking water sources and ensure access to adequate, safe, and affordable basic services, aligning with Target 11.1 of SDG 11. Indirectly, this restoration contributes to soil formation and carbon sequestration (Grizzetti et al., 2016; Khorchani et al., 2022; Mosquera et al., 2023) by reducing land degradation (target 15.3; SDG 15) (Muller et al., 2016), and promoting the conservation of mountain ecosystems (target 15.4; SDG15) (Brauman et al., 2019).

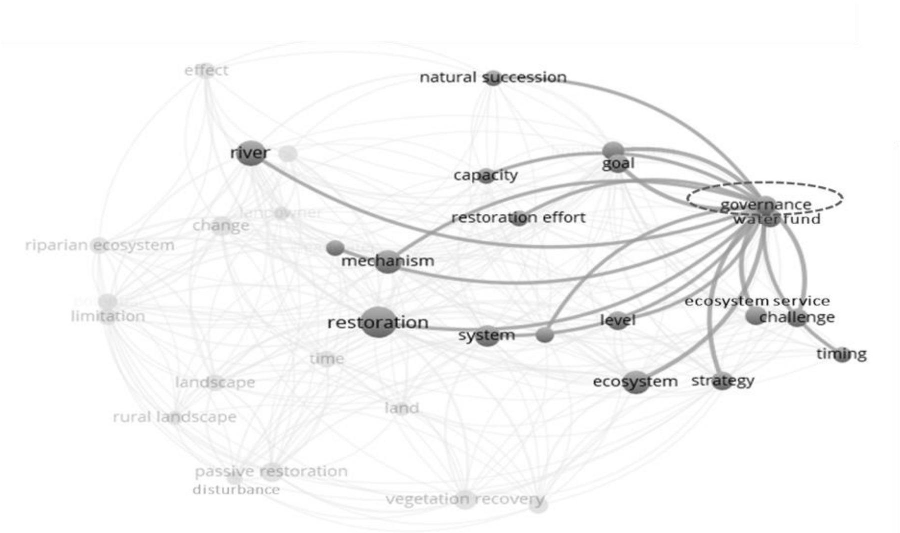


Fig. 4-6. Choosing “governance” in the passive river restoration network view map highlights connected items.

Restoration mechanisms are gradually emerging as a useful concept within the law, which can develop and enforce mechanisms to manage degraded ecosystems to restore them to desirable states (Sapkota et al., 2018). In this context, external financing mechanisms like water funds, whose goal is to invest in priority strategies such as private land acquisition, communal and private agreements, and coordinated or co-managed public lands, have facilitated long-term passive restoration efforts and managed conflicts between these actors in Ecuador and other developing countries (Brauman et al., 2019; Castanier, 2015; Coronel, 2019). Also, the focus on monetary valuation and payment mechanisms contributes to attracting political support for supporting conservation and commodifying a growing number of ecosystem services (Coral et al., 2021). In terms of investing in passive river restoration, Brauman et al. (2019) anticipate a positive financial return over the next twenty years for Water Utility Company in Quito. This projection is based on savings in water treatment costs resulting from reduced levels of nutrients, bacteria, turbidity, and sediments in the source water. Combining this strategy with enforcement mechanisms, adapting large-scale restoration programs, re-framing conventional top-down management, and prioritizing areas for restoration contribute to the sustainability of water ecosystem services (Csákvári et al., 2022; Khorchani et al., 2022; Sippi & Parmar, 2024).

4.3.3 Future research directions for AFIs and passive river restoration

The network view maps of AFIs and passive river restoration facilitated alignment with multiple PESTEL dimensions of local communities (Fig. 4-2). Nonetheless, crucial elements such as local community involvement, stakeholders, and experts were absent.

Integrating community knowledge into phytotechnology advancements could considerably influence AFIs (Fletcher et al., 2024; Gutierrez-Gines et al., 2021). Research should focus on understanding local communities' views on systems designed by manipulating macrophyte composition to provide various ecosystem services, a significantly underexplored field (Fletcher et al., 2024; Yongabi et al., 2018). Combining local place-based perspectives with expert and scientific insight is essential in ecosystem management (Balzan et al., 2022; Gbedemah, 2023). For example, AFIs are perceived as simple in their construction, operation and maintenance; however, numerous complex processes directly influence system performance and pollutant removal efficiency. Factors such as inlet contaminant concentrations, water retention time, pH, salinity condition, and microorganisms in the rhizosphere of macrophytes impact the system (Colares et al., 2020; Yeh et al., 2015) and require expert analysis. Public participation, including local communities and experts, also helps to identify concerns on AFIs installation, such as degradation of the plastic matrix, long-term maintenance and disturbance of native macrophytes (Ware & Callaway, 2019).

Hence, AFIs as a macrophyte-based technology could incentivise multi-stakeholder involvement to address local community dissatisfaction and resistance to conservation strategies (S2) and limited local community involvement in ecosystem management (P2) in our study area (Fig. 4-2). Involving multiple stakeholders not only helps in understanding accurate perceptions of ecosystem recovery but also provides operational and guidance information for future ecological restoration with an adaptive approach (Arsénio et al., 2020; Mercado et al., 2024). Furthermore, by enhancing local community livelihoods following Target 1.4 of SDG 1 and by mobilising and sharing knowledge, expertise, technology, and financial resources, AFIs support the achievement of these SDGs.

Moreover, a significant knowledge gap exists regarding AFIs on headwater streams for drinking water sources, a critical necessity upon which local communities heavily rely. Only a few studies highlight AFIs implementation on aquatic vegetation near riversides and mosaic floating islands, mainly addressing the mitigation of nutrient enrichment (Yeh et al., 2015; Zhao et al., 2012). It led us to combine AFIs with passive river restoration to tackle the PESTEL dimensions that the AFIs alone do not effectively address, as shown in Fig. 4-2 Nonetheless, the lack of political will to support strategies for maintaining healthy aquatic ecosystems (P1), the mismatches between data and technology (T1) and regulations historically favouring large-scale agriculture in crucial aquatic ecosystems (L3) of the PESTEL dimensions of local communities extends beyond the AFIs and passive river restoration. Since these challenges cannot be addressed by local community solutions alone, there is a need for future research to explore NbS in regional or national frameworks.

4.4 Conclusion

We developed a novel framework that demonstrates how integrating artificial floating islands (AFIs) with passive restoration addresses the complexity of local community PESTEL dimensions that influence the sustainability of water ecosystem services. AFIs, a cost-effective macrophyte-based technology, are adapted to political, economic, social, technological and environmental dimensions. They offer short- and long-term results with low investment in technology and a perspective of incorporating community plant knowledge. By improving the health of lakes, ponds and reservoirs in rural and urban landscapes in the local communities, AFIs contribute directly to vital water ecosystem services such as irrigation water supply, fisheries, and aquaculture and indirectly to water purification and maintaining populations and habitats.

Moreover, passive river restoration primarily addresses the legal dimension and complements the environmental and social dimensions AFIs cannot align. It promotes the natural regeneration of headwater vegetation to sustain drinking water sources while indirectly contributing to soil formation and carbon sequestration. At the same time, it demonstrates how restoration efforts have prompted laws to safeguard aquatic ecosystems in other local realities. Improving aquatic ecosystems also could help achieve the cultural values embedded in indigenous cosmovision. Integrating AFIs with passive river restoration can also achieve several SDGs, including promoting sustainable and efficient natural resource use (Target 12.2, SDG 12), reducing water pollution (Target 6.3, SDG 6), restoring aquatic ecosystems (Target 6.6, SDG 6), mitigating land degradation (Target 15.3, SDG 15), and conserving mountain ecosystems (Target 15.4, SDG 15). This integration also supports sustaining drinking water sources and ensuring access to adequate, safe, and affordable basic services (Target 11.1, SDG 11), improving local community livelihoods (Target 1.4, SDG 1), and mobilizing and sharing knowledge, expertise, technology, and financial resources to further the SDGs (Target 17.16, SDG 17).

The framework will facilitate informed decision-making for water ecosystem protection by a critical NbS assessment in a developing country while remaining adaptable to countries with similar contexts. Furthermore, as a policy tool, it highlights the benefits of AFIs and passive river restoration for water ecosystem services and advocates their integration into conservation and restoration projects.

References

Acharki, S. (2022). PlanetScope contributions compared to Sentinel-2, and Landsat-8 for LULC mapping. *Remote Sensing Applications: Society and Environment*, *27*, 100774. https://doi.org/10.1016/j.rsase.2022.100774

Acosta, D. (2018). *Identificación de especies de diatomeas epilíticas asociadas al grado de eutrofización del río cutuchi en la provincia de Cotopaxi* [Technical University Cotopaxi]. http://repositorio.utc.edu.ec/handle/27000/8429

Adriaenssens, V., Baets, B. D., Goethals, P. L. M., & Pauw, N. D. (2004). Fuzzy rule-based models for decision support in ecosystem management. *Science of The Total Environment*, *319*(1–3), 1–12. https://doi.org/10.1016/S0048-9697(03)00433-9

Afzal, M., Rehman, K., Shabir, G., Tahseen, R., Ijaz, A., Hashmat, A., & Brix, H. (2019). Large-scale remediation of oil-contaminated water using floating treatment wetlands. *Npj Clean Water*, *2*. https://doi.org/10.1038/s41545-018-0025-7

Aglomerados Cotopaxi. (2021). *Sembrando futuro: Memoria de sostenibilidad*. https://www.cotopaxi.com.ec/en

Ahmed, S., Khurshid, S., Madan, R., Abu Amarah, B. A., & Naushad, M. (2020). Water quality assessment of shallow aquifer based on Canadian Council of Ministers of the environment index and its impact on irrigation of Mathura District, Uttar Pradesh. *Journal of King Saud University - Science*, *32*(1), 1218–1225. https://doi.org/10.1016/j.jksus.2019.11.019

Al Amin, M., Adrianto, L., Kusumastanto, T., & Imran, Z. (2021). Community knowledge, attitudes and practices towards environmental conservation: Assessing influencing factors in Jor Bay Lombok Indonesia. *Marine Policy*, *129*, 104521. https://doi.org/10.1016/j.marpol.2021.104521

Albert, C., Brillinger, M., Guerrero, P., Gottwald, S., Henze, J., Schmidt, S., Ott, E., & Schröter, B. (2021). Planning nature-based solutions: Principles, steps, and insights. *Ambio*, *50*(8), 1446–1461. https://doi.org/10.1007/s13280-020-01365-1

Alcamo, J. (2019). Water quality and its interlinkages with the Sustainable Development Goals. *Current Opinion in Environmental Sustainability*, *36*, 126–140. https://doi.org/10.1016/j.cosust.2018.11.005

Allou, S., Cazamajor, P., Godard, H., Gómez, N., Gravelin, B., & León, J. (1987). El espacio urbano en Ecuador. *IGM de Ecuador*, 303.

Alvarez, M. O., & Sanchez, J. P. D. (2018). An overview of urbanization in Ecuador under functional urban area definition. *REGION*, *5*(3), 38–48. https://doi.org/10.18335/region.v5i3.235

Amores, M. (2019). *Evaluación de la Variación de la Calidad del Agua en la Cuenca Alta del Río Pita.* [Universidad Técnica de Cotopaxi]. https://rraae.cedia.edu.ec/Record/UTC_bf76c888738e379a7a542fd8162fdb16

Anderson, D. M., Gupta, A. K., Birken, S., Sakas, Z., & Freeman, M. C. (2021). Successes, challenges, and support for men versus women implementers in water, sanitation, and hygiene programs: A qualitative study in rural Nepal. *International Journal of Hygiene and Environmental Health*, *236*, 113792. https://doi.org/10.1016/j.ijheh.2021.113792

Andrade, J., Cunha, J., Silva, J., Rufino, I., & Galvão, C. (2021). Evaluating single and multi-date Landsat classifications of land-cover in a seasonally dry tropical forest. *Remote Sensing Applications: Society and Environment*, *22*, 100515. https://doi.org/10.1016/j.rsase.2021.100515

APHA, AWWA, WEF. (2005). *Standard Methods for the Examination of Water and Wastewater* (21st ed.). American Public Health Association/American Water Works Association/Water Environment Federation.

APHA, AWWA, WEF. (2017). *Standard Methods for the Examination of Water and Wastewater* (23rd ed.). American Public Health Association/American Water Works Association/Water Environment Federation.

Apostolaki, S., Koundouri, P., & Pittis, N. (2019). Using a systemic approach to address the requirement for Integrated Water Resource Management within the Water Framework Directive. *Science of The Total Environment*, *679*, 70–79. https://doi.org/10.1016/j.scitotenv.2019.05.077

Araújo, C. V. M., & Cedeño-Macias, L. A. (2016). Heavy metals in yellowfin tuna (Thunnus albacares) and common dolphinfish (Coryphaena hippurus) landed on the Ecuadorian coast. *Science of The Total Environment*, *541*, 149–154. https://doi.org/10.1016/j.scitotenv.2015.09.090

Arsénio, P., Rodríguez-González, P., Bernez, I., Dias, F., Bugalho, M., & Dufour, S. (2020). Riparian vegetation restoration: Does social perception reflect ecological value? *River Research and Applications*, *36*(6), 907–920. https://doi.org/10.1002/rra.3514

Asamblea Constituyente. (2008). *Constituci ́on de la República del Ecuador. Registro Oficial 449. Montecristi, Ecuador.* https://www.cancilleria.gob.ec/wp-content/uploads/2013/06/constitucion_2008.pdf

Asamblea Nacional. (2010). *Código Orgánico de Organización Territorial (COOTAD)*. http://www.bomberoslatacunga.gob.ec/site/transparencia2019/enero/a2/COOTAD.pdf

Asamblea Nacional. (2014). *Ley Orgánica de Recursos Hídricos Usos y Aprovechamiento del Agua.* http://www.regulacionagua.gob.ec/wp-content/uploads/2019/06/Ley-Org%C3%A1nica-de-Recursos-H%C3%ADdricos-Usos-y-Aprovechamiento-del-Agua.pdf

Asamblea Nacional Constituyente. (1998). *Constitución Política de la República de Ecuador, 1998*. https://www.acnur.org/fileadmin/Documentos/BDL/2002/0061.pdf

Ashok, A., Rani, H. P., & Jayakumar, K. V. (2021). Monitoring of dynamic wetland changes using NDVI and NDWI based landsat imagery. *Remote Sensing Applications: Society and Environment*, *23*, 100547. https://doi.org/10.1016/j.rsase.2021.100547

Ávila, B., & Gallo, M. N. (2021). Morphological behavior of the Magdalena River delta (Colombia) due to intra and interannual variations in river discharge. *Journal of South American Earth Sciences*, *108*, 103215. https://doi.org/10.1016/j.jsames.2021.103215

Balzan, M. V., Geneletti, D., Grace, M., De Santis, L., Tomaskinova, J., Reddington, H., Sapundzhieva, A., Dicks, L. V., & Collier, M. (2022). Assessing nature-based solutions uptake in a Mediterranean climate: Insights from the case-study of Malta. *Nature-Based Solutions*, *2*, 100029. https://doi.org/10.1016/j.nbsj.2022.100029

Barco, A., & Borin, M. (2020). Treatment performances of floating wetlands: A decade of studies in North Italy. *Ecological Engineering*, *158*. https://doi.org/10.1016/j.ecoleng.2020.106016

Baumgartner, T., & Pahl-Wostl, C. (2013). UN–Water and its Role in Global Water Governance. *Ecology and Society*, *18*(3). https://doi.org/10.5751/ES-05564-180303

Baustian, M. M., Jung, H., Bienn, H. C., Barra, M., Hemmerling, S. A., Wang, Y., White, E., & Meselhe, E. (2020). Engaging coastal community members about natural and nature-based solutions to assess their ecosystem function. *Ecological Engineering: X*, *5*, 100015. https://doi.org/10.1016/j.ecoena.2019.100015

BCE. (2021). *Cuentas nacionales regionales. Serie anuales de cuentas provinciales, 2009-2020.* https://contenido.bce.fin.ec/documentos/Estadisticas/SectorReal/CuentasProvinciales/Indice.htm

Beitl, C. M., Rahimzadeh-Bajgiran, P., Bravo, M., Ortega-Pacheco, D., & Bird, K. (2019). New valuation for defying degradation: Visualizing mangrove forest dynamics and local stewardship with remote sensing in coastal Ecuador. *Geoforum*, *98*, 123–132. https://doi.org/10.1016/j.geoforum.2018.10.024

Benvenuti, T., Hamerski, F., Giacobbo, A., Bernardes, A. M., Zoppas-Ferreira, J., & Rodrigues, M. A. S. (2018). Constructed floating wetland for the treatment of domestic sewage: A real-scale study. *Journal of Environmental Chemical Engineering*, *6*(5), 5706–5711. https://doi.org/10.1016/j.jece.2018.08.067

Bisaga, I., Parikh, P., Tomei, J., & To, L. S. (2021). Mapping synergies and trade-offs between energy and the sustainable development goals: A case study of off-grid solar energy in Rwanda. *Energy Policy*, *149*, 112028. https://doi.org/10.1016/j.enpol.2020.112028

Blacketer, M. P., Brownlee, M. T. J., Baldwin, E. D., & Bowen, B. B. (2021). Fuzzy Cognitive Maps of Social-Ecological Complexity: Applying Mental Modeler to the Bonneville Salt Flats. *Ecological Complexity*, *47*, 100950. https://doi.org/10.1016/j.ecocom.2021.100950

Blahna, D. J., Asah, S. T., & Deal, R. L. (2017). An Ecosystem Services Framework. In D. H. Olson & B. Van Horne (Eds.), *People, Forests, and Change: Lessons from the Pacific Northwest* (pp. 62–75). Island Press/Center for Resource Economics. https://doi.org/10.5822/978-1-61091-768-1_5

Brauman, K. A., Benner, R., Benitez, S., Bremer, L., & Vigerstøl, K. (2019). Water Funds. In *Green Growth That Works* (1st ed., p. 319). Island Press.

Bremer, L. L., Auerbach, D. A., Goldstein, J. H., Vogl, A. L., Shemie, D., Kroeger, T., Nelson, J. L., Benítez, S. P., Calvache, A., Guimarães, J., Herron, C., Higgins, J., Klemz, C., León, J., Sebastián Lozano, J., Moreno, P. H., Nuñez, F., Veiga, F., & Tiepolo, G. (2016). One size does not fit all: Natural infrastructure investments within the Latin American Water Funds Partnership. *Ecosystem Services*, *17*, 217–236. https://doi.org/10.1016/j.ecoser.2015.12.006

Brockwell, E., Elofsson, K., Marbuah, G., & Nordmark, S. (2021). Spatial analysis of water quality and income in Europe. *Water Resources and Economics*, *35*, 100182. https://doi.org/10.1016/j.wre.2021.100182

Brück, S. A., Torres, B. D. M., & de Moraes Polizeli, M. de L. T. (2023). The Ecuadorian paramo in danger: What we know and what might be learned from northern wetlands. *Global Ecology and Conservation*, *47*, e02639. https://doi.org/10.1016/j.gecco.2023.e02639

Buccina, S., Chene, D., & Gramlich, J. (2013). Accounting for the environmental impacts of Texaco's operations in Ecuador: Chevron's contingent environmental liability disclosures. *Accounting Forum*, *37*(2), 110–123. https://doi.org/10.1016/j.accfor.2013.04.003

Bundschuh, J., Armienta, M. A., Morales-Simfors, N., Alam, M. A., López, D. L., Delgado Quezada, V., Dietrich, S., Schneider, J., Tapia, J., Sracek, O., Castillo, E., Marco Parra, L.-M., Altamirano Espinoza, M., Guimarães Guilherme, L. R., Sosa, N. N., Niazi, N. K., Tomaszewska, B., Lizama Allende, K., Bieger, K., … Ahmad, A. (2021). Arsenic in Latin America: New findings on source, mobilization and mobility in human environments in 20 countries based on decadal research 2010-2020. *Critical*

Reviews in Environmental Science and Technology, *51*(16), 1727–1865. https://doi.org/10.1080/10643389.2020.1770527

Buytaert, W., Célleri, R., De Bièvre, B., Cisneros, F., Wyseure, G., Deckers, J., & Hofstede, R. (2006). Human impact on the hydrology of the Andean páramos. *Earth-Science Reviews*, *79*(1), 53–72. https://doi.org/10.1016/j.earscirev.2006.06.002

Buytaert, W., Iñiguez, V., & Bièvre, B. D. (2007). The effects of afforestation and cultivation on water yield in the Andean páramo. *Forest Ecology and Management*, *251*(1–2), 22–30. https://doi.org/10.1016/j.foreco.2007.06.035

Buytaert, W., Sevink, J., De Leeuw, B., & Deckers, J. (2005). Clay mineralogy of the soils in the south Ecuadorian páramo region. *Geoderma*, *127*(1), 114–129. https://doi.org/10.1016/j.geoderma.2004.11.021

Calispa, M., van Ypersele, R., Pereira, B., Páez-Bimos, S., Vanacker, V., Villacís, M., Molina, A., de Bièvre, B., Muñoz, T., & Delmelle, P. (2021). *Soil organic carbon stocks under different páramo vegetation covers in Ecuador's northern Andes*. EGU21-4121. EGU General Assembly Conference Abstracts. https://doi.org/10.5194/egusphere-egu21-4121

Calle, P., Monserrate, L., Medina, F., Calle Delgado, M., Tirapé, A., Montiel, M., Ruiz Barzola, O., Cadena, O. A., Dominguez, G. A., & Alava, J. J. (2018). Mercury assessment, macrobenthos diversity and environmental quality conditions in the Salado Estuary (Gulf of Guayaquil, Ecuador) impacted by anthropogenic influences. *Marine Pollution Bulletin*, *136*, 365–373. https://doi.org/10.1016/j.marpolbul.2018.09.018

Cammelli, F., Garrett, R. D., Barlow, J., & Parry, L. (2020). Fire risk perpetuates poverty and fire use among Amazonian smallholders. *Global Environmental Change*, *63*, 102096. https://doi.org/10.1016/j.gloenvcha.2020.102096

Capparelli, M. V., Moulatlet, G. M., Abessa, D. M. de S., Lucas-Solis, O., Rosero, B., Galarza, E., Tuba, D., Carpintero, N., Ochoa-Herrera, V., & Cipriani-Avila, I. (2020). An integrative approach to identify the impacts of multiple metal contamination sources on the Eastern Andean foothills of the Ecuadorian Amazonia. *Science of The Total Environment*, *709*, 136088. https://doi.org/10.1016/j.scitotenv.2019.136088

Carrillo, U., & Díaz-Villanueva, V. (2021). Impacts of volcanic eruptions and early recovery in freshwater environments and organisms. *Biological Reviews*, *96*(6), 2546–2560. https://doi.org/10.1111/brv.12766

Carrion, A., Goetschel, A. M., & Sánchez, N. (1997). *Breve historia d elos servicios de Quito/ Brief history of Quito's public services*. 147.

Carroll, J., Beck, J., Boyer, E., Dhanorkar, S., & Gupta, S. (2019). Empowering Community Water Data Stakeholders. *Interacting with Computers*, *31*(5), 492–506. https://doi.org/10.1093/iwcomp/iwz032

Castanier, H. (2015). Economic Valuation for Decision Making on the Protection of Water Sources. In K. W. Hipel, L. Fang, J. Cullmann, & M. Bristow (Eds.), *Conflict Resolution in Water Resources and Environmental Management* (pp. 81–108). Springer International Publishing. https://doi.org/10.1007/978-3-319-14215-9

Castelo-Cabay, M., Piedra-Fernandez, J. A., & Ayala, R. (2022). Deep learning for land use and land cover classification from the Ecuadorian Paramo. *International Journal of Digital Earth*, *15*(1), 1001–1017. https://doi.org/10.1080/17538947.2022.2088872

Castillejo, P., Chamorro, S., Paz, L., Heinrich, C., Carrillo, I., Salazar, J. G., Navarro, J. C., & Lobo, E. A. (2018). Response of epilithic diatom communities to environmental gradients along an Ecuadorian Andean River. *Comptes Rendus Biologies*, *341*(4), 256–263. https://doi.org/10.1016/j.crvi.2018.03.008

Castillo, L. L. C. (2022). *Análisis de las brechas de implementación del modelo normativo de gobernanza del agua en Ecuador. Estudio de caso de la subcuenca del río Cutuchi.* [Universidad Central de Ecuador]. http://www.dspace.uce.edu.ec/bitstream/25000/25666/1/UCE-FAG-CPO-CASTILLO%20LUCIA.pdf

CCME. (2017). *Canadian water quality guidelines for the protection of aquatic life: CCME Water Quality Index (UPDATE).* Canadian Council of Ministers of the Environment. https://ccme.ca/en/res/wqimanualen.pdf

Chance, L., Hall, C., & White, S. (2022). Viability assessment for the use of floating treatment wetlands as alternative production and remediation systems for nursery and greenhouse operations. *Journal of Environmental Management*, *305*. https://doi.org/10.1016/j.jenvman.2021.114398

Chazdon, R. L., Falk, D. A., Banin, L. F., Wagner, M., J. Wilson, S., Grabowski, R. C., & Suding, K. N. (2021). The intervention continuum in restoration ecology: Rethinking the active–passive dichotomy. *Restoration Ecology*, e13535. https://doi.org/10.1111/rec.13535

Chen, C.-T., & Chiu, Y.-T. (2021). A study of dynamic fuzzy cognitive map model with group consensus based on linguistic variables. *Technological Forecasting and Social Change*, *171*, 120948. https://doi.org/10.1016/j.techfore.2021.120948

Chen, J., & Chang, H. (2023). A review of wildfire impacts on stream temperature and turbidity across scales. *Progress in Physical Geography: Earth and Environment*, *47*(3), 369–394. https://doi.org/10.1177/03091333221118363

Chen, W., & Cho, F. (2019). Environmental information disclosure and societal preferences for urban river restoration: Latent class modelling of a discrete-choice experiment. *Journal of Cleaner Production*, *231*, 1294–1306. https://doi.org/10.1016/j.jclepro.2019.05.307

Chen, W., He, B., Nover, D., Lu, H., Liu, J., Sun, W., & Chen, W. (2019). Farm ponds in southern China: Challenges and solutions for conserving a neglected wetland ecosystem. *Science of The Total Environment*, *659*, 1322–1334. https://doi.org/10.1016/j.scitotenv.2018.12.394

Cheng, C., Zhang, F., Shi, J., & Kung, H.-T. (2022). What is the relationship between land use and surface water quality? A review and prospects from remote sensing perspective. *Environmental Science and Pollution Research*, *29*(38), 56887–56907. https://doi.org/10.1007/s11356-022-21348-x

Christmann, T., & Oliveras, I. (2020). Nature of Alpine Ecosystems in Tropical Mountains of South America. In *Encyclopedia of the World's Biomes* (pp. 282–291). Elsevier. https://doi.org/10.1016/B978-0-12-409548-9.12481-9

Clausen, J. C., & Meals, D. W. (1989). Water quality achievable with agricultural best management practices. *Journal of Soil and Water Conservation*, *44*(6), 593–596.

CODERECO. (2002). *Manejo integral de los recursos hídricos y Tratamiento de las aguas servidas – Cuenca del río Cutuchi* (p. 50). https://www.dspace.espol.edu.ec/bitstream/123456789/5988/1/PITOLO_RIO_CUTUCHI.pdf

Colares, G. S., Dell'Osbel, N., Wiesel, P. G., Oliveira, G. A., Lemos, P. H. Z., da Silva, F. P., Lutterbeck, C. A., Kist, L. T., & Machado, Ê. L. (2020). Floating treatment wetlands: A review and bibliometric analysis. *Science of The Total Environment*, *714*, 136776. https://doi.org/10.1016/j.scitotenv.2020.136776

Conable, J. (2015). Foreign Land Acquisitions, Corruption, and Sustainable Livelihood in Sub-Saharan Africa: The Cases of Mozambique and Tanzania. In *Handbook of Research on In-Country Determinants and Implications of Foreign Land Acquisitions:* IGI Global. https://doi.org/10.4018/978-1-4666-7405-9

Congreso Nacional. (1999). *Ley Nº 37 de 30 de julio de 1999—Gestión Ambiental.* https://www.acnur.org/fileadmin/Documentos/BDL/2008/6618.pdf

Coral, C., Bokelmann, W., Bonatti, M., Carcamo, R., & Sieber, S. (2021). Understanding institutional change mechanisms for land use: Lessons from Ecuador's history. *Land Use Policy*, *108*, 105530. https://doi.org/10.1016/j.landusepol.2021.105530

Corburn, J. (2007). Community knowledge in environmental health science: Co-producing policy expertise. *Environmental Science & Policy*, *10*(2), 150–161. https://doi.org/10.1016/j.envsci.2006.09.004

Cornejo, C., & Wilkie, A. C. (2010). Greenhouse gas emissions and biogas potential from livestock in Ecuador. *Energy for Sustainable Development*, *14*(4), 256–266. https://doi.org/10.1016/j.esd.2010.09.008

Coronel, L. (2019). *The path of water- FONAG: projects and lessons.* https://www.fonag.org.ec/web/wp-content/uploads/2019/11/Sistematizaci%C3%B3n-FONAG-Ingl%C3%A9s-Web.pdf

Correa, A., Ochoa-Tocachi, B. F., Birkel, C., Ochoa-Sánchez, A., Zogheib, C., Tovar, C., & Buytaert, W. (2020). A concerted research effort to advance the hydrological understanding of tropical páramos. *Hydrological Processes*, *34*(24), 4609–4627. https://doi.org/10.1002/hyp.13904

Crespo, M., Rodríguez Díaz, J. A., García Morillo, J., & McNabola, A. (2019). Pump-as-Turbine Selection Methodology for Energy Recovery in Irrigation Networks: Minimising the Payback Period. *Water*, *11*(1), Article 1. https://doi.org/10.3390/w11010149

Csákvári, E., Molnár, Z., & Halassy, M. (2022). Estimates of regeneration potential in the Pannonian sand region help prioritize ecological restoration interventions. *Communications Biology*, *5*(1). https://doi.org/10.1038/s42003-022-04047-8

Cui, H., Yang, Y., Zhang, X., Dong, L., Yang, Y., Huang, M., He, Y., Lu, X., & Zhen, G. (2022). Nitrogen removal and microbial mechanisms in a novel tubular bioreactor-enhanced floating treatment wetland for the treatment of high nitrate river water. *Water Environment Research*, *94*(8). https://doi.org/10.1002/wer.10767

Culhane, F., Teixeira, H., Nogueira, A. J. A., Borgwardt, F., Trauner, D., Lillebø, A., Piet, G., Kuemmerlen, M., McDonald, H., O'Higgins, T., Barbosa, A. L., van der Wal, J. T., Iglesias-Campos, A., Arevalo-Torres, J., Barbière, J., & Robinson, L. A. (2019). Risk to the supply of ecosystem services across aquatic ecosystems. *Science of The Total Environment*, *660*, 611–621. https://doi.org/10.1016/j.scitotenv.2018.12.346

Danelon, A. F., Augusto, F. G., & Spolador, H. F. S. (2021). Water resource quality effects on water treatment costs: An analysis for the Brazilian case. *Ecological Economics*, *188*, 107134. https://doi.org/10.1016/j.ecolecon.2021.107134

Davies, J.-M., & Mazumder, A. (2003). Health and environmental policy issues in Canada: The role of watershed management in sustaining clean drinking water quality at surface sources. *Journal of Environmental Management*, *68*(3), 273–286. https://doi.org/10.1016/S0301-4797(03)00070-7

De Moraes, K. R., Souza, A. T., Muška, M., Hladík, M., Čtvrtlíková, M., Draštík, V., Kolařík, T., Kučerová, A., Krolová, M., Sajdlová, Z., Šmejkal, M., & Kubečka, J. (2023).

Artificial floating islands: A promising tool to support juvenile fish in lacustrine systems. *Hydrobiologia*, *850*(9), 1969–1984. https://doi.org/10.1007/s10750-023-05204-8

De Sousa, G. C., & Castañeda-Ayarza, J. A. (2022). PESTEL analysis and the macro-environmental factors that influence the development of the electric and hybrid vehicles industry in Brazil. *Case Studies on Transport Policy*, *10*(1), 686–699. https://doi.org/10.1016/j.cstp.2022.01.030

De Stefani, G., Tocchetto, D., Salvato, M., & Borin, M. (2011). Performance of a floating treatment wetland for in-stream water amelioration in NE Italy. *Hydrobiologia*, *674*(1), 157–167.

Delgado-Aguilar, Ma. J., Konold, W., & Schmitt, C. B. (2017). Community mapping of ecosystem services in tropical rainforest of Ecuador. *Ecological Indicators*, *73*, 460–471. https://doi.org/10.1016/j.ecolind.2016.10.020

Den Heijer, C., & Coppens, T. (2023). Paying for green: A scoping review of alternative financing models for nature-based solutions. *Journal of Environmental Management*, *337*, 117754. https://doi.org/10.1016/j.jenvman.2023.117754

Donthu, N., Kumar, S., Mukherjee, D., Pandey, N., & Lim, W. M. (2021). How to conduct a bibliometric analysis: An overview and guidelines. *Journal of Business Research*, *133*, 285–296. https://doi.org/10.1016/j.jbusres.2021.04.070

Echavarria, M. (2002). Financing Watershed Conservation: The FONAG Water Fund in Quito, Ecuador. In *Selling Forest Environmental Services*. Routledge.

Echavarria, M., Cassin, J., & Bento da Rocha, J. (2021). Protecting source waters in Latin America. In *Nature-based Solutions and Water Security* (pp. 215–239). Elsevier. https://doi.org/10.1016/B978-0-12-819871-1.00001-4

Emich, K. J., Kumar, S., Lu, L., Norder, K., & Pandey, N. (2020). Mapping 50 Years of Small Group Research Through Small Group Research. *Small Group Research*, *51*(6), 659–699. https://doi.org/10.1177/1046496420934541

EPMAPS. (2021). *Informe de Gestión: Agua Potable*. Municipio de Quito. https://www.aguaquito.gob.ec/Alojamientos/RRdCC%202021/INFORME%20DE%20GESTI%C3%93N%202021%20EPMAPS.pdf

FAO. (1994). *Water quality for agriculture*. Water Quality Evaluation. https://www.fao.org/3/t0234e/T0234E01.htm

Feller, I. C., Lovelock, C. E., Berger, U., McKee, K. L., Joye, S. B., & Ball, M. C. (2010). Biocomplexity in Mangrove Ecosystems. *Annual Review of Marine Science*, *2*(Volume 2, 2010), 395–417. https://doi.org/10.1146/annurev.marine.010908.163809

Fernández-Cadena, J. C., Andrade, S., Silva-Coello, C. L., & De la Iglesia, R. (2014). Heavy metal concentration in mangrove surface sediments from the north-west coast of South America. *Marine Pollution Bulletin*, *82*(1), 221–226. https://doi.org/10.1016/j.marpolbul.2014.03.016

Fletcher, J., Willby, N., Oliver, D., & Quilliam, R. S. (2024). Engineering aquatic plant community composition on floating treatment wetlands can increase ecosystem multifunctionality. *Environmental Research*, *243*, 117818. https://doi.org/10.1016/j.envres.2023.117818

Floreano, I. X., & De Moraes, L. A. F. (2021). Land use/land cover (LULC) analysis (2009–2019) with Google Earth Engine and 2030 prediction using Markov-CA in the Rondônia State, Brazil. *Environmental Monitoring and Assessment*, *193*(4), 239. https://doi.org/10.1007/s10661-021-09016-y

FONAG. (2019). *Área de Conservación Hídrica Alto Pita*. Avances Alto Pita. https://www.fonag.org.ec/web/avances-alto-pita/

Fonseca, K., Acero Triana, J. S., Ramírez, M., Martínez, W., Ilbay, M., Espitia-Sarmiento, E., & Breuer, L. (2024). Assessing the potential of nature-based solutions as sustainable land and water management strategies in the high tropical Andean páramo ecosystem. *Journal of Environmental Management*, *372*, 123350. https://doi.org/10.1016/j.jenvman.2024.123350

Fonseca, K., & Clairand, M. (2018). *Caracterización preliminar de la calidad del agua del Reservorio del Centro Experimental Académico Salache*.

Fonseca, K., Espitia, E., Breuer, L., & Correa, A. (2022). Using fuzzy cognitive maps to promote nature-based solutions for water quality improvement in developing-country communities. *Journal of Cleaner Production*, *377*, 134246. https://doi.org/10.1016/j.jclepro.2022.134246

Fonseca, K. M., Ruiz Depablos, J. L., Espitia-Sarmiento, E. F., & Llugsha Moreta, N. M. (2020). Artificial Floating Island with Vetiver for Treatment of Arsenic-Contaminated Water: A Real Scale Study in High-Andean Reservoir. *Water*, *12*(11), 3086. https://doi.org/10.3390/w12113086

Fonseca, K., Ruiz, J., Espitia, E., Campaña, E., & Moreno, E. (2021). Phytoremediation of arsenic-contaminated waters by artificial floating island: Literature review. *Revista de La Facultad de Agronomía, Universidad Del Zulia*, *38*(1), 199–215. https://doi.org/10.47280/RevFacAgron(LUZ).v38.n1.010

Foody, G. M. (2020). Explaining the unsuitability of the kappa coefficient in the assessment and comparison of the accuracy of thematic maps obtained by image classification. *Remote Sensing of Environment*, *239*, 111630. https://doi.org/10.1016/j.rse.2019.111630

Forget, G., Carreau, C., Le Coeur, D., & Bernez, I. (2013). Ecological Restoration of Headwaters in a Rural Landscape (Normandy, France): A Passive Approach Taking Hedge Networks into Account for Riparian Tree Recruitment. *Restoration Ecology*, *21*(1), 96–104. https://doi.org/10.1111/j.1526-100X.2012.00868.x

Fu, Y., Zhang, J., Zhang, C., Zang, W., Guo, W., Qian, Z., Liu, L., Zhao, J., & Feng, J. (2018). Payments for Ecosystem Services for watershed water resource allocations. *Journal of Hydrology*, *556*, 689–700. https://doi.org/10.1016/j.jhydrol.2017.11.051

Gaballah, M. S., Ismail, K., Aboagye, D., Ismail, M. M., Sobhi, M., & Stefanakis, A. (2021). Effect of design and operational parameters on nutrients and heavy metal removal in pilot floating treatment wetlands with *Eichhornia Crassipes* treating polluted lake water. *Environmental Science and Pollution Research*, *28*(20), 25664–25678. https://doi.org/10.1007/s11356-021-12442-7

GADR San Pablo. (2015). *Plan de desarrollo y ordenamiento territorial*.

García, V. J., Márquez, C. O., Isenhart, T. M., Rodríguez, M., Crespo, S. D., & Cifuentes, A. G. (2019). Evaluating the conservation state of the páramo ecosystem: An object-based image analysis and CART algorithm approach for central Ecuador. *Heliyon*, *5*(10), e02701. https://doi.org/10.1016/j.heliyon.2019.e02701

García, V. J., Márquez, C. O., Rodríguez, M. V., Orozco, J. J., Aguilar, C. D., & Ríos, A. C. (2020). Páramo Ecosystems in Ecuador's Southern Region: Conservation State and Restoration. *Agronomy*, *10*(12), Article 12. https://doi.org/10.3390/agronomy10121922

Garrett, R. D., Gardner, T. A., Morello, T. F., Marchand, S., Barlow, J., de Blas, D. E., Ferreira, J., Lees, A. C., & Parry, L. (2017). Explaining the persistence of low income and environmentally degrading land uses in the Brazilian Amazon. *Ecology and Society*, *22*(3). https://www.jstor.org/stable/26270164

Gbedemah, S. F. (2023). Eruditing from indigenous adaptation strategies for resilient and sustainable coastal erosion management in southeastern Ghana. *Discover Sustainability*, *4*(1), 12. https://doi.org/10.1007/s43621-023-00123-z

Ghaboulian Zare, S., Alipour, M., Hafezi, M., Stewart, R. A., & Rahman, A. (2022). Examining wind energy deployment pathways in complex macro-economic and political settings using a fuzzy cognitive map-based method. *Energy*, *238*, 121673. https://doi.org/10.1016/j.energy.2021.121673

Giles, M. P., Michelutti, N., Grooms, C., & Smol, J. P. (2018). Long-term limnological changes in the Ecuadorian páramo: Comparing the ecological responses to climate warming of shallow waterbodies versus deep lakes. *Freshwater Biology*, *63*(10), 1316–1325. https://doi.org/10.1111/fwb.13159

Giordano, R., Pluchinotta, I., Pagano, A., Scrieciu, A., & Nanu, F. (2020). Enhancing nature-based solutions acceptance through stakeholders' engagement in co-benefits identification and trade-offs analysis. *Science of The Total Environment*, *713*, 136552. https://doi.org/10.1016/j.scitotenv.2020.136552

Glaser, G. L., Miller, M. C., Healy, S. D., & Shuker, D. M. (2021). The rationality of decisions depends on behavioural context. *Behavioural Processes*, *182*, 104293. https://doi.org/10.1016/j.beproc.2020.104293

Gómez Martín, E., Giordano, R., Pagano, A., van der Keur, P., & Máñez Costa, M. (2020). Using a system thinking approach to assess the contribution of nature based solutions to sustainable development goals. *Science of The Total Environment*, *738*, 139693. https://doi.org/10.1016/j.scitotenv.2020.139693

González-Zeas, D., Rosero-López, D., Muñoz, T., Osorio, R., De Bièvre, B., & Dangles, O. (2022). Making thirsty cities sustainable: A nexus approach for water provisioning in Quito, Ecuador. *Journal of Environmental Management*, *320*, 115880. https://doi.org/10.1016/j.jenvman.2022.115880

Goodwin, G. (2017). The Quest to Bring Land under Social and Political Control: Land Reform Struggles of the Past and Present in Ecuador: Land Reform Struggles in Ecuador. *Journal of Agrarian Change*, *17*(3), 571–593. https://doi.org/10.1111/joac.12181

Gordon, L. J., Finlayson, C. M., & Falkenmark, M. (2010). Managing water in agriculture for food production and other ecosystem services. *Agricultural Water Management*, *97*(4), 512–519. https://doi.org/10.1016/j.agwat.2009.03.017

Gorelick, N., Hancher, M., Dixon, M., Ilyushchenko, S., Thau, D., & Moore, R. (2017). *Google Earth Engine: Planetary-scale geospatial analysis for everyone. Remote Sensing of Environment. 202*, 18–27. https://doi.org/10.1016/j.rse.2017.06.031

Gray, S. A., Gray, S., De Kok, J. L., Helfgott, A. E. R., O'Dwyer, B., Jordan, R., & Nyaki, A. (2015). Using fuzzy cognitive mapping as a participatory approach to analyze change, preferred states, and perceived resilience of social-ecological systems. *Ecology and Society*, *20*(2), art11. https://doi.org/10.5751/ES-07396-200211

Gray, S. A., Zanre, E., & Gray, S. R. J. (2014). Fuzzy Cognitive Maps as Representations of Mental Models and Group Beliefs. In E. I. Papageorgiou (Ed.), *Fuzzy Cognitive Maps* *for Applied Sciences and Engineering* (Vol. 54, pp. 29–48). Springer Berlin Heidelberg. https://doi.org/10.1007/978-3-642-39739-4_2

Grigg, N. S. (2016). Social Aspects of Water Management. In N. S. Grigg (Ed.), *Integrated Water Resource Management: An Interdisciplinary Approach* (pp. 319–338). Palgrave Macmillan UK. https://doi.org/10.1057/978-1-137-57615-6_17

Grizzetti, B., Lanzanova, D., Liquete, C., Reynaud, A., & Cardoso, A. C. (2016). Assessing water ecosystem services for water resource management. *Environmental Science & Policy*, *61*, 194–203. https://doi.org/10.1016/j.envsci.2016.04.008

Groll, M. (2017). The passive river restoration approach as an efficient tool to improve the hydromorphological diversity of rivers – Case study from two river restoration projects in the German lower mountain range. *Geomorphology*, *293*, 69–83. https://doi.org/10.1016/j.geomorph.2017.05.004

Gude, V. G. (2017). Desalination and water reuse to address global water scarcity. *Reviews in Environmental Science and Bio/Technology*, *16*(4), 591–609. https://doi.org/10.1007/s11157-017-9449-7

Guo, Y., Liu, Y., Zeng, G., Hu, X., Li, X., Huang, D., Liu, Y., & Yin, Y. (2014). A restoration-promoting integrated floating bed and its experimental performance in eutrophication remediation. *Journal of Environmental Sciences*, *26*(5), 1090–1098. https://doi.org/10.1016/S1001-0742(13)60500-8

Gutierrez-Gines, M. J., Alizadeh, H., Alderton, E., Ambrose, V., Meister, A., Robinson, B. H., Halford, S., Prosser, J. A., & Horswell, J. (2021). Phytoremediation of microbial contamination in soil by New Zealand native plants. *Applied Soil Ecology*, *167*, 104040. https://doi.org/10.1016/j.apsoil.2021.104040

Gyawali, S., Techato, K., Monprapussorn, S., & Yuangyai, C. (2013). Integrating Land Use and Water Quality for Environmental based Land Use Planning for U-tapao River Basin, Thailand. *Procedia - Social and Behavioral Sciences*, *91*, 556–563. https://doi.org/10.1016/j.sbspro.2013.08.454

Haigh, M. J., & Křeček, J. (1991). Headwater management: Problems and policies. *Land Use Policy*, *8*(3), 171–176. https://doi.org/10.1016/0264-8377(91)90028-H

Hambira, W. L., & Kolawole, O. D. (2021). Botswana stock exchange listed companies and sustainable water demand management. *Physics and Chemistry of the Earth, Parts A/B/C*, *124*, 103078. https://doi.org/10.1016/j.pce.2021.103078

Hanson, H. I., Wickenberg, B., & Alkan Olsson, J. (2020). Working on the boundaries—How do science use and interpret the nature-based solution concept? *Land Use Policy*, *90*, 104302. https://doi.org/10.1016/j.landusepol.2019.104302

Hartmann, J., van der Aa, M., Wuijts, S., de Roda Husman, A. M., & van der Hoek, J. P. (2018). Risk governance of potential emerging risks to drinking water quality: Analysing current practices. *Environmental Science & Policy*, *84*, 97–104. https://doi.org/10.1016/j.envsci.2018.02.015

Hasan, M. M., Mondol Nilay, M. S., Jibon, N. H., & Rahman, R. M. (2023). LULC changes to riverine flooding: A case study on the Jamuna River, Bangladesh using the multilayer perceptron model. *Results in Engineering*, *18*, 101079. https://doi.org/10.1016/j.rineng.2023.101079

Hastie, T., Tibshirani, R., & Friedman, J. (2009). *The Elements of Statistical Learning: Data Mining, Inference, and Prediction, Second Edition* (2nd ed. 2009, Corr. 9th printing 2017 edition). Springer.

Hechavarría Hernández, J. R., Rodríguez, P. B., Fuentes, B. F., & Castell, Y. P. (2018). Social Inclusion: A proposal from the University of Guayaquil to design popular housing with citizen participation. *2018 World Engineering Education Forum - Global Engineering Deans Council (WEEF-GEDC)*, 1–5. https://doi.org/10.1109/WEEF-GEDC.2018.8629778

Herzog, M. H., Francis, G., & Clarke, A. (2019). Correlation. In M. H. Herzog, G. Francis, & A. Clarke (Eds.), *Understanding Statistics and Experimental Design: How to Not Lie with Statistics* (pp. 95–102). Springer International Publishing. https://doi.org/10.1007/978-3-030-03499-3_8

Hoffman, K. N. (2023). Justifying nature-based solutions. *Biology & Philosophy*, *38*(5), 38. https://doi.org/10.1007/s10539-023-09926-w

Hofstede, R., Calles, J., López, V., Polanco, R., Torres, F., Ulloa, J., Vásquez, A., & Cerra, M. (2014). *Los Páramos Andinos ¿Qué sabemos? Estado de conocimiento sobre el impacto del cambio climático en el ecosistema páramo*. UICN. https://portals.iucn.org/library/sites/library/files/documents/2014-025.pdf

Hofstede, R., Mena-Vásconez, P., & Suárez, E. (Eds.). (2023). *Los páramos del Ecuador Pasado, presente y futuro*. USFQ Press. https://libros.usfq.edu.ec/index.php/usfqpress/catalog/view/32/67/177

Hong, H.-Y., & Scardamalia, M. (2014). Community knowledge assessment in a knowledge building environment. *Computers & Education*, *71*, 279–288. https://doi.org/10.1016/j.compedu.2013.09.009

Hoogesteger, J., Boelens, R., & Baud, M. (2016). Territorial pluralism: Water users' multi-scalar struggles against state ordering in Ecuador's highlands. *Water International*, *41*(1), 91–106. https://doi.org/10.1080/02508060.2016.1130910

Hough-Snee, N., Roper, B. B., Wheaton, J. M., Budy, P., & Lokteff, R. L. (2013). Riparian vegetation communities change rapidly following passive restoration at a northern Utah stream. *Ecological Engineering*, *58*, 371–377. https://doi.org/10.1016/j.ecoleng.2013.07.042

Huang, X., Zhao, F., Song, C., Gao, Y., Geng, Z., & Zhuang, P. (2017). Effects of stereoscopic artificial floating wetlands on nekton abundance and biomass in the Yangtze Estuary. *Chemosphere*, *183*, 510–518. https://doi.org/10.1016/j.chemosphere.2017.05.091

Hudson, G., Hart, S., & Verbeek, A. (2023). *Investing in nature-based solutions*. https://www.eib.org/attachments/lucalli/20230095_investing_in_nature_based_solutions_en.pdf

Hurley, T., Sadiq, R., & Mazumder, A. (2012). Adaptation and evaluation of the Canadian Council of Ministers of the Environment Water Quality Index (CCME WQI) for use as an effective tool to characterize drinking source water quality. *Water Research*, *46*(11), 3544–3552. https://doi.org/10.1016/j.watres.2012.03.061

Hwang, J.-I., Li, Z., Andreacchio, N., Ordonez Hinz, F., & Wilson, P. C. (2020). Potential use of floating treatment wetlands established with *Canna flaccida* for removing organic contaminants from surface water. *International Journal of Phytoremediation*, *22*(12), 1304–1312. https://doi.org/10.1080/15226514.2020.1768511

Iacovidou, E., & Zorpas, A. A. (2022). Exploratory research on the adoption of composting for the management of biowaste in the Mediterranean island of Cyprus. *Cleaner and Circular Bioeconomy*, *1*, 100007. https://doi.org/10.1016/j.clcb.2022.100007

Ibarra, D. (2018). *Propuesta para la mejora de la potabilización de agua de la planta de tratamiento loma de Alcoceres de la ciudad de Latacunga, provincia de Cotopaxi* [Universidad Internacional SEK]. https://repositorio.uisek.edu.ec/bitstream/123456789/3173/1/DIANA%20IBARRA%20potabilizaci%C3%B3n%20de%20agua%20alcoceres.pdf

IDEAM. (2010). *Leyenda nacional de coberturas de la tierra. Metodología CORINE Land Cover adaptada para Colombia, escala 1:100.000* (pp. 1–16). IDEAM.

Idris, A., Liew, A., Hung, Y.-T., & Wang, L. (2010). Wetlands for Wastewater Treatment. In *Environmental Bioengineering*. Humana Press. https://doi.org/10.1007/978-1-60327-031-1

Ilbay-Yupa, M., Ilbay, F., Zubieta, R., García-Mora, M., & Chasi, P. (2021). Impacts of Climate Change on the Precipitation and Streamflow Regimes in Equatorial Regions: Guayas River Basin. *Water*, *13*(21), Article 21. https://doi.org/10.3390/w13213138

Ilbay-Yupa, M., Lavado-Casimiro, W., Rau, P., Zubieta, R., & Castillón, F. (2021). Updating regionalization of precipitation in Ecuador. *Theoretical and Applied Climatology*, *143*(3–4), 1513–1528. https://doi.org/10.1007/s00704-020-03476-x

INAMHI. (2017). *Isotermas serie 1981-2010. Información temática del Instituto Nacional de Meteorología e Hidrología. Escala 1:100000. Proyección UTM WGS Zona 17S. https://www.inamhi.gob.ec/geoinformacion-hidrometeorologica/.* http://www.inamhi.gob.ec/gisweb/ISOTERMAS_SERIE_1981_2010/JPEG/ISOTERMAS_SERIE_1981-2010.jpg

INEC. (2019). *Boletín Técnico. Encuesta Nacional de Salud y Nutrición. Indicadores de Salud y Nutrición de la población ecuatorian. Banco de Información.* https://aplicaciones3.ecuadorencifras.gob.ec/BIINEC-war/index.xhtml

INEC. (2022). *Cifras Claves del Censo del Ecuador 2022. Resultados Principales Nacionales. Boletin Nacional Segunda Entrega.* https://inec.censoecuador.gob.ec/public/BoletinNacional_SegundaPublicacion.html#414_Distribuci%C3%B3n_geogr%C3%A1fica

INEN. (2013a). *Guidance on sampling techniques -NTE INEN 2176:2013*. https://www.trabajo.gob.ec/wp-content/uploads/2012/10/NTE-INEN-2176-AGUA.-CALIDAD-DEL-AGUA.-MUESTREO.-T%C3%89CNICAS-DE-MUESTREO.pdf?x42051

INEN. (2013b). *Sampling-Handling and conservation of samples. NTE INEN 2169:2013*. https://www.trabajo.gob.ec/wp-content/uploads/2012/10/NTE-INEN-2169-AGUA.-CALIDAD-DEL-AGUA.-MUESTREO.-MANEJO-Y-CONSERVACI%C3%93N-DE-MUESTRAS.pdf?x42051

InfoMIES. (2022). *Información Estadística. Aseguramiento no contributivo. Consulta de datos interactivos. Pobreza de desigualdad.* https://info.inclusion.gob.ec/

Jähnig, S. C., Brabec, K., Buffagni, A., Erba, S., Lorenz, A. W., Ofenböck, T., Verdonschot, P. F. M., & Hering, D. (2010). A comparative analysis of restoration measures and their effects on hydromorphology and benthic invertebrates in 26 central and southern European rivers. *Journal of Applied Ecology*, *47*(3), 671–680. https://doi.org/10.1111/j.1365-2664.2010.01807.x

Jarosiewicz, P., Fazi, S., & Zalewski, M. (2022). How to boost Ecohydrological Nature-Based Solutions in water quality management. *Ecohydrology & Hydrobiology*, *22*(2), 226–233. https://doi.org/10.1016/j.ecohyd.2021.11.005

Jiménez-Oyola, S., Chavez, E., García-Martínez, M.-J., Ortega, M. F., Bolonio, D., Guzmán-Martínez, F., García-Garizabal, I., & Romero, P. (2021). Probabilistic multi-pathway human health risk assessment due to heavy metal(loid)s in a traditional gold mining area in Ecuador. *Ecotoxicology and Environmental Safety*, *224*, 112629. https://doi.org/10.1016/j.ecoenv.2021.112629

Joseph, L., Jun, B.-M., Flora, J. R. V., Park, C. M., & Yoon, Y. (2019). Removal of heavy metals from water sources in the developing world using low-cost materials: A review. *Chemosphere*, *229*, 142–159. https://doi.org/10.1016/j.chemosphere.2019.04.198

Joslin, A. J., & Jepson, W. E. (2018). Territory and authority of water fund payments for ecosystem services in Ecuador's Andes. *Geoforum*, *91*, 10–20. https://doi.org/10.1016/j.geoforum.2018.02.016

Jourdain, D., Namakando, N., Mungatana, E., Mirzabaev, A., & Njiraini, G. (2023). Revealing salient aquatic ecosystem services bundles in the Olifants River catchment, South Africa. *Wetlands Ecology and Management*, *31*(2), 267–286. https://doi.org/10.1007/s11273-023-09915-6

Junker, B., & Buchecker, M. (2008). Aesthetic preferences versus ecological objectives in river restorations. *Landscape and Urban Planning*, *85*(3), 141–154. https://doi.org/10.1016/j.landurbplan.2007.11.002

Kang, S., Kroeger, T., Shemie, D., Echavarria, M., Montalvo, T., Bremer, L. L., Bennett, G., Barreto, S. R., Bracale, H., Calero, C., Cardenas, A., Cardona, J., Cardozo García, I. C., Crespo, R., da Rocha, J. B., de Bièvre, B., Díaz González, J. D., Estévez, W., Hernandez, D., … Zhang, H. (2023). Investing in nature-based solutions: Cost profiles of collective-action watershed investment programs. *Ecosystem Services*, *59*, 101507. https://doi.org/10.1016/j.ecoser.2022.101507

Kansongue, N., Njuguna, J., & Vertigans, S. (2023). A PESTEL and SWOT impact analysis on renewable energy development in Togo. *Frontiers in Sustainability*, *3*. https://doi.org/10.3389/frsus.2022.990173

Karstens, S., Nazzari, C., Bâlon, C., Bielecka, M., Grigaitis, Ž., Schumacher, J., Stybel, N., & Razinkovas-Baziukas, A. (2018). Floating wetlands for nutrient removal in eutrophicated coastal lagoons: Decision support for site selection and permit process. *Marine Policy*, *97*, 51–60. https://doi.org/10.1016/j.marpol.2018.08.030

Kauffman, C. M. (2014). Financing watershed conservation: Lessons from Ecuador's evolving water trust funds. *Agricultural Water Management*, *145*, 39–49. https://doi.org/10.1016/j.agwat.2013.09.013

Kaur, M., Das, S. K., & Sarma, K. (2023). Water quality assessment of Tal Chhapar Wildlife Sanctuary using water quality index (CCME WQI). *Acta Ecologica Sinica*, *43*(1), 82–88. https://doi.org/10.1016/j.chnaes.2021.09.017

Keese, J. R. (1998). International NGOs and Land Use Change in a Southern Highland Region of Ecuador. *Human Ecology*, *26*(3), 451–468. https://doi.org/10.1023/A:1018708300053

Keesstra, S., Nunes, J., Novara, A., Finger, D., Avelar, D., Kalantari, Z., & Cerdà, A. (2018). The superior effect of nature based solutions in land management for enhancing ecosystem services. *Science of The Total Environment*, *610–611*, 997–1009. https://doi.org/10.1016/j.scitotenv.2017.08.077

Kenter, J. O., Hyde, T., Christie, M., & Fazey, I. (2011). The importance of deliberation in valuing ecosystem services in developing countries—Evidence from the Solomon Islands. *Global Environmental Change*, *21*(2), 505–521. https://doi.org/10.1016/j.gloenvcha.2011.01.001

Khatibi, F. S., Dedekorkut-Howes, A., Howes, M., & Torabi, E. (2021). Can public awareness, knowledge and engagement improve climate change adaptation policies? *Discover Sustainability*, *2*(1), 18. https://doi.org/10.1007/s43621-021-00024-z

Khorchani, M., Nadal-Romero, E., Lasanta, T., & Tague, C. (2022). Carbon sequestration and water yield tradeoffs following restoration of abandoned agricultural lands in Mediterranean mountains. *Environmental Research*, *207*, 112203. https://doi.org/10.1016/j.envres.2021.112203

Kim, K. E. (1996). Adaptive majority filtering for contextual classification of remote sensing data. *International Journal of Remote Sensing*, *17*(5), 1083–1087. https://doi.org/10.1080/01431169608949070

Kleemann, J., Koo, H., Hensen, I., Mendieta-Leiva, G., Kahnt, B., Kurze, C., Inclan, D. J., Cuenca, P., Noh, J. K., Hoffmann, M. H., Factos, A., Lehnert, M., Lozano, P., &

Fürst, C. (2022). Priorities of action and research for the protection of biodiversity and ecosystem services in continental Ecuador. *Biological Conservation*, *265*, 109404. https://doi.org/10.1016/j.biocon.2021.109404

Kobayashi, N., Tani, H., Wang, X., & Sonobe, R. (2020). Crop classification using spectral indices derived from Sentinel-2A imagery. *Journal of Information and Telecommunication*, *4*(1), 67–90. https://doi.org/10.1080/24751839.2019.1694765

Kosko, B. (1986). *Fuzzy cognitive maps*. International Journal of Man-Machine Studies. http://dx.doi.org/10.1016/ S0020-7373(86)80040-2

Kottek, M., Grieser, J., Beck, C., Rudolf, B., & Rubel, F. (2006). World Map of the Köppen-Geiger climate classification updated. *Meteorologische Zeitschrift*, 259–263. https://doi.org/10.1127/0941-2948/2006/0130

Kuhn, C., de Matos Valerio, A., Ward, N., Loken, L., Sawakuchi, H. O., Kampel, M., Richey, J., Stadler, P., Crawford, J., Striegl, R., Vermote, E., Pahlevan, N., & Butman, D. (2019). Performance of Landsat-8 and Sentinel-2 surface reflectance products for river remote sensing retrievals of chlorophyll-a and turbidity. *Remote Sensing of Environment*, *224*, 104–118. https://doi.org/10.1016/j.rse.2019.01.023

Kusin, F. M., Hasan, S. N. M. S., Nordin, N. A., Mohamat-Yusuff, F., & Ibrahim, Z. Z. (2019). FLOATING VETIVER ISLAND (FVI) AND IMPLICATION FOR TREATMENT SYSTEM DESIGN OF POLLUTED RUNNING WATER. *Applied Ecology and Environmental Research*, *17*(1), 497–510. https://doi.org/10.15666/aeer/1701_497510

La Hora. (2018). *Minería es vigilada en Latacunga*. https://www.lahora.com.ec/secciones/mineria-es-vigilada-en-latacunga/

Ladislas, S., Gérente, C., Chazarenc, F., Brisson, J., & Andrès, Y. (2015). Floating treatment wetlands for heavy metal removal in highway stormwater ponds. *Ecological Engineering*, *80*, 85–91. https://doi.org/10.1016/j.ecoleng.2014.09.115

Lebron, I., Robinson, D. A., Oatham, M., & Wuddivira, M. N. (2012). Soil water repellency and pH soil change under tropical pine plantations compared with native tropical forest. *Journal of Hydrology*, *414–415*, 194–200. https://doi.org/10.1016/j.jhydrol.2011.10.031

Levac, D., Colquhoun, H., & O’Brien, K. K. (2010). Scoping studies: Advancing the methodology. *Implementation Science*, *5*(1), 69. https://doi.org/10.1186/1748-5908-5-69

Li, D., Zuo, Q., Wu, Q., Li, Q., & Ma, J. (2021). Achieving the tradeoffs between pollutant discharge and economic benefit of the Henan section of the South-to-North Water Diversion Project through water resources-environment system management under uncertainty. *Journal of Cleaner Production*, *321*, 128857. https://doi.org/10.1016/j.jclepro.2021.128857

Li, S., Gu, S., Liu, W., Han, H., & Zhang, Q. (2008). Water quality in relation to land use and land cover in the upper Han River Basin, China. *CATENA*, *75*(2), 216–222. https://doi.org/10.1016/j.catena.2008.06.005

Liquete, C., Udias, A., Conte, G., Grizzetti, B., & Masi, F. (2016). Integrated valuation of a nature-based solution for water pollution control. Highlighting hidden benefits. *Ecosystem Services*, *22*, 392–401. https://doi.org/10.1016/j.ecoser.2016.09.011

Liu, J., Liu, J., Zhang, R., Zou, Y., Wang, H., & Zhang, Z. (2014). Impacts of aquatic macrophytes configuration modes on water quality. *Water Science & Technology*, *69*(2), 253–261. https://doi.org/10.2166/wst.2013.573

Lopolito, A., Falcone, P. M., Morone, P., & Sica, E. (2020). A Combined method to model policy interventions for local communities based on people knowledge. *MethodsX*, *7*, 100877. https://doi.org/10.1016/j.mex.2020.100877

Lu, C., & Tian, H. (2017). Global nitrogen and phosphorus fertilizer use for agriculture production in the past half century: Shifted hot spots and nutrient imbalance. *Earth System Science Data*, *9*(1), 181–192. https://doi.org/10.5194/essd-9-181-2017

Lu, H.-L., Ku, C.-R., & Chang, Y.-H. (2015). Water quality improvement with artificial floating islands. *Ecological Engineering*, *74*, 371–375. https://doi.org/10.1016/j.ecoleng.2014.11.013

Lynam, T., De Jong, W., Sheil, D., Kusumanto, T., & Evans, K. (2007). A Review of Tools for Incorporating Community Knowledge, Preferences, and Values into Decision Making in Natural Resources Management. *Ecology and Society*, *12*(1), art5. https://doi.org/10.5751/ES-01987-120105

MAE, (Ministerio del Ambiente del Ecuador). (2007). *Plan de manejo del Área Nacional de Recreación el Boliche (ANRB).* SIMBIOE. http://suiadoc.ambiente.gob.ec/documents/10179/242256/32+PLAN+DE+MANEJO+BOLICHE+ANR+El+Boliche.pdf/4a553f3b-8563-4f93-b8fd-2da6ab50fc1b

MAE, (Ministerio del Ambiente del Ecuador). (2015). *Texto Unificado de Legislacion Secundaria de Medio Ambiente*. https://www.gob.ec/sites/default/files/regulations/2018-09/Documento_Registro-Oficial-No-387-04-noviembre-2015_0.pdf

MAG. (2019). *Objetos geográficos y productos cartográficos agropecuarios del Ministerio de Agricultura y Ganaderia. Geoportal del Agro Ecuatoriano. Volumen IV*. http://geoportal.agricultura.gob.ec/

Marselina, M., Wibowo, F., & Mushfiroh, A. (2022). Water quality index assessment methods for surface water: A case study of the Citarum River in Indonesia. *Heliyon*, *8*(7), e09848. https://doi.org/10.1016/j.heliyon.2022.e09848

Martelo, J., & Borrero, J. A. L. (2012). Macrófitas flotantes en el tratamiento de aguas residuales: Una revisión del estado del arte. *Ingeniería y Ciencia*, *8*(15), 221–243.

Martínez, C. (2006). *Atlas de Cotopaxi* (Atlas Socioambiental de Cotopaxi). EcoCiencia. https://books.google.de/books/about/Atlas_socioambiental_de_Cotopaxi.html?id=1JOZMwEACAAJ&redir_esc=y

Masi, F., Rizzo, A., & Regelsberger, M. (2018). The role of constructed wetlands in a new circular economy, resource oriented, and ecosystem services paradigm. *Journal of Environmental Management*, *216*, 275–284. https://doi.org/10.1016/j.jenvman.2017.11.086

Maúre, E. de R., Terauchi, G., Ishizaka, J., Clinton, N., & DeWitt, M. (2021). Globally consistent assessment of coastal eutrophication. *Nature Communications*, *12*(1), Article 1. https://doi.org/10.1038/s41467-021-26391-9

Maurice, L., López, F., Becerra, S., Jamhoury, H., Le Menach, K., Dévier, M.-H., Budzinski, H., Prunier, J., Juteau-Martineau, G., Ochoa-Herrera, V., Quiroga, D., & Schreck, E. (2019). Drinking water quality in areas impacted by oil activities in Ecuador: Associated health risks and social perception of human exposure. *Science of The Total Environment*, *690*, 1203–1217. https://doi.org/10.1016/j.scitotenv.2019.07.089

Maxwell, S. K., & Sylvester, K. M. (2012). Identification of "ever-cropped" land (1984–2010) using Landsat annual maximum NDVI image composites: Southwestern Kansas case study. *Remote Sensing of Environment*, *121*, 186–195. https://doi.org/10.1016/j.rse.2012.01.022

Mazumder, D. N. G. (2008). Chronic arsenic toxicity & human health. *INDIAN J MED RES*, 12.

McIver, J., & Starr, L. (2001). Restoration of degraded lands in the interior Columbia River basin: Passive vs. Active approaches. *Forest Ecology and Management*, *153*(1–3), 15–28. https://doi.org/10.1016/S0378-1127(01)00451-0

Mercado, G., Wild, T., Hernandez-Garcia, J., Baptista, M. D., van Lierop, M., Bina, O., Inch, A., Ode Sang, Å., Buijs, A., Dobbs, C., Vásquez, A., van der Jagt, A., Salbitano, F., Falanga, R., Amaya-Espinel, J. D., de Matos Pereira, M., & Randrup, T. B. (2024). Supporting Nature-Based Solutions via Nature-Based Thinking across European and Latin American cities. *Ambio*, *53*(1), 79–94. https://doi.org/10.1007/s13280-023-01920-6

Metcalf, J. L., Cooper, A., & Wheeler, J. C. (2014). Alpaca and Llama: Domestication. In C. Smith (Ed.), *Encyclopedia of Global Archaeology* (pp. 145–147). Springer. https://doi.org/10.1007/978-1-4419-0465-2_2212

Ministerio del Ambiente. (2018). *Actualización del Plan de Manejo del Parque Nacional Cajas*. https://www.ambiente.gob.ec/wp-content/uploads/downloads/2018/03/ACUERDO-001-ANEXO-PAQUE-NACIONAL-CAJAS.pdf

Mitra, S., Chakraborty, A. J., Tareq, A. M., Emran, T. B., Nainu, F., Khusro, A., Idris, A. M., Khandaker, M. U., Osman, H., Alhumaydhi, F. A., & Simal-Gandara, J. (2022). Impact of heavy metals on the environment and human health: Novel therapeutic insights to counter the toxicity. *Journal of King Saud University - Science*, *34*(3), 101865. https://doi.org/10.1016/j.jksus.2022.101865

Molina-Vera, A., Pozo, M., & Serrano, J. C. (2018). *Agua, saneamiento e higiene: Medición de los ODS en Ecuador*. . Instituto Nacional de Estadística y Censos y UNICEF (INEC-UNICEF). https://www.ecuadorencifras.gob.ec/documentos/web-inec/Bibliotecas/Libros/AGUA,_SANEAMIENTO_e_HIGIENE.pdf

Molle, F. (2008). *Nirvana Concepts, Narratives and Policy Models: Insights from the Water Sector*. *1*(1).

Moosavi, S., Browne, G. R., & Bush, J. (2021). Perceptions of nature-based solutions for Urban Water challenges: Insights from Australian researchers and practitioners. *Urban Forestry & Urban Greening*, *57*, 126937. https://doi.org/10.1016/j.ufug.2020.126937

Morales-Simfors, N., Bundschuh, J., Herath, I., Inguaggiato, C., Caselli, A. T., Tapia, J., Choquehuayta, F. E. A., Armienta, M. A., Ormachea, M., Joseph, E., & López, D. L. (2019). Arsenic in Latin America: A critical overview on the geochemistry of arsenic originating from geothermal features and volcanic emissions for solving its environmental consequences. *Science of The Total Environment*, 135564. https://doi.org/10.1016/j.scitotenv.2019.135564

Morone, P., Yilan, G., & Imbert, E. (2021). Using fuzzy cognitive maps to identify better policy strategies to valorize organic waste flows: An Italian case study. *Journal of Cleaner Production*, *319*, 128722. https://doi.org/10.1016/j.jclepro.2021.128722

Mosquera, G. M., Hofstede, R., Bremer, L. L., Asbjornsen, H., Carabajo-Hidalgo, A., Célleri, R., Crespo, P., Esquivel-Hernández, G., Feyen, J., Manosalvas, R., Marín, F., Mena-Vásconez, P., Montenegro-Díaz, P., Ochoa-Sánchez, A., Pesántez, J., Riveros-Iregui,

D. A., & Suárez, E. (2023). Frontiers in páramo water resources research: A multidisciplinary assessment. *Science of The Total Environment*, *892*, 164373. https://doi.org/10.1016/j.scitotenv.2023.164373

Mourhir, A. (2021). Scoping review of the potentials of fuzzy cognitive maps as a modeling approach for integrated environmental assessment and management. *Environmental Modelling & Software*, *135*, 104891. https://doi.org/10.1016/j.envsoft.2020.104891

Muller, I., Delisle, M., Ollitrault, M., & Bernez, I. (2016). Responses of riparian plant communities and water quality after 8 years of passive ecological restoration using a BACI design. *Hydrobiologia*, *781*(1), 67–79. https://doi.org/10.1007/s10750-015-2349-3

Mulligan, M., Rubiano, J., Hyman, G., White, D., Garcia, J., Saravia, M., Gabriel Leon, J., Selvaraj, J. J., Guttierez, T., & Leonardo Saenz-Cruz, L. (2010). The Andes basins: Biophysical and developmental diversity in a climate of change. *Water International*, *35*(5), 472–492. https://doi.org/10.1080/02508060.2010.516330

Nahuelhual, L., Saavedra, G., Henríquez, F., Benra, F., Vergara, X., Perugache, C., & Hasen, F. (2018). Opportunities and limits to ecosystem services governance in developing countries and indigenous territories: The case of water supply in Southern Chile. *Environmental Science & Policy*, *86*, 11–18. https://doi.org/10.1016/j.envsci.2018.04.012

Nare, L., Odiyo, J., Francis, J., & Potgieter, N. (2011). Framework for effective community participation in water quality management in Luvuvhu Catchment of South Africa. *Physics and Chemistry of the Earth*, *36*(14–15), 1063–1070. https://doi.org/10.1016/j.pce.2011.08.006

Negrete, J., Varela, J., Heras, E., & Alcívar, M. (2019). *ESTRATEGIAS IMPLEMENTADAS PARA LA RECUPERACIÓN DEL ESTERO PALANQUEADO, RAMAL INTERIOR DE LA RESERVA DE PRODUCCIÓN DE FAUNA MANGLARES EL SALADO*. 13.

Nikas, A., Ntanos, E., & Doukas, H. (2019). A semi-quantitative modelling application for assessing energy efficiency strategies. *Applied Soft Computing*, *76*, 140–155. https://doi.org/10.1016/j.asoc.2018.12.015

Nuruzzaman, M., Anwar, A. H. M. F., Sarukkalige, R., & Sarker, D. C. (2021). Review of hydraulics of Floating Treatment Islands retrofitted in waterbodies receiving stormwater. *Science of The Total Environment*, *801*, 149526. https://doi.org/10.1016/j.scitotenv.2021.149526

Nyaki, A., Gray, S. A., Lepczyk, C. A., Skibins, J. C., & Rentsch, D. (2014). Local-Scale Dynamics and Local Drivers of Bushmeat Trade: Participatory Modeling in Conservation. *Conservation Biology*, *28*(5), 1403–1414. https://doi.org/10.1111/cobi.12316

Nyiwul, L. (2021). Innovation and adaptation to climate change: Evidence from the water sector in Africa. *Journal of Cleaner Production*, *298*, 126859. https://doi.org/10.1016/j.jclepro.2021.126859

O’Brien, K. K., Colquhoun, H., Levac, D., Baxter, L., Tricco, A. C., Straus, S., Wickerson, L., Nayar, A., Moher, D., & O’Malley, L. (2016). Advancing scoping study methodology: A web-based survey and consultation of perceptions on terminology, definition and methodological steps. *BMC Health Services Research*, *16*(1), 305. https://doi.org/10.1186/s12913-016-1579-z

OECD. (2018). OECD Water Governance Indicator Framework. In OECD, *Implementing the OECD Principles on Water Governance* (pp. 49–105). OECD. https://doi.org/10.1787/9789264292659-5-en

Oladipo, J. O., Akinwumiju, A. S., Aboyeji, O. S., & Adelodun, A. A. (2021). Comparison between fuzzy logic and water quality index methods: A case of water quality assessment in Ikare community, Southwestern Nigeria. *Environmental Challenges*, *3*, 100038. https://doi.org/10.1016/j.envc.2021.100038

Olguín, E., Sánchez-Galván, G., Melo, F., Hernández, V., & González-Portela, R. (2017). Long-term assessment at field scale of Floating Treatment Wetlands for improvement of water quality and provision of ecosystem services in a eutrophic urban pond. *Science of The Total Environment*, *584*, 561–571. https://doi.org/10.1016/j.scitotenv.2017.01.072

Oña, I. M., Guasti, R., & Tocte, S. G. N. (2021). *PDOT de la parroquia de Canchagua*. 296.

ONU. (2018). *Informe de la ONU sobre el desarrollo de los recursos hídricos en el mundo 2018, Soluciones basadas en la naturaleza para el agua*. http://www.unesco.org/new/es/natural-sciences/environment/water/wwap/wwdr/2018-nature-based-solutions/

Oral, H. V., Carvalho, P., Gajewska, M., Ursino, N., Masi, F., Hullebusch, E. D. van, Kazak, J. K., Exposito, A., Cipolletta, G., Andersen, T. R., Finger, D. C., Simperler, L., Regelsberger, M., Rous, V., Radinja, M., Buttiglieri, G., Krzeminski, P., Rizzo, A., Dehghanian, K., … Zimmermann, M. (2020). A review of nature-based solutions for urban water management in European circular cities: A critical assessment based on case studies and literature. *Blue-Green Systems*, *2*(1), 112–136. https://doi.org/10.2166/bgs.2020.932

Özesmi, U., & Özesmi, S. L. (2004). Ecological models based on people’s knowledge: A multi-step fuzzy cognitive mapping approach. *Ecological Modelling*, *176*(1–2), 43–64. https://doi.org/10.1016/j.ecolmodel.2003.10.027

Pagano, A., Pluchinotta, I., Pengal, P., Cokan, B., & Giordano, R. (2019). Engaging stakeholders in the assessment of NBS effectiveness in flood risk reduction: A participatory System Dynamics Model for benefits and co-benefits evaluation. *Science of The Total Environment*, *690*, 543–555. https://doi.org/10.1016/j.scitotenv.2019.07.059

Partridge, T. (2016). Water Justice and Food Sovereignty in Cotopaxi, Ecuador. *Environmental Justice*, *9*(2), 49–52. https://doi.org/10.1089/env.2016.0003

Patiño, S., Hernández, Y., Plata, C., Domínguez, I., Daza, M., Oviedo-Ocaña, R., Buytaert, W., & Ochoa-Tocachi, B. F. (2021). Influence of land use on hydro-physical soil properties of Andean páramos and its effect on streamflow buffering. *CATENA*, *202*, 105227. https://doi.org/10.1016/j.catena.2021.105227

Pazmiño Manrique, P., Barragán, J. M., & García Sanabria, J. (2018). Progress on coastal management in ecuador (2007–2017). *Environmental Science & Policy*, *90*, 135–147. https://doi.org/10.1016/j.envsci.2018.09.016

Pila, P. V. (2018). Gestión del agua en Azuay: Base de la organización rural y la trascendencia hacia la incidencia nacional. *Eutopía, Revista de Desarrollo Económico Territorial*, *13*, 71–88. https://doi.org/10.17141/eutopia.13.2018.3288

Pilon-Smits, E. (2005). Phytoremediation. *Annual Review of Plant Biology*, *56*(1), 15–39. https://doi.org/10.1146/annurev.arplant.56.032604.144214

Porras, G., Stringer, L., & Quinn, C. (2018). Unravelling Stakeholder Perceptions to Enable Adaptive Water Governance in Dryland Systems. *Water Resources Management*, *32*(10). https://doi.org/10.1007/s11269-018-1991-8

Possantti, I., & Marques, G. (2022). A modelling framework for nature-based solutions expansion planning considering the benefits to downstream urban water users. *Environmental Modelling & Software*, *152*, 105381. https://doi.org/10.1016/j.envsoft.2022.105381

Prach, K., & Hobbs, R. J. (2008). Spontaneous Succession versus Technical Reclamation in the Restoration of Disturbed Sites. *Restoration Ecology*, *16*(3), 363–366. https://doi.org/10.1111/j.1526-100X.2008.00412.x

Prashant, & Billore, S. K. (2020). Macroinvertebrates associated with artificial floating islands installed in River Kshipra for water quality improvement. *Water Science and Technology*, *81*(6), 1242–1249. https://doi.org/10.2166/wst.2020.219

Prion, S., & Haerling, K. A. (2014). Making Sense of Methods and Measurement: Pearson Product-Moment Correlation Coefficient. *Clinical Simulation In Nursing*, *10*(11), 587–588. https://doi.org/10.1016/j.ecns.2014.07.010

Pueblo Kichwa de Rukullakta (PKR), A. (2018). *PLAN DE COMANEJO DE LOS RECURSOS NATURALES DEL PUEBLO KICHWA DE RUKULLAKTA (PKR), CONFORMADO POR 17 COMUNIDADES, EN EL BOSQUE PROTECTOR CERRO SUMACO, CUENCA ALTA DEL RÍO SUNO DE LA PROVINCIA DE NAPO.* 155.

Pulford, E., Polidoro, B. A., & Nation, M. (2017). Understanding the relationships between water quality, recreational fishing practices, and human health in Phoenix, Arizona.

Journal of Environmental Management, *199*, 242–250. https://doi.org/10.1016/j.jenvman.2017.05.046

Qian, S., Hou, R., Yuan, R., Zhou, B., Chen, Z., & Chen, H. (2022). Removal of Escherichia coli from domestic sewage using biological sand filters: Reduction effect and microbial community analysis. *Environmental Research*, *209*, 112908. https://doi.org/10.1016/j.envres.2022.112908

Quishpe, S. (2018). *Identificación de especies de diatomeas epilíticas asociadas al grado de eutrofización del río cutuchi en la provincia de Cotopaxi, periodo octubre a marzo 2018* [Universidad Técnica de Cotopaxi]. http://repositorio.utc.edu.ec/handle/27000/8429

Rahm, B. G., Vedachalam, S., Shen, J., Woodbury, P. B., & Riha, S. J. (2013). A watershed-scale goals approach to assessing and funding wastewater infrastructure. *Journal of Environmental Management*, *129*, 124–133. https://doi.org/10.1016/j.jenvman.2013.06.053

Rey-Romero, D. C., Domínguez, I., & Oviedo-Ocaña, E. R. (2022). Effect of agricultural activities on surface water quality from páramo ecosystems. *Environmental Science and Pollution Research*, *29*(55), 83169–83190. https://doi.org/10.1007/s11356-022-21709-6

Richardson, J. S., & Danehy, R. J. (2007). A Synthesis of the Ecology of Headwater Streams and their Riparian Zones in Temperate Forests. *Forest Science*, *53*(2), 131–147. https://doi.org/10.1093/forestscience/53.2.131

Rode, J. (2021). When payments for ecosystem conservation stop. *Nature Sustainability*, *5*(1), 15–16. https://doi.org/10.1038/s41893-021-00812-5

Rojas, C. (2020). *Evaluación de las tendencias de la contaminación del recurso hídrico de la parte alta de la microcuenca del río Cutuchi, en la provincia de Cotopaxi, periodo 2019-2020.* [Universidad Técnica de Cotopaxi]. http://repositorio.utc.edu.ec/handle/27000/7072

Rolfe, J., & Harvey, S. (2017). Heterogeneity in practice adoption to reduce water quality impacts from sugarcane production in Queensland. *Journal of Rural Studies*, *54*, 276–287. https://doi.org/10.1016/j.jrurstud.2017.06.021

Rudel, T. K., & Richards, S. (1990). Urbanization, roads, and rural population change in the Ecuadorian Andes. *Studies In Comparative International Development*, *25*(3), 73–89. https://doi.org/10.1007/BF02687180

Salmoral, G., Khatun, K., Llive, F., & Lopez, C. M. (2018). Agricultural development in Ecuador: A compromise between water and food security? *Journal of Cleaner Production*, *202*, 779–791. https://doi.org/10.1016/j.jclepro.2018.07.308

Samal, A., Kar, S., Bhattacharya, P., & Santra, S. (2011). Human exposure to arsenic through foodstuffs cultivated using arsenic contaminated groundwater in areas of West Bengal, India. *Journal of Environmental Science and Health. Part A, Toxic/Hazardous Substances & Environmental Engineering*, *46*, 1259–1265. https://doi.org/10.1080/10934529.2011.598810

Samal, K. (2019). Ecological floating bed (EFB) for decontamination of polluted water bodies_ Design, mechanism and performance. *Journal of Environmental Management*, 13.

San Llorente Capdevila, A., Kokimova, A., Sinha Ray, S., Avellán, T., Kim, J., & Kirschke, S. (2020). Success factors for citizen science projects in water quality monitoring. *Science of The Total Environment*, *728*, 137843. https://doi.org/10.1016/j.scitotenv.2020.137843

Sansa, M., Badreddine, A., & Romdhane, T. B. (2021). Chapter 18—Sustainable design based on LCA and operations management methods:SWOT, PESTEL, and 7S. In J. Ren (Ed.), *Methods in Sustainability Science* (pp. 345–364). Elsevier. https://doi.org/10.1016/B978-0-12-823987-2.00019-2

Santoro, S., Pluchinotta, I., Pagano, A., Pengal, P., Cokan, B., & Giordano, R. (2019). Assessing stakeholders' risk perception to promote Nature Based Solutions as flood protection strategies: The case of the Glinščica river (Slovenia). *Science of The Total Environment*, *655*, 188–201. https://doi.org/10.1016/j.scitotenv.2018.11.116

Sapkota, R. P., Stahl, P. D., & Rijal, K. (2018). Restoration governance: An integrated approach towards sustainably restoring degraded ecosystems. *Environmental Development*, *27*, 83–94. https://doi.org/10.1016/j.envdev.2018.07.001

Sarkar, S. K., Saha, M., Takada, H., Bhattacharya, A., Mishra, P., & Bhattacharya, B. (2007). Water quality management in the lower stretch of the river Ganges, east coast of India: An approach through environmental education. *Journal of Cleaner Production*, *15*(16), 1559–1567. https://doi.org/10.1016/j.jclepro.2006.07.030

Saud, M., Guo, P., Haq, I. ul, Pan, G., & Khan, A. (2019). Do government expenditure and financial development impede environmental degradation in Venezuela? *PLoS ONE*, *14*(1), e0210255. https://doi.org/10.1371/journal.pone.0210255

Schiavon, E., Taramelli, A., & Tornato, A. (2021). Modelling stakeholder perceptions to assess Green Infrastructures potential in agriculture through fuzzy logic: A tool for participatory governance. *Environmental Development*, 100671. https://doi.org/10.1016/j.envdev.2021.100671

SENAGUA. (2021). *Banco Nacional de Autorizaciones*. Https://docs.google.com/spreadsheets/d/18Gmj0IB5tG0ecS4AU9F3pyam7T5xn8JJ/edit#gid=450715876.

Shah, N. W., Baillie, B. R., Bishop, K., Ferraz, S., Högbom, L., & Nettles, J. (2022). The effects of forest management on water quality. *Forest Ecology and Management*, *522*, 120397. https://doi.org/10.1016/j.foreco.2022.120397

Shahgedanova, M., Adler, C., Gebrekirstos, A., Grau, H. R., Huggel, C., Marchant, R., Pepin, N., Vanacker, V., Viviroli, D., & Vuille, M. (2021). Mountain Observatories: Status and Prospects for Enhancing and Connecting a Global Community. *Mountain Research and Development*, *41*(2). https://doi.org/10.1659/MRD-JOURNAL-D-20-00054.1

Shahvi, S., Mellander, P.-E., Jordan, P., & Fenton, O. (2021). A Fuzzy Cognitive Map method for integrated and participatory water governance and indicators affecting drinking water supplies. *Science of The Total Environment*, *750*, 142193. https://doi.org/10.1016/j.scitotenv.2020.142193

Sharma, R., Vymazal, J., & Malaviya, P. (2021). Application of floating treatment wetlands for stormwater runoff: A critical review of the recent developments with emphasis on heavy metals and nutrient removal. *Science of The Total Environment*, *777*, 146044. https://doi.org/10.1016/j.scitotenv.2021.146044

Sheldon, F., Leigh, C., Neilan, W., Newham, M., Polson, C., & Hadwen, W. (2019). Chapter 11 - Urbanization: Hydrology, Water Quality, and Influences on Ecosystem Health. In A. K. Sharma, T. Gardner, & D. Begbie (Eds.), *Approaches to Water Sensitive Urban Design* (pp. 229–248). Woodhead Publishing. https://doi.org/10.1016/B978-0-12-812843-5.00011-3

Sippi, S., & Parmar, D. (2024). Water quality simulation under river restoration measures for the Delhi stretch of river Yamuna, India. *Environment, Development and Sustainability*, *26*(3), 5891–5914. https://doi.org/10.1007/s10668-023-02943-5

Skrydstrup, J., Madsen, H., Lowe, R., Gregersen, I., Pedersen, A., & Arnbjerg-Nielsen, K. (2020). Incorporating objectives of stakeholders in strategic planning of urban water management. *Urban Water Journal*, *17*(2), 87–99. https://doi.org/10.1080/1573062X.2020.1748204

SNAP. (2015). *Cotopaxi National Park | Sistema Nacional de Áreas Protegidas del Ecuador*. http://areasprotegidas.ambiente.gob.ec/en/areas-protegidas/cotopaxi-national-park

Solana-Gutiérrez, J., Rincón, G., Alonso, C., & García-de-Jalón, D. (2017). Using fuzzy cognitive maps for predicting river management responses: A case study of the Esla River basin, Spain. *Ecological Modelling*, *360*, 260–269. https://doi.org/10.1016/j.ecolmodel.2017.07.010

Solo De Zaldívar, V. B. (2015). Tempest in the Andes? Part 1: Agrarian Reform and Peasant Differentiation in Cotopaxi (Ecuador): Agrarian Reform and Peasant Differentiation in Cotopaxi (Ecuador). *Journal of Agrarian Change*, *15*(1), 89–115. https://doi.org/10.1111/joac.12072

Some, S., Mondal, R., Mitra, D., Jain, D., Verma, D., & Das, S. (2021). Microbial pollution of water with special reference to coliform bacteria and their nexus with environment. *Energy Nexus*, *1*, 100008. https://doi.org/10.1016/j.nexus.2021.100008

Sonobe, R., Yamaya, Y., Tani, H., Wang, X., Kobayashi, N., & Mochizuki, K. (2018). Crop classification from Sentinel-2-derived vegetation indices using ensemble learning. *Journal of Applied Remote Sensing*, *12*(02), 1. https://doi.org/10.1117/1.JRS.12.026019

Souliotis, I., & Voulvoulis, N. (2022). Operationalising nature-based solutions for the design of water management interventions. *Nature-Based Solutions*, *2*, 100015. https://doi.org/10.1016/j.nbsj.2022.100015

Sowińska-Świerkosz, B., & García, J. (2022). What are Nature-based solutions (NBS)? Setting core ideas for concept clarification. *Nature-Based Solutions*, *2*, 100009. https://doi.org/10.1016/j.nbsj.2022.100009

Srivastava, S., Suprasanna, P., & D'Souza, S. F. (2012). Mechanisms of Arsenic Tolerance and Detoxification in Plants and their Application in Transgenic Technology: A Critical Appraisal. *International Journal of Phytoremediation*, *14*(5), 506–517. https://doi.org/10.1080/15226514.2011.604690

Stenfert Kroese, J., Batista, P. V. G., Jacobs, S. R., Breuer, L., Quinton, J. N., & Rufino, M. C. (2020). Agricultural land is the main source of stream sediments after conversion of an African montane forest. *Scientific Reports*, *10*(1), 14827. https://doi.org/10.1038/s41598-020-71924-9

Stewart, F. M., Muholland, T., Cunningham, A. B., Kania, B. G., & Osterlund, M. T. (2008). Floating islands as an alternative to constructed wetlands for treatment of excess nutrients from agricultural and municipal wastes–results of laboratory-scale tests. *Land Contamination & Reclamation*, *16*(1), 25–33.

Suárez-Perales, I., Valero-Gil, J., Leyva-de la Hiz, D. I., Rivera-Torres, P., & Garcés-Ayerbe, C. (2021). Educating for the future: How higher education in environmental management affects pro-environmental behaviour. *Journal of Cleaner Production*, *321*, 128972. https://doi.org/10.1016/j.jclepro.2021.128972

Taniwaki, R. H., Cassiano, C. C., Fransozi, A. A., Vásquez, K. V., Posada, R. G., Velásquez, G. V., & Ferraz, S. F. B. (2019). Effects of land-use changes on structural characteristics of tropical high-altitude Andean headwater streams. *Limnologica*, *74*, 1–7. https://doi.org/10.1016/j.limno.2018.10.002

Tanner, C. C., & Headley, T. R. (2011). Components of floating emergent macrophyte treatment wetlands influencing removal of stormwater pollutants. *Ecological Engineering*, *37*(3), 474–486. https://doi.org/10.1016/j.ecoleng.2010.12.012

Telwala, Y. (2023). Unlocking the potential of agroforestry as a nature-based solution for localizing sustainable development goals: A case study from a drought-prone region in rural India. *Nature-Based Solutions*, *3*, 100045. https://doi.org/10.1016/j.nbsj.2022.100045

Terán Cazar, P. R. (2017). *La aplicación de la ley orgánica de recursos hídricos uso y aprovechamiento del agua, vulnera los derechos de la comunidad de Chilla Chico cantón Saquisili provincia de Cotopaxi, en el año 2016*. http://www.dspace.uce.edu.ec/handle/25000/13306

Thompson, J. B., Zurita-Arthos, L., Müller, F., Chimbolema, S., & Suárez, E. (2021). Land use change in the Ecuadorian páramo: The impact of expanding agriculture on soil carbon storage. *Arctic, Antarctic, and Alpine Research*, *53*(1), 48–59. https://doi.org/10.1080/15230430.2021.1873055

Trujillo-Miranda, A. L., Toledo-Aceves, T., López-Barrera, F., & Gerez-Fernández, P. (2018). Active versus passive restoration: Recovery of cloud forest structure, diversity and soil condition in abandoned pastures. *Ecological Engineering*, *117*, 50–61. https://doi.org/10.1016/j.ecoleng.2018.03.011

Uddin, Md. G., Nash, S., & Olbert, A. I. (2021). A review of water quality index models and their use for assessing surface water quality. *Ecological Indicators*, *122*, 107218. https://doi.org/10.1016/j.ecolind.2020.107218

UN. (2015). *Transforming our world: The 2030 Agenda for Sustainable Development | Department of Economic and Social Affairs*. https://sdgs.un.org/2030agenda

UNCTAD. (2022). *Now 8 billion and counting: Where the world's population has grown most and why that matters | UNCTAD*. https://unctad.org/data-visualization/now-8-billion-and-counting-where-worlds-population-has-grown-most-and-why

UNEP. (2023). *Adaptation Gap Report 2023: Underfinanced. Underprepared. Inadequate investment and planning on climate adaptation leaves world exposed*. United Nations Environment Programme. https://doi.org/10.59117/20.500.11822/43796

UNICEF. (2020). *Access to drinking water*. UNICEF DATA. https://data.unicef.org/topic/water-and-sanitation/drinking-water/

United Nations Development Programme (UNDP). (2021). *Fondo de manejo de páramos y lucha contra la pobreza de Tungurahua* (https://www.equatorinitiative.org/wp-content/uploads/2019/07/Fondo-de-Paramos-Case-Study-English-FNL.pdf).

UN-Water. (2016). *Towards a worldwide assessment of freshwater quality: A UN-water analytical brief.* UN-Water. https://www.unwater.org/publications/towards-worldwide-assessment-freshwater-quality

UN-Water. (2023). *Partnerships and cooperation for water | UN World Water Development Report 2023*. https://www.unesco.org/reports/wwdr/2023/en

U.S. EPA, (U.S. Environmental Protection Agency). (1988). *Phosphorus: Water Quality Standards Criteria Summaries: A Compilation of State/Federal Criteria* [Overviews and Factsheets].

U.S. EPA, (U.S. Environmental Protection Agency). (2012). *Guidelines for Water Reuse* [Overviews and Factsheets]. https://www.epa.gov/sites/default/files/2019-08/documents/2012-guidelines-water-reuse.pdf

Valladares, C., & Boelens, R. (2019). Mining for Mother Earth. Governmentalities, sacred waters and nature’s rights in Ecuador. *Geoforum*, *100*, 68–79. https://doi.org/10.1016/j.geoforum.2019.02.009

Vergini, E. S., & Groumpos, P. P. (2021). Advanced State Fuzzy Cognitive Maps applied on nearly Zero Energy Building model. *IFAC-PapersOnLine*, *54*(13), 533–538. https://doi.org/10.1016/j.ifacol.2021.10.504

Videira, N., Schneider, F., Sekulova, F., & Kallis, G. (2014). Improving understanding on degrowth pathways: An exploratory study using collaborative causal models. *Futures*, *55*, 58–77. https://doi.org/10.1016/j.futures.2013.11.001

Vinueza, D., Ochoa-Herrera, V., Maurice, L., Tamayo, E., Mejía, L., Tejera, E., & Machado, A. (2021). Determining the microbial and chemical contamination in Ecuador's main rivers. *Scientific Reports*, *11*(1), 17640. https://doi.org/10.1038/s41598-021-96926-z

Vollmer, D., Burkhard, K., Adem Esmail, B., Guerrero, P., & Nagabhatla, N. (2022). Incorporating Ecosystem Services into Water Resources Management—Tools, Policies, Promising Pathways. *Environmental Management*, *69*(4), 627–635. https://doi.org/10.1007/s00267-022-01640-9

Wang, G., Mang, S., Cai, H., Liu, S., Zhang, Z., Wang, L., & Innes, J. L. (2016). Integrated watershed management: Evolution, development and emerging trends. *Journal of Forestry Research*, *27*(5), 967–994. https://doi.org/10.1007/s11676-016-0293-3

Wang, S., Li, J., Zhang, B., Spyrakos, E., Tyler, A. N., Shen, Q., Zhang, F., Kuster, T., Lehmann, M. K., Wu, Y., & Peng, D. (2018). Trophic state assessment of global inland waters using a MODIS-derived Forel-Ule index. *Remote Sensing of Environment*, *217*, 444–460. https://doi.org/10.1016/j.rse.2018.08.026

Wang, W.-H., Wang, Y., Sun, L.-Q., Zheng, Y.-C., & Zhao, J.-C. (2020). Research and application status of ecological floating bed in eutrophic landscape water restoration. *Science of The Total Environment*, *704*, 135434. https://doi.org/10.1016/j.scitotenv.2019.135434

Ware, J., & Callaway, R. (2019). Public perception of coastal habitat loss and habitat creation using artificial floating islands in the UK. *PLoS ONE*, *14*(10), e0224424. https://doi.org/10.1371/journal.pone.0224424

Warner, J. F., Wester, P., & Hoogesteger, J. (2014). Struggling with scales: Revisiting the boundaries of river basin management: Struggling with scale. *Wiley Interdisciplinary Reviews: Water*, *1*(5), 469–481. https://doi.org/10.1002/wat2.1035

Warren, G., & Metternicht, G. (2005). Agricultural Applications of High-Resolution Digital Multispectral Imagery. *Photogrammetric Engineering & Remote Sensing*, *71*(5), 595–602. https://doi.org/10.14358/PERS.71.5.595

Watkins, G., Mueller, S.-U., Ramirez, M. C., Meller, H., Blatsos, I., Carvalho Fernandes de Oliveira, J., Contreras Casado, C., Georgoulias, A., Georgoulias, N., & Rodriguez, J.

(2017). *Lessons from Four Decades of Infrastructure Project-Related Conflicts in Latin America and the Caribbean*. Inter-American Development Bank. https://doi.org/10.18235/0000803

WHO. (2019). *Drinking-water*. https://www.who.int/news-room/fact-sheets/detail/drinking-water

Wiegant, D. (2022). Ecuadorian water funds' use of scale-sensitive strategies to stay on course in forest and landscape restoration governance. *Journal of Environmental Management*, *311*, 114850. https://doi.org/10.1016/j.jenvman.2022.114850

Wiegant, D., Peralvo, M., van Oel, P., & Dewulf, A. (2020). Five scale challenges in Ecuadorian forest and landscape restoration governance. *Land Use Policy*, *96*, 104686. https://doi.org/10.1016/j.landusepol.2020.104686

Wilson, N. J., Mutter, E., Inkster, J., & Satterfield, T. (2018). Community-Based Monitoring as the practice of Indigenous governance: A case study of Indigenous-led water quality monitoring in the Yukon River Basin. *Journal of Environmental Management*, *210*, 290–298. https://doi.org/10.1016/j.jenvman.2018.01.020

Xiao, Z., Chen, W., & Li, L. (2012). An integrated FCM and fuzzy soft set for supplier selection problem based on risk evaluation. *Applied Mathematical Modelling*, *36*(4), 1444–1454. https://doi.org/10.1016/j.apm.2011.09.038

Yang, G., Li, Y., Huang, T., Fu, B., Tang, J., Zhang, X., & Wu, J. (2021). Multi-scale evaluation of ecological restoration effects in the riparian zone using Landsat series images from 1980 to 2019. *Ecological Indicators*, *132*, 108342. https://doi.org/10.1016/j.ecolind.2021.108342

Yang, L., Meng, X., & Zhang, X. (2011). SRTM DEM and its application advances. *International Journal of Remote Sensing*, *32*(14), 3875–3896. https://doi.org/10.1080/01431161003786016

Yeh, N., Yeh, P., & Chang, Y.-H. (2015). Artificial floating islands for environmental improvement. *Renewable and Sustainable Energy Reviews*, *47*, 616–622. https://doi.org/10.1016/j.rser.2015.03.090

Yongabi, K., Nagabhatla, N., & Rios, P. C. S. (2018). Phytoremediation Eco-models Using Indigenous Macrophytes and Phytomaterials. In N. Nagabhatla & C. D. Metcalfe (Eds.), *Multifunctional Wetlands: Pollution Abatement and Other Ecological Services from Natural and Constructed Wetlands* (pp. 253–273). Springer International Publishing. https://doi.org/10.1007/978-3-319-67416-2_9

Zahawi, R. A., Reid, J. L., & Holl, K. D. (2014). Hidden Costs of Passive Restoration. *Restoration Ecology*, *22*(3), 284–287. https://doi.org/10.1111/rec.12098

Zapata, D., Oleas, N. H., Páez-Vacas, M., & Tobes, I. (2021). Water Quality Assessment of the Cutuchi River Basin (Ecuador): A Review of Technical Documents. *IOP Conference Series: Earth and Environmental Science*, *690*(1), 012058. https://doi.org/10.1088/1755-1315/690/1/012058

Zhao, F., Xi, S., Yang, X., Yang, W., Li, J., Gu, B., & He, Z. (2012). Purifying eutrophic river waters with integrated floating island systems. *Ecological Engineering*, *40*, 53–60. https://doi.org/10.1016/j.ecoleng.2011.12.012

Acknowledgements

This project was funded by the German Academic Exchange Service (DAAD) from funds of the Federal Ministry for Economic Cooperation (BMZ), SDGnexus Network (grant number 57526248), the program "exceed - Hochschulexzellenz in der Entwicklungszusammenarbeit. I am very grateful to these institutions for this once-in-a-lifetime opportunity.

Throughout this research journey, I have received invaluable support from the wonderful people at SDG Nexus, the Center for International Research on Development and Environment (ZEU), the Technical University Cotopaxi, various Ecuadorian governmental institutions and local communities in Ecuador. Thanks to them, this dream has come true.

I deeply appreciate the guidance of my supervisor, Prof. Dr. Lutz Breuer, who has been my mentor over these three years, helping me view science from a new perspective. I also want to thank Prof. Dr. Ramona Teuber for serving as my second supervisor.

Finally, I am grateful to Dr Matthias Höher for his administrative support, as well as to my colleagues and friends at ZEU and SDG Nexus in Germany and to my family, colleagues and friends in Ecuador.

Declaration

Ich erkläre: Ich habe die vorgelegte Dissertation selbständig und ohne unerlaubte fremde Hilfe und nur mit den Hilfen angefertigt, die ich in der Dissertation angegeben habe. Alle Textstellen, die wörtlich oder sinngemäß aus veröffentlichten Schriften entnommen sind, und alle Angaben, die auf mündlichen Auskünften beruhen, sind als solche kenntlich gemacht. Bei den von mir durchgeführten und in der Dissertation erwähnten Untersuchungen habe ich die Grundsätze guter wissenschaftlicher Praxis, wie sie in der „Satzung der Justus-Liebig-Universität Gießen zur Sicherung guter wissenschaftlicher Praxis“ niedergelegt sind, eingehalten.

I declare that I have completed this dissertation single-handedly without the unauthorized help of a second party and only with the assistance acknowledged therein. I have appropriately acknowledged and cited all text passages that are derived verbatim from or are based on the content of published work of others, and all information relating to verbal communications. I consent to the use of an anti-plagiarism software to check my thesis. I have abided by the principles of good scientific conduct laid down in the charter of the Justus Liebig University Giessen „Satzung der Justus-Liebig-Universität Gießen zur Sicherung guter wissenschaftlicher Praxis“ in carrying out the investigations described in the dissertation”

Kalina Marcela Fonseca Largo
Giessen, January 2025