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Multi-risk assessment of freshwater ecosystem services under climate change conditions

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Coordinatore del Dottorato
ch. prof. Carlo Carraro

Supervisore
ch. prof. Antonio Marcomini

Supervisore cotutela
ch. prof. Andrea Critto
dott.ssa Silvia Torresan

Dottorando
Hung Vuong PHAM
Matricola 956329

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CONTENTS

EXECUTIVE SUMMARY

Freshwater provides a wide range of Ecosystem Services (ES) such as provisioning services (e.g. water for agriculture), regulating services (e.g. buffering of flood flows), cultural services (e.g. river viewing) and supporting services (e.g. role in nutrient cycling). Climate change (CC) has altered both the supply and demand of the freshwater ecosystem due to the changes in temperatures, precipitations, and water flow regimes. Consequently, these variations had changed the availability of freshwater, shifted river discharge, and intensified the magnitude and frequency of potential hazards such as storms, floods, droughts, and heatwaves. Beside CC, freshwater ecosystems are threatened by many anthropogenic stressors such as overfishing, water pollution, flow regimentation, invasion of exotic species transported either intentionally or accidentally by human-mediated vectors, and hydroelectric power stations. Therefore, there is an urgent need to understand the future dynamics of potential freshwater-related ES and the conjoined effects of climate and land-use change on these services to support the identification of proper management actions aiming to maintain the good ecological status of the ecosystem.

In this context, this work developed an integrated model, incorporated ecosystem services model (InVEST) and Bayesian Networks (BNs) with climate and land-use change models. This study aims to (i) dynamically project the capacity of the selected ES such as water yield (WY), total nitrogen (TN), and total phosphorus (TP) retention; (ii) identify and quantify the joint impacts of climate change and land-use change on these services; and (iii) assess the potential space to improve the capacity of ES toward sustainable water. These goals were achieved by the accomplishment of three independent research papers. Specifically, in the first paper, we selected and reviewed the most relevant peer reviewed publications and reports to identify major drivers and most relevant services, and conceptually synthesized the fundamental effects of these drivers on those services. Then, in the second paper, we developed an integrated modeling approach to quantify the impacts of CC and land-use change on selected ES such as WY, TN, and TN. This model incorporated InVEST with climate and land-use change models. Finally, in the third paper, we accommodated Bayesian Networks into this assessment in order to consider the uncertainty and the complexity of the real world, supporting decision-makers in the identification of the best adaptation plans. This work was conducted in the Taro River basin (TRB), Italy, where several pressures for ES have been identified.

The primary outcomes of this research are a conceptual framework, a set of indicators, and an integrated modeling approach coupling climate, land-use, and soil data into an ecosystem model and Bayesian Networks. Applying this conceptual framework and the integrated model to the TRB case study, we found some interesting insights. First, freshwater-related ES have not been explicitly targeted as the assessment endpoint. Moreover, the supply of ecosystem services draws much more attention than the demand. Second, we would expect 20% and 3% of reduction in water yield and total nitrogen retention, respectively while an increase of 3% for the total phosphorus retention in the TRB. The rates of these changes were found to differ over time with the most pronounced differences between 2020-2030 and slower variations afterward. Finally, the outcomes of thousands of simulations suggested that the probability of the high value of ES will be lower than those for low values. Moreover, there will be limited space to improve these services, especially for nutrient retention.

The advancement and innovation of this approach were the usage of free spatiotemporal data and open software such as R, Python, and Q-GIS, allowing to conduct quick assessments, visualizing the results in time and space. The application in the Taro River basin (Italy), within the activities of PROLINE project, showed that this approach provided useful insights for the decision-makers in the selection of best management practices toward a sustainable ecosystem.

1 INTRODUCTION

1.1 MOTIVATIONS AND OBJECTIVES

Climate change has altered freshwater ecosystems on both supply and demand due to changing temperatures, precipitations, and water flow regimes (Pham et al., 2019). Consequently, these anomalies had changed the availability of freshwater (Colombani, Dinelli, and Mastrocicco, 2016), shifted river discharge (Suen, 2010), and intensified the magnitude and frequency of extreme events such as heavy rainfalls (Coppola et al., 2014; Madsen et al., 2014; Sorg et al., 2012; Trenberth, 2011), floods, and droughts (Kundzewicz et al., 2008). Besides, deforestation, poor land use management, and manufacturing practices are destroying carbon sinks, imposing more stress on the ecosystem (Tobergte and Curtis, 2013). These problems draw massive attention from the scientific community, authorities, and decision-makers.

After the establishment of the Representative Concentration Pathways – RCP (Vuuren et al., 2011), the Shared Socioeconomic Pathways – SSPs (Neill, 2014), abundant spatiotemporal projected data have been related from climate, land use, soil, and other socio-economic models (Vuuren, 1014). Therefore, these data inspire the development of an integrated model to utilize this information to project ES and the related problems in the future, supporting decision-making processes in the complex world.

The objective of this thesis is to seek the answers for three main research questions (*i*) What are the problems related to ES? (*ii*) How do we know them? (*iii*) And what can we do to solve these problems? Specifically, the first question focuses on the identification of the complex effects of climatic and non-climatic factors on the supply and demand of the freshwater ecosystem. Then, the second task aims to assess the conjoined impacts of land-use and climate changes, identified from above, on the selected potential ecosystem services (i.e. water yield and nutrients retention). Finally, the last assignment means to project the capacity of these ES under thousands of random scenarios to assist decision-makers in the selection of the best management action.

1.2 OUTLINE OF THE THESIS

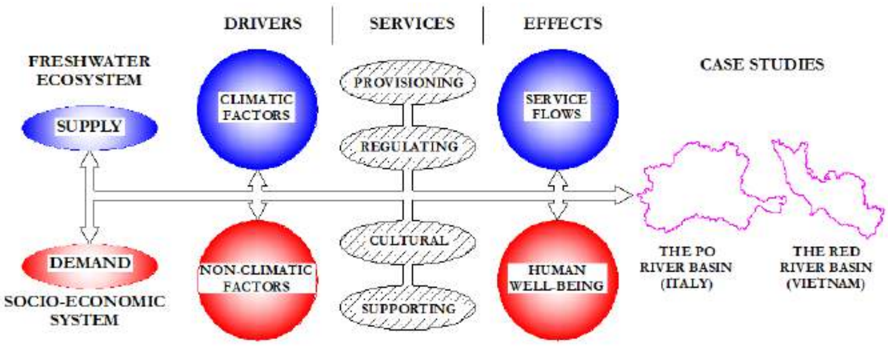
The thesis is composed of three independent research papers that aim to respond to the three pre-defined research questions, focusing on the topic of freshwater-related ecosystem services.

THE FIRST PAPER , titled "*Alteration of freshwater ecosystem services under global change – A review focusing on the Po River basin (Italy) and the Red River basin (Vietnam)*", proposed a conceptual framework and a set of indicators for assessing the impacts of climate change and human activities on freshwater-related ES. Then, this proposed framework was applied to the provisioning services of two well-known case studies, namely the Po River basin (Italy) and the Red River basin (Vietnam).

THE SECOND PAPER , titled "*Coupling scenarios of climate and land-use change with assessments of potential ecosystem services at the river basin scale*", developed an integrated modeling approach to assess the conjoined impacts of land-use and climate changes on the potential ecosystem services (i.e. water yield and nutrients retention) until 2050. The integrated model consists of 3 main components, namely climate projections, land-use projections, and the ecosystem model. Then, this approach was applied to the Taro River basin in Italy. Thanks to the simulation, the assessment of the impacts of land-use and climate change on potential freshwater ES can support decision-makers and authorities in the identification of the most suitable management measures.

THE THIRD PAPER , titled "*Integrating Bayesian Networks into ecosystem services assessment to support decision-makers at the river basin scale*", aimed to integrate Bayesian Networks in ecosystem service assessment which enables identification of main factors that affect the ecosystem, trade-offs among ecosystem services, and evaluation ES under different scenarios. This integrated model was applied to the Taro River basin in Italy, assisting the identification and prioritization of the best management practices for sustainable water use, balancing the tradeoffs among services.

2 PAPER 1: ALTERATION OF FRESHWATER ECOSYSTEM SERVICES UNDER GLOBAL CHANGE



GRAPHICAL ABSTRACT OF THE FIRST PAPER

HIGHLIGHTS

- We assessed the impacts of climate change and human activities on freshwater services;
- Indicators and a conceptual framework of climatic/non-climatic effects are analyzed;
- The literature focuses more on the provisioning and regulating services;
- Consistent terminologies and classifications are needed for future risk assessments.

Alteration of freshwater ecosystem services under global change – A review focusing on the Po River basin (Italy) and the Red River basin (Vietnam)

Hung Vuong Pham[a,b], Silvia Torresan[a,b], Andrea Critto[a,b], Antonio Marcomini[a,b,*]

[a]*Fondazione Centro Euro-Mediterraneo sui Cambiamenti Climatici (CMCC), via Augusto Imperatore 16, 73100 Lecce, Italy*
[b]*Department of Environmental Sciences, Informatics and Statistics, University Ca Foscari Venice, Via Torino 155, 30172 Venezia-Mestre, Venice, Italy*

Abstract

Freshwater ecosystem services are negatively affected by factors such as climate change (e.g. changes in temperature, precipitation, and sea level rise) and human interventions (e.g. agriculture practices, impoundment of dams, and land use/land cover change). Moreover, the potential synergic impacts of these factors on ecosystems are unevenly distributed, depending on geographical, climatic and socio-economic conditions. The paper aims to review the complex effects of climatic and non-climatic drivers on the supply and demand of freshwater ecosystem services. Based on the literature, we proposed a conceptual framework and a set of indicators for assessing the above-mentioned impacts due to global change, i.e. climate change and human activities. Then, we checked their applicability to the provisioning services of two well-known case studies, namely the Po River basin (Italy) and the Red River basin (Vietnam).
To define the framework and the indicators, we selected the most relevant papers and reports; identified the major drivers and the most relevant services; and finally summarized the fundamental effects of these drivers on those services. We concluded that the proposed framework was applicable to the analyzed case studies, but it was not straightforward to consider all the indicators since ecosystem services were not explicitly considered as key assessment endpoints in these areas. Additionally, the supply of ecosystem services was found to draw much more attention than their demand. Finally, we highlighted the importance of defining a common and consistent terminology and classification of drivers, services, and effects to reduce mismatches among ecosystem services when conducting a risk assessment.

Keywords: climate change, human activities, ecosystem services framework, risk assessment

1. Introductory review

Freshwater accounts for only 0.01% of water volume in the world and about 0.8% of the land surface, but this small amount of water is supporting at least 100,000 species (Cazzolla Gatti, 2016). There are about 263 international rivers, which occupy 45% of the land surface area of the Earth, supporting more than 40% of the global human population (Cazzolla Gatti, 2016). Freshwater provides a wide range of Ecosystem Services (ES) such as provisioning services (e.g. water for agriculture), regulating services (e.g. buffering of flood flows), cultural services (e.g. river viewing) and supporting services (role in nutrient cycling) (Aylward et al., 2005). Unfortunately, Climate Change (CC) has altered freshwater ecosystems on both supply side and demand side due to changing temperatures, precipitations, and water flow regimes (Carpenter et al., 2011; Däll and Zhang, 2010). These changes would cause more severe floods, droughts, and Sea Level Rise (SLR) in the future respect to the reference period (1980–1999), under the climate change scenarios A2 (high CO_2 emission) and B2 (medium CO_2 emission) (Giang et al., 2013). In regard to CC, the concentrations of greenhouse gases such as carbon dioxide (CO_2), methane (CH_4) and nitrous oxide (N_2O) have shown large increases since 1750 (40%, 150%, and 20%, respectively) (IPCC, 2014a). These changes led to an increase in average temperature of 0.85 (0.65–1.06) °C over the period 1880 to 2012, a rise in global mean sea level of 0.19 m (0.17–0.21) m over the period (1901–2010). Consequently, these anomalies had changed the availability of freshwater (Colombani et al., 2016a), shifted river discharge (Suen, 2010), and intensified the magnitude and frequency of extreme events such as heavy rainfalls (Coppola et al., 2014; Madsen et al., 2014; Sorg et al., 2012; Trenberth, 2011), floods , and droughts (Kundzewicz et al., 2008). For instance, the IPCC (2014b) projected that the annual runoff would decrease in the Southern Europe of 6–36% by 2070s and the low flow in summer may decrease up to 80% in some rivers in southern Europe. Yet, the timing of the maximum monthly river discharge would be shifted by at least one month on one-third of the global land area. Additionally, since temperature and precipitation patterns affect the rates of nutrient cycling and toxin breakdown, and thermal niches of organisms, CC could further affect the quality of freshwater (Carpenter et al., 2011), and aquatic species (Däll and Zhang, 2010). For instance, a higher temperature could increase the rate of uptake of pollutants by changing ventilation rate, increasing metabolic rate and decreasing oxygen solubility, which could lead to eutrophication (Schiedek et al., 2007). The decrease in river dis-

*Corresponding author
Email addresses: vuong.pham@unive.it (**Hung Vuong Pham**), marcom@unive.it (Antonio Marcomini)

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charges could reduce the genetic flow and variation of aquatic biota population due to stream fragmentation. Moreover, precipitation patterns could modify sediment flow and the erosivity rate, which could further modify the freshwater quality. Specifically, heavy rainfall and high flows could increase loads of suspended solids, sediment yields, and contaminant metal fluxes because of soil erosion and sediment transport from the land surface. In contrast, droughts could exacerbate acidification by lowering water tables, creating aerobic conditions and enhancing oxidation of sulfur to sulfate (Whitehead et al., 2009). Furthermore, different effects of CC (e.g. temperature anomalies, precipitation anomalies and SLR) not only affect individually FreshWater Ecosystems Services (FWES), but they may also interact with each other quite differently (Woodward et al., 2010). As a result, CC would have impacts on water provision, transportation, hydropower production, flood regulation, natural filtration, and water treatment (Kernan et al., 2011; Moore et al., 1997; Woodward et al., 2010). Beside CC, freshwater ecosystems are threatened by many anthropogenic stressors such as overfishing, water pollution, flow regimentation, degradation of habitats, invasion of exotic species transported either intentionally or accidentally by human-mediated vectors, and hydroelectric power stations (Cazzolla Gatti, 2016; Däll and Zhang, 2010). In addition, deforestation, poor land use management and manufacturing practices are destroying carbon sinks, imposing more stress on FWES (Tobergte and Curtis, 2013). On the one side, requiring higher demand of freshwater, the upward trend of population and urbanization could modify the availability and distribution of freshwater. For instance, anthropogenic water consumption resulted in a decrease in mean annual runoff by 5% or more in many river basins, e.g. the heavily irrigated areas at middle latitudes in Asia and the western part of the United States, during the examined period (1971–2000) (Haddeland et al., 2014). On the other side, these anthropogenic stressors could affect the quality of freshwater. For instance, chemicals and pollutants associated with human activities (i.e. agriculture practices, industrial areas and urban traffic) were recognized as relevant threats of freshwater quality (Ahmad et al., 2016; McCoy et al., 2015). Specifically, agricultural practices were considered as major non-point sources of macronutrient and pesticides transported to water bodies by rainfall and water runoff. When applied to crops, pesticides intercepted by the soil surface undergo partitioning between the soil, water and air. A proportion of pesticides may leave the soil/water system by volatilization to the atmosphere. Some pesticides remain in the soil partition between soil and rainfall water. The ratio of the concentration in the soil to the concentration in the water is called the soil sorption coefficient. It is an equilibrium between pesticide concentration divided between soil and water. For pesticides highly absorbed by soil particles, erosion and mechanical runoff are the main processes of transport to water bodies (Bloomfield et al., 2006). Additionally, humans directly influence the dynamics of the water cycle and river flow regimes through dams construction for water storage, and through water withdrawals for industrial, agricultural, or domestic purposes. In particular, the presence of over 45,000 dams exceeding 15 m height in the world would have influences on biodiversity and ecological function of river systems as well as disrupting sediment flux, thermal regimes, and the fertile soils of the flood basins (Mooney et al., 2009). For instance, Haddeland et al. (2014) found out that the impacts of human disturbances (i.e. impoundment and operation of dams and water consumption) on annual runoff in some river basins are greater than the impacts of expected CC over the next 40 years. Similarly, in some irrigated areas in Asia and the western United States, the effects of current anthropogenic interventions on mean annual runoff are stronger than those driven by the increased temperature of a 2 or 3°C (Haddeland et al., 2014). Indeed, the effects of CC on ES vary in time and space and are rarely homogeneous (Barker et al., 2010). For instance, while an increase in precipitation was observed since 1901 over land north of 30°N, a decrease in precipitation over land between 10°S and 30°N was detected after the 1970s (Carpenter et al., 2011). In the end, together with increasing pollution, overpopulation and human activities, climate change could lead to the deterioration and depletion of available water sources, with negative consequences on water security. Therefore, it is noteworthy that the impacts of climatic and non-climatic factors on FWES result from different combinations of climatic, geographical and socio-economic conditions. Thereafter, we choose the two well-known case studies, namely the Po River Basin (PRB) in Italy and the Vietnamese part of the Red River basin (hereafter it refers to the Red River basin or RRB), to investigate the above-mentioned aspects. Although they are located in two different continents, they have some relevant common characteristics. Firstly, they are both transnational basins. While the PRB is shared among Italy, Switzerland and France, the RRB is shared among Vietnam, China and Laos. Besides, they have a similar size (i.e. 74,000 km^2 and 86,700 km^2, respectively). Secondly, they are important to the national economy, contributing to the gross domestic product (GDP) about 40% and 20%, respectively. Thirdly, freshwater is intensively consumed by several conflicting water uses, such as agriculture, environment protection (e.g. the water demand for maintaining the minimum environmental flow) and hydropower generation. Finally, FWES on these case studies are negatively affected both by climatic factors and non-climatic factors. Noticeably, most of the CC impact assessment studies on FWES developed in recent years have focused on the impacts of temperature anomalies (Haddeland et al., 2014). Others consider the changes in mean precipitation and in river flow regimes, including long-term average flows, low flows and high flows (Däll and Zhang, 2010), and SLR (Carpenter et al., 2011). Thus, this review focuses on these relevant climate factors, namely temperature anomaly, precipitation anomaly and SLR . Meanwhile, anthropogenic factors, which vary locally and regionally from one case study to the other, will be further discussed in each case study, i.e. agriculture activities in the PRB. and the impoundment and operation of cascade reservoirs and dams in the RRB. In this context, the objectives of this paper are the following:

- Propose a conceptual framework and a set of indicators to assess the impacts of climatic/non-climatic factors on

FWES, considering both the supply and the demand side of such services (Section 2.1 and 2.2);

- Check the feasibility of the framework and indicators for the two selected river basins, namely the Po River basin in Italy and the Vietnamese part of the Red River basin, which are highly affected by several global change drivers and featured by different geographical, climatic and socio-economic conditions (Section 3);
- Highlight some gaps in the literature and recommend further research directions to bridge these gaps, especially with the reference to the selected case studies (Section 4).

2. Methods and materials

The literature-based review started with a search of scientific papers in the field of ecosystem services. To find the publications related to FWES, we used the following keywords, either alone or in combination: ecosystem service, ecosystem services, ecosystem function, ecosystem process, ecosystem benefit, and freshwater. To find the publications related to the selected case studies, namely the Po River (Italy) and the Red River (Vietnam), we used additional keywords: Po river basin, Po river delta, Po river valley, Red river basin, and Red river delta.
Then, for each publication, we identified the main drivers of changes and classified them into two groups, i.e. the climatic drivers (temperature, precipitation, rainfall, etc.) and the non-climatic drivers (land use change, population dynamics, dams construction, etc.). At the same time, all the categories of FWES were classified following the classification of the Millennium Ecosystem Assessment (Aylward et al., 2005). Finally, we summarized the effects of the aforementioned drivers on each category of FWES using the conceptual framework (Section 2.1 and Figure 1) and a set of indicators (Section 2.2 and Table 1). Subsequently, we checked the feasibility of this conceptual schematization and the proposed indicators by applying them to assess the impacts of global change on FWES in the selected case studies, namely the PRB (Section 2.3.1) and the RRB (Section 2.3.2). The results are extensively described in Section 3.

2.1. *Conceptual framework to assess the impacts of global change on FWES*

The conceptual framework draws the main components of the freshwater ecosystem and the socio-economic system, the main interactions between them and the relevant processes affecting both the supply and demand of FWES (Figure 1). In the vertical axis of this figure, the freshwater ecosystem is represented by the part A while the socio-economic system is represented by the part B. In the horizontal axis, all the drivers of change are listed in the block I with the climatic factors on the left and non-climatic factors on the right. Then, all the FWES are reported in block II, classified into four categories, namely provisioning, regulating, cultural and supporting services. Finally, the effects of all drivers on the FWES services are reported in block III.

2.1.1. *Drivers of changes*

Drivers of changes are the main natural and anthropogenic forces, which can determine variations in the state of the environment and/or human system. These drivers cause certain pressures on freshwater resources which result in impacts on FWES, either by changing the quantity and/or by changing the quality of freshwater. Thus, it is important to identify the main drivers and to classify them in suitable categories (e.g. climatic and non-climatic drivers). In this conceptual framework, the main climatic drivers are temperature, precipitation, rainfall pattern, extreme events, land cover changes and SLR. These drivers interact with the natural system through natural processes including photosynthesis, pollination, decomposition, and others. On the other side, the main non-climatic drivers are land use changes, population dynamic, dams construction, urbanization and agriculture practices.

2.1.2. *Freshwater ecosystem services*

FWES are the benefits people obtain from freshwater ecosystems, including provisioning, regulating, cultural, and supporting services (Aylward et al., 2005). First, freshwater is a provisioning service as it refers to the human use of fresh water for domestic, agriculture and industrial use, power generation, and transportation. Additionally, the hydrological cycle also sustains inland water ecosystems, including rivers, lakes, and wetlands. Therefore, important regulating FWES include water regulation (i.e. hydrological flow), water purification and waste treatment, erosion regulation and water-related natural hazard regulation. Then, freshwater ecosystem also provides the cultural services such as the human benefits associated with river rafting, kayaking, hiking, fishing as a sport, and river viewing. Finally, freshwater contributes to other supporting services such as primary production, predator/prey relationships and ecosystem resilience.

2.1.3. *Effects of global change on FWES*

Effects of global change on FWES are the changes in the physical, chemical or biological state of the freshwater system, determining the quality of ecosystem services and the welfare of human beings. The drivers of changes affect the FWES not only on the supply side through specific ecosystem functions[1] (e.g. producing biomass, decomposing organic matters, and dispersing plant seeds), but also on the demand side, through direct/non-direct uses of freshwater. For instance, extreme rainfall events can lead to hydrogeological hazards such as inundation, flood and landslides, affecting agriculture, transportation and communication infrastructure. In addition, the water demand in this period may increase due to the needs of the population for sanitary purposes.

2.1.4. *Interaction between freshwater ecosystem and socio-economic system*

It can be seen clearly from Figure1 that the supply of and the demand for ES are separated. While the supply is provided

[1]Ecosystem functions are the biological, geochemical and physical processes and components that take place or occur within an ecosystem.

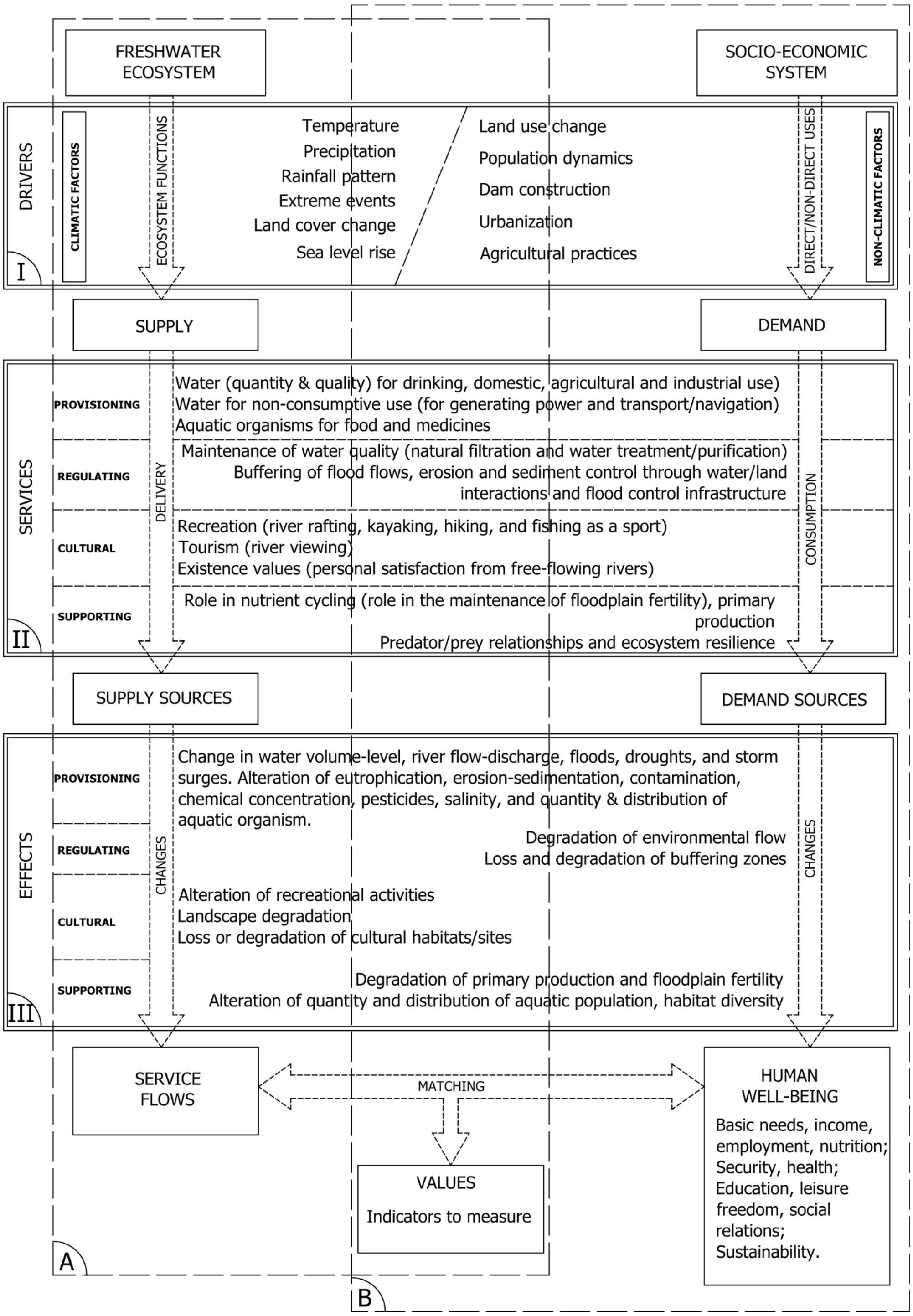


Figure 1: Conceptual schematization of freshwater ecosystem and socio-economic system (part A, and B in vertical axes, respectively) and their components, including the main drivers, their services, and the effects of these drivers on FWES (block I, II, and III in horizontal axes, respectively). Source: adapted from Aylward et al. (2005); Wolff et al. (2015).

by freshwater ecosystems, the demand is required by socio-economic systems. Nevertheless, this separation is not completely sharp since the systems share some common parts such as drivers of changes, services and effects. For instance, temperature, as a climatic driver, affects the water cycle (and supply) by changing water vapor concentrations, clouds, precipitation patterns, and streamflow patterns. Similarly, temperature changes affect the socio-economic system by altering the water consumption and demand rates.
Regarding the services, lakes and rivers supply the recreational services such as rafting, kayaking, hiking, fishing and river watching. Additionally, they also provide maintenance services such as the regulation of flow and level during the dry season. These services are driven by both climatic and non-climatic factors. Noticeably, the climatic drivers are usually driven by the natural system (e.g. circulation of water in the ocean, weather and climate, and water drainage) while the non-climatic drivers are often governed by the socio-economic system (e.g. human settlements, transportation routes, communication systems, and economics). However, the climatic and non-climatic drivers interact with each other and, consequently, intensify the impacts on FWES. On the one side, climate drivers (e.g. temperature and precipitation) are expected to alter the hydrological cycle, resulting in further changes in the supply of food, water and energy. On the other side, non-climatic factors (e.g. land use and urban expansion) may affect climatic factors by changing infiltration and evapotranspiration rates (Sample et al., 2016).
Consequently, the climate and non-climate drivers modify services flows on the supply side and the demand side through the delivery (e.g. from lakes and rivers to irrigation networks) and the consumption of freshwater (e.g. domestic, irrigation, and energy), respectively. Noticeably, on the one side, freshwater ecosystem can provide supplies in a specific location at a given time regardless of the demand from the socio-economic system. On the other side, demands from socio-economic system are independent from the supplies provided by freshwater ecosystem. Therefore, the basic needs of human well-being (e.g. income, employment, nutrition, security, health, education, leisure, freedom, etc.) would be achieved if the supplies are maintained over time and the demands are adapted to ensure no net loss of biodiversity and ecosystem services (European Commission, 2011). Nevertheless, the supplies and demands vary in time and in space depending on geographical, climatic and non-climatic conditions. Thus, we propose a list of indicators (Table 1) to quantify, map, assess, and compare the changes in supply and demand of FWES under climate change and human interventions.

2.2. *Indicators to quantify the effects of global change on FWES*

There is a wide range of indicators to assess the impacts of global changes on ES and, specifically, FWES. Nevertheless, they are not consistent and well-distributed among the four categories of FWES (i.e. provisioning, regulating, cultural and supporting services), depending on authors, fields of study and purpose of research. In fact, one indicator can be used for more than one type of FWES (e.g. annual average water yield can be used to assess the impacts of both climatic factors and non-climatic factors on provisioning services). Moreover, as shown in Table 1, most of the FWES categories can be associated with more than one indicator to quantify, assess or map their services.
Among FWES, provisioning services are the most commonly reported in the literature, followed by regulatory services and cultural services. Supporting services are the least common because they usually overlap with other service categories or it is not straightforward to differentiate them. For instance, pollination service can be considered as either the regulating service (i.e. it is an agriculture input that ensures the production of crops) or the supporting service (i.e. a necessary service for the production of other services such as the provisioning of food) (Kandziora et al., 2013). Therefore, the number of indicators for this category is much lower than those for other categories.
Furthermore, the selection of indicators depends on the purpose of a study, the applied methodological approach and the data availability (Olubode-Awosola, 2017; Wolff et al., 2015). Thus, it is noteworthy to study not only the number of indicators but also the quality of indicators and the methods to obtain them.
Regarding the quality of indicators, they can be classified into four categories, namely *(i)* indicator available and easily understood by non-technical audiences; *(ii)* indicator available but not easily understood by non-technical audiences; *(iii)* indicator not easily available and/or not easily understood by non-technical audiences; and *(iv)* indicator being proposed and may not be readily available (Olubode-Awosola, 2017). The quality of each indicator in Table 1 is represented by a symbol marked before it and by the legend in the caption of this table. It can be seen clearly from Table 1 that the majority of indicators is available and understandable without or with a little technical explanation. For instance, water level and water consumption per sector can be easily understood by readers while annual average water yield and annual river runoff can be understood by non-technical audiences when the definitions of water yield and runoff are given. Interestingly, the cultural services and the supporting services have many indicators which are proposed but may not be readily available. For example, it might be difficult to find suitable indicators for some generic services like life cycle maintenance and maintenance of genetic diversity (Egoh et al., 2012). Regarding the methods to obtain these indicators, they can be classified into three categories, namely *(i)* physically based; *(ii)* empirical; and *(iii)* participatory or expert-based (Wolff et al., 2015). Physically based methods are suitable for the scale on which relevant physical processes occur, such as the river basin scale for hydrological processes. Empirical methods work better with larger scale case studies with less detailed data requirements. Participatory approaches are most suitable for small-scale researches. These approaches consider the preferences of the participants such as stakeholders, experts and administrators. Therefore, the choice of the method depends on the scale of the case study and the data availability. In Table 1, the method used to quantify each indicator is represented by the format of the text and by the legend in the caption of this table. Noticeably, since one indicator can be obtained by more than one method, the classification of an indicator was

performed based on the popularity of these methods. It can be seen from Table 1 that the empirical methods are the most popular method across all FWES because these methods are suitable for a wide range of studies and they require less detailed data. The physically based methods are widely applied for provisioning services and regulating services, while the participatory and expert-based approaches are more popular in the cultural services.

Finally, it is important to point out that the use of these indicators does not necessarily allow the comparison among services (e.g. one service is more valuable than the others) because of the differences in metrics and methods. These indicators provide useful information for trade-off analysis, enabling markets for ES, supporting the establishment of environmental regulations and payments for ES (Olubode-Awosola, 2017).

2.3. Case studies

2.3.1. The Po River basin

The Po River is the longest Italian river (i.e. about 652 km long). Its basin (i.e. about 74,000 km^2) is a crucial resource for the Italian economy, accounting for about 40%, 35%, and 37% of the gross domestic product, agricultural product, and industrial product, respectively. It has 141 main tributaries which have a total length of about 6,750 km and 31,000 km for the natural and artificial channels, respectively. About 450 lakes are located in the Po River watershed (Montanari, 2012).

Freshwater of the Po River is intensely used for irrigation, hydropower production, and domestic purposes (Coppola et al., 2014). The average volume of annual precipitation is about 78 km^3, in which 47 km^3 (60%) is converted in outflow volume at the closure section, 20-25 km^3 is accounted for evapotranspiration, 17 km^3 is supplied for irrigation, 5 km^3 is supplied for civil and industrial users, and the rest is charged into groundwater. The mean daily discharge at Pontelagoscuro terminal station is 1,470 m^3 s$^-$1. The monthly distribution of rainfall is typically minimum in February and July, in both the lowlands and in the mountain areas, and maximum in late spring (MayJune) and in the middle of autumn (November) (Baruffi et al., 2012). Recently, this river basin has drawn huge attention from scientists and authorities on assessing the availability and distribution of freshwater resources in term of water quantity and water quality. Specifically, the extreme events (i.e. intensive rainfalls, floods and droughts) and their related impacts have been well reported (Coppola et al., 2014; Domeneghetti et al., 2015; Masoero et al., 2013; Montanari, 2012; Ravazzani et al., 2014). Other studies focused on the problems related to water quality, i.e. pollutants and eutrophication (Castaldelli et al., 2013; Colombani et al., 2016a; Corazzari et al., 2015; Facca et al., 2014; Luigi et al., 2015; Tesi et al., 2013).

Figure 2: The Po River basin and its catchment.

2.3.2. The Red River basin

The RRB is a transnational basin covering an area of 169,000 km2 between Vietnam (51.3%), China (48%), and Laos (0.7%). The three main tributaries of the Red River (Da, Lo, and Thao) rise in the northern part of the basin and join before reaching the large floodplain in the delta region, which extends over 21,000 km2 (Giuliani et al., 2016). Due to the limitation of data availability and accessibility, this review focuses only on the Vietnamese part of the river. Out of the three main tributaries, the Da River is the most important water source, contributing 42% to the total discharge at Son Tay (Castelletti et al., 2012). Additionally, the Red River Delta (RRD) is an important part of the RRB, accounting for approximately 23% of the population and 56% of rice products in Vietnam (Duc and Umeyama, 2011). About 47% of the RRD area (6,700 km2) is used for agriculture or aquaculture, mainly for annual crops, aquaculture, perennial crops, and pasture (90%, 6.6%, 3.1%, and 0.6%, respectively) (Luu et al., 2010). The water resources in this region are used mainly for human consumption, agricultural irrigation, and hydropower generation (Le et al., 2017).

There are four large reservoirs located in the RRB, i.e. the Hoa Binh and the Son La reservoirs are in the Da River; the Tuyen Quang and the Thac Ba reservoirs are in the Lo River. In addition, two more dams are recently constructed in the upstream Da River, namely the Huoi Quang dam (operating since December 2015) and the Lai Chau dam (operating since November 2016) (Le et al., 2017). The RRB has two distinct seasons,

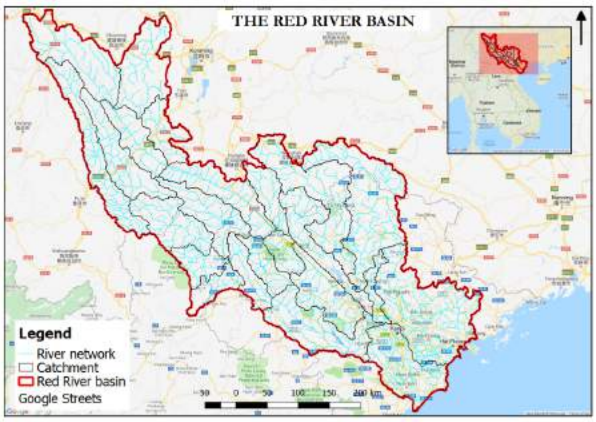


Figure 3: The Red River basin and its catchment.

Freshwater ecosystem services	Supply side		Demand side	
	Climate factors	Non-climate factors	Climate factors	Non-climate factors
Provisioning services				
Water (quantity and quality) (for drinking, domestic use, and agriculture and industrial use)	►**Annual average water yield (mm yr^{-1})** ▲*Number of extreme events (n. yr^{-1})* ►**Annual average sediment (kg yr^{-1})** ►**Nutrient rate, e.g. N, P (kg ha^{-1})**	►**Annual sectoral water yield (e.g. domestic, agriculture and industry) (mm yr^{-1})** ►**Annual river runoff (m^3 s^{-1} yr^{-1})** ▼**Leakage of nutrients (kg ha^{-1})**	▲*Withdrawal of freshwater (m^3 ha^{-1})* ▲*Water consumption per sectors (e.g. for cooling, for heating) (m^3 yr^{-1})*	▲*Water consumption per sectors (e.g. drinking, agriculture and industry) (m^3 yr^{-1})*
Water for non-consumptive use (for generating power, transport and navigation)	▲*Seasonal variation of hydropower production (kWh yr^{-1})* ▲**Water level (m)** ▲*Number of interrupted days for navigation (days)*	▲*Hydropower production before and after human interventions (kWh yr^{-1})*	▲*Seasonal variation of water consumption for hydropower (m^3 yr^{-1})* ▲*Water level for navigation (m)*	▲*Variation of water consumption for hydropower (m^3 yr^{-1})* ▲*Transportation product (t yr^{-1})*
Aquatic organisms for food and medicines	▼Amount or number of wild plants used for medicine/biochemical (kg ha^{-1}, n. ha^{-1})	►*Number and identity of selected species (n.)* ►*Number/area of habitants (n. ha^{-1})*	▼*Variation of wild plants demand for food and medicines due to climate change (kg ha^{-1})*	▼*Variation of wild plants demand for food and medicines due to socio-economic development (kg ha^{-1})*
Regulating services				
Maintenance of water quality (natural filtration and water treatment)	►**Groundwater recharge rate (mm ha^{-1})** ►**Total dissolved solids (mg/l)** ▼**Decomposition rate of organic matter (kg ha^{-1})** ▲*Area occupied by riparian forests (ha)*	►**Turnover rates of nutrients, e.g. N, P (kg yr^{-1})** ▼**Leakage of nutrients (kg ha^{-1})** ►*Introduced riparian and aquatic plants (ha)*	▲*Total amount of water use for water treatment (m^3 yr^{-1})* ▼*Average water flow as a potential source of waste disposal (m^3 s^{-1})*	▲*Total amount of wastewater (m^3 yr^{-1})* ▲*Number and efficiency of treatment plants (n.)*
Buffering of flood flows, erosion control through water/land iterations and flood control infrastructure	▲*Natural barriers (dunes, mangroves, wetlands, coral reefs) (ha)* ►**Annual average erosion (kg yr^{-1})** ►**Annual average sediment retention (kg yr^{-1})**	►**Annual sediment retention to reservoirs (kg yr^{-1})** ▲*Vegetation cover (%)* ▲*Number of prevented hazards (n. yr^{-1})* ▲*Conservation of river bank (km)*	▼Required floodplains areas (ha) ►**Sediment exports from catchments (kg ha^{-1})**	▼Required retention areas (ha) ►**Area of wetlands located in flood risk zones (ha)** ►**Sediment exports from catchments (kg ha^{-1})**
Cultural services				
Recreation (river rafting, kayaking, hiking, and fishing as a sport)	►**Turnover from tourism (\$ ha^{-1})** ▼Median water clarity as a measure of swimming suitability (m) ▲*Status of fish population (kg ha^{-1})* ▲*Number of sites with excellent bathing water quality (n.)*	▲*Number of visitors or facilities (e.g. hotels, restaurants, etc.) (n. ha^{-1})* ►*Status of fish population (species and age structure)*	△ Demand of recreation needs (day yr^{-1}) ▲*Average distance from recreation place (km)*	▲*Number of facilities (e.g. hiking paths, and fishing lake…) (n. ha^{-1})* ▲*Number of licenses, permits (e.g. fishing, rafting) (n.)*
Tourism (river viewing)	►**Turnover from tourism (\$ ha^{-1})** ▲*Number of river watching sites (n.)* △ Contrasting landscapes (lakes close to mountains)	▲*Number of visitors or facilities (e.g. hotels, restaurants, etc.) (n. ha^{-1})* ▲*Length of walkway and cycle way (km)*	△ Demand of recreation needs as surrogate indicator of water use for recreation purpose (day yr^{-1})	▲*Number of facilities as surrogate indicator of water use for recreation purpose (e.g. views, hotel, camps, etc.) (n.)*
Existence values (personal satisfaction from free-flowing rivers)	▼Number of spiritual facilities per landscape (n. ha^{-1}) ▲*Number of national parks (n.)*	△ Number of environmental educational-related facilities (n. ha^{-1}) △ Proximity to urban areas of scenic rivers or lakes (km) ▼Monitoring sites (by scientists) (n.)	△ Willingness to protect landscape, national park against extreme events (preferences, e.g. yes, no, not sure) ▲*Number of visitors (to national parks including lakes or rivers) (n.)*	△ Willingness to pay for restoration of natural ecosystem (preferences, e.g. yes, no, not sure) △ Number of scientific project, articles and studies (n.)
Supporting services				
Role in nutrient cycling (role in maintenance of floodplain fertility), primary production	►**Nutrient retention rate (kg yr^{-1})** ▲*Floodplain areas (ha)*	►**Nutrient retention rate in lakes (kg yr^{-1})** ▲*Floodplain areas (ha)*	►**Nutrient export (kg yr^{-1})**	►**Nutrient export to lakes (kg yr^{-1})**
Predator/prey relationships and ecosystem resilience	►*Number and identity of selected species (n.)* ►*Number/area of habitants (n. ha^{-1})* ▼Biodiversity value (e.g. species richness, species composition)	►*Number and identity of selected species in lakes (n.)* △ Ecological-morphological status (preferences, e.g. good, neutral, bad)	►*Harvesting productivity (t yr^{-1})* △ **Natural morphological change (%)**	►*Harvesting data for each species (t yr^{-1})* △ **Anthropogenic morphological change (%)**

Table 1: Indicators to quantify the effects of climate change and human interventions on FWES. The symbols represent the quality of indicators (▲ indicator available and easily understood by non-technical audiences; ► indicator available but not easily understood by non-technical audiences; ▼ indicator not easily available and/or not easily understood by non-technical audiences; △ indicator being proposed and may not be readily available). The font formats represent the type of method (**bold: physically-based**; *italic: empirical*; underlined: participatory and expert-based approach). Abbreviation: n. = number; yr = year. Sources: adapted from Egoh et al. (2012); Kandziora et al. (2013); Maes et al. (2013); Olubode-Awosola (2017); Terrado et al. (2014); Ekka and Pandit (2012).

namely the wet season (from May to October) and the dry season (from November to April). The average river flow in the basin at Son Tay, a meteorological station in Ha Noi, fluctuates from 8000 m^3 s^{-1} to 1500 m^3 s^{-1} (Giuliani et al., 2016). The annual rainfall in the RRB is about 1600 mm yr^{-1} of which about 75% concentrates in the wet season. The annual average temperature, humidity, and evaporation are around 24°C, 80%, and 900 mm, respectively (Duc and Umeyama, 2011).

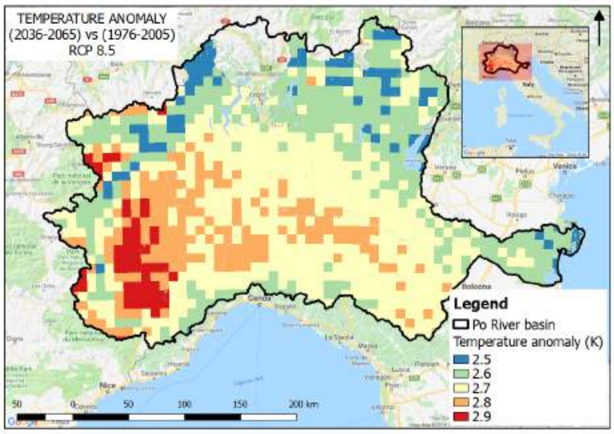
3. Application of the conceptual framework to the case studies

In this section, we discuss the above-mentioned methodological framework and indicators for the selected case studies, namely the RRB and the PRB.. For each case study, we first report the projections of changes, including climate change, land use change, and socio-economic dynamics. Then, we discuss the effects of these changes on FWES using the MEA classification of FWES reported in Figure 1 and the indicators reported in Table 1. Moreover, we focus our review on the provisioning services and two regulating services (i.e. the water purification and sediment control services) because these are the most important services in the two case studies. Yet, as discussed in the following sections, we report as many indicators as possible for these services both in absolute (e.g. nutrient rate - kg ha^{-1}) or relative value (e.g. the reduction of 53% in the monthly discharge) where they are available in the literature.

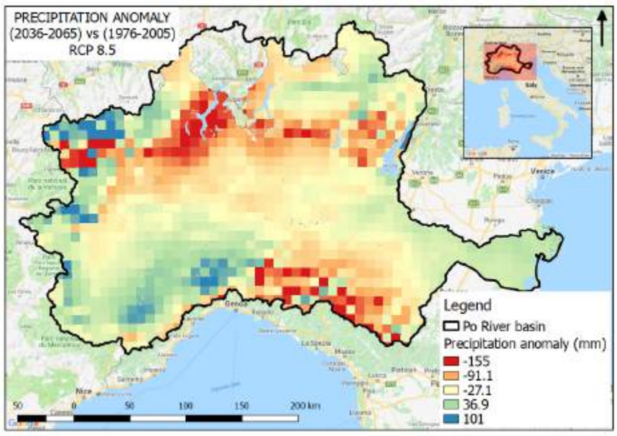
3.1. Projections of climate change in the Po River basin

The results from recent researches demonstrated that the temperature would increase almost in the whole PRB. during the projected period (until 2100) (Aerts et al., 2013; Vezzoli et al., 2015). Moreover, Aerts et al. (2013) demonstrated that positive temperature anomalies (i.e. increased temperature) in the mountain regions are stronger than those in the delta regions. Additionally, the temperature anomaly is strengthened in the further time frame. For instance, the anomaly of the period (2041–2070) is higher than the one of the period (2021–2050). This projection is consistent with the historical data reported by Ciccarelli et al. (2008), where the average temperature of north-western Italy significantly increased (about 1°C) in the period 1952–2002. This trend was also confirmed by Tibaldi et al. (2010), estimating that the maximum temperature of the PRB. increased constantly with a rate of 0.5°C every 10 years, equal to 2°C since 1960, and could reach an average increase of 3–4°C at the end of this century.

Similarly, the precipitation of majority areas of the PRB. would decrease in the future, especially in the mountain areas (Aerts et al., 2013). Coppola and Giorgi (2009) used downscaled AR4 (the Fourth Assessment report) climate scenarios (i.e. A2, A1B, B1[2]) to simulate the rainfall anomaly in Northern



(a)



(b)

Figure 4: Temperature anomaly in K (panel a) and precipitation anomaly in mm (panel b) in the period 2036–2065 (reference period 1976–2005, scenario RCP 8.5). Sources: adapted from Bucchignani et al. (2016)

Italy between 1920 and 2100, and projected an overall reduction in rainfall of about 20%, with local peaks of 40%. This result agreed well with observational data reported in Toreti et al. (2009), showing a decrease in rainfall with a rate of –1.47 mm yr^{-1} in the observational period 1961–2006.

Concerning seasonal variation, a climate model that was developed by Ravazzani et al. (2014) predicted a remarkable decrease in the precipitation in summer and spring. Yet, the results from RCM COSMO-CLM model projected a reduction of precipitation in summer and an increase in autumn and winter within time frame 2041–2070 and 2071–2100 under two climate scenarios, namely RCP4.5 and RCP 8.5[3] (Vezzoli et al., 2015). These results were consistent with another study in the

[2] They are the six emission scenarios, which are published and used by IPCC until the Fourth Assessment report. They include A1 (describes a rapid economic growth, global population) with three families (A1FI, A1T, and A1B), A2 (describes a very heterogeneous world), B1 (describes a convergent world), and B2 (describes an economic, social and environmental sustainability world).

[3] Representative Concentration Pathways (RCPs) describe four different 21st century pathways of greenhouse gas (GHG) emissions and atmospheric concentrations, air pollutant emissions and land use. They include a stringent mitigation scenario (RCP2.6), two intermediate scenarios (RCP4.5 and RCP6.0), and one scenario with very high GHG emissions (RCP8.5).

Veneto and Friuli plain (Baruffi et al., 2012), where the rainfall in the period 2071–2100 would decrease in the summertime while precipitation appeared to be 15% weaker with reference to the period 1971–2000. Finally, it is noteworthy that southern Europe and the Mediterranean region showed a high interannual variability in rainfall and temperatures, and an increase of flood events in the last decades (Coppola and Giorgi, 2009; Ulbrich et al., 2013).

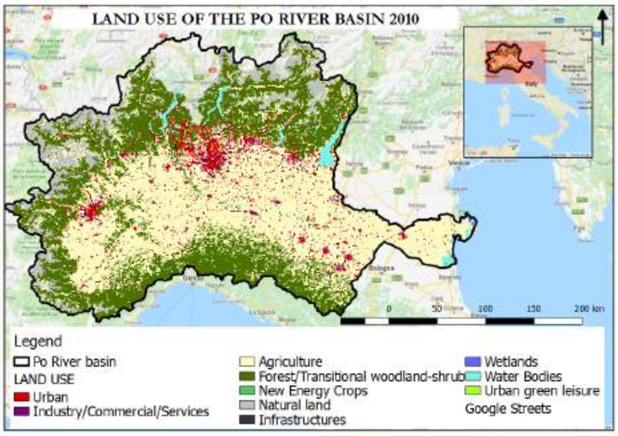
Regarding SLR, the observational data in the North Adriatic Sea region indicated that the rate of SLR varied from 1.2 mm yr^{-1} (in Trieste) to 2.5 mm yr^{-1} (in Venice). Additionally, the estimate of SLR by a multi-model chain from the Euro-Mediterranean Center on Climate Change (CMCC) and the National Research Council-Institute of Marine Sciences (CNR-ISMAR) for the future scenario 2070–2100 are about 17 cm and 42 cm for low and high SLR scenario, respectively (Rizzi et al., 2017).

In conclusion, the major climate-related changes in the PRB. include the increase of temperature, evapotranspiration, river load of nitrogen, seawater salinity, and the decrease of precipitation and river discharge (Giani et al., 2012). These above-mentioned changes (either single factor or combination of some factors), on the one side, are affecting FWES in the PRB. such as the reduction of freshwater availability, decline of water quality, increased frequency of anoxic events, increased frequency and magnitude of extreme events, increased coastal erosion, and salinization of soil, groundwater and surface water. On the other side, CC and SLR are modifying the demands of freshwater services, i.e. higher water demand for irrigation, water treatment, maintaining environmental flow, and for hydropower.

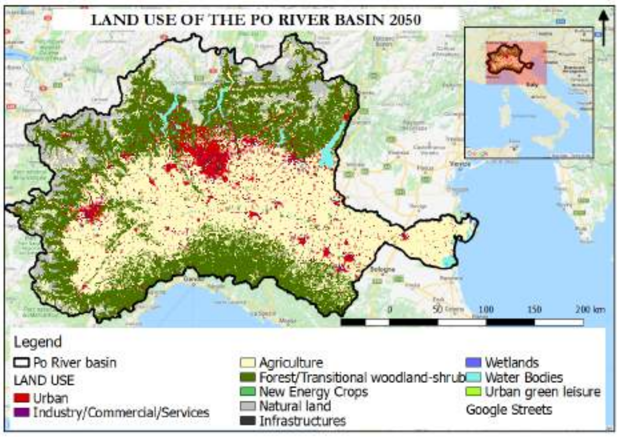
3.2. *Projections of land use and socio-economic change in the Po River basin*

The historical data showed that the PRB. experienced massive changes in land use and land cover. Over the period 2000–2006, the largest loss of forest and semi-natural areas was recorded in Lombardy (26 km^2). The largest increases in the artificial surfaces were observed in Veneto (78.72 km^2), Lombardy (62.52 km^2) and Emilia-Romagna (53.37 km^2). The agricultural lands declined by 78 km^2 in Veneto, 29 km^2 in Emilia Romagna, and 36 km^2 in Lombardy (Amadio, 2012). Regarding land use/land cover in the future, the Land-Use based Integrated Sustainability Assessment model (LUISA) data provides the projections of land use in the future (2010–2050) respect to the updated 2014 configuration (Baranzelli et al., 2014). Figure 5 indicates that artificial areas of the PRB. are expected to increase by about 26.07% in the period 2010–2050 while urban and industry-commercial-services areas are estimated to increase of about 6.2% in the same timeframe. On the contrary, natural land, urban green leisure areas and water bodies are projected to decrease of about 9.61%, 2.15%, and 0.22%, respectively. Regardless of CC, these trends may increase the stress on both the supply side of FWES (e.g. decreasing water availability due to the progressive decline of water bodies) and their demand side (e.g. increasing water demand due to the expansion of urban areas). Nevertheless, land use changes could also trigger some positive trends, which may strengthen FWES. Specifically, a decrease in agricultural areas could lead to decreased water consumption and stress on water quality due to the reduced use of pesticides. In addition, an increase in forest and transitional woodland may increase the precipitation and increase the infiltration capacity of the surrounding rivers, waterfalls, and wetlands.

Regarding the socio-economic dynamics in the PRB., the demographic trend recorded over the 1951–2001 period showed a positive peak (12%) during the 70s, followed by a stabilization of the growth during 1991–2001 (with a negative trend in metropolitan provinces of Milan and Turin, i.e. 5.0% and 3.4%, respectively). The majority of the population of the PRB. (about 64%) lives in the lowland portion of the basin, with higher rates in the metropolitan cores of Northern Lombardy and the Southern Emilia-Romagna (Amadio, 2012). Additionally, the recorded data in the period 1991-2001 indicate that the economic structure was stable with a positive trend in all sectors such as primary (10.7%), tertiary (18.9%), and touristic sec-



(a)



(b)

Figure 5: Land use of the Po River basin in 2010 (panel a) and the projection in 2050 (panel b). Source: adapted from LUISA platform Baranzelli et al. (2014).

tors (27.7%). In conclusion, despite the future socio-economic of the PRB. depends on the European and national policies (e.g. policies on the role/competitiveness of EU countries in the global market), we would expect both a demographic and economic increase in the major cities (e.g. Rome, Milan and Florence), together with an extended suburban sprawl and a diffuse land cover due to the intensification of logistic activities and infrastructures supporting the production (Amadio, 2012).

3.3. *Effects of climatic factors on FWES in the Po River basin*

3.3.1. *Effects on water provisioning services*

Some studies based on the historical data demonstrated that there were decreases in the water volume (Coppola et al., 2014; Ravazzani et al., 2014) and increases in the frequency and the magnitude of extreme events (e.g. floods and droughts) (Coppola et al., 2014). For instance, the observational data indicated a reduction of the 20% of the annual Po River discharge, measured between 1975 and 2010 (Carrera et al., 2013). Moreover, the analysis of longer data record (1920–2009) demonstrated that there was an increase in the maximum annual river runoff (i.e. 9.24 m^3 s^{-1} yr^{-1}), while a decrease in the minimum one (i.e. 0.18 m^3 s^{-1} yr^{-1}) (Montanari, 2012).
Recently, some authors projected a similar trend for the future. For instance, Pedro-Monzonís et al. (2016) projected an outflow reduction of about 33 km^3 in the period (2040–2041) with respect to the reference period (2010–2011) under the climate scenario RCP 4.5, assuming an increase in CO_2 emissions until 2040 and a later decrease to less than the present, approximately 4.2 Pg C yr^{-1}. Furthermore, these negative changes became more complex with seasonal variations. In fact, Ravazzani et al. (2014) demonstrated that monthly discharge in the time period (2041–2050) is expected to increase remarkably in the winter and autumn (279% in February at Villafranca), and to decrease significantly in the summer (–53% in August at Tavagnasco) with respect to the period (2001–2010). Coppola et al. (2014) concluded that the annual discharge in the period (2020–2050) is expected to decrease in all seasons except winter (December, January and February) with respect to the period (1960–1990). The runoff of the northern parts of the basin is projected to increase by 20% during winter in low elevation areas and of 40% at higher elevations. In the fall season (September, October and November), a 20% discharge decrease was found everywhere, with a maximum of –40% in few areas to the far north and south of the basin. Baruffi et al. (2012) reported that, with respect to the reference period (1971–2000), the projected mean winter surface runoff appears to increase by about 60%, while decreasing of about 25–45% during summer in the period (2071–2100). As a consequence of the variations in precipitation and runoff, the frequency of extreme events is projected to increase in the future. The projection of the Standard Precipitation Index (SPI) (Aerts et al., 2013) confirmed that extreme events such as extreme wet (SPI greater or equal to 2) and extreme dry (SPI smaller than –2) may occur more often in the future, with a higher magnitude. These changes are likely to have negative impacts on crop growth and crop yields, water consumption for irrigation and hydropower, the spreading of pests and diseases (Brilli et al., 2014; Lionello et al., 2014). For instance, the outcome of the sub-national Computable General Equilibrium (CGE) model demonstrated that the impacts of flood events were considerable in both absolute and relative terms (i.e. from 3.94 to 10.75 billion euro in values of the year 2000) (Carrera et al., 2015). A study on the socio-economic impact of drought events in the PRB. confirmed that although some farmers experienced very severe effects, agricultural sectors as a whole benefited in economic terms due to the price effect (Musolino et al., 2017). Other case studies in Val DAosta demonstrated that the variation in precipitation pattern would result in a reduction of 10% in the annual hydropower production (equivalent to 200 GWh) (Maran et al., 2014). This reduction could affect the electricity demand of 170,000 home users, the stability of electricity market and other water users.

3.3.2. *Effects on water purification services*

Recent studies indicated that there was a strong correlation between extreme events (i.e. floods and droughts) and the transportation of pollutants to surface freshwater. For instance, flood events accounted for at least one-third of the particulate annual export (organic carbon and nitrogen). The suspended organic material during the flood periods was dominated by soil organic matter (Tesi et al., 2013). Chemicals transported by the river, such as dissolved nitrate (NO_3^-), exhibited a strong relationship with the water flow. Their concentration decreased with increasing flow rate due to the dilution processes. During flood events, the concentration of nitrite (NO_3^-) and ammonia (NH_3) showed the same trend but different compared to because of the positive correlation with the concentration of the suspended material. Additionally, heavy rains and floods caused an increase in sediment load and water turbidity thus reducing water transparency. Variations in light availability in the mixed layer could not only limit algal development but also favor the presence of species tolerant to low light. In contrast to the case of high discharge, lower discharge could lead to a reduction of nutrient loads. For instance, during extremely low discharges like the one occurred in 2005, nutrient loads in the Po river decreased by –70% for dissolved inorganic nitrogen, –58% for , –73% for SiO_2, –50% for total nitrogen and –59% for total phosphorus, comparing with high discharge load as in 1996 (Cozzi and Giani, 2011). Yet, under the extreme dry and warm conditions (e.g. in 2015) the concentrations of nitrates were relatively high (i.e. higher than 10 mg l^{-1}) (Marchina et al., 2017).
Besides, SLR was predicted to have negative effects on water quality by accelerating upward flux of saline water and increasing the presence of arsenic (As), lead (Pb) and Zinc (Zn) in the shallow portion of the aquifers (Colombani et al., 2016a). Furthermore, saline water was further distributed on the surface water in the catchment through the irrigation networks. Additionally, SLR affected negatively the quality of groundwater via seepage of saline groundwater. Colombani et al. (2016b) confirmed that seepage fluxes could increase and, consequently, the salinization of shallow groundwater and surface waters would intensify in the Po River delta. The authors also projected that salt loads in drainage canals would increase by 30% in some areas, and fresh groundwater volumes would be reduced by 46% by 2050.

As a consequence of SLR, the mass fluxes of all the examined trace elements (i.e. As, Pb, and Zn) increased in Po river delta under the analyzed climate change scenarios (Colombani et al., 2016a). This trend was explained by the increase of those elements in the shallow portion of the aquifer, drained by the reclamation network for irrigation.

3.3.3. Effect on erosion and sediment control services

The effects of CC on the coastal sediment dynamics were investigated by Bonaldo et al. (2015) with a complex wave and sediment transport model, fostered by IPCC A1B emission scenario within the period (2070–2099). The authors concluded that there was a reduction of 10–20% in the long-shore transport regime, while the effects on cross-shore transport were generally negligible for ordinary storms (with a return period shorter than 1 year) but could decrease up to 10% for extraordinary events (with a return period longer than 10 years). These modifications would affect morphology and coastal line evolution in the long timescale.

3.4. Effects of non-climatic factors on FWES in the Po River basin

3.4.1. Effects on water provisioning services

Anthropogenic interventions in the PRB. (e.g. land use changes and agricultural activities) induced some negative consequences on water provisioning services. For instance, the results of a case study on a reclaimed area located in the PRB. shown that land use changes had significant impacts on hydrological-hydraulic behaviors. Particularly, the simulated peak flows markedly increased from the 1955 land use scenario to that of 1992, significantly increasing the flood risk in this reclaimed area (Camorani et al., 2005). Moreover, these authors suggested that the expansion of agriculture and or industrial areas could result in an increase of flood risk for less severe frequency rainfall events.

Indeed, the management plan of the Po River Basin District identified that the expansion of agricultural areas had negative effects on the provisioning of freshwater. Specifically, in the Mediterranean area, the water demand of the agricultural system would increase in response to CC, due to the need of compensating the reduction of rainfall. Additionally, the higher temperature may shift the activity period of pests and diseases in the growing season, causing higher consumptions of chemical products (i.e. pesticides) (Brilli et al., 2014).

3.4.2. Effects on water purification services

The last survey of the Italian National Institute for Environmental Protection and Research (ISPRA) over the national territory reported that pesticides were found in 63.9% of 1284 monitoring points distributed on surface waters of the whole Italy. In 274 monitoring points, the pesticide concentrations were found over the quality limit and 89.4% of these were located in the region of the Po River Basin District. The main pollutants of the Po River were total nitrogen, i.e. NO_2^-, NO_3^-, NH_4^+; total phosphorus, i.e. $P_3O_4^{3-}$; SiO_2; total organic carbon; and total suspended matter (Cozzi and Giani, 2011). Published data for the Po river indicated that total nitrogen transport increased by about two times during the last three decades. The highest record was about 173.103 t yr^{-1}, which was reported for the years 1996–2000, due to a constant increase of anthropogenic emissions not fully compensated by the improvements of wastewater management. Castaldelli et al. (2013) argued that farming practices and intensive agricultural activities posed the risk of excess soil mineralization and progressive loss of denitrification capacity in this area.

Concerning macronutrients, Pieri et al. (2011) investigated the quality of water draining from three watersheds, totally or partially cultivated, by determining chemical indicators (NO_3^- and NH_4^+ concentration, N balance), trophic status (chlorophyll-a concentration) and benthic population indexes. From their research, they found the correlation between chemical parameters and land use management and farming practices. Specifically, intensive agricultural activities are linked to the high NO_3^- concentration in water. Moreover, the chlorophyll-a concentration followed the same trend, depending on nitrogen loads. The macronutrient pollution was also investigated by the Po River Basin Authority in five sub-basins at municipality level: Agogna, Oglio (lake downstream), Trebbia, Enza and Secchia. From a preliminary estimation performed by a multiple regression analysis, a significant correlation was found between the amount of the total dissolved nitrogen and the population density/livestock load ($R^2 = 0.85$); and between the amount of the total dissolved phosphorus and population density ($R^2 = 0.93$) (Autorità di Bacino del Fiume Po, 2015).

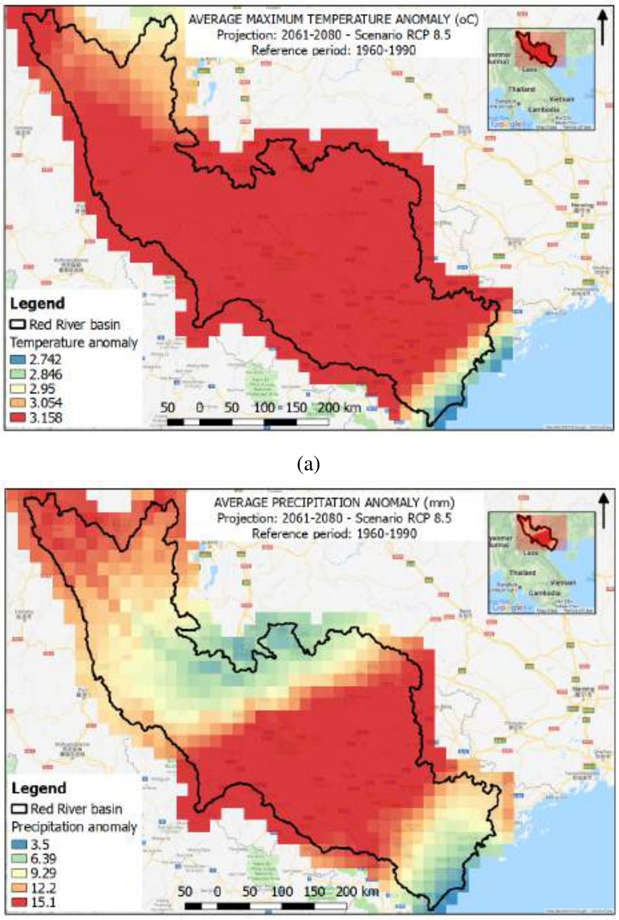
3.4.3. Effect on erosion and sediment control services

A case study on Reno River, located in the lower part of the PRB., investigated the impacts of intensive engineering works (e.g. headwaters, dam construction, torrent control works, and bed material mining) on channel modification and sediment budget. The comparison of four different longitudinal profiles surveyed in 1928, 1951, 1970 and 1998 shown that the bed incision rate in the lower parts of the Reno river, from Bologna to the mouth (about 120 km), was about 3.78 m, with peaks of more than 6 m (Preciso et al., 2012). This streambed degradation led to a reduction of bedload transport rates, which might limit sediment supply downstream of Reno River. Besides, while the impoundment of reservoirs and mobile barrage for hydropower resulted in the local interruption of sediment transport, sand mining changed significantly river morphology and affected transportation activities (Lanzoni et al., 2015).

3.5. Projections of climate change in the Red River basin

Figure 6 reports the spatial distribution of anomalies of maximum monthly temperature and those of precipitation between historical data period (1960–1990) and projection period (2061–2080) under the climate scenario RCP8.5.

This figure demonstrates that the maximum monthly temperature would increase in the whole basin, from 2.74 to 3.16°C by 2080 under the climate scenario RCP 8.5. Giuliani et al. (2015) also concluded that the mean daily temperature is projected to increase in the entire basin, reaching an anomaly of



(a)

(b)

Figure 6: Maximum monthly temperature anomaly (panel a) and monthly precipitation anomaly (panel b) in the period 2061–2080 (referenced period 1960–1990, scenario RCP 8.5). Source: adapted from http://www.worldclim.org/ (Hijmans et al., 2005).

4°C by 2098 under the A1B emission scenario. This projection is consistent with the results reported in Weiss (2009), i.e. the temperature anomaly would reach about 2–4°C by 2100 with respect to the reference period 1980–1999.

Total mean monthly precipitation is expected to increase over time in the entire catchment with a monotonic trend in almost all the sub-basins, varying from 3.5 to 15.1 mm. This result is consistent with Chaudhry and Ruysschaert (2008), where annual total rainfall is expected to increase in the range of 2.5–4.8% by 2050 and 4.7–8.8% by 2100, compared to 1990. Nevertheless, these anomalies have not any even distribution, i.e. decrease in July and August and increase in September and November (Weiss, 2009). More generally, in the northern part of Vietnam, rainfall and daily rainfall intensity would increase of 10–20% over in the future period 2061–2090, compared with the reference period 1961–1990 (Raghavan et al., 2017).

Regarding SLR, Vietnam is experiencing an upward trend of about 2–3 mm per year (Weiss, 2009). The report of Ministry of Natural Resources and Environment stated that the sea level would increase about 30 cm by 2050 and about 75–100 cm by 2100, compared with the average mean sea level of the period 1980–1999 (Duc and Umeyama, 2011; Ha and Pintor, 2014). Consequently, the coastal areas could be affected by saltwater intrusion into aquifers and groundwater resources, leading to the reduction of freshwater availability. Specifically, the coastal areas of the RRB would experience varying levels of damages due to SLR and storm surge flooding. Especially, areas at lower elevations would be flooded more completely and for longer periods, causing greater damages, while areas at higher elevations would experience less severe impacts.

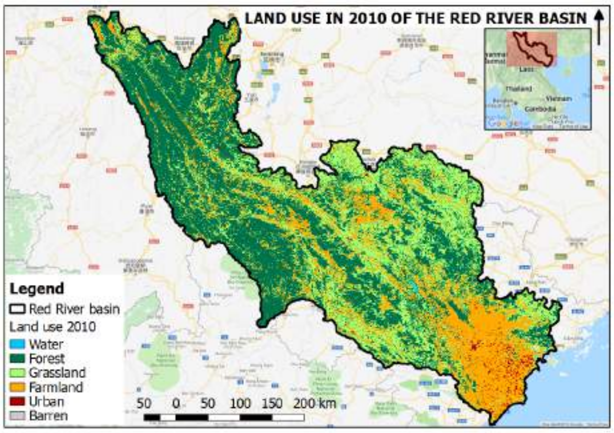
Overall, CC is expected to increase climate variability in Vietnam, decreasing rainfall during the dry season and increasing rainfall in the wet season. Additionally, SLR would lead to permanent inundation of a significant portion of the low-lying Red River delta areas (Neumann et al., 2015). CC is also expected to have considerable impacts on Vietnams fishery and aquaculture sectors, which accounted for 3.9 percent of GDP in 2005 (Chaudhry and Ruysschaert, 2008).

3.6. Projections of land use and socio-economic change in the Red River basin

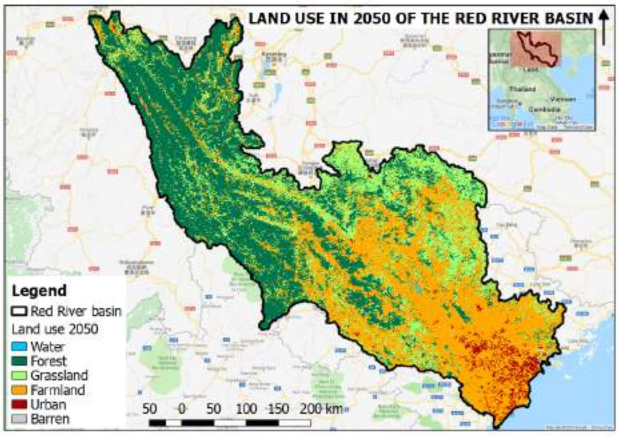
Figure 7 shows the distribution of different types of land use in the RRB in 2010 and the projection of land use in 2050 under the climate scenario B2 (Li et al., 2017). Under this scenario, farmland and urban areas are expected to expanse (i.e. 37.29% and 58.17%, respectively) while forest and grassland areas are expected to collapse (i.e. 20.36% and 34.52%) in the RRB. The intensively cultivated areas such as irrigated, rainfed, mosaic croplands and aquatic regions are mainly located in the lower part of the RRB (Arino et al., 2010).

The historical data showed significant changes in some land use classes such as agriculture, forest and exclusive land. Specifically, agricultural land area dropped by about 5.5%, from 1.97 million ha in 2003 to 1.86 million ha in 2011 (Simone et al., 2013). In the same period, forestland increased from 3.27 million ha to 4.42 million ha, leading to a percentage increase of forest coverage from 36.2% to 48.9%. Additionally, exclusive land area increased from 0.63 million ha to 0.72 million ha in the whole basin. These changes would become more complex in the future depending on climate and socio-economic conditions. For instance, the build-up land would expand in clusters around the largest cities such as Ha Noi and Hai Phong due to urban population growth (Van Dijk et al., 2012). This expansion was driven by the presence of industrial areas and infrastructure located in suburban areas at a distance of about 70–140 km from these big cities.

Regarding socio-economic dynamic of the RRB, according to the Provinces report in 2009, the economic growth rate was about 8–12% per year, in which the industrial growth was about 15–30% (higher than the national average rate) and the urban population growth was about 35%. Nevertheless, the industrialization was not uniform. While higher industrialization rates (over 20%) were observed in Hanoi, Quang Ninh, Hai Phong,



(a)



(b)

Figure 7: Land use the Red River basin in 2010 (panel a) and the projection in 2050 under the climate scenario B2 (panel b). Source: adapted from Li et al. (2017)

Hung Yen, Hai Duong and Vinh Phuc, the lower ones were detected in the agricultural-based provinces such as Thai Binh, Ninh Binh, Nam Dinh, Phu Tho, and Ha Nam (Trinh, 2009). In addition, many towns were upgraded to cities, such as Thai Binh, Phu Ly, Ninh Binh, Hai Duong, Hung Yen, Bac Ninh and Vinh Yen.

Several development objectives were targeted in the Master Plan for the socio-economic development in the RRD through 2020 (Minister, 2013). Specifically, the socio-economic targets were set for 2020, such as increasing the regions contribution to countrys GDP from 24.7% (in 2010) to 28.7%; increasing the average income per capita to around USD 4180 (1.3 times of the national average); the regions average population growth would be around 0.93% in the 2011–2020 period and regions population would reach 21.7 million. Obviously, all these targets would increase stress on FWES on both supply side and demand side. In addition, numerous environmental promotion objectives were targeted to eliminate environmental pollution in both rural and urban areas to ensure sustainable development, such as treating at least 95% of medical waste, and centralizing wastewater treatment systems for at least 85% of operating industrial parks and export processing zones.

3.7. Effects of climatic factors on FWES in the Red River basin

3.7.1. Effects on water provisioning services

The main climatic factor affecting freshwater provisioning in the RRB is the precipitation related to the South Asia Summer Monsoon (SASM) and El Nio Southern Oscillation (ENSO) (Gao et al., 2015). These authors found a positive correlation between precipitation and river discharge, confirming the high dependence of discharge on precipitation. They also observed the decrease in river flows and water levels in many dams and reservoirs, particularly during El Nino years. These changes were dominated by precipitation and closely related to the SASM and ENSO. Specifically, El Nino events over the RRB might induce lower water discharge with possible droughts. La Nia events might cause higher water discharge with possible flood events (Gao et al., 2015). Duong et al. (2016) found that the mean monthly discharge increased of about 20–40% in the wet season and decreased up to 30% during the dry season in the period (2026–2035), respect to reference period (1979–2003). Additionally, these authors projected that the magnitude of high flow regimes would increase of 20–60%, while the magnitude of low flow regimes would slightly decrease of 10–20%. Consequently, these changes would increase water deficit, water shortages, conflicts and competitiveness among water users (e.g. agriculture, water supply and hydropower) with relevant impacts on the supply side of FWES. Since irrigation is the most important freshwater usage, the lowering of the water level in the Red River and its canal network would reduce the irrigated ability and, consequently, the crop productivity. Yet, decreased river water levels and river water discharges would lead to further saltwater intrusion in the coastal areas and their neighbors. It is also expected that flood events would increase in both magnitude and frequency due to increased high flow regimes in the flood season. In addition to water volume variations, extreme events (e.g. storm surge events) were expected to occur more frequently due to SLR (Neumann et al., 2015). These authors augured that the historical 100–year event in Ha Noi would happen every 65 years, 59 years and 49 years by 2050, corresponding to the low SLR, medium SLR, and high SLR scenarios, respectively. As a consequence of these extreme events, over 70% of areas with high-valued land uses, such as urban residential, rural residential, commercial and industrial uses would be at risk of flooding. In addition, over 90% of rice paddies would suffer from floods.

3.7.2. Effects on water purification services

One of the main drivers affecting the freshwater purification services in the RRB is SLR. The Ministry of Natural Resources and Environment reported that the saltwater intrusion extent had already increased of 7–15% in the period (1965–1985), with respect to the reference period (1993–2007). Duc and Umeyama (2011) predicted that saltwater intrusion would increase up to 31–50% when SLR would reach 100 cm, by the year 2100. Positive correlations between the extent of salinity intrusion and SLR were found in all branches of the Red River such as Tra Ly, Ninh Co and Day. At Tra Ly branch, the location of 1-psu salinity (the upper threshold value for drinking water) would increase of 7%, 26%, or 40% over the present distance when the SLR would be +30 cm, +75 cm, or +100 cm, respectively. The corresponding values would be 13%, 24%, and 50% for the Red River; 8%, 23%, and 31% for the Ninh Co; and 10%, 23%, and 29% for the Day. The location of 4–psu salinity (the marginal value of irrigation water) would change noticeably when the sea level would increase up to 100 cm. Consequently, the increased rate of the extent of salinity intrusion would be 43% for the Tra Ly, 47% for the Red River, 33% for the Ninh Co, and 37% for the Day (Duc and Umeyama, 2011). Thus, it was foreseen that SLR and its alterations would affect FWES. On the supply side, SLR would reduce inland territory as well as cultivation areas. Moreover, the salinity intrusion would further decrease the availability and quality of surface and groundwater in coastal areas and their nearby.

3.7.3. *Effect on erosion and sediment control services*

Duc et al. (2012) analyzed the impact of SLR on the erosion rate in coastal areas of the Red River delta. The raw estimation data at the south Hai Thinh commune were approximately 0 and 11 m yr^{-1} during the period 1965–1985 and 1985–1995, respectively. The author concluded that the erosion rate increased by 12% from 1995 to 2005 due to SLR.

Regarding sediment load, with respect to the conditions of the period 1997–2004, the model predicted an increase of about 20% of the suspended matter load, i.e. from 40 to 48.106 t yr^{-1}, due to enhanced rainfall, with an increase of 10% in rainfall and an increase of 5% in potential evapotranspiration (Le et al., 2007). van Maren (2007) reported that sediment was transported along the Red River and was deposited close to the river mouth with the rate of 20 to 25.106 t yr^{-1} at Ba Lat river mouth.

3.8. *Effects of non-climatic factors on FWES in the Red River basin*

3.8.1. *Effects on water provisioning services*

Dams construction is the main factor affecting the water availability and the water distribution in the RRB (Gao et al., 2015). First, the impoundment and the regulation of the Hoa Binh dam (HBD) has led to the reduction of water discharge in the wet season (14% in the Red River at Son Tay) and the increased of water discharge in the dry season (12% at the same station) (Vinh et al., 2014). This result was consistent with (Gao et al., 2015), showing that after the regulation of the HBD (from 1990), water discharge increased in January–May while decreased in June–December. Overall, a downward trend of annual discharge was observed, from 3541 $m^3\ s^{-1}$ to 3300 $m^3\ s^{-1}$ at the Son Tay station before and after the construction of the Hoa Binh reservoir, respectively (Lu et al., 2015).

Additionally, the impoundment of the Hoa Binh reservoir affected the water distribution among tributaries of the RRB. Specifically, the water discharge ratio of the estuaries in the north of the RRD slightly increased after impoundment, ranging from 15.5% to 16.5% for the Cam and Bach Dang estuaries, from 3.5 to 3.7% for the Lach Tray River, from 13.6 to 14.5% for the Van Uc River, and from 6.0 to 6.4% in the Thai Binh River (Vinh et al., 2014).

Consequently, the impoundment and operation of reservoirs would have strong impacts on FWES and human being. For instance, depending on the climate scenario and the time horizon, the operation of reservoirs would change on average from –7 to +5% in hydropower production, +35 to +520% in flood damages (Giuliani et al., 2016). By integrating future climate on the current operations, we could reduce the loss of hydropower to 5%, potentially saving around 34.4 million US\$ yr^{-1} at the national scale.

Other factors affecting the provisioning services are the change in land use/land cover (e.g. the increase of forest cover) and soil conservation practices (e.g. strip grass barrier, contour hedgerow system, and crop residue) (Ngo et al., 2015). A study concerning the distribution of land cover in the RRB, based on the Globcover 2009 dataset (Arino et al., 2010), indicated that the total irrigated area was about 869,029 ha (Simons et al., 2016). Therefore, changes in land use/land cover (e.g. from agricultural fields to industrial areas) could significantly affect water consumption, water quality, evaporation rate and, ultimately, FWES. For instance, change in land use type between 1995 and 2005 from forest to field crop and urban areas strongly contributed to an increase in the average annual runoff from 182.5 to 342.7 mm. Finally, a decrease of runoff from 342.7 to 167.7 mm between 2005 and 2010, was due to the expansion of forested areas and the application of soil conservation practices (Ngo et al., 2015).

3.8.2. *Effects on water purification services*

Land use and land cover are the main factors affecting the nutrient cycle and biomass production in the RRB. A study in the Red River estuary, using LOICZ–CABARET budget model, indicated that mangrove system was a net sink of nutrient, i.e. its sequestration rate was about 26,000 kg N d^{-1} and 3100 kg P d^{-1} for their biomass production (Wösten et al., 2003). The authors also explored that these rates would change in the future corresponding to different scenarios. For instance, either increasing river discharge in combination with constant river nutrient concentration or constant river discharge in combination with increasing river nutrient concentration would result in increasing nutrient concentration in estuary areas. Beside the role of the primary source of energy and nutrients, mangrove forests had positive roles in the stabilization of shorelines, minimizing wave damages and trapping sediments (Bao, 2011). The protection level of this ecosystem depended on their structure, characterized by the high, the density and the bandwidth. Moreover, the conversion from mangrove forests to agriculture or aquatic areas induced strong alterations in soil properties and carbon dynamics, i.e. mangrove clearing not only turned a mangrove from a carbon sink to a carbon source, but also from a sink to a source of trace metals (Grellier et al., 2017). This result was consistent with Ha et al. (2018), i.e. mangrove forests had a

high potential for storing below ground.
Additionally, human disturbances such as untreated sewage from metropolitan cities and runoff from agricultural fields had significant impacts on FWES in term of quantity (e.g. higher demand for agriculture) and quality (e.g. through anoxic conditions, high nutrient concentration and low dissolved oxygen condition) (Hoang et al., 2018; Viet et al., 2018). The water demand for agriculture would increase about 8 bills m^3 yr^{-1} (from 13.65 bill m^3 yr^{-1} to 21.5 bill m^3 yr^{-1}) in 2020 due to the expansion of agriculture. Yet, the distribution and structure of phytoplankton and zooplankton in the Day River, a part of the Red River, depended on physical, chemical and biological factors, affecting by human activities such as changes in the nutrient form of freshwater effluence.

3.8.3. *Effect on erosion and sediment control services*

The impoundment of the Hoa Binh reservoir had strong effects on sediment discharge and carbon budget in the Da, Red and Duong rivers. The annual total suspended sediment discharge of the Da River before and after the presence of HBD were 65.0×10^6 t yr^{-1} and 5.8×10^6 t yr^{-1}, respectively. Consequently, the annual total suspended sediment of the Red River at Son Tay reduced to 46×10^6 t yr^{-1} after the impoundment (i.e. a reduction of 61%), corresponding to a decrease in the average carbon from 1030 mg l^{-1} to around 400 mg l^{-1} (Vinh et al., 2014). Dang et al. (2010) concluded that the presence of the Hoa Binh reservoir on the Da River system reduced the flow velocity, consequently, leading to the accumulation of sediments in the reservoir. Approximately, the mean annual sediment trapped by the Hoa Binh reservoir was estimated at 53.7–81.6×10^6 t yr^{-1}, accounting for a reduction of 52%–80% of the total suspended particulate matters flux at the Son Tay site. Furthermore, the impoundment and operation of the Son La dam, on the Da River, upstream of the Hoa Binh reservoir, would further decrease the suspended load by about 50%. Finally, the construction of the Thac Ba reservoir in the Chay river and the Tuyen Quang reservoir in the Gam river resulted in a reduction of about 95% and 71% of suspended sediment load, respectively (Ranzi et al., 2012).
Besides, land use and land cover change affected the sediment load in the RRB. Specifically, the conversion of 11.07% forestland to agricultural land caused an increase of 8.94% in sediment load in Cau River Catchment (Khoi and Suetsugi, 2014). Ranzi et al. (2012) showed that a 35% decrease in forest areas, of which 20% was converted into rice fields and agricultural crop areas and 15% was converted into bushes, shrubs and meadows, resulted in a 28% increase in sediment load of the Lo River. Khoi and Suetsugi (2014) also reported that a decrease of 14.07% in forest and an increase of 14.89% in cropland led to an increase of 25.4% of sediment load.

4. Discussion and conclusions

The proposed conceptual framework, together with the list of selected indicators, provides a comprehensive overview of FWES including their interaction with the socio-economic system. Yet, it also highlights the main impacts of global changes on the supply side and the demand side of the ecosystems. These tools provide the basic instruments to identify and classify the main stressors and impacts of global drivers (e.g. climate change and human activities) on FWES, while conducting a risk assessment. When applying this framework to the Red River basin and the Po River basin, we encountered difficulties to find information about all the selected indicators, due to the incompleteness of literature and the inconsistency of terminology and classification among different authors and publications. Nevertheless, these challenges highlighted the following important aspects in both case studies.
In the PRB, CC could affect the ability of freshwater ecosystem to provide supplies by altering water quantity (e.g. water volume/level) and quality (e.g. salt instruction of surface and groundwater). Yet, CC could increase the frequency and magnitude of extreme events in autumn and spring, with an intensification of damages on human activities. Moreover, the higher temperature could shift the time of flows, with a shift from a winter minimum to a late summer minimum, and an increase in the severity of droughts in summer. Yet, the higher temperature would increase water demand for all water usages, namely irrigation, domestic and hydropower. Consequently, we would expect some negative changes on both the supply and the demand side of ecosystem services, i.e. decreasing in water availability and increasing water consumption. Therefore, we argue that extensive regulation and technological improvements are needed to reduce the conflict and competitiveness among water users in the future.
Besides the climatic factors (Section 3.3), the expected demographic growth, urbanization and industrial progress would have great impacts on FWES through land use change, landscape modification, increased demand and pollutions. Because of industrial, agricultural and household pollutants, excessive organic content in surface water could cause eutrophication in rivers and lakes. Yet, groundwater is continuously contaminated by a high concentration of nitrates originated from fertilizer consumption in agriculture. Therefore, there is a need for reinforcing water regulations to cope with these non-climatic drivers and to maintain the function of FWES.
In the case of the RRB, the main climatic factors affecting FWES are the reduction of water availability in the delta part due to decreased precipitation; the SLR and storm surge events altering these services in the coastal areas due to saltwater intrusion. Noticeably, these impacts are not separable, but they are interacting and magnifying each other. For instance, the lower water levels in rivers, canals, and groundwater table lead to the intrusion of saltwater, that is further altered by SLR .
Besides the climatic factors (Section 3.7), the impoundment and regulation of cascade reservoirs along the main streams is the major anthropogenic factor altering FWES in the RRB. On the one side, the operation of the four big dams (i.e. Hoa Binh, Thac Ba, Tuyen Quang, and Son La) protected the population of the RRB, especially the Ha Noi capital, from flood risks, thus reducing significantly damages and losses. On the other side, this operation changed the natural regime, water availability, water level and, consequently, aquatic diversity.

Moreover, the operation of upstream reservoirs in China had clear influences on the flow regimes in the Vietnamese part (Phuong Nam and Van Anh, 2010). Therefore, we emphasize the importance of the regulation of multi-reservoir and the cooperation with Vietnamese neighbors in managing shared river basins.

Additionally, the demographic and urbanization development are other stressors for FWES. On the supply side, urbanization and industrialization would reduce water body areas, leading to the reduction of freshwater availability. Yet, water pollution from the domestic and industrial areas is the main factor contributing to the degradation of water quality in many rivers and lakes such as the part of Nhue River in Ha Noi, creating pressures on the freshwater ecosystems. On the demand side, urbanization and industrialization are expected to increase the water consumption (e.g. demand for wastewater treatment). Thus, in order to ensure sustainable development and maintenance of FWES in RRB, it is necessary to comply with the environmental objectives targeted in the Vietnamese Master Plan (e.g. treating medical waste, centralizing wastewater treatment systems) and the related environmental regulations.

In summary, the effects of global change on FWES vary in time and space, depending on geographical, climatic and socio-economic conditions. Specifically, while CC is considered as one of the main drivers of changes for FWES in the PRB., human activities (i.e. impoundments and dams operations) are considered as the major drivers of ES alterations in the RRB. Secondly, in these case studies, most of the current researches focus on the provisioning and regulating of FWES, namely water quantity and water quality, while other services such as cultural, and supporting services have not been well investigated yet. Finally, even though CC and human interventions affect FWES on both the supply and demand side, these impacts are mainly investigated on the supply side while those on the demand side are under-presented in both case studies. In fact, it is not straightforward to separate the impacts into these two categories, due to the complex dynamic interactions between the natural system and the socio-economic system.

To conclude, since FWES have not been explicitly targeted as the assessment endpoints in the analyzed case studies, we propose that future research on FWES shall define a common and consistent terminology, and classification regarding drivers (e.g. climatic and non-climatic drivers), FWES components (e.g. provisioning, regulating, cultural, and supporting services), and freshwater ecosystem categories (e.g. supply and demand side). These efforts could enhance the utility of past and current knowledge for the definition of climate change adaptation measures and risk mitigation strategies at different scales (e.g. local, national, regional and global scale). Moreover, it is important to address the potential compound effects of different CC components (i.e. the impacts related to combinations and interactions of climate-related hazards) as well as the synergy and potential cumulative effects of climate and non-climate factors, by using interdisciplinary multi-risk modelling approaches.

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List of acronyms

ES Ecosystem Services

FWES FreshWater Ecosystems Services

CC Climate Change

SLR Sea Level Rise

RRB Red River Basin

PRB Po River Basin

ENSO El Nio Southern Oscillation

CMCC Euro-Mediterranean Center on Climate Change

HBD Hoa Binh dam

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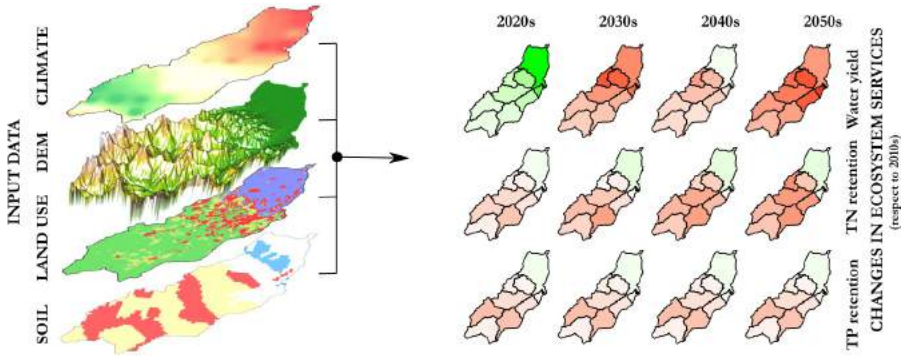
PAPER 2: COUPLING SCENARIOS OF CLIMATE AND LAND USE CHANGE WITH ASSESSMENT OF ECOSYSTEM SERVICES AT RIVER BASIN SCALE



GRAPHICAL ABSTRACT OF THE SECOND PAPER

HIGHLIGHTS

- Integrated risk assessment of climate and land use changes on ecosystem services;
- Comparing to land use change, climate change has a greater effect on water yield;
- Nutrient retentions are more sensitive to land use change than climate change;
- Useful insights for water resources management and climate adaptation are provided.

Coupling scenarios of climate and land–use change with assessments of potential ecosystem services at the river basin scale

Hung Vuong Pham[a,b], Anna Sperotto[a,b], Silvia Torresan[a,b], Vicen Acua[c], Ddac Jorda-Capdevila[c], Guido Rianna[b], Antonio Marcomini[a,b], Andrea Critto[a,b,*]

[a]*Euro-Mediterranean Center on Climate Change (CMCC), via Augusto Imperatore 16, 73100 Lecce, Italy*
[b]*Department of Environmental Sciences, Informatics and Statistics, University Ca Foscari Venice, Via Torino 155, 30172 Venezia-Mestre, Venice, Italy*
[c]*Catalan Institute for Water Research (ICRA), Carrer Emili Grahit 101, 17003 Girona, Spain*

Abstract

Climate and land-use changes are posing increasing threats to freshwater-related ecosystem services, acting both on the supply and the demand side. A better understanding of the dynamics of these potential services, driven by the interactions between the factors mentioned above, could bring benefits to water resources management, the environment, and human well-being. In this work, we developed an integrated modeling approach to assess the conjoined impacts of land-use and climate changes on the potential ecosystem services (i.e. water yield and nutrients retention) until 2050. This approach was applied to the Taro River basin in Italy. Firstly, the results showed a 20% reduction in water yield was driven mainly by the increases in evapotranspiration demand and changes in rainfall patterns. Furthermore, a mean decrease of approximately 3% of the total nitrogen retention and a mean increase of 3% for the total phosphorus retention could be mainly attributed to land-use changes. Secondly, the rate of change would be different over time with the most pronounced differences between 2020-2030 and slower variations afterward. Finally, the obtained results could be a valuable support to identify and prioritize the best management practices for sustainable water use, balancing the tradeoffs among services.

Keywords: climate change, human activities, potential ecosystem services, water yield, nutrient retention,

1. Introduction

Freshwater ecosystems (i.e. rivers, lakes, and groundwaters) support the delivery of potential ecosystem services (ES) crucial for the hydrological cycle including water provision, nutrient regulation, and sediment retention (Aylward et al., 2005). When these potential ES bring actual benefits to humans, they become real ecosystem services or ecosystem services, which were used by most authors (Braat and de Groot, 2012). For instance, water yield (i.e. potential water from a landscape) can generate water flow providing water for irrigation in cultivated areas; nutrient retentions needed for water purification and clean drinking water.

Europe's waters are affected by several factors, such as water abstraction, water pollution, droughts, and floods (Kristensen et al., 2018). Specifically, the effects of climate change (e.g. variations in temperature and precipitation) on the hydrological cycle are recognizable in several areas worldwide in terms of occurrence and magnitude of extreme events and related impacts, water availability, retention and export of chemical pollutants (Khoi and Suetsugi, 2014). Land-use and land cover may exacerbate these changes by the modification of soil water balance and nutrient inputs as the result of agricultural transformation and urbanization (Vaighan et al., 2017). All these changes are likely to strongly affect ES provision, putting human well-being at risk (Däll and Zhang, 2010). Therefore, the identification of significant pressures and the quantification of their impacts on freshwater-related ES have become crucial in adapting to global change (Pervez and Henebry, 2015).

The concept of ecosystem and freshwater-related ES has drawn extensive attention among scientists due to its relevance to management options associated with the synergy and trade-offs between ES (Xu et al., 2018). Nevertheless, there were still some gaps and limitations such as terminology confusion, inconsistent indicators, parameter verification and non-systematic evaluation of ES (Braat and de Groot, 2012; Xu et al., 2018).

In this context, several case studies on freshwater-related ES have examined the quantification of supply changes (Karabulut et al., 2016; Khoi and Suetsugi, 2014; Lin et al., 2019; Pasini et al., 2012; Pervez and Henebry, 2015; Polce et al., 2016; Vaighan et al., 2017; Weber et al., 2016), risk screening (Paroissien et al., 2015; Sample et al., 2016; Sperotto et al., 2019), mitigation policies (Giakoumis and Voulvoulis, 2018; Jorda-Capdevila et al., 2019; Nel et al., 2017) and potential tradeoff among ES (Hoyer and Chang, 2014; Hu et al., 2019; Wang et al., 2019). These works were carried out by different advanced models such as InVEST, ARIES, SolVES, MIMES, EPM, InFOREST, Envision, EcoMetrix, and LUCI (Xu et al., 2018). In Europe, the Water Information System for Europe (WISE) platform provided the assessment of status and pressures on European water based on the Water Framework Di-

*Corresponding author
Email addresses: vuong.pham@unive.it (**Hung Vuong Pham**), critto@unive.it (Andrea Critto)

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rective (WFD) and River Basin Management Plans (RBMPs) data (Kristensen et al., 2018). Despite these, further efforts are required to address existing gaps concerning the scale of these case studies, their time frame, and the assumptions about land-use change. Firstly, most studies have focused on the large scale, such as at the European level (Polce et al., 2016) or basin level (Pervez and Henebry, 2015), but not on the local scale (e.g. sub-basin). Secondly, most studies have projected the long-term impacts of global change on ES between 2050 and 2100. Furthermore, land-use change data was often derived from simple land models such as CLM (Polce et al., 2016), NLCD 2006 (Hoyer and Chang, 2014), iCLUE (Jorda-Capdevila et al., 2019) and CLUE-S (Paroissien et al., 2015). Finally, many authors investigated the single effects or sole stressors on ES (Cozzi and Giani, 2011; Mazzoleni et al., 2014b; Migani et al., 2015; Vezzoli et al., 2015, 2014) and neglected to consider the potential cumulative impacts between multiple drivers such as global climate variations and local land-use change.

To bridge these gaps, the aims of this study are: *(i)* to understand the future dynamics of potential freshwater-related ES at the sub-basin level; *(ii)* to quantify the conjoined effects of climate and land-use change on these potential services as well as to break down the relative contribution of each driver; *(iii)* to provide appropriate indicators to support the identification of proper practices to improve ES management at the local level.

To achieve these objectives, starting from the conceptual framework for ES proposed in Pham et al. (2019), we developed an integrated modeling approach. This work incorporated the Integrated Valuation of Ecosystem Services and Tradeoffs model (InVEST) with climate and land-use change models. This approach allowed us to dynamically project water yield (WY), total nitrogen (TN), and total phosphorus (TP) retentions until 2050. The proposed model was applied to the Taro River basin, a sub-basin of the Po River basin in Italy. This approach is transparent and reproducible since the inputs are freely available. Input data included land-use data from the Land Use-based Integrated Sustainability Assessment modeling platform (LUISA) (Baranzelli et al., 2014), climate data from a Regional Climate Model (COSMO-CLM), optimized over Italy by Bucchignani et al. (2016), and soil data from European Soil Data Center (ESDAC) (Panagos et al., 2012).

After a brief introduction to the case study area (Section 2.1), the paper describes the methodological steps and input data used to implement the integrated modeling approach (Section 2.2). It then reports the results obtained for the Taro river basin (Section 3), discusses these results (Section 4) and highlights their implications for sustainable management of ES with future global changes scenarios (Section 5).

2. Materials and methods

2.1. Case study

2.1.1. Geographical features

The Taro River, located in the Italian Northern Apennines in the Emilia-Romagna region, flows 126 km from the Apennine's main divide to the Po River. Its total drained area covers 2026 km^2, of which 800 km^2 are in the floodplain areas (see Figure 1).

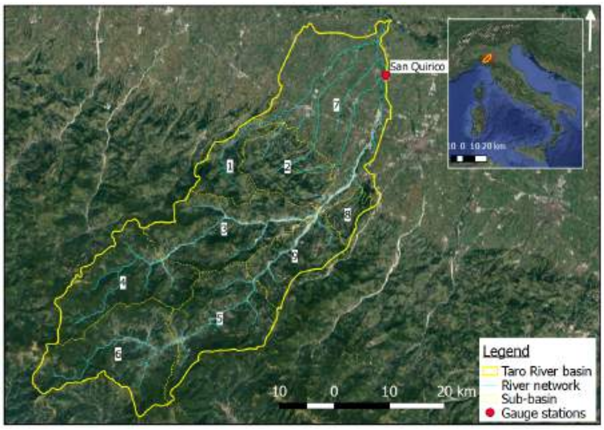

Figure 1: Taro River basin and its sub-basins.

The mean annual precipitation and temperature over the period (1999-2011) averaged 1200 mm and 12°C, respectively (Clerici et al., 2015). The yearly mean discharge over the period 1916-2013 at the gauge station of San Quirico (see Figure 1), the closest to the river mouth, was 31 $m^3 s^{-1}$ (Clerici et al., 2015).

2.1.2. Pressures for ecosystem services

The Taro River Basin (TRB) is mainly characterized by woodlands (53%) in the upper and the middle parts, agriculture (42%) in the lower parts while urban areas account for only 3% of the total land-use. Its surface water is used primarily for agriculture and is directly abstracted from the stream by the Reclamation Consortium (i.e. Consorzio di irrigazione parmense, Societ Canale Naviglio Taro). Besides, freshwater is served for the industrial sector, transportation, water regulation, and wastewater treatment (ADBPo, 2016; Dio et al., 2010; Gaglio et al., 2017). Drinking water, on the other hand, is extracted from the groundwaters and managed by Emiliambiente and IRETI. Accordingly, the lowland area of TRB has been heavily modified due to the construction of engineered flood defenses, irrigation channels, and inland waterways for navigation. The intensive agricultural practices, characterized by the use of fertilizers and the rising demand for irrigation, are increasingly threatening the environmental status of freshwaters and ecosystems in the TRB. The most significant threats are morphological changes (Clerici et al., 2015), flooding (Mazzoleni et al., 2014a), chemical pollution (Sammartino et al., 2007), changes in the quality of the surface water (Madoni and Zangrossi, 2005), and the quality of the groundwater (Iacumin et al., 2009; Toscani et al., 2007). In recent years, the TRB has suffered from droughts in summer periods, making it impossible to satisfy the demands for irrigation and maintaining the minimum environmental flow.

These issues could affect relevant potential ES and the actual uses of those services of the rural/urban community of the

TRB (see Table 1). These related services were identified based on the literature review, the analysis of available data, the participatory engagement of local policymakers, and stakeholders during workshops and national meetings within the PROLINE-CE project . The participants include local authorities from the Po river basin authority (ADBPo), the Interregional Agency for the Po River (AIPo), Commune of Parma, the multiutility leader in environmental water and energy services (HERA), Consortium of energy and water (CEA) and Soil protection service of Emilia Romagna Region.

These identified freshwater-related services are crucial for the development of the TRB and Emilia-Romagna region. Irrigated agriculture plays an essential role in contributing to domestic food security in the TRB. Moreover, crop productivity depends on adequate allocations and quality of water to agriculture. Besides, water demands for non-agricultural users, namely households and industry, have increased during the last decades due to the growth of the population and industrialization process. Therefore, there is a need to improve water quantity and water quality by enhancing the potential freshwater ES such as water yield and nutrient retention. This improvement may reduce the cost of water reallocation and water treatment. Moreover, it is also essential to understand how ecosystems services provided by freshwater ecosystems in the TRB could be affected by land-use activities and the effects of climate change. This information allows to re-allocate the water resource in more equitable distribution and to mitigate the potential conflicts among water users.

2.2. *Integrated modeling approach*

To address the gaps mentioned above, we developed an integrated modeling approach to provide a useful tool for exploring the likely outcomes of alternative management options, climate, and land-use change scenarios, and for evaluating trade-offs among freshwater services (Figure 1).

The integrated model consists of 3 main components, namely climate projections, land-use projections, and the ecosystem model. The climate and land-use data were obtained from simulations developed by CMCC and LUISA platform, respectively, while the ecosystem model was the core of this integrated modeling approach. Firstly, the LUISA platform projected land-use at 1km resolution based on the EU reference year 2014, which was set as the baseline scenario for every ten years until 2050 (Baranzelli et al., 2014). This baseline scenario configuration of LUISA was aligned with socio-economic trends, business as usual urbanization processes, and the effect of established European policies. Secondly, the Regional Climate Model COSMO-CLM, with the configuration optimized by Bucchignani et al. (2016) on Italy, provided high-resolution data (i.e. 8km x 8km) under the greenhouse gases concentration scenario RCP 8.5 IPCC (2014). Thirdly, ESDAC provides accurate, updated, and reliable information on soil, taking into consideration EU directives and policies (Panagos et al., 2012). These sources of data provided dynamic inputs for the ecosystem model (i.e. InVEST), which allowed assessing the potential ES in the reference year, and future changes within the baseline scenario. In this analysis, the baseline scenario refers to configurations of socio-economic trends, urbanization processes, and established European policies (Baranzelli et al., 2014) under the greenhouse gases concentration scenario RCP 8.5 (IPCC, 2014). The outputs of the InVEST model were the sets of dynamic maps and indicators, which explored the likely outcomes of alternative management options, climate, and land-use change scenarios. These results allow evaluating the trade-offs among water users and freshwater services.

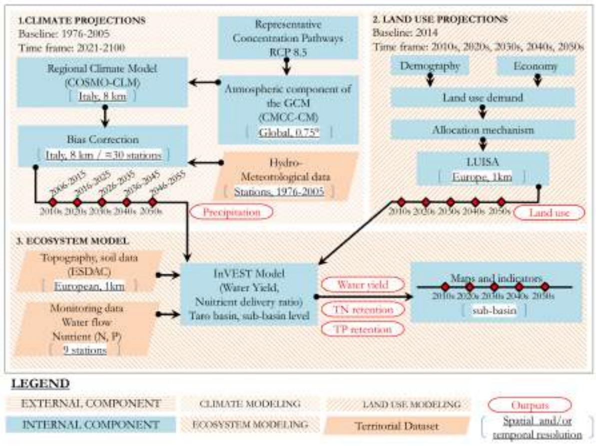

Figure 2: Integrated modeling approach to quantify the impacts of climate change and land use change on freshwater ecosystem services. Sources: Adapted from (Baranzelli et al., 2014; Ronco et al., 2017)

Noticeably, these input data had different spatial and temporal resolutions. Therefore, they were aggregated into a standard scale of the analysis, based on the requirement of the model. Specifically, the ecosystem model (i.e. InVEST) required annual datasets; thus, daily precipitation was converted into the mean annual precipitation. The technical description of the input data, supplying to each component of the integrated modeling approach, is described in sections 2.2.1, 2.2.2, and 2.2.3.

2.2.1. *Climate projections*

In this study, climate projections were provided by adopting the well-consolidated downscaling approach. The outputs of the General Circulation Model (Scoccimarro et al., 2011), with a spatial resolution of $0.75° \times 0.75°$, were dynamically downscaled utilizing the Regional Climate Model COSMO-CLM (Rockel et al., 2008). The COSMO-CLM was nested on the global model for the whole Italy with a spatial resolution of $0.0715° \times 0.0715°$ (≈ 8 km $\times$ 8 km). For the test case (i.e. 2026 km^2), we retrieved the daily time series of total precipitation.
To minimize the biases in the climate simulations, their outputs were bias-adjusted by adopting the approach proposed by Bachner et al. (2008). As a reference dataset for current conditions, E-OBS was exploited (see B.4 for more details).
The data in Figure 3 show the average precipitation for every ten years with the central years shown in the labels. Since this short interval could be disputed, thirty years should be ac-

Table 1: The freshwater-related services in the Taro river basin

No	Ecosystem function (potential ecosystem service)	Ecosystem service	Reference
1	Water yield	Irrigation Water supply Industrial use Transportation Water regulation (i.e. environmental flow) Flood regulation Recharge ground water	Autorit di Bacino del Fiume Po, 2016; Di Dio et al., 2010 Mazzoleni et al., 2014b Iacumin et al., 2009; Toscani et al., 2007
2	Nutrient retentions (TN and TP)	Water purification Waste water treatment	Sammartino et al., 2007; Madoni and Zangrossi, 2005 Autorit di Bacino del Fiume Po, 2015; Di Dio et al., 2010; Gaglio et al., 2017

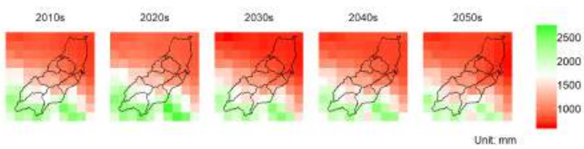

Figure 3: Yearly precipitation in the Taro River basin over the period 2010-2050.

counted for proper considering interannual variability. Nevertheless, the results should be read in terms of trends for a single frame of a transient condition. Over the study period (2010-2050), the average yearly precipitation in the TRB would vary from 637 mm in the lower parts (i.e. the effluent of the Taro river with the Po river) to 2259 mm in the upper parts (i.e. near Genova). Comparing with the reference year 2010s (2006-2015), precipitation would decrease over the period 2030-2050 with higher absolute variations in the upper parts (i.e. about -304 mm) and lower values in the lower areas (see Figure A.1). On the other hand, it would increase in the period 2010-2020 with the highest value of 216 mm. Noticeably, in terms of relative change respect to the 2010s, the highest reduction of precipitation (i.e. -18% and -16% at pixel and sub-basin level, respectively) would occur in the middle parts of the TRB (see Figure A.2).

2.2.2. Land use projections

To consider the impact of land-use change on freshwater ES, we used data from the Land Use-based Integrated Sustainability Assessment modeling platform - LUISA (Baranzelli et al., 2014). LUISA was configured as a baseline scenario of land-use changes up to 2050, assuming likely socio-economic trends, business as usual urbanization processes, and the effect of established European policies. Specifically, land-use distributions were obtained from the recently refined CORINE land cover maps. The actual economic figures used in LUISA were taken from the GEM-E3 model, which modeled the sector composition of the future economy consistently with the DG ECFINs projections (EC, 2013). Regional land demands were obtained from Eurostats demographic outlooks (EUROPOP2010) and long-term economic projections from DG ECFIN (EC, 2012). Urban land-use demands were obtained by combining the demand for residential and tourist accommodation (Baranzelli et al., 2014). The projected land-use maps were classified into ten categories with a spatial resolution of 1km x 1km (see Figure 4).

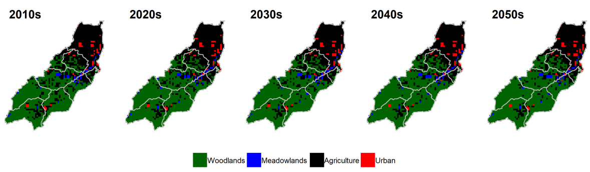

Figure 4: Projections of land use in the Taro River basin over the period 2010-2050. Source: adapted from (Baranzelli et al., 2014)

According to LUISA simulations, the spatial pattern of land-use in the period 2020-2050 would remain the same as that in the 2010s. Specifically, the agricultural and urban areas would be concentrated in the lower part of the TRB while the woodlands would spread over the upper part of this basin. Woodland areas would show a relative increase from about +5% in the 2020s to +13% in the 2050s while agriculture would decrease from about -6% in the 2020s to -18% in the 2050s. Urban zones would show a slight increase up to 2030 and keep constant afterward.

2.2.3. Ecosystem model

The ecosystem model, InVEST, is a spatially-explicit model that can be used to map and value potential ES, and compare tradeoffs among development scenarios (Posner et al., 2016).

InVEST was used for assessing a wide variety of ecosystem services such as carbon storage (Goldstein et al., 2012), water yield (Canqiang et al., 2012; Marquès et al., 2013), nutrient and sediment retention (Sánchez-Canales et al., 2015), habitat quality (Sallustio et al., 2017) and the potential tradeoff among ES (Hoyer and Chang, 2014; Hu et al., 2019; Jorda-Capdevila et al., 2019). For the TRB, we use the Water Yield (WY) and the Nutrient Delivery Ratio (NDR) module to quantify the current condition and changes of water provisioning and water purification. The outputs of these modules are the volume of water yield (m^3yr^{-1}), total nutrient production in the watershed ($kg\ yr^{-1}$), and total nutrient export from the watershed ($kg\ yr^{-1}$). These modules are calibrated and validated by using the historical data of the year 2010 (see Figure A.5 for the location of the stations for calibration and validation). Then the simulations are generated by using the projections of climate and land-use data, which are described in section 2.2.1 and 2.2.2, respectively. The technical descriptions of the models inputs are given in Table B.1, while the results are described in section 3.

The final outputs of the InVEST model are reported in absolute terms (i.e. $m^3\ yr^{-1}$ and $kg\ yr^{-1}$ for WY and NDR module, respectively) at sub-basin and basin levels. These results are used to build spatial maps of potential ES. Since it is essential to highlight the future spatial changes of potential ES with the reference year, we use these outputs to build the thematic maps of relative changes at the sub-basin level for all potential ES (see Figure 7). These maps illustrate the spatial-temporal impacts of climate change and land-use change on potential ES and, consequently, the delivery of actual ES.

2.3. Study design

To achieve the objectives mentioned in Section 1, our study was designed along with the following steps:

- Analysis of uncertainty related to the use of climate data;
- Model calibration and validation;
- Quantification of potential ES in the reference year and their relative changes under the baseline scenario;
- Assessment of drivers of changes in potential ES.

2.3.1. Analysis of uncertainty related to the use of climate data

World Meteorological Organization recommends assessing climate conditions and related impacts by using 30 years; nevertheless, such prescription could be prevented when the required simulation chains are time o resource consuming as in this study. To this aim, as all the available data for calibration and validation are made available for 2010, we analyzed the historical data in the 30 years (1986-2015) to retrieve the main features of 2010 in terms of annual precipitation and then to select the equivalent year in the projection data in the same period. After recognizing 2010 as a particularly wet year in the reference period, a year with similar features was retrieved by considering the bias-corrected dataset. The comparison between the two analyses was aimed at understanding the influence of replacing the actual monitoring data with the simulated ones, characterized by similar statistical properties within the interested period. Of course, data from a single year alone could not permit clear identification of all the issues and variations among the input datasets. However, for the moment, the stocktaking phase of data concerning water availability and quality within the basin represents the most limiting factor. The uncertainty was represented in the models outputs with different configurations (see section 3.1).

2.3.2. Model calibration and validation

All three components of the integrated model were calibrated and validated. The validation of climate data on the whole Italian domain was performed using the E-OBS dataset and other high-resolution datasets such as EURO4M-APGD and gridded dataset provided by ARPA Emilia Romagna and ARPA Piemonte (Bucchignani et al., 2016). The validation of land-use data (LUISA) was carried out by using the refine Corine Land Cover (CLC) to correct the underrepresentation of land cover related to transport infrastructure, and inappropriate classification of built-up tourist infrastructures as green urban areas (Baranzelli et al., 2014).

Regarding the InVEST model, the calibration (in 2010) and validation (in 2006) of this model were performed by comparing model outputs against observed values of the average annual water yield ($m^3\ yr^{-1}$), mean annual nitrogen and phosphorus concentration in the watershed ($mg\ l^{-1}$) from gauging and monitoring stations spread across the TRB (see Figure A.5). The observational data of average daily water flow ($m^3\ s^{-1}$) was obtained from the DEXT3D platform of the Regional Agency for Prevention, Environment, and Energy of Emilia-Romagna, Italy (ARPA-E). The observational data of nitrogen and phosphorus concentration ($g\ l^{-1}$) were obtained from ARPA-E . The watersheds corresponding to these stations were generated in the GRASS module in QGIS, using Digital Elevation Model with the resolution of 25m from the European Environment Agency (EEA) . The quality of these modules was assessed through different metrics, namely the Nash-Sutcliffe efficiency (NSE) and the coefficient of determination (R^2). The NSE determines the magnitude of the residual variance compared to the measured data variance (Nash and Sutcliffe, 1970). The R^2 provides a measure of how well the observational data is replicated by the model, based on the proportion of total variation of outputs explained by the model.

2.3.3. Quantification of potential freshwater ecosystem services

To understand the future dynamics of freshwater-related ES, we ran the InVEST model for every ten years in the period 2010-2050 when the LUISA data was available. In these simulations, the year 2010 was selected for the calibration, and the other years were used for the projections. Moreover, to capture the temporal variation of precipitation around the simulation years and to cover all the studied period 2010-2050, we used a moving window of 10 years (e.g. the mean precipitation of the 2010s is the average value of the period 2006-2015). We also extracted the minimum/maximum value of yearly precipitation (hereafter, they are denoted as min and max condition, respectively see Table A.3) to estimate the range of selected services. Additionally, to investigate the effects of climate and land-use change on ES as well as the relative contribution of each driver, we configurated the InVEST model with different sets of inputs. Specifically, the first configuration was to analyze the impact of climate change on ES by keeping land-use as constant, as in the 2010s, and changing climate data over time. Then, we kept climate data as constant, as in the 2010s, and changed land-use data to quantify the effects of land-use. Lastly, we changed both climate and land-use data in the third configuration to investigate the overall impacts of both drivers on ES (see section 3.3).

2.3.4. *Assessment of drivers of changes*

We compared the results of these simulations (i.e. only CC, only LU change and the combination of both drivers) to identify the main drivers that affected each ES. This identification allowed us to quantify the relative contribution of each factor to the alterations of these services (section 4), and to propose proper management practices to reduce the impacts of CC and LU change on potential ES as discussed in the conclusion part (see section 5).

3. Results

3.1. *Uncertainty of climate data*

Since we used simulated climate data to run the simulations for the reference year 2010 and the future period up to 2050, it is necessary to analyze the uncertainty induced by comparing them with the historical data reported in gridded dataset ArCIS. Over 30 years (1986-2015), Table A.1 shows that the average yearly precipitation of ArCIS data was higher than those returned by climate data also after bias correction, the average error is +176 mm. Such bias can be, in part, explained, by considering the underestimations of E-OBS dataset (also characterized by a coarser resolution) and used for bias-adjusting climate simulations. Within these 30 years, the year 2010 was the second wettest year in the ArCIS data set, which is equivalent to the year 2007 in the COSMO dataset. Then, we ran the simulations for these years with the calibrated InVEST model to assess the differences in the outputs, corresponding to different climate datasets (i.e. ArCIS and COSMO). Table A.3 shows that the simulations with ArCIS data resulted in a higher variation in WY (+19%) but a limited change in both TN and TP retention (only +0.1%).

3.2. *Results of calibration and validation*

3.2.1. *Calibration of the InVEST model*

The results of the calibration for WY and NDR module in 2010 are shown in Figure 5. Each point represents a monitoring station, showing in Figure A.5. The red line indicates the perfect fit between observational and simulated data (i.e. NSE = 1 and R^2 =1). Figure 5 shows that the output of the WY module of InVEST follows quite well the observational data (i.e. R2 =0.98). For the NDR module, the correlation coefficients

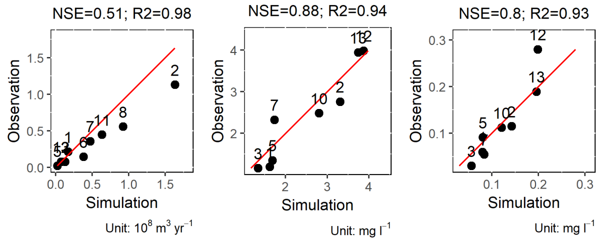

Figure 5: Results of the calibration for the year 2010

are lower than those in the WY module because the simulation outputs are converted from nutrient load to nutrient concentration using the water volume from the WY module. Thus, the residual errors from the WY module are inherited in the NDR module.

3.2.2. *Validation of the InVEST model*

The calibrated InVEST model was tested using bias-corrected data in 2006, the second wettest year in the 1986-2015 dataset for climate simulated data (the same plotting position of 2010 in the dataset of actual data). Of course, the parameters obtained from the calibration process for the year 2010 were maintained. Figure 6 illustrates that the calibrated

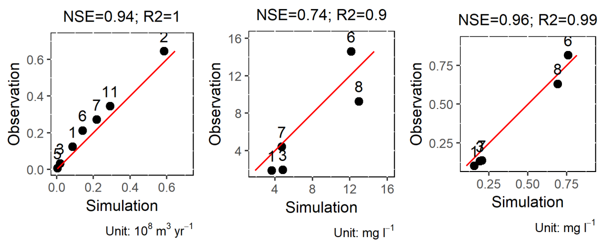

Figure 6: Results of the validation for the year 2006

model correlated quite well with the observational data in 2006. Specifically, the WY module underestimated WY services in most stations, but it showed a good correlation. Meanwhile, the NDR module resulted in a lower correlation than expected in the calibration process.

3.3. *Freshwater ecosystem services in the reference year 2010s and their relative changes under the baseline scenario*

Table 2 shows the results of all simulations with different configurations such as climate change (CC) alone, land-use change (LU) alone, and both climate and land-use change (CCLU). In the reference year 2010s, water yield in TRB was 1.5 10^9 m^3yr^{-1} within a range from 7.0 10^8 to 2.6 10^9 m^3yr^{-1} for the minimum and maximum values, respectively (see Table A.3 and Table 2). Regarding nutrient retention, the values were 1.8 10^7 kg^3yr^{-1} and 8.8 10^5 kg^3yr^{-1} for the TN and TP retention, respectively (see Table A.3 and Table 2).
Table 2 also reports the relative change, respect to the 2010s, of

ES from 2020 to 2050 at the basin level. These results are obtained from the InVEST model under different configurations of the input variables, driven by either climate change, land-use change, or both factors. At the basin level, WY decreases over the period 2030-2050 in the mean condition which ranges from -20 to -8%, while it increases in the period 2010-2020. Similarly, we observe a slight reduction of nitrogen retention over time (i.e. from -3 to -0.6%), except for the year 2030s when we observe a small increase of 0.8%. On the contrary, phosphorus retention slightly increases over time (i.e. from +0.4 to +3%) until the 2050s when we observe a decrease of -0.4%.
From the above, we identified that the changes of WY in CC simulation were much more significant than those in LU simulation. Specifically, the average variations were about -8% and -0.3%, respectively. On the contrary, the changes of TN and TP retention in LU simulation were much more significant than those in the CC simulation. For instance, the variations of TN in two types of simulations were -1.1% and 0.2%, respectively (see Figure A.6 for the detail at the sub-basin level). Most notably, these changes in the CCLU simulation in the 2040s and 2050s would be greater than those in the CC or LU simulation.

Figure 7 shows the relative changes of the three ES at sub-basin level respect to 2010s under the CCLU simulation. The red-white-green color ramp palette is used to represent negative, neutral, and positive change, respectively.

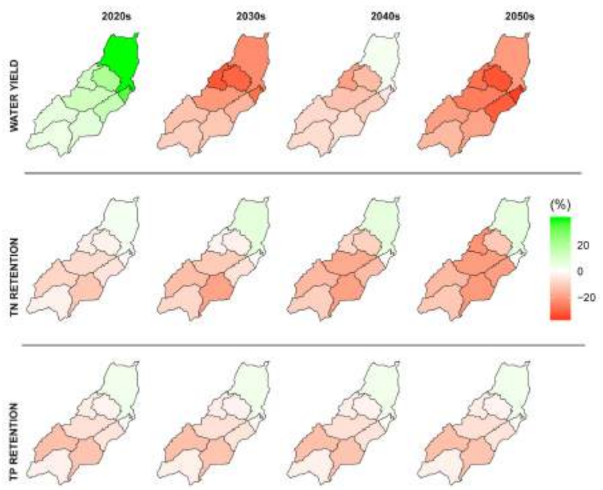

Figure 7: Relative changes of potential ecosystem services at sub-basin level respect to 2010s, consider both climate and land use change.

Compared with the reference year 2010s, at the sub-basin level, WY is expected to have the most substantial changes (from about -36 to +40%), followed by TP retention (from -22 to +7%) and TN retention (from -24 to +9%). Overall, WY would decrease in all sub-basins over the period 2030-2050 while increase in all sub-basin in the period 2010-2020. The significant reduction would occur in sub-basin 8, 2, and 9 (see Figure 1) with the values of -36%, -33% and -31% in the 2050s, respectively. Regarding TN and TP retention, they would have the same trend, i.e. the retention would decrease in all sub-basins over time except for the lower part of the TRB (sub-basin 7 in Figure 1). It worth noting, also in this case, that the strong inter-variations could be partly due to considering 10 years in each analysis.

4. Discussion

4.1. Strengths and limitations of the integrated modeling approach

This integrated approach utilized the advantages of freely available data on global change (e.g. land-use from LUISA platform and soil data from ESDAC) and open source software (e.g. InVEST, Q-GIS, and R). They allowed us to calibrate and validate the InVEST model, and to conduct quick assessment over the study period, providing information that could assist environmental managers in developing adaptation strategies. This approach is spatially explicit, transparent, and applicable to other case studies where climate, land-use, and soil data are available.
Nevertheless, this approach had some limitations due to the configuration of the InVEST model. Firstly, although this model worked well within the annual time step, it was not able to capture the seasonal variation of climate variables and extreme events. Secondly, we considered only three services; thus, we were unable to assess the complete impact of climate, and land-use change on the ecosystem and to analyze the trade-offs among related services. This limitation was due to the lack of data for calibration and validation. Thirdly, the soil input data (e.g. depth to root restricting layer and maximum root depth) was kept constant in the whole simulation period, although they should be simulated over time. Therefore, there is a need for improving the temporal resolution of the ecosystem model (e.g. seasonal or monthly time step), coupling with a soil model (e.g. SWAT) in future research. These would allow us to maintain the dynamic of the whole system and to capture the seasonal variation and the impact of extreme events on ES.

4.2. Uncertainty of ES related to the use of climate data

As stated earlier, we need to be carefully assessed the uncertainty when using climate projections for future impact assessment. To accurately quantify uncertainty in future variations and errors, ensemble approaches should be preferred (e.g. using data from multiple climate simulations). Nonetheless, it was beyond the scope of this study for which the changes in land-use were also accounted. In our work, bias-corrected data were compared with historical observations in order to estimate the errors affecting the simulations. Moreover, this task was carried out under the assumption that the simulation chain could be characterized by comparable performances in the current and future conditions. Therefore, the variations were estimated by comparing the outputs of the impact model (InVEST), forced by climate simulations in the future versus the current period. This approach permits discriminating potential errors affecting the analysis by variations induced by changes in atmospheric forcing.

Table 2: Absolute value of potential freshwater ecosystem services in the 2010s and their relative changes in 2020-2050 at basin level (respect to the 2010s) under different drivers of changes such as only climate change (CC), only land-use change (LU), and both climate and land-use change (CCLU)

2010s (reference year)	CC and LU 2010s	Water yield (m^3yr^{-1})	1.5E+09	Nitrogen retention (kg^3yr^{-1})	1.8E+07	Phosphorus retention (kg^3yr^{-1})	8.8E+05
2020s	CC	Relative change of water yield respect to 2010s (%)	10.4	Relative change of water yield respect to 2010s (%)	0.0	Relative change of water yield respect to 2010s (%)	-0.1
	LU		-0.2		-0.6		0.5
	CC&LU		10.2		-0.6		0.4
2030s	CC		-14.3		0.2		0.3
	LU		-0.2		0.5		2.7
	CC&LU		-14.5		0.8		2.9
2040s	CC		-7.5		0.1		0.0
	LU		-0.3		-1.5		0.9
	CC&LU		-7.8		-1.4		0.9
2050s	CC		-19.6		0.1		0.1
	LU		-0.5		-3.0		-0.5
	CC&LU		-20.1		-2.9		-0.4

Table A.3 shows that WY was more sensitive to climate data at the basin and sub-basin level. Specifically, we found that WY in the simulation using ArCIS data were higher than in those using COSMO data. The difference among them was about from 5% to 40% at the sub-basin level and 19% at the basin level. Therefore, we concluded that we might misrepresent WY service in the analysis when using projected climate data as the input. However, in terms of variations, such errors are assumed to be minimized.
On the contrary, nutrient retention services were conservative to the climate data in all simulation. Overall, the difference among simulations was about from -4% to 3% at the sub-basin level and 0.1% at the basin level. These changes proved that the use of different climate dataset had a limited effect on both TN and TP retention.

4.3. Drivers of changes in water yield service

Several studies investigated the link between the alteration of water yield and the distribution of land-use as well as the changes in precipitation pattern (Canqiang et al., 2012; Chang and Bonnette, 2016; Gaglio et al., 2017; Karabulut et al., 2016; **?**). For instance, Paudyal et al. (2019) concluded that the reduction of WY was due to the higher rate of evapotranspiration and groundwater recharge driven by the changes of the land-use pattern. Regarding the influence of climate factors, Polce et al. (2016) argued that climate change had profound impacts on the different ecological functions and ES across river basins in Europe.
In our case study, by comparing the simulations with different scenario configurations (e.g. CC, LU and CCLU simulation), we identified that precipitation was the main factor that would affect WY service. As shown in Figure 7 and Figure A.2, the reductions in WY were due to the decline in precipitation in the whole basin, especially in the 2050s with a reduction of -20% in some regions. Specifically, the reductions of WY in CC simulations were much higher than those in LU simulations, and they are close to the CCLU simulations. For instance, in 2050s, these values were -20%, -0.5% and -20% for CC, LU, and CCLU simulation, respectively.
The decline of WY was also confirmed at the sub-basin level when we observed that the changes in water yield followed the temporal and spatial variation of precipitation. Within each sub-basin, we observed that the temporal changes in water yield over time followed the overall trend at the basin level and the trend of precipitation. For instance, the most considerable change (i.e. about -36%) happened in the central part of the TRB (i.e. label 2 in Figure 1) and its surroundings where the most substantial change in precipitation (i.e. -16%) occurred. This correlation between water yield and precipitation was also found in other case studies worldwide such as the Be River basin in south Vietnam (Khoi and Suetsugi, 2014), Brahmaputra River basin in South Asia (Pervez and Henebry, 2015), the Kor River Basin in Southwest of Iran (Vaighan et al., 2017), and several basins in Europe (Polce et al., 2016).
Nevertheless, at the sub-basin level, we observed some positive changes of WY in some areas near the effluence of the TRB and Po River basin in the 2040s. These gains were due to the slight increase in precipitation in the north-east of the TRB in these years (see Figure A.2). This positive correlation reconfirmed the dependency between WY and precipitation. This increase in WY was also found in the Adige basin, located in the central Southern Alps, and its sub-basin (Jorda-Capdevila et al., 2019). Despite these increases, water yield in the whole TRB still declined over time because they were not large enough to compensate for the reductions in other sub-basins.

4.4. Drivers of change in nutrient retention services

The relationship between the spatial or temporal change of land-use, and nutrient load and retention was well established by some authors in different case studies (Gaglio et al., 2017; Holland et al., 2015; Mokondoko et al., 2016; Viaroli et al., 2018). For instance, Mokondoko et al. (2016) found a positive correlation between forest cover and water regulatory services in central Veracruz, Mexico. Viaroli et al. (2018) highlighted that TN and TP load in the Po river basin were related to land-use change in the last fifty years. Gaglio et al. (2017) concluded that land-use conversions related to direct human influences (e.g. croplands and urban areas expansion) were mostly associated with the loss of specific ES such as waste treatment in the Po river basin.

In the TRB, by comparing the InVEST simulations with different configurations (see Figure A.6), we identified that LU was the dominant driving factor of TN and TP retention which was also found in Ren et al. (2015) and Shen et al. (2013). Specifically, the alterations of nutrients retention in LU simulations were much higher than those in the CC simulation. For instance, TN would decrease -3% in the LU simulation while it would increase +0.1% in the CC simulation in the 2050s. These trends were mainly driven by the distribution of land-use and the sources of nutrient-related to them (Vaighan et al., 2017). Agriculture is the primary source of nitrogen, while the main sources of phosphorus are from municipal wastewater treatment plans and agriculture (Mockler et al., 2017). In Europe, diffuse sources such as agriculture are accounted for 38% of pollution on surface water bodies (Kristensen et al., 2018). Thus, the reduction of agricultural areas overtime, i.e. from -6% in the 2020s to -18% in the 2050s (see Figure A.3), led to the decline of TN production.

At the river basin level, we observed different trends for TN and TP retention. Specifically, TN retention would decrease slightly with a maximum variation of -3% while TP retention would increase slightly with a maximum increase of +3%. These trends were driven by the changes in land-use patterns associated with the sources of nutrients. Wastewater from urban zones is the primary source of phosphorus (Mockler et al., 2017). Thus, the increase of urban areas in TRB (e.g. about +0.4% in the 2030s and afterward) would lead to an increase of phosphorus production and, consequently, TN retention. These different behaviors in TN and TP were also found by Wang et al. (2018) in Hubei province, China (i.e. the load of TN decreased by 4%, and TP increased by 12% in 25 years). The outcomes of our work were consistent with the published results found in the Po river basin, i.e. N and P loads increased until the 1990s, then P decreased while N remained high (Viaroli et al., 2018). These authors concluded that the notable reduction of P load was due to the increase TP retention by soil and sediments in the watersheds.

At the sub-basin level, we observed a similar trend for TN and TP retention despite the opposite trend for TN and TP retention that we detected at the basin level. Specifically, TN and TP retention would decrease in all sub-basins except for the lower part of the Taro river (i.e. label 7 in Figure 1) where we found some improvements in nutrient retention overtime.

Firstly, the negative changes, which were found in most sub-basins, confirmed the correlation between nutrient concentration and precipitation. The reduction in precipitation led to the reduction of the dilution capacity of nutrients from the sources (e.g. agriculture and wastewater effluents) and, consequently, nutrient production and retention capacity in rivers and on land. Although TN and TP had different sources, the dilution, transportation, and export processes were the same, which led to the same trend. Additionally, we also observed the correlation between the reduction of agriculture area and TN retention. Since agriculture is the primary source of TN pollution due to the intensified use of mineral fertilizers (Johnson et al., 2013; Mockler et al., 2017; Vaighan et al., 2017), the reduction of agriculture led to the decrease of TN production and, consequently, TN retention. Specifically, we found the more considerable reductions of TN retention in the central parts of the TRB, where the most significant conversion from agriculture to woodland occurred.

Second, we found that the local improvement of nutrient retention in the lower part of the TRB (i.e. label 7 in Figure 1) was due to the expansion of urban areas in this sub-basin (see Figure A.4). Point sources of nutrient pollutions, especially phosphorus, from municipal wastewater treatment plans were the main suppliers of nutrients (Mockler et al., 2017). Moreover, TP and TN from point sources (e.g. untreated sewage) were much higher than those from nonpoint sources (e.g. from runoff) (Qin et al., 2019). Therefore, small expansion of urban areas (i.e. up to +2%) in the lower sub-basin could lead to an increase in nutrient production and, consequently, nutrient retention. Moreover, these positive changes were significant enough to compensate for the reductions of nutrient retentions due to the decline of agriculture in this sub-basin. Because of this combination, the lower part of the TRB experienced increases of +7% and +9% for TN and TP retention, respectively.

Finally, we found that the contrast trends of TN and TP retention at river basin level were not statistically significant. Firstly, the magnitudes of relative changes were quite small (i.e. -3% and +3% for TN and TP retention, respectively). Secondly, at the sub-basin level, the trends of TN and TP retention were similar, but they did not contrast with each other in all sub-basins. Third, we found in many case studies worldwide that TN load usually had one order of magnitude higher than TP load (Chen et al., 2017; Ma et al., 2017, 2015; Mockler et al., 2017; Reza et al., 2016; Zeng et al., 2016; Zhang et al., 2018). In the Po river basin, to which the TRB belongs, the ratio TN/TP was 48-208 (Cozzi and Giani, 2011). Thus, TP retention was more sensitive to climate change and land-use change. Although the reductions of TP retention were predicted in almost all sub-basins (except for the sub-basin 7 in Figure 1), the increase in the lower part of the TRB was high enough to compensate for those reductions, which led to the overall increase at the basin level.

5. Conclusions and recommendations for environmental management

This work proposed an innovative integrated modeling approach to quantify the coupling impacts of climate and land-use change on potential freshwater ES. This approach can be applied to any case study where climate, land-use, soil, and monitoring data are available to set up, calibrate, and validate the models. The selected services, namely water yield, and nutrient retentions, were identified by local authorities through national meetings and workshops in the relevance to environmental management and policies. Thanks to the simulation, the assessment of the impacts of land-use and climate change on potential freshwater ES can support decision-makers and

authorities in the identification of the most suitable management measures. These solutions are implemented concerning different land-use types (e.g. wetland, agriculture, urban and water bodies), topographic settings (plain, mountain or both), their adaptation targets (single or combined actions among water quantity and quality)and planning time horizon (operational or strategic).
In this study, we found that the effects of climate and land-use change on freshwater ES varied in terms of impacts, location, and time. Among services, water yield had a more significant reduction compared to nutrient retentions under the baseline scenario. Yet, nutrient retention was highly correlated to nutrient production and dilution capacity, which are driven by water availability. Therefore, water managers are informed about prioritizing actions aimed at improving water yield because this service is more sensitive to CC and LU change. Moreover, enhancing WY implies the enrichment of other related services such as TN and TP retention. Water yield could be considered as an indicator measuring the effectiveness of adaptation plans to prevent or significantly reduce water stress. Thus, it assists in complying with approved environmental directives (e.g. Water Framework Directives) and achieving the Sustainable Development Goals. Specifically, different adaptation strategies (e.g. the reduction of water consumption) can be implemented in this integrated model to quantify the contribution of these measures to the SDG 6.3 Water Stress Score.
Second, the trends of TN and TP retention were quite different among sub-basins due to the different sources of pollutants associated with land-use distribution. Thus, managers and authorities are informed about which sectors (e.g. agriculture, urban, wetlands, etc.) they should implement management actions to improve water quality to comply with regulatory policies and directives such as Nitrates directive (e.g. nutrient cycle, including nitrogen and phosphorus, is managed in a more sustainable and resource-efficient way) and the SDGs. To be more precise, we could introduce different adaptation strategies (e.g. stabilize agriculture trends, control nitrogen fertilizer and manage wastewater) into our integrated model and use the outputs (i.e. nitrogen and phosphorus retention) as indicators to quantify the effectiveness of these measures to the achievement of SDG 6.1 Improved water source.
Finally, ecosystem service assessments enable more accurate assessments of the dynamic of ES and the effects of multiple stressors on the ecosystem, providing useful information regarding the planning time horizon. Therefore, further assessments should integrate not only climate and land-use change data but also the change in pattern and characteristic of soil to capture the dynamics of this complex system. Moreover, these assessments need to consider the importance of extreme events by moving the temporal resolution from the annual to monthly or seasonal time steps. This improvement would allow greater accuracy in assessing uncertainty in future scenarios and the complex effects of compound events that would affect potential ES at sub-basin or basin scale. In this regard, decision-makers are informed about the time and duration to implement management practices to improve the capacity of potential freshwater ES.

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Appendices

A. Appendix A: Climate, land use and observational data

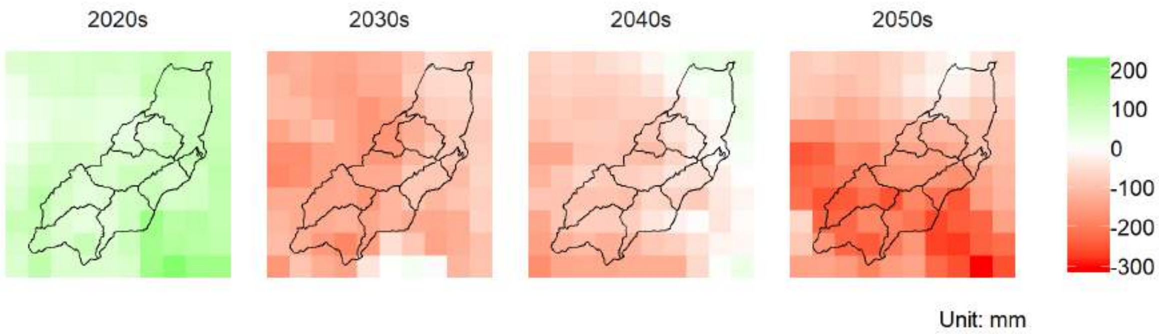


Figure A.1: Absolute change of yearly precipitation in the Taro River basin respect to 2010s.

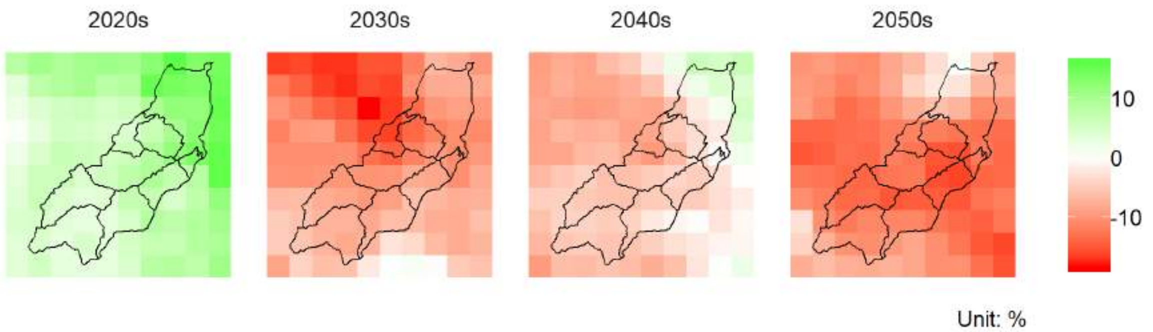


Figure A.2: Relative change of yearly precipitation in the Taro River basin respect to 2010s.

Table A.1: Average annual precipitation of historical data (ARCIS) and projected data (COSMO)

Year	Yearly precipitation (mm yr-1)		Year	Yearly precipitation (mm yr-1)	
	ARCIS	COSMO		ARCIS	COSMO
1986	978	1013.9	2001	920.5	886.9
1987	1124.8	667.6	2002	1322.1	716
1988	912.4	824.5	2003	816.4	898.5
1989	907.9	810.1	2004	1054	818.5
1990	927.6	653.8	2005	931.7	857.2
1991	994.4	693.4	2006	837.4	682.7
1992	1137.3	657.5	2007	811.6	1100.4
1993	1025.8	926	2008	1276.7	1052.8
1994	1038.9	1049.5	2009	1154.2	1175.5
1995	1018.3	976.1	2010	1404.4	1075.2
1996	1248	860.1	2011	890.2	661.3
1997	919	867.4	2012	1054.5	917.9
1998	926.9	956.9	2013	1270.7	1080.7
1999	1102.4	836.2	2014	1500.9	825.2
2000	1215.3	971.9	2015	879.8	824.1

Table A.2: Potential ecosystem services using different input data of precipitation

Sub-basin	Water yield (m3 yr-1)		Nitrogen retention (kg yr-1)		Phosphorus retention (kg yr-1)	
	COSMO 2010	ARCIS 2007	COSMO 2010	ARCIS 2007	COSMO 2010	ARCIS 2007
1	4.88E+07	6.82E+07	764879.2	750707.3	31572	30954.3
2	5.56E+07	6.85E+07	1812840	1819883	86091.4	86399.1
3	1.73E+08	2.06E+08	1540280	1544317	69712.1	69860.1
4	3.08E+08	3.86E+08	451577	441636.7	14251.8	14046.7
5	2.88E+08	3.20E+08	854197.8	861392.9	39304.9	39410.4
6	4.59E+08	5.48E+08	868155.4	837224.5	37821.8	36162.5
7	8.91E+07	1.12E+08	10252400	10276495	546474.6	547972.2
8	1.05E+07	1.10E+07	374617.9	382441.3	15848.3	16204.1
9	8.47E+07	8.86E+07	789904.8	811933.9	34479.2	35441.9
Taro basin	**1.52E+09**	**1.81E+09**	**17708852**	**17726032**	**875556.2**	**876451.3**

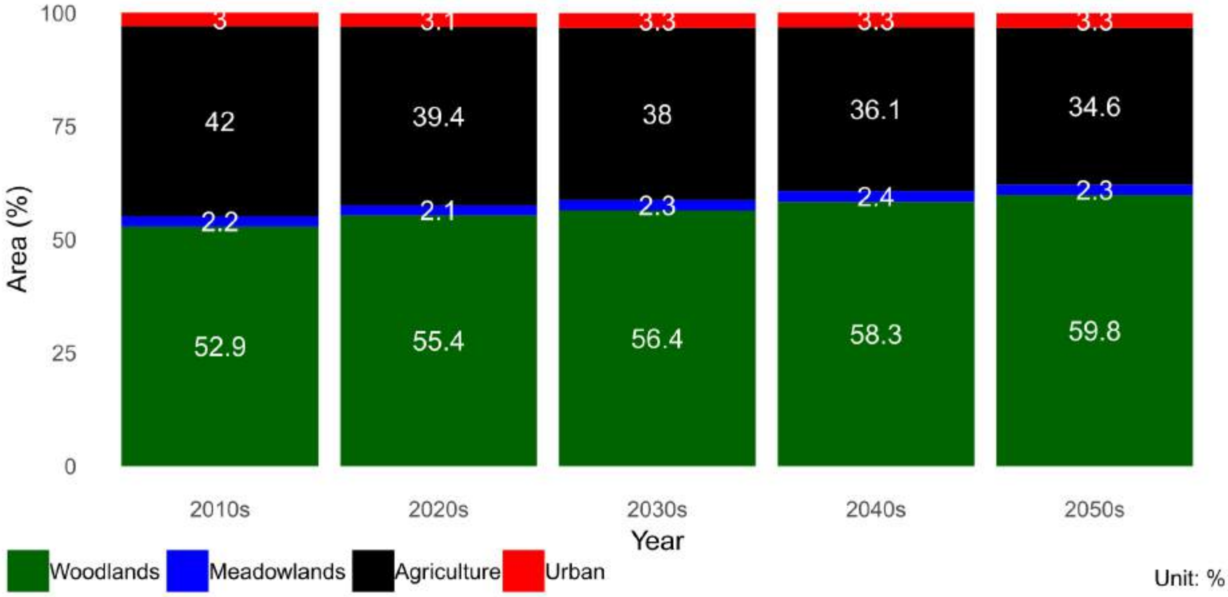


Figure A.3: Distribution of land use in the Taro River basin over the period 2010-2050

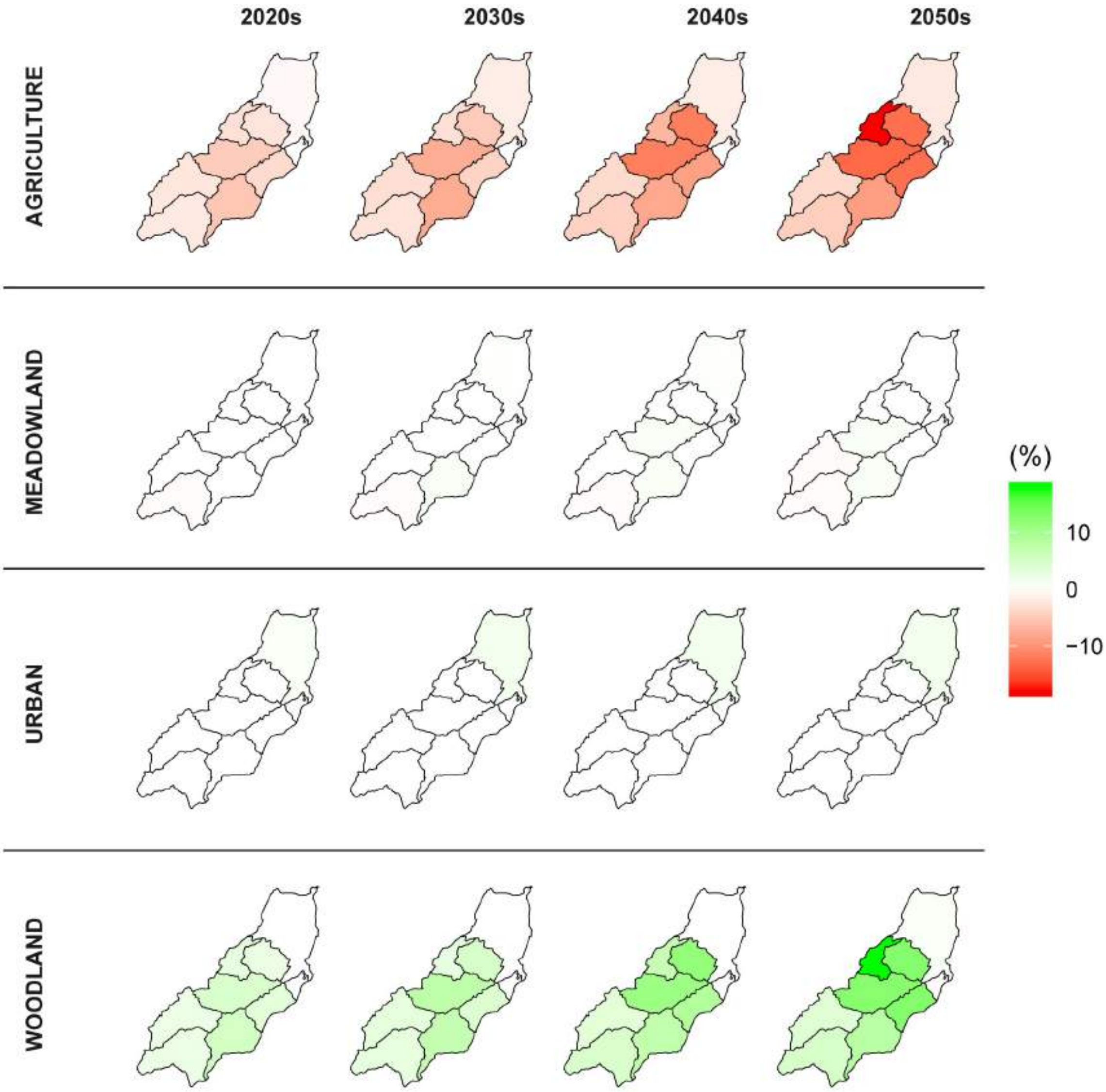
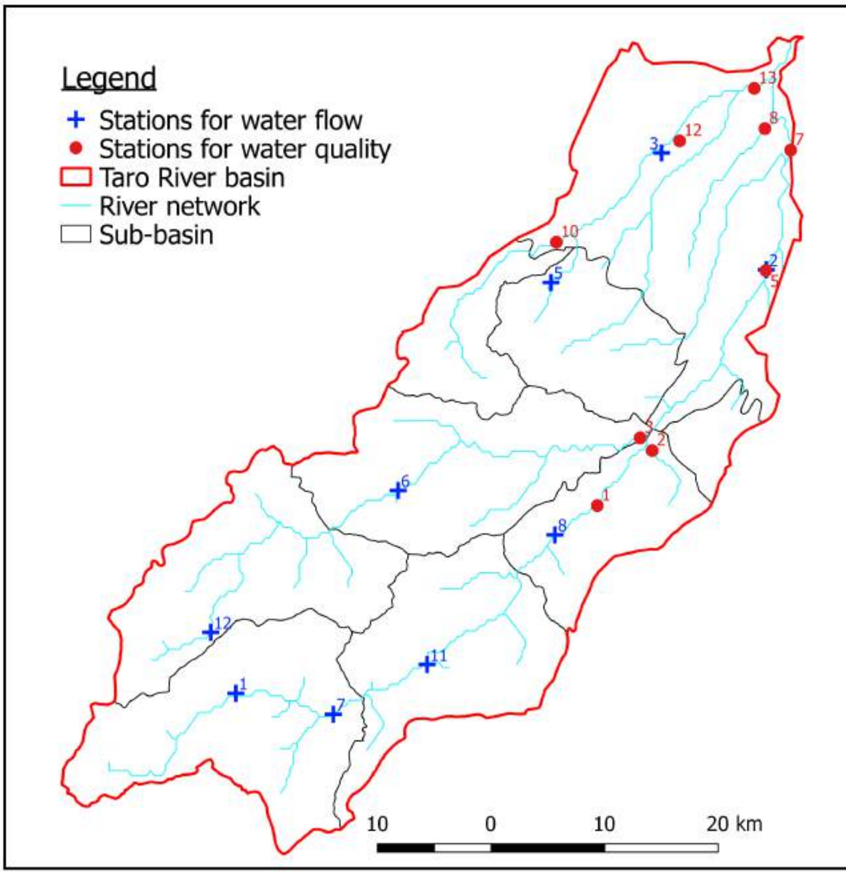


Figure A.4: Changes of relative contribution of each land use class at sub-basin level respect to 2010s



Stations for water flow

Id	Name
1	Tornolo
2	Parma Ovest
3	Castellina di Soragna
5	Salsomaggiore sul Ghiara
6	Ponte Lamberti
7	Pradella
8	Solignano
11	Ostia Parmense
12	Ponteceno

Stations for water quality

Id	Name
1	Ponte sul Taro Citerna - Oriano
10	imm. T. Ghiara
12	Soragna
13	Fontanelle - S. Secondo Parmense
2	Fornovo
3	Ramiola - Varano de Melegari
5	Pontetaro
7	San Quirico - Trecasali
8	San Secondo Parmense

Figure A.5: Positions of the calibration stations

Table A.3: Valuation of potential freshwater ecosystem services in 2010s and their relative change in 2020-2050 at sub-basin and basin level respect to 2010s using different values of climate data: minimum, mean and maximum values

Time/Space	2010s			2020s			2030s			2040s			2050s		
	Min	Mean	Max	Min	Mean	Max	Min	Mean	Max	Min	Mean	Max	Min	Mean	Max
Sub-basin	Water yield (m3 yr-1)			Relative change of water yield respect to 2010s (%)											
1	1.4E+07	4.9E+07	9.3E+07	130.2	11.6	-6.5	-15.0	-32.2	-34.8	32.2	-19.1	-8.9	-50.6	-27.8	-37.3
2	1.1E+07	5.6E+07	1.2E+08	265.0	20.8	-5.3	39.4	-29.9	-40.3	89.2	-13.6	-19.3	-45.5	-32.5	-44.1
3	6.3E+07	1.7E+08	3.0E+08	101.8	12.6	1.8	-12.8	-20.8	-25.4	18.3	-12.6	-4.4	-31.1	-25.9	-35.3
4	1.6E+08	3.1E+08	4.8E+08	42.3	6.3	-1.2	-13.8	-12.2	-21.7	2.6	-7.8	1.4	2.0	-19.6	-37.8
5	1.5E+08	2.9E+08	4.5E+08	42.9	7.6	-5.6	-21.4	-11.5	-20.9	0.9	-6.0	-3.0	4.6	-18.8	-33.7
6	2.6E+08	4.6E+08	7.4E+08	26.2	5.1	-8.4	-16.3	-9.7	-20.1	2.3	-7.0	-1.3	19.3	-14.3	-37.9
7	1.6E+07	8.9E+07	2.2E+08	302.4	39.6	3.3	54.7	-23.2	-34.5	98.0	2.7	-0.7	-43.6	-20.4	-28.1
8	1.6E+06	1.1E+07	2.7E+07	389.7	34.6	5.7	54.1	-28.0	-42.1	96.5	-4.8	-32.2	-53.4	-35.5	-46.4
9	3.2E+07	8.5E+07	1.5E+08	96.3	14.5	5.6	-11.3	-15.1	-24.5	13.5	-9.9	-19.0	-38.8	-31.1	-42.3
Taro basin	**7.0E+08**	**1.5E+09**	**2.6E+09**	**56.2**	**10.2**	**-3.2**	**-13.6**	**-14.5**	**-24.4**	**8.4**	**-7.8**	**-3.9**	**1.1**	**-20.1**	**-36.7**
Sub-basin	Nitrogen retention (kg yr-1)			Relative change of nitrogen retention respect to 2010s (%)											
1	7.7E+05	7.6E+05	7.6E+05	-5.1	-3.1	-2.8	-4.3	-1.6	-2.2	-10.7	-8.0	-8.4	-18.8	-22.0	-23.1
2	1.9E+06	1.8E+06	1.8E+06	-6.4	-2.5	-1.6	-7.6	-2.7	-1.7	-13.3	-9.6	-7.8	-8.9	-11.1	-12.0
3	1.6E+06	1.5E+06	1.5E+06	-8.9	-7.3	-8.0	-12.6	-10.8	-11.5	-18.1	-17.3	-17.1	-17.3	-20.0	-21.3
4	4.4E+05	4.5E+05	4.6E+05	-7.2	-8.6	-10.1	-11.4	-13.2	-13.3	-12.9	-14.4	-16.6	-16.6	-14.2	-13.3
5	8.2E+05	8.5E+05	8.6E+05	-5.4	-11.5	-9.5	-10.2	-18.6	-17.6	-13.4	-19.0	-18.3	-25.8	-20.5	-19.0
6	8.3E+05	8.7E+05	8.5E+05	4.1	-2.9	-1.9	-2.7	-8.1	-6.9	-5.4	-10.0	-11.8	-24.4	-11.4	-3.5
7	1.0E+07	1.0E+07	1.0E+07	2.1	2.8	2.9	5.7	6.8	6.9	6.0	6.6	7.0	7.1	6.4	6.1
8	3.8E+05	3.7E+05	3.7E+05	-3.9	-1.2	-1.4	-3.5	0.4	1.4	-2.3	-0.7	3.6	2.6	0.5	0.0
9	7.8E+05	7.9E+05	7.9E+05	-5.1	-5.5	-7.3	-5.6	-6.1	-6.1	-12.6	-13.6	-10.2	-15.2	-19.6	-19.7
Taro basin	**1.8E+07**	**1.8E+07**	**1.8E+07**	**-1.0**	**-0.6**	**-0.5**	**0.0**	**0.8**	**1.0**	**-1.8**	**-1.4**	**-0.9**	**-2.5**	**-2.9**	**-2.8**
Sub-basin	Phosphorus retention (kg yr-1)			Relative change of phosphorus retention respect to 2010s (%)											
1	3.2E+04	3.2E+04	3.2E+04	-5.5	-3.4	-3.1	-4.7	-1.8	-2.3	-11.8	-8.9	-9.2	-20.8	-24.2	-25.4
2	8.9E+04	8.6E+04	8.5E+04	-6.7	-2.3	-1.3	-7.5	-2.2	-1.1	-13.0	-8.7	-6.9	-7.7	-10.2	-11.3
3	7.0E+04	7.0E+04	7.0E+04	-8.0	-6.2	-6.9	-11.6	-9.6	-10.0	-16.7	-16.0	-15.4	-15.6	-18.5	-19.6
4	1.4E+04	1.4E+04	1.4E+04	-12.0	-12.9	-14.0	-17.3	-18.7	-19.1	-20.3	-21.1	-22.7	-22.0	-21.1	-21.1
5	3.7E+04	3.9E+04	3.9E+04	-4.2	-11.9	-9.2	-8.8	-19.0	-17.6	-12.6	-19.4	-18.3	-27.1	-21.0	-18.8
6	3.6E+04	3.8E+04	3.7E+04	4.7	-2.7	-1.6	-3.3	-8.2	-6.6	-4.7	-10.0	-11.9	-25.0	-11.1	-2.5
7	5.5E+05	5.5E+05	5.5E+05	3.0	3.8	3.8	7.9	9.1	9.3	8.2	8.8	9.3	9.6	8.8	8.5
8	1.6E+04	1.6E+04	1.6E+04	-4.2	-1.3	-1.5	-3.8	0.4	1.5	-2.5	-0.7	3.9	2.7	0.5	-0.1
9	3.4E+04	3.4E+04	3.4E+04	-4.9	-5.5	-7.4	-5.5	-6.0	-6.0	-12.9	-14.2	-10.5	-15.2	-20.1	-19.9
Taro basin	**8.8E+05**	**8.8E+05**	**8.7E+05**	**-0.1**	**0.4**	**0.6**	**2.0**	**2.9**	**3.2**	**0.4**	**0.9**	**1.6**	**0.1**	**-0.4**	**-0.4**

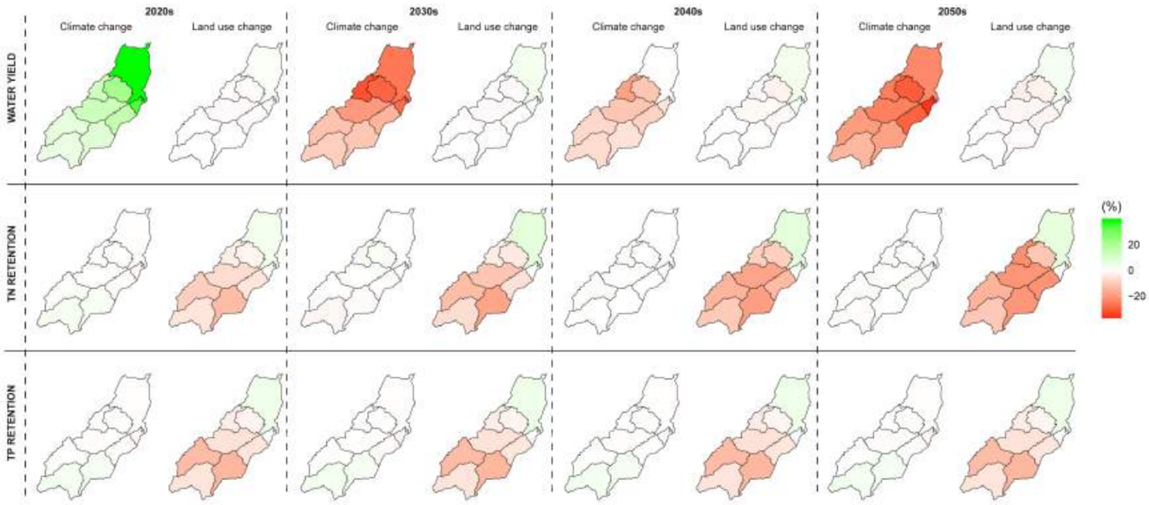


Figure A.6: Relative changes of potential ecosystem services at sub-basin level respect to 2010s, consider either climate change or land use change

B. Appendix B: Input of InVEST model

Table B.1: Data requirement

Input data	Units	Sources
I. Common data		
Watersheds and sub-watersheds	dimensionless	AdbPo
Precipitation	mm	CMCC
Land use	dimensionless	LUISA platform
Biophysical table	-	Calibrate
II. Water Yield module		
Reference evapotranspiration	mm	JRC
Depth to root restricting layer	mm	ESDAC
Plant available water fraction	-	ESDAC
Maximum root depth	mm	(Jorda-Capdevila et al., 2019)
Seasonally factor (Z)	-	(Snchez-Canales et al., 2015)
Water demand for consumptive uses	m3 yr-1	ISTAT
III. Nutrient Delivery Ratio module		
Digital elevation model (DEM)	m.a.m.s.l.	EEA
Threshold flow accumulation	-	(Jorda-Capdevila et al., 2019)
Borselli k parameter	-	Calibrate
Subsurface critical length	m	InVEST User Guide
Subsurface maximum retention efficiency	-	InVEST User Guide

B.1. Watershed and sub-watershed

Watershed is a shapefile, with one polygon per watershed. Sub-watershed is a shapefile, with one polygon per sub-watershed within the main watersheds specified in the watershed shapefile. These layers were downloaded from the website of the Po River basin authority AdbPo

B.2. Precipitation

See Section 2.2.1 for the details.

B.3. Land use

See Section 2.2.2 for the details. The classification of land use was adapted from Pelorosso et al. (2009), a case study on land cover and land use change in the Italian central Apennines.

B.4. Reference evapotranspiration

A GIS raster dataset, with an annual average evapotranspiration value for each cell. Reference evapotranspiration is the potential loss of water from soil by both evaporation from the soil and transpiration by healthy alfalfa (or grass) if sufficient water is available. The reference evapotranspiration values are in millimeters.
This data was obtained from the EU Open Data Portal . They are 12 maps of monthly potential evapotranspiration used for soil water balance. The potential evapotranspiration in the dataset was computed using Langbein's formula (see p. 55 of Pistocchi et al., 2006). This formula relates annual potential evapotranspiration ($mm\ yr^{-1}$) to temperature as $PET = 300 + 25T + 0.05T^3$, where T is temperature in degrees C. Although we used monthly mean temperatures, PET for a month is still computed in mm yr^{-1}.

B.5. Depth to root restricting layer

A GIS raster dataset with an average root restricting layer depth value for each cell. Root restricting layer depth is the soil depth at which root penetration is strongly inhibited because of physical or chemical characteristics. The root restricting layer depth values should be in millimeters. This data was obtained from the European Soil Data Center (ESDAC) . Specifically, the layer Depth to rock (DR) was used as a proximation of root restricting layer depth. This layer was a raster file of 1km spatial resolution in ETRS_1989_LAEA with 5 values corresponding with 5 classes .

Table B.2: Land use classification. Source: Adapted from Pelorosso et al. (2009)

LU code	LU label	CLC code	CLC label
1	Woodlands	311	Broad-leaved forest
		312	Coniferous forest
		313	Mixed forest
2	Meadowlands	321	Natural grasslands
		322	Moors and heathland
		323	Sclerophyllous vegetation
		324	Transitional woodland-shrub
		331	Beaches, dunes, sands
		332	Bare rocks
		333	Sparsely vegetated areas
		334	Burnt areas
		335	Glaciers and perpetual snow
		231	Pastures
3	Agriculture	241	Annual crops associated with permanent crops
		243	Land principally occupied by agriculture, with significant areas of natural vegetation
		244	Agro-forestry areas
		221	Vineyards
		222	Fruit trees and berry plantations
		223	Olive groves
		242	Complex cultivation patterns
		213	Rice fields
		211	Non-irrigated arable land
		212	Permanently irrigated land
4	Urban	111	Continuous urban fabric
		112	Discontinuous urban fabric
		121	Industrial or commercial units
		122	Road and rail networks and associated land
		123	Port areas
		124	Airports
		131	Mineral extraction sites
		132	Dump sites
		133	Construction sites
		141	Green urban areas
		142	Sport and leisure facilities
5	Water bodies	411	Inland marshes
		511	Water courses
		512	Water bodies

Table B.3: Attribute value of depth to rock (DR) layer.

No	Value	Short class	Full class	Depth value (cm)	Assign value (mm)
1	1	D	Deep	80-120	1000
2	2		Non-soil	0	0
3	3	M	Moderate	40-80	600
4	4	S	Shallow	< 40	200
5	5	V	Very deep	> 120	1400

B.6. Plant available water fraction

It is a GIS raster dataset with a plant available water content value for each cell. It is the fraction of water that can be stored in the soil profile for plants use. It is defined as the difference between the fraction of volumetric field capacity and permanent wilting point, which ranges from 0 to 1. This data was calculated from some indicators of soil hydraulic properties which were downloaded from ESDAC . From this source, two indicators were obtained, namely water content at field capacity (cm^3 cm^{-3}) and water content at wilting point (cm^3 cm^{-3}). For each variable, two raster layers were produced, one with the true value of the soil property and one with the values reclassified as equidistant classes ranging from 1 to 10. They were stored in GeoTif files with two bands each (band 1 is the absolute value while band 2 is the relative value):

- $ths\_fao\_octop.tif$: saturated water content
- $fc\_fao.tif$: water content at field capacity
- $wp\_fao.tif$: water content at wilting point
- $ks\_fao\_octop.tif$: saturated hydraulic conductivity

Then, the plant available water fraction (PAWF) is calculated by the formula in (Tarnik and Igaz, 2015):

$$PAWF = W_{fc}W_{wp}$$

Where: W_{fc} : water content at field capacity
W_{wp} : water content at wilting point

B.7. Water demand for consumptive uses

It is a table which indicates consumptive water use for each land use type. Consumptive water use is the part of water used that is incorporated into products or crops, consumed by humans or livestock, or otherwise removed from the watershed water balance. This data was obtained from the Italian National Institute of Statistics (ISTAT) in the section water supply use and volume of water use for irrigation for urban and irrigation, respectively. Then, we aggregated and calculated these demands for each sub-basin using QGIS.

B.8. Digital elevation model (DEM)

The DEM layer with resolution of 25m was obtained from EEA platform .

B.9. Other parameters

Other scalar parameters were obtained from calibrated values from the literature. Specifically, the maximum root depth and threshold flow accumulation were obtained from a similar case study in Adige basin, Italy (Jorda-Capdevila et al., 2019). The subsurface critical length and subsurface maximum retention efficiency were the recommended value from InVEST User Guide.

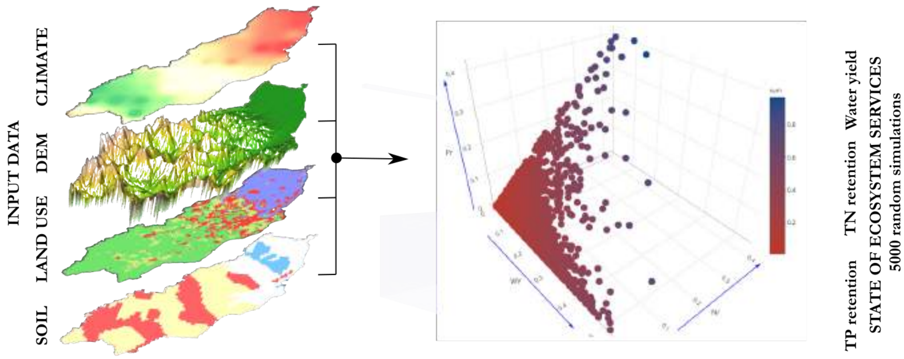
B.10. Other related data sets

Table B.4: Other related data sets

No	Datasets	Short description	References
1	E-OBS	E-OBS is a daily gridded observational dataset over containing daily maps spanning the period from 1 January 1950 to the present, at four different grid resolutions for five weather variables (daily mean temperature, daily minimum temperature, daily maximum temperature, daily precipitation sum and daily averaged sea level pressure). It was developed within the European Union Framework 6 ENSEMBLES project [van der Linden and Mitchell, 2009] with the main goal to support the validation of Regional Climate Models (RCMs) and improve understanding of climate change impacts.	Haylock et al. (2008)
2	EURO4M-APGD	APGD (Alpine precipitation grid dataset) is a high-resolution grid dataset of daily precipitation developed at MeteoSwiss and covering the entire Alpine region. It was established as part of the EU FP7 The European Reanalysis and Observations for Monitoring - EURO4M project	Isotta et al. (2014)
3	LUISA	The Land Use-based Integrated Sustainability Assessment modeling platform is a land function model that is developed by JRC, which is primarily used for the ex-ante evaluation of EC policies that have a direct or indirect territorial impact. The core outputs are projected land use, population and accessibility distributions at the 100m grid cell level.	Baranzelli et al. (2014)
4	CORINE	The CORINE Land Cover (CLC) inventory was initiated in 1985 (the reference year 1990). Updates have been produced in 2000, 2006, 2012, and 2018. It consists of an inventory of land cover in 44 classes. CLC uses a Minimum Mapping Unit (MMU) of 25 hectares (ha) for areal phenomena and a minimum width of 100 m for linear phenomena	(CORINE Land Cover Copernicus Land Monitoring Service, n.d.)
5	DG ECFIN	Directorate General for Economic and Financial Affairs provided the necessary analysis and calculations used in The 2012 Ageing Report - Economic and budgetary projections for the 27 EU Member States (2010-2060). This report provided projections of age-related expenditure items such as pensions, health care, long-term care, education, and unemployment benefits. These projections were used as inputs for LUISA platform.	EC (2012)

4

PAPER 3: INTEGRATING BAYESIAN NETWORKS INTO ECOSYSTEM SERVICES ASSESSMENT TO SUPPORT DECISION-MAKERS AT THE RIVER BASIN SCALE



GRAPHICAL ABSTRACT OF THE THIRD PAPER

HIGHLIGHTS

- Integrate Bayesian Networks into ecosystem service assessment;
- The probability of the high value of ES will be lower than those for low values;
- There will be limited space to improve ES, especially for the nutrient retention;
- Useful tool for decision-makers in designing the best management practices.

Integrating Bayesian Networks into ecosystem services assessment to support decision-makers at the river basin scale

Hung Vuong Pham[a,b,*], Anna Sperotto[a,b], Silvia Torresan[a,b], Andrea Critto[a,b], Antonio Marcomini[a,b]

[a]*Fondazione Centro Euro-Mediterraneo sui Cambiamenti Climatici (CMCC), via Augusto Imperatore 16, 73100 Lecce, Italy*
[b]*Department of Environmental Sciences, Informatics and Statistics, University Ca Foscari Venice, Via Torino 155, 30172 Venezia-Mestre, Venice, Italy*

Abstract

Freshwater ecosystem is negatively affected by climate change and human interventions through the alteration of supply and demand. Recently, both quantitative and qualitative research on ecosystem services concentrates on assessing potential risks of those stressors, using empirical data incorporate with expert knowledge. This works aims to integrate Bayesian Networks in ecosystem service assessment which enables identification of main factors that affect the ecosystem, trade-offs among ecosystem services, and evaluation ecosystem services under different scenarios. This integrated model will be applied to the Taro River basin in Italy. The outcome of this model is the projection of ecosystem services under thousands of different climate, land-use and water demand scenarios. The results show that the probability of having high values of these services would be lower than the low values. Moreover, there will be a limit of space to improve those values, especially for nutrient retention services. The obtained results could be a valuable support to identify and prioritize the best management practices for sustainable water use, balancing the tradeoffs among services.

Keywords: Climate change, Human activities, Ecosystem services, Water yield, Nutrient retention,

1. Introduction

Ecosystem services (ES) are the services that ecosystems deliver and that contribute to human well-being (MEA, 2005). They include provisioning services (e.g. water for agriculture), regulating services (e.g. buffering of flood flows), cultural services (e.g. river viewing) and supporting services (role in nutrient cycling) (Aylward et al., 2005). Since the publication of (MEA, 2005), ES became a well-established topic in scientific research with many relevant sub-topics such as quantification of supply changes (Karabulut et al., 2016; Khoi and Suetsugi, 2014; Lin et al., 2019; Pervez and Henebry, 2015; Polce et al., 2016; Vaighan et al., 2017; Weber et al., 2016), risk screening (Paroissien et al., 2015; Sample et al., 2016; Sperotto et al., 2019), mitigation policies (Giakoumis and Voulvoulis, 2018; Jorda-Capdevila et al., 2019; Nel et al., 2017) and potential tradeoff among ES (Hoyer and Chang, 2014; Hu et al., 2019; Wang et al., 2019). Nevertheless, the scientific community faced challenges to model ES, arising from the complexity of the ES and the lack of consistent data (Landuyt et al., 2013). Additionally, climate change and land-use change induce more pressure on that complex system by altering freshwater ecosystems on both the supply and demand (Pham et al., 2019b).
In this context, the introduction of Bayesian belief networks (BBNs or BNs) had the potential to bridge these gaps by integrating qualitative and quantitative data. Traditionally, BNs had various applications in environmental modeling, medical diagnoses, classification problems, and machine learning (Landuyt et al., 2013).
Applications of BNs in environmental modeling include habitat suitability models, risk assessments (Plomaritis et al., 2018; Pollino et al., 2007), management evaluation (Dang et al., 2019), decision support (Domínguez-Tejo and Metternicht, 2019; Jäger et al., 2018) and, more recently, ES modeling (Aguilera et al., 2011; Dang et al., 2019; Tang et al., 2019; Zeng and Li, 2019). Landuyt et al. (2013) reviewed the application of BNs in ES modeling and highlighted that there was a growth in the number of publications related to ES, integrating BNs in their application, due to the advances in model structure and computational time. First, BNs provide the possibility to separately model single ES production processes and to couple several sub-models. For instance, Tang et al. (2019) combined BNs with Soil and Water Assessment Tool (SWAT), Carnegie-Ames-Stanford Approach (CASA), Universal Soil Loss Equation (USLE), and Crop Productivity (CP) model to optimize four ecosystem services. Yet, BNs allows integrating various data such as expert knowledge, stakeholder involvement, model simulations, observational and empirical data (Kerebel et al., 2019). Second, BNs include a wide range of learning technics (i.e. constraint-based, score-based, and hybrid) and many inference algorithms (i.e. diagnostic and prognostic) (Scutari, 2017a, 2010, 2017b). Finally, BNs enhance the computational capacity and calculation speed of ES models (Landuyt et al., 2013).
The aim of this study is to demonstrate the advanced capability of BNs in ES modeling to (*i*) identify critical factors that al-

*Corresponding author
Email address: `vuong.pham@unive.it` (**Hung Vuong Pham**)

 DOI link will be available soon

low optimizing the supply of ES; (*ii*) assess the potential space to improve the capacity of ES; and (*iii*) quantify the capacity of ES under thousand scenarios, assisting decision-makers in developing management actions toward sustainable water. The Taro River basin (Italy) is selected as the case study to demonstrate this approach.

2. Materials and methods

2.1. Case study

2.1.1. Physical features of the study area

The Taro is a river in Emilia-Romagna region, in the northern part of Italy. It has 126km long and flows almost in the Parma province. The Taro River basin (TRB) is a sub-basin of the Po river basin, the most crucial basin in Italy. This basin has an area of 2026 km^2 of which 800 km^2 are in the floodplain areas (see Figure 1).

Figure 1: Taro River basin and its sub-basins.

Over the period 1999-2011, the average annual rainfall was about 1200 mm. The mean temperature over the land was around 12°C (Clerici et al., 2015). The mean annual discharge over the period 1916-2013 at the gauge station of S. Quirico, the closest to the river mouth, was 31 $m^3\ s^{-1}$ (Clerici et al., 2015). Land-use of the TRB is mainly characterized by woodlands (53%) in the upper and the middle parts, by agriculture (42%) in the lower parts while urban areas account for only 3.0% of the total land use (Pham et al., 2019a)

2.1.2. Pressures for ecosystem services

The surface water of the PRB is used primarily for agriculture and is directly abstracted from the stream by the Reclamation Consortium (i.e. Consorzio di irrigazione parmense, Societ Canale Naviglio Taro). Drinking water, on the other hand, is extracted from the groundwaters and managed by two integrated service companies, namely Emiliambiente and IRETI. Several threats to the ES have been found in the TRB such as morphological changes (Clerici et al., 2015), flooding (Mazzoleni et al., 2014), chemical pollution (Sammartino et al., 2007), quality of the surface water (Madoni and Zangrossi, 2005), and the quality of the groundwater (Iacumin et al., 2009; Toscani et al., 2007). These threads were induced by the excessive use of fertilizers and the rising demand for water for irrigation, together with changing patterns of precipitation. Therefore, it is important to understand how climate change and human intervention could affect the ecosystem services provided by freshwater ecosystems.

2.2. Bayesian Belief Networks

2.2.1. Theory background

Bayesian Belief Networks (BBNs), also known as Bayesian Networks (BNs), are multivariate statistical models, representing a set of random variables and their conditional interdependencies (Scutari, 2010). Each variable is presented in the BNs by a node. BNs includes two mandatory components, namely (*i*) a directed acyclic graph (DAG) that denotes dependencies and independencies between the models nodes; and (*ii*) conditional probability tables (CPTs) denoting the strengths of the links in the graph (Landuyt et al., 2013). A node is a graphical representation of the system's variables in BNs (Landuyt et al., 2015). The DAG can be constructed by expert-based knowledge on system understanding or can be learned by empirical observation. The statistical dependencies between different nodes are indicated by directed arrows that represent the cause-effect relations between the systems variables. Each arrow starts in a parent node and ends in a child node. The graph is acyclic, and therefore no feedback arrows from child nodes to parent nodes exist. Conditional Probability Tables (CPTs), indicating the strengths of the links in the graph by denoting the likelihood of the state of a child node given the states of its parent nodes. Each node contains a limited number of states to which their realized value can belong. The summary of the most essential terms using in BNs is reported in Table 1.

The most advanced of a BBN is to address uncertainties by integrating quantitative and qualitative data (Bicking et al., 2019; Landuyt et al., 2013). To obtain quantitative model outputs, Bayesian inference is used to propagate these probabilities through the network. Bayesian inference is based upon the Bayesian theorem as the follow:

$$Pr(A|B) = \frac{Pr(B|A)\,Pr(A)}{Pr(B)} \tag{1}$$

Equation 1 indicates that the conditional probability, or posterior probability, of an event A after event B ($Pr(A|B)$) is observed in terms of the prior probability of A ($Pr(A)$), the conditional probability of B given A ($Pr(B|A)$) and the prior or marginal probability of B ($Pr(B)$). More detail about BNs is well-documented in (Ronquist, 2004).

2.2.2. Input data for the BNs

The quality of the input data is key for the applicability of the BBN-based assessment application (Bicking et al., 2019). Data sets for the TRB case study were inherited from the previous work done by (Pham et al., 2019a). They are mixed of climate data (i.e. precipitation), land-use (LU), soil data (root depth, depth to rock, etc.), and measurement data (i.e. water

Table 1: Frequently used Bayesian belief network terminology. Adapted from Landuyt et al. (2015)

Terminology	Description
Nodes	Graphical representation of the system's variables in a Bayesian belief network model
State	A value, discrete class or qualitative level a variable can be assigned to. Each variable in a Bayesian belief network model has a set of states it can manifest
Probability distribution	The set of probabilities assigned to the states of a variable that expresses the probability that the variable is in one of its states. This set of probabilities always sums up to 1
Arrows	Graphical representation of the causal relations among the system's variables. Each arrow flows from a parent node to a child node. Together with the nodes, they define a Directed Acyclic Graph (DAG)
DAG	Directed Acyclic Graph illustrates the dependencies between the different variables in the BBN. DAG contains only directed arcs but does not include any loop or cycle
Conditional probability	The probabilities that quantify the model's causal relations and express the probability distribution of a child node given the status of its parent nodes
Conditional probability table	A table that contains a child node's conditional probability distributions for all possible combinations of its parent nodes' states. Abbreviated as CPT
To instantiate	Assigning a 100\% probability to one of the states of a variable. This can be done, for example, in case the exact value of that variable is known

demand). These data sets have different temporal and spatial resolutions. Thus, they have been cleaned, processed and discretized in order to fulfill the input requirements of the BBN. Specifically, spatial data sets were visualized and processed in Q-GIS. Then, these treated data, together with other non-spatial data sets, were integrated in R. Finally, these data were discretized into several states which are labeled with appropriate legends. For instance, the states of most variables, except for YEAR and LU, were denoted as L, M, and H, corresponding to the low, medium, and high state. The list of the selected variables and their abbreviations in BNs is listed in Table 2 while the value and the legend of the states are given in A.1. They include 22 key variables of the Water Yield and the Nutrient Retention module (Sharp et al., 2016). The technical detail of these variables can be found in (Pham et al., 2019a).

2.3. Study design

The BNs for the Taro river basin was implemented and run using the package bnlearn in R (Scutari, 2017a). To achieve our objectives, our workflow was organized as follows:

- BNs structure learning and training
- BNs calibration and validation
- Scenario building

2.3.1. BNs structure learning and training

The phase of model learning aims at developing an influence (i.e. box and arrow) diagram providing a graphical representation of the system under consideration. The network conceptualization, therefore, includes the identification of the main system variables (i.e. nodes) as well as the links between them (i.e. directed arcs). The identification of relevant variables and links can be typically based on a literature review, expert knowledge, and consultation with local stakeholders. For each variable, appropriate indicators as much as possible measurable, observable and predictable have to be identified or define by means of specific structure learning methods. Once the variables and relative indicators are defined, the links between them are identified and represented. In this case, structure learning and training processes were performed as the flows.

First, the DAG was elicited by expert knowledge and derived from the mathematical equations of the WY and NR module of the InVEST model. Accordingly, as stated earlier, we selected 22 key variables to develop the BNs with three target nodes (i.e. WY, Nr, and Pr). Then, these DAGs were improved by two different structure leaning algorithms, namely constraint-based (e.g. grow-shrink and incremental association) and score-based algorithm (e.g. hill-climbing and tabu search) (Scutari, 2010). The constraint-based algorithms use statistical tests to learn conditional independence relationships from the data and assume that the DAG is a perfect map to determine the correct network structure. Meanwhile, in the Score-based algorithms, each candidate DAG is assigned a score reflecting its goodness of fit, which is then taken as an objective function to maximize. All these learning algorithms aim to improve the DAG by suggesting some further arcs we may have overlooked.

Second, the DAG had to be trained to translate from the conceptual model into a probabilistic one. BN training and involves assigning states, prior and conditional probabilities to all nodes of the networks, For each node, a certain number of states must be identified. States represent potential values or conditions that the variable can assume in the analyzed system (Kragt, 2009) and can be featured differently, representing Boolean functions (e.g. true, false), categorical definitions (e.g. low, medium, high), continuous or discrete numeric intervals (Martín de Santa Olalla et al., 2005). Once the type and number of states have been defined, the prior probability associated with each state of

Table 2: List of variables and their code in BNs

No	Short name	Full name	Unit	Source
1	PRE	Precipitation	mm	CMCC
2	AET	Actual evapotranspiration	mm	InVEST model
3	LU	Land-use	-	LUISA
4	DTR	Depth to root restricting layer	mm	ESDAC
5	PAWC	Plant available water content	-	ESDAC
6	ETO	Reference evapotranspiration	mm	JRC
7	KC	Evapotranspiration coefficient	-	adapted from Pham 2019
8	RD	Root depth	mm	(Jorda-Capdevila et al., 2019)
9	WD	Water demand	m3/yr	ISTAT
10	YEAR	Year	-	-
11	Neff	Nitrogen retention efficiency	-	adapted from Pham 2019
12	Peff	Phosphorous retention efficiency	-	adapted from Pham 2019
13	DEM	Digital elevation model	mm	EEA
14	Ns	Nitrogen source	kg/yr	InVEST model
15	Ps	Phosphorous source	kg/yr	InVEST model
16	Nl	Nitrogen load	kg/yr	InVEST model
17	Pl	Phosphorous load	kg/yr	InVEST model
18	Ne	Nitrogen export	kg/yr	InVEST model
19	Pe	Phosphorous export	kg/yr	InVEST model
20	Nr	Nitrogen retention	kg/yr	InVEST model
21	Pr	Phosphorous retention	kg/yr	InVEST model
22	WY	Water Yield	m3/yr	InVEST model

the node has to be calculated based on available information and knowledge (Pollino et al., 2007). Accordingly, the prior probability distribution describes the starting point for each node in the network and thus the expectation of the node being in a particular condition given current knowledge and data. Finally, to operationalize the network, Conditional Probabilities (CPs) of nodes must be specified for all combinations of states of its parent nodes. This parameter training process was implemented under the `bn.fit` function on the `bnlearn` package in R.

2.3.2. BNs calibration and validation

The calibration process was performed along with the structure leaning and parameter learning procedure by assessing the classification error of the target nodes, presenting the goodness of fit. To be more specific, this task was performed by comparing the predicted value of trained BN and the observed data of the target nodes. The aim of calibration is to minimize the error by improving the DAG and the local parameter throughout the iterations of structure and parameter learning procedure. Once the BN was calibrated, we then check the predictive accuracy of the classifier assessing the ability of the BN to correctly predict instances on an independent dataset. Specifically, we used performed cross-validation to obtain an estimate of the predictive classification error. The golden standard is 10 runs of 10-fold cross-validation, using `bn.cv` function with method k-fold (Scutari, 2017a). This procedure was applied to all selected nodes.

2.3.3. Scenario building

Once trained and evaluated, the BN model can be used for scenarios analysis allowing to assess the relative changes in outcomes probabilities associated with changes in input variables (i.e. predictive function) or to define the state in which input variable should be to obtain the desired outcome (i.e. diagnostic function). Therefore, different scenarios were built to respond to three critical questions such as (*i*) what are the main drivers that affect ES; (*ii*) what are the likely outcomes of ES given that these drivers are optimized; and (*iii*) what are the possible outcomes of ES with different conditions. There are three groups of scenarios, namely Optimal World, Ideal World, and Possible World.

The first group (**Optimal world OPT**) focuses on the maximization of ES using diagnostic inference to find the best combination of other key drivers such as PRE, LU, and WD, etc. We select different optimization target nodes to identify the effects of different drivers on the ES. Specifically, the first three scenarios (OPT1, OPT2A, and OPT2B) focus on the optimization of single ES, namely WY, nutrient retention (Nr, Pr), and nutrient source (Ns, Ps). The last two scenarios (OPT3A, OPT3B) focus on the optimization of all ES such as the maximization of WY in the combination with the maximization of Nr and Pr or the minimization of Ns and Ps.

The second group (**Ideal world - IDE**) focuses on the quantification of ES given that the main drivers are in the best conditions. For instance, the scenario IDE1 considers the best condition of the WD node, i.e. 100% is assigned for the state high (H). The scenario IDE2 and IDE3 assume the best condition of precipitation (i.e. 100% for the state high - H) and land use (i.e. 100% for the state woodlands - WLS), respectively. Then, the last one (IDE4) examines the combination of all the above aspects.

Finally, the last group (**Possible world - IDE**) focuses on the quantification of ES with all possible scenarios that may happen. In the first scenario (POS1), we consider the projection of precipitation from COSMO@CMCC RCP 4.5 under the assumption that the emission level will reach this scenario instead of RCP 8.5. Then, the second scenario (POS2), land-use distribution was obtained from LU@CMCC, considering the influence of demographic change and the preservation of protected areas. Finally, the scenario POSn is developed to consider the uncertainty of input variables in the real world. Specifically, the distributions of the main inputs (e.g. LU, PRE, and WD) are randomly assigned to capture the uncertainty of these variables and the combination among them. This procedure is repeated 5000 times to increase the robustness of this approach. The summary of those scenarios is given in Table 3.

3. Results

3.1. BNs learning

3.1.1. Structure learning

Figure 2 presents different learned BNs with different learning algorithms such as hill-climbing (hc) and tabu (tabu) search grow-shrink (gs) and incremental association (iamb). The gray arcs represent the original BN that built from an expert election and the mathematical equation of the InVEST model.

It can be seen from Figure 2 that the constraint-based algorithms added more arcs to the original BN than those suggested by the score-based methods. Moreover, two score-based methods suggested the same six potential arcs. Based on these results, the structure was improved by adding some meaning DAGs (e.g. DTR PAWC, ET0 PAWC, PRE WY, and ET0 WY). The final structure of BN after the learning process is shown in the bottom right panel of Figure 2.

3.1.2. Parameter learning

The final BN was trained with data sets describing in section 2.2.2. The input data sets were discrete values; thus, they were assigned to specific state, using discretize function in R. The numbers of states for each variable (or node) depended on its natural characters. For instance, the node YEAR was discretized into five states since the original data was collected from these years. The node LU was discretized into only four states, corresponding to four types of land use, namely woodlands (WLS), meadowlands (MDS), agriculture (AGR), and urban (URB). Other nodes were classified into three or two states to reduce the complexity of the networks and to avoid non-value in each state. For convenience, these states were assigned short labels such as low (L), medium (M), and high (H). The detail of all states for all nodes is described in Table A.1. The overview of the trained BN, together with all states and related marginal distribution, is shown in Figure 3.

3.2. Model calibration and validation

The classification errors were 7.04%, 0.47%, and 0.46% for the WY, Nr, and Pr, respectively. These numbers demonstrated a high quality of goodness of fit. BNs were able to provide a good prediction of target nodes.

Table 4 shows the average classification errors of each key variable for each iteration of cross-validations. The mean losses of precipitation, nitrogen retention and phosphorous retentions are 17.6%, 0.2% and 0.2%, respectively. This means that the performance of the designed BN was much better for nutrient retention than those for the water yield service. Overall, this BN could provide good inferences and optimizations in different scenarios.

3.3. Scenarios

3.3.1. Diagnostic inference

Figure 4 shows the marginal probability distribution of the critical variables (e.g. LU and PRE) in order to maximize the target nodes (i.e. WY, Nr, and Pr). Regarding land-use, the results suggest that the conversion from all types of land-use to urban (URB) would lead to the increase of nutrient retention due to the abundance of nutrient sources from industry activities (see the scenario OPT2A and OPT3A). On the contrary, the conversion from urban to other states such as woodlands could maximize water yield services (see the scenario OPT2B and OPT3B).

Concerning precipitation, Figure 4b shows that the water yield service could be maximized by converting state Low to High in the node PRE as in OPT1, OPT3A, and OPT3B. While precipitation acted as the key variable to optimize WY, it did not show a significant role in maximizing nutrient services as in OPT2A and OPT2B.

3.3.2. Prognostic inference

Figure 5 reports the marginal probability of the state H of water yield and nutrient retention services (i.e. reported as the average value of Nr and Pr). The same charts for the state M and L can be found in Figure B.1 and Figure B.2. The black bubble represents the prior probability of the state H that learned from the input data while the orange ones imply the posterior probability that deriving from different scenarios. The size of the bubble the summation of the marginal probabilities of both services. It can be seen from Figure 5 that IDE2 and IDEA4 are the best scenarios because we observe significant improvement for both services or at least for the water wield. The other scenarios show some negative changes in respect to the reference PrP.

Figure 6 reports the results of WY and NR for 5000 random scenarios (POSn). It can be seen that the distribution of the Low L state is on the upper-right corner while the high H and medium M states concentrate on the bottom-left of the graph. The black dots represent a high value of WY and Nr. Therefore, the higher the value of this state, the better for the environment. The mean values of the state H, M, and L of WY are 10.2%, 30.9%, and 58.5%, respectively. Those numbers for NR are 1.8%, 0.9%, and 97.3%, respectively.

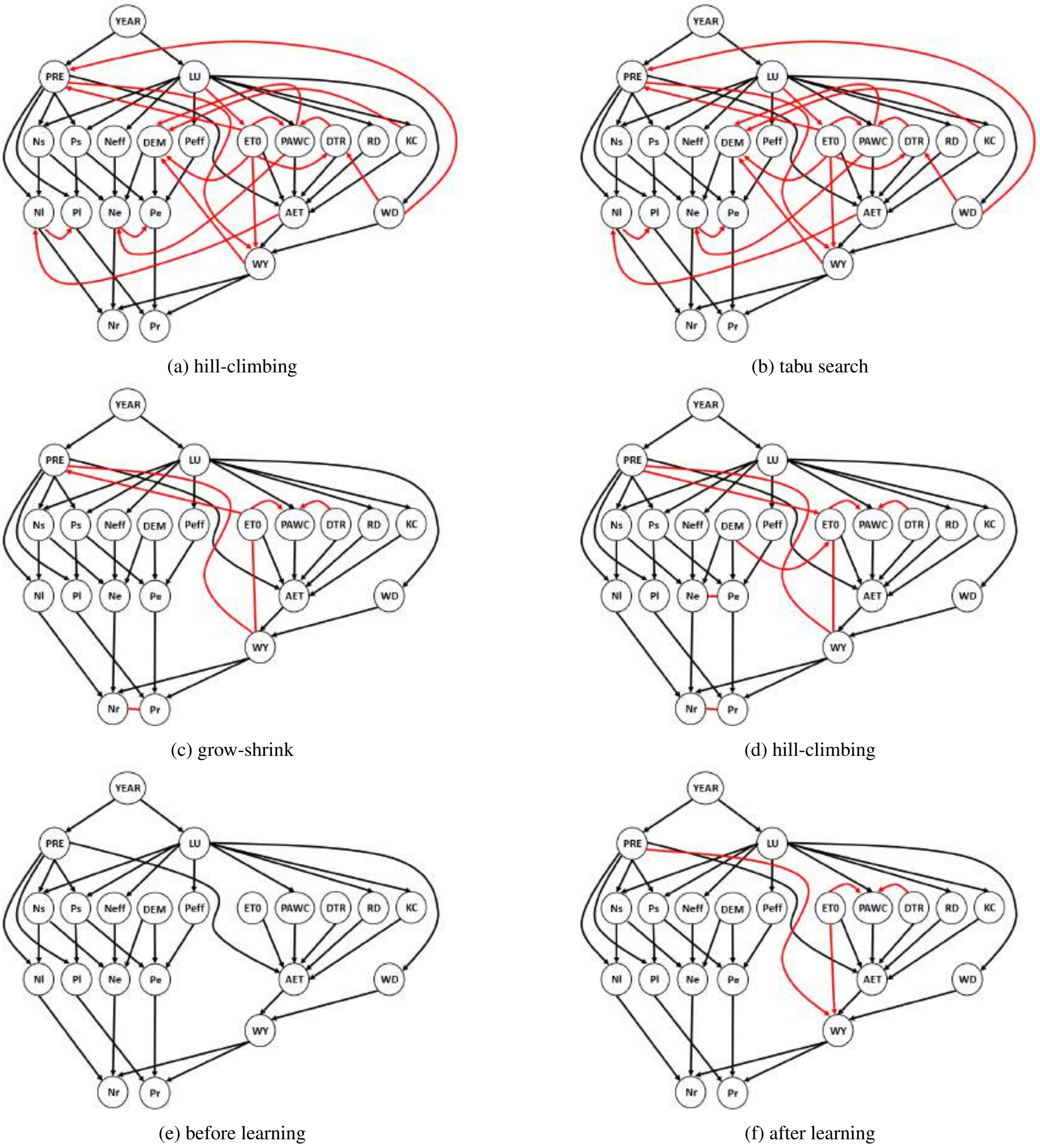


Figure 2: BNs with different learning algorithms (hill-climbing (hc), tabu (tabu), search grow-shrink (gs), and incremental association (iamb)). The black arrows represent the pre-defined arcs; the red arrows represent the further arcs suggested by different learning algorithms. The original and final structure are shown in the bottom left and bottom right panel, respectively.

Table 3: List of scenarios (Abbreviation: OPT - Optimal; IDE - Ideal; POS - Possible)

Group	Short name	Description	Type of inference	Evidence
Optimal world(OPT)	OPT 1	Maximization of Water Yield	Diagnostic	nodes="WY",states="H"
	OPT 2A	Maximization of nutrient retention (Nr and Pr)	Diagnostic	nodes=c("Nr","Pr"), states=c("H","H")
	OPT 2B	Minimization of nutrient source (Ns and Ps)	Diagnostic	nodes=c("Ns","Ps"), states=c("L","L")
	OPT 3A	Maximization of WY, Nr and Pr	Diagnostic	nodes=c("WY","Nr","Pr"), states=c("H","H","H")
	OPT 3B	Maximization of WY and minimization of Ns and Ps	Diagnostic	nodes=c("WY","Ns","Ps"), states=c("H","L","L")
Ideal world (IDE)	IDE 1	Best scenario of water demand	Prognostic	nodes=c("WD"), states=c("L")
	IDE 2	Best scenario of precipitation	Prognostic	nodes=c("PRE"), states=c("H")
	IDE 3	Best scenario of land use	Prognostic	nodes=c("LU"), states=c("WLS")
	IDE 4	Best scenario of WD, PRE and LU	Prognostic	nodes=c("WD","PRE","LU"), states=c("L","H","WLS")
Possible world(POS)	POS 1	Projection of future precipitation under RCP 4.5	Prognostic	PRE=c(0.25,0.35,0.45)
	POS 2	Projection of future land use based on LU@CMCC model under RCP 8.5	Prognostic	LU=c(0.705,0.025,0.26,0.01)
	POS n	All possible combination of the key inputs (WD, LU, PRE)	Prognostic	LU =random; PRE = random, WD = random

Table 4: Results of ten folds cross-validation technique

Variable	Classification error for each iteration									
	(1)	(2)	(3)	(4)	(5)	(6)	(7)	(8)	(9)	(10)
PRE	17.679	17.615	17.696	17.713	17.601	17.601	17.635	17.554	17.615	17.567
Nr	0.207	0.204	0.211	0.194	0.204	0.200	0.207	0.194	0.214	0.211
Pr	0.170	0.173	0.173	0.177	0.180	0.180	0.180	0.173	0.183	0.163

4. Discussion and conclusions

4.1. Strengths and limitations

The major advantages of this Bayesian network modeling approach are the model transparency which enables stakeholder involvement in model development and evaluation, the possibility to incorporate both empirical data and expert knowledge, a straightforward combination with valuation studies and the inherent consideration of uncertainties in a transparent way. In this work, BNs showed their advantages in integrating different kinds of data with different units such as climate data (PRE), land-use (LU), soil data (RD, and DRT), and measurement data (WD).

Nevertheless, a significant weakness of BNs to model ES is their limited capacity to model systems spatially and dynamically (Landuyt et al., 2013). In our case study, although the input data sets expand for the period (2010-2050), with a spatial resolution of 1kmx1km, the model lost this spatiotemporal resolution in the final outputs.

4.2. Main finding

In this case study, we applied two different Bayesian inferences (i.e. diagnostic and prognostic) for different purposes. The former aimed to explore the probability distribution of the most important input variables in order to maximize the interested nodes (i.e. WY, Nr, and Pr). Meanwhile, the latter inference aims to query the posterior probability distribution of the output nodes, given that the inputs were known. The results of these query groups are discussed in sections 4.2.1, and 4.2.2, respectively.

4.2.1. The influence of key variables to the ES

The results of the optimal (OPT) scenarios lead to some directions to maximize ES. First, the scenario OPT2 and OPT3 suggested that Nr and Pr could reach the maximum value if all land-use classes were converted to urban (URB) (see Figure 4a). This conversion would lead to the abundance sources of N, P and, consequently, the improvement of Nr and P, as the results of the increase in the load of N and P. Nevertheless, the

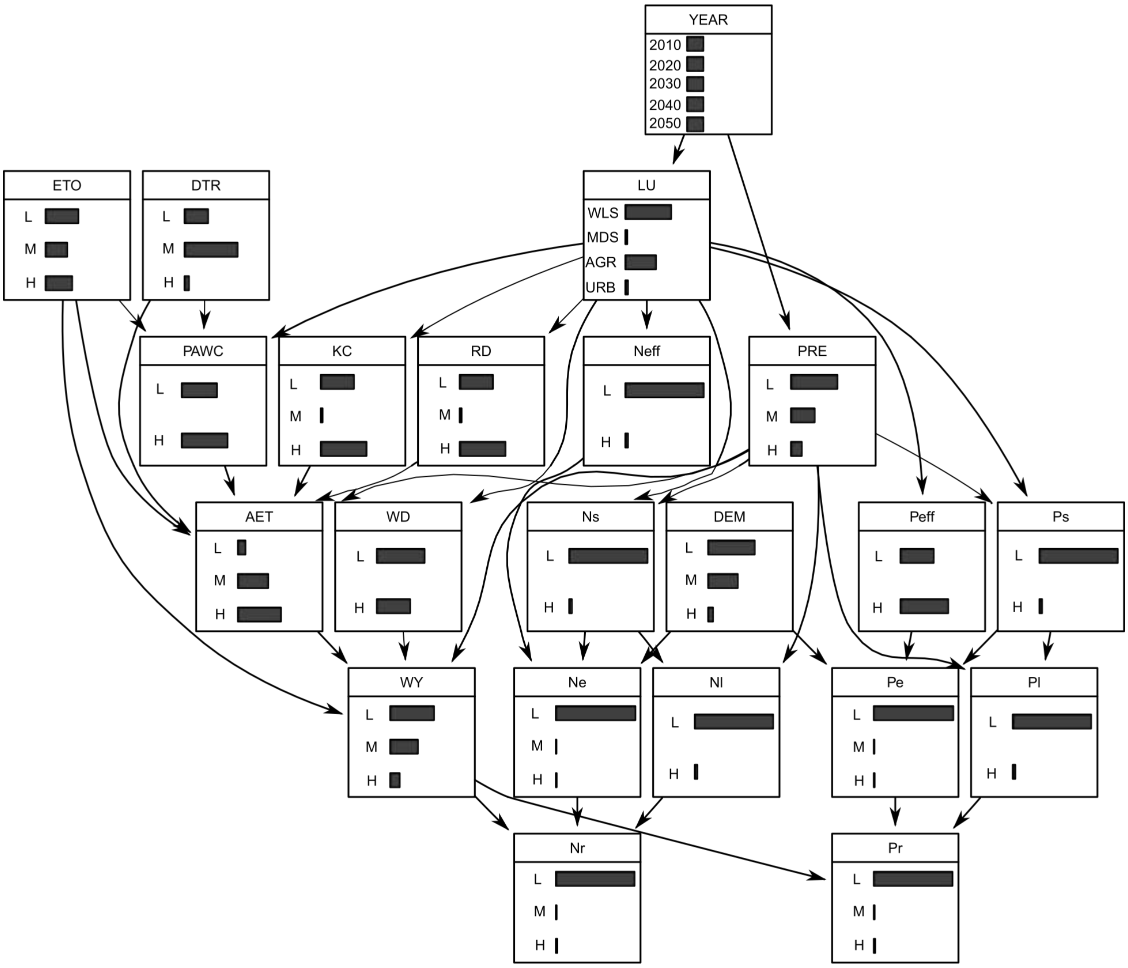


Figure 3: Trained BN and the marginal distributions associated with the nodes.

transformation to urban for the whole TRB is definitely a non-feasible solution. Therefore, these scenarios did not provide useful information for environmental managers and decision-makers. We also argued that the selection of Nr and Pr as the nodes for maximization was not an appropriate choice and it may lead to the misrepresentation of the outputs.

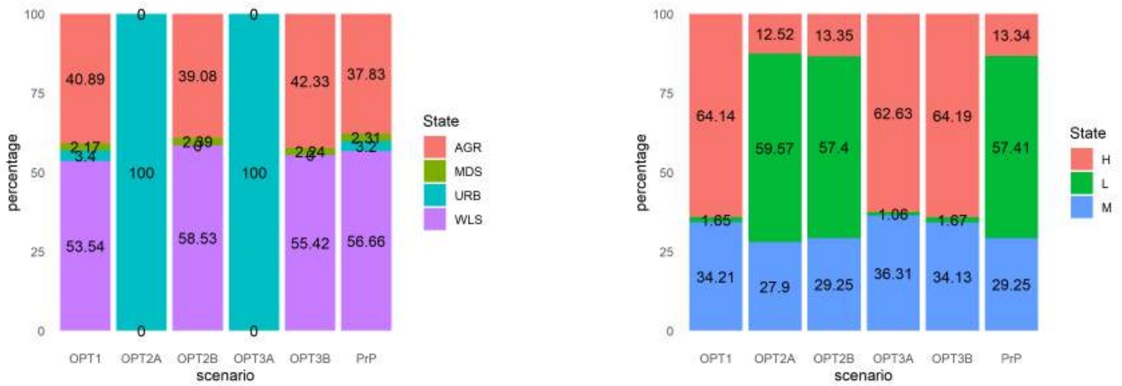
Second, the results from the scenario OPT2B and OPT3B, where we minimized the nutrient sources (Ns and Ps), proposed that we could improve ES by the transformation of urban into greener land-use such as 1.8% and 1.3% for wetlands and agriculture, respectively. Although urban accounted for only 3.2% among other land-use types, the conversion of this class had a significant impact on ES. Thus, we concluded that urbanization is the major factor that affects the quality of ES, especially nutrient retention.

Regarding the effect of precipitation on ES, Figure 4b illustrates that rainfall patterns had significant impacts on water yield. Specifically, scenario OPT1, OPT3A, and OPT3B suggested that shifting the state of precipitation (PRE) distribution from low (L) to high (H) of about 51% could lead to the high value of state H of WY in those simulations. Nevertheless, this transformation did not have an important impact on Nr and Pr.

4.2.2. *Projection of ES with different scenarios*

While section 4.2.1 investigates the effect of the most important variable on ES, this section explores the possible distribution of ES under predefined different scenarios by implementing an ideal hypothesis and possible distribution of input data into BN. Prognostic queries were used to find the distribution of ES under those conditions. Since our ambition was to increase the value of ES, the state H was selected as the indicator to quantify. Figure 5 reports the probability of state H of these ES with different scenarios.

First, Nr and Pr had a similar trend in all simulations (see Table B.4 and Table B.5). This phenomenal is because nitrogen and phosphorous have the same original sources from urban and agriculture. Even though their primary sources might differ (i.e. point source from urban and distribution sources from agriculture for phosphorus and nitrogen, respectively), they had the same dilution and transportation process. Thus, they have



(a) Node LU (b) Node PRE

Figure 4: Marginal probability distribution of key variables (LU on the left panel and PRE on the right panel) using diagnostic inference

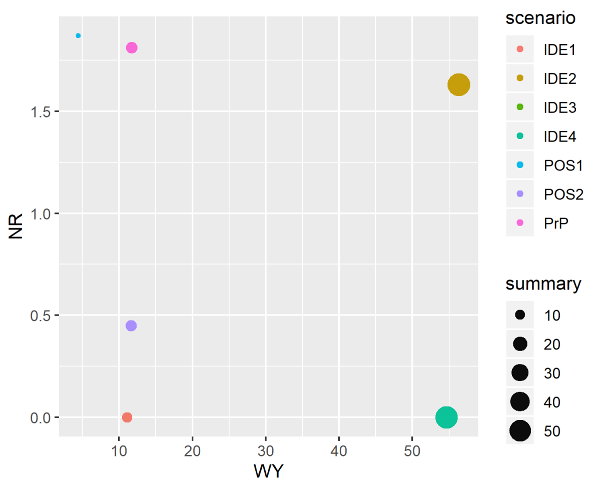


Figure 5: Marginal probability of state High (H) of WY and NR services. The size of the bubble represents the summation of these two marginal probabilities

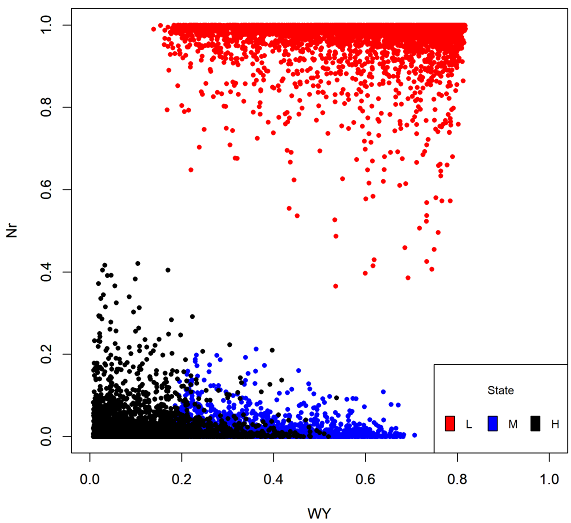


Figure 6: Results of prognostic inference for the ES nodes (WY vs Nr) under the scenario POSn (The number of iterations is 5000)

the same behavior in the load, export and retention capacity. Second, WY had higher probabilities to reach high value (i.e. state H) than those in Nr and Pr. Specifically, the mean values for WY and nutrient retention were about 20.6% and 0.6%, respectively. Therefore, we argued that there was a trade-off among these services, and we might not improve them equally. Third, as shown in Figure 5, scenario IDE2 and IDE4 were the preferable scenarios since we found the highest probability to reach the H state for both ES. Among these, the performance of IDE2 was better than those of IDE4, with an improvement of 1.6% for Nr. Yet, the results from IDE2 indicated that this scenario could lead to an increase of 44% in total for both services, respect to the scenario PrP. Then, we argue that there will be lots of space to improve these services.

Fourth, the overall performances of the simulations POS1 and POS2, with different possible configurations of projected land-use and climate change, were worse than those in the PrP. This deterioration was due to the decline of water yield, as the consequence of the reduction of precipitation when moving from RCP8.5 to the RCP4.5. We also observed this behavior in the performances of Nr in most scenarios, except for the POS2. From above, we argued that mitigation strategies (i.e. to reach emission level RCP4.5) did not imply the improvement of WY.

Finally, the results from 5000 random simulations gave us some interesting insights. First, the black dots in Figure 6 represent the good ecological status, associating with the high value of ES. Unfortunately, there is a limited space to improve this good state since the maximum values of this state for WY and NR are 53% and 41%, respectively. Therefore, decision-maker can rely on the scenarios which bring the black dots to move to the upper-right of the graph while design adaptation plan by investigating the configuration of those scenarios. Second, the H and M state concentrates mainly on the bottom of the graph, near the WY axis. This phenomenon reflects that there is more probability to improve WY services rather than NR. Specifically, the maximum probability of the state H for the NR is about 42%. Thus, the management actions should prioritize improving the NR services rather than WY.

4.3. Conclusions

This work demonstrated the application of BN in ES assessment, utilizing different kinds of data, different learning, and inference algorithms. Thanks to their advantages, thousands of simulations were conducted with different configurations, presenting the uncertainty of climate condition, land-use change and water demand in the real world. The outcomes suggested that the probability of the high value of ES will be lower than those for low values. Moreover, there will be limited space to improve these services, especially for the NR. Thus, the measures should focus on the improvement of NR to keep the balance among ES.

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Appendices

A. Appendix A: BNs and input data

Table A.1: List of variables and their states

No	Variable	State	Legend	No	Variable	State	Legend
1	PRE	[647-1.27e+03)	L	11	Peff	[0.015-0.408)	L
		[1.27e+03-1.89e+03)	M			[0.408-0.8]	H
		[1.89e+03-2.52e+03]	H	12	DEM	[27.3-569)	L
2	AET	[176-357)	L			[569-1.11e+03)	M
		[357-538)	M			[1.11e+03-1.65e+03]	H
		[538-719]	H	13	Ns	[18-616)	L
3	LU	Woodlands	WLS			[616-1.22e+03]	H
		Meadowland	MEA	14	Ps	[0.4-41.3)	L
		Agriculture	AGR			[41.3-82.3]	H
		Urban	URB	15	Nl	[1.12-38.5)	L
4	DTR	[0-533)	L			[38.5-75.9]	H
		[533-1.07e+03)	M	16	Pl	[0.025-2.58)	L
		[1.07e+03-1.6e+03]	H			[2.58-5.14]	H
5	PAWC	[0.129-0.149)	L	17	Ne	[0-22.7)	L
		[0.149-0.169]	H			[22.7-45.4)	M
6	ETO	[588-666)	L			[45.4-68.1]	H
		[666-744)	M	18	Pe	[0-1.55)	L
		[744-822]	H			[1.55-3.09)	M
7	KC	[0.85-0.9)	L			[3.09-4.64]	H
		[0.9-0.95)	M	19	Nr	[0-25.3)	L
		[0.95-1]	H			[25.3-50.6)	M
8	RD	[980-2.82e+03)	L			[50.6-75.9]	H
		[2.82e+03-4.66e+03)	M	20	Pr	[0-1.72)	L
		[4.66e+03-6.5e+03]	H			[1.72-3.43)	M
9	WD	[0-1.25e+03)	L			[3.43-5.14]	H
		[1.25e+03-2.49e+03]	H	21	WY	[55.1-727)	L
10	Neff	[18-616)	L			[727-1.4e+03)	M
		[616-1.22e+03]	H			[1.4e+03-2.07e+03]	H

B. Appendix B: Results of the diagnostic and prognostic inferences

B.1. Results of the diagnostic inferences

Table B.1: Results of the diagnostic inferences for the LU

State	PrP	OPT1	OPT2A	OPT2B	OPT3A	OPT3B
WLS	56.66	53.54	0.00	58.53	0.00	55.42
MDS	2.31	2.17	0.00	2.39	0.00	2.24
AGR	37.83	40.89	0.00	39.08	0.00	42.33
URB	3.20	3.40	100.00	0.00	100.00	0.00

B.2. Results of the prognostic inferences

Table B.2: Results of the diagnostic inferences for the PRE

State	PrP	OPT1	OPT2A	OPT2B	OPT3A	OPT3B
L	57.4	1.7	59.6	57.4	1.1	1.7
M	29.3	34.2	27.9	29.3	36.3	34.1
H	13.3	64.1	12.5	13.4	62.6	64.2

Table B.3: Results of the prognostic inferences for the WY

States	PrP	IDE1	IDE2	IDE3	IDE4	POS1	POS2
L	54.11	55.7	20.0	55.7	20.1	67.8	54.4
M	34.16	33.2	23.6	33.2	25.3	27.8	34.0
H	11.72	11.1	56.4	11.1	54.6	4.4	11.6

Table B.4: Results of the prognostic inferences for the Nr

States	PrP	IDE1	IDE2	IDE3	IDE4	POS1	POS2
L	97.27	100.0	97.5	100.0	100.0	97.2	97.2
M	0.92	0.0	0.9	0.0	0.0	0.9	0.9
H	1.81	0.0	1.6	0.0	0.0	1.9	1.9

Table B.5: Results of the prognostic inferences for the Pr

States	PrP	IDE1	IDE2	IDE3	IDE4	POS1	POS2
L	97.27	100.0	97.5	100.0	100.0	97.2	99.3
M	0.97	0.0	0.9	0.0	0.0	1.0	0.2
H	1.76	0.0	1.6	0.0	0.0	1.8	0.4

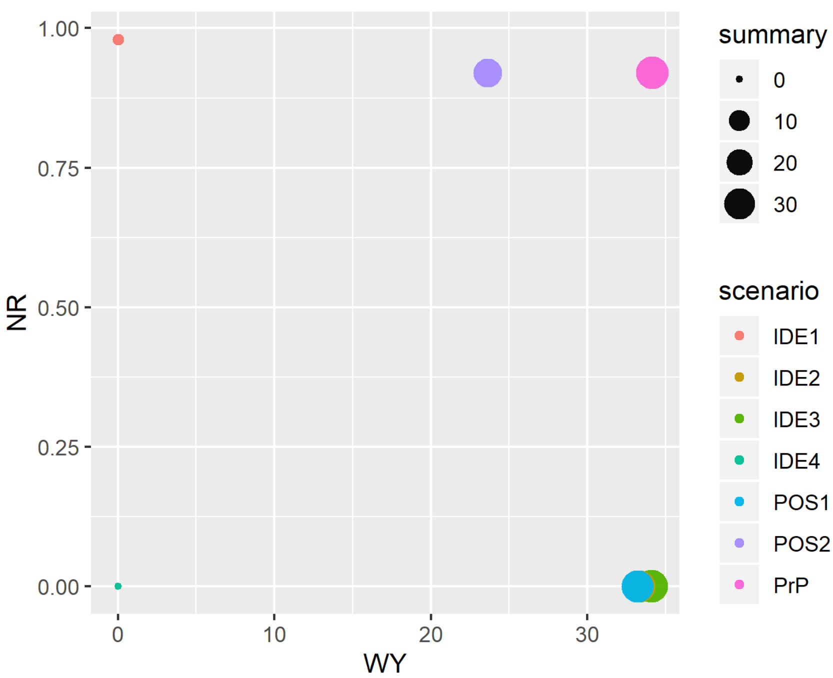

Figure B.1: Marginal probability of state Medium (M) of WY and NR services. The size of the bubble represents the summation of these two marginal probabilities

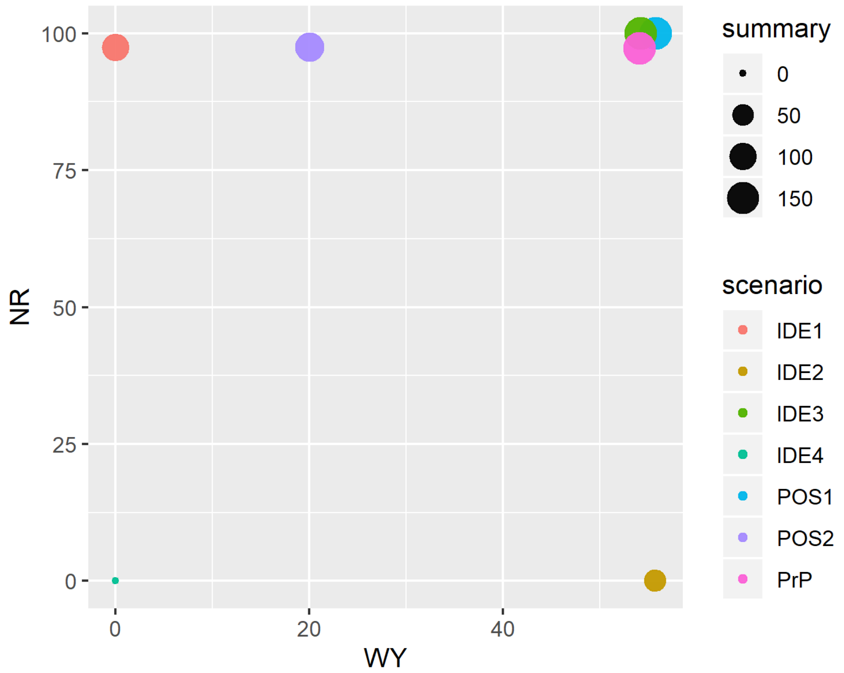

Figure B.2: Marginal probability of state Low (L) of WY and NR services. The size of the bubble represents the summation of these two marginal probabilities

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Estratto per riassunto della tesi di dottorato

Studente: Hung Vuong PHAM matricola: 956329

Dottorato: Science and Management of Climate Change

Ciclo: 32

Titolo della tesi[1] :

Multi-risk assessment of freshwater ecosystem services under climate change conditions

Abstract:

Freshwater provides a wide range of ecosystem services such as provisioning, regulating, cultural, and supporting services. The supply and demand of freshwater are negatively affected by climate, land-use change, and human interventions. In this work, we propose a conceptual framework and a set of indicators for assessing the impacts mentioned above on freshwater-related ecosystem services. Then, based on this framework, we developed an integrated modeling approach, coupling climate, and land-use data dynamically into ecosystem model, to assess the conjoined impacts of land-use and climate change on the selected services (i.e. water yield and nutrients retention). Finally, we integrated Bayesian Networks into ecosystem model to evaluate thousands of random scenarios, representing the sophisticated world. The outcomes of this work could be a valuable support to identify and prioritize the best management practices for sustainable water use, balancing the tradeoffs among services.

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