Zara Fani Gonçalves Teixeira

RESILIENCE TO LAND USE INDUCED IMPACTS

Tese de Doutoramento em Biociências, na especialidade de Ecologia, orientada pelo Professor Doutor João Carlos Marques e apresentada ao Departamento de Ciências da Vida da Faculdade de Ciências e Tecnologia da Universidade de Coimbra.

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Universidade de Coimbra

Resilience to land use induced impacts.

A contribution to improve precautionary ecological management in coastal ecosystems

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Dedicada às mulheres da minha vida

e ao meu caçula

Contents

Abstract

The European Water Framework Directive enhances the role of land use in determining water quality of all waters, but the use of land cover data, regarding surface waters, is restricted to the estimation of land use patterns for the identification of pressures on surface waters from diffuse source pollution.

The main goal of this study is to uncover specificities of land cover in the assessment of state changes (Chapter I), pressures (Chapter I and II) and drivers (Chapter III and IV) of water quality of surface waters.

For this purpose, we use data from the Mondego river basin, a coastal watershed in Portugal, where socio-economic changes observed since the 1990s resulted in the loss of representativeness from the agricultural sector and gain from the construction sector. Despite the observed changes, agriculture remained a source of pressure on surface waters which, associated with hydro-morphological modifications, caused a decline of the ecological quality of the Mondego estuary. In 1998, the implementation of mitigation measures enabled the recovery of the system but routine monitoring identified high concentrations of nutrients deserving further assessment to understand its sources. For this reason, the Mondego river basin, as a case study, provided conditions to reveal wider applications of land cover data within the assessment of drivers and pressures of water quality.

Chapter I establishes the baseline for our research applying indicators of nutrients and oxygen-consuming-substances to analyze progress in the concentrations of ammonia, nitrate and phosphate in the Mondego estuary from a former period (2003-2007) to a recent period (2012-

2013) (ANOVA in R software); to analyze differences between the annual mean concentrations of nutrients in the tributary rivers in 2012-2013 and the average annual mean concentrations found in European rivers in three different periods (1992, 2000, 2012); and to evaluate differences in the variability of nutrient concentrations between the estuary and the tributaries in 2012-2013 (ANOVA in R software). Additionally, Chapter I evaluates the dependency of estuarine nutrient concentrations on riverine freshwater inputs in two different periods. A linear relationship between the nutrient concentrations and salinity, precipitation and temperature was sought (linear regression in R software). Altogether the results indicate differences between time intervals and are able to establish a dependency of nutrient concentrations on riverine freshwater inputs, indicating pressure from nutrient loadings on the estuary.

In Chapter II pressure from nutrient flushing on the river basin is evaluated using land cover data as an indicator. The suitability of landscape metrics to describe the spatial variability of nitrate across the Mondego river basin is assessed using seasonal data from the years 2001 and 2006 (linear mixed model - R software). The results reveal that land cover patterns are weak descriptors of nitrate spatial variability in the Mondego river basin.

In Chapter III driving forces of water quality are characterized using processes of landscape change as proxy. CORINE Land cover maps at 1990, 2000 and 2006 are used to analyze changes among eight categories through an extended analysis of transition matrices. The magnitude of change and consistency of transitions reveal that the most relevant driving forces from land transitions include categories with low percentage of occupation but high potential effect on hydrological processes.

In Chapter IV, acknowledging that the accuracy of maps may influence our perception of the main driving forces acting in the system, the Intensity Analysis approach is applied to evaluate the suitability of the CORINE land cover maps as indicators of land change. Inconsistent transitions reveal the misclassification errors that could propagate to other land cover change applications, as in the assessment of hydrological processes.

Through the analysis of the results we conclude that there is still space for the development of indicators, namely policy effectiveness indicators; land cover data as indicator of pressure may reveal the effect of policies' implementation in the long-term; the magnitude and consistency of the processes of land change reveal additional information regarding the assessment of driving

forces from land transitions and that CORINE maps, though being high quality datasets, have accuracy limitations that are easily accessed through Intensity Analysis. Regarding nutrient concentrations, our study suggests that mitigation measures, preferably through policies at the European level, are needed to reduce the inputs of phosphate concentrations into the estuary.

Keywords: Integrated river basin management, Water Framework Directive, DPSIR, environmental indicators, water quality, anthropogenic pressures, land cover, coastal systems

Resumo

A Directiva-Quadro da Água reforça o papel dos usos do solo na avaliação da qualidade da água, mas a utilização de dados de ocupação do solo, no que respeita a águas de superfície, restringe-se à estimativa dos padrões de uso do solo para a identificação de pressões provenientes de fontes de poluição difusas.

O principal objetivo deste estudo é identificar especificidades da ocupação dos usos do solo na avaliação das mudanças de estado (Capítulo I), pressões (Capítulo I e II) e forças motrizes (Capítulo III e IV) da qualidade das águas de superfície.

Para o efeito, usamos dados da bacia hidrográfica do Rio Mondego, uma bacia costeira localizada em Portugal, onde as mudanças socioeconómicas têm sido observadas desde a década de 1990 e que resultaram na perda de representatividade do setor agrícola e ganho do setor da construção. Apesar das mudanças, a agricultura mantém-se como uma fonte de pressão sobre as águas de superfície que, associada a modificações hidromorfológicas, causaram um declínio da qualidade ecológica do estuário do Mondego. Em 1998, a implementação de medidas de mitigação permitiu a recuperação do sistema, mas a monitorização identificou elevadas concentrações de nutrientes que pediam uma avaliação da sua proveniência. Por esta razão, a bacia do Rio Mondego, emerge como um caso de estudo com as condições necessárias para identificar aplicações mais amplas de dados de ocupação do solo no âmbito da identificação de forças motrizes e pressões sobre a qualidade da água.

O Capítulo I estabelece a linha de base para este trabalho através da aplicação de indicadores de nutrientes e de substâncias-consumidoras-de-oxigénio para analisar a evolução das

concentrações de amónia, nitrato e fosfato no estuário do Mondego entre os períodos 2003-2007 e 2012-2013 (ANOVA no software R); para analisar as diferenças entre as concentrações médias anuais de nutrientes nos rios tributários em 2012-2013 e as concentrações médias anuais encontradas em rios europeus em três períodos diferentes (1992, 2000, 2012); e para avaliar as diferenças na variabilidade das concentrações de nutrientes no estuário e nos tributários em 2012-2013 (ANOVA em software R). Adicionalmente, o Capítulo I avalia a dependência das concentrações de nutrientes estuarinas das entradas de água doce provenientes dos tributários, em dois períodos distintos. Procurou-se a relação linear entre as concentrações de nutrientes e a salinidade, a precipitação e a temperatura (regressão linear no software R). Os resultados revelam diferenças entre os intervalos de tempo e mostram uma dependência das concentrações de nutrientes das entradas de água doce proveniente dos rios, o que indica pressão de cargas de nutrientes sobre o estuário.

No Capítulo II a pressão exercida pela descarga de nutrientes nos rios da bacia hidrográfica é avaliada utilizando dados de ocupação do solo. Analisa-se a adequação de métricas de paisagem para descrever a variabilidade espacial de nitratos em toda a bacia do rio Mondego usando dados sazonais de 2001 e 2006 (modelo linear misto - software R). Os resultados revelam que os padrões de ocupação do solo são descritores fracos da variabilidade espacial de nitratos na bacia do rio Mondego.

No Capítulo III caracterizam-se as forças motrizes da qualidade da água usando como proxy os processos de mudança da paisagem. Os mapas CORINE de 1990, 2000 e 2006 são utilizados para analisar as mudanças entre oito categorias de ocupação, através de uma análise alargada de matrizes de transição. A magnitude da mudança e a consistência das transições revelam que as forças motrizes mais relevantes incluem categorias com baixa percentagem de ocupação, mas elevado efeito potencial nos processos hidrológicos.

No Capítulo IV, reconhecendo que a precisão dos mapas pode influenciar a nossa percepção acerca das forças motrizes que atuam no sistema, aplica-se uma Análise de Intensidade para avaliar a adequação dos mapas CORINE como indicadores de mudanças de ocupação. Transições inconsistentes indicam erros de classificação que se poderiam propagar para outras aplicações que utilizam mapas de mudança de ocupação do solo.

Analisando os resultados concluímos que ainda há espaço para o desenvolvimento de indicadores, nomeadamente indicadores de eficácia-de-políticas; que os dados de ocupação do solo como indicadores de pressão podem revelar o efeito, a longo prazo, da implementação de políticas; que a magnitude e a consistência dos processos de mudança de ocupação do solo revelam informações adicionais relativamente à avaliação das forças motrizes de transições de ocupação e que os mapas CORINE, apesar de serem dados de elevada qualidade, têm limitações de precisão facilmente identificadas através de uma Análise de Intensidade. Em relação à concentração de nutrientes, o nosso estudo sugere que são necessárias medidas de mitigação para reduzir as entradas de concentrações de fosfato para o estuário, de preferência por meio de políticas a nível europeu.

Palavras-chave: Gestão integrada de bacia hidrográfica, Directiva-Quadro da Água, DPSIR, indicadores ambientais, qualidade da água, pressões antrópicas, ocupação do solo, sistemas costeiros

General introduction

"It's the process of science that has changed the world.
Science rules!"
Bill Nye (2013)

Integrated river basin management

River basins are geographical areas delineated by watersheds, within which the surface runoff will flow towards a specific location. River basins have played an important role in sustaining communities of people and other forms of life. The use of its resources – for food, water and shelter - has intensified across civilizations and currently many river basins are under pressure due to human activities (Freire et al. 2009). The impacts have been felt in water supply processes (Yang et al. 2015), water demand (Priess et al. 2011) and water quality (León-Muñoz et al. 2013). As a consequence, water and the services provided by aquatic systems have become degraded and scarce and competition for its use has increased, calling the need for river basin management (Moss 2004). As a natural system that links all water-related decisive factors, river basin has been adopted as the management unit to address several environmental problems associated to the hydrologic function and that determine water quantity and quality (EC 2000).

To be effective, river basin management needs to integrate all the complexity of the physical river system, the exchange between groundwater and surface water and the continuous interaction between environmental elements, relating the natural system to all human activities occurring within the hydrologic unit (Evers and Nyberg 2013). Management actions, in turn, need to establish a balance between the existing natural functions and the developed aspects of the system to fulfil all the expectations of the society for industrial use, recreation, nature management, and agricultural purposes. Integrated river basin management allows coordinating these multiple activities and provides a framework to resolve the conflicts (Moss 2004).

At the European level, the increasing demand for cleaner water natural systems as led to the implementation of the Water Framework Directive (WFD) (EC 2000). The WFD is a river basin management framework aiming to expand the scope of water protection to all waters; achieve “good status” for all waters by 2015; implement water management based on river basins; implement a "combined approach" of emission limit values and quality standard and get drinking water prices right. Regarding the objective of achieving the ecological protection of all surface waters, a general requirement for ecological protection, and a general minimum chemical standard, were introduced to cover all surface waters resulting in two elements: "good ecological status" and "good chemical status". Good ecological status is defined in terms of chemical characteristics, hydrological characteristics and biological community. The achievement of the target goal is controlled allowing a slight departure from the biological community which would be expected in conditions of minimal anthropogenic impact. Absolute standards for biological communities, and for the chemical and hydrological characteristics of the systems, were avoided due to ecological variability. To assess whether the broad objectives and targets of the EU water policy have been achieved and to indicate policy gaps a number of indicators and indices have been suggested. Some developed under the scope of the WFD and others yet under development (EEA 2014).

The water pollution problems from urban waste water and from agriculture have been addressed by European legislation since the 1980’s with the adoption of the Urban Waste Water Treatment Directive (EC 1991a) and the Nitrates Directive (EC 1991b). Currently several directives tackle particular pollution problems and the goal of the WFD is to coordinate the application of these to meet the objectives mentioned above. If the existing legislation is not sufficient to solve an identified problem and meet the objectives, then Member States must implement additional measures (EC 2000).

DPSIR framework

The DPSIR (Driver–Pressure-State-Impact-Response) framework is a structuring approach used to assess, manage and communicate the impact of environmental policy changes and associated problems. The EEA applied this conceptual framework to promote structural thinking within the analysis of environmental problems (EEA 1999). In the DPSIR framework social and economic developments *Drive* (D) changes that exert *Pressure* (P) on the environment. As a consequence, changes occur in the *State* (S) of the environment, which lead to *Impacts* (I) on human health, ecosystem functioning and the economy that will in turn require a societal *Response* (R). The response may feedback, directly or indirectly, to the drives, the pressures, the state or the impacts (EEA 2014) (Figure 1).

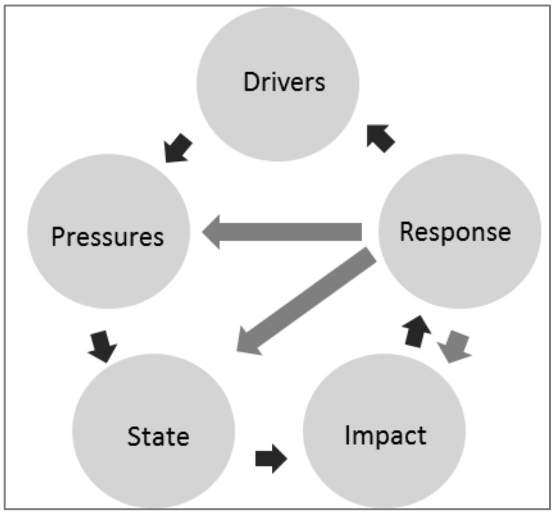


Figure 1. The DPSIR framework.

The DPSIR framework has widespread application besides the one to environmental indicators (offshore wind power (Elliott 2002), sustainability in coastal zones (Bidone and Lacerda 2004) and marine aggregates extraction (Atkins et al. 2011) and the framework continues to be used despite the criticisms (Bell 2012, Cooper 2013, Rekolainen et al. 2003). A comprehensive review on the critiques of DPSIR is provided by Gari et al. (2015). We highlight criticisms concerning the static behavior of indicators, not taking into account the changing dynamics of the system; the lack of capacity to capture trends; the synergistic relations between DPSIR categories ignored by the framework; the unclear boundary between state and impact categories and the lack of a precisely defined set of information categories (Gari et al. 2015).

In the context of the coastal and estuarine environment, the over-arching Drivers of social and economic development change refers to basic human needs such as shelter and food provided by sectors in industry (e.g. farming) and activities in the sector (e.g. cultivation). Each of these Drivers has the potential to create Pressures on the system (e.g. nutrient load). As a result, the State of the system (e.g. phytoplankton biomass) is changed and this may lead to actual or potential Impacts on society (e.g. decreasing capture rates of shellfish for local communities).

Identifying causal links between drivers, pressures and state change can be a complex task (Knights et al. 2013) as the relationships between activities and pressures and between pressures and state changes can emerge in the form of many-to-many relationships, i.e., for example, a single activity may potentially cause many pressures and a single pressure may be caused by several different activities. The uncertainty that may come from the high number of relationships that may be established hinders our capacity to find the most adequate long-term solutions to enhance the resilience of social-ecological systems, i.e, the capacity of systems to absorb stress and yet still maintain ‘‘function” (Folke 2006).

Within a causal link framework, the adoption and development of indicators is essential to determine the level of pressure, and changes in state and in the impact (e.g. Aubry and Elliott 2006). Environmental indicators summarize, generally quantitatively, complex environmental phenomena enabling the communication of environmental state. The development of indicators was initially developed by the Organization for Economic Co-operation and Development (OECD) (OECD 2003) in the early 1990s and further developed by other entities. Examples are the United Nations Statistical Division (UNSD), the United Nations Economic Commission for Europe (UNECE) (UNECE 2007), the Eurostat and the Joint Research Centre (JRC) and the European Environment Agency (EEA), both from the European Commission (EC).

Environmental indicators are commonly used to support policy development, priority-setting, progress and effectiveness of policy responses and play an important role when reference values, thresholds and/or policy targets have been established, as they allow to measure progress and performance against them. In assessments, indicators provide information on environmental state, including trends and progress over time (EEA 2014). Though the indicators selected ought to fit the assessment purposes, the selection is frequently a subjective process and may depend on data availability and researcher preference. Conceptual frameworks, such as the DPSIR framework, have helped in the process of selection of indicators for assessments, but also to ensure that all aspects of a specific issue are covered and also to identify gaps for which indicators are not available (Smith et al. 2014).

The role of land cover

The WFD (EC 2000) has created opportunities to overcome problems related to lack of communication between water management, which struggles to maintain water quality and quantity at adequate levels for human use and ecological sustainability, and other policy fields, such as agriculture, forestry, land use planning, hydro-electric power and economic development. By introducing instruments that broaden the spatial dimension of water management – river basin units, combined pollution control and necessity to consider hydro-morphological aspects of the basin - the more territorially integrated approach of the WFD has created obligations to consider the role of land use in causing water stress (EC 2000). Specifically, the WFD requires a) the estimation of land use patterns for the identification of pressures on surface waters from diffuse source pollution; b) estimation of land use in the catchments from which the groundwater body receives its recharge, and information on pollutant inputs and anthropogenic alterations to the recharge characteristics such as rainwater and run-off diversion through land sealing, artificial recharge, damming or drainage (EC 2000).

In this context, land cover data has become a key instrument to understand the impacts of human use of the landscape and of natural phenomena, such as impacts from land impermeability (Wu et al. 2013), nutrient runoff (Morrison and Kolden 2015, Tang et al. 2011), floods and storm surges and sea level rise (Ferreira et al. 2014), through the assessment of urban growth (Kumar et al. 2013), agricultural land changes (Hutchins 2012), deforestation (Öztürk et al. 2013) and wetland losses (Records et al. 2014). Land cover maps - which indicate the proportion of a geographical area covered by agriculture, forests and other natural areas, artificial surfaces and water types - and land cover change maps are useful to evaluate progress of management decisions and to gain insight into possible effects of current or prospective management decisions before implementation. Though conceptually different from land use, which documents how society is using the landscape – from development, conservation or mixed uses -, both terms, land cover and land use, are frequently applied interchangeably.

Regarding water quality, land cover, land cover change and land cover patterns represent an implicit piece of the puzzle of the several factors that may cause water degradation: point- and diffuse-source pollution, sediment yields, runoff, and infiltration capacity.

The Mondego river basin

The Mondego river basin was used as a case study and the results obtained ought to contribute to the systematic synthesis of sub-global research on the linkage between land system science and estuarine and coastal science. The basin is located in the central region of Portugal, Europe and has an approximate area of 6658 square kilometers and a NE–SW orientation. Its functional structure ranges from mountainous areas (Serra da Estrela, Lousã and Caramulo) to a large alluvial plain discharging into the Atlantic Ocean. The main geomorphologic characteristics of the Mondego river basin are available on table 1.

Table 1.
Main geomorphologic characteristics of the Mondego river basin.

Perimeter (km)		757.39
Area (km2)		6 658.58
Altitude (m)	Average	381.77
	Maximum	1 992.72
	Minimum	0.00
Average slope		16.99

The river basin is occupied mainly by agricultural (32%) and forest (64%) areas that are distributed throughout the basin, whereas urban (2.34%) and industrial (0.68%) areas are located mainly on the coastal strip (Teixeira et al. 2014). Major urban centres within the region are the cities of Coimbra (population 139 151), Leiria (126 348), Viseu (98 778) and Figueira da Foz (61 505); two other cities have populations higher than 50 000 (INE 2014). In total, the basin encompasses 36 municipalities with an estimated population of 165 inhabitants per square kilometre, (INE 2014). The number of inhabitants showed only a slight negative net change (-0.04%) between the years 1992 and 2011 (INE 2014) (Figure 2).

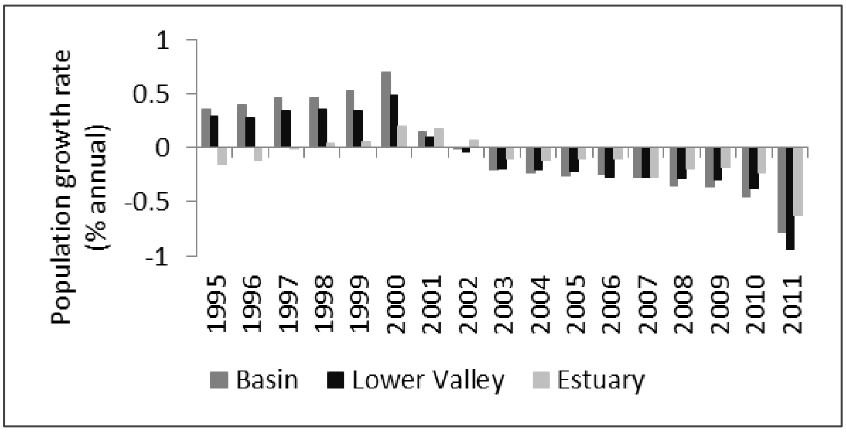


Figure 2. Population annual growth rate in the Mondego river basin,

Lower Mondego and in the estuary region, between 1995 and 2011.

Strong pressures in the basin are caused by the primary economic sector. Specifically, agriculture plays a major role in the basin due to highly productive rice fields in the Lower Mondego region. The agricultural sector has nonetheless suffered some changes since the 1990s. In the river basin, the utilized agricultural area decreased from approximately 17% to 8% of the total basin area and similar trends were observed in the Lower Mondego and in the estuary region, downstream the basin. The observed trend is mainly a result of the decline of farms smaller than 3 ha (INE 2014).

Other pressures are exerted by the secondary and tertiary economic sectors, which are well represented in the basin with approximately 57% of the employed population working in the tertiary sector and 37% in the secondary sector of activity (INE 2014). Within the secondary economic sector, we highlight activities related to construction. Between 1995 and 2001 the number of buildings increased in the basin, slowly decreasing afterwards until 2010. The estuary region, where the municipality of Figueira da Foz is located, shows the highest number of buildings per hectare (Figure 3).

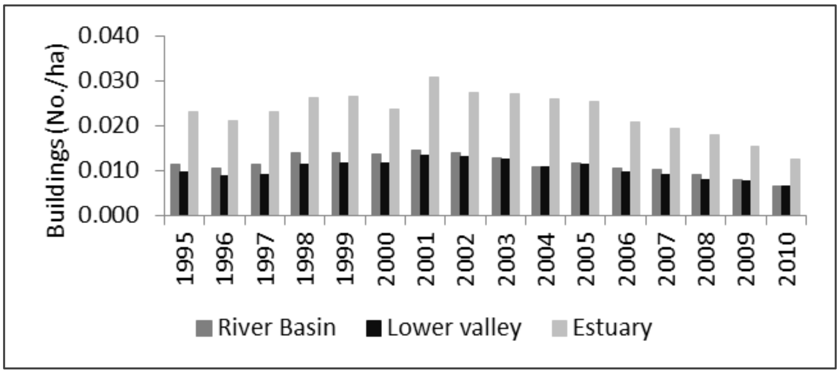


Figure 3. Number of buildings per hectare in the Mondego river basin, Lower Mondego and estuary region, between 1995 and 2010.

The Mondego estuary

The Mondego Estuary is a warm-temperate, polyhaline, intertidal system with about 7 km long and an area of approximately 1072ha. The estuary consists of two different arms: the northern, with depths between 4 to 8 m during high tide, and the southern arm, with depths between 2 to 4 m. An alluvium-formed island (Murraceira Island) separates the two arms. The estuary receives water from the Mondego River, entering the estuary through the Remolha water body, and from tributaries joining it at both the north and the south banks. Water circulation in the south arm was, until 2006, mostly dependent on tides and on the fresh water input from the Pranto River, which is artificially controlled by a sluice (Marques et al. 2013) and regulated according to water requirements in rice fields from the Pranto Valley.

The Mondego Estuary has been under environmental stress by eutrophication processes (Baeta et al. 2011, Dolbeth et al. 2003, Marques et al. 2003, Sousa et al. 2008). In the 1990s the communication between the two arms became totally interrupted in the upstream area, and the water circulation in the south arm became dependent on tides and in the Pranto river. As a consequence, eutrophication became a problem in the south arm and macroalgal blooms, especially of *Ulva spp*., occurred repeatedly (Dolbeth et al. 2007). This species may be found in the estuary throughout the whole year, but its growing season starts in late winter and its maximal biomass usually occurs in spring. A smaller biomass peak may also occur in early autumn (Martins and Marques 2002).

A comprehensive study on the Mondego estuary environmental quality has been carried out for more than 25 years, comprising regular monitoring of the system with regard to water quality, biological communities, hydraulics and sediment dynamics. Studies allied to hydrodynamic modelling, concluded that environmental degradation observed in the South arm had been mostly related with the increase in water residence time after the interruption of the communication between the two arms (Martins et al. 2001, Kenov et al. 2012). In 1998, to reduce the eutrophication symptoms, the upstream connection between the two arms of the estuary was partially re-established ($1m^2$), as a mean to improve the hydraulic regime and to decrease the water residence time; and the freshwater inputs from the Pranto River were reduced through diversion of Pranto freshwater to the northern arm by another sluice located further upstream. In the Spring of 2006, a large-scale intervention was implemented and the connection between the two arms was fully re-established to allow for a more efficient nutrients flushing, reduce the water residence time in the south arm and prevent pollution related problems in this section of the estuary (Veríssimo et al. 2012). Studies evaluating the progress of eutrophication symptoms suggested that after the implementation of the mitigation measures, the nutrient balance and status of this coastal system depended both on biogeochemical mineralization processes (Coelho et al. 2004, Otero et al. 2013) and on additional external point and diffuse sources within the south arm or through the Mondego river north arm (Lillebø et al. 2005).

General aims and thesis outline

Land patterns and changes are among the several causes of water degradation and thus land cover data has become a key instrument to understand the impacts of human use of the landscape on water quality. The WFD implicitly and explicitly, enhances the role of land use in determining water quality of all waters (EC 2000), but the use of land cover data, regarding surface waters, is restricted to the estimation of land use patterns for the identification of pressures on surface waters from diffuse source pollution. The main goal of this study was thus to uncover specificities of land cover in the assessment of drivers, pressures and state changes of water quality of surface waters.

Previous research has identified land use as a source of pressure within our case study, the Mondego river basin (Pinto et al. 2013), where 32 % of land is occupied by agriculture, but a clear link between land cover occupation and water quality has not been explored at the watershed level; neither the potential of land cover changes as drivers of water quality.

Our study was driven by the following specific questions:

- After the successful mitigation measures implemented in the estuarine system to overcome its environmental problems, do the tributary rivers remain a source of pressure?
- Is it possible to establish a relationship between the landscape descriptors of the Mondego river basin and the stream water quality, at the watershed scale?

- Recognizing that land cover is a source of stress with potential effects on the condition of water bodies, what are the main processes of land change that potentially drove the environmental state of the system, in recent years?

- At what extent the Error in the land cover maps may be interfering with our capacity of identify drivers of water quality from land processes of change?

The four chapters of the present PhD thesis address the abovementioned research questions. For each one a scientific paper was produced, which constitute the chapters of this thesis, and therefore content overlapping was unavoidable. All papers are currently published or submitted for publication. A general discussion, built on the findings of the four papers, is available at the end of the document which served to highlight gaps that ought to be addressed in future research as a contribution for the development of coastal and estuarine management actions and policies.

Publications

This thesis is based on the following published, submitted or ready for submission publications:

Chapter I

Teixeira Z, N Leite N, AC Garcia, JC Marques. Changes in chemical parameters of a southern European temperate estuarine ecosystem: evidence of dependency of water quality on freshwater inputs. (to be submitted)

Chapter II

Teixeira Z, JC Marques. Relating landscape to stream nitrate concentrations in a coastal eastern-Atlantic watershed (Portugal). Ecological Indicators (Submitted). Editorial state: Accept pending revisions.

Chapter III

Teixeira Z, H Teixeira, JC Marques. 2014. Systematic processes of land use/land cover change to identify relevant driving forces: Implications on water quality. Science of Total Environment. 470-471:1320-1335. doi:10.1016/j.scitotenv.2013.10.098

Chapter IV

Teixeira Z, JC Marques, RG Pontius Jr.. Evidence for deviations from uniform changes in estuarine and adjacent coastal areas in Portugal illustrated by CORINE maps: an Intensity Analysis approach. (Submitted).

Chapter I

Changes in chemical parameters of a southern European temperate estuarine ecosystem: evidence of dependency of water quality on freshwater inputs.

Abstract

The European Freshwater Quality Report, published in 2015, indicates an overall decline in the concentrations of nutrients and oxygen-consuming-substances influencing the physico-chemical characteristics of European rivers. Despite such improvements, more than a half of the water bodies, namely estuarine, are reported to be in less than good ecological status (or potential), and eutrophication remains a challenge. Our goal is to evaluate the progress of the concentrations of nitrate, phosphate and ammonia in a coastal system in Portugal and determine whether the evaluated chemical parameters depend on freshwater inputs from tributary rivers. To evaluate progress, assessment indicators from the core set of the European Environment Agency were applied. In the estuarine system, the concentrations of ammonia, nitrate and phosphate in a post-mitigation period (2003-2007) were compared to a recent time interval (2012-2013), whereas in the tributary rivers, the concentrations of the same three parameters were evaluated in relation to the average annual mean concentrations in European rivers. To evaluate pressure from nutrient loadings, a pressure indicator, relating salinity, precipitation and temperature to estuarine concentrations of ammonia, nitrate and phosphate, was applied to determine the dependency on freshwater inputs. Results have shown that in the estuary, ammonia significantly decreased, phosphate significantly increased and nitrate did not show significant differences. In the tributary rivers, the annual mean concentrations of phosphate are higher than the average annual mean concentrations in European rivers in 1992, concentrations of ammonia are higher than the average in European rivers in 2000 and concentrations of nitrate are lower than the average in 2012. No significant differences were found between the estuarine concentrations and those found in the upstream freshwater sections. Results have also demonstrated that the estuarine concentrations of phosphate and nitrate depended on riverine freshwater inputs in the post-mitigation period (2003-2007), but not recently (2012-2013). This suggests that other external sources are currently contributing to the increase of phosphate concentrations in the estuary. In conclusion, evaluating progress of the physico-chemical parameters remains important even when previous mitigation measures have been implemented and the system has overcome its major problems. Evaluating the dependency of chemical parameters' concentrations on freshwater inputs through comparison to other physical-chemical characteristics provides a suitable pressure indicator and a basis for development of further mitigation measures.

Keywords: Freshwater, coastal systems, physico-chemical characteristics, nutrients, nitrate, phosphate, ammonia, Portugal, water quality, indicators

Introduction

The European Environment Agency (EEA) maintains indicators of water quality to answer policy questions and support all phases of environmental policy making (EEA 2014). The most recent reports on the status and pressures of freshwater and transitional waters were launched this year, 2015, relating water indicator development with relevant directives. Among the state indicators assessed by the EEA are the nutrients in freshwater (EEA 2015a) and the oxygen-consuming-substances (i.e., BOD and ammonia) in rivers (EEA 2015b), whose information is provided at the European level. The reports indicate a decline in the average nutrient concentration in European rivers between 1992 and 2012, as well as of oxygen-consuming-substances (EEA 2015c), ascribing the observed decrease to changes in pressures such as land use and waste water treatment.

To tackle the impacts caused by pressures such as waste water treatment plants, industrial effluents and agricultural runoff and improve the environmental quality of surface waters, namely the reduction of eutrophication and nutrient concentrations, several directives have been implemented at the European level. Measures applied under the scope of these directives, and further national laws, have managed to reduce the concentrations of ammonia, nitrates and phosphate in surface waters in the last 20 years. From 1992 to 2012, the average ammonia concentration in European rivers decreased by 0.0116 milligrams per litre of nitrogen (mg N/L) per year (EEA 2015c), mainly as a result of the implementation of the Urban Waste Water Treatment Directive (EC 1991a) and the Integrated Pollution Prevention and Control Directive (EC 1996). During the same interval, the average nitrate concentration in European rivers declined by 0.03 mg N/L per year and the average phosphate concentration has decreased by 0.003 mg N/L per year (EEA 2015b). The reduction of river nitrate concentrations reflects the success of the EU Nitrate Directive (EC 1991b) to reduce nitrogen pollution from agriculture, whereas the decrease of phosphate is mainly a result of the implementation of legislative measures to reduce the emissions of phosphorus, such as the Urban Waste Water Treatment Directive (EC 1991a). Also, they have all benefited from the implementation of the Water Framework Directive (EC 2000), which required the achievement of good ecological status or good ecological potential of rivers by 2015.

The reports indicate an improvement of the water quality of rivers within the Member States, but countering eutrophication remains a challenge (EEA 2015a). In this context, regional and local monitoring plans play a key role in the assessment of changes in the water quality of coastal systems, frequently led by nutrients' release from land surrounding areas, entering in the transitional systems through freshwater inputs (Martins et al. 2001).

The main goals of our study are to evaluate progress in the concentrations of nitrate, phosphate and ammonia in a southern European temperate estuarine ecosystem, the Mondego estuary, in the Atlantic coast of Portugal, and in its tributaries; and to determine the dependency of nutrient concentrations on freshwater inputs, as an indicator of pressure from nutrient loadings into the estuary.

Eutrophication in the Mondego estuary

The Mondego estuary is a small, warm temperate, polyhaline, intertidal system located in the centre of Portugal, Europe. It consists of two arms: the northern, with depths between 4 to 8 m during high tide, and the southern arm, with depths between 2 to 4 m. The estuary receives water from the Mondego River, entering the estuary through the Remolha water body, and from tributaries joining it at both the north and the south banks (Figure I.1).

The Mondego estuary has been under environmental stress by eutrophication processes (Baeta et al. 2011, Dolbeth et al. 2003, Marques et al. 2003, Sousa et al. 2008). In the 1990s an opportunistic macroalgae, *Ulva spp*., replaced the seagrass *Zostera noltii* in the southern arm of the estuary and the area was considered a "Potential Problem Area" regarding to eutrophication (Lillebø et al. 2007). *Ulva* individuals are present in the estuary throughout the all year, but its growing season starts in late winter and its maximal biomass usually occurs in spring. Autumn may also present a smaller biomass peak (Martins et al. 2001).

Several mitigation measures were implemented in the estuary back in 1998: a) enlargement of upstream connection between the two arms of the estuary, to improve the hydraulic regime and to decrease the water residence time; b) reduction of freshwater inputs from the Pranto River by reducing the openings of its sluice and by diversion of Pranto freshwater to the northern arm by another sluice located further upstream.

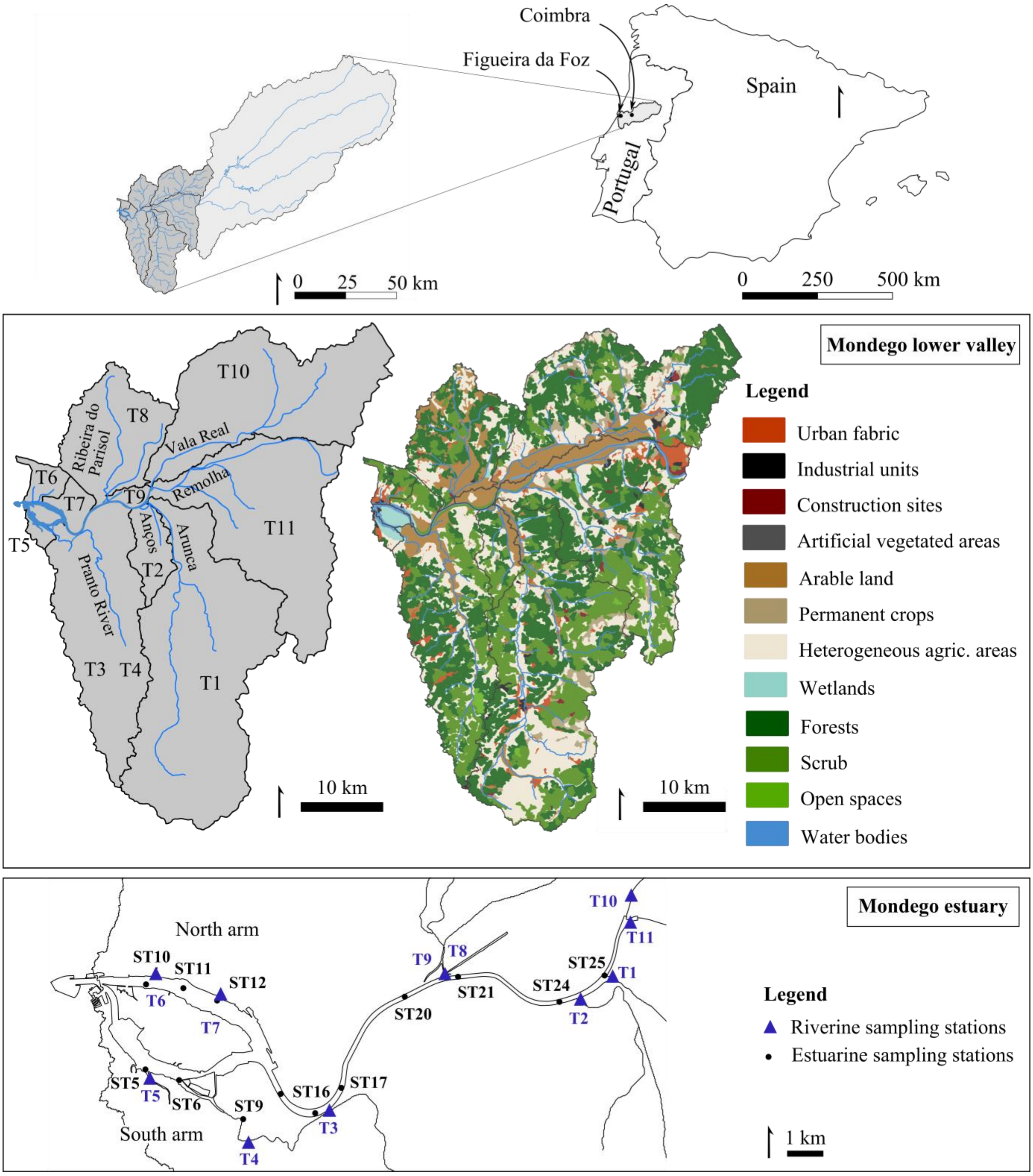


Figure I.1.
Map in the upper right shows the location of the study site in the centre of Portugal. The study site is the Mondego estuary. Map in the upper left shows the Mondego river basin (light grey) and the Mondego lower valley region (dark grey). Map in the middle left shows the sub-basins that drain to each tributary of the Mondego estuary. Map in the middle right shows the land cover in the Mondego lower valley region based on the 2006 inventory of the CORINE Land Cover project. The lower map shows the Mondego estuary and the location of the riverine and estuarine sampling stations.

Prior to the implementation of the mitigation measures, the water circulation in the southern arm was mostly dependent on the tides and on freshwater discharges from the Pranto River (Flindt et al. 1997). The flow of this small tributary was controlled by a sluice, which was regulated according to water requirements from upstream rice fields, and represented an important input of organic and inorganic matter to the estuary (Flindt et al. 1997), mainly due to fertilisers used in upstream agricultural land.

Previous studies assessing eutrophication of the Mondego estuary after the 1998 interventions suggested that the nutrient balance and status of this coastal system depended both on biogeochemical mineralization processes (Coelho et al. 2004, Otero et al. 2013) and on additional external point and diffuse sources within the south arm or through the Mondego river north arm (Lillebø et al. 2005). In either case, the studies suggested that the concentrations of both phosphate ($P\text{-}PO_4$) and nitrate ($N\text{-}NO_3$) depended on freshwater inputs (Lillebø et al. 2005). Thus, a comprehensive assessment of the new eutrophic conditions after interventions, the last one in May 2006, should take into account riverine nutrient loads. However, as far as we know, until now, no data has been collected at the mouth of all tributaries of the Mondego estuary. River input data has relied on national databases, but the service intermittence and the low number of monitoring stations with relevant data frequently hinders the ability to use such environmental information.

Pressures in the Mondego river basin

Despite the interventions in the Mondego estuary the system remains under pressure by the release of nutrients from agriculture, from urban drainage and from domestic and industrial waste waters. The Mondego estuary receives agricultural runoff from 15 000 ha of upstream crop areas, mainly rice and corn fields (Teixeira et al. 2014). Agriculture is one of the main sources of diffuse pollution in the Mondego, but also of point source pollution through the two sluices, which open according to farmers' water requirements. Urban and industrial areas occupy 3% of the Mondego river basin, but two of the most populated municipalities, Coimbra and Figueira da Foz, have grown along the river margins and play an important role within the Mondego river dynamics as urban and industrial areas reduce soil permeability, causing the increase of runoff and promoting diffuse pollution to surface water, and are point sources of domestic and industrial waste waters. Within the Mondego river basin, two waste water treatment plants operate in the Figueira da Foz municipality, providing only secondary water treatment and without capacity to treat industrial waste water (PGRH4 2012). The lack of industrial waste water treatment is yet to be addressed by local authorities (PGRH4 2012) as well as certain areas regarding domestic waste water. The Murraceira Island, in between the two arms of the estuary is one such example.

Methodology

Sampling and laboratory procedures

Surface water samples were collected, on a monthly basis, between July 2012 and June 2013. Sampling was performed at the mouth of eleven tributaries of the Mondego estuary and in twelve estuarine subtidal sampling stations (Figure I.1 and Table I.1). A total of 276 samples were collected. Water salinity and temperature (^{o}C) were measured *in situ*. Water samples (approximately 1.5L) were collected for analysis of nitrate, ammonia and phosphate. Samples were immediately filtered (Whatman GF/F glass–microfibre filter. Pore size 0.45 µm) and stored frozen at −18 ^{o}C until analysis, following standard methods described in Limnologisk Metodik (1992) for ammonia (N-NH_3, mg L^{-1}) and phosphate (P-PO_4, mg L^{-1}) and in Strickland and Parsons (1972) for nitrate (N-NO_3, mg L^{-1}).

Data analysis

Normality and homogeneity were tested by the Shapiro-Wilk and Levene tests respectively. One-way ANOVA, with Welch's correction when suitable, and the post-hoc Games-Howell method (Peters 2015) based on Welch's correction, appropriate for samples with unequal variances and unequal sample sizes, were used to a) establish the existence of spatial and seasonal variations in the mean concentrations of nitrate, phosphate and ammonia in river stations and in the estuary; b) establish the existence of differences among two time intervals (2003-2007 and 2012-2013), in the estuarine annual and seasonal mean concentrations of nitrate, phosphate and ammonia; c) and to establish the existence of differences between the annual and seasonal mean concentrations of nitrate, phosphate and ammonia in river stations and the adjacent estuarine stations. Linear regression was applied to assess the significance of linear relationships between variables. Boxplots were used to represent profiles of the variables along the sampling stations. All statistical analyses were performed using R software.

Table I.1.
Location and description of freshwater sampling sites.

Station code	Entrance	Water body hydrologic code	River Name and/or freshwater source	Land use description	Adjacent estuary station(s)	
					Upstream	Downstream
T1	South	PT04MON0680	Arunca river	Rice fields surrounded by natural vegetation	--	ST25
T2	South	PT04MON0683	Vala de Anços	Rice fields surrounded by natural vegetation	ST25	ST24
T3	South	PT04MON0691	Pranto river	Rice fields	ST17	ST16
T4	South	PT04MON0691	Pranto river	Rice fields	--	ST9
T5	South	PT04MON0682	Armazéns channel	Salines	ST6	ST5
T6	North	PT04MON0681	Small tibutary adjacent to the Figueira da Foz harbour	Industrial and urban areas, as well as non-irrigated arable land	ST11	ST10
T7	North	--	WWTP	Water comes directly from the WWTP, which tretas both domestic and industrial water	ST12	ST11
T8	North	PT04MON0677	Foja water pumping station. East side	Mineral extraction sites and rice fields surrounded by woodland-shrub and mixed forest	ST21	ST20
T9	North	PT04MON0677	Foja water pumping station. West side	Mineral extraction sites and rice fields surrounded by woodland-shrub and mixed forest	ST21	ST20
T10	Upstream	PT04MON0674	Vala Real	Discontinuous urban areas, rice fields and other annual crops associated with permanent crops, surrounded by broad-leaved forest	--	ST25
T11	Upstream	PT04MON0675	Vala da Remolha	Land principally occupied by agriculture - rice fields, olive groves and other - with significant areas of broad-leaved forest and natural grassland	--	ST25

Results

Mondego estuary quality

Several mitigation measures during the late 1990s and the early 2000s were implemented in the Mondego estuary to cope with eutrophication problems in the south arm of the estuary, assigned to the excess of nutrients. During the 2003-2007 time interval, after the efforts to reduce the concentrations of nutrients in the south arm, the maximum seasonal mean concentrations observed were 0.49 mg/l for ammonia; 2.47 mg/l for nitrate and 0.16 mg/l for phosphate. The highest concentrations of ammonia were observed in ST9 (Figure I.2), which is the closest estuary station to the Pranto River.

The concentrations of ammonia in ST9 exhibited significant differences from all other estuarine stations. In fact, the estuary showed a clear spatial pattern regarding the concentrations of ammonia, but also the concentrations of nitrate and phosphate. Statistically significant differences between stations show evidence of increasing concentrations from downstream to upstream areas (Table I.2), both in the north and in the south arm (Figure I.2).

Table I.2.
Levene's test and one-way ANOVA to analyse differences among estuarine stations for ammonia, nitrate and phosphate concentrations

	Levene's test		One-way ANOVA	
	F	P(F)	F	P(F)
Ammonia	1.6417	0.094	22.826	< 0.001
Nitrate	0.8046	0.635	31.603	< 0.001
Phosphate	0.974	0.473	31.912	< 0.001

One of the intervention measures was the reestablishment of the connection between the two arms of the estuary, reducing the water residence time in the south arm. Results show that after mitigation, during the 2003-2007 time interval, the seasonal and inter-annual salinity of the estuary had little influence from precipitation. Both parameters show significant ($P<0.001$) negative linear relationship (Figure I.3), but with low correlation (R^2=0.22, N= 55). The linear relation between salinity and precipitation appears to disappear in the time interval 2012-2013 ($P=0.06$), but the low number of observations hinders our capacity of interpretation (Figure I.3).

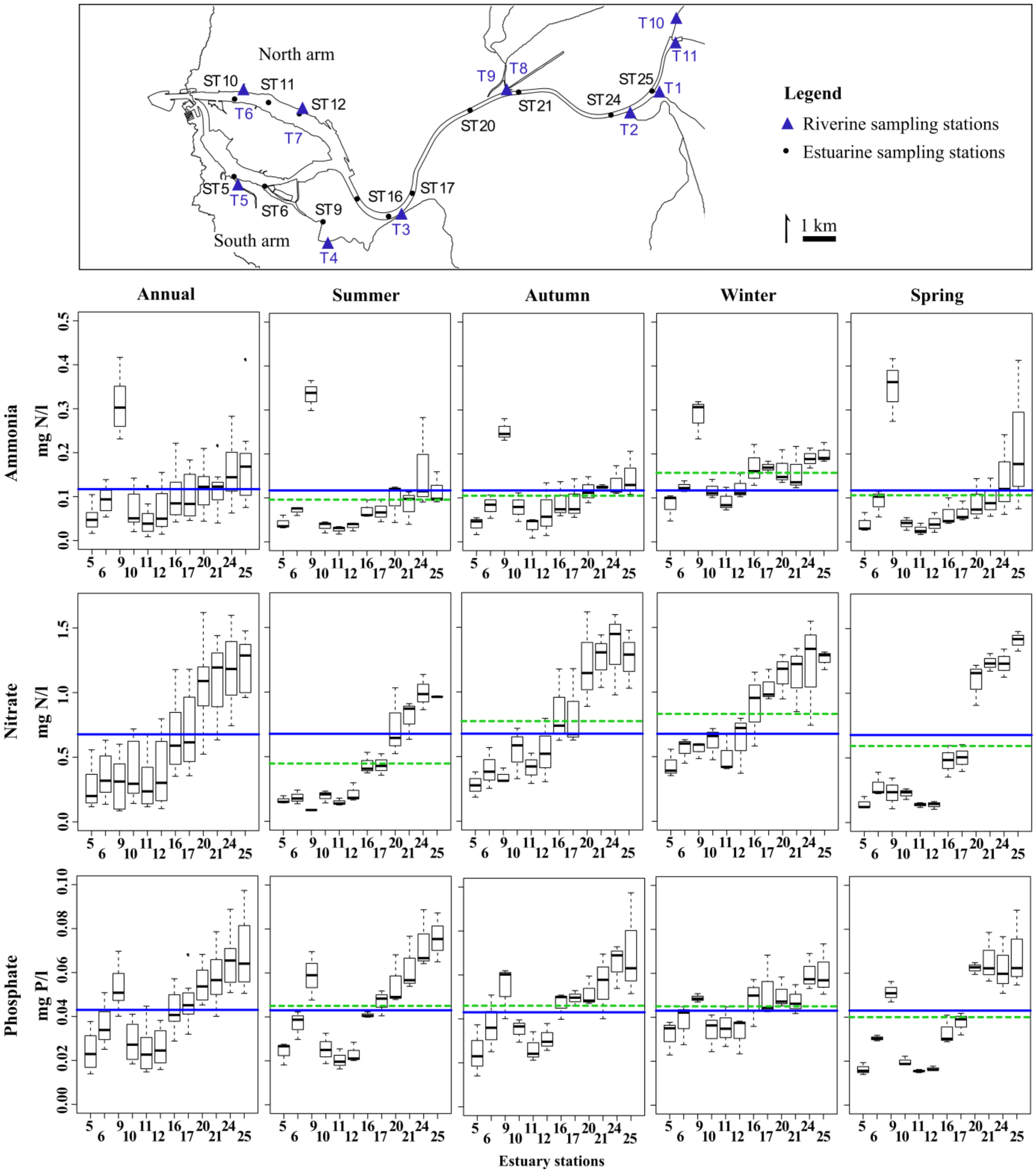


Figure I.2.

Boxplots of average annual and seasonal mean concentrations (mg/l) of ammonia, nitrate and phosphate sampled in the Mondego estuary subtidal stations, between 2003 and 2007. Sampling stations are ordered from downstream to upstream stations. Blue line is the average annual mean concentration of the estuary for the 2003-2007 time interval. Green dashed lines are the average seasonal mean concentrations for estuarine stations for the 2003-2007 time interval. The upper map shows the location of the sampling stations in the Mondego estuary. T-riverine stations. ST-estuarine stations.

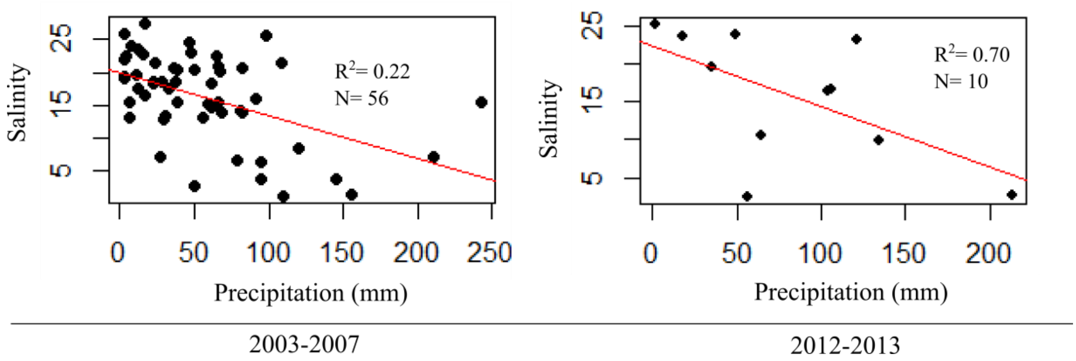


Figure I.3.
Relation between precipitation and salinity in the 2003-2007 time interval and the 2012-2013 time interval.

Likewise, seasonal variability of salinity showed significant (P<0.001) negative linear relation and low correlation (R^2=0. 016, N=144) with ammonia in the 2003-2007 time interval (Figure I.4), which disappears in the 2012-2013 time interval (Figure I.5).

On the contrary, seasonal variability of salinity exhibited highly significant (P<0.001) negative linear relationship with nitrate (R^2=0.86, N= 144) and phosphate (R^2=0.66, N=144) with high correlations in the 2003-2007 time interval (Figure I.4). In 2012-2013 the linear relation of salinity with nitrate disappears and becomes weak for phosphate (R^2=0.05, N=144) (Figure I.5). Seasonal variability of precipitation, however, did not show significant linear relation with phosphate and ammonia during both periods of analysis, nor with nitrate in the 2012-2013 period, but showed significant (P<0.01) positive linear relation, though with very low correlation, with nitrate (R^2=0.09, N=55) in 2003-2007 time interval.

Both phosphate and nitrate showed significant (P<0.01) linear relationships with water temperature in the 2003-2007 time interval, but only phosphate showed this relation in the 2012-2013 period. Positive relation with low correlation for phosphate (R^2=0.05, N=144), and negative relation with low correlation for nitrate (R^2=0.03, N=144) were observed in the 2003-2007 time interval (Figure I.4). Negative relation, again with very low correlation, for phosphate was observed more recently (R^2=0.05, N=144) (Figure I.5).

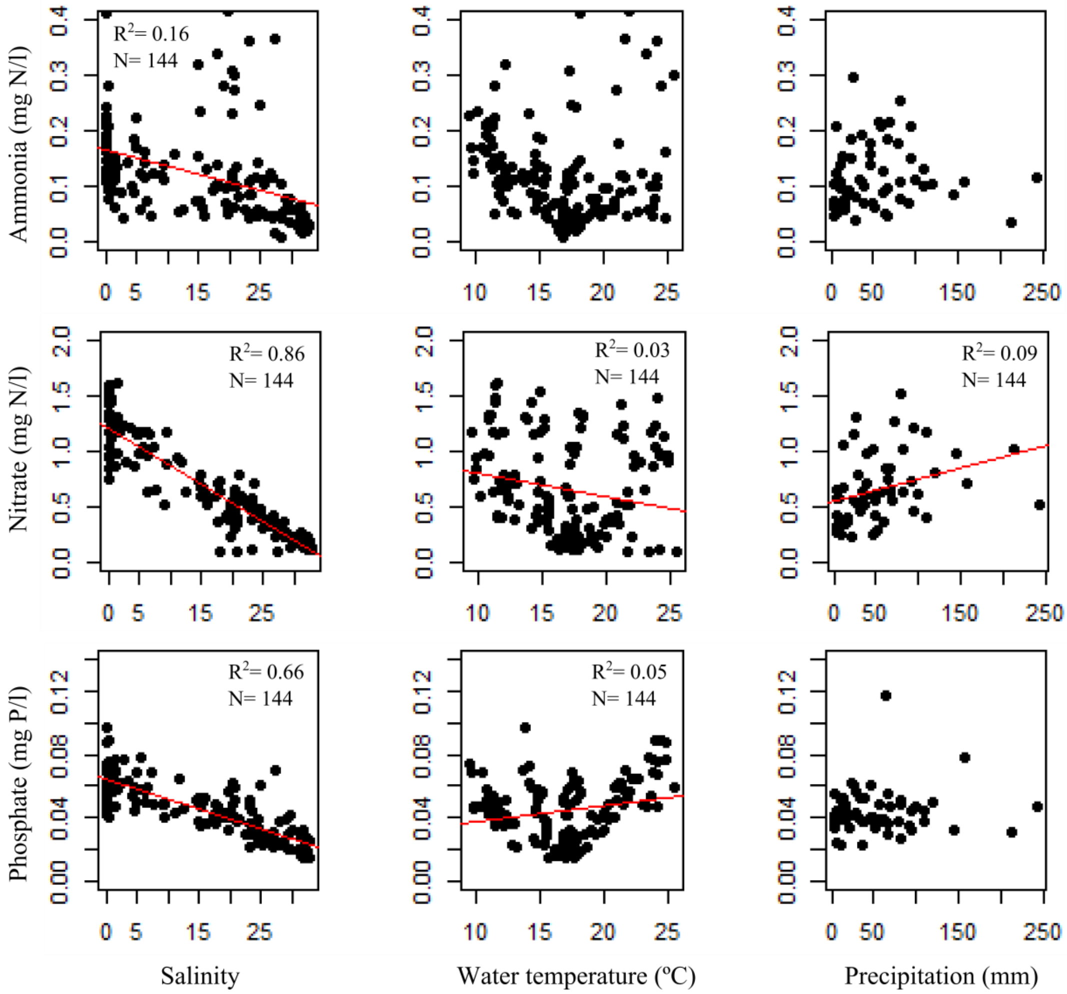


Figure I.4.

Relation between salinity, water temperature and precipitation with the concentrations of ammonia, nitrate and phosphate of the Mondego estuary during the 2003-2007 time interval. Trend line shown only for significant relationships.

Changes in the Mondego estuary quality

We analysed the differences on the estuarine mean concentrations of ammonia, nitrate and phosphate among two time intervals: 2003-2007 and 2012-2013 (Table I.3). Significant differences were found between the two periods for the annual mean concentrations of ammonia and phosphate (Table I.4). Ammonia shows lower annual mean concentrations (red dashed line in figure I.6). Phosphate shows higher annual mean concentrations (red dashed line in figure I.6). All three parameters show significant differences between periods among winter mean concentrations and none of the three show differences among summer concentrations. Nitrate and phosphate show significant differences among spring concentrations (Table I.4), but the number of samples is very low and therefore caution must be taken when analysing spring concentrations (Figure I.6).

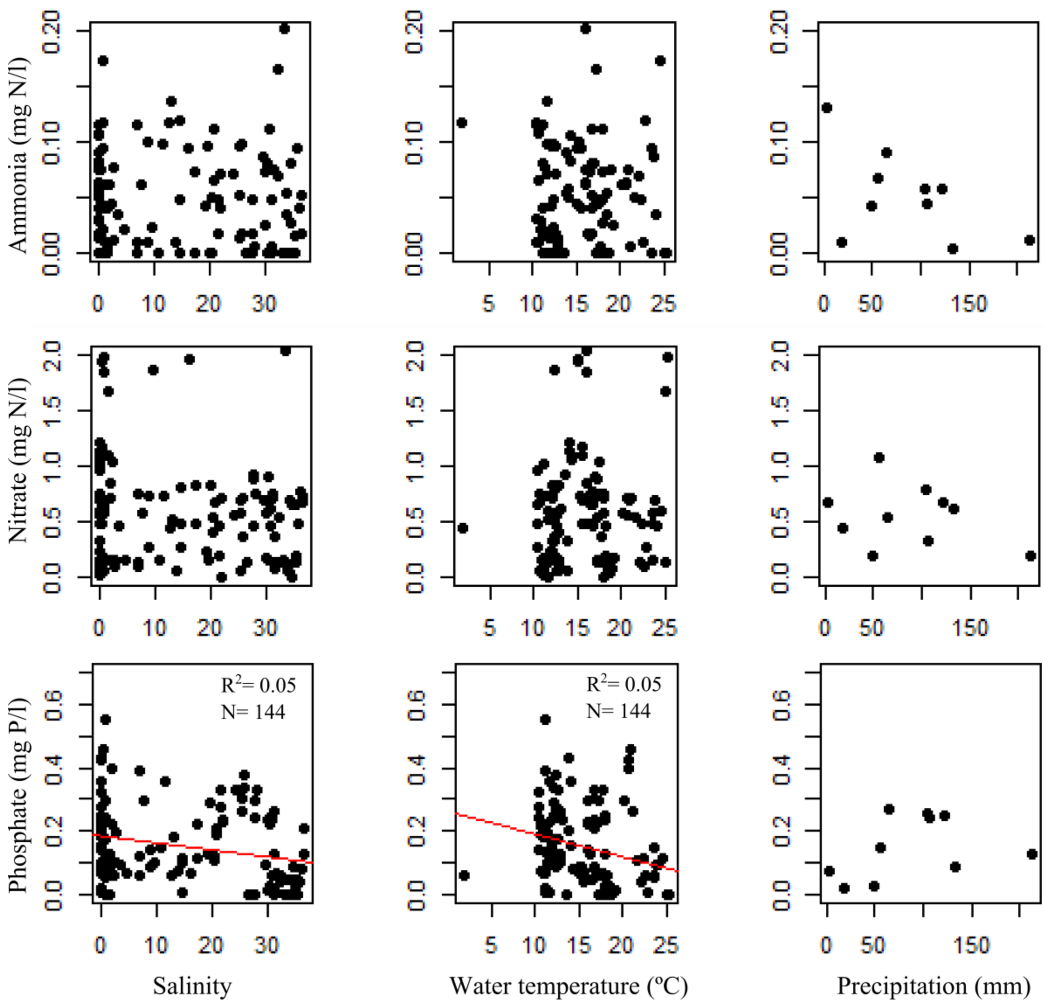


Figure I.5.
Relation between salinity, water temperature and precipitation with the concentrations of ammonia, nitrate and phosphate of the Mondego estuary for the 2012-2013 period. Trend line shown only for significant relationships.

Table I.3.
Physico-chemical parameters of estuarine (ST) stations of the Mondego estuary in the 2003-2007 time interval and in the 2012-2013 time interval and of riverine (T) stations in the 2012-2013 time interval..

		Temperature (^{o}C)		Salinity		Ammonia ($N-NH_3$) (mg/l)		Nitrate ($N-NO_3$) (mg/l)		Phosphate ($P-PO_4$) (mg/l)	
Interval /System	Stations	Mean	SD	Mean	SD	Mean	SD	Mean	SD	Mean	SD
2003-2007 Estuary	ST5	15.71	2.40	29.19	6.71	0.05	0.05	0.26	0.20	0.02	0.01
	ST6	16.38	3.31	25.64	6.73	0.08	0.06	0.35	0.25	0.04	0.02
	ST9	18.23	5.31	20.37	8.46	0.10	0.08	0.30	0.28	0.04	0.02
	ST10	15.64	2.49	26.69	10.67	0.10	0.08	0.38	0.39	0.04	0.02
	ST11	15.49	2.32	27.76	10.73	0.11	0.09	0.30	0.34	0.05	0.02
	ST12	15.55	2.61	26.11	11.98	0.12	0.09	0.37	0.41	0.05	0.02
	ST16	16.48	3.81	16.49	11.62	0.12	0.09	0.66	0.49	0.05	0.02
	ST17	16.67	4.06	14.29	11.16	0.12	0.09	0.70	0.47	0.06	0.03
	ST20	17.51	4.92	5.37	6.76	0.14	0.12	1.05	0.54	0.06	0.04
	ST21	17.84	5.26	2.57	4.33	0.15	0.11	1.12	0.51	0.06	0.03
	ST24	17.85	5.37	0.45	1.12	0.16	0.16	1.21	0.54	0.06	0.04
	ST25	17.80	5.52	0.14	0.47	0.18	0.17	1.22	0.51	0.07	0.05
	All	16.75	4.20	16.46	13.77	0.12	0.12	0.65	0.56	0.04	0.03
2012-2013 Estuary	ST5	15.30	2.53	28.92	7.61	0.05	0.05	0.63	0.56	0.11	0.13
	ST6	15.58	3.15	24.28	10.32	0.03	0.04	0.51	0.29	0.14	0.12
	ST9	17.95	5.73	20.49	10.35	0.04	0.05	0.56	0.32	0.13	0.11
	ST10	15.65	2.60	22.99	12.01	0.04	0.05	0.51	0.34	0.15	0.14
	ST11	15.00	2.40	21.81	13.58	0.05	0.06	0.74	0.74	0.11	0.10
	ST12	15.43	2.46	23.11	12.57	0.06	0.07	0.53	0.59	0.15	0.12
	ST16	15.65	3.18	18.22	12.69	0.08	0.07	0.44	0.34	0.13	0.12
	ST17	15.49	3.32	16.57	13.51	0.07	0.07	0.42	0.29	0.15	0.10
	ST20	14.22	6.28	6.06	7.38	0.06	0.04	0.48	0.34	0.16	0.09
	ST21	16.81	5.21	3.29	5.04	0.04	0.04	0.63	0.60	0.26	0.17
	ST24	16.86	5.97	0.65	1.07	0.06	0.04	0.55	0.52	0.17	0.18
	ST25	17.30	5.79	0.21	0.24	0.06	0.06	0.69	0.54	0.13	0.13
	All	15.95	4.24	15.69	13.62	0.05	0.05	0.56	0.46	0.13	0.13
2012-2013 Tributaries	T1	16.36	4.13	0.21	0.10	0.38	0.30	1.45	1.88	0.12	0.09
	T2	15.80	4.66	0.26	0.18	0.35	0.35	1.08	1.25	0.14	0.03
	T3	16.50	5.61	0.26	0.14	0.32	0.30	1.00	0.90	0.22	0.14
	T4	17.43	5.63	0.99	1.04	0.33	0.26	0.77	1.28	0.12	0.06
	T5	16.71	4.81	19.80	12.29	0.29	0.32	1.47	1.69	0.27	0.07
	T6	16.09	4.19	6.06	10.11	0.20	0.19	0.40	0.51	0.22	0.06
	T7	21.94	5.37	4.69	8.99	0.31	0.27	0.89	1.34	0.20	0.10
	T8	16.95	5.61	0.21	0.10	0.34	0.31	1.49	1.69	0.19	0.05
	T9	16.96	5.73	0.42	0.41	0.25	0.28	1.50	1.54	0.21	0.05
	T10	17.01	5.58	0.26	0.04	0.29	0.23	1.66	1.89	0.19	0.05
	T11	16.77	6.45	0.11	0.04	0.23	0.21	1.12	1.65	0.26	0.05
	All	17.15	5.33	3.05	7.80	0.31	0.27	1.17	1.44	0.20	0.09

Table I.4.
One-way ANOVA, with Welch's correction for heterogeneous groups, to assess differences between the 2003-2007and the 2012-2013 time intervals.

		Levene's test		ANOVA	
		F	Pr(>F)	F	Pr(>F)
Ammonia	Annual	11.316	0.001*	55.217	0.000*
	winter	1.729	0.193	84.845	0.000*
	spring	2.440	0.125	1.608	0.211
	summer	0.014	0.905	3.178	0.079
	autumn	11.980	0.001*	20.076	0.000*
Nitrate	Annual	0.433	0.511	3.482	0.063
	winter	0.211	0.647	15.922	0.000*
	spring	0.159	0.692	8.048	0.007*
	summer	0.158	0.693	0.027	0.870
	autumn	6.892	0.011*	4.004	0.050
Phosphate	Annual	153.190	0.000*	81.606	0.000*
	winter	33.980	0.000*	37.655	0.000*
	spring	8.067	0.007*	8.554	0.014*
	summer	13.368	0.000*	0.809	0.373
	autumn	25.824	0.000*	127.578	0.000*

* Significant values (<0.05)

Analysing the differences between the two periods for each estuarine sampling station, we found that ST9 shows significant differences between periods. The annual mean concentrations of ammonia in ST9 significantly decreased (t-value= 11.912, P(t)<0.001), as well as the summer mean concentration (t-value= 8.517, P(t)= 0.028) and the autumn mean concentration (t-value= 9.146, P(t)= 0.022). On the contrary, the annual mean concentration of nitrate in ST9 significantly (t-value= 8.634, P(t)= 0.026) increased between periods, from 0.342 mg N/l to 0.705 mg N/l. Only other two stations show significant differences between periods. The annual mean concentration of ammonia in ST24 significantly decreased (t-value= 4.513, P(t)= 0.030) as well as the summer mean concentration of ammonia in ST6 (t-value= 8.807, P(t)= 0.020). There were not enough spring replicates to test, for each estuarine station, the difference between periods (Figure I.6).

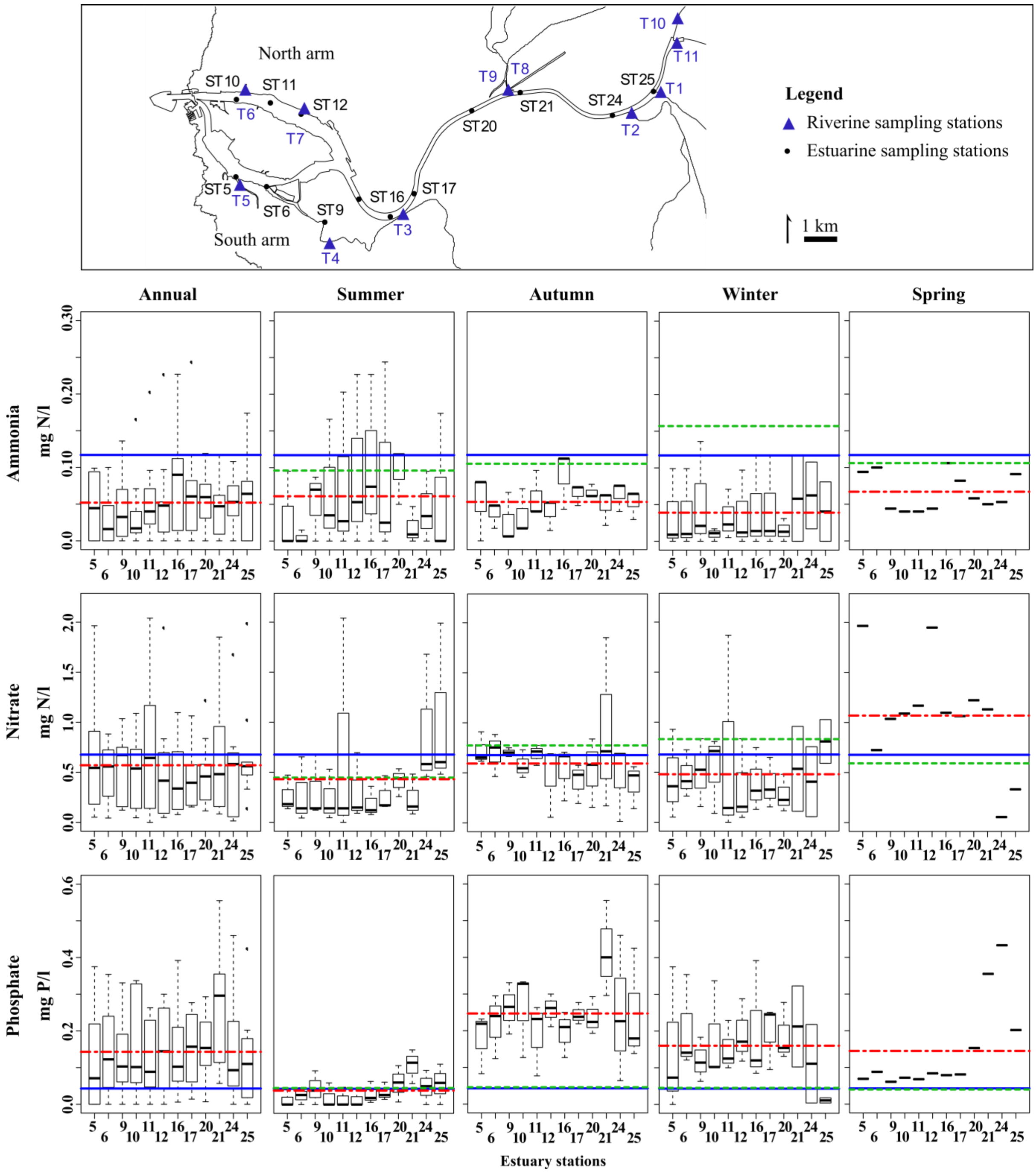


Figure I.6.

Boxplots of average annual and seasonal mean concentrations (mg/l) of ammonia, nitrate and phosphate sampled in the Mondego estuary subtidal stations, between the summer of 2012 and the spring of 2013. Sampling stations are ordered from downstream to upstream stations. Blue line is the average annual mean concentration of the estuary for the 2003-2007 tiime interval. Green dashed lines are the average seasonal mean concentrations for estuarine stations for the 2003-2007 time interval. Red dashed line is the annual (first column in figure) and seasonal (last four columns in figure) mean concentrations of the estuary of the recent sampling campaigns (2012-2013). Spring concentrations without replicates. The upper map shows the location of the sampling stations in the Mondego estuary. T-riverine stations. ST- estuarine stations.

Mondego freshwater quality and the status of European rivers

The European Indicator Assessment (EEA 2014) defined the annual means of ammonia, nitrogen (nitrate) and phosphorous (phosphate) as physico-chemical indicators of freshwater quality. The EEA reports (EEA 2015a, EEA 2015b) indicate that the concentrations of European rivers have declined steadily over the period 1992 to 2012 (Table I.5).

Table I.5.
Average annual mean concentrations for river stations in Europe (EEA 2015a, EEA 2015b) and in the tributaries of the Mondego estuary.

	European rivers			Mondego tributaries
	1992	2000	2012	2012/2013
Nitrate (mg N/l)	2.66	2.08	1.82	1.17
Ammonia (mg N/l)	0.41	0.31	0.16	0.31
Phosphate (mg P/l)	0.13	0.11	0.07	0.20

The annual and the seasonal mean concentrations of the tributaries of the Mondego estuary were compared to the average annual concentrations for European rivers in 1992, 2000 and 2012. Phosphate is the indicator with the lowest performance in the tributaries of the Mondego estuary, with annual concentrations higher than the average concentrations of 1992 (blue line in figure I.7). Ammonia shows annual mean concentrations higher than the European average concentrations in 2012. Autumn is the season whose ammonia concentrations show the highest values, with almost all stations high above the European 1992 mean concentrations. Nitrate is the indicator with the lowest concentrations when comparing to the European averages. Nitrate annual mean concentrations are lower than the 2012 concentrations (Figure I.7).

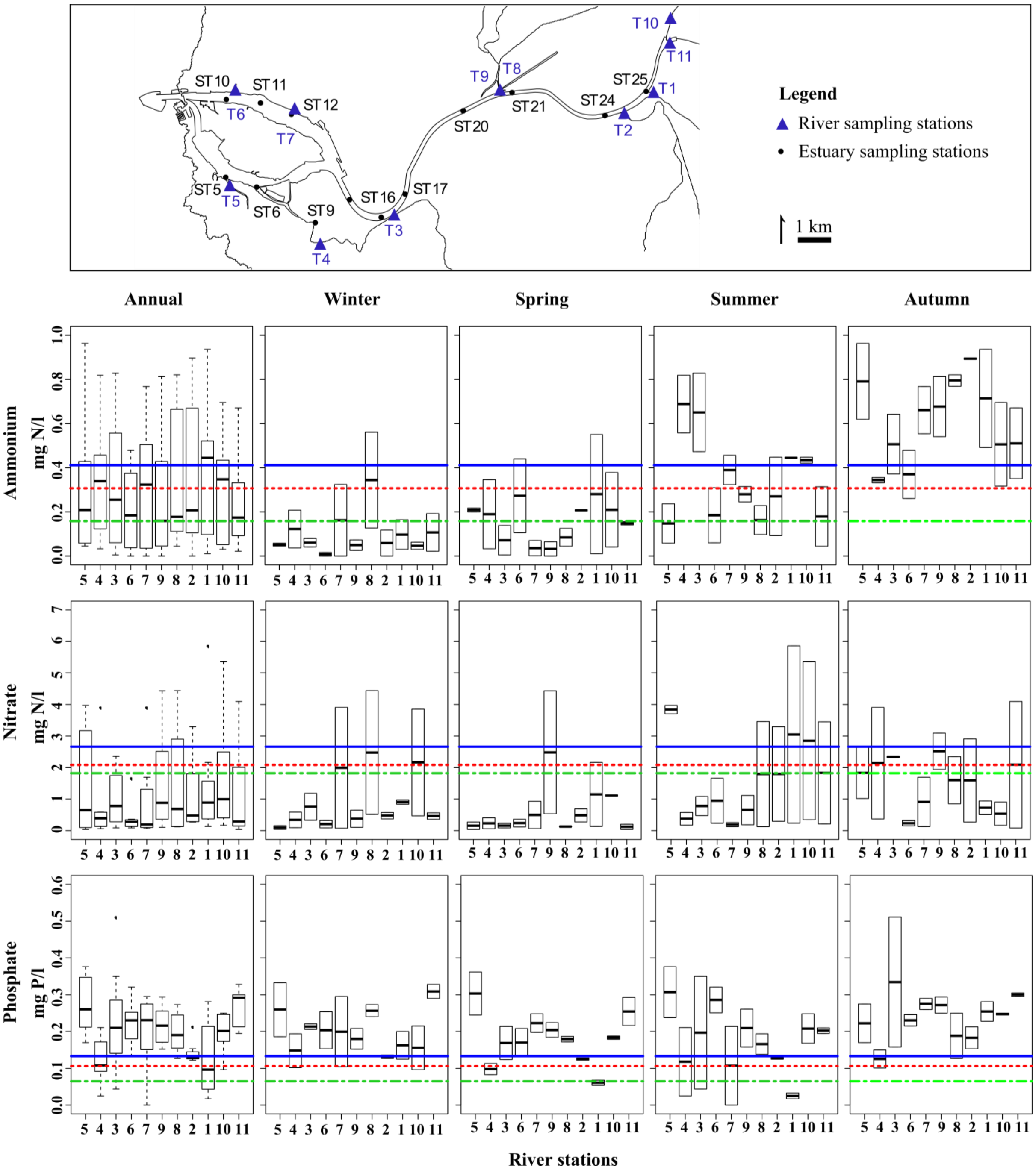


Figure I.7.

Boxplots of annual and seasonal mean concentrations (mg/l) of nitrate, ammonia and phosphate sampled in the mouth of the Mondego estuary tributaries, between summer 2012 and spring 2013. In the graphs stations are ordered from downstream to upstream stations. The upper map shows the location of the sampling stations. T-river stations. ST- estuary stations. Dashed lines are the average annual mean concentrations for European rivers in 1992 (blue), 2000 (red) and 2012 (green).

Differences between river and estuary

Freshwater inputs are one of the main sources of nutrients to the Mondego estuary. The differences between the concentrations of ammonia, nitrate and phosphate measured in the Mondego tributaries and the concentrations measured in the adjacent estuarine stations during the 2012-2013 field campaign were analysed (Table I.3).

Table I.6.
One-way ANOVA with Welch's correction and pairwise comparisons (Games Howell t-test for unequal variances and unequal sample sizes) to assess differences between river stations and their in adjacent estuarine stations.

			ANOVA		Pairwise comparisons		
River station	Estuary station(s)	Parameter	F	Pr(>F)	Group of stations	t	Pr(>t)
T1	ST25	$N\text{-}NH_3$	8.815	0.019*			
		$N\text{-}NO_3^-$	1.373	0.260			
		$P\text{-}PO_4$	0.000	0.995			
T2	ST24 and ST25	$N\text{-}NH_3$	2.727	0.105			
		$N\text{-}NO_3^-$	0.943	0.404			
		$P\text{-}PO_4$	0.184	0.835			
T3	ST16 and ST17	$N\text{-}NH_3$	2.529	0.116			
		$N\text{-}NO_3^-$	1.472	0.263			
		$P\text{-}PO_4$	1.462	0.251			
T4	ST9	$N\text{-}NH_3$	9.733	0.016*			
		$N\text{-}NO_3^-$	0.252	0.623			
		$P\text{-}PO_4$	0.073	0.790			
T5	ST5 and ST6	$N\text{-}NH_3$	2.610	0.110			
		$N\text{-}NO_3^-$	1.349	0.295			
		$P\text{-}PO_4$	5.227	0.013*			
					T5:ST5	3.400	0.010*
					T5:ST6	2.800	0.033*
T6	ST10 and ST11	$N\text{-}NH_3$	2.998	0.083			
		$N\text{-}NO_3^-$	0.856	0.437			
		$P\text{-}PO_4$	2.465	0.105			
					T6:ST11	2.920	0.027*
T7	ST11 and ST12	$N\text{-}NH_3$	0.049	0.953			
		$N\text{-}NO_3^-$	0.373	0.692			
		$P\text{-}PO_4$	0.244	0.785			
T8	ST20 and ST21	$N\text{-}NH_3$	0.449	0.643			
		$N\text{-}NO_3^-$	0.277	0.760			
		$P\text{-}PO_4$	2.067	0.147			
T9	ST20 and ST21	$N\text{-}NH_3$	2.492	0.121			
		$N\text{-}NO_3^-$	1.689	0.225			
		$P\text{-}PO_4$	1.868	0.190			
T10	ST25	$N\text{-}NH_3$	7.977	0.023*			
		$N\text{-}NO_3^-$	2.203	0.158			
		$P\text{-}PO_4$	2.028	0.175			
T11	ST25	$N\text{-}NH_3$	5.544	0.047*			
		$N\text{-}NO_3^-$	0.565	0.464			
		$P\text{-}PO_4$	7.659	0.014*			

(continues from Table I.6)
* Significant values (<0.05)
Pairwise comparisons only performed for river stations with two adjacent estuary stations and only shown whenever significant differences between stations were present.

Three upstream river stations - T1, T10 and T11- and one in the south arm – T4 - reveal significantly higher concentrations of ammonia than their adjacent estuarine stations (Table I.6). No significant differences were found between river and adjacent estuary stations regarding the nitrate concentrations. The most downstream stations- T5 and T6- and one of the most upstream - T11- show significantly higher phosphate concentrations than their adjacent estuarine stations (Table I.6).

Discussion

Progress in time

In his study we apply assessment indicators from the Environmental European Agency core set, namely nutrients and oxygen-consuming-substances, both in transitional waters and in freshwater, to evaluate the progress of estuarine and riverine concentrations of nitrate, phosphate and ammonia between two time intervals (2003-2007 and 2012-2013).

The mitigation measures implemented in the Mondego estuary, in 1998, were able to change the physico-chemical quality of the south arm water body from Moderate to Good, based on the WFD criteria (Lillebø et al. 2007). The eutrophication symptoms decreased, but the concentrations of chlorophyll *a*, of oxidised forms of nitrogen and of dissolved inorganic phosphorous did not show any significant changes and it was suggested that the system became nitrogen limited due to the reduction of ammonia from the Pranto River (Lillebø et al. 2005). The indicators applied in this study reveal further decrease of ammonia concentrations in the estuary, between 2003-2007 and 2012-2013, and no significant differences in the concentrations of nitrate, but show an increase on the concentrations of phosphate.

Regarding the tributary rivers, our study presents the first assessment in all the direct tributaries of the Mondego estuary regarding nitrate, phosphate and ammonia. Progress was evaluated comparing the concentrations in the tributaries with the average annual mean concentrations of ammonia, nitrate and phosphate in European rivers. The indicators have shown that the annual mean concentrations of phosphate are higher than the average annual mean concentrations in European rivers in 1992, concentrations of ammonia are higher than the average in European rivers in 2000 and concentrations of nitrate are lower than the average in 2012. Among the three parameters analysed, the concentrations of phosphate have shown the lowest performance both in the estuary and in the tributary rivers and indicate that the estuary, as well as the upstream freshwater sections, remains under pressure due to the concentrations of phosphate. Pressure over the estuary will remain as long as the physico-chemical status, supporting the final evaluation of the ecological status, presents low quality.

Pressure from nutrient loadings

In this study, we evaluated pressure from nutrient loadings into the estuary analyzing the dependency of the estuarine concentrations of nitrate, ammonia and phosphate on freshwater inputs. To achieve our goal, we analyzed the linear relationships of nitrate, ammonia and phosphate with salinity, assuming that significant negative relations indicate dependency on riverine freshwater inputs (effect of precipitation of estuarine concentrations was previously discarded). The relationships with water temperature were also analyzed, and when positive, indicate dependency on biogeochemical processes (Otero et al. 2013).

Results support the dependency of nitrate and phosphate on freshwater inputs during the 2003-2007 time interval as both parameters show significant negative linear relation with salinity. The absence of significant positive linear relation of phosphate with temperature and the persistence of high concentrations of phosphate during autumn and winter are also an indication of phosphate dependency on external sources, rather than on biogeochemical processes (Otero et al. 2013). Lillebø et al. (2005) also demonstrated lack of relation between phosphate and temperature between 1999 and 2003, but, contrary to our study, phosphate did not show significant relations with salinity and/or precipitation and the authors suggested that the high concentrations of phosphate were only a result of diffuse pollution from aquaculture farms and/or small industries. Ammonia did not show any relation either with salinity or temperature, indicating that this nitrogen-source is not freshwater-dependent and that the primary producers are able to control the sediment/water flux (Lillebø et al. 2002).

On the contrary, in the most recent time interval (2012-2013) results indicate that none of the three parameters was dependent on freshwater inputs from rivers. Absence of linear relation between phosphate and temperature also indicates lack of dependency of phosphate on mineralization processes. Results suggest that the higher phosphate concentrations in 2012-2013 are a result of diffuse and/or point sources of pollution, rather than of freshwater inputs. Further analyses with a longer-term dataset should be performed to verify these conclusions. We have also shown that, with exception for two very small tributaries, the mean concentration of phosphate in estuarine stations is not significantly different from the nearest tributary of the Mondego estuary.

The need for mitigation measures

As long as the physico-chemical status, supporting the final evaluation of the ecological status, presents low quality, the estuarine system will remain a "Potential Problem Area" (Lillebø et al. 2007). In our study we have shown that the system remains under pressure due to concentrations of phosphate, although the changes in the hydrodynamics of the estuary have successfully eliminated the eutrophication symptoms in the estuary. This suggests that further mitigation measures ought to be applied in order to reduce the concentrations of phosphate.

As shown in the abovementioned section, inputs from the tributaries and other diffuse external sources are an important source of phosphate to the Mondego estuary. In turn, the agricultural fields surrounding the tributaries are the most probable source of nutrients into the hydrographic network. European Directives have successfully contributed to the reduction of point source pollution of phosphate through the reduction of phosphorous in detergents, but diffuse runoff from agricultural land remains an important source in Europe (EEA 2015c). In the Mondego lower valley 45% of land is covered by agricultural land and the estuary is surrounded by rice and corn fields (Teixeira et al. 2014). Pollution from industries is also expected, as the WWTPs that operate in the area lack capacity to treat industrial waste water.

On the contrary, at the International level, policies to reduce the concentration of nitrate in agriculture have been in action since 1991 and have been successful. The decline in the Mondego river basin is most likely associated to the designation of the “Nitrate-Vulnerable Zone of the Littoral Center” and to the implementation of good agricultural practices (EC 1992, EC 2010) to reduce nitrate in agricultural runoff, as required by the directive (EC 1991b). Likewise, the reduction of ammonia into waters has been successful at the European and at the local level, revealing efficiency of domestic waste water treatment (EEA 2015c). Nonetheless, further improvements regarding waste water treatment should be implemented in the Mondego lower valley, as some urban and industrial facilities still lack waste water treatment (PGRH4 2012).

Conclusion

The WFD states that European waters must achieve, and or maintain, the “good ecological status” by 2015. The good ecological status is defined in terms of chemical characteristics, hydrological characteristics and biological community. In this study we have shown that assessment indicators to evaluate progress in the physico-chemical parameters of a coastal system remain important even when previous mitigation measures have been implemented and the system has overcome its major problems.

Understanding the main sources of pressure and in what manner they affect estuarine systems is another key step for improvement. The pressure indicator applied in this study, showing that the concentrations of certain chemical parameters in estuary depend on freshwater inputs from rivers but also on other external sources, has provided a basis for the development of further mitigation measures to ensure the reduction of nutrient loadings and reduce pressure in the system.

Chapter II

Relating landscape to stream nitrate concentrations in a coastal eastern-Atlantic watershed (Portugal).

Abstract

We apply a linear regression mixed effects model to explore the influence of landscape factors on nitrate concentrations in a coastal watershed of Portugal. Landscape composition and configuration metrics, together with variables assessing the physical characteristics of the study area, were used. The analysis was performed using seasonal data from the years 2001 and 2006. The seasonal influence was included as a random effect to account for temporal correlations. Together, the fixed and the random factors explain 78% of the variance, whereas the fixed factors alone explain 10%. Urban, slope, elevation and aggregation index of urban class contribute to the differences found in the nitrate concentrations. Urban has the weakest effect, whereas slope and elevation show a conditioned negative effect on nitrate. The effect of slope gets stronger for higher standard deviations of elevation and the effect of the standard deviation of elevation, measuring the variation of elevation within a sub-watershed, gets stronger for steeper slopes. Of the configuration class level metrics included in the analysis, only aggregation index of urban played a significant role in the final model, and it revealed to be related to urban percentage. The influence of landscape configuration metrics, though observed by others, was not obvious in this study. Future analysis evaluating the effect of metrics selection could be performed.

Keywords: Land cover, landscape metrics, nitrate, linear mixed-effect model, river basin, Mondego

Introduction

Elements of stream water, such as nitrate and phosphorous, whose concentration determines the quality of the system may derive from a variety of sources. Din et al. (2014), for instance, distinguished the contribution of five different nitrate sources to total stream nitrate in an agricultural field (atmospheric deposition, soil organic matter nitrification, chemical fertilizer nitrification, groundwater and manure and sewage). The supply and availability of these elements to the stream biota is influenced by both landscape-level processes and in-stream processes. Landscape-level processes include watershed geology (Luo et al. 2013, Selle et al. 2013), hydrology (Bartoli et al. 2012, Ficklin et al. 2013), soil processes (McDowell and Liptzin 2014), land-use practices and management activities (Gundersen et al. 2006, Monaghan et al. 2009) and landscape composition and pattern (Krupa et al. 2011, Wan et al. 2014). River water chemistry is therefore controlled by both forest and anthropogenic factors that can either be diffuse (e.g. from crop cultivation and urban drainage) or concentrated (e.g. wastewater treatment plants [WWTP]). Addressing the diffuse sources has been one of the major challenges for researchers, which have often taken the landscape approach to evaluate the relationship between land use and land cover (LUC) and nutrient concentrations on water (Ahearn et al. 2005, Chen and Lu 2014, Huang et al. 2013).

Several statistical methods have been employed to reveal the relationships between landscape characteristics and water quality parameters. Frequently applied traditional statistical methods include multivariate analysis (Chen and Lu 2014, Fučík et al. 2012, Selle et al. 2013), constrained least square (CLS) regression models (Kang et al. 2010) and multiple linear regression (Huang et al. 2013, Sangani et al. 2015). The simplicity and robustness of these techniques makes them appealing to estimate independent variables. The disadvantage of these methods is that the relationship between landscape and water quality is scale-dependent, temporal-dependent and spatial-dependent (Zhou et al. 2012) and the implicit correlations are not always taken into account (Krupa et al. 2011). Mixed effects models can provide an alternative to the above-mentioned methods. These are suitable for hierarchical data allowing the introduction of correlation structures between observations (Zuur et al. 2009). The mixed effects model combines a regression model with a random effects analysis of variance model (Ahearn et al. 2005, Cabezas et al. 2010). Madriñan et al. (2012) applied a linear mixed model to analyse more rigorously a decreasing trend in water turbidity across four sub-watersheds within the Tampa Bay watershed, where consistent changes in land use and land cover were identified. Taranu and Gregory-Eaves (2008) applied a meta-analytical analysis to synthesise overall across-study effect of agriculture on water quality based on study-specific correlations and then applied a linear mixed-effects model to address within-study variability. The authors tested whether the slopes and intercepts of the relationships between variables differed across studies.

More recently, spatial regression models coupled with kriging techniques have been used to account for spatial autocorrelation (Chan 2008, Yang and Jin 2010), and geographically weighted regression (GWR) has been applied to explain spatial variation by incorporating spatial coordinates into the regression model (Tu 2013, Yu et al. 2013). Spatial regression methods provide accurate predictions along with uncertainty estimation but the process is computationally complicated and these methods are not appropriate for small datasets. More recently, the Bayesian hierarchical framework has been applied to account for the effect of independent variables on different spatial scales, spatially varying regression parameters and distribution of parameters. The Bayesian framework includes time varying covariates and semi-parametric spatial covariance structures and has proven to be a strong tool to model spatio-temporal dependencies (Ding et al. 2014, Wan et al. 2014). However, it often comes with high computational cost, especially in models with a large number of parameters and, historically, the Bayesian approach has been criticised because it requires "subjective" specification of prior information on the parameters (Cressie et al. 2009).

The Mondego River Basin case

In Portugal, the water quality of the Mondego river basin has been widely monitored both by national governmental institutions and by funded research projects (e.g. Marques et al. 2003, Feio et al. 2009, Patrício et al. 2009). However, water quality studies on this system have been focused mainly on the water environment and its communities without taking into account the surrounding landscape (e.g. Neto et al. 2008). Those studies that have addressed the landscape environment have been implemented at the estuarine scale. Cardoso et al. (2008), for instance, explored the interaction between extreme flooding events and anthropogenic stressors on the estuarine macrobhentic communities' dynamics. Ferreira et al. (2003) classified sensitive and/or vulnerable zones in ten Portuguese systems, describing the potential sources of nutrient inputs (treated or untreated domestic and industrial effluents and agricultural point and diffuse sources), but focus was only on transitional systems.

This study focuses on describing the relationship between stream water quality and landscape characteristics at the watershed scale. The linkage between nitrate and landscape descriptors in the Mondego river basin will be assessed. Though we have hierarchical data with temporal and spatial correlation, we selected the linear mixed effects model (LME) over others more computationally complicated due to our small sized dataset and also because we were focused on providing a descriptive, rather than predictive, model.

The following questions will be addressed: a) Does nitrate exhibit seasonal and spatial variability at the watershed-scale? b) Which landscape factors most contribute for nitrate variability? c) What is the relationship between landscape factors and nitrate?

Methodology

Study area

The Mondego river basin in Central Portugal (Figure II.1) has a North-South orientation and a total area of 6658 km2. The southeast upstream region of the watershed is mountainous, but only 20% of the total area has a topographic slope higher than 20%. The medium altitude is 324.84 m.

The Mondego river basin is mainly occupied by agricultural and forest areas that are distributed throughout the basin, whereas urban and industrial land are essentially located on the coastal strip. Major urban centres within the region are the cities of Coimbra (population 139 151), Leiria (126 348), Viseu (98 778) and Figueira da Foz (61 505); two other cities have populations higher than 50 000 (INE 2014).

Teixeira et al. (2014) characterised the main land uses in the river basin and assessed the most relevant transitions that occurred in the basin from 1990 to 2006. Though some of the changes observed showed a systematic pattern indicating that they might have evolved in a consistent manner due to some socio-economic processes, land persistence accounts for more than 95% of the total units of observation, i.e. raster cells, in the two time intervals analysed (1990-2000, 2000-2006).

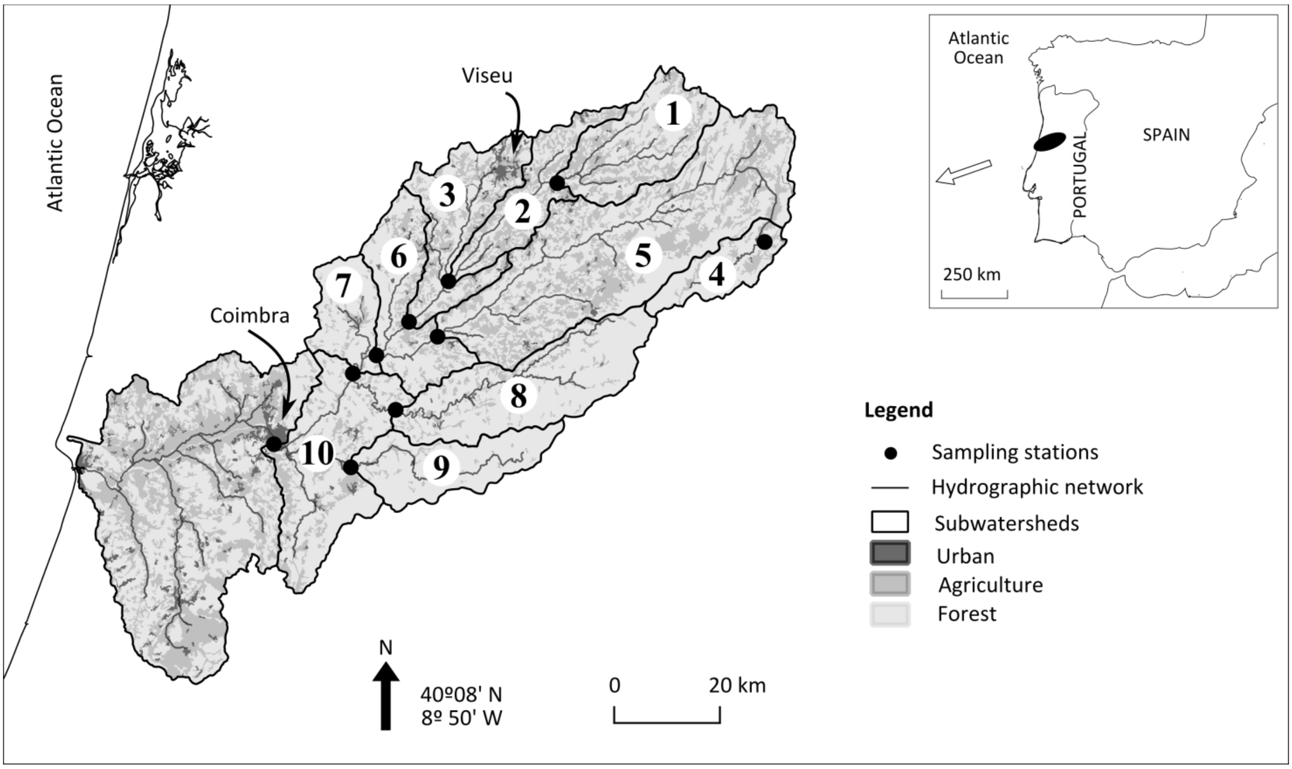


Figure II.1.
Map in the upper right shows the location of the study site in the centre of Portugal. The study site is the Mondego river basin. The map on left shows the land cover reclassification based on the 2006 inventory of the CORINE Land Cover project. Numbers 1-10 identify the sub-watersheds used in the analysis.

The mean annual temperature is 13.4°C, whereas the mean temperature in the coldest months of December and January is 8°C and in the warmest months of July and August is 20°C. The mean annual precipitation is 1073 mm, out of which 72 % occurs between October and March (PGBH dos Rios Vouga, Mondego e Lis 2012). Compared to these general climate patterns, observed for the period 1931-1990, several differences in the climate of Portugal were recorded between 1993 and 2006 (IPMA 2013, APA). During this period, heavy precipitation events and drastic shifts to very low levels of precipitation were observed. In particular, in the winter of 2000/01 unprecedented high values of precipitation were registered (1802.1 mm in Central Portugal against the average annual value of 1030.6 mm), causing one of the largest floods of the century. This hydrologic event was followed by a severe drought in 2005 (486.1 mm in Central Portugal)

(IPMA 2013, APA). Both events precede our nitrate concentrations dataset. 2001 and 2006 datasets were analysed.

Land cover

CORINE Land Cover raster data, resolution 100x100m, for the 2000 and 2006 inventories was used (EEA 2012). Because we were interested in understanding whether major land cover classes could detect and explain some of the variability on nitrate datasets, the 44 CORINE classes were aggregated into 4 categories: urban, agriculture, forest and water. Urban matches CORINE class 1 (i.e., artificial surfaces), agriculture matches CORINE class 2 (i.e., agricultural areas) and water corresponds to CORINE class 5 (i.e., water bodies), from the first nomenclature level. The forest class aggregates classes 3 (i.e, forest and natural areas) and 4 (i.e, wetlands), also from the CORINE first level. Water was not included in the analysis. Satellite imagery to produce the CORINE Land Cover maps were collected between the years 1999 and 2002 (JRC-IES 2005) for the clc2000 map and in the spring and summer of 2006 for the clc2006 map (Caetano et al. 2009).

Nitrate datasets

Water quality datasets should match the years of the satellite imagery used to produce the CORINE land cover maps. The Portuguese Environment Agency (APA) offers water quality data for all the river basins of Portugal, but datasets are incomplete.

For the Mondego river basin we managed to download datasets of monthly nitrate concentrations (mg N/l) for the years 2001 and 2006, for 10 gauge stations, though incomplete. To overcome data gaps, and whenever possible, seasonal values of nitrate concentrations were calculated and used herein (Figure II.2). Nitrate data was not available for the most downstream sub-watershed, where the Mondego estuary is located, and therefore this sub-watershed was removed from the analysis (Figure II.1).

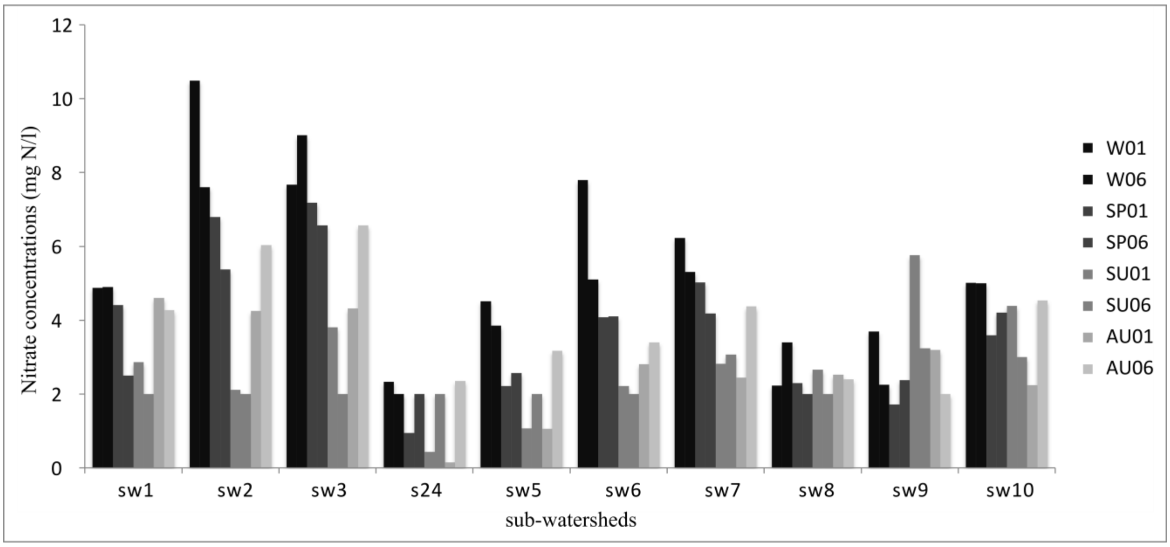


Figure II.2.
Mean seasonal nitrate concentrations per sub-watershed. sw - sub-watershed; 1:10- sub-watershed ID; W-winter; SP-spring; SU-summer; AU-autumn.

Sub-watersheds

The Mondego river basin was divided into 11 sub-watersheds according to the location of the 10 gauge stations for which nitrate data was collected and one more station located near the mouth of the estuary (Figure II.1). Sub-watersheds were defined using the watershed delineation plugin (Moya 2011) for MapWindow GIS (version 4.8.6) and were based on SRTM 30m digital elevation data (version 4.1.) derived from USGS/NASA SRTM data (USGS 2014). The Mondego river basin limits available from the APA site were used as a focusing mask and a threshold of 40 km2 was used for network delineation. The resulting stream net coincides with the hydrographic network available also from APA. Sub-watersheds vary in size from 18 000 ha to 135 000 ha (Figure II.1). Rivers range from first to fourth order.

Landscape factors

To describe the physical characteristics of each sub-watershed we used the mean sub-watershed slope (SLO) and the standard deviation of sub-watershed elevation (STDAVE) (Table II.1). Both variables were calculated based on the SRTM 30m (USGS 2014) using QGIS 2.2.0 *Valmiera* application (QGIS 2014). SRTM30 was first converted to the ETRS89 Lambert Azimuthal

Equal-Area projection coordinate reference system (ETRS89/ETRS-LAEA) to match European datasets (EC 2010) and resampled to 100m resolution, using the bilinear method, to match the CORINE Land Cover maps resolution.

To quantify landscape composition, the percentages of urban (PLANDU), agriculture (PLANDA) and forest (PLANDF) were derived through spatial analysis in QGIS (2014) (Table II.1). To describe landscape configuration 57 landscape metrics at the class level were calculated (Table II.2a and Table II.2b). Some are only distinguished because both the mean (MN) and the weighted-area mean (AM) were calculated. Because some of the metrics are so closely related, groups of metrics with Pearson correlation coefficients higher than 0.8 (Appendix A) were formed and one metric to represent each group was arbitrarily selected. This process eliminated 40 metrics.

A principal component analysis (PCA) with the orthogonal varimax rotation was applied to the remaining metrics as a mean to describe the covariance structure among pairs of metrics with medium and small correlations. From each retained factor, the metric with the highest loading was selected for further analysis (Table II.4). The following criteria were defined to select the principal factors to consider further: a) cumulative proportion up to 90% and b) eigenvalues higher than 1.

These criteria allowed us to retain seven factors. From each retained factor, the metric with the highest loading was selected for further analysis (Table II.4). The percentage of urban (PLANDU), agriculture (PLANDA) and forest (PLANDF) were not included in the factor analysis. Package FRAGSTATS 4.2 (McGarigal and Marks 1995) was used to calculate the class level metrics and package *psych* available for the R software was used to perform factor analysis.

Table II.1.
Land cover composition and physical characteristics in the Mondego river basin.

SW	Year	PLANDU	PLANDA	PLANDF	PLANDW	SLO	STDAVE
sw1	2000	0.7	35.01	64.07	0.21	12.81	107.62
	2006	1.09	34.76	63.94	0.22		
sw2	2000	1.84	47.25	50.9	0	12.86	113.03
	2006	2.8	46.58	50.62	0		
sw3	2000	5.84	36.1	57.87	0.2	11.89	124.59
	2006	7.65	34.76	57.33	0.26		
sw4	2000	0.3	20.92	78.48	0.3	22.08	175.61
	2006	0.44	17.83	81.37	0.36		
sw5	2000	0.99	37.64	61.32	0.06	14.62	209.13
	2006	1.53	36.94	61.47	0.06		
sw6	2000	1.53	25.98	68.97	3.52	15.58	162.37
	2006	2.61	24.41	69.37	3.61		
sw7	2000	0.45	14.74	83.97	0.84	20.41	120.15
	2006	1.3	13.93	83.93	0.84		
sw8	2000	0.37	13.61	85.46	0.55	29.99	421.41
	2006	0.55	13.26	85.62	0.57		
sw9	2000	0.2	6.54	93.26	0	34.4	262.28
	2006	0.46	6.14	93.4	0		
sw10	2000	1.29	24.45	73.59	0.66	22.5	171.4
	2006	1.94	23.8	73.61	0.66		

SW – sub-watershed; PLANDU – percentage of urban; PLANDA – percentage of agriculture; PLANDF – percentage of forest; PLANDW – percentage of water; SLO - mean sub-watershed slope; STDAVE - standard deviation of sub-watershed elevation

Table II.2a.
Name and description of class level landscape metrics (adapted from McGarigal and Marks 1995)
For some metrics, both the mean and the area-weighted mean were calculated. In total, considering the 3 land cover types, 60 metrics were calculated.

Name	Range	Description
Percentage of landscape (PLAN)	0 < PLAND ≤ 100	percentage of the landscape comprised of the corresponding patch type. approaches 0 when the corresponding patch type (class) becomes increasingly rare in the landscape.
Number of patches (NP)	NP ≥ 1, without limit	number of patches of the corresponding patch type (class). 1 when the landscape contains only 1 patch of the corresponding patch type
Patch Density (PD)	PD > 0	number of patches of the corresponding patch type divided by total landscape area (nr/ha)
Largest Patch Index (LPI)	0 < LPI ≤ 100	percentage of the landscape comprised by the largest patch. approaches 0 when the largest patch of the corresponding patch type is increasingly small. LPI = 100 when the largest patch comprises 100% of the landscape
Landscape Shape Index (LSI)	LSI ≥ 1, without limit	total length of edge involving the corresponding class, given in number of cell surfaces, divided by the minimum length of class edge possible for a maximally aggregated class LSI increases without limit as the patch type becomes more disaggregated
Patch Area (AREA)	AREA > 0, without limit	area of the patch (ha)
Radius of Gyration (GYRATE. MN and AM)	GYRATE ≥ 0, without limit	mean distance (m) between each cell in the patch and the patch centroid (m). 0 when the patch consists of a single cell and increases without limit as the patch increases in extent.
Shape Index (SHAPE. MN and AM)	SHAPE ≥ 1, without limit	patch perimeter (given in number of cell surfaces) divided by the minimum perimeter (given in number of cell surfaces) possible for a maximally compact patch (in a square raster format) of the corresponding patch area. 1 when the patch is maximally compact (i.e., square or almost square) and increases without limit as patch shape becomes more irregular.
Fractal Dimension Index (FRAC. MN and AM)	1 ≤ FRAC ≤ 2	2 times the logarithm of patch perimeter (m) divided by the logarithm of patch area (m^2) approaches 1 for shapes with very simple perimeters such as squares

MN (Mean) equals the sum, across all patches of the corresponding patch type, of the corresponding patch metric values, divided by the number of patches of the same type.
AM (area-weighted mean) equals the sum, across all patches of the corresponding patch type, of the corresponding patch metric value multiplied by the proportional abundance of the patch [i.e., patch area (m2) divided by the sum of patch areas].

Table II.2b. (continues from table II.2a.)
Name and description of class level landscape metrics (adapted from McGarigal and Marks 1995)
For some metrics, both the mean and the area-weighted mean were calculated. In total, considering the 3 land cover types, 60 metrics were calculated.

Name	Range	Description
Related Circumscribing Circle (CIRCLE. MN and AM)	0 ≤ CIRCLE < 1	1 minus patch area (m^2) divided by the area (m^2) of the smallest circumscribing circle. 0 for circular patches and approaches 1 for elongated
Contiguity Index (CONTIG. MN and AM)	0 ≤ CONTIG ≤ 1	average contiguity value for the cells in a patch (i.e., sum of the cell values divided by the total number of pixels in the patch) minus 1, divided by the sum of the template values minus 1. 0 for a one-pixel patch and increases to a limit of 1 as patch contiguity, or connectedness, increases.
Percentage of like adjacencies (PLADJ)	0 ≤ PLADJ ≤ 100	number of like adjacencies involving the focal class, divided by the total number of cell adjacencies involving the focal class. 0 when the corresponding patch type is maximally disaggregated
Interspersion and Juxtaposition Index (IJI)	0 < IJI ≤ 100	minus the sum of the length (m) of each unique edge type involving the corresponding patch type divided by the total length (m) of edge (m) involving the same type, multiplied by the logarithm of the same quantity, summed over each unique edge type; divided by the logarithm of the number of patch types minus 1. 100 when the corresponding patch type is maximally interspersed and juxtaposed to other patch types
Patch Cohesion Index (COHESION)	0 ≤ COHESION < 100	1 minus the sum of patch perimeter divided by the sum of patch perimeter times the square root of patch area for patches of the corresponding patch type, divided by 1 minus 1 over the square root of the total number of cells in the landscape. 0 as the proportion of the landscape comprised of the focal class decreases and becomes increasingly subdivided and less physically connected.
Aggregation Index (AI)	0 ≤ AI ≤ 100	number of like adjacencies involving the corresponding class, divided by the maximum possible number of like adjacencies involving the corresponding class, which is achieved when the class is maximally clumped into a single, compact patch. 0 when the focal patch type is maximally disaggregated

MN (Mean) equals the sum, across all patches of the corresponding patch type, of the corresponding patch metric values, divided by the number of patches of the same type.
AM (area-weighted mean) equals the sum, across all patches of the corresponding patch type, of the corresponding patch metric value multiplied by the proportional abundance of the patch [i.e., patch area (m2) divided by the sum of patch areas].

Data analysis

Exploratory analysis of landscape descriptors

The percentage of urban (PLANDU) and the standard deviation of elevation (STDAVE) were log_{10}-transformed to reduce the effect of outliers (Zuur et al. 2010). Correlations between all the explanatory variables (7 class level metrics from the previous section, plus the percentage of urban, agriculture and forest, plus the 2 physical factors) were assessed using the Pearson correlation coefficient. Combinations of significant correlated (r>0.9) co-variables were removed (Table II.3).

Table II.3.
Pearson correlation values for all variables.

	NO_3^--N	U_LOG10	PLANDA	PLANDF	SLO	STDAVE LOG10	COHESION3	IJI1	FRAC_MN1	SHAPE_MN3	AI1	CIRCLE_MN3
U_LOG10	0.51											
PLANDA	0.35	0.67										
PLANDF	-0.4	-0.77	**-0.99**									
SLO	-0.37	-0.73	-0.88	**0.91**								
STDAVE_LOG10	-0.43	-0.53	-0.6	0.61	0.78							
COHESION3	-0.4	-0.5	-0.79	0.78	0.61	0.59						
IJI1	0.05	-0.14	-0.17	0.18	0.47	0.29	-0.12					
FRAC_MN1	-0.41	-0.31	-0.27	0.27	0.18	0.2	0.21	-0.19				
SHAPE_MN3	-0.26	-0.52	-0.54	0.58	0.6	0.66	0.38	-0.01	-0.06			
AI1	-0.22	-0.41	-0.68	0.66	0.6	0.24	0.48	0.28	0.3	0.03		
CIRCLE_MN3	-0.08	0.13	0.27	-0.26	-0.27	0.07	-0.21	-0.59	0.35	0.35	-0.41	
LSI1	0.33	0.84	0.7	-0.76	-0.66	-0.24	-0.32	-0.17	-0.28	-0.39	-0.55	0.19

Significant correlations >0.9 (p>0.05) are in bold
NO_3^--N- nitrate; U_LOG10- log10 of the percentage of urban; PLANDA-percentage of agriculture; PLANDF-percentage of forest class type; SLO-mean sub-watershed slope; STDAVE_LOG10-log10 of the standard deviation of sub-watershed elevation; COHESION3-cohesion index of forest; IJI1-interspersion and juxtaposition index of urban; FRAC_MN1-mean fractal dimension index of urban; SHAPE_MN3-mean shape index of forest; AI1-aggregation index of urban; CIRCLE_MN3-mean related circumscribing circle of forest; LSI1-landscape shape index of urban.

The following co-variables, supporting low redundancy information, constituted the environmental dataset whose relationship with nitrate was assessed: U_LOG10, SLO,

STDAVE_LOG10, COHESION3, IJI1, FRAC_MN1, SHAPE_MN3, AI1, CIRCLE_MN3 and LSI1.A PCA was applied to these variables to identify groups of explanatory variables that are able to account for the variance among the sub-watersheds' characteristics.

Exploratory analysis of nitrate concentrations

Significance of the differences in the concentration of nitrate between seasons and between sub-watersheds was examined running one-way ANOVA tests with Welch's correction to account for unequal variances of nitrate concentrations (Figure II.3).

To evaluate which seasons and sub-watersheds were driving the differences observed with ANOVA, we applied the Games-Howell method (Peters 2015), which is a post-hoc test based on Welchs's correction and suitable for samples with unequal variances.

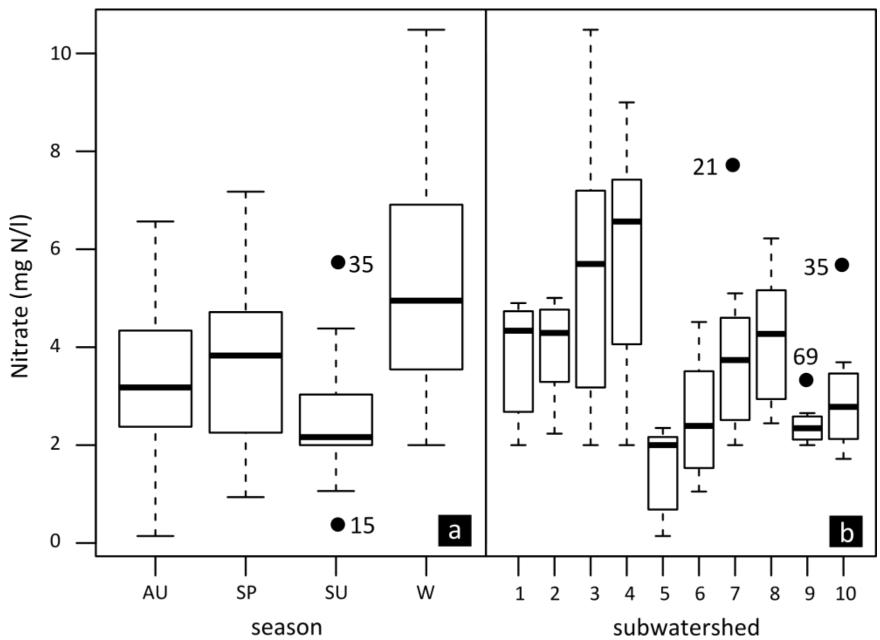


Figure II.3.
Boxplots of nitrate concentrations (mg N/l) by season (a) and subwatershed (b) showing unequal variances across seasons and sub-watersheds. Black dots are outliers. AU – autumn, SP – spring, SU – summer, W – winter.

Modelling the relationship

The exploratory analysis revealed problems with homogeneity but apparently no violation of the independence assumption. To allow for heterogeneity we first fitted a generalized least square (GLS) model to find the optimal variance structure (Zuur et al. 2009). Six variance structures were tested and validated through ANOVA, assuming a) homogeneity; b) heterogeneity between seasons, but homogeneity within seasons; c) homogeneity between seasons but heterogeneity within seasons along the mean sub-watershed slope; d) heterogeneity between seasons and within seasons along the mean sub-watershed slope; e) heterogeneity between seasons and within seasons along the mean sub-watershed slope, but heterogeneity is allowed to differ between seasons; f) homogeneity between seasons and heterogeneity within seasons along the mean sub-watershed slope, where heterogeneity is allowed to differ between seasons. The GLS model implying heterogeneity between seasons and heterogeneity within seasons along the mean sub-watershed slope (SLO) was selected after model validation comparing the AIC and also through visual interpretation (Table II.6).

The exploratory analysis also revealed that the relationship between nitrate concentrations and the explanatory variables was different between seasons. Because we were not interested in the assessment of the exact nature of the relationship between seasons and the explanatory variables, though we did not want to ignore it, we used an interaction between seasons (SEAS) and slope as a random effect. This random structure was compared to an intercept model using only SEAS as random effect.

The variance structure (GLS model) and the random effect (linear mixed model) were selected using the REML estimation method (Zuur et al. 2009) (Table II.6). Trial and error analysis revealed an interaction between the mean sub-watershed slope and the standard deviation of elevation, which was included in the model.

The linear mixed effect model with both the variance structure and the random effect did not reveal violation of homogeneity, independence or normality and therefore the ML estimation method was applied to find the optimal fixed component (Zuur et al. 2009). Package nlme (Pinheiro et al. 2014) from R software (R Core Team 2014) was used. The marginal and conditional R^2 were calculated using function rsquared.glmm (Johnson 2014). The resulting optimal model was again validated to verify whether the underlying assumptions were not violated. Histogram was used to verify the normality of the residuals; homogeneity of variance was evaluated by plotting the normalised residuals versus fitted values and independence was examined by plotting the normalised residuals versus each co-variable.

Results

Landscape metrics selection

The first seven factors of principal components analysis together explain 94% of the variation in the 19 class level metrics (Table II.4). They all have associated eigenvalues greater than one. The first rotated component (RC3) is most correlated with measures of aggregation and shape complexity associated with forest areas and agriculture. It indicates that as forest areas become increasingly subdivided (COHESION3) and less physically connected (FRAC_MN3), the shape complexity of agriculture (CIRCLE_AM2) decreases. The second rotated component (RC2) indicates that as interspersion of urban decreases (IJI1), this class becomes narrower and more elongated (CIRCLE_MN1 and CIRCLE_AM1) while the patch density (PD2) and interspersion (IJI2) of agriculture increases and its shape complexity (FRAC_MN2) decreases.

The third component (RC1) reveals the correlation between urban and forest. As the shape complexity of urban (FRAC_MN1) increases, it becomes more subdivided and less physically connected (COHESION1) or in other words more disaggregated (LSI1). At the same time the landscape connectivity of forest (GYRATE_AM3) increases. The fourth component (RC5) reveals the patterns of forest areas. As the shape complexity of this class increases (SHAPE_MN3), its patches become narrower and more elongated (CIRCLE_MN3). The last three components have low correlations with the class level metrics. RC7 is most correlated with the aggregation index of urban (AI1); RC6 with the mean fractal dimension of forest and the landscape shape index of urban, revealing that the aggregation of urban increases as the patches of forest become less elongated; RC4 is most correlated with the aggregation index of urban (LSI1). AI and LSI are similar measures of aggregation.

Table II.4.
Results of principal components factor analysis and varimax rotation of the first seven factors.

	RC3	**RC2**	**RC1**	**RC5**	**RC7**	**RC6**	**RC4**
	Eigenvalues and proportion of variance explained by principal components analysis before rotation						
Eigenvalue	3.021	3.540	4.553	1.745	1.315	1.596	2.092
Proportion Variance	0.159	0.186	0.24	0.092	0.069	0.084	0.11
	Factor pattern after varimax rotation						
LSI1	-0.272	0.224	**0.743**	-0.187	0.128	0.406	**-0.199***
GYRATE_MN1	0.494	-0.175	-0.244	0.686	-0.130	0.193	-0.112
FRAC_MN1	-0.116	-0.116	**0.947***	0.197			
CIRCLE_MN1	0.153	**0.715**	0.635				
CIRCLE_AM1	0.307	**0.791**	0.468				
IJI1	-0.171	**-0.931***					
COHESION1	0.173	0.179	**0.913**	-0.212			
AI1	0.472	-0.186	-0.527	0.308	**-0.448***	0.288	-0.119
PD2	0.250	**0.868**	0.327	0.174			
FRAC_MN2	0.279	**-0.821**	0.203	0.141	0.226	0.269	
CIRCLE_AM2	**-0.911**	-0.263	0.138	-0.153	-0.110		
IJI2	-0.302	**0.798**	-0.176	0.354	-0.229		
GYRATE_AM3	0.274	-0.259	**0.840**	-0.119	-0.132	0.253	
SHAPE_MN3	0.184	-0.228	-0.145	**0.928***			
FRAC_AM3	**0.861**	-0.196	0.356	0.117	-0.198		
CIRCLE_MN3	-0.316	0.148	0.325	**0.706**	0.214	**0.425***	
CIRCLE_AM3	-0.528	0.672	0.235	-0.322			
IJI3	0.543	**0.727**	0.188	0.105	0.130		
COHESION3	**0.923***	-0.233	0.122	0.203			
	Proportion of variance and cumulative proportion after rotation						
Proportion Variance	0.174	0.160	0.156	0.122	0.114	0.110	0.105
Cumulative Variance	0.174	0.334	0.490	0.612	0.725	0.835	0.940

RC# - Rotated component
Significant correlations >0.9 (p>0.05) are in bold.
* - selected metric
1-Urban, 2-Agriculture, 3-Forest areas, LSI-Landscape shape index
GYRATE_MN-Mean radius of gyration, FRAC_MN-Mean fractal dimension index, CIRCLE_MN-Mean related circumscribing circle, CIRCLE_AM-Area-weighted mean related circumscribing circle, IJI - Interspersion and juxtaposition index, COHESION-Patch cohesion index, AI-Aggregation index, PD-Patch density, GYRATE_AM-Area-weighted mean radius of gyration, SHAPE_MN-Mean shape index, FRAC_AM-Area-weighted mean fractal dimension index

Sub-watersheds characterization

The first four axes of the principal component analysis explain 85% of the variance among the explanatory variables included in our model.

The first axis separates sub-watersheds with high percentage (U_LOG10) and low aggregation of urban areas (LSI1) (sw1, sw2, sw3, sw6) from those with steeper slopes (SLO), high standard deviation of elevation (STDAVE_LOG10), high physical connectedness (COHESION3) and compactness (SHAPE_MN3) of forest areas and with shape complex urban areas (FRAC_MN1) (sw4, sw7, sw8 and sw9).

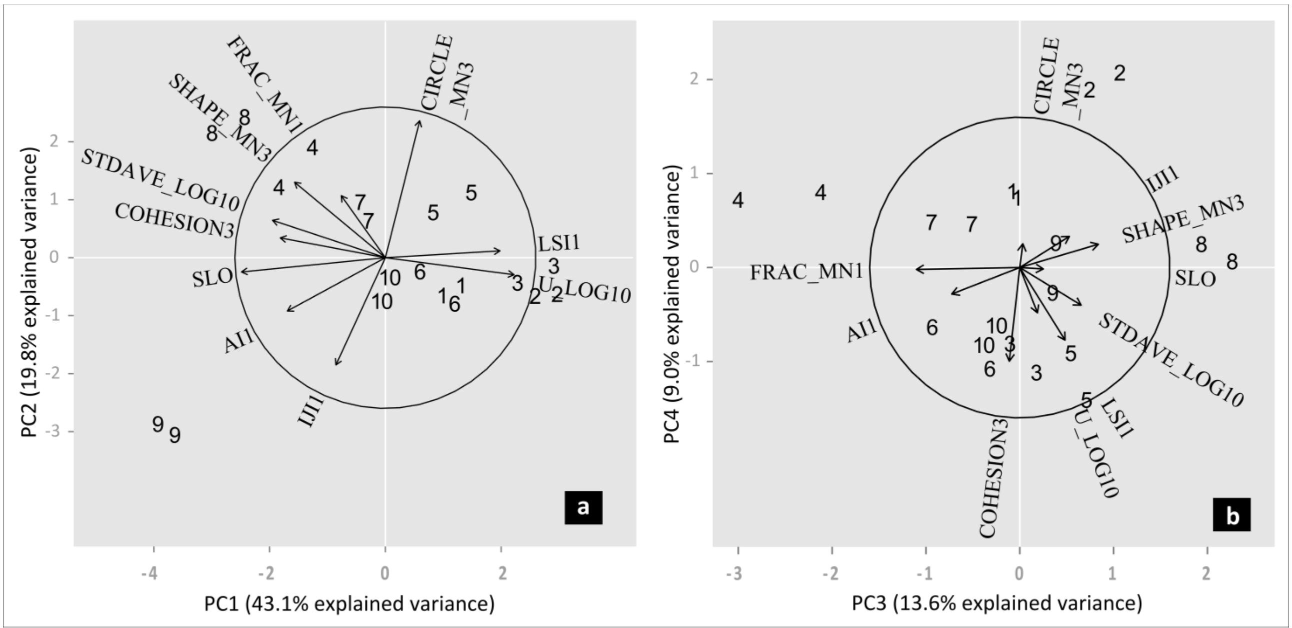


Figure II.4.

PCA of explanatory variables. **a.** First two axes of the principal component analysis for the explanatory variables retained in the final model. **b.** Axes 3 and 4 of the principal component analysis for the explanatory variables retained in the final model. **Legend**: 1 to 10-sub-watershed ID; U_LOG10-log10 of percentage of urban; SLO - mean sub-watershed slope; STDAVE_LOG10- standard deviation of elevation; COHESION3-patch cohesion index of forest; IJI1-interspersion and juxtaposition index of urban; FRAC_MN1-mean fractal dimension index of urban; SHAPE_MN3-mean shape index of forest; AI1-aggregation index of urban; CIRCLE_MN3-mean related circumscribing circle of forest; LSI1-landscape shape index of urban.

The second axis separates mainly sw9 from the rest due to high interspersion (IJI1) and low aggregation of urban areas. This sub-watershed is also related to very high percentage of forest areas and very low percentage of urban, as well as very low number of urban patches (2 patches in 2000 and 3 in 2006) (Figure II.4a). The third axis distinguishes sw8 from sw4. The former has higher STDAVE_LOG10 and lower shape complexity of forest areas (SHAPE_MN3), whereas sw4 has highly convoluted urban areas (FRAC_MN1). The fourth axis separates those sub-watersheds with highly physically connected forest areas (COHESION3), but at the same time with high percentage (U_LOG10) of shape complex urban areas (LSI1) (Figure II.4b).

Nitrate regional and seasonal pattern

Nitrate concentrations in the Mondego river basin, sampled in 2001 and 2006, exhibited seasonal significant differences ($p<0.001$). More precisely, the differences were found between winter and summer ($p<0.001$) and also between winter and autumn ($p<0.05$) (Table II.5). Significant differences were also found between sub-watersheds ($p<0.001$) (Table II.5).

Table II.5.
One-way ANOVA with Welch's correction and pairwise comparisons to assess seasonal and spatial differences.

ANOVA	**F value**	**Pr(>F)**							
SEAS	6.88	<0.001							
SW	6.85	3.747E-05							
Pairwise comparisons (Games Howell t-test for unequal variances)									
SEAS	**AU**	**SP**	**SU**						
SP	0.7								
SU	1.8	2.4							
W	**2.9**	2.2	**4.4**						
SW	**sw1**	**sw2**	**sw3**	**sw4**	**sw5**	**sw6**	**sw7**	**sw8**	**sw9**
sw2	1.64								
sw3	2.28	0.24							
sw4	**4.2**	**3.86**	**4.99**						
sw5	2.08	2.76	**3.59**	1.91					
sw6	0.17	1.37	1.86	3.30	1.74				
sw7	0.61	1.27	1.82	**4.71**	2.53	0.30			
sw8	3.10	3.09	**4.15**	2.60	0.25	2.21	3.52		
sw9	1.26	2.31	3.06	2.71	0.75	1.13	1.76	1.22	
sw10	0.36	1.49	2.13	**5.28**	2.57	0.08	0.32	**4.07**	1.68

Significant values (<0.05) are in bold.
SEAS – seasons; SW – sub-watersheds; SP – spring; SU – summer; W – winter; AU – autumn

Sw2 and sw3 had the highest concentrations of nitrate, with exception for concentrations sampled in the summer, whereas sw4 had consistently the lowest concentrations. Sw5 had the second lowest concentrations in the summer and autumn, whereas sw8 and sw9 had the second lowest concentrations in the winter and spring (Figure II.2).

Landscape-nitrate linkage

From the model selection, the addition of the variance structures significantly decreased the AIC from 314.8549 to 296.1154. The optimal model assumes heterogeneity between seasons and within seasons along the mean sub-watershed slope – GLS[d] (Table II.6).

Table II.6.
Summary of the forward selection referring to the Akaïke Information Criterion (AIC) to reach the final linear mixed-effect model (LME).

		AIC	p-value
Random part selection			
Variance structure	Full GLS [a]	314.8549	
	Full GLS [b]	300.3327	0.0001*
	Full GLS [c]	305.4643	0.0007*
	Full GLS [d]	**296.1154**	<.0001*
	Full GLS [e]	297.1861	<.0001*
	Full GLS [F]	296.9466	<.0001*
Random intercept model	Full LME (seasons)	291.6538	0.0237*
Random intercept and slope model	Full LME (seasons + slope)	**275.6936**	<.0001*
Fixed part selection			
	Full LME (seasons + slope)	273.2724	
	CIRCLE_MN3	266.1911	0.9218
	LSI1	266.1911	0.9494
	IJI1	268.8391	0.5415
	AI1	268.6291	0.1809
	U_LOG10	309.705	<.0001**
	SLO:STDAVE_LOG10	273.8415	0.0106**
	SLO	274.2445	0.0058
	STDAVE_LOG10	309.705	<.0001**
	COHESION3	309.705	<.0001**
	FRAC_MN1	309.705	<.0001**
	SHAPE_MN3	272.2509	0.0177**

[a]-model assuming homogeneity
[b]-model assuming heterogeneity between seasons, but homogeneity within seasons
[c]-model assuming homogeneity between seasons and heterogeneity within seasons along the SLO
[d]-model assuming heterogeneity between seasons and heterogeneity within seasons along the SLO
[e]-model assuming heterogeneity between seasons and heterogeneity within seasons along the SLO, where heterogeneity is allowed to differ between seasons
[f]-model assuming homogeneity between seasons and heterogeneity within seasons along the mean sub-watershed slope, where heterogeneity is allowed to differ between seasons

*Models with significantly lower AIC compared to the full model without random structure. REML estimation method.
**Models with significantly higher AIC compared to the model without non-significant variables, meaning that the null model is worse without the removed explanatory variable(s). ML estimation method.

The random intercept model, where the model intercept is allowed to change between seasons, decreased the AIC to 291.1861. However, validation of GLS[d] showed that the relationship of nitrate and the explanatory variables changed between seasons, and therefore a random effect where the relationship was allowed to change between seasons was added, decreasing the AIC to 275.6936 (Table II.6).

Table II.7.
Synthesis of the results of the final linear mixed-effect model (LME)

Response: NO_3^--N
Fixed component: urban + slope + standard deviation of elevation + cohesion index of forest + mean shape index of forest + aggregation index of urban + slope : standard deviation of elevation interaction
Variance structure: varComb(varIdent(form =~ 1 | SEAS), varPower(form =~ SLO)))
Random effect: ~1 + SLO | SEAS

		Parameter	Std. Error	DF	t-value	p-value	Confidence Intervals Lower	Confidence Intervals Upper
Variance structure								
Different standard deviations per season (~1 | SEAS)	AU	1						
	SP	1.098					0.667	1.808
	SU	1.322					0.787	2.219
	W	1.579					0.965	2.585
Power of variance covariate (~SLO)		-0.276					-0.778	0.225
Random effects								
Random intercept		3.248					1.395	7.563
Random effect (SLO)		0.114					0.048	0.273
Residual standard error		1.839					-1.000	-0.799
Fixed effects								
Intercept		-88.934	187.149	69	-0.475	0.636	-462.287	284.419
U_LOG10		2.407	0.424	69	5.684	0*	1.562	3.252
SLO		-0.964	0.352	69	-2.740	0.0078*	-1.665	-0.262
STDAVE_LOG10		-17.026	3.663	69	-4.649	0*	-24.332	-9.719
COHESION3		1.633	1.971	69	0.828	0.410	-2.300	5.566
SHAPE_MN3		-0.338	0.446	69	-0.757	0.452	-1.228	0.553
AI1		-0.374	0.155	69	-2.420	0.0182*	-0.682	-0.066
SLO : STADVE_LOG10		0.486	0.150	69	3.249	0.0018*	0.188	0.784

* significant coefficients for p-value <0.05. DF – degrees of freedom.

To improve our understanding of the variability in nitrate concentration, a successive exclusion of the explanatory variables was made using the ML estimation. The AIC revealed that CIRCLE_MN3, LSI1, IJI1 and AI1 are not important and they were eliminated (Table II.6). The final model has an AIC of 274.1181 (REML estimation). Together, the fixed and the random factors explain 78% of the variance, whereas the fixed factors alone explain 10%.

The log10 of urban has significant (p>0.05) positive effect on the nitrate concentrations, whereas the aggregation index of urban has significant negative effect (Table II.7). The model also reveals non-significant effects of COHESION3 and SHAPE_MN3.

The effect of SLO and STDAVE_LOG10 on the concentrations on nitrate can only be assessed analysing their interaction (Figure II.5). SLO has a weak negative effect on nitrate concentrations when the sub-watershed standard deviation of elevation is small (Figure II.5a), but the negative effect gets stronger as the standard deviation of elevation increases. According to Brambor et al. (2006) we can assume a statistical significant effect whenever the upper and lower bounds of the confidence interval are both above (or below) the zero line. Because, the confidence intervals around the line are both below the zero line, we assume that SLO has a statistically significant effect on nitrate concentrations for all standard deviation of elevations. In turn, until slope does not exceed 9%. Only above this percentage, the standard deviation has significant negative effect on nitrate concentrations, and it gets stronger as the slope gets steeper (FigureII. 5b).

The results do not allow us to confirm the linear dependence of nitrate concentrations on cohesion and shape index of forest areas (p>0.05).

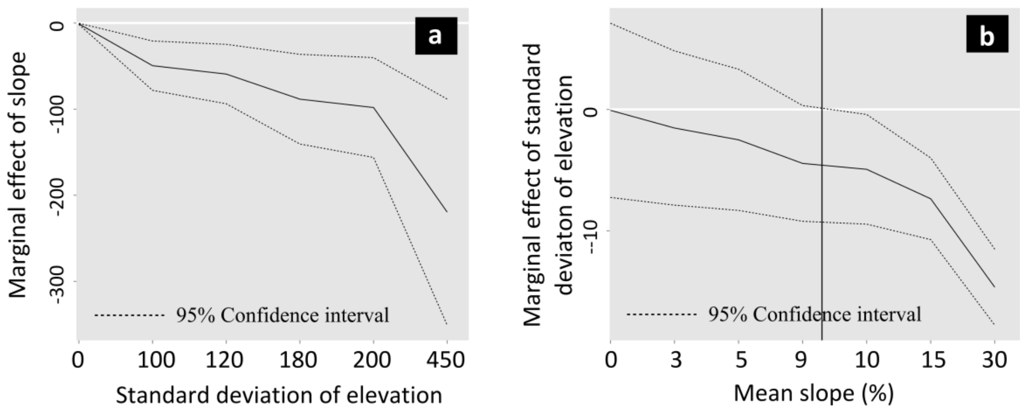


Figure II.5.
a. The marginal effect of the mean sub-watershed slope on nitrate concentrations; **b.** the marginal effect of the standard deviation of elevation on nitrate concentrations. To the left of the vertical black line the lower bound of the confidence interval is negative, whereas the upper bound is positive. To the right of the vertical black line both the lower and upper bounds are negative, indicating that the effect is statistically significant (Brambor et al. 2006).

Discussion

The main goal of our study was to determine the relationship between nitrate and landscape factors. Other factors, such as socio-economic determinants are expected to have a meaningful relationship with stream water quality. Others have successfully established such relationships (Aheam et al. 2005, Carpio and Fath 2011, Chen and Lu 2014). However, including them was out of the scope of this study, as our goal was to assess the relationship between nitrate concentrations and landscape composition and configuration. Results from our analysis are expected to support future work on the effect of landscape changes on stream water quality. The definition of suitable and realistic land use and land cover change scenarios to support water management also relies on previous knowledge on the systems functioning.

Regional and seasonal variability

The seasonal variability in nitrate concentrations observed can be explained by the temporal variation of the hydrological regime (Bernal et al. 2013). Though hydro-meteorological processes have a complex structure in space and time (Modarres and Ouarda 2013), both drought and wet periods have an effect on surface runoff (Langhans et al. 2011, Lange and Haensler 2012), which in turn may have an effect on nutrient transport to surface water bodies (Cooper et al. 2013). Regarding our study, the unprecedented high values of precipitation registered in the winter of 2000/2001, might have had a significant contribution for the differences found in the nitrate values between seasons.

Regarding the observed variability across sub-watersheds it can be explained by the underlying land cover structure which also has an effect on surface runoff. Maetens et al. (2012) concluded that the correlation of annual runoff with length and slope gradient of sampled plots depended on land-use type. Ali et al. (2011) found that the magnitude of peak discharge increment relates to the expansion rate of urban area. Bracken and Croke (2007) concluded that in heterogeneous forest vegetation areas, the loss of patchiness had an effect on the ability of hillslopes to reduce surface runoff. Nunes et al. (2011a), in a study performed in a marginal area of Portugal, concluded that vegetation dynamics is a key factor quantifying and interpreting the hydrological response of land use/cover. More precisely, these authors state that scrub and woodland are better for soil and water conservation, producing less surface runoff, whereas cereal cultivation and tree planting accelerate runoff as a result of reduced anti-erodibility.

In the Mondego river basin, sw2 and sw3, which show the highest concentrations of nitrate (Figure II.2) and significant differences from other sub-watersheds (Table II.5), have the higher percentage of urban, the higher percentage of agricultural land and gentler slopes (Table II.1). Both urban and agriculture, which are the two main land cover sources of nitrate, tend to be in flat areas. In addition, transitions from agriculture to urban are usual (Teixeira et al. 2014) causing landscape fragmentation. Notice that the percentage of like adjacencies of urban (PLADJ1) increases from 2000 to 2006, whereas the percentage of like adjacencies of agriculture (PLADJ2) decreases. As a consequence landscape connectivity of agriculture (GYRATE_MN2) also decreases (Table II.1). On the contrary, sw4, sw7, sw8 and sw9, with lower concentrations of nitrate, show steeper slope and high standard deviation of elevation tending to be occupied by higher percentage of forest (Table II.1). The presence of forest areas is also reflected by their high physical connectedness and compactness, as measured by the cohesion and the shape indexes. Agriculture and urban tend to avoid these areas (Teixeira et al. 2014).

Effect of landscape on nitrate

The results show the linkage between landscape physical characteristics and stream nitrate in a Mediterranean watershed in Central Portugal. Only main land cover types were distinguished: urban, agriculture and forest areas. Urban and agriculture are known drivers of stream water quality and therefore there were previous expectations to find a clear relationship between these land cover types and stream nitrate concentrations. The linear mixed model defined was able to establish the relationship, but the marginal effects found were only able to explain 10% of the nitrate concentration variability. We assumed this was related to three factors: a) the resolution of CORINE Land cover maps, which might be too coarse to detect the effects of land cover on water quality (Uuemaa et al. 2005); b) the land cover classification, which only distinguished 3 class types. More detailed class types, would probably be capable of distinguishing different sources of nitrate, but this was out of the scope of our study; c) the small dataset used. Only 10 sub-watersheds and two years, with high land cover persistence (Teixeira et al. 2014). The mixed model however, with its fixed and random part, explains 78% of the variability. It was assumed since the beginning that the seasonal pattern would be considered a random effect, in order to clearly separate the landscape effect, more persistent, from the climate effect, more unforeseeable.

Results suggest that the interaction between the standard deviation of elevation and slope plays an important role in the Mondego river basin. Steeper slopes and higher variation of elevation tend to have stronger negative effects on nitrate concentrations. Steeper slopes and higher elevation (more variation of elevation presupposes high and lowland in the same sub-watershed) are highly correlated with forest areas (Table II.3). Forest areas, which include mostly forests, reduce surface runoff and provide protection from nutrient and pesticide flow due to the extensive and deep root systems responsible for soils with high infiltration rates (Neary et al. 2009).

In contrast, flat areas and lowland are highly correlated with agricultural fields (Table II.3), which are diffuse sources of nitrate to water bodies due to the agricultural use of fertilizers (Ferreira et al. 2004). The selection of a group of variables with low correlation, to include in the mixed model, eliminated *a priori* all the variables that characterised this class. However, the Mondego river basin is known for its extensive agricultural areas, due to rice, corn and other cereals cultivations (Teixeira et al. 2014). The relationship of slope and elevation with nitrate concentrations reflects the pattern of occupation of both agricultural and forest areas.

Urban is also associated with flat areas, though its correlation is lower as it is more dispersed throughout the landscape topography. The expected positive effect of urban on nitrate might be due to domestic and/or industrial drainage and sewage. Similar relationships have been found in other locations (Chen and Lu 2014). But the relationship found with nitrate concentrations is weak (log 10 of model coefficient = 0.012, p-value=0). One possible explanation for the weak relationship could be the presence of wastewater treatment plants (WWTP). Aheam et al. (2005) had previously concluded that insufficient wastewater treatment results in a strong relationship between nitrate and urban areas. This means that, unless the point source is taken into account, results from significant urban coefficients could possibly give spurious results or weak relationships (Ahearn et al. 2005). But this does not occur in the Mondego River basin, where less than 32% of the population was served by WWTP (In 2010, 443 WWTP served about 32% of the population and there is evidence that the number of WWTP was lower in 2001 and 2006, though the exact number is not known) (PGBH dos Rios Vouga, Mondego e Lis 2012). We believe that the effect of imperviousness might be influencing the effect of urban. Ferreira et al. (2015, unpublished) found that a highly urbanised sub-catchment within the Mondego basin displayed higher concentrations of nitrate and the authors believe that the nearby agricultural fields are the actual nitrate source, but they have also found that specific loads of nitrate increased linearly with percentage of impermeable area.

The negative effect of the aggregation index of urban on the concentrations of nitrate indicates that sub-watersheds with aggregated urban patches tend to have lower nitrate concentrations. The interpretation of AI cannot be dissociated from the total area occupied by the focal class. High aggregation index and high percentage of urban might indicate that we are in the presence of a major urban centre. This could in turn, mean that WWTP were present, reducing the effect of urban, but could also mean large impermeable areas and thus large urban effects as a result of drainage, especially in the rainy seasons (Chu et al. 2013). This is not the case for the Mondego basin. Sw4 and sw9 have the highest AI but also the lowest percentage of urban area. In this case, low occupation also means high AI because disaggregation is less probable to occur. The capacity to detect aggregation is scale-dependent, as higher resolutions are more capable of detecting disaggregated patches.

The percentage of urban occurs in association (Table II.3) with higher urban patch density (PD1), landscape connectivity (GYRATE_AM1) and shape complexity (SHAPE_AM1). PD is a measure of fragmentation and sub-watersheds with higher patch density of urban, which is the case for sw2 and sw3, tend to have more interspersed agricultural areas (IJI2) and more disaggregated (PLADJ3, AI3) and less connected (CONTIG3) forest areas, due to the transitions from agriculture and from forest to urban (Texeira et al. 2014). Urban development fragments, isolates and degrades forest habitats, disrupting hydrological systems (Braud et al. 2013) and modifying nutrient cycling, with impact on stream water quality (Mejía and Moglen 2010, Nie et al. 2011).

The linkage between nitrate, as well as other water quality parameters, with landscape configuration metrics is still not very well understood as different combinations of land use classes are expected to have different impacts on water. Take PD as an example, Huang et al. (2013) found that PD at the landscape level, in the Jiulong River watershed, China, was negatively correlated with $NH4^+$-N. On the other hand, Wang et al. (2014) found that PD, in the Jinjing catchment, China, was significantly positively correlated to all N nutrient concentrations in the stream water, both at the landscape and class level. Johnson et al. (1997) found that PD at the landscape level, in the Saginaw Bay catchment in Michigan, USA, was negatively correlated with total nitrogen in summer. Uuemaa et al. (2005), in turn, recognised that total nitrate depends on both land use and landscape metrics, but these authors were not able to determine the detailed influence of either factor on total-N runoff, in the Porijögi River catchment. Landscape configuration, however, should have a role in determining the effect of each class on water quality, but its effects are expected to be more evident for studies analysing landscapes with similar composition of land use classes.

Landscape metrics' selection

There is an array of landscape metrics to characterise landscape composition and configuration, of which the percentage occupied by each class is probably the most obvious and the most used. It is well known that some of the available metrics are highly correlated and some sort of selection criteria is needed to eliminate groups of correlated metrics (Ritters et al. 1995, Hargis et al. 1998). The decision behind metrics selection is, nevertheless, often an arbitrary decision with unrevealed selection criteria (e.g. Sun et al. 2014). The selection of metrics, however, should not be neglected since it has an influence on model results. To guarantee the optimal group of metrics that best characterise the study area we consider that selection should be performed based on the region under study and not on previous work and should be made following some methodology that clearly identifies the reasons for variable reduction. Of the three class level metrics retained in our model, only one shows a significant effect, AI1, and it is providing more information on urban occupation than actually on its aggregation. This occurs because urban occupies a very small percentage of the watershed.

Conclusion

A mixed model was applied to find the relationship between stream nitrate concentrations and sub-watersheds' physical characteristics and landscape structure. Only main land cover types were distinguished: urban, agriculture and forest areas. The model was able to establish the anticipated relationships, though the selected variables only explained 10% of the variability observed. Slope and elevation showed conditional negative effect on nitrate concentrations and revealed the general configuration of landscape composition: urban and agriculture, which are nitrate sources, tend to occupy flat areas, whereas forest areas, which provide protection from nutrient flow, tend to occupy steeper slopes and highland. Of the configuration class level metrics included in the analysis, only aggregation index of urban played a significant role in the final model, and it revealed to be related to urban percentage. The influence of landscape configuration metrics, though observed by others, was not obvious in this study. Though the model validation indicated satisfactory results, future analysis evaluating the effect of metrics selection could be performed.

Though our findings are meaningful for watershed-scale water quality management, the results should be taken with caution due to gaps in water quality data. We did not manage to compile data for the most downstream catchment, within our time interval, as well as complete datasets for other water quality parameters. Similar problems will emerge from more recent datasets, as the Portuguese national monitoring program was suspended for a few months during the last recent years.

Chapter III

Systematic processes of land use/land cover change to identify relevant driving forces: Implications on water quality.

Abstract

Land use and land cover (LUC) are driving forces that potentially exert pressures on water bodies, which are most commonly quantified by simply obtained aggregated data. However, this is insufficient to detect the drivers that arise from the landscape change itself. To achieve this objective one must distinguish between random and systematic transitions and identify the transitions that show strong signals of change, since these will make it possible to identify the transitions that have evolved due to population growth, industrial expansion and/or changes in land management policies. Our goal is to describe a method to characterize driving forces both from LUC and dominant LUC changes, recognizing that the presence of certain LUC classes as well as the processes of transition to other uses are both sources of stress with potential effects on the condition of water bodies. This paper first quantifies the driving forces from LUC and also from processes of LUC change for three nested regions within the Mondego river basin in 1990, 2000 and 2006. It then discusses the implications for the environmental water body condition and management policies. The fingerprint left on the landscape by some of the dominant changes found, such as urbanization and industrial expansion, is, as expected, low due to their proportion in the geographic regions under study, yet their magnitude of change and consistency reveal strong signals of change regarding the pressures acting in the system. Assessing dominant LUC changes is vital for a comprehensive study of driving forces with potential impacts on water condition.

Keywords: Water framework directive, DPSIR, LUC, Systematic transitions, Mondego river basin, Portugal

Introduction

Under the assumption that the relationship between humans and ecosystems relies on a complex, dynamic web of interactions, a change in the human condition might serve to change ecosystems both directly and indirectly. An understanding of such interactions can be described in terms of “drivers” of ecosystem change. Drivers, or driving forces, are any natural (e.g. rainfall, temperature) or human-induced factors that cause a change in an ecosystem.

Human-induced driving forces, in particular, are human activities and economic sectors responsible for pressures acting on an ecosystem (Elliott 2002), whose identification is essential to evaluate the current and the potential impacts of human activity on the status of surface waters (Carey et al. 2011, Lowicki 2012, Zhou et al. 2012). Moreover, the identification and understanding of those driving forces are essential to the design of interventions that enhance positive and minimize negative impacts on the ecosystem. The water framework directive (WFD) itself presented a guidance document for the pressure and impact analysis adopting the Driver, Pressure, State, Impact, Response (DPSIR) framework, according to which information on drivers is highly necessary in order to identify pressures and their environmental effect (IMPRESS 2003).

Land use and land cover (LUC) are important drivers of change to biogeochemical cycles, biodiversity and water quality. Li et al. (2011) found that changes in global land vegetation affected the silicon (Si) uptake by land biomass, causing changes in Si river inputs. The impacts of such land use changes on functional guilds of benthic invertebrates were then evaluated through Eco-Exergy (Li et al. 2013). Wang et al. (2013) assessed spatial–temporal water quality variations, identifying LUC sources of water pollution. Measuring LUC and its rates and patterns of change requires a spatial–temporal assessment of LUC data, which is most commonly provided through the analysis of transitional matrices (Lu et al. 2004). The traditional analysis of transitional matrices provides information on the most prominent landscape changes, but is insufficient for distinguishing between random and systematic transitions (Pontius et al. 2004). Random transitions are influenced by coincidental or unique processes of change, whereas systematic transitions are those that tend to evolve in a consistent and/or progressive manner due to population growth, industrial or commercial expansion, or changes in land management policies (Braimoh 2006, Lambin et al. 2003). The identification of systematic transitions makes it possible to focus on the strongest signals of landscape change and ultimately to link pattern to process (Manandhar et al 2010, Pontius et al. 2004). Pontius et al. (2004) proposed a methodology to assess inter-category transitions based on systematic transitions while accounting for land persistence. Among other applications, such methodology has been applied to explore the impacts of land use on regional water balance and revegetation strategies (Versace et al. 2008); to assess landscape dynamism to be considered in models of LUC change (Lira et al. 2012); to link patterns to processes of LUC changes based on levels of intensity analysis (Huang et al. 2012); and to detect the dynamic linkage between landscape characteristics and water quality evaluating the statistical relationship between landscape metrics and physical–chemical parameters (Huang et al. 2013). It has also been used to provide a set of pressures on biodiversity derived from LUC changes covering a metropolitan area in Chile (Rojas et al. 2013) and for a spatial–temporal land use change analysis in a peri-urban area within the same river basin used as a case study in this paper (Tavares et al. 2012). Yet, as far as we know, it has never been used as a tool to explicitly identify and quantify drivers of environmental change linked both to LUC and LUC change with potential effects on the condition of water bodies.

Driving forces linked to LUC are regularly quantified by the surface occupied by a specified class (IMPRESS 2003). However, this method might be insufficient since a LUC class with a small area might leave a larger than expected fingerprint on the landscape if we also consider its rate of transition. Moreover, this method provides information on the drivers linked to the sole presence of the specified LUC, but is unable to provide information on the drivers linked to the process of land transition. Our major goal is therefore to characterize driving forces linked both to LUC and LUC change, with potential impacts on the condition of water bodies. Ultimately, our study intends to contribute to the assessment of land use and land use change in the scope of the WFD, providing instruments to improve the analysis of pressures and impacts (IMPRESS 2003). In this study, LUC is characterized and quantified by exploring traditional cross-tabulation matrices over a sixteen-year period (1990–2006), in three nested regions in the Mondego river basin, Portugal. From these results, the role of each LUC class as a driving force of environmental change with impact on the state of water bodies is discussed. Our study then follows the methodology proposed by Pontius et al. (2004) and further extended by Braimoh (2006) as a mean to detect systematic transitions and dominant signals of landscape change, providing the basis for the identification of dominant driving forces from processes of LUC change.

Methodology

Study Site

The Mondego river basin is located in the central region of Portugal, Europe (Figure II.1). With an area of 6658 km2 and a NE–SW orientation, it encompasses 36 municipalities with an estimated population of 165 inhabitants per km2 (INE 2014). Coimbra and Figueira da Foz are two of the most populated municipalities and, because they have grown along the river margins, they play an important role within the Mondego river dynamics.

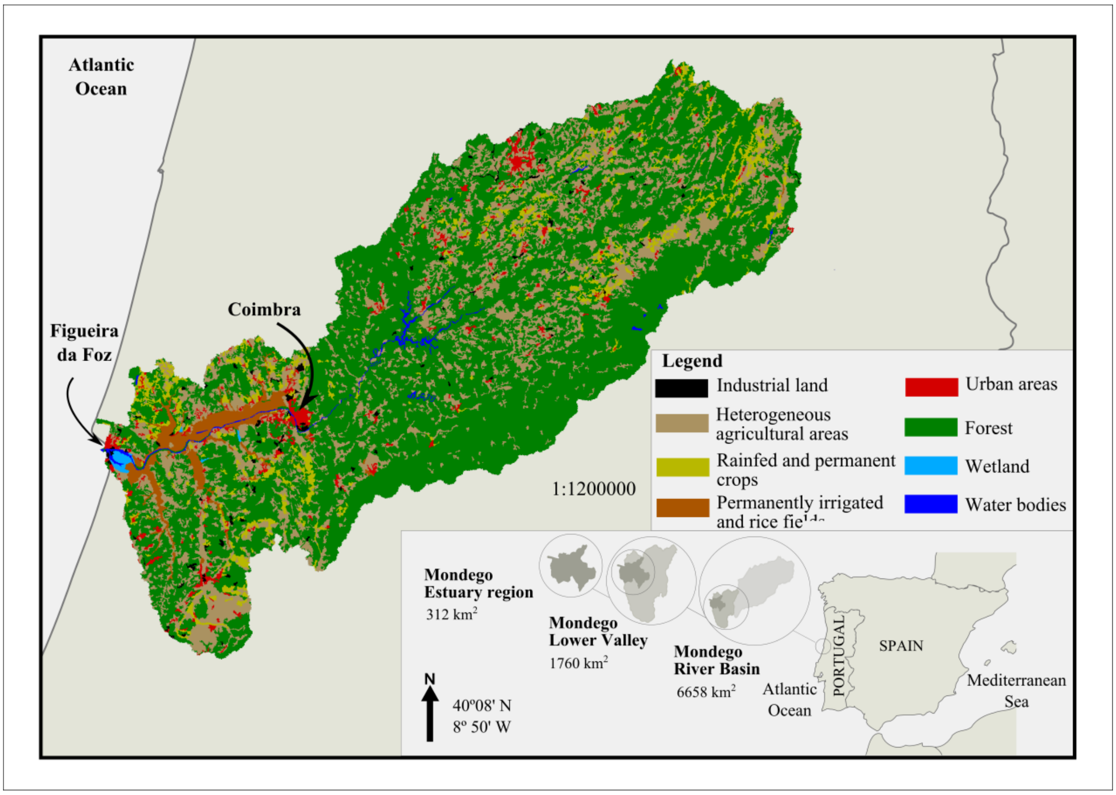


Figure II.1.
Location of the study area and land use/land cover classes. Geographic location of the three nested regions: Mondego river basin, lower valley and estuary region. Land use/land cover reclassification based on the 2006 inventory of the CORINE Land Cover project.

Study Regions

A first analysis of the study area indicated that a reduced number of LUC classes together occupy more than 88% of the total river basin area. To overcome the dominance of these LUC classes, as well as clumpiness, which could mask relevant driving forces acting in the lower part of the Mondego river basin, analysis was performed for three nested regions within the Mondego river basin: the river basin itself, the Mondego lower valley and the Mondego estuary region (Figure II.1). Identical regions have been previously employed for studies under this same system (Pinto et al. 2010). The Mondego lower valley comprises the subwatersheds draining into the Mondego river and its tributaries, downstream from the city of Coimbra. This region is integrated in the Lower Mondego NUTSIII subregion (EUROSTAT). The Mondego estuary region comprises the subwatersheds draining into the Mondego estuary, plus adjacent subwatersheds draining into the Mondego estuary tributaries. Subwatersheds were defined using the watershed delineation plugin (Moya 2011) available on MapWindow GIS (version 4.8.6) and were based on SRTM 90m digital elevation data (version 4.1.) derived from USGS/NASA SRTM data (Jarvis et al. 2008). The Mondego river basin limits available from the Portuguese Environment Agency (APA) were used as a focusing mask and a threshold of 25 km2 was used for network delineation. The resulting stream net coincides with the hydrographic network available also from APA, except for floodplains in the lower part of the Mondego river. The current hydrographic network is not consistent with the expected bounded hydrologic systems, because the Mondego river has undergone several regularization works since the sixteenth century, which have modified its lower part (PBH do Rio Mondego 1998).

LUC reclassification

To quantify LUC driving forces we defined six LUC classes based on water retention capacity and potential pressures on water bodies, plus two more classes characterizing the water environment (water bodies and wetlands). Of the six LUC classes, two characterize artificial surfaces (urban areas and industrial land), three characterize agricultural areas (rainfed and permanent crops; permanently irrigated land and rice fields; heterogeneous agricultural areas) and the last one characterizes forests (see Table A.III.1 for a more detailed description of each class). Hereafter these classes are referred to as urban, industrial land, rainfed, rice fields, heterogeneous, forest, wetlands and water bodies. The analysis was based on CORINE Land Cover raster data, resolution 100 × 100m, for the 1990, 2000 and 2006 inventories. CORINE was selected for the analysis because it is a ready-to-use dataset, allowing for replications and comparison to other European sites. The latest versions available, from May 2012, were used (EEA 2012). The 44 classes of CORINE were reclassified using Quantum GIS 1.8.0 'Lisboa' (OSGeo4W).

LUC characterization

LUC was characterized by calculating the proportion of total landscape occupied by each class in 1990, 2000 and 2006, in the three nested regions. In order to obtain this information, transition matrices were built. Transition matrices are tables displaying classes of time period 1 in rows and classes of time period 2 in columns. The Total column shows the proportion of a class in time period 1, while the Total row shows the proportion of a class in time period 2. Entries on the diagonal of the matrix indicate the proportion of landscape that remained unchanged during the time period analyzed, whereas the remaining cells indicate the proportion of landscape surface of a given LUC class that changed to a different class. This means that off-diagonal entries indicate a transition from a given class in time period 1 to a different class in time period 2. For a better characterization of landscape changes, the annual rate of landscape change, the net change and swap were also calculated. The annual rate of landscape change measures the amount of LUC change per year, for each time interval. It was calculated following the approach proposed by Puyravaud (2003) and described in Eq. (1)

$$r = (1 / t_2 - t_1) \ln (A_2/A_1) \qquad (1)$$

where A_1 and A_2 are the landscape cover of a given LUC class at time t_1 and t_2, respectively. The net change measures the definite change between two periods of time (Pontius et al. 2004). It was determined by calculating the difference between the Total column and the Total row. The swapping component of landscape change, i.e., the proportion of a given class that changes location, while the total surface area remains the same, was calculated as the difference between the total change, i.e. gain plus loss, and the absolute value of net change. The relevance of swap lies in the fact that a lack of net change does not necessarily mean a lack of change in the landscape.

In order to assess the total disagreement between maps of two different time periods, quantity disagreement and allocation disagreement were also determined. Total disagreement provides a measure of the total differences between two maps. Quantity disagreement measures the amount of difference in the proportions of the classes. Allocation disagreement measures the amount of difference in the spatial allocation of the classes, given the proportions of the classes (Pontius and Millones 2011). Such parameters were calculated in terms of number of cells, which represent our units of observation.

Systematic transitions

LUC systematic transitions were identified by examining off diagonal entries given any level of landscape's degree of persistence and taking into consideration the size of each LUC class. It was assumed that the gain of a given class (difference between the column totals and the unchanged landscape) and its proportion at time 2 was fixed. Likewise, it was assumed that the loss of the same class (difference between row totals and unchanged landscape) and its proportion at time 1 was also fixed. These assumptions, together with the assumption that expected and observed unchanged areas are equal, allowed us to calculate expected values under random processes of gain and loss. The difference between the observed and the expected values provides information on the rate that a given class is to gain – or lose – randomly. This difference is zero if gains – or losses – occurred randomly, and it is not near zero if gains – or losses – are systematic transitions. The magnitude of this difference quantifies the systematic patterns of change and “indicates the size of the fingerprint left on the landscape due to a systematic transition” (Pontius et al. 2004). To simplify the interpretation of the results, only differences higher than 1%were considered relevant and further discussed. Furthermore, the ratio between this difference and the expected value provides information on the rate that a class gains – or loses – compared to the rate that would be expected if the same class was to gain – or lose – randomly. In this case, the magnitude of the ratio “indicates the strength of the systematic transition”. Notice that the factors that promote gains in LUC are most likely different from those that lead to losses and, for this reason, results from the analysis of gains can be different from the analysis of losses. (For further information on systematic transition methodology, please see Pontius et al. 2004.)

Dominant processes of LUC change

Although systematic transitions identify non-random landscape changes, a specific systematic transition might still not be of special importance. Relevance of a systematic transition between two explicit LUC classes is only acknowledged when, for instance, one or several patches of a class of 2006 consistently gain surface from patches of a class of 2000; while in the same time period, one or several patches of the class of 2000 consistently lose surface to patches of that same class of 2006 (Braimoh 2006). Once these dominant signals of change were identified, they were compared to 10 potential processes of LUC change (Table A.III.2) which allowed us to identify the dominant processes of LUC change. Processes of LUC change are broad categories of landscape change and not the actual causes of land modifications, as considered in previous studies (Huang et al. 2012, Manandhar et al. 2010). The approach followed in this study is more appropriate for the identification of driving forces linked to LUC change. The potential processes of LUC change were defined based on the reclassification of our 8 classes and mainly taking into consideration their potential effect on runoff. Due to the increase of impervious areas and reduction of evapotranspiration and water infiltration, runoff intensification is of special concern because it may cause changes in water flow and chemistry, increase sedimentation, and cause other impacts on biological communities (EPA 2005, Zhang et al. 2007). The effect on runoff and other potential pressures can be deduced from the description of each LUC available on Table A.III.1. The potential processes of LUC change are urbanization, industrial expansion, afforestation, agricultural shift, other agricultural changes, agriculturalization, deforestation, land restoration, siltation/deposition and dredging/erosion (see Table A.II.2 for potential processes of LUC change according to the type of transition). Urbanization refers to the expansion of urban settlement at the expense of other LUC types. Industrial expansion refers to the increase of industrial settlement at the expense of other LUC types. Afforestation refers to conversion from any other LUC type to forest cover. Agricultural shift refers to conversions between rainfed class and rice fields class. Other agricultural changes refer to conversions between one of these two classes and heterogeneous. Agriculturalization refers to the expansion of agricultural areas at the expense of non-agricultural LUC types, except forest. Deforestation refers to the expansion of agricultural areas at the expense of forest. The expansion of agricultural areas from artificial surfaces (agriculturalization) and forest (deforestation) was distinguished because the impact on the hydrologic cycle is potentially different. Land restoration refers to the expansion of wetlands or water bodies at the expense of other LUC types. Siltation/deposition refers to the expansion of wetlands at the expense of water bodies and dredging/erosion refers to the loss of wetlands with subsequent increase in water body surface.

Results

LUC characterization

The proportion of total landscape occupied by each class characterizes and quantifies LUC in the Mondego river basin, Mondego lower valley and Mondego estuary region, in 1990, 2000 and 2006 (Table III.1). All LUC classes show differences in the proportion occupied in each regional scale no matter the year considered. The largest class in all three regions is forest, the surface of which decreases from around 64% in the river basin, to 47% in the lower valley and 36% in the estuary region. The second-largest class – heterogeneous – in turn occupies an area between 23% and 29%, with only slight differences between regions. On the other hand, the proportion occupied by rice fields increases from around 2.5% in the river basin to approximately 9% in the lower valley and 21% in the Mondego estuary. Artificial surfaces, i.e., urban and industrial land, occupy the lowest landscape proportion in all three regions, with exception for the classes characterizing the water environment. These, as expected, occupy larger proportions downstream from the river basin. Regarding the differences between years, total disagreement is higher in the last six years (2000–2006) than in the first ten years (1990–2000), in all three regions (Figure III.2).

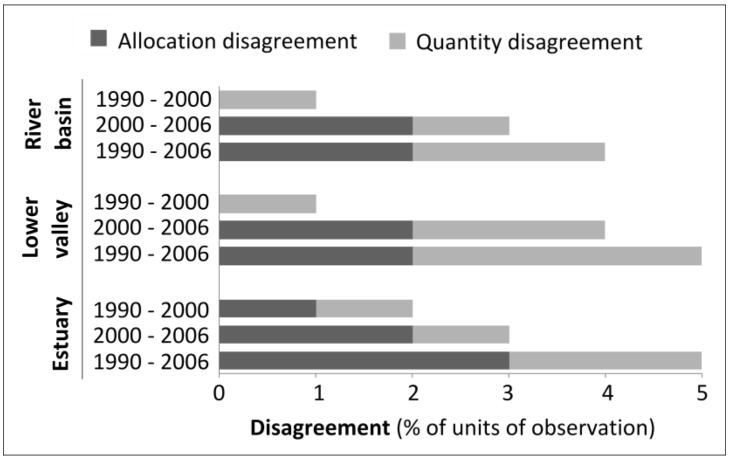


Figure III.2.
Total disagreement between LULC maps. Total disagreement expressed by allocation disagreement and quantity disagreement measured for three periods of time and three nested regions in the Mondego river basin.

Also, from 1990 until 2000, only quantity disagreement contributes to total disagreement in the river basin and lower valley, whereas the estuary region shows both quantity and allocation disagreement. Overall, total disagreement does not exceed 5% of the total units of observation. This is in accordance with the high levels of persistence and low annual rates of land change observed for all classes, in all periods of time, in all regions (Table III.1). There are some exceptions, though, which are related to artificial surfaces. These show positive and high annual rates of land change during the sixteen-year period. Changes in industrial land took place mostly during the first ten years, while changes in urban took place mostly during the last six years (Table III.1). Despite the low annual rates of change, rainfed, heterogeneous and forest classes show the highest swap, suggesting that these classes, together with artificial surfaces, are the most dynamic LUC classes (Table III.1).

Systematic transitions

When a LUC class systematically gains surface from, or loses surface to, another class, the difference between the observed and the expected values is different from zero. Results show that the smaller the geographic area under analysis, the higher the number of different systematic transitions found (Tables III.2a–III.4b). Furthermore, in some cases, for the same geographic area, a transition is only systematic for one time period: for only the first ten years or only the last six, or only when considering the sixteen-year time period.

Systematic transitions observed from other classes to urban are consistent with its positive net change, low swap and high annual rates of change (Table III.1). Comparing the observed with the expected transitions based on a random process of loss, heterogeneous, industrial land, rainfed, and forest, they all attracted, at least in the lower valley and estuary regions, a systematic replacement by urban. At the same time, and based on a random process of gain, urban gained surface by systematically replacing heterogeneous (all three regions), rice fields (lower valley and estuary) and industrial land (estuary).Moreover, from 2000 to 2006, the rate of transition from industrial land was over 24 times the rates expected if urban were to gain randomly (Table III.4a).

Table III.1

Land use/land cover characterization and main landscape changes occurred in the Mondego river basin, lower valley and estuary region, from 1990 to 2006.

Region	LULC Class[a]	Total Area (% of landscape)			Persistence (% of class in time 1)			r[b] (x100)			Loss (% of landscape)			Gain (% of landscape)			Total Change (% of landscape)			Absolute value of Net Change[c]			Swap		
	Time Period	1990	2000	2006	1990-2000	2000-2006	1990-2006	1990-2000	2000-2006	1990-2006	1990-2000	2000-2006	1990-2006	1990-2000	2000-2006	1990-2006	1990-2000	2000-2006	1990-2006	1990-2000	2000-2006	1990-2006	1990-2000	2000-2006	1990-2006
River Basin	U	1.155	1.478	2.335	99.83	98.05	98	2.46	7.63	4.4	0	0.03	0.02	0.32	0.89	1.2	0.33	0.91	1.23	0.32	0.86	1.18	0	0.06	0.05
	I	0.22	0.56	0.675	94.1	87.94	82.36	9.36	3.13	7.02	0.01	0.07	0.04	0.35	0.18	0.49	0.37	0.25	0.53	0.34	0.12	0.46	0.03	0.13	0.08
	R	5.615	5.942	5.476	98.86	89.14	88.86	0.57	-1.4	-0.2	0.06	0.65	0.63	0.39	0.18	0.49	0.45	0.82	1.11	0.33	0.47	0.14	0.13	1.29	1.25
	P	2.526	2.536	2.528	99.58	97.86	97.42	0.04	-0.1	0	0.01	0.05	0.07	0.02	0.05	0.07	0.03	0.1	0.13	0.01	0.01	0.00	0.02	0.11	0.13
	H	25.26	24.72	24.01	97.37	94.57	92.18	-0.2	-0.5	-0.3	0.66	1.34	1.98	0.13	0.63	0.73	0.79	1.97	2.7	0.53	0.71	1.25	1.33	2.69	3.95
	F	64.53	63.99	64.19	99.1	99.12	98.27	-0.1	0.05	-0	0.58	0.56	1.11	0.05	0.76	0.78	0.63	1.32	1.89	0.53	0.20	0.33	1.17	1.12	2.23
	W	0.186	0.194	0.194	97.73	100	97.73	0.46	0	0.29	0	0	0	0.01	0	0.01	0.02	0	0.02	0.01	0	0.01	0.01	0	0.01
	WB	0.517	0.575	0.592	98.98	99.66	98.63	1.05	0.5	0.85	0.01	0	0.01	0.06	0.02	0.08	0.07	0.02	0.09	0.06	0.02	0.07	0.01	0	0.01
	Total	100	100	100	98.65	97.3	96.15				1.35	2.7	3.85	1.35	2.7	3.85	1.35	2.7	3.85	1.07	1.19	1.72	1.35	2.7	3.85
Lower Valley	U	2.429	2.793	4.582	99.69	98.03	97.76	1.4	8.25	3.97	0.01	0.06	0.05	0.37	1.84	2.21	0.38	1.9	2.26	0.36	1.79	2.15	0.01	0.11	0.11
	I	0.593	1.298	1.471	96.03	89.12	89.07	7.82	2.09	5.67	0.02	0.14	0.06	0.73	0.31	0.94	0.75	0.46	1.01	0.70	0.17	0.88	0.05	0.28	0.13
	R	9.378	9.258	8.654	97.9	90.63	88.73	-0.1	-1.1	-0.5	0.2	0.87	1.06	0.08	0.26	0.33	0.27	1.13	1.39	0.12	0.60	0.72	0.15	0.53	0.67
	P	9.31	9.349	9.31	99.56	97.83	97.39	0.04	-0.1	0	0.04	0.2	0.24	0.08	0.16	0.24	0.12	0.37	0.49	0.04	0.04	0.00	0.08	0.33	0.49
	H	29.29	28.92	27.23	98.37	91.66	90.21	-0.1	-1	-0.5	0.48	2.41	2.87	0.11	0.73	0.81	0.58	3.14	3.68	0.37	1.69	2.06	0.21	1.45	1.63
	F	47.56	46.94	47.28	98.48	99.12	97.65	-0.1	0.12	-0	0.72	0.41	1.12	0.09	0.76	0.83	0.82	1.17	1.95	0.63	0.34	0.29	0.19	0.83	1.67
	W	0.708	0.741	0.741	97.73	100	97.73	0.46	0	0.29	0.02	0	0.02	0.05	0	0.05	0.07	0	0.07	0.03	0	0.03	0.03	0	0.03
	WB	0.727	0.707	0.729	97.24	99.76	97	-0.3	0.51	0.01	0.02	0	0.02	0	0.02	0.02	0.02	0.03	0.05	0.02	0.02	0.00	0	0	0.04
	Total	100	100	100	98.49	95.91	94.55				1.51	4.09	5.45	1.51	4.09	5.45	1.51	4.09	5.45	1.14	2.33	3.07	0.37	1.77	2.38
Estuary	U	3.673	4	4.602	100	96.76	96.56	0.85	2.34	0.85	0	0.13	0.13	0.33	0.73	1.06	0.33	0.86	1.18	0.33	0.60	0.33	0	0.26	0
	I	0.913	1.469	1.246	100	74.01	79.43	4.76	-2.7	4.76	0	0.38	0.19	0.56	0.16	0.52	0.56	0.54	0.71	0.56	0.22	0.56	0	0.32	0
	R	7.379	7.318	6.224	99.17	82.53	81.84	-0.1	-2.7	-0.1	0.06	1.28	1.34	0	0.18	0.18	0.06	1.46	1.52	0.06	1.09	0.06	0	0.37	0
	P	21.4	21.56	21.61	99.23	99.94	99.17	0.07	0.04	0.07	0.17	0.01	0.18	0.32	0.06	0.38	0.48	0.08	0.56	0.15	0.05	0.15	0.33	0.03	0.33
	H	23.91	23.4	23.79	97.74	96.31	94.15	-0.2	0.28	-0.2	0.54	0.86	1.4	0.03	1.26	1.28	0.57	2.12	2.68	0.51	0.39	0.51	0.06	1.73	0.06
	F	36.45	35.91	36.18	97.81	98.95	96.84	-0.1	0.12	-0.1	0.8	0.38	1.15	0.26	0.65	0.88	1.06	1.03	2.04	0.54	0.27	0.54	0.52	0.76	0.52
	W	3.628	3.816	3.816	97.5	100	97.5	0.5	0	0.5	0.09	0	0.09	0.28	0	0.28	0.37	0	0.37	0.19	0	0.19	0.18	0	0.18
	WB	2.644	2.531	2.531	95.72	99.87	95.59	-0.4	0	-0.4	0.11	0	0.12	0	0	0	0.11	0.01	0.12	0.11	0	0.11	0	0.01	0
	Total	100	100	100	98.23	96.95	95.41				1.77	3.05	4.59	1.77	3.05	4.59	1.77	3.05	4.59	1.22	1.32	1.22	0.55	1.73	0.55

[c] U- urban areas; I- industrial land; R- rainfed and permanent crops; P- permamently irrigated and rice fields; H- heterogeneous agricultural areas; F- forest; W- wetlands; WB- water bodies.

[b] r - annual rate of landscape change

[c] Values in light grey boxes depict net gains (positive net change); values in white boxes depict net losses (negative net change); values in dark grey boxes depict no net change

Table III.2a.
River basin gains: Most relevant systematic transitions in the Mondego river basin considering the percentage of land change in terms of gains.

	Systematic transition (from to) [a]		Time period	% Obsv. [b]	% Expect. if gain random [c]	Obsv. minus expected [d]	Difference divided by expected [e]	Interpretation of systematic transition
River basin relevant gains	Heterog.	Urban	1990-2000	0.231	0.083	0.148	1.786	When Urban gains, it replaces Heterog.
			2000-2006	0.681	0.222	0.459	2.063	
			1990-2006	0.927	0.307	0.619	2.015	
	Forest		1990-2000	0.067	0.212	-0.145	-0.685	When Urban gains, it avoids replacing Forest.
			2000-2006	0.083	0.576	-0.492	-0.856	
			1990-2006	0.158	0.785	-0.627	-0.798	
	Heterog.	Rainfed	1990-2000	0.308	0.105	0.204	1.946	When Rainfed gains, it replaces Heterog.
			1990-2006	0.320	0.130	0.190	1.460	
	Forest		1990-2000	0.083	0.267	-0.184	-0.690	When Rainfed gains, it avoids replacing Forest.
			1990-2006	0.165	0.333	-0.168	-0.503	
	Rainfed	Heterog.	2000-2006	0.323	0.050	0.274	5.517	When Heterog. gains, it replaces Rainfed.
			1990-2006	0.332	0.055	0.277	5.062	
	Forest		2000-2006	0.255	0.534	-0.279	-0.523	When Heterog. gains, it avoids replacing Forest.
			1990-2006	0.347	0.629	-0.282	-0.448	
	Rainfed	Non-Rainfed	2000-2006	0.645	0.243	0.402	1.654	When non-rainfed gain, they replace Rainfed. Rainfed loses
			1990-2006	0.626	0.284	0.342	1.207	
	Heterog.	Non-Heterog.	1990-2000	0.663	0.337	0.326	0.967	When non-heterog. gain, they replace Heterog. Heterog. loses.
			2000-2006	1.343	0.852	0.491	0.576	
			1990-2006	1.976	1.159	0.817	0.704	
	Forest	Non-Forest	1990-2000	0.583	0.883	-0.300	-0.340	When non-forest gain, they avoid replacing Forest. Forest does not lose.
			2000-2006	0.560	1.392	-0.832	-0.598	
			1990-2006	1.114	2.172	-1.058	-0.487	
			1990-2000	0.758	1.672	-0.914	-0.546	
			1990-2006	0.780	2.184	-1.405	-0.643	

[a] Urban - urban areas; Industrial - industrial land; Rainfed - rainfed and permanent crops; Rice - permanently irrigated and rice fields; Heterog. - heterogeneous agricultural areas; Forest – forest.
[b] % Obsv. - percentage of class observed in time 1. [c] % Expect. - % of class expected in time period 2 if gain (or loss) had been random.
[d] Obsv. minus expected - % observed minus % expected. [e] Difference divided by expected - % observed minus % expected divided by the % expected. [f] Gray boxes identify relevant systematic transitions.

Table III.2b.
River basin losses: Most relevant systematic transitions in the Mondego river basin considering the percentage of land change in terms of losses.

	Systematic transition (from	to) [a]	Time period	% Obsv. [b]	% Expect. if gain random [c]	Obsv. minus expected [d]	Difference divided by expected [e]	Interpretation of systematic transition
River basin relevant losses	Heterog.	Urban	1990-2000	0.231	0.013	0.218	16.742	When Heterog. loses, Urban replaces it.
			2000-2006	0.681	0.041	0.640	15.504	
			1990-2006	0.927	0.061	0.866	14.262	
	Non-urban		1990-2000	0.325	0.038	0.286	7.444	When non-urban lose, Urban replaces them. Urban gains.
			2000-2006	0.886	0.097	0.789	8.166	
			1990-2006	1.203	0.152	1.052	6.937	
	Forest	Industrial	1990-2000	0.259	0.009	0.249	27.526	When Forest loses, Industrial replaces it.
			2000-2006	0.124	0.011	0.114	10.779	
			1990-2006	0.368	0.021	0.347	16.498	
	Non-industrial		1990-2000	0.353	0.015	0.339	23.359	When non-industrial lose, Industrial replaces them. Industrial gains.
			2000-2006	0.183	0.028	0.155	5.608	
			1990-2006	0.495	0.044	0.451	10.309	
	Heterog.	Rainfed	1990-2000	0.308	0.052	0.256	4.883	When Heterog. loses, Rainfed replaces it.
			1990-2006	0.320	0.142	0.178	1.250	
	Non-rainfed		1990-2000	0.391	0.151	0.240	1.594	When non-rainfed lose, Rainfed replaces them. Rainfed gains.
			1990-2006	0.487	0.321	0.166	0.519	
	Non-rice	Rice	1990-2006	0.067	0.163	-0.096	-0.590	When non-rice lose, they avoid replacement by Rice. Rice does not gain.
	Rainfed	Heterog.	2000-2006	0.323	0.164	0.159	0.973	When Rainfed loses, Heterog. replaces it.
			1990-2006	0.332	0.159	0.173	1.087	
	Forest		1990-2000	0.115	0.400	-0.285	-0.713	When Forest loses, it avoids replacement by Heterog.
			2000-2006	0.255	0.376	-0.120	-0.321	
			1990-2006	0.347	0.747	-0.400	-0.536	
	Non-heterog.		1990-2000	0.130	0.426	-0.296	-0.695	When non-heterog. lose, they avoid replacement by Heterog.. Heterog. does not gain.
			1990-2006	0.728	0.940	-0.212	-0.225	
	Rainfed	Forest	2000-2006	0.216	0.438	-0.222	-0.507	When Rainfed loses, it avoids replacement by Forest.
			1990-2006	0.146	0.425	-0.279	-0.656	
	Heterog.		1990-2000	0.035	0.564	-0.529	-0.939	When Heterog. loses, it avoids replacement by Forest.
			2000-2006	0.526	1.134	-0.608	-0.536	
			1990-2006	0.618	1.669	-1.052	-0.630	
	Non-forest		1990-2006	0.050	0.630	-0.580	-0.920	When non-forest lose, they avoid replacement by Forest. Forest does not gain.
			1990-2000	0.758	1.672	-0.914	-0.546	
			1990-2006	0.780	2.184	-1.405	-0.643	

[a] Urban - urban areas; Industrial - industrial land; Rainfed - rainfed and permanent crops; Rice - permanently irrigated and rice fields; Heterog. - heterogeneous agricultural areas; Forest – forest.
[b] % Obsv. - percentage of class observed in time 1. [c] % Expect. - % of class expected in time period 2 if gain (or loss) had been random.
[d] Obsv. minus expected - % observed minus % expected. [e] Difference divided by expected - % observed minus % expected divided by the % expected. [f] Gray boxes identify relevant systematic transitions.

Table III.3a.
Lower valley gains: Most relevant systematic transitions in the Mondego lower valley considering the percentage of land change in terms of gains.

	Systematic transition (from to) [a]		Time period	% Obsv. [b]	% Expect. if gain random [c]	Obsv. minus expected [d]	Difference divided by expected [e]	Interpretation of systematic transition
Lower valley relevant gains	Rice	Urban	2000-2006	0.021	0.177	-0.157	-0.884	When Urban gains, it avoids replacing Rice.
			1990-2006	0.021	0.211	-0.190	-0.902	
	Heterog.		1990-2000	0.218	0.111	0.107	0.957	When Urban gains, it replaces Heterog.
			2000-2006	1.437	0.549	0.889	1.620	
			1990-2006	1.699	0.663	1.036	1.564	
	Forest		1990-2000	0.071	0.181	-0.110	-0.610	When Urban gains, it avoids replacing Forest.
			2000-2006	0.123	0.890	-0.768	-0.862	
			1990-2006	0.218	1.076	-0.859	-0.798	
	Heterog.	Industrial	1990-2006	0.139	0.278	-0.138	-0.498	When Industrial gains, it avoids replacing Heterog.
	Forest		1990-2000	0.504	0.348	0.156	0.447	When Industrial gains, it replaces Forest.
			1990-2006	0.643	0.451	0.192	0.426	
	Heterog.	Rainfed	2000-2006	0.223	0.084	0.139	1.652	When Rainfed gains, it replaces Heterog.
			1990-2006	0.285	0.108	0.178	1.651	
	Forest		2000-2006	0.026	0.136	-0.110	-0.810	When Rainfed gains, it avoids replacing Forest.
			1990-2006	0.046	0.175	-0.129	-0.737	
	Rainfed	Heterog.	2000-2006	0.478	0.094	0.383	4.054	When Heterog. gains, it replaces Rainfed.
			1990-2006	0.511	0.108	0.403	3.736	
	Forest		2000-2006	0.082	0.479	-0.398	-0.830	When Heterog. gains, it avoids replacing Forest.
			1990-2006	0.137	0.547	-0.410	-0.750	
	Rice	Forest	2000-2006	0.002	0.134	-0.131	-0.983	When Forest gains, it avoids replacing Rice.
			1990-2006	0.014	0.148	-0.134	-0.907	
	Heterog.		2000-2006	0.647	0.414	0.234	0.566	When Forest gains, it replaces Heterog.
			1990-2006	0.701	0.465	0.236	0.506	
	Rainfed	Non-rainfed	1990-2006	1.057	0.590	0.467	0.792	When non-rainfed gain, they replace Rainfed. Rainfed loses.
	Rice	Non-rice	1990-2000	0.041	0.147	-0.106	-0.723	When non-rice gain, they avoid replacing Rice. Rice does not lose.
			2000-2006	0.203	0.466	-0.263	-0.565	
			1990-2006	0.243	0.595	-0.352	-0.591	
	Heterog.	Non-heterog.	2000-2006	2.411	1.197	1.213	1.013	When non-heterog. gain, they replace Heterog.. Heterog. loses.
			1990-2006	2.869	1.613	1.255	0.778	
	Forest	Non-forest	2000-2006	0.415	1.751	-1.336	-0.763	When non-forest gain, they avoid replacing Forest. Forest does not lose.
			1990-2006	1.119	2.411	-1.293	-0.536	

[a] Urban - urban areas; Industrial - industrial land; Rainfed - rainfed and permanent crops; Rice - permanently irrigated and rice fields; Heterog. - heterogeneous agricultural areas; Forest – forest.
[b] % Obsv. - percentage of class observed in time 1. [c] % Expect. - % of class expected in time period 2 if gain (or loss) had been random.
[d] Obsv. minus expected - % observed minus % expected. [e] Difference divided by expected - % observed minus % expected divided by the % expected. [f] Gray boxes identify relevant systematic transitions.

Table III.3b.
Lower valley losses: Most relevant systematic transitions in the Mondego lower valley considering the percentage of land change in terms of losses.

	Systematic (from to) [a]	transition	Time period	% Obsv. [b]	% Expect. if gain random [c]	Obsv. minus expected [d]	Difference divided by expected [e]	Interpretation of systematic transition
Lower valley relevant losses	Industrial	Urban	2000-2006	0.117	0.007	0.111	16.832	When Industrial loses, Urban replaces it.
	Rainfed		2000-2006	0.146	0.044	0.102	2.351	When Rainfed loses, Urban replaces it.
			1990-2006	0.211	0.053	0.158	2.972	
	Heterog.		1990-2000	0.218	0.019	0.199	10.638	When Heterog. loses, Urban replaces it.
			2000-2006	1.437	0.152	1.285	8.468	
			1990-2006	1.699	0.181	1.518	8.405	
	Forest		1990-2006	0.218	0.097	0.120	1.237	When Forest loses, Urban replaces it.
	Non-urban		1990-2000	0.371	0.066	0.305	4.638	When non-urban loses, Urban replaces them. Urban gains.
			2000-2006	1.844	0.248	1.596	6.429	
			1990-2006	2.207	0.348	1.860	5.344	
	Rainfed	Industrial	1990-2006	0.125	0.017	0.108	6.349	When Rainfed loses, Industrial replaces it.
	Heterog.		1990-2000	0.133	0.009	0.124	14.292	When Heterog. loses, Industrial replaces it.
	Forest		1990-2000	0.504	0.018	0.486	27.448	When Forest loses, Industrial replaces it.
			2000-2006	0.174	0.012	0.163	14.069	
			1990-2006	0.643	0.031	0.612	19.595	
	Non-industrial		1990-2000	0.728	0.030	0.697	22.945	When non-industrial lose, Industrial replaces them. Industrial gains.
			2000-2006	0.315	0.078	0.236	3.010	
			1990-2006	0.942	0.112	0.831	7.446	
	Forest	Rainfed	1990-2000	0.020	0.126	-0.107	-0.846	When Forest loses, it avoids replacement by Rainfed.
			1990-2006	0.046	0.184	-0.138	-0.750	
	Non-rainfed		1990-2000	0.076	0.199	-0.123	-0.616	When non-rainfed lose, they avoid replacement by Rainfed. Rainfed does not gain.
			2000-2006	0.263	0.392	-0.128	-0.327	
			1990-2006	0.333	0.562	-0.229	-0.408	
	Heterog.	Rice	2000-2006	0.048	0.308	-0.261	-0.846	When Heterog. loses, it avoids replacement by Rice.
			1990-2006	0.044	0.367	-0.323	-0.881	
	Forest		1990-2006	0.075	0.198	-0.122	-0.619	When Forest loses, it avoids replacement by Rice.
	Non-rice		1990-2000	0.080	0.217	-0.137	-0.632	When non-rice lose, they avoid replacement by Rice. Rice does not gain.
			2000-2006	0.164	0.489	-0.325	-0.665	
			1990-2006	0.243	0.687	-0.444	-0.646	
	Rainfed	Heterog.	2000-2006	0.478	0.259	0.219	0.847	When Rainfed loses, Heterog. replaces it.
			1990-2006	0.511	0.315	0.196	0.621	
	Forest		1990-2000	0.069	0.395	-0.326	-0.826	When Forest loses, it avoids replacement by Heterog.
			2000-2006	0.082	0.214	-0.133	-0.620	
			1990-2006	0.137	0.578	-0.441	-0.764	
	Non-heterog.		1990-2000	0.107	0.490	-0.383	-0.782	When non-heterog. lose, they avoid replacement by Heterog. Heterog. does not gain.
			2000-2006	0.725	0.589	0.136	0.232	When non-heterog. lose, Heterog. replaces them. Heterog. gains.
			1990-2006	0.813	1.010	-0.197	-0.195	When non-heterog. lose, they avoid replacement by Heterog. Heterog. does not gain.
	Rainfed	Forest	2000-2006	0.090	0.449	-0.359	-0.801	When Rainfed loses, it avoids replacement by Forest.
			1990-2006	0.104	0.547	-0.443	-0.810	
	Rice		2000-2006	0.002	0.106	-0.103	-0.978	When Rice loses, it avoids replacement by Forest.
			1990-2006	0.014	0.127	-0.113	-0.891	
	Heterog.		1990-2000	0.069	0.315	-0.246	-0.781	When Heterog. loses, it avoids replacement by Forest.
			2000-2006	0.647	1.566	-0.919	-0.587	
			1990-2006	0.701	1.864	-1.163	-0.624	
	Non-forest		1990-2000	0.095	0.470	-0.375	-0.798	When non-forest lose, they avoid replacement by Forest. Forest does not gain.
			2000-2006	0.759	2.217	-1.458	-0.658	
			1990-2006	0.833	2.614	-1.781	-0.681	

[a] Urban - urban areas; Industrial - industrial land; Rainfed - rainfed and permanent crops; Rice - permanently irrigated and rice fields; Heterog. - heterogeneous agricultural areas; Forest – forest.
[b] % Obsv. - percentage of class observed in time 1. [c] % Expect. - % of class expected in time period 2 if gain (or loss) had been random.
[d] Obsv. minus expected - % observed minus % expected. [e] Difference divided by expected - % observed minus % expected divided by the % expected. [f] Gray boxes identify relevant systematic transitions.

Table III.4a.

Estuary gains: Most relevant systematic transitions in the Mondego estuary region considering the percentage of land change in terms of gains.

	Systematic (from to) [a]	transition	Time period	% Obsv. [b]	% Expect. if gain random [c]	Obsv. minus expected [d]	Difference divided by expected [e]	Interpretation of systematic transition
Estuary region relevant gains	Industrial	Urban	2000-2006	0.294	0.011	0.283	24.750	When Urban gains, it replaces Industrial.
			1990-2006	0.184	0.010	0.174	17.453	
	Rice		2000-2006	0.000	0.166	-0.166	-1.000	When Urban gains, it avoids replacing Rice.
			1990-2006	0.000	0.234	-0.234	-1.000	
	Heterog.		1990-2006	0.411	0.262	0.149	0.570	When Urban gains, it replaces Heterog.
	Forest		2000-2006	0.116	0.277	-0.160	-0.579	When Urban gains, it avoids replacing Forest.
	Rice	Industrial	1990-2000	0.000	0.121	-0.121	-1.000	When Industrial gains, it avoids replacing Rice.
			1990-2006	0.000	0.113	-0.113	-1.000	
	Forest		1990-2000	0.437	0.206	0.231	1.123	When Industrial gains, it replaces Forest.
			1990-2006	0.337	0.192	0.145	0.756	
	Heterog.	Rice	1990-2006	0.006	0.116	-0.110	-0.944	When Rice. gains, it avoids replacing Heterog.
	Rainfed	Heterog.	2000-2006	1.103	0.121	0.982	8.118	When Heterog. gains, it replaces Rainfed.
			1990-2006	1.104	0.124	0.979	7.879	
	Rice		2000-2006	0.010	0.355	-0.345	-0.973	When Heterog. gains, it avoids replacing Rice.
			1990-2006	0.010	0.361	-0.351	-0.973	
	Forest		2000-2006	0.136	0.591	-0.455	-0.770	When Heterog. gains, it avoids replacing Forest.
			1990-2006	0.159	0.614	-0.455	-0.742	
	Rice	Forest	2000-2006	0.003	0.217	-0.214	-0.985	When Forest gains, it avoids replacing Rice.
			1990-2006	0.003	0.298	-0.294	-0.989	
	Heterog.		1990-2000	0.239	0.099	0.141	1.426	When Forest gains, it replaces Heterog.
			2000-2006	0.534	0.236	0.297	1.258	
			1990-2006	0.767	0.332	0.435	1.308	
	Rice	Wetland	1990-2000	0.165	0.062	0.103	1.673	When Wetland gains, it replaces Rice.
			1990-2006	0.165	0.062	0.103	1.670	
	Forest		1990-2000	0.000	0.105	-0.105	-1.000	When Wetland gains, it avoids replacing Forest.
			1990-2006	0.000	0.105	-0.105	-1.000	
	Water bodies		1990-2000	0.113	0.008	0.106	13.860	When Wetland gains, it replaces Water bodies.
			1990-2006	0.113	0.008	0.106	13.832	
	Industrial	Non-industrial	2000-2006	0.388	0.055	0.333	6.064	When non-industrial gain, they replace Industrial. Industrial loses.
			1990-2006	0.188	0.047	0.141	2.996	
	Rainfed	Non-rainfed	2000-2006	1.284	0.270	1.014	3.759	When non-rainfed gain, they replace Rainfed. Rainfed loses.
			1990-2006	1.340	0.404	0.936	2.317	
	Rice	Non-rice	1990-2000	0.165	0.351	-0.186	-0.530	When non-rice gain, they avoid replacing Rice. Rice does not lose.
			2000-2006	0.013	0.815	-0.803	-0.984	
			1990-2006	0.178	1.110	-0.932	-0.840	
	Heterog.	Non-heterog.	2000-2006	0.860	0.521	0.340	0.653	When non-heterog. gain, they replace Heterog.. Heterog. loses.
			1990-2006	1.398	0.954	0.445	0.466	
	Forest	Non-forest	1990-2000	0.799	0.595	0.204	0.343	When non-forest gain, they replace Forest. Forest loses.
			2000-2006	0.378	1.027	-0.648	-0.631	When non-forest gain, they avoid replacing Forest. Forest does not lose.
			1990-2006	1.152	1.561	-0.409	-0.262	
	Wetland	Non-wetland	2000-2006	0.000	0.148	-0.148	-1.000	When non-wetland gain, they avoid replacing Wetland. Wetland does not lose.
			1990-2006	0.091	0.195	-0.105	-0.536	

[a] Urban - urban areas; Industrial - industrial land; Rainfed - rainfed and permanent crops; Rice - permanently irrigated and rice fields; Heterog. - heterogeneous agricultural areas; Forest – forest.

[b] % Obsv. - percentage of class observed in time 1. [c] % Expect. - % of class expected in time period 2 if gain (or loss) had been random.

[d] Obsv. minus expected - % observed minus % expected. [e] Difference divided by expected - % observed minus % expected divided by the % expected. [f] Gray boxes identify relevant systematic transitions.

Table III.4b.
Estuary losses: Most relevant systematic transitions in the Mondego estuary region considering the percentage of land change in terms of losses.

	Systematic (from to) [a]	transition	Time period	% Obsv [b]	% Expect. if gain random [c]	Obsv. minus expected [d]	Difference divided by expected [e]	Interpretation of systematic transition
Estuary region relevant losses	Industrial	Urban	2000-2006	0.294	0.018	0.276	15.235	When Industrial loses, Urban replaces it.
			1990-2006	0.184	0.009	0.176	20.087	
	Heterog.		1990-2000	0.181	0.028	0.153	5.379	When Heterog. loses, Urban replaces it.
			2000-2006	0.230	0.052	0.178	3.408	
			1990-2006	0.411	0.084	0.327	3.868	
	Forest		1990-2006	0.327	0.083	0.244	2.934	When Forest loses, Urban replaces it.
	Non-urban		1990-2000	0.327	0.098	0.229	2.343	When non-urban loses, Urban replaces them. Urban gains.
			2000-2006	0.741	0.162	0.579	3.582	
			1990-2006	1.055	0.262	0.793	3.022	
	Heterog.	Industrial	1990-2000	0.123	0.011	0.112	10.691	When Heterog. loses, Industrial replaces it.
	Forest		1990-2000	0.437	0.018	0.418	22.656	When Forest loses, Industrial replaces it.
			1990-2006	0.337	0.022	0.314	13.962	
	Non-industrial		1990-2000	0.560	0.036	0.523	14.464	When non-industrial loses, Industrial replaces them. Industrial gains.
			2000-2006	0.158	0.041	0.118	2.899	
			1990-2006	0.521	0.070	0.451	6.412	
	Non-rainfed	Rainfed	1990-2000	0.000	0.174	-0.174	-1.000	When non-rainfed lose, they avoid replacement by Rainfed. Rainfed does not gain.
	Rainfed	Rice	2000-2006	0.003	0.296	-0.292	-0.989	When Rainfed loses, it avoids replacement by Rice.
			1990-2006	0.003	0.309	-0.306	-0.990	
	Heterog.		1990-2000	0.000	0.153	-0.153	-1.000	When Heterog. loses, it avoids replacement by Rice.
			2000-2006	0.006	0.244	-0.237	-0.973	
			1990-2006	0.006	0.396	-0.390	-0.984	
	Forest		1990-2006	0.275	0.390	-0.115	-0.295	When Forest loses, it avoids replacement by Rice.
	Non-rice		1990-2000	0.317	0.481	-0.164	-0.340	When non-rice lose, they avoid replacement by Rice. Rice does not gain.
			2000-2006	0.065	0.782	-0.717	-0.917	
			1990-2006	0.382	1.211	-0.829	-0.685	
	Rainfed	Heterog.	2000-2006	1.103	0.326	0.777	2.380	When Rainfed loses, Heterog. replaces it.
			1990-2006	1.104	0.340	0.764	2.247	
	Forest		1990-2000	0.029	0.292	-0.263	-0.900	When Forest loses, it avoids replacement by Heterog.
			1990-2006	0.159	0.430	-0.271	-0.631	
	Non-heterog.		1990-2000	0.029	0.406	-0.377	-0.928	When non-heterog. lose, they avoid replacement by Heterog. Heterog. does not gain.
			2000-2006	1.262	0.598	0.663	1.109	When non-heterog. lose, Heterog. replaces them. Heterog. gains.
			1990-2006	1.282	0.951	0.331	0.348	
	Industrial	Forest	2000-2006	0.019	0.142	-0.123	-0.863	When Industrial loses, it avoids replacement by Forest.
	Rainfed		2000-2006	0.078	0.495	-0.417	-0.843	When Rainfed loses, it avoids replacement by Forest.
			1990-2006	0.100	0.517	-0.417	-0.806	
	Heterog.		2000-2006	0.534	0.408	0.125	0.307	When Heterog. loses, Forest replaces it.
			1990-2006	0.767	0.664	0.103	0.156	
	Non-forest		1990-2000	0.262	0.429	-0.167	-0.390	When non-forest lose, they avoid replacement by Forest. Forest does not gain.
			2000-2006	0.647	1.102	-0.455	-0.413	
			1990-2006	0.884	1.457	-0.573	-0.394	
	Rice	Wetland	1990-2000	0.165	0.008	0.157	19.576	When Rice loses, Wetland replaces it.
			1990-2006	0.165	0.009	0.156	18.050	
	Water bodies		1990-2000	0.113	0.004	0.109	24.559	When Water bodies loses, Wetland replaces it.
			1990-2006	0.113	0.005	0.109	23.833	
	Non-wetland		1990-2000	0.278	0.090	0.189	2.106	When non-wetland loses, Wetland replaces them. Wetland gains.
			1990-2006	0.000	0.139	-0.139	-1.000	When non-wetland lose, they avoid replacement by Wetland. Wetland does not gain.

[a] Urban - urban areas; Industrial - industrial land; Rainfed - rainfed and permanent crops; Rice - permanently irrigated and rice fields; Heterog. - heterogeneous agricultural areas; Forest – forest. [b] % Obsv. - percentage of class observed in time 1. [c] % Expect. - % of class expected in time period 2 if gain (or loss) had been random. [d] Obsv. minus expected - % observed minus % expected. [e] Difference divided by expected - % observed minus % expected divided by the % expected. [f] Gray boxes identify relevant systematic transitions.

Regarding industrial land and from the perspective of gains, this class systematically replaces forest, but when focusing only on the lower valley and the estuary regions. At the same time, industrial land avoids replacing heterogeneous in the lower valley and rice fields in the estuary regions (Tables III.3a and III.4a). From the perspective of losses, however, forest tends to be replaced by industrial land in all three regions, by rates from over 13 times the expected ones in the Mondego estuary (Table III.4b) to over 16 times in the river basin, from 1990 to 2006 (Table III.2b). Such rates are even higher from 1990 to 2000 reaching values over 27 times the expected if forest were to lose randomly (Table III.3b). Heterogeneous also tend to be replaced by industrial land from 1990 to 2000, with rates higher than the expected if heterogeneous were to lose randomly: between 10 times in the Mondego estuary (Table III.4b) and 14 times in the lower valley (Table III.3b). As a result, industrial land shows positive and high net change and annual rates of change, except in the estuary region during the 2000–2006 period, when industrial land loses surface due to replacement by urban at rates over 15 times the expected if industrial land were to lose randomly (Table III.4b).

With regard to rainfed, this class systematically replaces heterogeneous and avoids replacing forest. Transitions from heterogeneous to rainfed occur at rates from over 1.4 times up to 1.9 times the rates expected if rainfed class were to gain randomly; and rates from over 1.2 times up to 4.8 times the rates expected if heterogeneous were to lose randomly. Systematic transitions to rainfed occur only at the river basin and in the lower valley regions (Tables III.2 and III.3). Despite the systematic transitions to rainfed, this class shows net loss and negative annual rates in all three regions, except for the river basin from 1990 to 2000. This is explained by the systematic transitions to urban, industrial land and also heterogeneous classes (Table III.1).

With respect to heterogeneous, they lose total surface from 1990 to 2006, in all three regions, due to the aforementioned systematic replacements by urban, industrial land and rainfed and also by forest. However, its swapping dynamics indicates that heterogeneous also replace other classes (Table III.1). Results show that when heterogeneous gain, they systematically replace rainfed at rates higher than 3%, in all three regions, while they avoid replacing forest. At the same time, rainfed systematically attracted replacement by heterogeneous, but at lower rates.

Rice fields show low annual rates of change and low swap. This is in accordance with the low number of systematic transitions found, which only shows that, in the lower valley and estuary region, heterogeneous, forest and rainfed avoid replacement by rice fields (Tables III.3b and III.4b) and that, in the estuary region, rice fields avoid replacing heterogeneous (Table III.4a). At the same time, in the estuary region, wetlands systematically replace rice fields, at rates over 19 times the expected rates if rice fields were to lose randomly (Table III.4b).

The only class that forest replaces in a systematic manner is heterogeneous. Nonetheless the highest rate found was 1.4 times the expected if forest were to gain randomly, which was found in the estuary region. At the same time, forest avoids replacement by rice fields and also rainfed (Table III.4a).Moreover, and as mentioned before, the only classes that systematically replace forest are urban and industrial land. Overall, forest has low annual rates of change, shows net loss from 1990 to 2000 and net gain from 2000 to 2006, but has high swap (Table III.1). Wetlands, whose proportion is highest in the Mondego estuary region show very low annual rates of change and hardly any swapping changes. However, our study was able to detect that, in the estuary region, when wetlands gain surface it is due to systematic replacement of rice fields and also water bodies. 3.3. Dominant processes of LUC change According to Braimoh (2006) there is a dominant signal of change from class A to class B, if class B systematically loses surface to class A, while at the same time class A systematically gains surface from class B. From this perspective, not all systematic transitions found in this study are actually strong signals of change. For some pairs of classes, this assumption was not fulfilled in any time period or region. In other cases, this assumption was only fulfilled for a specific time period. For all dominant signals of change found, they were always detected when considering the sixteen-year time period, but some were only detected during the first ten years, or during the last six (Figure III.3).

Regarding the entire river basin, the first ten years witnessed strong signals of change from heterogeneous to urban and to rainfed; whereas during the last six years a strong signal was again found from heterogeneous to urban but also from rainfed to heterogeneous.

Regarding the lower valley, during the first ten years dominant signals of change were found from heterogeneous to urban and also from forest to industrial land; whereas during the last six years signals of strong change were found again from heterogeneous to urban, but also from rainfed to heterogeneous. Finally, and focusing only on the estuary region, during the first ten years strong signals of change were observed from forest to industrial land and from rice fields and water bodies to wetland; whereas during the last six years dominant signals of change were found from industrial land to urban, from rainfed to heterogeneous and from heterogeneous to forest. A dominant signal of change was also detected from heterogeneous to urban, but only when considering the sixteen-year time period. The dominant signals of change allow us to determine the dominant processes of LUC change (Table A.III.2). Urbanization and other agricultural changes are common to the three nested regions. Yet, if we take into account only the Mondego lower valley and the estuary regions, then industrial expansion also arises as a dominant process of change. But focusing only on the estuary region, apart from the processes mentioned before, afforestation together with land restoration and siltation/deposition also emerge as dominant processes of change.

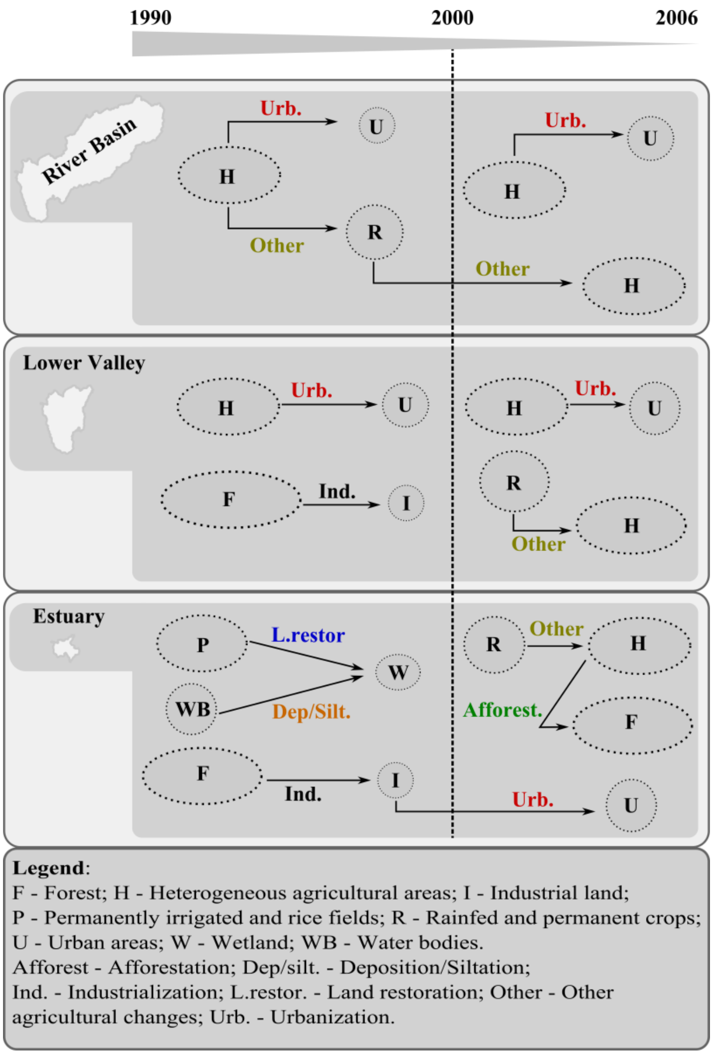


Figure III.3.

Dominant processes of LULC change. Systematic transitions revealing strong signals of change found in the Mondego river basin, lower valley and estuary region between 1990 and 2006. The size of each circle approximately represents the proportion occupied by the class at the beginning of the time period, if class loses area, or at the end of the time period if class gains area.

Discussion

LUC as driving forces and dynamics over the years

Each LUC class, with exception to those related to the water environment, has the potential to exert different types of pressure affecting the state of water bodies—from pollution to alteration of hydrologic regime, to changes in morphology and other types of pressure, such as the introduction of new diseases in local fauna and flora (Aguilera et al. 2012, FAO 1996, Fiquepron et al. 2013). Quantifying the proportion of each class and its dynamics over the years allows for a first glance over the type and intensity of driving forces acting in an aquatic system. The driving forces from LUC, i.e., LUC classes with potential impact on aquatic systems, were quantified for three nested regions of the Mondego river basin, in Central Portugal. Forest, which occupies the largest surface area in all three regions and is associated with good runoff retention and with low diffuse pollution potential (Neary et al. 2009), loses representativeness in the lower valley and estuary regions. At the same time, agricultural areas and artificial surfaces, which might critically affect water systems, occupy larger proportions in downstream regions (Table A.III.1). This general pattern is observable for all the three years analyzed (1990, 2000 and 2006). This means that the type of driving forces did not change over the years or across the regions. Despite this, because the total surface occupied by each class and its overall spatial allocation have changed, we expect changes in the type or intensity of pressures acting on the system. During the sixteen-year period forest lost surface in all three regions, whereas artificial surfaces gained area and showed positive high annual rates of landscape change. Urban and industrial land classes are not only a proxy for impervious areas, but are also a proxy for household and for industrial/commercial estates. As a consequence of both forest decrease and artificial surface increase, there is a potential for runoff increase, promoting the intensification of non-point source pollution from urban drainage, commercial forestry and agriculture (Vidal-Dorsch et al. 2012). At the same time, point-source pollution from wastewater, waste management, industry and contaminated land has also potentially increased. As a result, pesticides, pharmaceuticals (Leston et al. 2011, Santos et al. 2010), endocrine disruptors (Baptista et al. 2013, Nunes et al. 2011b, Ribeiro et al. 2009), metals (Couto et al. 2013), organic material, salt, ammonia and other urban contaminants might have increased in the system, though treatment plants can be successful in removing some of them

(EPA 2010). Such alterations in the type and pattern of LUC driving forces could have implications not only on future management policies, but also on monitoring plans and on the selection of biological indicators. Consider, as an example, the increase in industrial areas and the consequent emergence and/ or intensification of sources of pollution, such as pharmaceuticals used in the food production industry to ensure animal welfare. Though several pharmacological substances are already within the scope of researchers, new monitoring programs and biological indicators are needed to assess the wide variety of substances used in the food production industry. Leston et al. (2011, 2013), for instance, advocate that *Ulva lactuca* should be included as an indicator for nitrofuran and chloramphenicol, two illegal antibiotics still in use in Europe. With respect to agricultural classes, a decrease in the total surface occupied by rainfed and permanent crops indicates a decline on the pressures derived from the application of fertilizers and other agrochemicals, especially during the irrigation seasons. In contrast, an increase in nutrients could be expected due to the increase of surface occupied by rice fields in the Mondego estuary region. In fact, the Mondego estuary has been under environmental stress by eutrophication processes, in part due to nutrient inputs from surrounding rice fields (Marques et al. 2003). However, mitigation measures implemented in 1998 caused an effective reduction in the N:P atomic ratio leading to a decrease in green macroalgae biomass and an increase in seagrass biomass and cover (Leston et al. 2008, Lillebø et al. 2005). Nonetheless, to return the system to its original state of seagrass dominance, further mitigation measures need to be taken (Marques et al. 2003) and, concordantly, the current Hydrographic Region Management Plan for the Mondego river basin proposes the reduction of nutrient loads into the estuary as one of the main actions to implement in order to achieve good status in all water bodies (ARH do Centro 2012). Though our paper quantifies swap between classes, it does not assess landscape patterns and therefore we are unable to discuss the impacts of different configurations in the Mondego river basin. However, it is widely recognized that LUC configuration poses challenges to aquatic systems (Alberti 2005, Wiens 2002). As an example, Alberti et al. (2007) showed that the configuration of impervious area and forest influences the ecological conditions of streams.

Systematic transitions

Whether or not the LUC changes observed are a result of random or non-random processes of change is of ultimate importance. Such findings help us focus on transitions that have evolved as a result of consistent processes that can be targeted; described and quantified (e.g. population growth, industrial expansion and changes in land management policies). This study describes the most prevalent systematic transitions of LUC following the methodology proposed by Pontius et al. (2004), which prevents us from focusing mainly on large transitions, usually between the largest classes. In our case study, this would mean focusing only on transitions between heterogeneous and forest, which, in reality occur at rates not much higher than those that would be expected if the transition were to occur randomly (at maximum, around 1.2 times). On the other hand, identifying the systematic transitions allowed us to realize that the transitions with the highest rates, compared to those expected if the transition had been random, are in fact those involving artificial surfaces, which are also the smallest classes, apart from wetlands and water bodies. This means that despite the low fingerprint left on the landscape by these small classes, transitions to urban and industrial land classes might suggest a transition of pressures acting on the system at a rate higher than expected considering their total area. Our case study focused on three periods of time– 1990–2000, 2000– 2006 and 1990–2006 – for which we sought systematic transitions. One of the first evidences was that we did not find the same systematic transitions for all time periods, meaning that temporal resolution affects results and therefore interpretation. On the one hand, if a transition is considered systematic only for the entire sixteen-year period, it could mean that the processes causing that transition operated throughout the entire period and were not strong enough to be detected when analyzing the two separate time periods. On the other hand, if a transition is systematic only for one of the time periods under study, it could mean that the causes linked to that specific transition are probably related to some management policy prevailing during that period of time. The goal of this study was not to evaluate the causes of systematic transitions. Nonetheless, identifying the time periods and the geographic regions where transitions had a non-random behavior is a first step towards identifying such causes. From a precautionary point of view, identifying such causes is of utmost importance, since they are critical for the definition of social–ecological policies and management scenarios (Marques et al. 2009).

The methodology followed also compares systematic transitions among three nested regions — the Mondego river basin, Mondego lower valley and Mondego estuary region. This approach allowed us to focus on LUC relevant systematic transitions downstream the river basin, which would have been masked by transitions involving larger and well distributed classes across the basin. Systematic transitions involving rice fields and wetlands are examples of potentially overlooked transitions. Rice fields are an extremely important driver in the Mondego estuary. In fact, management policies implemented to improve the Mondego estuary water quality have long focused on the reduction of nutrient loadings from rice fields (Dolbeth et al. 2007, Lillebø et al. 2005). Our analysis with nested regions shows that, in the estuary, despite the overall increase of this class, it was, to some extent, replaced by wetland due to a non-random transition. Similarly, water bodies were also replaced by wetlands in the estuary region, both from the perspective of wetland gains and water body losses. What should be noticed with respect to wetlands is that, no matter what the causes of wetland increase, their presence is important in the Mondego Estuary region for two main reasons: 1) they are representative of wetland values in the west coast of Portugal, being important for birds, while supporting a diverse intertidal macroinvertebrate community (Lopes 2006); and 2) they can act as natural wastewater treatment plans reducing the nutrient loadings into the estuarine system (Marques et al. 2003). Previous studies showed that LUC detection analysis should consider different spatial scales, since landscape patterns might change with the resolution of maps (Manandhar et al. 2010). Coarser resolutions tend to show less swap and less inter-class transitions, and yet they can be very useful in finding the distances over which the change occurs (Pontius et al. 2004). Our study was only implemented for 100 m pixel resolution, though CORINE land cover maps are also provided with 250 m resolution. Even though CORINE land cover only delivers these two raster products, one with higher resolution, consistent with European standards, is also available from the Portuguese Geographic Institute (IGP 2010). However, the more detailed levels of this land cover map are not free of cost.

Dominant processes of LUC change as driving forces

Our work assumes that not only the presence of a certain LUC class has an effect on the state of the water bodies, but also that the transition to another use or practice is in itself a source of stress (IMPRESS 2003). Urbanization and industrial expansion associated with loss of forest and agricultural areas alter hydrology, water chemistry and habitat, which contribute to the degradation of biological communities. Chu et al. (2013), for instance, found that the frequency of average- and high flow events increased with urbanization and decreased with vegetation cover. Wang et al. (2012) found that stream benthic macroinvertebrate metrics are significantly correlated with the percent of impervious area. Though imperviousness is always foreseeable after the expansion of artificial surfaces, a higher magnitude of response of aquatic systems is expected if artificial areas replace forests than if they replace agricultural areas, since higher hydrologic impacts are expected with the loss of forests (Salazar et al. 2013, Trabucco et al. 2008, Zhang et al. 2001). Additionally, the magnitude of the impact from an urbanization or industrial expansion process due to loss of agricultural area will depend on the agricultural activities employed previous to transition. Agricultural areas are known to degrade water quality due to impacts such as siltation, turbidity, salinization, erosion, sedimentation and contamination with agrochemicals and toxic leaches, which are a consequence of the agricultural activity employed. For instance, activities with high levels of irrigation promote runoff of salts, fertilizers and pesticides (EPA 2005). This means that, after an urbanization process, a previously degraded aquatic system might face greater or new environmental problems which might demand new mitigation measures. Likewise, the magnitude of the impact from a change on the agricultural practice due to a change on the type of crop will depend on the practices employed previously. Agricultural transitions between different types of agricultural areas mean a change in water consumption behavior (ARH do Centro 2012), erosion rates (O'Geen 2006), type of fertilizers or pesticides, or other sources of pollution (Zhao et al. 2013). The rates at which a transition occurs should also be at the core of our attention since high transition rates might act in the system as if they were unique and extreme events from which the system will need to recover (Folke 2006). Transition rates could be used as proxy for the pace at which new pressures emerge or the intensity of a certain pressure increases/decreases. In the Mondego case study, urbanization due to loss of heterogeneous areas and industrial expansion due to loss of forest are two of the most relevant driving forces arising from transitions between LUC classes. Class heterogeneous aggregates associations of annual and permanent crops; areas with juxtaposition of annual and permanent crops; agricultural areas interspersed with natural vegetation and also annual crops under forestry species (EEA 2012). For its inherent characteristics, heterogeneous classes have uncertain water consumption behavior patterns, and therefore a more detailed analysis would be needed to evaluate the impact of this particular transition. However, impacts typical to any process of urbanization, such as runoff magnification, are still expected. With respect to industrial expansion due to loss of forest, we believe that critical environmental problems might have emerged due to this transition. LUC areas which suffered this transition, changed from an area with good runoff and evapotranspiration characteristics to an impervious area with high potential for contamination. In addition, the high rates at which the transitions to these artificial classes have occurred suggests that the magnitude of urban and industrial land fingerprints is, and could be in the future, of special concern, specifically with respect to flooding. Our study also revealed other agricultural changes as dominant processes of LUC change, mainly due to transitions between heterogeneous and rainfed. In terms of this study, other agricultural changes mean both a heterogenization of agricultural areas as well as the reverse process. In the case of the Mondego river basin, transition rates were higher from rainfed to heterogeneous, than the reverse. This means that the uncertainty with respect to water consumption and water retention behavior as well as agricultural pollution sources has increased at rates higher than those expected if these transitions were to occur randomly. Notice that rainfed tends to be replaced by heterogeneous and that heterogeneous tends to be replaced by urban. This could mean that abandonment of rainfed and permanent crops could ultimately promote urbanization. However, to clearly assess this relationship further research must be performed. Focusing only on the estuary region, afforestation at the expense of heterogeneous areas, siltation/deposition and land restoration also stands as dominant processes of LUC change. If afforestation is to fulfill commercial forestry needs, then we can expect an increase or at least an exchange of pesticides or fertilizers in this region and also an increase in pollution sources from planting/ground preparation. Nevertheless, it will always represent an increase in the evapotranspiration and infiltration levels, and therefore a change in the hydrologic and subsequent impacts. Further research focusing on forests, and, particularly, on the geographical area downstream from the city of Coimbra, distinguishing between commercial and non-commercial forestry should be performed. With respect to the process of land restoration, the fact that it is due to the systematic replacement of rice fields by wetlands indicates that a potential change on the pressures associated with intensive agriculture with high consumption of water and high levels of diffuse pollution might have occurred. Moreover, this specific transition is an indication that the expansion of artificial surfaces has not been sustained by land reclamation. Whether or not wetland restoration is a cause or a consequence of rice field disappearance would also need further exploration. With respect to siltation/deposition results show that an existing salt marsh patch increases very close to where the two arms of the estuary communicate. Although finding causes for transitions is not a subject for this study, in this case it is clear that such an increase is consistent with regularization works on the Mondego estuary, during the 90s, when the margins were grounded (Cunha et al. 1997).

Conclusion

The main objective of this study was to characterize the driving forces linked both to LUC and LUC change, with potential impacts on the condition of water bodies. Driving forces linked to the sole presence of LUC were obtained by quantifying the proportion occupied by each LUC class in three nested regions in the Mondego river basin, in Central Portugal, which was based on the three available CORINE Land Cover projects — 1990, 2000 and 2006. Results showed that agricultural areas and artificial surfaces, which are the driving forces that pose the most challenges to aquatic systems, are also the ones whose representativeness increases in downstream regions. Though this evidence might be useful to quantify the importance of each driving force in each region and in each year analyzed, it does not give information on the drivers linked to the processes of land transition. To obtain this information we identified the most relevant driving forces from dominant processes of LUC change through identification and quantification of systematic transitions. The magnitude of change and consistency of transitions revealed that the most relevant driving forces from LUC changes are not necessarily transitions between large classes. We also considered that these transitions revealed changes regarding the pressures acting in the system that might have been overlooked. Systematic transitions indicate that special attention should be paid to magnification of runoff, due both to loss of forests and increase of impervious areas and also to contamination of water bodies either due to new contaminants emerging from urbanized and industrialized areas, or to changes in agricultural practices. Our work characterized driving forces assessing differences in quantity, but a thorough analysis should also be performed to analyze changes in the configuration, since LUC patterns also play a key role on the type and intensity of pressures acting in the system. Additionally, future work should focus on the underlying processes that caused the observed dominant changes (Huang et al. 2012), since this is also crucial information for the development of effective management strategies.

Chapter IV

Evidence for deviations from uniform changes using CORINE maps: an Intensity Analysis approach.

Abstract

Land change affects processes that determine water supply, water demand and water quality. We apply a method to evaluate the strength of the evidence for deviations from uniform land change in a coastal area, in the context of Intensity Analysis. The errors in the CORINE maps at 1990 and 2006 can influence the apparent change, but the errors are unknown because error assessment of the 1990 map has never been released, while the error of the 2006 map has been checked for only some countries. The 1990 and the 2006 maps of a coastal watershed in Portugal served as the data to compute the intensities of changes among eight categories. We evaluate the sizes and types of errors that could explain deviations from uniform intensities. Errors in 2.0% of the 2006 map can explain all apparent deviations from uniform gains. Errors in 1.5% of the 1990 map can explain all apparent deviations from uniform losses. Errors in less than 0.7% of the 1990 map can explain all apparent deviations from uniform transitions to each gaining category. We analyse the strength of the evidence for deviations from uniform intensities in light of historical processes of change. Historical processes can explain some transitions that the data show, while the hypothesized errors in the data are the explanation for other transitions that are not consistent with known processes. Inconsistent transitions are an indication of the misclassification errors that could propagate to other land cover change applications, as in the assessment of hydrological processes.

Keywords: Accuracy, coastal system, estuary, Europe, hypothetical, land cover, transitions, watershed

Introduction

The increasing demand for land and water resources in coastal areas (Freire et al. 2009) has triggered land cover changes imposing pressures on coastal systems (Palmer et al. 2011). The impacts of land change include changes in water supply from alterations in the processes of runoff, infiltration and groundwater recharge (Ampe et al. 2012, Sajikumar and Remya 2015, Yang et al. 2015); changes in water demands from changes in land use practices (Priess et al. 2011); and changes in water quality from urban growth and agricultural runoff (León-Muñoz et al. 2013, Mouri et al. 2011, Seeboonruang 2012).

The assessment of impacts in the water balance and quality may have consequences in coastal management and thus requires accurate interpretation of land cover changes (LCC), for which the accuracies of land cover maps at two time points and the map of change are key elements (Loosvelt et al. 2014). However, accuracy assessment has not always been considered in the interpretation of LCC (Kuemmerle et al. 2009). The error pattern observed in a LCC map reflects the errors at the first time point as well as their interactions with the errors at the second time point. The error of the map of change is expected to be lower than the accuracy of either of the two maps from which it derives (Burnicki 2011). The change-detection error matrix is the most reported accuracy assessment tool (Foody 2010), but important advances are still under development (Aldwaik and Pontius 2012, Burnicki 2011, Liu and Zhou 2004, Zhang and Tang 2012).

The CORINE Land Cover (CLC) products are a series of land cover maps developed and released by the European Environment Agency since the 1990s. Three maps are currently available (CLC1990, CLC2000 and CLC2006), and one more is under development (CLC2012) (EEA 2012). CLC maps have been used widely for land cover change assessment, not only at the European level (Feranec et al. 2010), but also at the level of regions (Hewitt and Escobar 2011), river basin catchments (Teixeira et al. 2014), coastal zones (Freire et al. 2009) and bio-geographic areas (Kozac et al. 2007). The accuracy assessment of CLC maps has been standardized and reported, but only after the CLC1990 map. There is no available information regarding the accuracy of the CLC1990 map and the accuracy assessment of the CLC2006 has been checked for only some countries (Büttner et al. 2012, Caetano et al. 2009). Feranec et al. (2010) assessed land cover change in Europe using CORINE Land Cover at 1990 and 2000, then interpreted the results in light of the accuracies of the individual land categories in the single 2000 map. The possible error at 1990 was ignored by Feranec et al. (2010) though no explanation was provided.

Higher single map accuracies are expected to create higher accuracy in the map of change (Feranec et al. 2010), but if the available accuracy information is unavailable or imprecise, then how can we evaluate the influence of classification errors on land change assessments? The answer depends on the method of assessment. Aldwaik and Pontius (2012) proposed a method for land change assessment, called Intensity Analysis, which computes deviations between observed changes and uniform changes. Aldwaik and Pontius (2013) give a method to compute the minimum hypothetical error that could account for those deviations between observed changes and uniform changes. The goal of this article is to compute the size of the hypothetical errors that could account for the deviations between the observed intensities and the corresponding uniform intensity and discuss the implications of the identification of misclassification errors on the assessment of hydrologic processes. If the hypothetical errors are small, then the evidence for deviations from uniform intensities is weak. The hypothetical errors provide a framework to explain deviations that no known historical processes of change can explain.

We use the CLC1990 and CLC2006 maps of eight land categories to examine change in the Mondego river basin, which is a coastal watershed in Portugal. In a previous study of the same Basin, Teixeira et al. (2014) applied a methodology proposed by Pontius et al. (2004), but changes at the category level were not evaluated, accuracies of the single CLC maps was ignored and the disagreement between maps of different time points was the only statistic reported. This previous study was important for the management of land-water interaction, therefore we use the same data to demonstrate the ability of Intensity Analysis and its associated error measures to interpret land change patterns. The results from Teixeira et al. (2014) indicated lack of uniformity for the transition intensities across losing categories, given a particular category’s gain, and across gaining categories, given a particular category’s loss. Our present manuscript refrains from analysing transition intensities across gaining categories, thus refrains from using the word “systematic”, based on the advice of Pontius et al. (2013) and Enaruvbe and Pontius (2015).

CORINE Land Cover

The CLC1990_PT (for Portugal), hereafter just CLC1990, was produced based on satellite images from the years 1985/86/87. It was initially a pilot project and experienced several modifications since it was first released. The accuracy of the CLC1990 map was assessed, but the assessment report is not available. Büttner et al. (2012) state that a thematic accuracy of at least 85% was probably not achieved. Between 2002 and 2005, geometric and thematic corrections were implemented and a revised product was generated (Caetano et al. 2006). Though the CLC1990 base data corresponds to three different time points, we assumed that the changes that might have occurred during the three consecutive years are negligible. Land persistence tends to dominate landscapes (Pontius et al. 2004) and the Mondego river basin is an example of this phenomenon (Teixeira et al. 2014).

The CLC2006_PT, hereafter just CLC2006, was produced following a first approach to change mapping (Caetano et al. 2009). According to this approach, the revision and correction of the CLC2000 map was performed simultaneously with the visual interpretation of images at both 2000 and 2006. Afterwards, the CLC2006 map was derived based on the intersection of the revised CLC2000 and the change map (CLC-Changes$_{2000\text{-}2006}$) (Büttner et al. 2004, Caetano et al. 2009). Delineation of changes was based on the polygons of the CLC2000 map to avoid the creation of sliver polygons when performing intersection (Caetano et al. 2009). The Portuguese CLC2006 map has an estimated overall accuracy percentage of 90.2±1.3 at the 95% confidence level (Caetano et al. 2009). Regarding user's accuracies, only five categories (231, 132, 222, 313 and 332) out of the forty-four level 3 categories have 95% confidence intervals that lie completely below the minimum goal of 85 percent. Regarding producer's accuracies, only two categories (231 and 423) have 95% confidence intervals entirely below 85 percent.

One of the key issues affecting land cover configuration and composition is the minimum mapping unit (MMU). The MMU of CLC maps is set to 25ha, which is attained through generalization procedures (Caetano et al. 2009, EEA 2007). Likewise, the minimum distance between lines is set to 100m. Spatial generalization reduces the complexity of the data structure influencing both location and attribute accuracy, as a result of reclassification, aggregation, amalgamation (technical-unreal change) and boundary smoothing and simplification (Congalton 1997). Despite the expected errors associated with these procedures, the methodology followed by the CORINE project proved to be successful for the location accuracy and reasonably controlled for the attribute accuracy (EEA 2007).

The raster format of the CLC maps was used in this study. Though the conversion from vector to the raster might be another source of error, the positional errors near boundaries (Foody 2002) introduced by the raster model used to generate the CLC raster files are not substantial, because the pixel size (1ha) is much smaller than the MMU (25ha) (EEA 2007).

Intensity Analysis

Intensity Analysis compares observed intensities of changes to a uniform intensity (Aldwaik and Pontius 2012, 2013; Pontius et al. 2013). Intensity Analysis has three levels of detail: interval, category and transition. The interval level examines overall change during each time interval for cases with more than two time points. The category level examines each category's gross gains and gross losses. The transition level examines intensities of transitions from losing categories to any particular gaining category. Each level has its own uniform hypothesis that the change intensity is uniform across intervals, categories and transitions.

Intensity Analysis quantifies the deviation between an observed intensity and the uniform intensity. Map error might account for the calculated difference between an observed intensity and the uniform intensity. If the observed intensity is greater than the uniform intensity, then the data show more change than the uniform hypothesis implies and thus a hypothetical commission error is calculated. Commission error intensity is 100% minus User's accuracy. In contrast, if the observed intensity is smaller than the uniform intensity, the data show less change than the uniform hypothesis implies and thus a hypothetical omission error is calculated. Omission error intensity is 100% minus Producer's accuracy. Larger hypothetical errors give weaker evidence that true change is uniform.

Mondego river basin case study

The Mondego river basin is located in the central region of Portugal, Europe (Figure 1). The studied river basin has an area of 6658 square kilometers and a NE–SW orientation. It

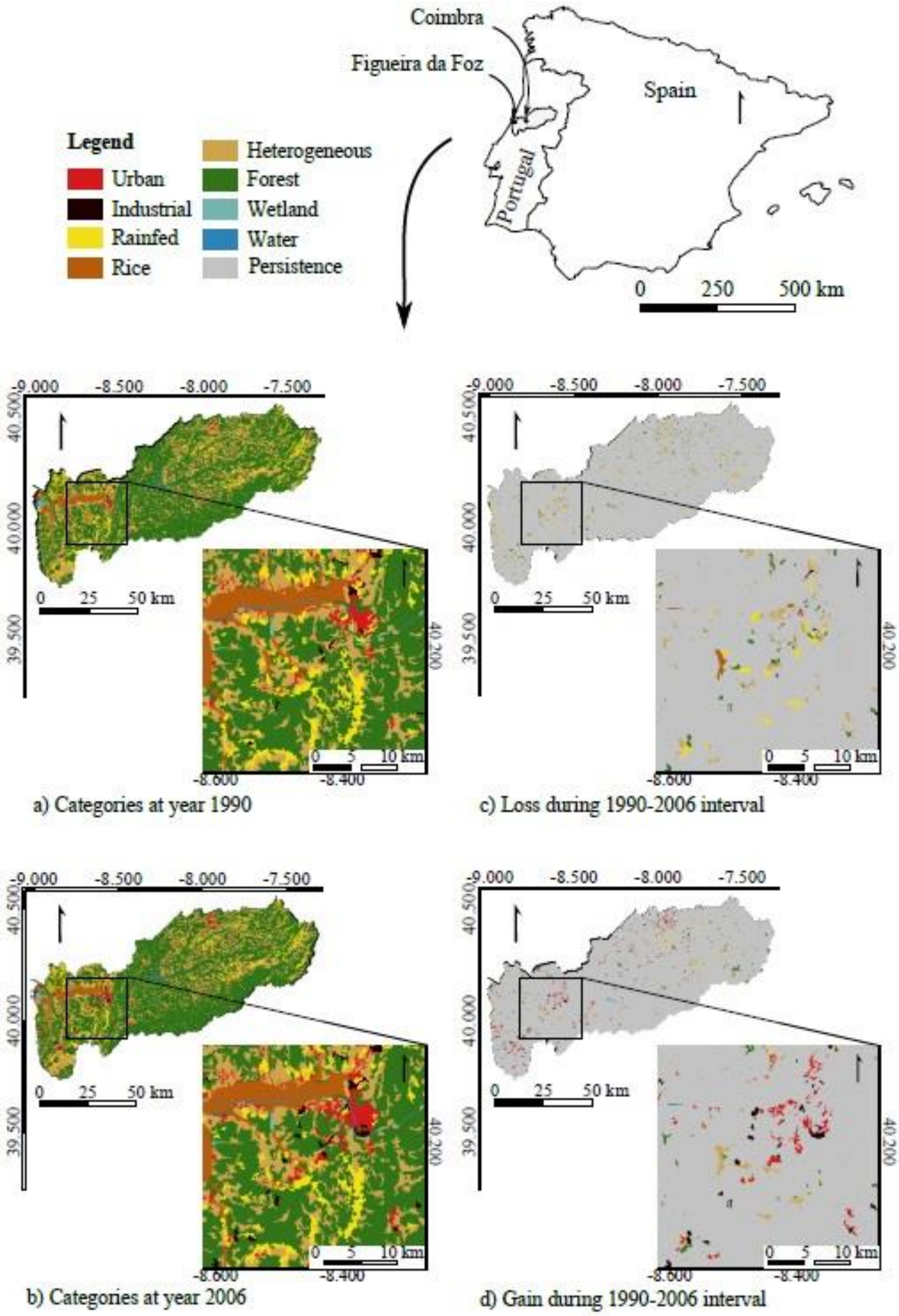


Figure IV.1.
Map in the upper right shows the location of the study site in the centre of Portugal. The study site is the Mondego river basin. The four lower maps on the left show land-cover categories at 1990 and 2006 in the entire river basin and in a small portion of the study site. Maps on the right show changes during 1990-2006 in the entire river basin and in a small portion of the study site.

encompasses 36 municipalities with an estimated population of 165 inhabitants per square kilometer (INE 2014). The river basin is occupied mainly by agricultural and forest areas that are distributed throughout the basin, whereas urban and industrial areas are located mainly on the coastal strip (Teixeira et al. 2014). Coimbra and Figueira da Foz are two of the most populated municipalities, and they play an important role within the Mondego river dynamics because they have grown along the river margins.

Methodology

Land cover dataset

The analysis was performed using CORINE Land Cover raster data, resolution 100×100m, for the 1990 and 2006 inventories (EEA 2012). The same eight categories analysed by Teixeira et al. (2014) were evaluated in the present manuscript: urban (Urban - U), industrial (Industrial - I), rainfed and permanent crops (Rainfed - R), permanently irrigated and rice fields (Rice - P), heterogeneous agriculture (Heterogeneous - H), forest (F), wetland (W) and water bodies (Water - B). Teixeira et al. (2014) describe how the 44 CORINE categories were reclassified to these eight categories.

Intensity Analysis and hypothetical error

A transition matrix was produced for the time interval 1990-2006. Table IV.1 gives the Mathematical notation. Equation 1 gives overall change during the time interval.

Table IV.1.
Mathematical notation for Intensity Analysis

Symbol	Meaning
J	number of categories
i	index for a category at the interval's initial time point
j	index for a category at the interval's final time point
n	index for the gaining category for the selected transition
Cij	number of pixels that transition from category *i* to category *j*
S	overall change as percent of the spatial extent, which equals the uniform intensity for the category level
Gj	intensity of gain of category *j* relative to size of category *j* at final time
Li	intensity of loss of category *i* relative to size of category *i* at initial time
Rin	intensity of transition from category *i* to category *n*, relative to size of category *i* at initial time where *i≠n*
Wn	uniform intensity of transition from all non-*n* categories to category *n*, relative to size of all non-*n* categories at the initial time

$$\boldsymbol{S} = \frac{\sum_{j=1}^{J}\left\{\left(\sum_{i=1}^{J} C_{ij}\right) - C_{jj}\right\}100\%}{\sum_{j=1}^{J}\sum_{i=1}^{J} C_{ij}} \quad (1)$$

Category level

Equation 2 gives gross gain intensities and Equation 3 gives gross loss intensities. These were compared to the uniform intensity of change from Equation 1. If all land categories were to gain and to lose with the same intensity given the size of overall change, then category gain intensities (*Gj*) and loss intensities (*Li*) would equal the overall intensity *S*. If *Gj* > *S*, then category *j* is an active gainer; and if *Li* > *S*, then category *i* is an active loser. If *Gj* < *S*, then category *j* is a dormant gainer; and if *Li* < *S*, then category *i* is a dormant loser.

$$\boldsymbol{G_j} = \frac{\{area\ of\ gain\ of\ j\}100\%}{area\ of\ j\ at\ t{+}1} = \frac{\left\{\left(\sum_{i=1}^{J} C_{ij}\right) - C_{jj}\right\}100\%}{\sum_{i=1}^{J} C_{ij}} \quad (2)$$

$$\boldsymbol{L_i} = \frac{\{area\ loss\ of\ i\}100\%}{area\ i\ at\ t} = \frac{\left\{\left(\sum_{j=1}^{J} C_{ij}\right) - C_{ii}\right\}100\%}{\sum_{j=1}^{J} C_{ij}} \quad (3)$$

If *Gj* > *S*, then equation 4 gives the hypothesised commission error intensity of category *j* at the final time. If *Gj* < *S*, then equation 5 gives the hypothesised omission error intensity of category *j* at the final time.

$$\textbf{Commission of } j \textbf{ intensity at final time} = \frac{\left(\sum_{i=1}^{J} C_{ij}\right)(G_j - S)/100\% - S}{\left(\sum_{i=1}^{J} C_{ij}\right) - C_{jj}} \mathbf{100\%} \tag{4}$$

$$\textbf{Omission of } j \textbf{ intensity at final time} = \frac{\left(\sum_{i=1}^{J} C_{ij}\right)(S - G_j)/100\% - S}{\left\{\left(\sum_{i=1}^{J} C_{ij}\right) - C_{jj}\right\} + \left\{\left(\sum_{i=1}^{J} C_{ij}\right)(S - G_j)/100\% - S\right\}} \mathbf{100\%} \tag{5}$$

If $Li > S$, then equation 6 gives the hypothesised commission error intensity of category *i* at the initial time. If $Li < S$, then equation 7 gives the hypothesised omission error intensity of category *i* at the initial time.

$$\textbf{Commission of } i \textbf{ intensity at initial time} = \frac{\left(\sum_{j=1}^{J} C_{ij}\right)(L_i - S)/100\% - S}{\left(\sum_{j=1}^{J} C_{ij}\right) - C_{ii}} \tag{6}$$

$$\textbf{Omission of } i \textbf{ intensity at initial time} = \frac{\left(\sum_{j=1}^{J} C_{ij}\right)(S - L_i)/100\% - S}{\left\{\left(\sum_{j=1}^{J} C_{ij}\right) - C_{ii}\right\} + \left\{\left(\sum_{j=1}^{J} C_{ij}\right)(S - L_i)/100\% - S\right\}} \tag{7}$$

Transition level

We consider the transition from an arbitrary category *i* to a particular gaining category *n*. Equation 8 gives the observed intensity of transition from *i* to *n* relative to the size of *i* at the initial time. Equation 9 gives the hypothesised uniform intensity for the gain of category *n*. If *n* were to gain with the same intensity from all non-*n* categories, then the uniform intensity *Wn* would equal *Rin* for all *i*. If $Rin > Wn$, then the gain of *n* targets *i*. If $Rin < Wn$, then the gain of *n* avoids *i*.

$$\boldsymbol{R_{in}} = \frac{\{\textbf{area of transition from i to n}\}100\%}{\textbf{area of i at t}} = \frac{\{C_{in}\}100\%}{\sum_{j=1}^{J} C_{ij}} \tag{8}$$

$$\boldsymbol{W_n} = \frac{\{\textbf{area of gain to n}\}100\%}{\textbf{area of j at t+1}} = \frac{\left\{\left(\sum_{i=1}^{J} C_{in}\right) - C_{nn}\right\}100\%}{\sum_{j=1}^{J}\left\{\left(\sum_{i=1}^{J} C_{ij}\right) - C_{nj}\right\}} \tag{9}$$

If $Rin > Wn$, then equation 10 gives the hypothesised commission error intensity of category *i* at the initial time that could account for the deviation from uniform. If $Rin < Wn$, then equation 11 gives the hypothesised omission error intensity of category *i* at the initial time that could account for the deviation from uniform.

$$\boldsymbol{Commission\ of\ i\ intensity\ at\ initial\ time} = \frac{\left(\sum_{j=1}^{J} C_{ij}\right)(R_{in}-W_n)/100\% - W_n}{C_{in}} \mathbf{100}\% \quad (10)$$

$$\boldsymbol{Omission\ of\ i\ intensity\ at\ initial\ time} = \frac{\left(\sum_{j=1}^{J} C_{ij}\right)(W_n-R_{in})/100\% - W_n}{\left\{\left(\sum_{j=1}^{J} C_{ij}\right)(W_n-R_{in})/100\% - W_n\right\} + (C_{in})} \mathbf{100}\% \quad (11)$$

Results

Category level

Gain. Figure IV.2-a shows the gain intensity of each category. Five categories are active gainers: urban, industrial, rainfed, wetland and water. The remaining categories are dormant gainers: rice, heterogeneous and forest. Figure IV.2-b shows a segmented bar for the gain of each category. If the category is active, then the size of the gain is the sum of its two segments, which would be its black segment and a commission segment. If the category is dormant, then the size of the gain is the black segment, which is accompanied by an omission segment. For example, if change were uniform, then urban would have gained 301 pixels, but the observed urban gain was 7986 pixels. The difference could be explained by commission of urban error on 7685 pixels at 2006. If change were uniform, then forest would have gained 16873 pixels, but forest actually gained 5176 pixels. The difference could be explained by omission of forest error on 11697 pixels at 2006. We assume each pixel of error is commission of an active category and omission of a dormant category, thus the size of all the commission errors equals the size of all the omission errors in figure IV.2-b. Errors on 2.0% of the 2006 map could account for all deviations from uniform gains.

Loss. Figure IV.2-c shows the loss intensity of each category. Three categories are active losers: industrial, rainfed, heterogeneous. The remaining categories are dormant losers. Figure IV.2-d shows a segmented bar for the loss of each category. If the category is active, then the size of the loss is the sum of its two segments, which would be its black segment and a commission segment.

If the category is dormant, then the size of the gain is the black segment, which is accompanied by an omission segment. For example, if change were uniform, then forest would have lost 16873 pixels, but forest actually lost 7396 pixels. Omission of forest error on 9477 pixels at 1990 could account for the difference. Errors on 1.5% of the 1990 map could account for all deviations from uniform losses.

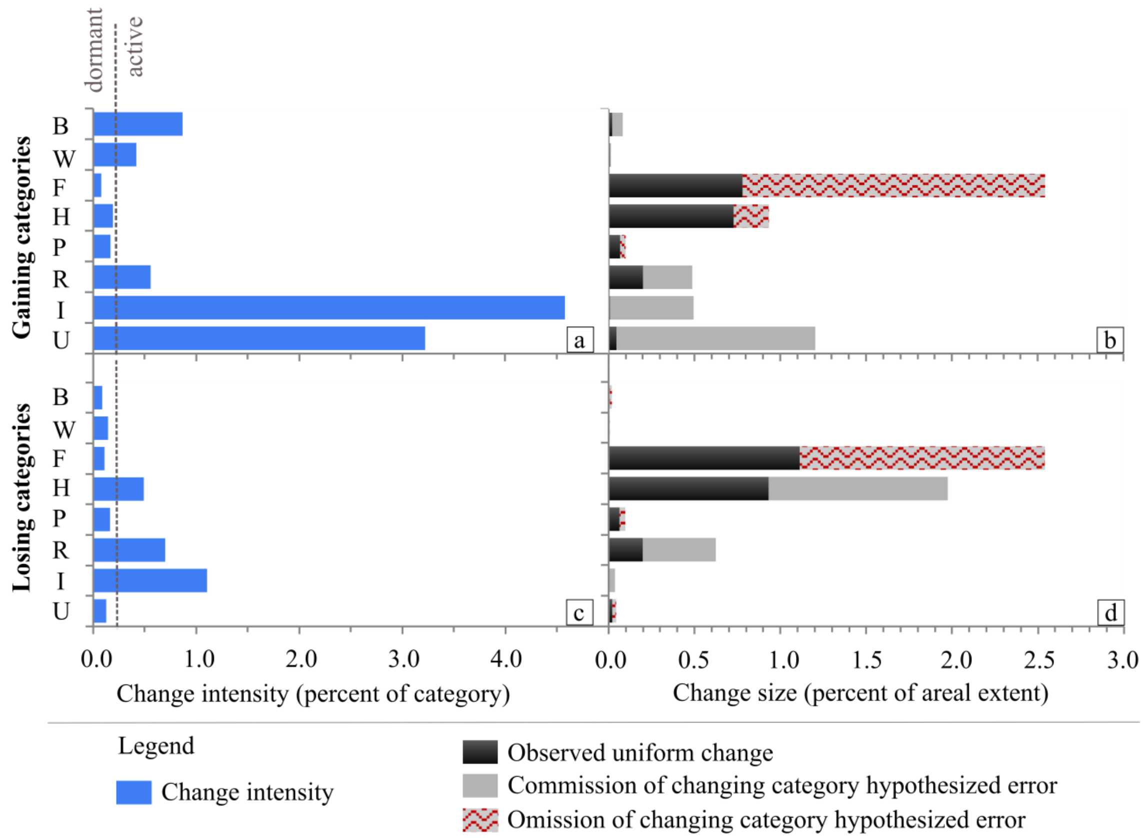


Figure IV.2.

Intensity of a) gains and c) losses by category. Gain intensity is a percentage of the category at 2006, while loss intensity is a percentage of the category at 1990. The dashed line is the uniform change intensity. Sizes of the gains and losses along with hypothetical errors that could account for deviations from uniform category level b) gains and d) losses. U-urban, I-industrial, R-rainfed and permanent crops, P-Permanently irrigated and rice fields, H-heterogeneous agriculture, W-wetland, B-water bodies.

Transition level

Figure IV.3 shows the results of the transition level analysis for each gaining category. Each gaining category has a pair of graphs. Figures IV.3a-b show graphs for urban gain, figures IV.3c-d show graphs for industrial gain, etc. The first graph in each pair shows the transition intensity, while the second graph in each pair shows the transition size. If the transition intensity from a particular losing category is greater than the uniform intensity, then we say the gaining category targets that particular losing category. If the transition intensity from a particular losing category is less than the uniform intensity, then we say the gaining category avoids that particular losing category. For example, figure IV.3-a shows that the gain of urban targets industrial, rainfed and heterogeneous, while avoids the remaining categories.

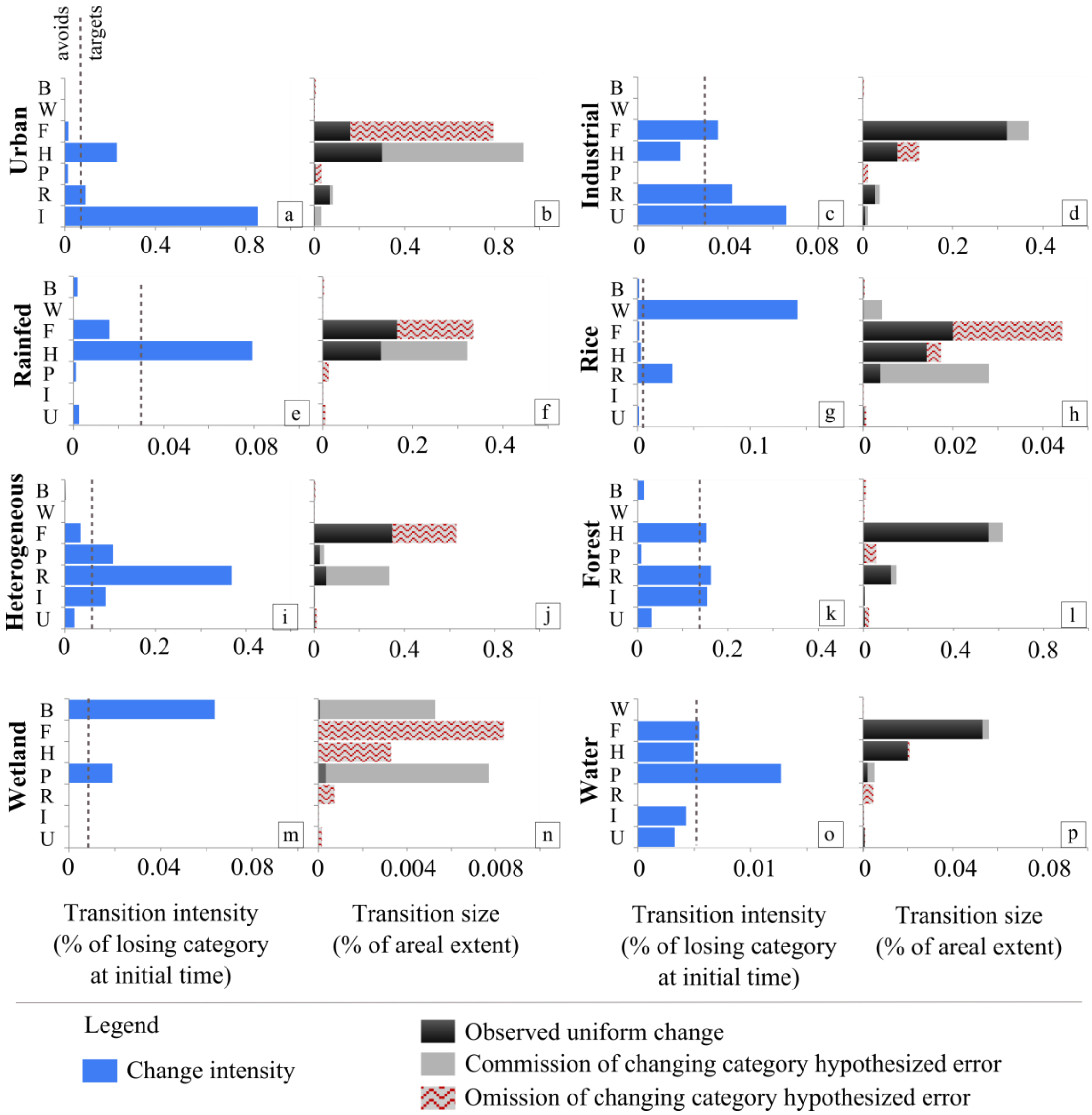


Figure IV.3.

Transition intensity for the gain of: a) urban, c) industrial, e) rice, g) rainfed, i) heterogeneous, k) forest, m) wetland, o) water. The vertical line indicates the uniform transition intensity, given the category's gain. Sizes of the transitions along with hypothetical errors at 1990 that could account for deviations from uniform intensities for the gain of: b) urban, d) industrial, f) rice, h) rainfed, j) heterogeneous, l) forest, n) wetland, p) water. U-urban, I-industrial, R-rainfed and permanent crops, P-Permanently irrigated and rice fields, H-heterogeneous agriculture, W-wetland, B-water bodies.

The graphs concerning transition size indicate the hypothetical errors in the 1990 map that could account for deviations between uniform and observed transition intensities. Figure IV.3-b shows that hypothetical error on 0.7% of the areal extent could account for deviations from uniform transitions to urban, where the errors are simultaneously commission of industrial, rainfed and heterogeneous and omission of rice, forest, wetland and water. For each other gaining category, errors on less than 0.7% of the areal extent could account for all other deviations from uniform transitions.

Discussion

Counter-intuitive results

Results indicate that the urban category is active in terms of gains. Data from the National Statistical Institute indicate that house holding increased in Portugal, at least, since 2001 (INE 2014), which is expected in a context of rapid economic growth, such as the one occurred in Portugal between 1990 and 2006 (Amaral 2011). Given what we know about urban development and its most probable trends, we consider it plausible that urban is truly active in gains, regardless of possible map error. However, counter-intuitively, the transition level intensities show that the gain of urban seems to target industrial more than heterogeneous or rainfed. We expect urban areas to grow more from heterogeneous because developers tend to overtake farming or rural open space around major growth centres (Delbecq and Florax 2010, Ives and Kendal 2013). In fact, the hypothesised commission error at 1990 of heterogeneous is larger than the hypothesized commission error of rainfed and industrial at 1990 (Figure 3-b), meaning that there is evidence that the gain of urban targets heterogeneous more strongly than the gain of urban targets rainfed or industrial. Also, due to spectral similarity, urban and industrial are categories that could be easily confused during the mapping process (Su and Du 2012). It is plausible that the river basin has substantial omission error of urban and commission of industrial in the 1990 map. In such situation, the commission of industrial at 1990 on 0.002% of the areal extent could account for urban appearing to target industrial. This 0.002% hypothetical error would account for 92.2% of the apparent transition from industrial to urban. This could also explain the counter-intuitive result of industrial active loss, which is unexpected in a context of rapid-economic growth. In this case, the hypothetical commission error of industrial at 1990 on 0.031% of the areal extent could account for industrial's loss appearing active. This hypothetical error would account for 81.3% of industrial apparent loss. Results also indicate that industrial is active in terms of gains. The active gain of industrial is consistent with the period of prosperity observed in Portugal after the adhesion to the European Economic Community in 1986 (Duarte et al. 2013), which tended to favour processes of change to industrial from forest and, at some degree from agriculture (Batista e Silva et al. 2014). However, the transition level intensities (Figure 3-c) indicate that the gain of industrial targets urban, followed by rainfed and forest, though we would expect industrial to target forest more than urban. The hypothetical commission errors at 1990 seem to support this hypothesis, because we observe larger commission error for forest (Figure 3-d) indicating that the gross gain of industrial derives more from forest than from urban or rainfed. It is again plausible to assume that urban and industrial have been confused in the maps (Su and Du 2012), and that the commission error at 1990 on 0.007% of the areal extent could account for industrial appearing to target urban. This 0.007% hypothetical error would account for 53.3% of the apparent transition from urban to industrial.

The active gain of water is another counter-intuitive result considering the temporal changes in the Mondego river basin. Results indicate that the gain of water targets forest, which would be expected with the construction of dams. In fact, the set of interventions under the Hydraulic Harnessing plan for the Mondego basin included the construction of upstream dams for flood control and power generation (LNEC 2012). However, those dams began to operate before 1986 and the CORINE Land Cover images for the 1990 map were taken between 1987 and 1989 (Caetano et al. 2009). It is plausible that the map of 1990 has substantial omission of water and commission of forest error. Results indicate that error at 1990 on 0.003% of the areal extent could account for water appearing to target forest. This 0.003% hypothetical error would account for 5.6% of the apparent transition from forest to water. Given the location of the transition, it is evident that the gain of water is related to the construction of the Fronhas dam in a forested area (LNEC 2012). This dam began operating in 1985, so we would expect that its reservoir would already be identified in the 1990 map, but this is not the case. We hypothesise that the Fronhas water reservoir had, by the time the CORINE Land Cover images for the CLC 1990 map were taken, a total area smaller than 25ha and a width smaller than 100m, i.e., smaller than the CORINE MMU and the smallest mapped width. As a result, the Fronhas reservoir was not identified in the 1990 map and thus perhaps truly is error consisting of omission of water and commission of forest.

Finally, results show that heterogeneous is dormant in terms of gains, though the gain of heterogeneous targets industrial, rainfed and rice. Our understanding of historical processes indicates that heterogeneous category could either emerge from forest if a part of it would be converted to agriculture; or from agriculture if a part of its area was abandoned and left intact for natural recovery. Processes of change that can lead to the gain of heterogeneous areas are highly plausible because crop abandonment and crop size reduction were a reality in the Mondego river basin during the period of implementation of severe measures of the European Common Agricultural Policy (CAP) (EC 2003). Results indicate that omission of heterogeneous error at 2006 on 0.205% of the areal extent could account for the gain of heterogeneous appearing dormant.

Error Analysis

Büttner et al. (2004) indicate that the error of CLC1990 maps could be close to 15% or be even higher. Caetano et al. (2009) registered an error of 9.8% for the Portuguese CLC2006 map. Teixeira et al. (2014) used the same CORINE land-cover maps and registered, for the Mondego river basin, a total disagreement between 1990 and 2006 of 2.0%. Are the CORINE maps sufficiently accurate such that the temporal differences indicate true deviations from uniform change? The results show that: error on 1.5% of the 1990 map could potentially explain all deviations from uniform losses, error on 2.0% of the 2006 map could potentially explain all deviations from uniform gains, and error on less than 0.7% of the 1990 map could potentially explain all deviations from uniform transitions to each gaining category. These hypothetical errors are smaller than the amount of error we suspect in the maps, in which case map error might be able to explain the apparent temporal changes. However, we will never be certain whether error explains all deviations from uniform changes. It is easy to imagine how error could explain apparent changes that are inconsistent with supplemental historical information. However, some of the apparent changes are consistent with supplemental information concerning land change history. Thus we do not automatically assume that error accounts for all deviations from uniform changes, because some apparent changes are consistent with our understanding of historical processes.

CORINE maps as base data

The results give information concerning land cover changes at the category and transition levels using two CLC data layers, the CLC1990 and the CLC2006. The specifications of all CLC products are similar, though some improvements have been made since the CLC1990 (EEA 2007). Such specifications influence the final map accuracy and interpretation of the results.

Thematic accuracy. Thematic accuracy is the correspondence between the category label assigned by the classification and that observed in reference information. Higher thematic accuracies are expected to positively influence the identification of land cover changes (Feranec et al. 2010). Regarding our case study, the information concerning the CLC1990 accuracy is vague and has probably not achieved the 85% target (Büttner et al. 2004). Our analysis measured the strength of the evidence for the deviations from uniform intensities, in order to shed light on the categories whose errors in the CLC1990 map could account for non-uniform transitions. We found that it is plausible that the error of the CLC1990 map might account for non-uniform transitions from urban to industrial and from water to forest.

The assessment of the thematic accuracy of the CLC2006 has shown an overall accuracy of 90.2% for the Portuguese territory, though some land categories show very low user and/or producer accuracy (Caetano et al. 2009). In our study, we assumed that the omission error of heterogeneous at 2006 could account for the gain of heterogeneous appearing dormant. According to Caetano et al. (2009), the omission error at 2006 of the five individual categories that compose the heterogeneous category varies between 48.1% and 12.2%. The high omission errors reported by Caetano et al. (2009) could be negatively influencing the ability to identify land change (Feranec et al. 2010).

Minimum mapping unit. Despite the accuracy attained for the CLC maps in terms of both location and attribute (EEA 2007), larger MMU may lead to misrepresentation of sparse and fragmented land cover categories (Saura 2002). In the Mondego river basin, forest and heterogeneous agriculture are dominant categories, which tend to occupy large continuous areas

(Teixeira et al 2014). On the other hand, urban and industrial areas are sparse and fragmented, especially in the rural areas, which tend to be interspersed with natural and agricultural areas (Mateus 2009). As a consequence, generalization may have led to underestimation (Büttner et al. 2004) of urban and industrial, as well as other less dominant land categories, because isolated areas smaller than 25ha were not incorporated in the final map. Generalization is the most probable explanation for water's gain appearing to target forest.

Implications for coastal management

The apparent changes that are inconsistent with historical processes are an indication of the misclassification errors that could propagate to other land cover map applications, as in the assessment of processes affecting water supply, demand and quality (Loosvelt et al. 2014). In our case study, results indicate that there is evidence that the gain of urban targets heterogeneous more strongly than industrial. The three categories affect the hydrological processes of runoff, infiltration and groundwater recharge, but urban and industrial strongly reduce the permeability of soils. Thus, transitions from heterogeneous to urban are expected to have larger effects on water supply than transitions from industrial. Likewise, the three categories affect water quality, but the type of impact expected from each category is different. From urban growth we expect an impact on water from higher concentrations of ammonia; from industrial growth we expect impacts from ammonia and chemical contaminants; and from agricultural runoff we expect impacts due to higher concentrations of nitrate and phosphate.

Coastal and estuarine systems are highly dynamic, with complex interactions and feedback loops. Developing management plans for these areas highly depends on a clear understanding of the problem to be addressed as well as on the degree of certainty (Townend 2004). The counter-intuitive results concerning urban, industrial and heterogeneous are an example of how our perception of the problems affecting coastal systems could change with consequences on coastal management.

Conclusion

The processes that influence water supply, water demand and water quality are affected by land change, whose assessment can benefit from the use of Intensity Analysis. Intensity Analysis' approach to compute hypothetical errors provides a structure to evaluate the strength of the evidence for deviations from uniform intensities. Larger hypothetical errors indicate stronger evidence. All apparent deviations from uniform gains could be explained by errors on 2.0% of the 2006 map. All apparent deviations from uniform losses could be explained by errors on 1.5% of the 1990 map. All apparent deviations from uniform transitions to each gaining category could be explained by errors on less than 0.7% of the 1990 map. The map of 1990 is different than the map of 2006 on 2.0% of the areal extent. These percentages are lower than the overall error percentage of 9.7±1.3% found for the CLC2006 (Caetano et al. 2009) and lower than the hypothesized error percentage of 15% for the CLC1990 (Büttner et al. 2004). We analysed the processes of changes that are known to have occurred in our case study in order to interpret whether the hypothetical errors could account for deviations from uniform intensities. We found that some apparent changes are consistent with the supplemental historical record concerning land change processes, in which case errors are not necessarily the reason of the apparent changes. However, errors that confuse urban and industrial might account for the counter-intuitive apparent transitions between urban and industrial. Omission of heterogeneous error at 2006 could account for the unexpected observation that the gain of heterogeneous appears dormant. Generalization procedures for the CLC1990 map might explain the apparent transition from water to forest. The method to quantify hypothetical errors has allowed us to explain counter-intuitive land changes that the raw data indicate but that no known historical processes can explain. Interpreting inconsistent transitions supports the identification of pressures in coastal areas.

General discussion

This section of the thesis synthesizes and discusses its contribution to increase the knowledge on the use of land cover as indicator within the Driver-Pressure-State-Impact-Response framework. Major findings will be discussed in light of current progresses regarding indicator development, in light of new evidences regarding the concentrations of nutrients and oxygen-consuming-substances in European waters, and in light of European policies aiming to reduce pollution in surface waters.

The use of indicators and the role of land cover

Assessing progress

Assessment and performance indicators provide information into the state of the environment and allow us to analyze progress in meeting targets (EEA 2014). In Chapter I indicators of physico-chemical characteristics have been applied to transitional and freshwater systems, namely, nutrients and oxygen-consuming-substances. Both are included in the core set of indicators selected by the European Environment Agency (EEA 2014), and similar indicators are defined by the OECD (OECD 2003), the UNSD and the UNECE (UNECE 2007) as key environmental indicators for the assessment of freshwater and/or transitional and marine waters. Comparing time intervals, Chapter I evaluates whether progress has been achieved in reducing the concentrations of ammonia, nitrate and phosphate. In the estuary, progress is evaluated by comparison to a previous time interval, after the implementation of mitigation measures. In the tributaries, progress is evaluated by comparison to European average concentrations in three years, 1992, 2000, 2012. Indicators within the DPSIR framework, alone, lack capacity to capture trends (Gari et al. 2015). But comparing performance across time and space is an effective mean to evaluate progress allowing the analysis of trends, as long as assessment is provided at different time steps, or regions.

Assessing the progress of chemical parameters is part of the process to evaluate the ecological status of transitional and freshwater systems, which is assessed by evaluating both physical-chemical and biological characteristics of each water body, as defined by the WFD (EC 2000). Due to the complexity of ecological processes, changes resulting in high concentrations of nutrients, as was the case for the Mondego system, do not necessarily mean low ecological status. Other factors of the system may contribute to maintain the ecosystem integrity, such as hydrodynamics (Martins et al. 2001). However, high concentrations of nutrients may reduce the resilience of the systems, in the sense that, even a slight change in some other key factor contributing to ecological integrity, may cause the system to change “function”, as it occurs when severe symptoms of eutrophication appear (Marques et al. 2003, Dolbeth et al. 2007).

Effectiveness of measures

The mitigation measures implemented in the Mondego estuary from 1998 to 2006 were able to eliminate serious eutrophication symptoms in the south arm, but the system remained a “Potential Problem Area” (Lillebø et al. 2007). Included in the set of interventions were a) the diversion of river freshwater inputs from the south arm sluice to the north arm sluice and b) the re-establishment of the connection between the two arms. To evaluate whether freshwater inputs were causing changes in the concentrations of chemical parameters of the estuary we established a relationship between the estuarine chemical parameters and other water parameters (salinity, precipitation and water temperature). We assumed that variations in salinity and water temperature were related to inputs of freshwater (both precipitation and from rivers) and used the relationship as an indicator of pressure from nutrient loadings.

Comparing the relationship in two different periods enabled us to evaluate the effectiveness of the policy response in reducing pressure from nutrient loadings after more than 10 years of the first mitigation measures in 1998. Such indicators that relate change in environmental variables to policy measures are commonly referred to as policy-effectiveness indicators (EEA 2014) and allow monitoring whether a specific action is having an effect in reducing, or solving, an environmental problem. These indicators provide a link between response indicators and state, driving force, pressure or impact indicators and can be instrumental in changing policies. In our case, the societal response (i.e., diversion of freshwater to northern sluice and re-establishment of connection) was linked to pressure indicators (i.e., nutrient loadings).

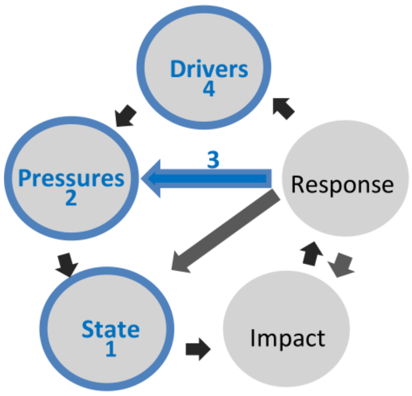
The development of this type of indicators is still ongoing and several reasons might explain the delay in the production of this type of indicators. The EEA, for instance, does not describe policy-effectiveness indicators to evaluate water quality progress in light of the implementation of policy measures (EEA 2014). One of the reasons for the low development of this type of indicators might be related to the frequent ambiguity of the policy goals and objectives. Wilson and Buller (2001) described this problem within the scope of the evaluation of Agri-Environment Regulation (EC 1992, EC 2010). In the context of environmental assessment, and more precisely, ecological quality of waters, policy "ambiguity" can be explained by the natural variability of ecosystems that makes it difficult to definite goals based on thresholds (Irvine 2004). This is closely linked to scale, in the sense that policy measures defined at the European level ought to be suitable to be applied at local and regional levels (Irvine 2004). Likewise, the production of policy-effectiveness indicators must account for the high variability of systems at the European level. Another reason might be related to the fact that indicators may reveal trends, but do not necessarily explain them. The policy-effectiveness indicator applied in this thesis to evaluate whether freshwater inputs remained a source of pressure to the estuary is such an example. Explaining the causes for the high concentration of nutrients - and more specifically of phosphate in the Mondego estuary and tributaries - remains necessary in order to outline appropriate measures. A third, and last reason, could be related to the necessity to include socio-cultural policy goals when evaluating the effectiveness of policies. This problem is less important when evaluating a particular action (e.g. re-establishment of the connection between the two arms to historical characteristics). But policies defined at the national and European level, are usually defined based on equilibrium among environmental, social and economic development issues and all of which must be accounted for when evaluating the success of a particular policy (Wilson and Buller 2001).

Assessing pressure and driving forces from land cover maps

Excess of nutrients in water are frequently a result of agricultural runoff and urban drainage and, thus, land cover is an indicator of pressure regarding diffuse-source pollution (EC 2000). In this study, we assessed the suitability of land cover and land cover patterns to describe the spatial variability of nitrate across the river basin (complete datasets for ammonia and phosphate were not available). The percentage occupied by agriculture, urban and industrial areas was used as an indicator of nutrient runoff, whereas the percentage of forest was included as an indicator of watershed health as it prevents storm water runoff and promotes groundwater recharge. Landscape patterns, measured by landscape metrics, were included as indicators of fragmentation. It was hypothesized that higher fragmentation results in land degradation with negative impacts on the hydrologic processes controlling the transport of nutrients from land to water, which in turn would lead to higher nutrient loadings. Land cover emerged as a potential indicator of efficiency of long-term policy measures in reducing nitrate release to water, namely the EU Nitrate Directive (EC 1991b). Lack of relationship could reveal successful outcomes from the implementation of measures, but further research to confirm such conclusion is necessary.

The state of ecosystems may suffer modifications as a result of change in pressures, which in turn may be altered due to changes in the driving forces. Land cover, an indicator of pressure, is influenced by changes in the over-arching drivers of social and economic development, which in this study were accounted for using processes of landscape change as proxies. These are broad categories of landscape change such as urbanization, afforestation and land restoration (Chapter III). The methodology was able to identify the dominant processes of landscape change, revealing changes in the driving forces of water quality in the Mondego river basin. However, land cover maps, used to identify land cover changes, have an associated error that might be able to account for the observed land changes. If this occurs, the uncertainty associated to the identified processes of landscape change is high. In Chapter IV we applied a method to evaluate the suitability of the CORINE land cover maps as indicators of land change. CORINE land cover datasets are a product from the EEA, freely available and widely used for assessments in Europe. Assessing the suitability of these maps to detect deviations from uniform changes, discussing its limitations and underlying possible reasons for systematic errors, is an asset for all CORINE users, but the methodology employed is suitable for all land cover maps and thus appropriate for all land change assessments.

The present study has provided valuable information to support the assessment and evaluation of changes of drivers (Chapters III and IV), pressures (Chapter I and Chaper II) and state (Chapter I) of water quality on coastal watersheds (Figure 4). The major focus was on the ability of land cover maps as a tool to support the identification of driving forces and pressures of surface water quality, complementing data previously provided by others (Pinto et al. 2013).



1 State change assessment indicators

- (Chapter I)Nutrients in transitional waters: nitrate and phosphate
- (Chapter I)Nutrients in freshwater: nitrate and phosphate
- (Chapter I)Oxygen-consuming-substances: Ammonia

2 Pressure assessment indicators

- (Chapter I) Nutrient loadings into estuary: Freshwater inputs from rivers
- (Chapter II) Nutrient runoff: % of agriculture, urban and industrial
- (Chapter II) Watershed health: % of forest (complementary indicator – reveals lack of pressure)
- (Chapter II) Landscape fragmentation: landscape metrics

3 Policy-effectiveness indicator

- (Chapter I)Progress of pressure indicator: persistence of pressure

4 Driving forces assessment indicators

- (Chapter III) Socio-economic development: Processes of landscape change

Figure 4.
Environmental indicators used in this study within the DPSIR framework. Figure on the left shows a simplified diagram of the DPSIR framework. In blue are the DPSIR categories, or feedbacks, for which environmental indicators have been applied. Boxes on the right specify the indicators per DPSIR category and per chapter.

Understanding the relationship of land cover with water quality is the first step towards the definition of land cover scenarios capable of providing insights into the implications of current management trajectories and, more importantly, capable of highlighting provocative alternative options for the future (Peterson et al. 2002). If based on topographic and geomorphological characteristics of the watershed, as well as on management policies - either implemented, under development or proposed for implementation -, land change scenarios can provide a key input for the assessment of potential impacts on coastal systems (Bossa et al. 2014, Guse et al. 2015).

The Mondego system in the European context

The tributary rivers of the Mondego estuary have been contributing as freshwater inputs to the concentrations of nitrate and phosphate in the estuary (Chapter I), though the results indicate that the nutrient dependency on riverine inputs has decreased or has become weak, from a former (2003-2007) to a more recent period (2012-2013). Both nitrate and phosphate are indicators of water chemical quality, as over-enrichment by nutrients may trigger several ecological changes. The primary symptoms are excessive algae growth and oxygen depletion (Dolbeth et al. 2007). The eutrophic and hypoxic events are associated with urban, industrial and agricultural areas, and more precisely to fertilizers, manure, human waste and biomass burning. Over the past two centuries, human activities have altered the cycles of several key elements, including the cycles of nitrogen and phosphorous. When compared to natural values, human activities have caused the near doubling of nitrogen and tripling of phosphorous flows to the environment (Millennium Ecosystem Assessment 2005). The results have also revealed that in the Mondego estuary phosphate has increased and ammonia has decreased, while nitrate showed no significant differences between a former time interval (2003-2007) and a recent time interval (2012-2013). In the tributary rivers, results have shown that, in 2012-2013, the nitrate annual mean concentrations are lower than the European 2012 average mean concentrations; the ammonia annual mean concentrations are lower than the European concentrations found in 2000 and that the phosphate annual mean concentrations are higher than the 1992 European concentrations. In the Mondego river basin, phosphate has emerged as the parameter that needs closer attention in the future regarding policy measures to encourage sustainable agricultural practices (Chapter I).

Concentrations higher than 0.1-0.2mg P/l were recorded, which are generally perceived to be sufficiently high to result in freshwater eutrophication (EEA 2015a). Frequently, eutrophication results from leaching of nutrients from soils managed for agriculture (Munn et al. 2010) due to the oversupply of nutrients to soil to support food production that has significantly altered the rate at which phosphorous accumulates in soils (Schoumans et al. 2015, Stutter et al. 2015), which may then be eroded into freshwater systems. Chapter I calls for the necessity to manage, in order to diminish, the inputs of phosphate into the estuary, in an European legislation context that is still insufficient with regard to pollution from phosphorous (P) compounds that affect transitional waters and freshwater (Schoumans et al. 2015). This contrasts with European legislation for the reduction of nitrates, for which a specific Directive (EC 1991b) has been in action since 1991. There has been however, a recent effort to propose and implement a coherent package of measures to manage P (Withers et al. 2015). The proposed strategy suggests acting in order to realign P inputs, reduce P losses to waters, recycle P in bio-resources, recover P from waste, and finally if necessary redefine our food system (Withers et al. 2015).

The transport of nutrients from land to water is a function of land cover and land cover patterns (Aguilera et al. 2012). In the Mondego river basin 32% of land cover is occupied by agricultural areas, rising to 52% if we only consider the sub-basins that directly drain to the estuary. Urban and industrial areas only occupy 3% of the basin, but have increased at high rates between 1990 and 2006. Despite the known relationship between land cover and nutrients concentrations, weak relationship of land cover percentage with nitrate was found and no relationship of landscape patterns with the concentrations of nitrate was detected (Chapter II). Results also indicated weak positive relation with urban areas and strong negative relation with steeper slopes and higher variation of elevation - which indicate low occupation of agricultural land and high occupation of forest. This is in accordance with what was expected, because the major source of nitrate to aquatic systems is agriculture (EEA 2015a). Though the factors suggested in Chapter II should explain, at some extent, the low variability described by our model, we believe that the implementation of nitrate-reduction measures could also explain the weak relationship between landscape and nitrate. The recent European reports on freshwater indicators assessment has revealed that, between 1992 and 2012, nitrate concentrations have declined steadily in European rivers as a result of the implementation of European Directives focused on the reduction of nitrate concentrations from agricultural land (EEA 2015a). It is plausible to assume that the decline of nitrate concentrations due to the implementation of European policies, namely the Nitrates Directive (EC 1991b), has changed the relationship between nitrate and its former major land sources, hindering the capability of the statistical model to detect strong relationships that were absent already, or at least weaker. For more than 20 years, the European Union (EU) has been implementing a comprehensive framework of EU legislation to protect the environment, within which the Nitrates Directive is a key element (EC 1991b), with successful outcomes indicated by the decline in the average concentrations of nitrate in European rivers (EEA 2015a). The Nitrates Directive, which established a code of Good Agricultural Practice (EC 2010), is backed up by the common agricultural policy (CAP) through direct support and rural development measures. The code established the general principles of rational fertilization of soil and crops, with emphasis on nitrogen fertilization, but it is voluntary out of vulnerable zones (EC 2010). In Portugal the use of fertilizers, including manure, has been reduced and crops are removing nitrate in a more effective manner, reducing the overall pressure over the aquatic systems (GPP 2014). At the same time, the number of farms and the utilized agricultural area decreased between 1999 and 2009 (GPP 2014), which might have also contributed for the reduction of fertilizers in use.

The crop abandonment trend was captured in the Mondego river basin through the analysis of transitions among land cover categories which detected systematic transitions from rainfed to heterogeneous agricultural areas (Chapter III). The socio-economic drivers behind crop abandonment were not assessed in this study, but the implementation, in the 1990s, of CAP policies encouraging the decrease of plant production and the compulsory set-aside of land (EC 1992, EC 2003) might have been at the core of the agricultural transitions observed during the 1990-2006 time-interval. This was a time of rapid economic growth in Portugal sustained by large transfers of EU structural funds (Becker 2012) and by private capital inflows boosted by a decrease in interest rates (Detragiache and Hamann 1997, Afonso 2007). These factors tend also to favor urban and industrial growth (Amaral 2011, Hott and Jokipii 2012) and could explain the intense urbanization and industrial expansion processes observed in Mondego river basin (Chapter III). Similar trends have been observed throughout Europe (EEA 2013). The processes of change to artificial areas indicate that soil permeability has decreased and that land fragmentation has increased, with consequences for hydrologic processes and for the transport of nutrients into water, as observed in other geographic areas (Aguilera et al. 2012). If this tendency is maintained, the policy measures needed to reduce the concentrations of phosphate in European waters will have to take into account that the tendency of phosphate to be eroded into freshwater may increase in the future, since large amounts of phosphorous have accumulated on land and their transport to water systems is slow and difficult to prevent (Joel et al. 2012).

When including the errors of the maps into the analysis the results show evidences that the gain of urban and industrial land are targeting heterogeneous agricultural areas more than any other category and that the errors in the maps cannot account for all deviations from uniform intensity (Chapter IV), meaning that some land cover categories and transitions are changing at rates higher than our uniform hypothesis postulated. This is the case of urban and industrial areas. These are the categories that occupy the smallest area but that have changed the most (Chapter III and Chapter IV). Future research relating the socio-economic drivers that are influencing land changes in the Mondego river basin should be performed. The interpretation of the results indicates that the processes of landscape change are suitable proxies for socio-economic development, however land changes can hardly be dissociated from population growth, economic development and agricultural and environmental policies assessment, as the impact of these over-arching drivers on ecosystem integrity may arise faster than land changes occur. The linkage with the socio-economic drivers of land change has been successfully achieved for other geographic areas (Changhong et al. 2011, MacLeod and Moller 2006, Newman et al. 2014) and is vital for integrated sustainable management (Evers and Nyberg 2013).

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Appendix

Table A1.

Pearson correlation coefficients for landscape metrics of class Urban (1-20).

Significant coefficients higher than 0.8 are highlighted in bold

URBAN																			
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
2	0.73																			
3	**0.89**	0.76																		
4	**0.92**	0.56	0.7																	
5	0.73	**0.98**	0.74	0.52																
6	0.3	-0.04	-0.07	0.45	-0.04															
7	-0.22	-0.41	-0.48	-0.09	-0.4	0.71														
8	**0.89**	0.55	0.61	**0.97**	0.55	0.54	-0.03													
9	-0.15	-0.23	-0.32	-0.06	-0.26	0.51	0.79	-0.04												
10	**0.87**	0.55	0.61	**0.97**	0.53	0.49	-0.07	0.99	0.02											
11	-0.24	-0.24	-0.32	-0.17	-0.28	0.35	0.68	-0.17	0.96	-0.1										
12	0.78	0.5	0.53	**0.88**	0.49	0.46	-0.01	0.91	0.2	0.96	0.07									
13	-0.16	-0.1	-0.16	-0.12	-0.15	0.11	0.37	-0.11	0.49	-0.1	0.62	-0.01								
14	0.51	0.33	0.33	0.61	0.34	0.2	0.03	0.67	0.03	0.67	-0.07	0.71	0.24							
15	-0.59	-0.66	-0.68	-0.48	-0.69	0.25	0.67	-0.49	0.5	-0.51	0.56	-0.49	0.56	-0.36						
16	0.44	0.13	0.1	0.53	0.15	**0.9**	0.44	0.63	0.11	0.53	-0.04	0.42	-0.05	0.19	0.06					
17	0.44	0.14	0.1	0.53	0.17	**0.85**	0.39	0.63	0.02	0.52	-0.14	0.39	-0.1	0.21	0.02	**0.99**				
18	-0.08	-0.14	0.01	-0.13	-0.17	0.1	0.07	-0.15	-0.25	-0.24	-0.19	-0.41	0.03	-0.29	0.17	0.26	0.26			
19	0.76	0.41	0.42	**0.86**	0.43	0.76	0.21	**0.93**	0.13	**0.9**	-0.04	**0.85**	-0.07	0.56	-0.3	**0.82**	**0.81**	-0.14		
20	-0.11	-0.49	-0.42	0.14	-0.55	0.77	0.67	0.17	0.4	0.13	0.3	0.08	0.12	-0.08	0.57	0.67	0.63	0.28	0.38	

1-PLAND1, 2-NP1, 3-PD1, 4-LPI1, 5-LSI1, 6-AREA_MN1, 7-GYRATE_MN1, 8-GYRATE_AM1, 9-SHAPE_MN1, 10-SHAPE_AM1, 11-FRAC_MN1, 12-FRAC_AM1, 13-CIRCLE_MN1, 14-CIRCLE_AM1, 15-CONTIG_MN1, 16-CONTIG_AM1, 17-PLADJ1, 18-IJI1, 19-COHESION1, 20-AI1

Table A2.

Pearson correlation coefficients between landscape metrics of class Urban (1-20) and class Agriculture (21-40).

Significant coefficients higher than 0.8 are highlighted in bold

	URBAN																			
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
21	0.48	0.68	0.69	0.28	0.7	-0.31	-0.55	0.25	-0.27	0.29	-0.27	0.29	-0.42	0.1	**-0.83**	-0.21	-0.21	-0.17	0.08	-0.68
22	0.18	0.57	0.1	0.04	0.66	0.01	-0.18	0.2	-0.14	0.16	-0.15	0.19	0.08	0.18	-0.3	0.19	0.23	-0.22	0.26	-0.37
23	0.43	0.11	0.36	0.3	0.23	0.14	-0.05	0.34	-0.02	0.33	-0.05	0.36	0.07	0.21	-0.09	0.18	0.2	-0.39	0.38	-0.14
24	0.02	0.27	0.32	-0.13	0.27	-0.5	-0.49	-0.2	-0.25	-0.14	-0.2	-0.11	-0.42	-0.15	-0.54	-0.48	-0.48	-0.1	-0.33	-0.6
25	0.29	0.76	0.3	0.13	**0.82**	-0.17	-0.38	0.24	-0.28	0.22	-0.29	0.23	-0.13	0.25	-0.59	0.03	0.06	-0.2	0.19	-0.58
26	0.14	0.56	0.35	0.02	0.54	-0.4	-0.46	-0.01	-0.23	0.04	-0.21	0.05	-0.41	0	-0.68	-0.34	-0.34	-0.11	-0.16	-0.61
27	0.03	0.42	0.18	-0.02	0.38	-0.3	-0.29	-0.03	-0.07	0.02	-0.08	0.06	-0.34	0.06	-0.59	-0.3	-0.32	-0.03	-0.15	-0.46
28	0.08	0.56	0.34	-0.1	0.54	-0.46	-0.54	-0.11	-0.34	-0.09	-0.28	-0.09	-0.34	-0.12	-0.61	-0.34	-0.33	-0.02	-0.25	-0.64
29	0.13	0.45	0.15	0.13	0.42	-0.15	-0.17	0.13	0	0.19	-0.06	0.21	-0.33	0.22	-0.58	-0.17	-0.18	-0.11	0.03	-0.37
30	0.17	0.58	0.42	0	0.58	-0.45	-0.55	-0.02	-0.36	0.01	-0.33	0	-0.43	-0.04	-0.69	-0.33	-0.32	-0.05	-0.17	-0.67
31	-0.17	0.01	-0.41	0	-0.01	0.21	0.36	0.06	0.37	0.09	0.25	0.17	-0.02	0.28	-0.01	0.07	0.06	-0.18	0.12	0.15
32	0.3	0.63	0.53	0.13	0.62	-0.38	-0.55	0.09	-0.32	0.13	-0.3	0.11	-0.45	0	-0.76	-0.27	-0.27	-0.05	-0.07	-0.65
33	-0.44	-0.45	-0.73	-0.2	-0.47	0.29	0.56	-0.12	0.38	-0.13	0.3	-0.04	0.28	0.23	0.51	0.14	0.15	-0.14	0.04	0.5
34	0.06	0.33	0.37	-0.04	0.27	-0.44	-0.5	-0.1	-0.32	-0.07	-0.23	-0.1	-0.16	-0.02	-0.53	-0.35	-0.37	0.27	-0.28	-0.48
35	-0.53	-0.57	-0.79	-0.33	-0.58	0.36	0.62	-0.26	0.36	-0.3	0.31	-0.26	0.28	-0.11	0.73	0.25	0.26	0.09	-0.04	0.66
36	0.23	0.54	0.38	0.1	0.55	-0.19	-0.31	0.09	0.05	0.16	0.04	0.23	-0.38	-0.06	-0.63	-0.23	-0.26	-0.34	0.01	-0.53
37	0.23	0.55	0.38	0.09	0.55	-0.2	-0.31	0.09	0.05	0.16	0.04	0.23	-0.38	-0.06	-0.63	-0.23	-0.26	-0.34	0.01	-0.53
38	0.77	0.53	**0.83**	0.58	0.55	0.06	-0.31	0.54	-0.26	0.52	-0.27	0.45	-0.14	0.25	-0.45	0.19	0.21	-0.13	0.44	-0.27
39	0.32	0.59	0.5	0.16	0.6	-0.25	-0.43	0.14	-0.13	0.19	-0.14	0.22	-0.48	-0.05	-0.72	-0.22	-0.23	-0.23	0.02	-0.58
40	0.19	0.48	0.34	0.07	0.48	-0.18	-0.27	0.05	0.1	0.13	0.09	0.2	-0.38	-0.11	-0.57	-0.24	-0.28	-0.34	-0.02	-0.47

1-PLAND1, 2-NP1, 3-PD1, 4-LPI1, 5-LSI1, 6-AREA_MN1, 7-GYRATE_MN1, 8-GYRATE_AM1, 9-SHAPE_MN1, 10-SHAPE_AM1, 11-FRAC_MN1, 12-FRAC_AM1, 13-CIRCLE_MN1, 14-CIRCLE_AM1, 15-CONTIG_MN1, 16-CONTIG_AM1, 17-PLADJ1, 18-IJI1, 19-COHESION1, 20-AI1, 21-PLAND2, 22-NP2, 23-PD2, 24-LPI2, 25-LSI2, 26-AREA_MN2, 27-GYRATE_MN2, 28-GYRATE_AM2, 29-SHAPE_MN2, 30-SHAPE_AM2, 31-FRAC_MN2, 32-FRAC_AM2, 33-CIRCLE_MN2, 34-CIRCLE_AM2, 35-CONTIG_MN2, 36-CONTIG_AM2, 37-PLADJ2, 38-IJI2, 39-COHESION2, 40-AI2

Table A3.

Pearson correlation coefficients between landscape metrics of class Urban (1-20) and class Forest (41-60).
Significant coefficients higher than 0.8 are highlighted in bold

	URBAN																			
	1	**2**	**3**	**4**	**5**	**6**	**7**	**8**	**9**	**10**	**11**	**12**	**13**	**14**	**15**	**16**	**17**	**18**	**19**	**20**
41	-0.59	-0.73	-0.77	-0.38	-0.76	0.24	0.53	-0.35	0.26	-0.39	0.28	-0.37	0.41	-0.15	**0.83**	0.13	0.13	0.18	-0.18	0.66
42	0.51	**0.83**	0.67	0.28	**0.85**	-0.23	-0.51	0.27	-0.29	0.29	-0.29	0.26	-0.37	0.05	-0.78	-0.07	-0.07	-0.09	0.14	-0.65
43	0.51	0.45	0.78	0.3	0.45	-0.26	-0.49	0.21	-0.24	0.25	-0.21	0.21	-0.33	-0.03	-0.65	-0.19	-0.21	0.08	0.03	-0.51
44	-0.58	-0.67	-0.78	-0.39	-0.68	0.25	0.54	-0.34	0.28	-0.38	0.28	-0.36	0.43	-0.15	**0.84**	0.15	0.15	0.12	-0.16	0.62
45	0.41	**0.84**	0.53	0.2	**0.87**	-0.27	-0.5	0.24	-0.3	0.25	-0.3	0.24	-0.29	0.14	-0.74	-0.1	-0.08	-0.19	0.12	-0.7
46	-0.43	-0.49	-0.51	-0.27	-0.54	0.03	0.18	-0.26	-0.17	-0.33	-0.12	-0.42	0.34	-0.04	0.58	0.1	0.13	0.5	-0.22	0.47
47	-0.43	-0.51	-0.5	-0.26	-0.57	0.03	0.19	-0.27	-0.16	-0.33	-0.11	-0.42	0.34	-0.05	0.6	0.08	0.11	0.49	-0.23	0.49
48	-0.17	0.37	-0.2	-0.25	0.39	-0.11	-0.08	-0.14	-0.12	-0.18	-0.07	-0.19	0.26	-0.01	0.04	0.03	0.06	0.03	-0.1	-0.24
49	-0.4	-0.37	-0.42	-0.24	-0.39	-0.25	0	-0.21	-0.11	-0.21	-0.06	-0.16	0.47	0.39	0.29	-0.27	-0.23	-0.01	-0.25	0.03
50	0.2	0.7	0.2	0.04	0.77	-0.14	-0.26	0.14	-0.16	0.12	-0.17	0.14	-0.13	0.12	-0.45	0	0.03	-0.29	0.11	-0.55
51	-0.22	-0.12	-0.16	-0.11	-0.14	-0.36	-0.12	-0.09	0.09	-0.02	0.14	0.14	0.41	0.56	-0.07	-0.49	-0.49	-0.38	-0.2	-0.3
52	0.33	0.69	0.34	0.13	0.79	-0.16	-0.32	0.23	-0.2	0.22	-0.24	0.24	-0.22	0.18	-0.58	-0.01	0.02	-0.35	0.18	-0.64
53	0.11	0.18	0.14	0.13	0.19	-0.17	-0.08	0.17	0.33	0.26	0.35	0.46	0.42	0.58	-0.26	-0.35	-0.38	-0.59	0.08	-0.41
54	0.29	0.54	0.45	0.13	0.52	-0.08	-0.23	0.11	-0.05	0.11	0.03	0.05	0.12	-0.08	-0.33	0.01	-0.03	0.35	0	-0.33
55	0.02	0.27	0.08	0.09	0.23	-0.42	-0.43	0.12	-0.21	0.18	-0.19	0.28	0.1	0.59	-0.45	-0.4	-0.38	-0.38	-0.04	-0.46
56	-0.57	-0.64	**-0.81**	-0.35	-0.66	0.32	0.58	-0.28	0.32	-0.32	0.31	-0.28	0.38	-0.09	0.79	0.2	0.21	0.07	-0.08	0.67
57	-0.57	-0.63	**-0.81**	-0.35	-0.65	0.32	0.57	-0.28	0.32	-0.32	0.31	-0.28	0.37	-0.09	0.78	0.2	0.21	0.07	-0.08	0.66
58	0.33	0.04	0.19	0.21	0.16	0.3	0.24	0.24	0.06	0.18	0.01	0.13	0.16	0.1	0.21	0.34	0.37	-0.1	0.32	0.05
59	-0.41	-0.35	-0.71	-0.25	-0.32	0.34	0.53	-0.14	0.27	-0.19	0.21	-0.16	0.25	-0.01	0.61	0.29	0.31	-0.12	0.06	0.48
60	-0.57	-0.65	**-0.81**	-0.34	-0.68	0.32	0.58	-0.29	0.34	-0.32	0.32	-0.28	0.38	-0.1	**0.8**	0.2	0.2	0.07	-0.08	0.67

1-PLAND1, 2-NP1, 3-PD1, 4-LPI1, 5-LSI1, 6-AREA_MN1, 7-GYRATE_MN1, 8-GYRATE_AM1, 9-SHAPE_MN1, 10-SHAPE_AM1, 11-FRAC_MN1, 12-FRAC_AM1, 13-CIRCLE_MN1, 14-CIRCLE_AM1, 15-CONTIG_MN1, 16-CONTIG_AM1, 17-PLADJ1, 18-IJI1, 19-COHESION1, 20-AI1, 40-AI2, 41-PLAND3, 42-NP3, 43-PD3, 44-LPI3, 45-LSI3, 46-AREA_MN3, 47-GYRATE_MN3, 48-GYRATE_AM3, 49-SHAPE_MN3, 50-SHAPE_AM3, 51-FRAC_MN3, 52-FRAC_AM3, 53-CIRCLE_MN3, 54-CIRCLE_AM3, 55-CONTIG_MN3, 56-CONTIG_AM3,57-PLADJ3, 58-IJI3, 59-COHESION3, 60-AI3

Table A4.

Pearson correlation coefficients for landscape metrics of class Agriculture (21-40).

Significant coefficients higher than 0.8 are highlighted in bold

AGRICULTURE																				
	21	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**	**31**	**32**	**33**	**34**	**35**	**36**	**37**	**38**	**39**	**40**
21																				
22	0.22																			
23	0.01	0.25																		
24	**0.8**	-0.11	-0.27																	
25	0.56	**0.88**	-0.02	0.26																
26	**0.87**	0.19	-0.43	**0.85**	0.6															
27	0.72	0.13	-0.61	0.71	0.54	**0.94**														
28	**0.83**	0.26	-0.39	**0.87**	0.62	**0.95**	**0.82**													
29	0.62	0.19	-0.57	0.54	0.58	**0.83**	**0.95**	0.67												
30	**0.88**	0.22	-0.34	**0.9**	0.61	**0.96**	**0.83**	**0.98**	0.72											
31	-0.12	0.2	-0.52	-0.11	0.3	0.21	0.48	0.02	0.68	0.05										
32	**0.94**	0.19	-0.28	**0.88**	0.59	**0.96**	**0.85**	**0.94**	0.74	**0.98**	0.04									
33	-0.75	0.03	-0.23	-0.58	-0.16	-0.47	-0.23	-0.55	-0.04	-0.56	0.68	-0.64								
34	0.71	-0.11	-0.5	0.79	0.28	**0.8**	0.75	**0.82**	0.56	**0.8**	-0.1	**0.81**	-0.58							
35	**-0.91**	-0.03	-0.17	-0.73	-0.33	-0.67	-0.47	-0.67	-0.33	-0.72	0.43	**-0.8**	**0.89**	-0.7						
36	**0.88**	0.24	-0.11	0.78	0.53	**0.88**	0.77	**0.82**	0.68	**0.83**	0.11	**0.87**	-0.54	0.58	-0.68					
37	**0.88**	0.25	-0.11	0.78	0.54	**0.88**	0.78	**0.82**	0.68	**0.83**	0.11	**0.87**	-0.53	0.58	-0.68	**1**				
38	0.46	0.06	0.67	0.17	0.07	0.05	-0.22	0.1	-0.27	0.16	-0.67	0.23	-0.67	0.08	-0.63	0.21	0.21			
39	**0.94**	0.19	-0.13	**0.85**	0.53	**0.91**	0.78	**0.87**	0.68	**0.91**	0.03	**0.95**	-0.64	0.67	-0.78	**0.97**	**0.97**	0.3		
40	**0.85**	0.15	-0.15	0.79	0.44	**0.86**	0.76	**0.8**	0.65	**0.81**	0.09	**0.84**	-0.53	0.58	-0.66	**0.99**	**0.99**	0.19	**0.96**	

21-PLAND2, 22-NP2, 23-PD2, 24-LPI2, 25-LSI2, 26-AREA_MN2, 27-GYRATE_MN2, 28-GYRATE_AM2, 29-SHAPE_MN2, 30-SHAPE_AM2, 31-FRAC_MN2, 32-FRAC_AM2, 33-CIRCLE_MN2, 34-CIRCLE_AM2, 35-CONTIG_MN2, 36-CONTIG_AM2, 37-PLADJ2, 38-IJI2, 39-COHESION2, 40-AI2

Table A5.

Pearson correlation coefficients between landscape metrics of class Agriculture (21-40) and Forest (41-60).

Significant coefficients higher than 0.8 are highlighted in bold

	AGRICULTURE																			
	21	**22**	**23**	**24**	**25**	**26**	**27**	**28**	**29**	**30**	**31**	**32**	**33**	**34**	**35**	**36**	**37**	**38**	**39**	**40**
41	**-0.99**	-0.24	-0.12	-0.73	-0.56	**-0.8**	-0.64	-0.76	-0.56	**-0.81**	0.15	**-0.9**	0.77	-0.62	**0.92**	**-0.85**	**-0.85**	-0.55	**-0.91**	**-0.81**
42	**0.94**	0.44	-0.03	0.68	0.74	**0.85**	0.69	**0.85**	0.61	**0.87**	-0.05	**0.92**	-0.69	0.63	**-0.8**	**0.85**	**0.85**	0.43	**0.91**	**0.8**
43	**0.86**	-0.15	0.13	0.7	0.15	0.59	0.44	0.57	0.3	0.63	-0.43	0.74	**-0.89**	0.67	**-0.94**	0.64	0.63	0.59	0.73	0.63
44	**-0.98**	-0.12	-0.07	-0.77	-0.46	-0.79	-0.64	-0.75	-0.56	**-0.82**	0.18	**-0.9**	0.78	-0.69	**0.94**	**-0.82**	**-0.82**	-0.55	**-0.9**	-0.79
45	**0.85**	0.63	-0.04	0.57	**0.9**	**0.82**	0.7	**0.83**	0.67	**0.84**	0.12	**0.85**	-0.5	0.53	-0.66	0.79	0.79	0.27	**0.82**	0.72
46	**-0.8**	-0.18	-0.33	-0.59	-0.35	-0.59	-0.44	-0.51	-0.39	-0.56	0.08	-0.64	0.57	-0.28	0.68	**-0.87**	**-0.87**	-0.51	**-0.83**	**-0.86**
47	**-0.8**	-0.26	-0.35	-0.56	-0.42	-0.58	-0.44	-0.5	-0.41	-0.55	0.05	-0.64	0.55	-0.25	0.67	**-0.86**	**-0.86**	-0.47	**-0.82**	**-0.84**
48	-0.03	0.79	-0.22	-0.18	0.73	0.18	0.2	0.26	0.24	0.16	0.35	0.08	0.21	-0.02	0.21	0.04	0.05	-0.36	-0.02	-0.02
49	-0.54	-0.02	-0.16	-0.36	-0.11	-0.36	-0.22	-0.36	-0.15	-0.37	0.23	-0.45	0.62	-0.14	0.44	-0.62	-0.62	-0.46	-0.63	-0.64
50	0.51	**0.85**	0.01	0.22	**0.95**	0.57	0.49	0.58	0.53	0.55	0.31	0.53	-0.15	0.15	-0.26	0.56	0.57	0.03	0.52	0.48
51	-0.04	0.01	-0.11	0.08	0.08	0.07	0.18	0	0.21	0.01	0.3	-0.02	0.36	0.18	0.01	-0.07	-0.07	-0.28	-0.11	-0.09
52	0.61	**0.82**	0.24	0.26	**0.91**	0.53	0.43	0.52	0.49	0.53	0.2	0.55	-0.27	0.12	-0.41	0.59	0.59	0.19	0.57	0.5
53	0.27	0.21	0.21	0.14	0.26	0.19	0.25	0.08	0.29	0.1	0.22	0.15	0.08	0.15	-0.27	0.28	0.28	0.01	0.19	0.25
54	0.54	0.3	-0.25	0.23	0.49	0.5	0.52	0.49	0.45	0.45	0.02	0.53	-0.52	0.55	-0.5	0.42	0.42	0.04	0.44	0.38
55	0.27	0.23	-0.22	0.28	0.43	0.4	0.45	0.34	0.48	0.37	0.34	0.33	0.17	0.4	-0.21	0.19	0.2	-0.14	0.2	0.15
56	**-0.96**	-0.09	-0.14	-0.73	-0.42	-0.73	-0.56	-0.7	-0.45	-0.76	0.3	**-0.85**	**0.86**	-0.67	**0.97**	-0.75	-0.75	-0.59	**-0.84**	-0.72
57	**-0.96**	-0.08	-0.14	-0.74	-0.41	-0.73	-0.56	-0.7	-0.45	-0.76	0.3	**-0.85**	**0.86**	-0.67	**0.97**	-0.75	-0.75	-0.59	**-0.84**	-0.71
58	-0.26	0.23	0.79	-0.53	-0.05	-0.58	-0.67	-0.55	-0.54	-0.51	-0.29	-0.47	-0.01	-0.7	0.13	-0.38	-0.38	0.41	-0.39	-0.43
59	-0.79	0.27	-0.01	-0.74	-0.05	-0.58	-0.45	-0.57	-0.28	-0.63	0.44	-0.71	**0.83**	-0.78	**0.91**	-0.56	-0.56	-0.51	-0.67	-0.56
60	**-0.96**	-0.11	-0.13	-0.73	-0.44	-0.74	-0.57	-0.71	-0.47	-0.77	0.29	**-0.86**	**0.85**	-0.67	**0.97**	-0.75	-0.75	-0.58	**-0.84**	-0.71

21-PLAND2, 22-NP2, 23-PD2, 24-LPI2, 25-LSI2, 26-AREA_MN2, 27-GYRATE_MN2, 28-GYRATE_AM2, 29-SHAPE_MN2, 30-SHAPE_AM2, 31-FRAC_MN2, 32-FRAC_AM2, 33-CIRCLE_MN2, 34-CIRCLE_AM2, 35-CONTIG_MN2, 36-CONTIG_AM2, 37-PLADJ2, 38-IJI2, 39-COHESION2, 40-AI2, 41-PLAND3, 42-NP3, 43-PD3, 44-LPI3, 45-LSI3, 46-AREA_MN3, 47-GYRATE_MN3, 48-GYRATE_AM3, 49-SHAPE_MN3, 50-SHAPE_AM3, 51-FRAC_MN3, 52-FRAC_AM3, 53-CIRCLE_MN3, 54-CIRCLE_AM3, 55-CONTIG_MN3, 56-CONTIG_AM3,57-PLADJ3, 58-IJI3, 59-COHESION3, 60-AI3

Table A6.
Pearson correlation coefficients between landscape metrics of class Agriculture (21-40) and Forest (41-60).
Significant coefficients higher than 0.8 are highlighted in bold

	NATURAL AREAS																			
	41	**42**	**43**	**44**	**45**	**46**	**47**	**48**	**49**	**50**	**51**	**52**	**53**	**54**	**55**	**56**	**57**	**58**	**59**	**60**
41																				
42	**-0.94**																			
43	**-0.87**	0.72																		
44	**0.98**	**-0.9**	**-0.91**																	
45	**-0.85**	**0.95**	0.52	-0.77																
46	**0.82**	-0.72	-0.67	0.78	-0.62															
47	**0.82**	-0.73	-0.66	0.78	-0.65	**0.99**														
48	0.05	0.24	-0.38	0.17	0.47	0.21	0.15													
49	0.58	-0.58	-0.56	0.55	-0.39	0.72	0.7	0.21												
50	-0.52	0.71	0.08	-0.38	**0.87**	-0.41	-0.47	0.77	-0.23											
51	0.1	-0.19	-0.15	0.06	-0.05	0.15	0.14	0.04	0.77	-0.06										
52	-0.64	0.74	0.26	-0.52	**0.88**	-0.55	-0.63	0.59	-0.29	**0.94**	-0.05									
53	-0.26	0.13	0.15	-0.25	0.22	-0.34	-0.36	0.02	0.35	0.18	**0.81**	0.28								
54	-0.54	0.63	0.52	-0.51	0.6	-0.22	-0.28	0.41	-0.29	0.46	-0.14	0.44	0.12							
55	-0.21	0.19	0.01	-0.23	0.33	0	0	0.2	0.59	0.25	**0.86**	0.22	0.69	0.01						
56	**0.97**	**-0.87**	**-0.95**	**0.98**	-0.74	0.74	0.74	0.16	0.53	-0.36	0.09	-0.51	-0.23	-0.55	-0.16					
57	**0.97**	**-0.87**	**-0.95**	**0.98**	-0.73	0.74	0.74	0.17	0.53	-0.35	0.09	-0.5	-0.23	-0.55	-0.16	**1**				
58	0.13	-0.18	-0.11	0.21	-0.15	-0.02	-0.07	0.03	-0.11	0.06	-0.33	0.21	-0.1	-0.12	-0.49	0.1	0.1			
59	0.77	-0.61	**-0.96**	**0.86**	-0.41	0.54	0.51	0.45	0.38	0.07	-0.03	-0.08	-0.21	-0.44	-0.16	**0.88**	**0.88**	0.3		
60	**0.97**	**-0.88**	**-0.94**	**0.98**	-0.76	0.74	0.74	0.13	0.53	-0.38	0.09	-0.53	-0.23	-0.57	-0.17	**1**	**1**	0.1	**0.87**	

41-PLAND3, 42-NP3, 43-PD3, 44-LPI3, 45-LSI3, 46-AREA_MN3, 47-GYRATE_MN3, 48-GYRATE_AM3, 49-SHAPE_MN3, 50-SHAPE_AM3, 51-FRAC_MN3, 52-FRAC_AM3, 53-CIRCLE_MN3, 54-CIRCLE_AM3, 55-CONTIG_MN3, 56-CONTIG_AM3,57-PLADJ3, 58-IJI3, 59-COHESION3, 60-AI3