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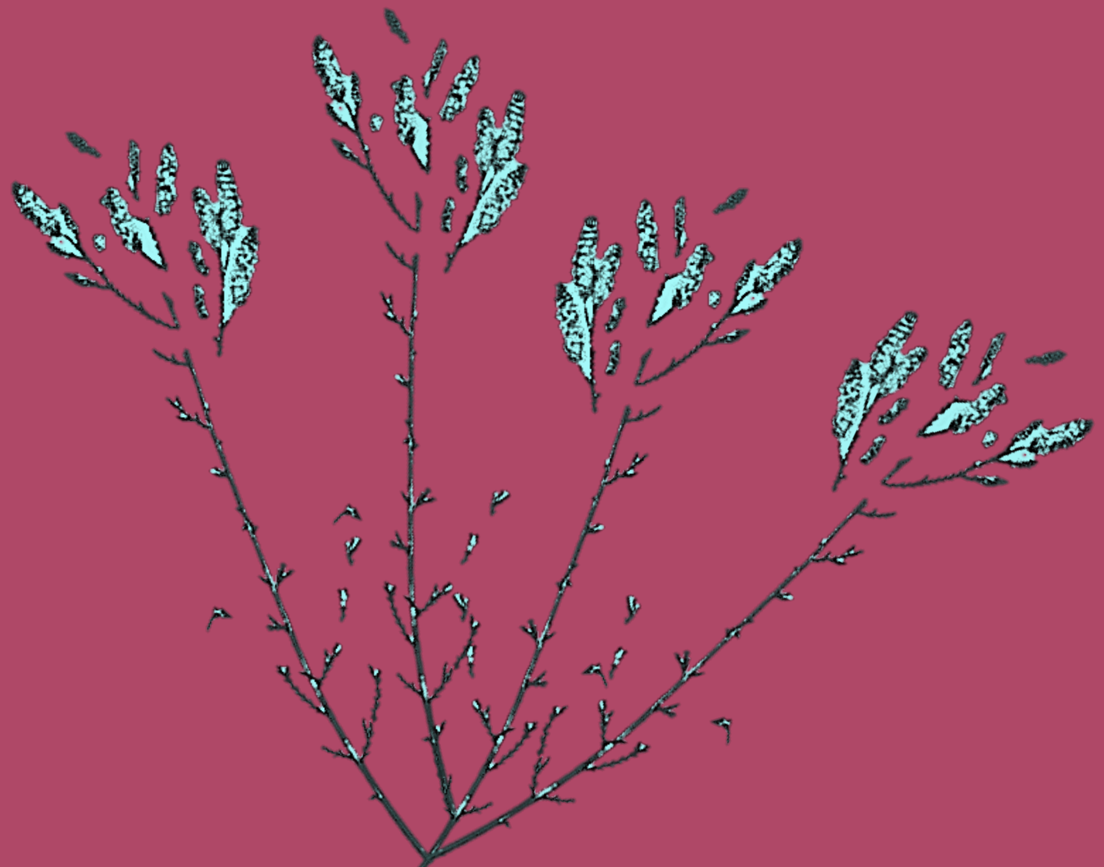
Riparian forest quality, land-use dynamics and their influence on macroinvertebrate communities. An evaluation of the ecological status of Pesquería River (N.E., Mexico)

Jaime Castro-López



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Daniel Castro-López

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Monterrey, Nuevo León, México 1983

Después de crecer en Monterrey, egresé de la carrera de Químico Farmacéutico Biólogo en la Universidad Autónoma de Nuevo León. Realicé mis estudios de máster en Ciencias con Orientación en Ingeniería Ambiental (U.A.N.L.), donde comencé a trabajar con el Dr. Víctor Hugo Guerra Cobián, de quien aprendí el gusto por la ciencia y los ecosistemas ribereños.

Después he recibido por parte del Consejo Nacional de Ciencia y Tecnología y del Instituto de Innovación y Transferencia de Tecnología (I^2T^2) del estado de Nuevo León, a una beca internacional para desarrollar mi tesis doctoral en el grupo de Investigación Freshwater Ecology Hidrology and Management de la Universidad de Barcelona. Aquí guiado por el Dr. Narcís Prat, de quien el aprendí el amor por la ecología y además reafirme mi pasión por los ríos.

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TESIS DOCTORAL

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Riparian forest quality, land-use dynamics and their influence on macroinvertebrate communities. An evaluation of the ecological status of Pesquería River (N.E., Mexico).

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Dr. Narcís Prat Fornells
Catedrático de Ecología
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Dr. Víctor Hugo Guerra Cobián
Profesor Titular
Universidad Autónoma de Nuevo León

La vida es nostra

A Jaime y Genoveva, Andrés, Eunice y Camila, Farina y Nina,

sobre todo, a Quentin

This river is wild

Run for the hills before they burn

Listen to the sound of the world

Don't watch it turn

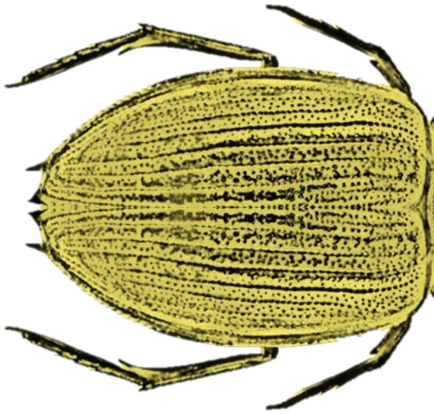
But shake a little

Sometimes I'm nervous when I talk, shake a little

Sometimes I hate the line I walk

I just want to show you what I know and catch you when the current lets you go

-This river is wild-
-The Killers-



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My house in Budapest,

My hidden treasure chest, golden grand piano

My beautiful Castillo, You

For You I'd leave it all

Barcelona, 23 de septiembre 2019

He tenido una beca del Consejo Nacional de Ciencia y Tecnología (CONACYT) en convenio con el Instituto de Innovación y Transferencia de Tecnología (I^2T^2). Parte de la investigación llevada a cabo ha sido financiada por el Centro Internacional del Agua y el Instituto de Ingeniería Civil de la Universidad Autónoma de Nuevo León y el grupo de Investigación Freshwater, Ecology, Hydrology and Management de la Universitat de Barcelona. Además, he obtenido la beca Santander de movilidad para jóvenes investigadores para la realización de la estancia internacional de investigación.

ADVISORS' REPORT

Dr. Narcís Prat Fornells and Dr. Víctor Hugo Guerra Cobián, advisers of the PhD thesis entitled “Riparian forest quality, land-use dynamics and their influence on macroinvertebrate communities. An evaluation of the ecological status of Pesquería River (N.E., Mexico)",

CERTIFY that Daniel Castro-López has carried out the dissertation presented here. The PhD candidate is the main author of the 3 chapters, and has acted as the principal researcher in all tasks regarding them (conceiving the research objectives, designing the studies, executing the field experiments, performing the lab analyses, analyzing the data, interpreting the results, writing the manuscripts, and reviewing and editing during the publication process).

INFORM that none of the information contained here will be used to elaborate other PhD thesis. Below, we detail the publication status of the chapters and indicate the impact factor (ISI Journal Citation Reports® Ranking) of the journals where the chapters have been published or submitted.

Chapter 1 (to be Submitted)

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Chapter 3 (Published)

The Influence of Riparian Corridor Land Use on the Pesquería River's Macroinvertebrate Community (N.E. Mexico)

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DOI: 10.3390/w11091930

Impact factor (2018): 2.66; (Q1) Water Science and Technology; (Q2) Aquatic Science.

For all of the above, we consider that the work of the PhD candidate grants him the right to defend his PhD thesis in front of a scientific committee.

Barcelona, 23 September 2019

Dr. Narcís Prat Fornells
Catedrático de Ecología
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Dr. Víctor Hugo Guerra Cobián
Profesor Titular
Universidad Autónoma de Nuevo León

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SUMMARY

Worldwide, biological diversity is declining at an unprecedented rate. The earth's ecosystems are currently being threatened due to both population growth and anthropogenic activity. The increasing exploitation and utilization of natural resources have led to excessive depletion of resources and increased environmental vulnerability. Current environmental changes are causing water, ecological and biodiversity crises that, coupled with climate change, are affecting landscape patterns and sustainable development efforts. Thus, anthropogenic pressures are changing the world's landscapes in pervasive ways. Additionally, these anthropogenic pressures on the natural landscape have been associated with changes in landscape and to the ecological condition of freshwater ecosystems. Thus, the riparian ecosystems should be considered the bloodstream of the planet since they determine the sustainability of living systems. Rivers provide environmental health, economic wealth and facilitate human wellbeing. Therefore, freshwater ecosystems, particularly rivers, must be examined in detail, and it is important to consider their relationship with terrestrial ecosystems, particularly the “transitional land-water” ecosystems (i.e. the riparian channel). This is particularly important for developing countries given to their ecosystems are constantly threatened by anthropogenic activities. Additionally, the lack of research, proper legislation and management hinders the restoration and conservation plans for their freshwater ecosystems.

Within this context, the overarching goal of this PhD thesis is to evaluate the ecological status of the Pesquería River and assess the effects of landscape degradation along its riparian channel. Since changes in landscape composition can significantly affect a riparian channel, we aimed to evaluate how said changes can also impact biological quality and subsequently affect the ecological integrity of a river ecosystem. We focused into provide simple and reliable methods to define the ecological status and discern the effects of

landscape degradation on the ecological state of the Pesquería River using a variety of tools for diagnosis and monitoring.

On this PhD thesis we explored land-use cover/change dynamics (LUCCD) in the riparian channel also we describe the land use processes in the riparian zone during the period 1976-2016 in order to ascertain whether land use dynamics can provide useful information about the current ecological status of a riparian zone **(Chapter 1)**. Additionally, we evaluated the riparian forest quality of the Pesquería River (Chapter 2) using an adaptation of the riparian forest quality index (QBR), modifying it for the specific local conditions (QBR-RNMX). Finally, we investigate how the macroinvertebrate communities in the Pesquería River are affected by different land use covers and assess their potential use as bioindicators to evaluate the ecological status of Mexican semi-arid rivers with similar conditions **(Chapter 3)**.

The findings of this PhD. thesis **(Chapter 1)** reflected how LUCCD analyses for the Pesquería River riparian zone described the main disturbances caused by the degradation of its riparian forest. The LUCCD assessment indicated a major increase in anthropogenic activity, especially in urban zones, grasslands and areas of secondary vegetation. The study did show some evidence of revegetation, but this was most likely due to the abandonment of agricultural practices caused by rural emigration and extreme flash flood events. This chapter also highlighted the most important changes in the riparian channel's land use dynamics in recent decades (1976-2016).

Our results for the evaluation of the riparian forest Quality using the QBR-RNMX **(Chapter 2)** showed five levels of riparian quality defined in the index, in the area surrounding the Pesquería we found poor or very poor conditions at 66% of the sampling sites, average-good conditions at 27% of the sites, and only one sampling site with excellent conditions. These results show that the riparian forest has been impacted significantly by urbanisation, agriculture and the presence of many invasive species.

On **Chapter 3** we demonstrate that the riparian channel is mainly influenced by agricultural and urban land use. Eighty-one invertebrate taxa were identified during the study. Permutational analysis of the variance analysis confirmed significant differences across the different land use classes and the macroinvertebrate community composition while no differences were found between seasons. The indicator species analysis revealed 31 representative taxa for natural land use, 1 for urban, and 4 for agricultural land use. Our modelling analysis showed that 28 of the 42 biological metrics tested responded significantly to land use disturbances. Our findings confirm the

influence of different land use changes on the Pesquería River's macroinvertebrate communities and suggesting that these metrics may have a use as bioindicators.

On this PhD thesis we demonstrated that LUCCD analysis at riparian channel level should be viewed as a key tool for the improvement of conservation policies targeting freshwater ecosystems in Mexico. We remark the importance of consider the riparian buffer width as an ecosystem, not as a federal area, in order to protect the freshwater ecosystems in Mexico. We corroborated the efficiency of the adaptation of the QBR-RNMX also we recommend the application of the index annually to evaluate the riparian forest's quality, and to assess its ecological status. This may be used for the establishment of restoration plans in high-impact zones and contingency plans to eliminate invasive species along the Pesquería River. This PhD thesis has provided eleven biological metrics and thirty-two representative taxa that can be used as bioindicators in future research. However, more studies, and especially the relationship of this index with the river pressures are necessary to establish an index for the Pesquería River in the future. Furthermore, in the general discussion we propose a macroinvertebrate-based index (BMWP-system) as a first step for the evaluation of the ecological status of the river.

Finally, this thesis highlights the lack of research and the bad current ecological status of the Pesquería River. Our findings suggest the creation of a Mexican framework similar to the European Water Framework Directive, where the ecological quality of the river should be the main objective of riparian ecosystems recovery.

RESUMEN

A nivel mundial, la diversidad biológica está disminuyendo a un ritmo sin precedentes. Los ecosistemas del planeta están actualmente amenazados debido al crecimiento acelerado de la población y a las actividades antropogénicas. La creciente explotación y utilización de los recursos naturales ha llevado a un agotamiento excesivo y a una mayor vulnerabilidad ambiental. Los cambios ambientales actuales están causando crisis hídricas, ecológicas y de biodiversidad que, junto con el cambio climático, están afectando los patrones del paisaje y los esfuerzos de desarrollo sostenible. Por lo tanto, las presiones antropogénicas están cambiando los paisajes del mundo de manera generalizada. Además, estas presiones antropogénicas en el paisaje natural se han asociado con cambios en el paisaje y en las condiciones ecológicas de los ecosistemas de agua dulce. Por lo que los ecosistemas ribereños deben considerarse como el torrente sanguíneo del planeta, ya que determinan la sostenibilidad de los sistemas vivos. Los ríos proporcionan salud ambiental, riqueza económica y facilitan el bienestar humano. Por lo tanto, los ecosistemas de agua dulce, particularmente los ríos, deben ser examinados en detalle, considerando su relación con los ecosistemas terrestres, particularmente los ecosistemas de "transición" (es decir, el canal ribereño). Esto es particularmente importante para los países en desarrollo, dado que sus ecosistemas están constantemente amenazados por actividades antropogénicas. Además, la falta de investigación, legislación y gestión adecuadas dificultan los planes de restauración y conservación de sus ecosistemas de agua dulce.

En este contexto, el objetivo general de esta tesis doctoral es evaluar el estado ecológico del río Pesquería y evaluar los efectos de la degradación del paisaje a lo largo de su canal ribereño. Dado que los cambios en la composición del paisaje pueden afectar significativamente un canal ribereño, nuestro objetivo fue evaluar cómo dichos cambios también pueden afectar la calidad biológica

y, posteriormente, afectar la integridad ecológica de un ecosistema fluvial. Nos enfocamos en proporcionar métodos simples y confiables para definir el estado ecológico y discernir los efectos de la degradación del paisaje en el estado ecológico del río Pesquería utilizando una variedad de herramientas para el diagnóstico y monitoreo.

En esta tesis doctoral exploramos las dinámicas de cobertura / cambio del uso de la tierra (LUCCD) en el canal ribereño, también describimos los procesos de uso de la tierra en la zona ribereña durante el período 1976-2016 para determinar si la dinámica del uso de la tierra puede proporcionar información útil sobre El estado ecológico actual de una zona ribereña (Capítulo 1). Además, evaluamos la calidad del bosque ribereño del río Pesquería (Capítulo 2) utilizando una adaptación del índice de calidad del bosque ribereño (QBR), modificándolo para las condiciones locales específicas (QBR-RNMX). Finalmente, investigamos cómo las comunidades de macroinvertebrados en el río Pesquería se ven afectadas por diferentes coberturas de uso de la tierra y evalúe su uso potencial como bioindicadores para evaluar el estado ecológico de los ríos semiáridos mexicanos con condiciones similares (Capítulo 3).

Los hallazgos de esta tesis doctoral (Capítulo 1) reflejaron cómo los análisis de LUCCD para la zona ribereña del río Pesquería describieron las principales perturbaciones causadas por la degradación de su bosque ribereño. La evaluación de LUCCD indicó un aumento importante en la actividad antropogénica, especialmente en zonas urbanas, pastizales y áreas de vegetación secundaria. El estudio mostró cierta evidencia de revegetación, pero esto probablemente se debió al abandono de las prácticas agrícolas causadas por la emigración rural y los eventos extremos de inundaciones repentinas. Este capítulo también destacó los cambios más importantes en la dinámica del uso de la tierra del canal ribereño en las últimas décadas (1976-2016).

Nuestros resultados para la evaluación de la calidad del bosque ribereño utilizando el QBR-RNMX (Capítulo 2) mostraron cinco niveles de calidad ribereña definidos en el índice, en el área alrededor de la Pesquería encontramos condiciones pobres o muy malas en el 66% de los sitios de muestreo, buenas condiciones promedio en el 27% de los sitios, y solo un sitio de muestreo con excelentes condiciones. Estos resultados muestran que el bosque ribereño se ha visto afectado significativamente por la urbanización, la agricultura y la presencia de muchas especies invasoras.

En el Capítulo 3 demostramos que el canal ribereño está influenciado principalmente por el uso de la tierra agrícola y urbana. Ochenta y un taxones de invertebrados fueron identificados durante el estudio. El análisis de permutaciones del análisis de varianza confirmó diferencias significativas entre las diferentes clases de uso de la tierra y la composición de la comunidad de macroinvertebrados, mientras que no se encontraron diferencias entre las estaciones. El análisis de especies indicadoras reveló 31 taxones representativos para el uso natural de la tierra, 1 para uso urbano y 4 para uso agrícola. Nuestro análisis de modelos mostró que 28 de las 42 métricas biológicas evaluadas respondieron significativamente a las perturbaciones del uso de la tierra. Nuestros hallazgos confirman la influencia de diferentes cambios en el uso de la tierra en las comunidades de macroinvertebrados del río Pesquería y sugieren que estas métricas pueden tener un uso como bioindicadores.

En esta tesis doctoral demostramos que el análisis de LUCCD a nivel de los canales ribereños debe verse como una herramienta clave para la mejora de las políticas de conservación dirigidas a los ecosistemas de agua dulce en México. Observamos la importancia de considerar el ancho del amortiguador ribereño como un ecosistema, no como un área federal, para proteger los ecosistemas de agua dulce en México. Corroboramos la eficiencia de la adaptación del QBR-RNMX y también recomendamos la aplicación del índice anualmente para evaluar la calidad del bosque ribereño y evaluar su estado ecológico. Esto puede usarse para el establecimiento de planes de restauración en zonas de alto impacto y planes de contingencia para eliminar especies invasoras a lo largo del río Pesquería. Esta tesis doctoral ha proporcionado once métricas biológicas y treinta y dos taxones representativos que pueden usarse como bioindicadores en futuras investigaciones. Sin embargo, se necesitan más estudios, y especialmente la relación de este índice con las presiones del río para establecer un índice para el río Pesquería en el futuro. Además, en la discusión general, proponemos un índice basado en macroinvertebrados (sistema BMWP) como un primer paso para la evaluación del estado ecológico del río.

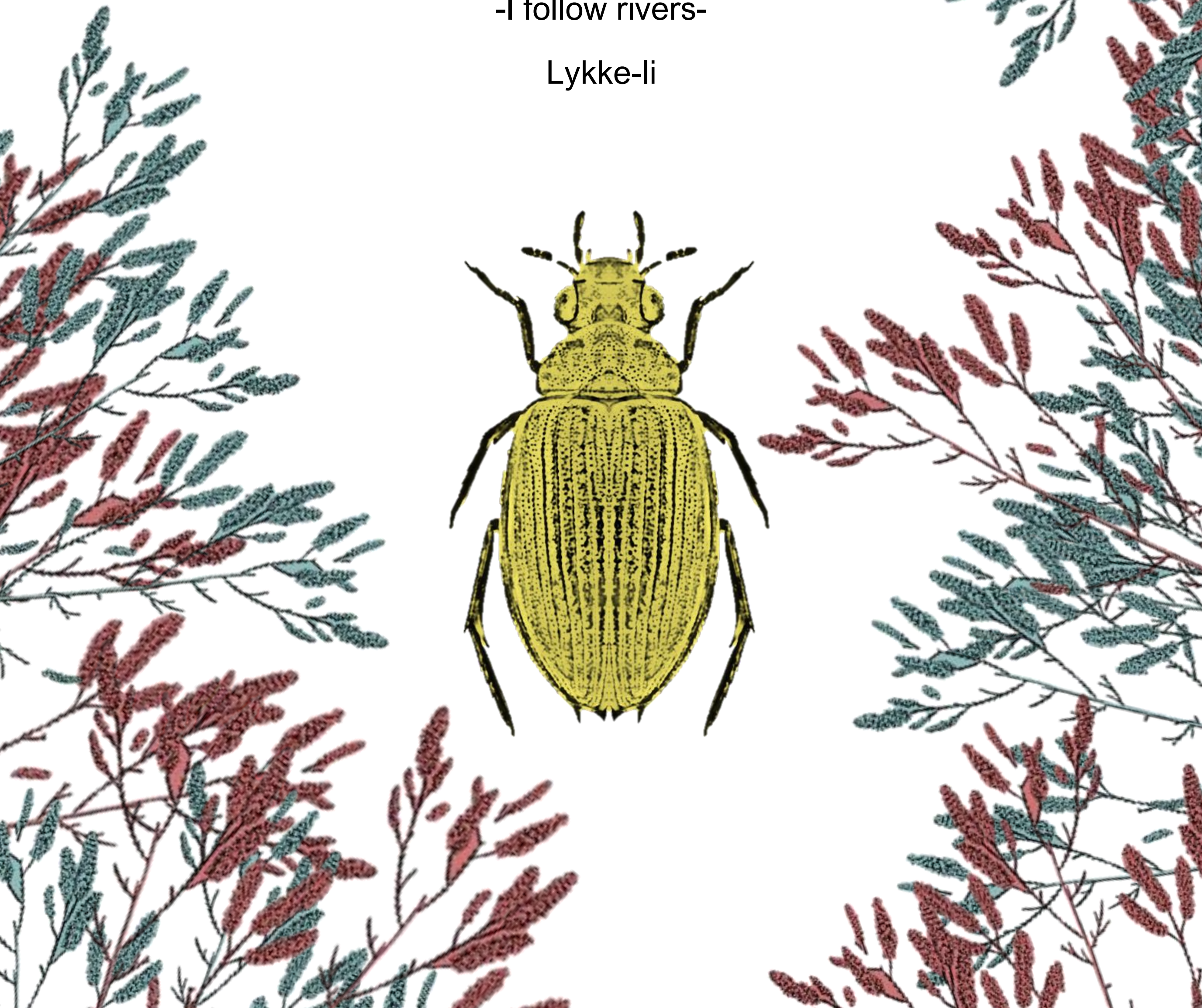
Finalmente, esta tesis doctoral destaca la falta de investigación y el mal estado ecológico actual del río Pesquería. Nuestros hallazgos sugieren la creación de un marco mexicano similar a la Directiva Marco del Agua Europea, donde la calidad ecológica del río debería ser el objetivo principal de la recuperación de los ecosistemas ribereños.

You're my river running high

Run deep, run wild

-I follow rivers-

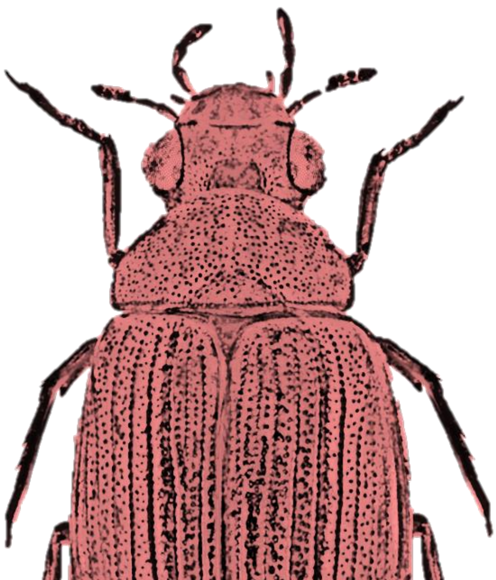
Lykke-li



General Introduction

Freshwater ecosystems and land use

Worldwide, biological diversity is declining at an unprecedented rate (Myers, 2000; Velázquez et al., 2003). The earth's ecosystems are currently being threatened due to both population growth and anthropogenic activity (Burdon et al., 2019; IPBES, 2019; Price et al., 2019). Rapid and intense land use changes have put considerable pressure on the natural environment (Figure 1) Li et al., 2010; Ma et al., 2019). The increasing exploitation and utilization of natural resources has led to an excessive depletion of resources and increased environmental vulnerability (Naveh, 1994). What is more, over a third of the earth's land is now used for agriculture, a practice which also consumes more than 75% of the planet's fresh water (IPBES, 2019). Further, many urban areas have doubled in size in recent years, and it is projected that by 2050 there will be more than 25 million kilometers of paved roads (IPBES, 2019).



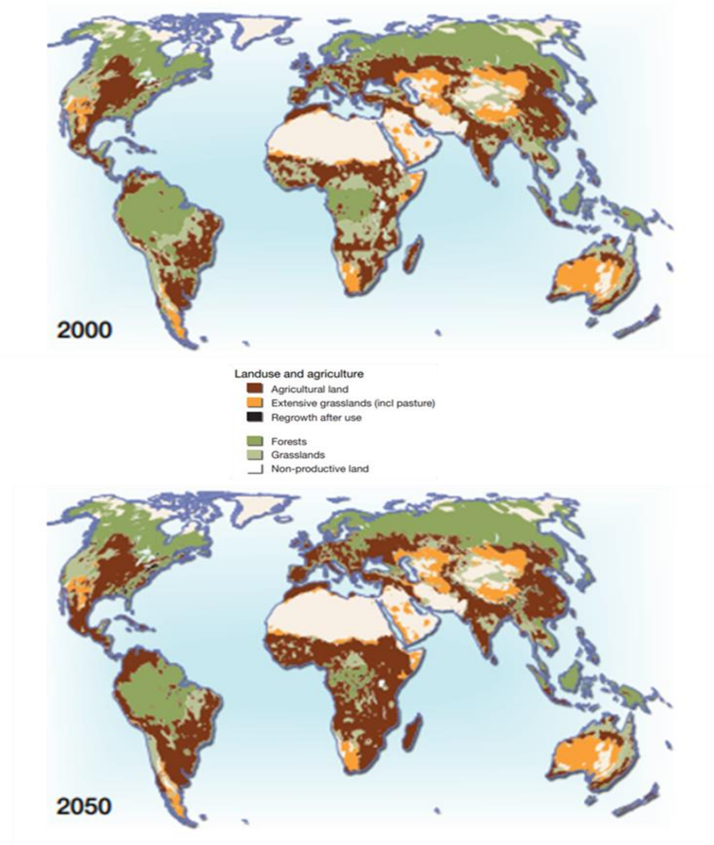


Figure 1 World land use projection obtained from (“Projected land use changes, World | ReliefWeb,” 2008)

Current global environmental changes are causing water, ecological and biodiversity crises that, coupled with climate change, are affecting landscape patterns and sustainable development efforts (Hodson & Marvin, 2009; Li et al., 2010). Consequently, in recent decades, the ecological crisis and water security (the availability of freshwater) have become key topics in global environmental research (Figure 2; taken form Steffen et al., 2015; and Vörösmarty et al., 2010; Wu, 2014). Habitat loss is considered to be one of the most important aspects of the global biodiversity crisis (Sala et al., 2000). Both habitat loss and alterations to land cover/change dynamics are a consequence of anthropogenic activities and climatic change (Falcucci et al., 2007; Vitousek et al., 1997).

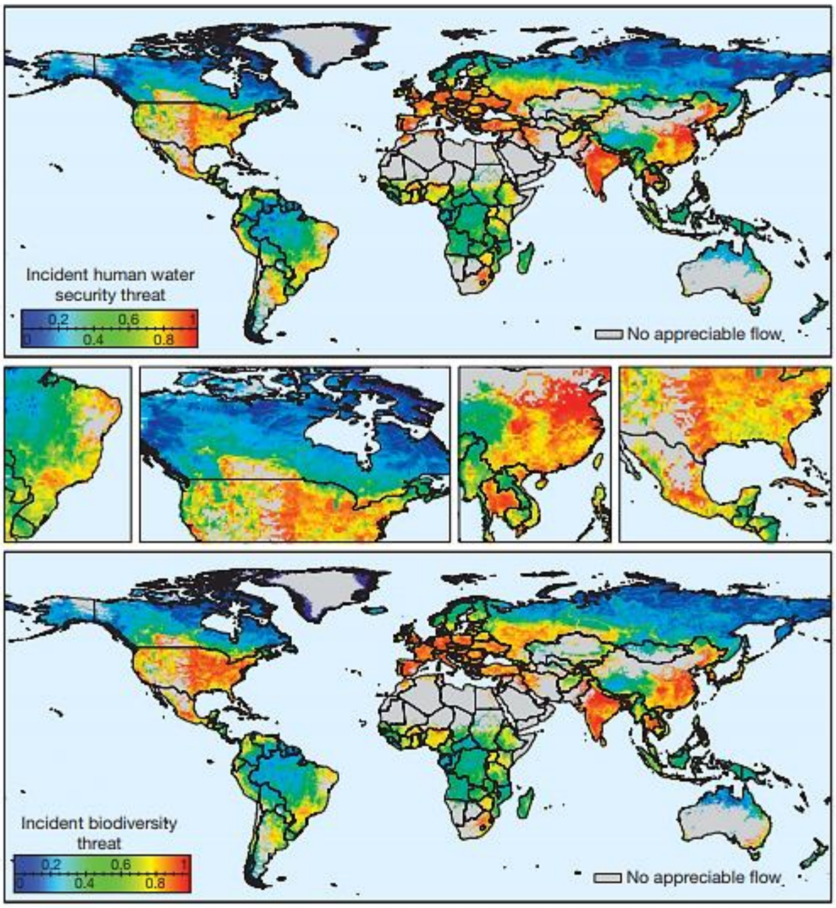


Figure 2 Global geography of incident threat to human water security and biodiversity obtained from (Vörösmarty et al., 2010). Where the term 'incident' refers to exposure to a diverse array of stressors at a given location.

Anthropogenic activities are therefore changing the world's landscapes in pervasive ways. Over time, anthropogenic activity can cause a given region to pass through several of the transitional states between different land use cover/change categories. Different regions of the world are in different transition states, which vary according to each region's history, social and economic conditions, and ecological contexts. Furthermore, not all regions of the world move linearly through these transitional states. Some places remain in one state for a long period of time, while others move quickly between the transition states (DeFries, Foley, & Asner, 2004). Thus, societies appear to follow a sequence of different land-use transitional states (Figure 3) from pre-settlement natural vegetation to frontier clearing, then to subsistence agriculture and small-scale farms, and finally to intensive agriculture, urban areas, and protected/recreational lands (Mustard, et al., 2004).

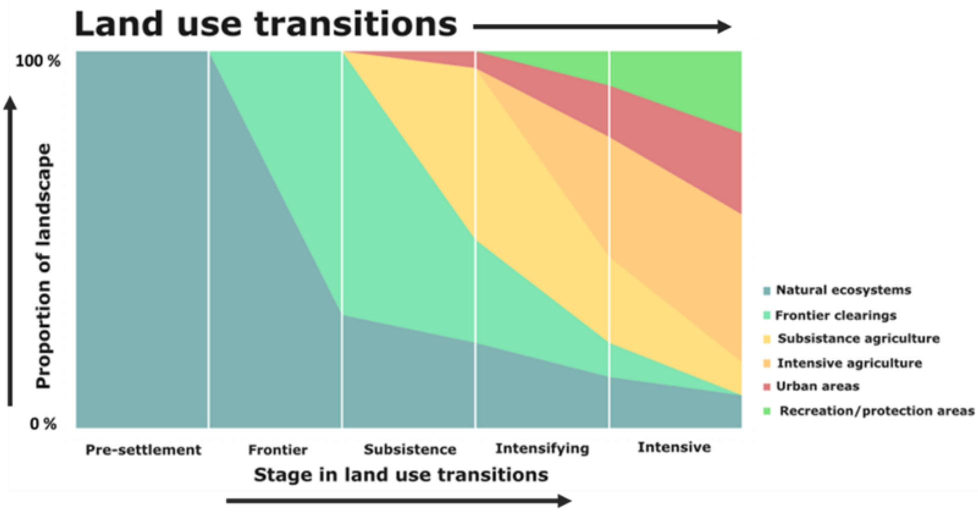


Figure 3 Schematic representation of land use transitions. Different parts of the world are at different transition stages which are usually representative of economic and ecological conditions; adapted from (DeFries et al. , 2004; Foley et al. , 2005; Mustard et al. , 2004)

Agricultural landscapes are considered to be homogenous matrices that have produced extensive environmental damage (Reidsma et al., 2006), converting perennial natural habitats like grasslands into arable fields, and destroying edge habitats such as field boundaries and buffer zones along creeks. In addition, natural habitat loss affects agricultural production by hampering pollinators, especially bees (Kremen, Williams, & Thorp, 2002). Urbanization dynamics offer some of the most clearly observable anthropogenic impacts on the Earth (Dawson et al., 2009). Urbanization replaces natural soil and vegetation covers with impervious surfaces. Agricultural activities are replaced by commercial and industrial activities, and rural structures are replaced with complex urban structures (Pandey et al., 2014).

These anthropogenic pressures on the natural landscape have been associated with changes in landscape and to the ecological condition of freshwater ecosystems (Allan, 2004; Krynak & Yates, 2018). The deterioration of freshwater ecosystems has affected water quality and said systems' capacity to provide reliable resources to maintain the natural hydrological cycle and ecosystem dynamics (Assessment (MEA), 2005; Edegbene, et al., 2019).

Riparian ecosystems ought to be considered the bloodstream of the planet since they determine the sustainability of living systems (Ripl, 2003). Rivers provide environmental health, economic wealth and facilitate human wellbeing (Grill et al., 2019). Historically, rivers have contributed to human development, providing food, water, industry and generating energy (Ripl, 2003). Nonetheless, freshwater ecosystems are one of the most threatened environs worldwide, particularly in developing countries, where population growth and changes in anthropogenic land-use activities are increasing (Ruiz-Picos, et al., 2017). Therefore, freshwater ecosystems, particularly rivers, must be examined in detail, and it is important to consider their relationship with terrestrial ecosystems (Margalef, 1990), particularly the “transitional land-water” ecosystems.

The “transitional land-water” ecosystem: the riparian zone

The effects of anthropogenic activities can be detected at distinct spatial strata, from small riparian zones to across an entire watershed (Zhou, Wu, & Peng, 2012). Riparian ecosystems are considered an extraordinarily diverse mosaic of landforms, communities, and environments within a larger landscape (Naiman & Décamps, 1997). Riparian zones serve as a framework for understanding the organization, diversity, and dynamics of communities associated with aquatic ecosystems (Gregory et al., 1991; Naiman & Décamps, 1997; Dufour, Rodríguez-González, & Laslier, 2019). Thus, the riparian zone is defined as an interface between terrestrial and aquatic ecosystems, and is also considered one of the ecosystems most vulnerable to both anthropogenic activities and climate change (Broadmeadow & Nisbet, 2004; Mendoza-Cariño et al., 2014; Pero & Quiroga, 2019). The ecological importance of this "transitional land-water" ecosystem lies in that it provides multiple services such as habitats for aquatic and terrestrial species, sediment filtering, flood control, stream channel stability and aquifer recharge (Naiman & Décamps, 1997; Valera et al., 2019).

The riparian zone is one of the ecosystems most disturbed by anthropogenic activities, and objectives for restoration plans should be clearly outlined in all cases of degeneration. Intact riparian forests can protect streams from nutrient or sediment pollution, and can also shield them from the effects of climate change such as river warming, while protecting the banks from erosion and flash flood events (R. K. Johnson & Almlöf, 2016; Turunen, Markkula, Rajakallio, & Aroviita, 2019). In addition, riparian forest ecosystems provide key habitats and ecological corridors for species migration (Figure 4 from; Naiman, Decamps, & Pollock, 1993).

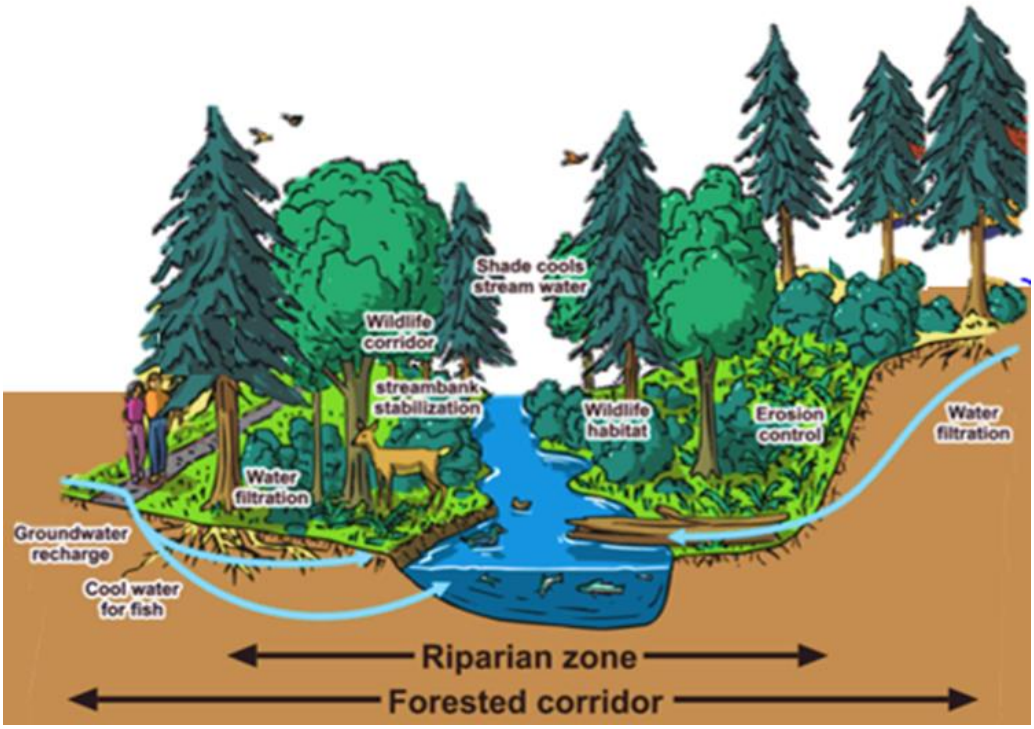


Figure 4 Riparian zone and forested corridor functions adapted from Regional District of Nanaimo (2014)

Alterations to landscape hydrology caused by anthropogenic activities lead to changes in the natural water and nutrient balance, which in turn contribute to the loss of important riparian habitats and to freshwater eutrophication. It is common for "agricultural rivers" to lose their natural riparian vegetation, which usually protects them from organic pollution, nutrients, fine inorganic sediments and pesticides from multiple diffuse locations. This loss has serious consequences for their ecosystem structure and function (Hladyz et al., 2011; Turunen et al., 2019). Urbanization often turns natural streams into channels; their riverbanks harden, and their natural riparian vegetation is either partially removed or replaced entirely with newly planted vegetation, reducing the riparian zone's ability to function as a natural ecosystem (Bernhardt & Palmer, 2007; Hu, Yue, & Zhou, 2019). Urbanization disturbs the natural nutrient filtering of riparian ecosystems, turning functioning ecosystems effectively into gutters that transport urban pollution downstream (Bernhardt & Palmer, 2007; Vought & Lacoursière, 2010). Furthermore, the vegetation of riparian zones can be disturbed in many other ways by urbanization, such as via the introduction/invasion of exotic plants used for forestry (e.g. conifer plantations, (Riipinen et al., 2010), the creation of specific forests for forestry (e. g. (Lecerf et al., 2005), forest clear-cutting (Mckie & Malmqvist, 2009), biofuel production, decorative landscaping, or through the provision of shelter for livestock (e. g. Rhododendron invasion (Hladyz et al. , 2011). Finally, the effects of land use cover/change dynamics are degrading riparian habitats and affecting their ecological functions and ecosystem services. Hence riparian zones stand out as vital for maintaining water quality and the "ecological status" of freshwater ecosystems (Almada et al., 2019).

ECOSYSTEM HEALTH AND "ECOLOGICAL STATUS" MEASUREMENT

The increasing threats to freshwater ecosystems and their management structures have garnered the attention of the international community. During the 1990s, actions were taken to begin to mitigate the impact of anthropogenic activities on freshwater ecosystems. International conferences like the "International Conference on Water and the Environment" (Dublin, 1992), which took place in the build up to the United Nations Conference on Environment and Development (UNCED, Brazil, 1992), both prompted the establishment of international water resource programs and institutions (Giordano & Wolf, 2003). The introduction of the Water Framework Directive (WFD) in 2000 aimed to create a new perspective in European water management to understand and integrate all aspects of the aquatic environment. The WFD requires an in-depth understanding of catchments and management under an ecological vision that considers human activities as a source of disturbance and water quality degradation (Kelly, 2013).

In support of this practice, the WFD adopted the Drivers-Pressures-State-Impacts-Responses framework (Oliveira, Lima, & Vieira, 2007). This aimed to provide a systematic understanding of anthropogenic impacts on, the environment, what causes them, and the subsequent measures taken to alleviate them (Figure 5). The framework also requires that a programme of future actions for managing anthropogenic pressures and improving ecosystem health be drawn up (European Commission, 2000; Nõges & Nõges, 2006).The WFD calls for a ‘catchment-based approach’ and ‘integrated river basin management’, both terms used to refer to the management of land and water as one system. Hence, WFD requires a paradigm shift in management towards system-based thinking and the adoption of an interdisciplinary, integrated, and holistic approach (Voulvoulis, 2012; Voulvoulis, Arpon, & Giakoumis, 2017). Systems are identified by their structure, their function and their state (health) is an expression of both of these aspects (Arnold & Wade, 2015; European Commission, 2000).

The WFD defines ecological status or potential as an “expression of the quality of the structure and functioning of surface water ecosystems”, i.e. a comprehensive indicator of an ecosystem's health (European Commission, 2000). Thus, the definition of good ecological status is a system free from anthropogenic pressures, or evidencing a slight biological deviation from what would be expected under reference conditions (European Commission, 2000). Additionally, the WFD created the concept of "Reference conditions", which is a high-quality example used in order to evaluate deviations of biological communities from the desired “good” conditions.

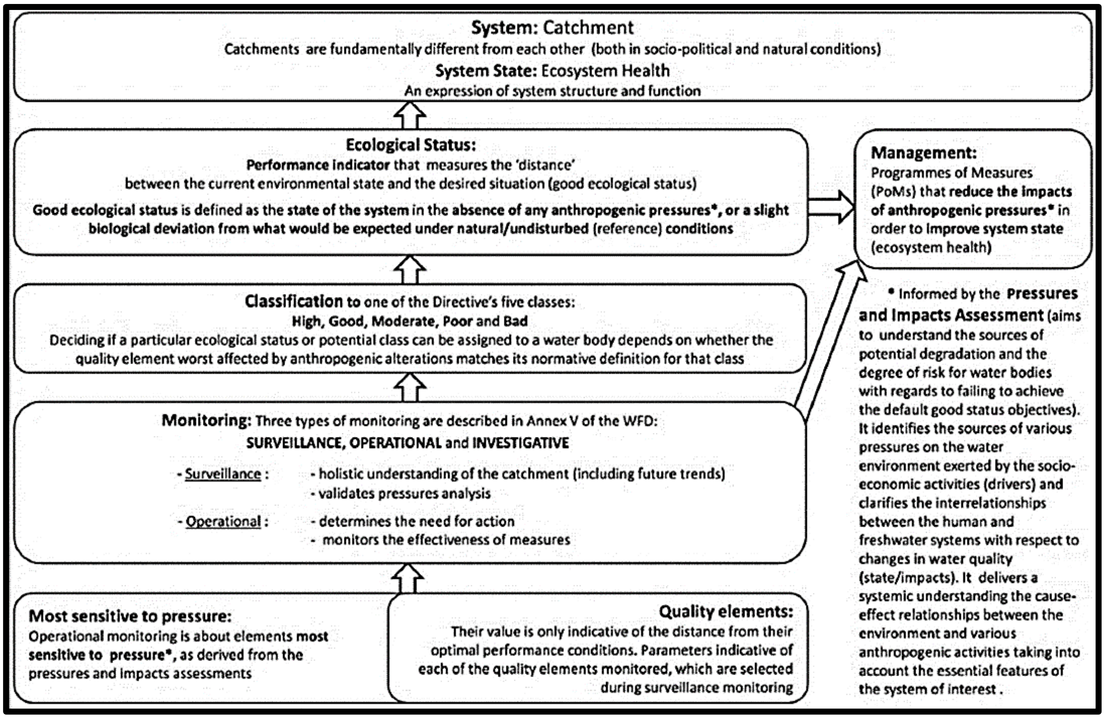


Figure 5 The Water Framework Directive (European Commission, 2000; Voulvoulis et al., 2017)

Hence the main objectives of the WFD for all bodies of water are: to reach good or high ecological status; to correctly monitor their current state; and to establish how far they are from good or high ecological status, thus indicating their management needs. The WFD proposes three phases for river management: diagnosis, monitoring and rehabilitation. Diagnosis of the ecological status is mainly done using biological and hydromorphological indicators (Domènech, 2016). Biological indicators can detect disturbances in the composition and functions of the assemblage of life forms that inhabit riparian ecosystems, and hydro-morphological indicators are based on the study of the physical-morphological structure, the riparian forest structure and the flow regimes associated with river ecosystems (Domènech, 2016).

WFD is considered the first European directive focused on environmental sustainability (Carter, 2007; C. W. Johnson, 2013). During the first WFD cycle, which operated from 2009 to 2015, the number of surface water bodies in a "good" state increased from 43% to 53% (Rijswick & Backes, 2015). The introduction of the WFD was revolutionary, and the Directive has been flagged as a benchmark for future environmental regulation (Josefsson, 2012).

CATALONIA AS A ROLE MODEL IN ECOHYDROLOGICAL MANAGEMENT

Since the introduction of the WFD and in some cases, even prior to its implementation, Catalonia (N. E. Spain) has contributed remarkably to the processes used in the diagnosis, rehabilitation and monitoring of rivers' ecological status. Biological monitoring has an important history in Spain, going back to the pioneering studies of Margalef (Margalef, 1969). Since 1979, tools for the ecological evaluation of two Catalan rivers, the Llobregat and the Besòs, have been rigorously developed and monitored (Prat & Rieradevall, 2006). The first method for measuring the biological quality of Mediterranean rivers using macroinvertebrates was developed in Catalonia from 1983-1984 (Prat, Puig & González,1983; Prat et al., 1984). However, in 1994, following on from studies completed, the "Diputació de Barcelona" approved the ECOBILL biomonitoring programme, which has been in place ever since. Prior to the introduction of the WFD indexes and of changes to biomonitoring tools, evaluation protocols has already been developed in both Catalonia and Spain (i. e. (BMWP´, Alba Tercedor et al., 2002); (FBILL ;(Muñoz & Prat, 1994)); (QBR index, Munné et al., 2003)). With the advent of the WDF, these new European ecological diagnosis/monitoring tools were adopted as official measures of ecological quality, and new tools were also developed (Munné et al., 2006; Munné & Prat, 2009; Pardo et al., 2002). All of this research, diagnosis and monitoring has contributed to the improvement of the ecological status of Catalan freshwater ecosystems.

Therefore, Spanish research, and Catalan research in particular, have long been at the forefront of the generation of new approaches and adaptations using these ecohydrological tools. This research is particularly important for developing countries which, as noted previously, have ecosystems that are threatened by anthropogenic activities, and suffer from a lack of research and consequently a lack of proper legislation and management (Acosta et al., 2009; Carrasco et al., 2014; Castro-López, Guerra-Cobián, & Prat, 2019; Ríos-Touma, Acosta, & Prat, 2014; Sirombra & Mesa, 2012; Villamarín, et al., 2013).

MEXICAN FRESHWATER AND ITS CURRENT ECOLOGICAL STATE

451585 hm^3 of renewable water is produced in Mexico every year. The southeast has two-thirds of the country's renewable water, but only one-fifth of the population, and contributes one-fifth of the national Gross Domestic Product (GDP). In contrast, the northern, central and northwestern regions have a third of Mexico's renewable water, are home to four-fifths of the population, and contribute four-fifths of the country's GDP (CONAGUA, 2018). Hence the renewable water available in the southeastern regions is per capita seven times higher than in the rest of Mexico (Figure 7).

Figure 7 Water availability in Mexico. In yellow regions with the lowest water availability and high national GDP. In green regions with the highest renewable water in Mexico, and the lowest national GDP. Adapted from (CONAGUA, 2018)

In Mexico, 216,593 hm^3-yr of water is extracted from the environment, of which 178,379 hm^3-yr is superficial (82%), 33,819 hm^3-yr is from underground waters (16%) and 4,395 hm^3-yr is rainwater (2%). Of this, 76% is used for agriculture, 14.4% is for household use, and 9.6 % is used by industry (CONAGUA, 2018). The total amount of wastewater that returns to the environment (i.e. rivers, lakes etc.) is 61,034 hm^3-yr, composed of 23, 877 hm^3-yr untreated wastewater (39%), 6,292 hm^3-yr from wastewater treatment plants (10%) and 30,866 hm^3-yr (51%) that is lost due to leaks in the collection and distribution systems. What is more, Mexico's lack of legislation concerning land use management has had negative consequences for the country's freshwater ecosystems and for biodiversity. It is also worth noting that Mexico is ranked 5th out of the 17 countries with the highest biodiversity and endemism levels in the world, meaning that Mexico is considered one of the world's few megadiverse countries (Mittermeier et al., 1997). This adds further importance to the evaluation, diagnosis and conservation of freshwater ecosystems in Mexico.

Since 1974, the Mexican National Water Commission (CONAGUA) has only monitored water quality in aquatic ecosystems using a physicochemical parameter methodology (Mathuriau et al., 2011). In 2003, biochemical oxygen demand (BOD), chemical oxygen demand (COD), and total suspended solids (TSS) were added to the monitoring methodology (Figure 8).

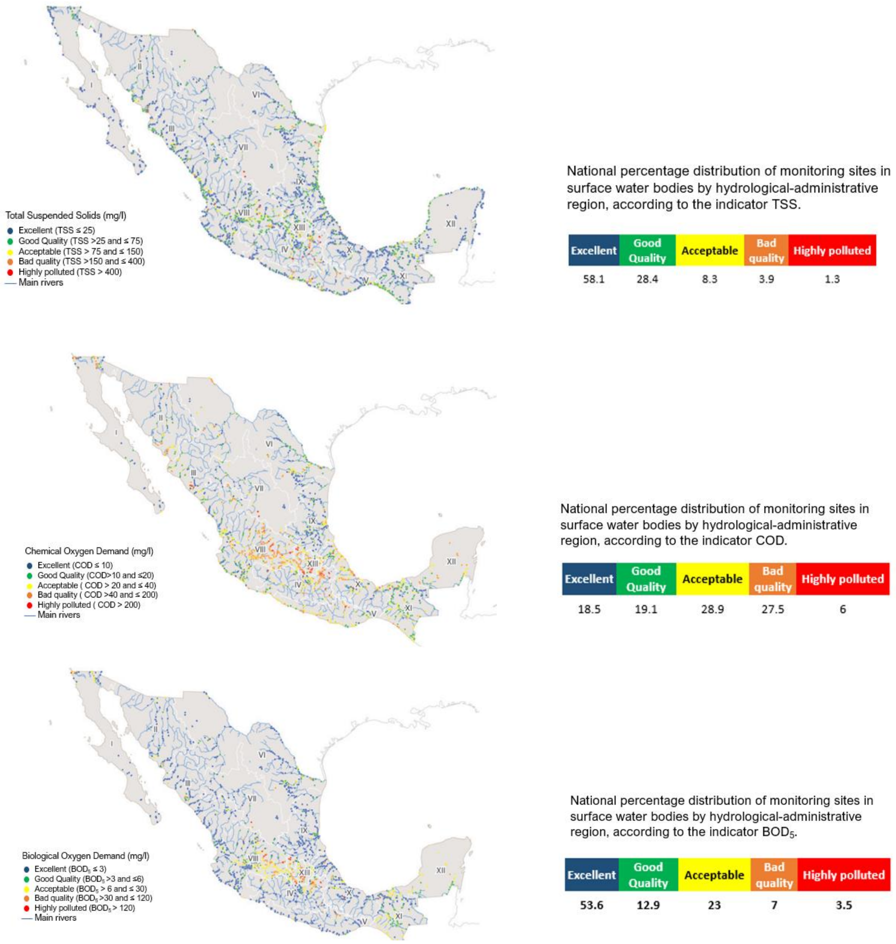


Figure 8 Parameters of water quality in Mexico adapted from CONAGUA (2018)

There are few studies in central Mexico that use biological indicators (e.g. macroinvertebrates, riparian forest/vegetation, fishes etc.) to diagnose and monitor riparian ecosystems (Mathuriau et al., 2011). This is due to the lack of ecological information available; for instance, in order to evaluate freshwater quality by using benthic macroinvertebrates, taxonomic keys are required, which are not available to researchers in the area (Caro-Borrero & Carmona-Jiménez, 2019; Mathuriau et al., 2011; Serrano-Balderas et al., 2016). Moreover, biomonitoring techniques are not included on the list of approved water evaluation methods outlined by the Mexican government.

On the other hand, whereas previous studies have underscored the importance of riparian vegetation buffers, the "Mexican National Water Law 1992” considers this zone as a federal area and not as an ecosystem (Almada et al., 2019; Chua et al., 2019; CONAGUA, 1992; Luke et al., 2019; Mendoza Cariño et al., 2014). Thus, the inadequate interpretation and application of this law creates a legislative gap that is hindering the implementation of sustainable management policies for riparian ecosystems in Mexico. Yet the most important limitation in Mexico is that it is becoming difficult to find riparian ecosystems without evidence of anthropogenic pressures that can function as reference sites (Mathuriau et al., 2011).

Given this lack of information, and the pressures that Mexican freshwater ecosystems are subject to, the National Commission for the Understanding and Usage of Biodiversity implemented the Priority Hydrological Regions (PHR) programme in 1998 (Arriaga-Cabrera, Aguilar-Sierra, & Alcocer-Durán, 2000). This program was focused on diagnosing the main catchments and freshwater ecosystems in Mexico. The PHR considered the biodiversity characteristics and the social and economic patterns of the identified areas in order to establish a reference framework that could be used by different sectors for the development of research plans and conservation/sustainable management programs. This program identified 110 priority hydrological regions, 75 of which have high biodiversity levels. 29 of these 75 had high biodiversity but little scientific information about them was available. Furthermore, the PHR identified 82 hydrological regions influenced by the different land use covers, 75 of which have been impacted considerably by anthropogenic activity (Arriaga-Cabrera et al., 2000). This program paved the way for integrating research and ecohydrological management in Mexico, highlighting the gaps and the future work needed for the diagnosis, monitoring and rehabilitation of Mexican freshwater ecosystems.

In brief, ascertaining the ecological state of the riparian ecosystems provides the basis for the sustainable management of water via the assessment of the performance and the structure of the ecosystem. The ecological state concept therefore ought to be considered in the diagnosis of Mexican freshwater ecosystems. The previous characterization of Mexican rivers completed using the PHR revealed significant gaps in the current knowledge and highlighted the actions required to protect riparian ecosystems in Mexico.

THE PESQUERÍA RIVER

Under the PHR framework, the Pesquería River was classified as a priority case for ecological evaluation given the high level of environmental degradation caused by anthropogenic activity in the region. In addition, the programme highlighted both the lack of a biological inventory for the region and a lack of biomonitoring in the river basin.

The Pesquería River is located in Hydrological region 24-Bravo Conchos (North-eastern Mexico) and flows through the states of Coahuila and Nuevo León. The Pesquería River has an area of 5,255.56 km^2, and its main course is 288.22 km in length. The river drains with a mean annual flow of 2.04 m3/s through the metropolitan area of Monterrey. The climate where the Pesquería River is located is extremely variable, but it is predominately semiarid, and the sub-basin has a mean elevation of 542 m above sea level with an average gradient of 0.4% [40]. The average temperature in the Pesquería River basin is between 20°C and 24°C, with a total annual rainfall of between 400 and 700 mm. The wet season occurs during the months of May to October, while the dry season is from November to April. The average annual temperature is usually between 18° and 22°C; the mean highest temperatures of 28° and 29°C are reached in July, and the minimum temperatures of 13° and 14°C are reached in January and December respectively. The predominant vegetation is sub-montane scrub with the Mesquite and halophilic vegetation typical of sandy deserts. Citrus production, livestock, aquaculture and rainfed agriculture are the main economic activities in the Pesquería River basin. The river water's natural conditions are consistent with mesohaline habitats.

In spite of the important role of the Pesquería in local ecosystems and its high level of degradation, only two master´s degree theses on the Pesquería have been published by the Autonomous University of Nuevo Leon (U.A.N.L.), both of which use biological indicators to evaluate the river water quality (Bermejo-Acosta, 2002; Torres-Muñiz, 2013). Although several studies of the Pesquería River that focus on the evaluation of the Diptera taxa composition due to the high rate of transmitting diseases caused by this order exist (Garza-Rodríguez, 2017; Rodríguez-Castro et al., 2004; Rodríguez-Castro, Garza-Rodríguez, & Martínez, 2018), there is a clear need for further investigation. Thus, a lack of research, management and policies are contributing to the continued degradation of the Pesquería River, and hindering its restoration, conservation and protection. This is what motivated us to evaluate the Pesquería River´s ecological status via an assessment of its riparian forest quality, land use dynamics and the influence of these two aspects on the River´s macroinvertebrate community.

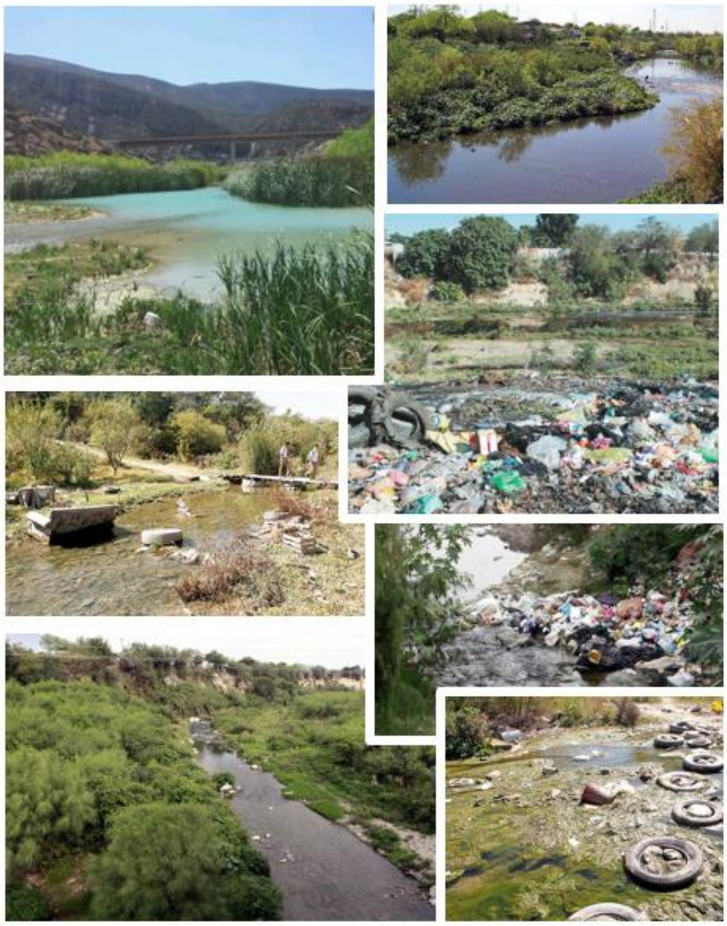


Figure 9 The Pesquería River´s current conditions

Objectives

The overarching goal of this PhD thesis is to evaluate the ecological status of the Pesquería River and assess the effects of landscape degradation along its riparian channel. Since changes in landscape composition can significantly affect a riparian channel, we aimed to evaluate how said changes can also impact biological quality and subsequently affect the ecological integrity of a river ecosystem. We focused on providing simple and reliable methods to define the ecological status and discern the effects of landscape degradation on the ecological state of the Pesquería River.

The thesis was divided into 3 chapters, corresponding to 3 article manuscripts (two of them already published, one to be submitted). More concretely, each chapter aimed to answer the following questions:

Chapter 1

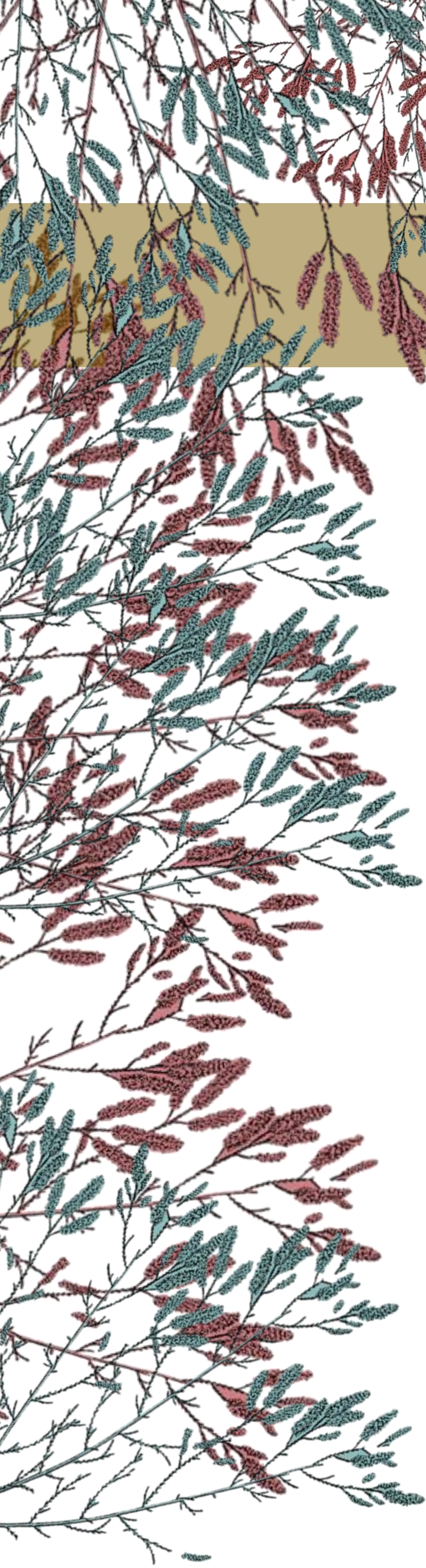
- Which land-use cover/change processes best describe the composition of the riparian channel over the last four decades (1976-2016)?
- Are these land-use cover/change processes able to elucidate the relationship between land use dynamics and the river's current ecological condition?

Chapter 2

- What are the main stressors that affect riparian forest quality in the Pesquería River?
- Is riparian forest quality a suitable indicator for the evaluation of the ecological status of the Pesquería river?

Chapter 3

- Is the macroinvertebrate community's composition influenced by the different categories of land use in the Pesquería River?
- Do other factors influence the macroinvertebrate composition (e.g. seasonality or the longitudinal natural variability)?
- Are the different taxa and the different biological metrics also impacted by these factors?
- Can we consider these taxa and metrics to be good bioindicators of the ecological status of the Pesquería River?



CHAPTER 1

On seràs demà?

On seràs quan tot comenci a girar

i el teu propi cos

et porti a algun lloc

on no has estat mai?

Que a idiotes com jo

ens fa perdre el cap

i els dies s'escapen.

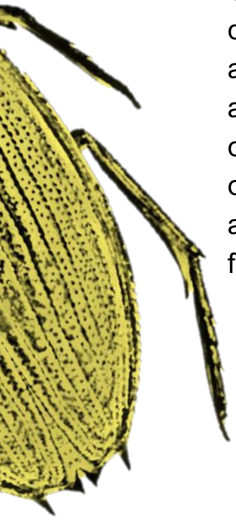
Les cendres cobreixen el parc.

-On serás demá?-

Joan Dausà i Riera

Dynamics of land use cover/change and their influence on the riparian channel ecosystem health: the case of the Pesquería River (N.E. Mexico)

In Mexico, intensive land use cover/change dynamics (LUCCD) and inadequate monitoring policies have resulted in the severe degradation of freshwater ecosystems. This study analyzed the LUCCD in the riparian zone of the Pesquería River sub-basin (PR), located in the northeast of the country. Two land use map series from 1976 and 2016 were used to characterize land use, and a 50-meter-wide riparian buffer along the main channel of the PRSB was delimited for analysis. After this, transition and probability of permanence matrices were calculated to identify the LUCCD of the PRSB's riparian zone. LUCCD analyses for the PRSB riparian zone showed disturbances caused by the degradation of its riparian forest. The LUCCD assessment indicated a major increase in anthropogenic activity, especially in urban zones, grasslands and areas of secondary vegetation. The study did show some evidence of revegetation, but this was most likely due to the abandonment of agricultural practices caused by rural emigration and extreme flash flood events. These results allowed for a comprehensive analysis of the LUCCD in the PRSB. They also highlight the most important changes in the riparian channel's land use dynamics in recent decades. We conclude that LUCCD analysis at riparian channel level should be viewed as a key tool for the improvement of conservation policies targeting freshwater ecosystems in Mexico.



INTRODUCTION

The earth's ecosystems are currently being damaged at an unprecedented rate due to both population growth and anthropogenic activity (Burdon et al., 2019; IPBES, 2019; Price et al., 2019). Since 1992, many urban areas have more than doubled in size, and it is expected that by 2050 there will be more than 25 million kilometers of paved roads (IPBES, 2019). What is more, over a third of the earth's land is now used for agriculture, a practice which also consumes more than 75% of the planet's fresh (IPBES, 2019). As these figures indicate, human activity has directly affected biotic diversity across the globe, resulting in deforestation, habitat fragmentation, freshwater pollution and, consequently, global warming (Fearnside, 2001; Sala et al., 2000). The impact of this rapid change on water sources has been particularly profound.

The effects of land use cover/change dynamics (LUCCD) can be seen at distinct spatial strata, including the watershed, the catchment, and the riparian zone (Zhou, Wu, & Peng, 2012)]. The riparian zone is the transition area between fluvial and terrestrial or riparian habitats, and is considered a vital part of aquatic ecosystems. It comprises the strip of vegetation adjacent to the body of water within a floodplain (Broadmeadow & Nisbet, 2004; Mendoza Cariño et al., 2014). Riparian zones have been impacted by anthropogenic landscapes that have developed in order to satisfy human needs (Polis, Anderson, & Holt, 1997; Xiang, Zhang, & Richardson, 2017). The degradation of riparian areas caused by changes in land use processes alters water surface hydrology, geomorphology, physico-chemistry and, as a result, the biodiversity of an area (Allan, 2004; Dudgeon et al., 2006; Masese et al., 2014; Vörösmarty et al., 2010). Changes in land use can lead to natural forests being replaced by agricultural pasture and exotic tree species at the basin and riparian corridor levels (Ferreira, Gulis, & Graça, 2006; Hladyz et al., 2010).

It has already been established that forest loss results in a low level of assimilation of nonpoint source pollutants in riparian ecosystems (Lowrance, Leonard, & Sheridan, 1985). Furthermore, several studies have found that forested riparian buffer zones have a positive impact on water quality, reducing the sediment load and nutrient concentrations in water (Dosskey et al., 2010; Fernández et al., 2014; Naiman, Decamps, & McClain, 2010; Scarsbrook & Halliday, 1999). Additionally, the riparian buffer forest offers shade that can protect streams from warming and from the effects of climate change (Johnson & Almlöf, 2016; Kiffney, Richardson, & Bull, 2003;

Sponseller, Benfield, & Valett, 2001). Riparian zone buffers are therefore increasingly being used as a management tool to protect streams from the effects of land use (Broadmeadow & Nisbet, 2004; Jones et al.,1999; Turunen et al., 2019).

Environmental degradation has often been documented through statistics that track rates of deforestation (Groombridge & Jenkins, 2000; Alejandro Velázquez et al., 2003; Wahlberg, Moilanen, & Hanski, 1996). Nonetheless, statistics are not always sufficient for the identification and prevention of further damage caused by LUCCD in the riparian zone. Hence, LUCCD must be considered in the ecological diagnosis and conservation of a riparian zone. Environmental degradation is particularly critical in developing countries where LUCCD are threatening natural resources (Velázquez et al., 2002). Although this is an urgent issue, many experts have highlighted the difficulty of creating a precise methodology for the evaluation of the effects and extent of LUCCD in a given area in order to protect natural resources (Velázquez et al., 2003).

Mexico is one of the world’s few megadiverse countries (FAO, 2015). It is ranked 5th out of the 17 countries with the highest biodiversity and endemism levels (Mendoza-Ponce et al., 2019; Mittermeier, et al., 1997). Nevertheless, the lack of legislation regarding land use management in Mexico has had negative consequences for the country’s riparian ecosystems. In Mexico, about 16% of the land is subject to periodic flooding, and is therefore considered part of the riparian ecosystem, or at least connected to it. More than 70% of these flood-prone areas have become urban and/or agricultural land, or are regularly flooded by reservoirs, and only about 2% remain a natural riparian ecosystem (Granados-Sánchez et al., 2006). The high rate of degradation in riparian channels caused by anthropogenic pressures makes it increasingly difficult to find “healthy” freshwater ecosystems in Mexico (Carmona-Jiménez & Caro-Borrero, 2017; Cotler-Ávalos, H., 2010; Mathuriau et al., 2011).

Within this context, we chose the Pesquería River Sub-basin (Nuevo Léon, México) for this study. According to the Priority Hydrological Regions (PHR) program, the Pesquería River Sub-basin (PRSB) was classified as a priority case for ecological evaluation given the high level of environmental degradation that has been caused by anthropogenic activity in the region (Arriaga-Cabrera et al., 2000). This study aims to understand the LUCCD in the riparian zone of the PRSB. In order to do so, we explore the dynamics of the land use cover of the PRSB riparian channel– that is; we define the main land use types; assess the extent of land use and cover change, and ascertain the most important land use change processes (i.e., degradation, revegetation) for the land use/cover classes between 1976 and 2016. The results elucidate how land use change dynamics can provide useful information about the current ecological status of the PRSB riparian zone. The discussion analyses the Pesquería River's riparian zone, its conversion trends and the relationship between land use dynamics and the River's current ecological condition.

METHODOLOGY

Study area and the PRSB riparian channel land use characterization.

The PRSB is located in Hydrological region 24-Bravo Conchos (Northeastern Mexico), and flows through the states of Coahuila and Nuevo León (Figure 1), (INEGI, 2017). The PRSB has an area of 5,255.56 km^2, and its main course is 288.22 km in length. The river drains with a mean annual flow of 2.04 m^3/s through the metropolitan area of Monterrey. The climate where the PRSB is located is extremely variable, but it is predominately semiarid, and the sub-basin has a mean elevation of 542 m above sea level with an average gradient of 0.4% (García, 2004). The average temperature in the PRSB is between 20°C and 24°C, with a total annual rainfall of between 400 and 700 mm (Fierro, 2015). The predominant vegetation is sub-montane scrub with the Mesquite and halophilic vegetation typical of sandy deserts. Citrus production, livestock, aquaculture and rainfed agriculture are the main economic activities in the PRSB (CONABIO, 2017).

The study area was delimited based on the geomorphological limits of the PRSB. We used the digital map database of the INEGI (National Mexican Institute of Statistics and Geography) to obtain the land use information data for the PRSB. This study used two national land cover maps (1976 and 2016) in vector format (INE - INEGI, 1997; INEGI, 2016a). The original classification included more than 15 classes of land use covers. In order to ensure the concordance of the land use cover map series, these were grouped into the five categories described in Table 1: Riparian Forest (RF), Secondary Forest (SF, i.e. secondary vegetation (Chokkalingam & de Jong, 2001), Grassland (GL), Agriculture (AG) and Urban Zones (UZ) (Alanís-Rodríguez et al., 2015).

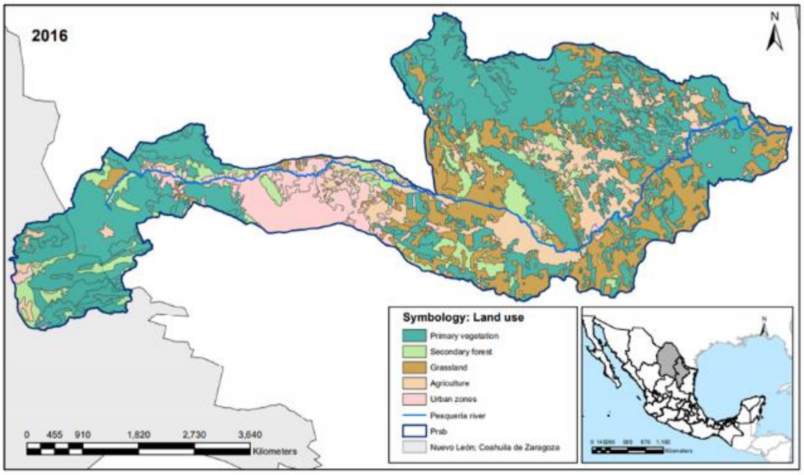


Figure 1 Study Area. Pesquería River sub-basin land use (2016).

Table 1 The Pesquería River´s land use cover categories

Land Use Typology	Classification Criteria for Each Land Use Cover
Microphyllous desert shrubland	
Rosetophyllous desert shrubland	
Tamaulipan thornscrub	Riparian Forest
Xerophitic scrubland mesquite	
Piedmont scrub	
Secondary vegetation of Tamaulipan thornscrub	
Secondary vegetation of Piedmont scrub	Secondary Forest
Secondary vegetation of Microphyllous desert shrubland	
Permanent grassland	Grassland
Induced grassland	
Annual rainfed agriculture	Agriculture
Annual irrigated agriculture	
Urban zone	
Human settlements	Urban
Non-vegetated area	

The LUCCD processes were subsequently grouped into different types of anthropogenic activity (transformation, deforestation, degradation) and revegetation processes (revegetation and recuperation). The first group, "anthropogenic degradation", describes riparian forest and secondary forest degradation as the land converts to anthropogenic land use cover (i.e. through deforestation, agriculture, livestock, urbanization); and includes the transitional state of degradation from a riparian forest into a secondary forest (secondary succession). The second group, "revegetation", comprises land that is recovering from anthropogenic land use cover and changing into a secondary forest or riparian forest (Velázquez et al., 2003).

For this study, we used ArcGIS to demarcate a 50m (width) buffer along both sides of the stream which incorporated the riparian zone (ESRI, 2012). This was in line with the HDRI methodology protocol for riparian hydromorphologic evaluation (Munné et al., 2006). Once the riparian zone had been defined, each category of land use in the riparian buffer was determined. Following this, we calculated the total area for each category in the riparian buffer of the PRSB (Figure 2). All cartographic documents were created using the geographical information system ArcGIS 10.1. (ESRI, 2012).

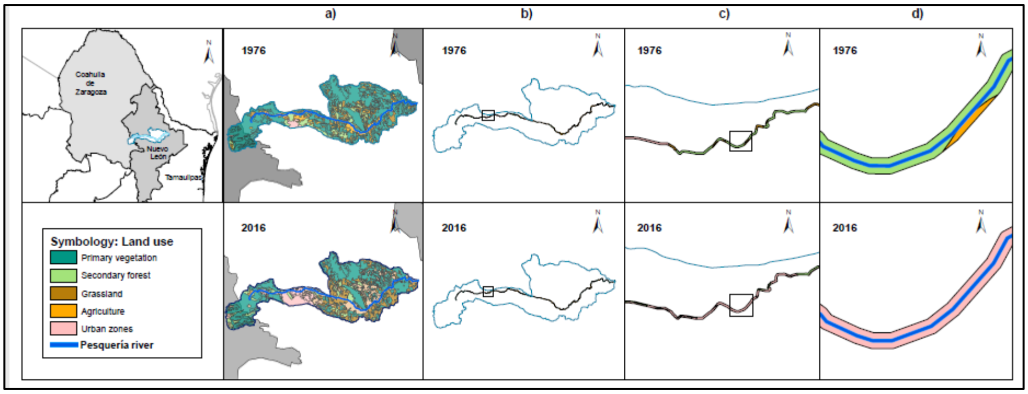


Figure 2 Illustration of the riparian buffer preparation for LUCCD for 1976-2016.

PRSB riparian channel conversion rates

To determine the conversion rate of the riparian channel land use, we used the equation cited below (FAO, 1996).

$$t = [1-(S_2/S_1)]1/n$$

Where t is the rate of conversion, S_1 and S_2 are the land use areas at the start and end of the period respectively, and the variable n is the difference of years between the studied periods (≈40 years).

Land use cover/change dynamics analysis

INEGI maps for land use and vegetation from series I (1976) and VI (2016) (hereafter referred to as TS_1 and TS_2 respectively) were used in order to perform the LUCCD analysis (INE - INEGI, 1997; INEGI, 2016a). Firstly, we consulted the previous classifications (Table1) in order to maintain the same categories for each series. Secondly, we integrated TS_1 and TS_2 using the same overlapping database maps at the same scale (1:250000). Thus, two overlapped maps could be used to construct a land use transition matrix to describe the dynamics of the different land use cover areas.

The land use transition matrix creates a quantitative description of the system's state of transition (Zhang et al., 2017). The computed transition matrix consists of rows that display categories at time T1 and columns that display categories at time T2 (Figure 3). The diagonal elements in the transition matrix indicate the area of the landscape that remained unchanged during the studied period, while the rest of the cells estimate the area for each type of land use that has changed, allowing us to understand the dynamics of the change at a regional or local scale (Castelán-Vega et al.,2007).

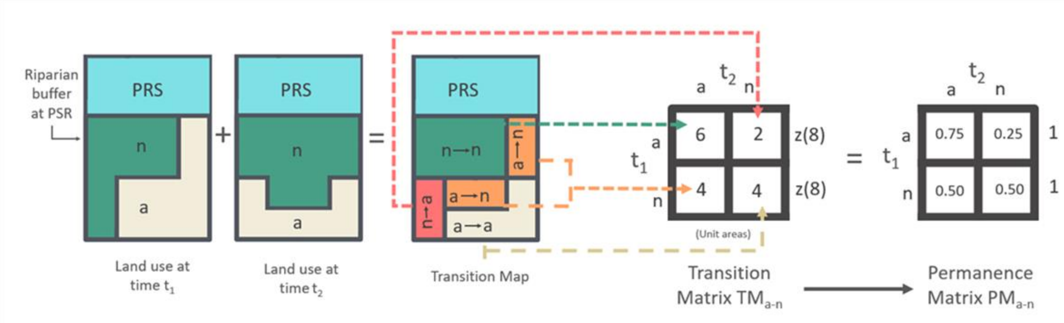


Figure 3 This geometric operation combines the information from each land use cover map into a transition map. The transition and permanence matrices are obtained from this map. For probability of permanence matrix we used the equation: PCa-n: Sa-n(t1)/(t2).

Where: Sa-n = Surface of "n-a" in the transition matrix of the land use surface for the first period (t1); z= land use total surface for the second period (t2). The sum of each box in the probability of permanence matrix should result in 1. (∑pa-n = 1.)

Once the transition matrix was complete, a probability of permanence matrix was calculated. This matrix is generated by dividing each of the cells from the transition matrix that show the area in hectares of each category of land use by the total area of the category analysed (Castelán-Vega et al., 2007; Müller-Hansen et al., 2017). For the probability of permanence matrix, we considered that the probability of permanence (PC_{a-n}) was proportional to the remaining area of the same category between S_{a-n} (t_1) and z (t_2). Figure 3 also shows the representative expression of all the processes of the **LUCCD** analysis. For the interpretation of our analysis, the following classes of probability were used: 0-33% (low), 34-66% (average) and 67-100% (high), (Sánchez et al., 2003).

RESULTS

Riparian channel land use characterization and conversion rates in the PRSB.

The total area calculated for the riparian buffer was 21.86 km^2. The land use area for each category revealed that the PRSB had undergone considerable changes over the last few decades (Table 2).

Table 2 Pesquería sub-basin area changes from 1976 to 2016 (figures in km2)

CATEGORY \ LAND USE SERIE	SERIE I 1976	SERIE VI 2016
Riparian forest	14.55	6.28
Secondary forest	1.44	2.49
Grasslands (cultivated/ induced)	1.52	4.69
Agriculture (rainfed / irrigated)	3.53	4.55
Urban land use	0.81	3.85
TOTAL Km^2	21.86	21.86

The estimated conversion rates between 1976 and 2016 are given in Figure 4. The bars above zero represent the land use classes that increased in surface area, whereas the bars below zero indicate the land use classes that decreased in surface area. 38 % of the riparian forest (i.e. 8.270 km^2) has been lost, with an annual conversion rate of -2.86 %. Secondary vegetation increased by 4.8 % (i.e. 1.05 km^2), with an annual conversion rate of 1.9 %. Grasslands increased by 14.48 % (i.e. 3.16 km^2), with an annual conversion rate of 3.95%. Agriculture increased by 4.65 % (i.e. 1.01 km^2), with an annual conversion rate of 0.8%. Finally, the urban area increased by 13.88% (i.e. 3.03 km^2), with an annual conversion rate of 5.49%.

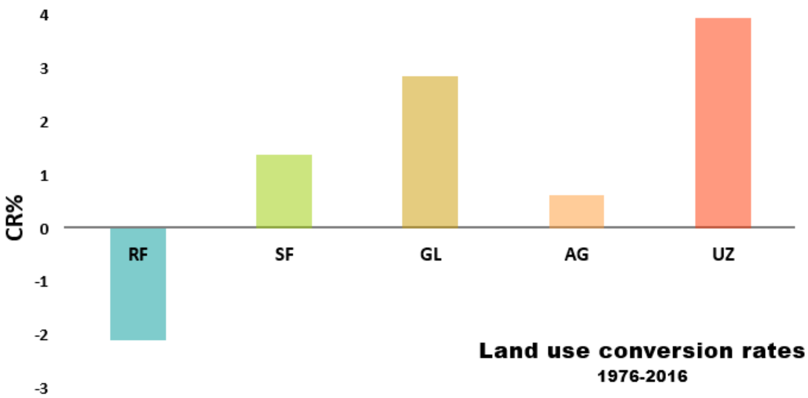


Figure 4 Conversion rates among each land use category expressed in percentages.

Depicting land use cover/change dynamics.

The transition matrix shown in Table 3 provides the conversion figures for each land use category as the result of crossing t1 (1976) and t2 (2016). Overall, in the evaluated area (21.86 km^2), over 28% of the riparian forest remained unchanged, whereas 3% of the secondary forest, 3% of the grassland, 11% of the agricultural land and 3% of the urban zones remained as anthropogenic land use classes during the period studied (1976-2016).

Table 3 Transition matrix for LUCCD 1976-2016. The columns on the far right indicate the surface area and total percentage of each category for 1976 (t1). The bottom two rows indicate the surface area and percentage for each category in 2016 (t2).

	LAND USE 2016 (t_2)						
LAND USE 1976 (t_1)	Riparian forest (km^2)	Secondary forest (km^2)	Grasslands (km^2)	Agriculture (km^2)	Urban zones (km^2)	**Total 1976 (km^2)**	Percentage
Riparian forest (km^2)	6.04	1.32	3.88	2.06	1.26	**14.55**	66.54
Secondary forest (km^2)	0.00	0.56	0.03	0.04	0.81	**1.44**	6.59
Grasslands (km^2)	0.11	0.15	0.70	0.06	0.50	**1.52**	6.97
Agriculture (km^2)	0.12	0.39	0.07	2.39	0.56	**3.54**	16.17
Urban zones (km^2)	0.01	0.07	0.00	0.00	0.73	**0.82**	3.73
Total 2016 (km^2)	**6.28**	**2.49**	**4.69**	**4.55**	**3.85**	**21.86**	100
Percentage	28.72	11.40	21.44	20.83	17.61	100	

Regarding conversions, land use cover dynamics are described in the probability of permanence matrix (Table 4).

Table 4 Probability of permanence matrix, 1976-2016. The probability of permanence for each land use category is represented in the diagonally highlighted boxes. The boxes above the diagonal line show the degradation probability, and the boxes below it shows the probability of reversion for each land use category.

LAND USE 1976	LAND USE 2016					
	Riparian forest	Secondary forest	Grasslands	Agriculture	Urban zones	TOTAL
Riparian forest	**0.415**	0.091	0.267	0.141	0.087	1
Secondary forest	0.000	**0.390**	0.021	0.030	0.559	1
Grasslands	0.072	0.100	**0.462**	0.040	0.325	1
Agriculture	0.034	0.111	0.021	**0.676**	0.158	1
Urban zones	0.018	0.085	0.000	0.000	**0.894**	1

About 59% of the riparian forest as it existed in 1976 was converted into anthropogenic classes (fig. d). 9% of this figure became secondary forest, 27% became grassland, 14% became agricultural land, and 9% became urban zones. In addition, 61% of the secondary forest was turned in grassland (2%), agricultural land (3%) and urban zones (56%) respectively. Further, 37% of the grassland became agricultural land (4%) and urban zones (33%). 2% of the agricultural land was converted into grassland and 16% into urban zones. With regards the reversion processes, the secondary forest did not present any changes, whereas 10% of the grassland became secondary forest and only 7% of the grassland reverted to riparian forest. Additionally, 2% of the agricultural land reverted to grassland, 11% to secondary forest and only 3% to riparian forest. Interestingly, 9% of urban zones reverted to secondary forest and 2% to riparian forest. The riparian forest, secondary forest and grassland presented average probabilities of permanence of 41%, 39% and 46% respectively. Nonetheless, the agricultural and urban zones both presented a high probability of permanence in the riparian zone with 68% and 89% in each case (Figure 5).

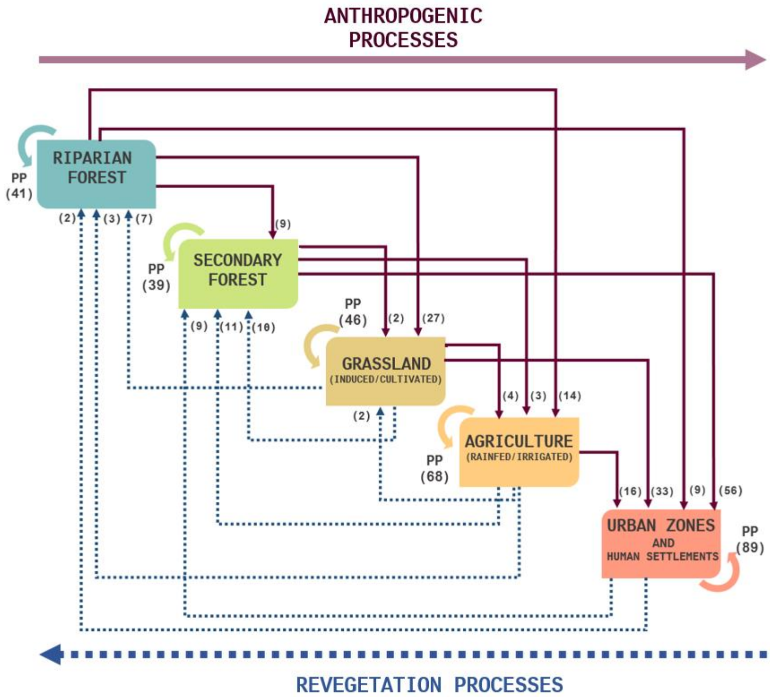


Figure 5 Flowchart depicting the probabilistic transition values (as a percentage) and the probability of permanence (PP, as a percentage). Only values >1% are presented.

Description of LUCCD processes PRSB riparian zone

The land use cover processes for the PRSB riparian channel in the period studied are represented in Figure 6. Anthropogenic degradation, either in the riparian forest or in the secondary forest, affects 37% of the total riparian forest in the PRSB. In addition, 6% of the riparian forest has converted to secondary forest (i.e. secondary succession). However, revegetation only took place across 3.2% of the PRSB surface. There was no evidence of transitional processes of recovery from secondary forest to riparian vegetation.

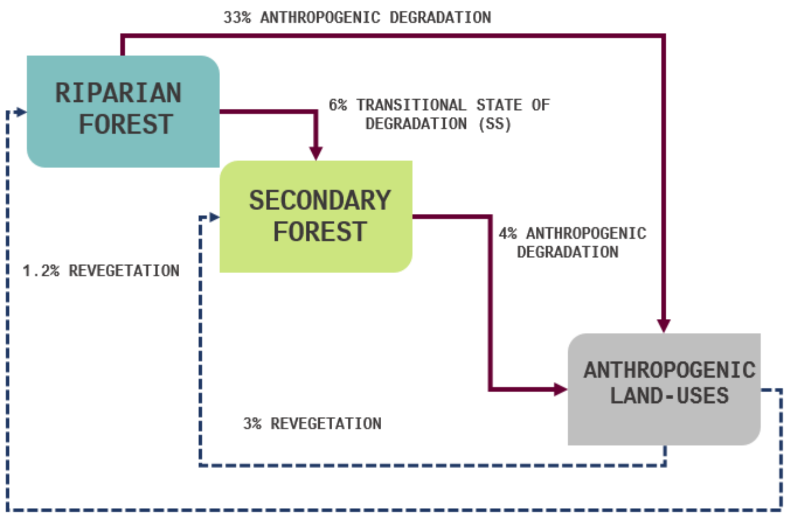


Figure 6 Conversion dynamics (as a percentage) in the PRSB.

Discussion

The land use cover dynamics and processes described in this study suggest that anthropogenic processes are directly influencing the ecological condition of the PRSB riparian zone. The riparian forest in the PRSB is changing into grassland and agricultural zones, while the secondary forest is mostly becoming part of urban zones. These findings coincide with the low probability of permanence statistics for the riparian forest and secondary forest in the riparian channel at the local and regional scales (Pérez-Miranda et al., 2012; Torre-Barajas et al., 2018). Given that urban and agricultural settlements are well defined in the PRSB, both categories presented a high probability of permanence into the riparian zone.

Land use conversion trends in the PRSB riparian channel

The PRSB land use characterization shows that the riparian forest is composed mainly of semiarid shrublands, (Alanís-Rodríguez et al., 2015) whereas the dominant anthropogenic land use activities are related to grassland and urban zones. The present findings are consistent with other research which has found that the loss pattern of shrublands is related to an increase in grassland and agricultural land (Pérez-Miranda et al., 2012; Torres-Barajas et al., 2018). According to Vela & Lozano (2010) an increase of 10% in grassland area and 5% in agricultural land area is expected in the northeast region of Mexico (where the PRSB is located) by 2030. In addition, the PRSB flows through the metropolitan area of Monterrey city which, according to the INEGI demographic survey, was the third most populated city in Mexico in 2010 (INEGI, 2016b). Population growth increases demand on the local ecosystems, which in turn degrades the riparian forest and converts larger areas into secondary forest (Vela & Lozano, 2007). Agricultural conversion rates were not notable for the time prior to the period studied when compared with other land use dynamics. A possible explanation for this might be that the agricultural zones in the PRSB have been concentrated in the outskirts of the city, and although the city has expanded since 1976, the agricultural zones have remained in these locations (Castro-López, Guerra-Cobián, & Prat, 2019; Torres-Barajas et al., 2018). Additionally, the abandonment of agricultural land due to rural emigration tends to increase secondary forest land use (Díaz, 2003; García-Ruiz et al., 2013). This factor may explain why the secondary forest presented a higher conversion rate than the agricultural land.

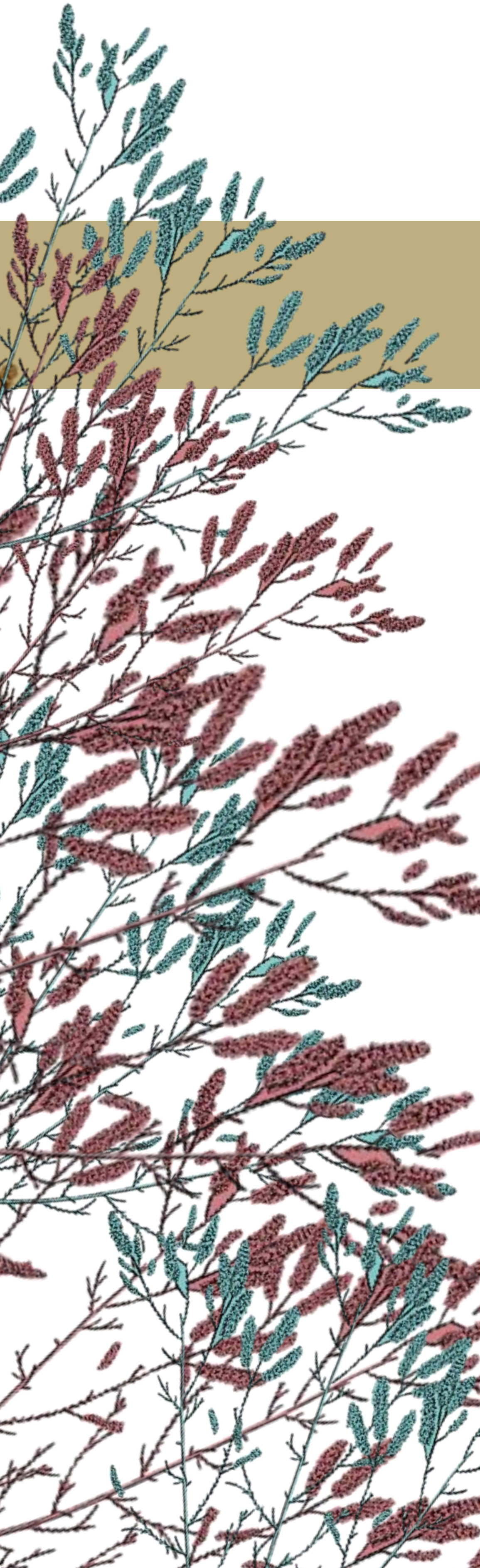
“Anthropogenic vs revegetation” processes in the Pesquería River Sub Basin

The **LUCCD** processes show that the PRSB´s riparian channel has been significantly disturbed given the level of natural land use degradation. Land use degradation is understood as a process were disturbances have gone beyond the resilience of the land, producing irreversible damage to the ecosystems (Hill et al., 2008). In addition, this gradually allows for further environmental decline, which can culminate in desertification (Brandt & Thornes, 1996; Puigdefabregas, 1995). For our particular case, anthropogenic activities degraded almost 39% of the riparian forest and over 4% of the secondary forest. The revegetation process can take several decades, depending on the degradation level and the evolution of the soil and climatic conditions. In addition, these conditions can delay or even impede natural revegetation (Bienes et al., 2016). For the PRSB, the revegetation process was split into two stages, the first being revegetation from anthropogenic land to secondary forest (3%). In this process, the reestablishment of vegetation is mainly down to the lack of agriculture, which is often abandoned due to the low profitability and rural emigration trends in semiarid regions (García-Ruiz et al., 2013). Nevertheless, during the second stage, 1.2% of anthropogenic land use (mainly grassland) became riparian forest, in line with prior studies undertaken in Mexico (Jaimes et al., 2003). Urban zones presented the lowest level of revegetation. Previous studies in the PRSB have highlighted the presence of alien species, local industry, human settlements and a clandestine waste dump as factors obstructing revegetation (Castro-López et al., 2019). Further, given that the PRSB is almost completely engulfed by the metropolitan area of Monterrey, revegetation of the urban land zones is extremely unlikely (Fierro, 2015). The metropolitan area of Monterrey is also vulnerable to flash flood events (Aguilar-Barajas et al., 2019). During the period studied (1976-2016), five extreme hydrometeorological events (1989-2010) affected the city (Aguilar-Barajas et al., 2015). These events "damaged" the anthropogenic land use in the riparian zone of the PRSB, devastating human settlements near the river margin, and consequently positively influencing the revegetation process (Aguilar-Barajas et al., 2015, 2019). Anthropogenic activity and episodic high-intensity natural disturbances generate significant changes in habitat distribution, which in turn produce a reduction in biodiversity by facilitating the establishment of invasive species (Bhattarai & Cronin, 2014; Calderon-Aguilera et al., 2012; Newton et al., 2009). Invasive species also hinder shrub restoration processes in these dry areas (Porensky et al., 2014). Finally, Castro-López et al., (Castro-López et al., 2019) have confirmed that one of the possible causes of low riparian forest quality in the PRSB is, in fact, the introduction of invasive species.

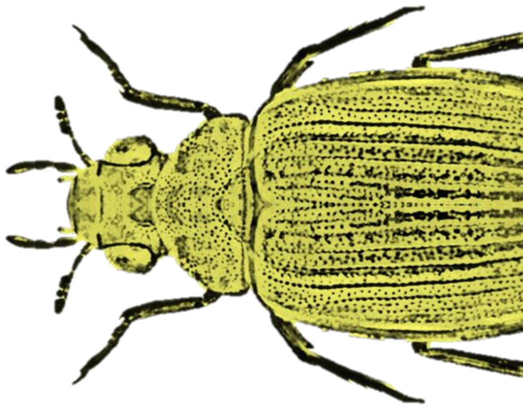
Management challenges for the riparian zones in Mexico. The PRSB's ecological status.

Even though Mexico is a megadiverse country, biodiversity and ecosystem integrity are threatened by anthropogenic activities such as agriculture and urban development. Thus, entire landscapes are changing in Mexico for a variety of reasons, which include the nation's current economic priorities (industrial expansion and hydrocarbon exploration), the increasing population, a lack of land use planning, poor monitoring and obsolete environmental laws governing freshwater ecosystems (Calderon-aguilera et al., 2012). The lack of land use planning and inadequate monitoring policies in Mexico have resulted in the severe degradation of freshwater ecosystems (Carmona-Jiménez & Caro-Borrero, 2017; Castro-López et al., 2019; Mathuriau et al., 2011; Mendoza-Ponce et al., 2019). While previous studies have underscored the importance of riparian vegetation buffers, the "Mexican National Water Law of 1992" considers this zone as a federal area and not as an ecosystem (Almada et al., 2019; Chua et al., 2019; CONAGUA, 1992; Luke et al., 2019; Mendoza-Cariño et al., 2014). Thus, the inadequate interpretation and application of this law creates a legislative gap that is hindering the implementation of sustainable management policies. LUCCD studies have proved to be useful for evaluating anthropogenic activity at riparian zone level (Dodds & Oakes, 2008; Valera et al., 2019; Zhou et al., 2012). LUCCD analyses at riparian channel level should, therefore, be a key consideration in freshwater ecosystem management in Mexico (Broadmeadow & Nisbet, 2004; Brogna et al., 2018; Dosskey et al., 2010).

In our particular case, a 50-meter-wide riparian buffer (on each bank) was enough to assess the LUCCD processes of the PRSB. However, a 50-meter-wide buffer will not be sufficient to protect the river's headwaters from the increasing urbanization of the metropolitan area in the future (Valera et al., 2019). Agricultural areas on the city's outskirts were found not to put added pressure on the riparian forest since there is only farmland on one bank of the river, and the other side usually still had vegetation. Agricultural land in the PRSB was found to be at some distance from the river, and so cannot have any significant impact. The restoration of degraded areas in the riparian zone could improve the current ecological condition of the PRSB. Nevertheless, restoration and preservation plans will not be effective so long as the Mexican government prioritizes economic affairs over environmental legislation and protection.



CHAPTER 2



I walked across an empty land

I knew the pathway like the back of my hand

I felt the earth beneath my feet

Sat by the river, and it made me complete

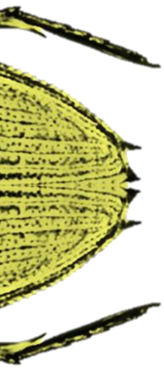
Oh, simple thing, where have you gone?

I'm getting old, and I need something to rely on

-Somewhere only we know-

Keane

THE ROLE OF RIPARIAN VEGETATION IN THE EVALUATION OF ECOSYSTEM HEALTH. THE CASE OF SEMIARID CONDITIONS IN NORTHERN MEXICO



The Pesquería River (North-eastern Mexico) has long been subjected to considerable anthropogenic pressures. For this reason, it has been identified by the Mexican National Commission for the Knowledge and Usage of Biodiversity (CONABIO) as a priority resource to be evaluated and restored. In order to establish the means required for the restoration of the river, the condition of its riparian ecosystem must be evaluated. To evaluate the quality of the riparian forest we adapted the QBR index methodology for Mediterranean rivers for the semiarid rivers of north-eastern Mexico (QBR-RNMX). The QBR-RNMX index included modifications to the four sections that comprise the original index, and their values range between 0 and 100. Using the five levels of riparian quality defined in the index, in the area surrounding the Pesquería we found poor or very poor conditions at 66% of the sampling sites, average-good conditions at 27% of the sites, and only one sampling site with excellent conditions. These results show that the riparian forest has been impacted significantly by urbanisation, agriculture and the presence of many invasive species. We recommend the application of the QBR-RNMX annually in order to evaluate the riparian forest's quality, and to assess its ecological status. This may be used for the establishment of restoration plans in high-impact zones and contingency plans to eliminate invasive species along the Pesquería River.

INTRODUCTION

An increased interest in the ecological condition of rivers has been observed around the world, since rivers are ecosystems that are subject to anthropogenic pressures and have significant consequences for freshwater ecosystems (Master et al., 1997; Naiman & Turner 2000; Allan, 2004). Anthropogenic impacts upon freshwater ecosystems alter their physical and biological characteristics, thus modifying their natural condition (Nilsson et al., 2007). The European Union Water Framework Directive (D.O.C.E., 2000) considers the ecological condition of freshwater bodies as a measure of ecosystem health. The river health can be evaluated using a series of biological, physicochemical, and hydromorphological indicators (Mendoza et al., 2014), including some specific methodologies for assessing riparian areas. Therefore, the characterization of the reference conditions is a key process in the successful evaluation of the ecological state of a river (Feio et al., 2014).

The riparian zone is one of the areas most disturbed by anthropogenic activities and objectives for restoration plans should be clearly stated. The riparian zone is defined as the area of transition between the river channel and the adjacent land-based ecosystem and includes both the flowing channel and the surrounding land that is influenced by fluctuations in water level (Malanson, 1993). The heterogeneity and complexity of riparian ecosystems makes studying, evaluating and restoring them difficult, and means that managing them sustainably is a complex endeavour (Chovanec et al., 2000; Reed & Carpenter 2002). Since they represent an interface between the land and the water, one of their functions is to regulate the river water quality by acting as a filter, preventing soil erosion, regulating temperature and light levels and decreasing the number of contaminants that enter the stream. Riverside vegetation is an important indicator in the evaluation of the ecological status of rivers used in land planning and ecosystem management (Suárez et al., 2002).

The most important pressure factors associated with a global reduction in biodiversity and the degradation of the riparian zones are deeply linked to loss of habitat due to anthropogenic impacts, combined with the invasion of non-native flora and fauna (Ruzycki et al., 2003; Millennium Ecosystem Assessment, 2005). These pressures deeply affect the natural environmental heterogeneity of riverbank environments, causing considerable damage to both the levels of biodiversity and to ecological processes (Ward 1998; Strayer et al., 2003; Townsend et al., 2003). This often has grave consequences for human health and for the local economy (Vitousek et al., 1996).

Various procedures for evaluating and measuring the quality of riparian vegetation have been developed, (Raven et al., 1998; Boon et al., 1998). The QBR index (Qualitat del Bosc de Ribera) was developed to assess the riparian forest quality of Mediterranean rivers in Spain (Munné et al., 1998a, 1998b, 2003). Numerous authors have adapted this methodology to different geographical regions because of its simplicity and efficiency (Acosta et al., 2009; Colwell, 2007; Kutschker et al., 2009; Sirombra & Mesa 2012; Carrasco et al., 2014).

The QBR index considers key aspects of riparian vegetation such as coverage and structure, and aspects of the morphology of the riparian zone, such as anthropogenic intervention in the landscape. It is worth noting that in the investigation carried out by Suárez & Vidal-Abarca (2000), they conclude that the index must be adapted in order to consider the local environment, placing considerable emphasis on ephemeral rivers.

In Mexico, the QBR index has been applied to the El Tunal and Sauceda rivers in the state of Durango, and the results showed the current pressures on the riparian forest ecosystem. The ease of application, the low costs incurred, and the reliability of the information generated were noted as important factors in the decision to use this methodology (Rodríguez-Téllez et al., 2012[a], 2016[b]). Previous work suggests that with the appropriate changes, the QBR index allows for the in-situ evaluation of the conservation value of riverbank vegetation, including the assessment of anthropogenic impacts on any riparian ecosystem (Fernández et al., 2009).

Given the lack of information and considerable pressures that Mexican rivers are subject to, the CONABIO (National Commission for the Understanding and Usage of Biodiversity) implemented the Priority Hydrological Regions (RHP) programme in 1998 (Arriaga et al., 2009). Currently, the programme considers the Pesquería River (Nuevo Léon, México) as a priority case for ecological evaluation. The aim of this investigation is to fully understand the current quality of the Pesquería River's riparian forest by typifying it and applying the QBR index, which has been adapted for specific local conditions.

MATERIALS AND METHODS

Study area and river characterization

The Pesquería River (within Hydrological Region 24-Bravo-Conchos) has a catchment area of 5255.56 km^2 and its main course is 288.22 km in length (Figure 1). The river drains its waters from West to East with an annual average flow of 2.04 m^3/s through the states of Coahuila and Nuevo Léon. The climate where the sub-basin is located is extremely variable, but is predominately semiarid, and has a mean elevation of 542 masl with an average gradient of 0.4%. The average temperature in the sub-basin is between 20°-24°C with a total annual rainfall of between 400-700mm (Ferriño, 2016).

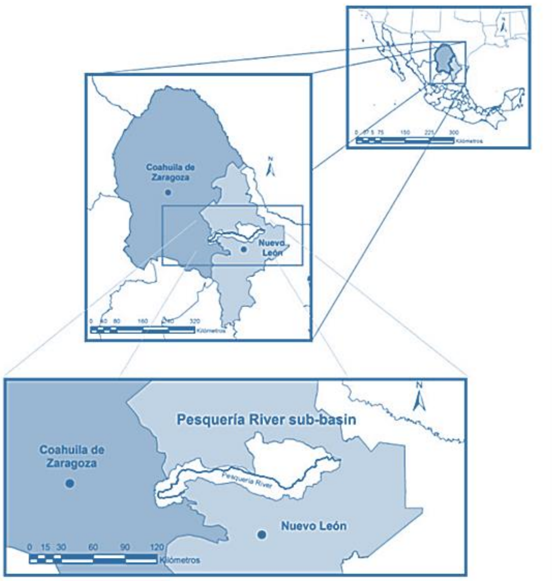


Figure 1 Area of study: Pesquería River sub-basin (located between coordinates 25°48´15.67” N; 100°39´15.80” O and 25°39´21.94” N; 99°41´11.87” O)

Prior to the study, land use in the basin was characterized using digital maps from the digital map database of the INEGI (National Institute of Statistics and Geography of Mexico) (INEGI, 2015). We established the area of the riparian zone as stipulated in section XLVII of the third article of the "Mexican National Water Law 2016", (LAN, 2016). The distance between sampling sites was chosen so that they did not exceed 10km (Acosta et al., 2009), given that at larger distances the factors being evaluated may lose continuity, and parts of the vegetation may remain unanalysed (Rodríguez-Téllez et al. 2012a). The river characterization was conducted in February 2016. It consisted of an inventory of the composition and structure of the vegetation as well as an *in situ* physicochemical analysis of the water in order to determine the nature of the water present in the study area, and the possible anthropogenic effects on the river. This study was conducted over 108 km of the hydrological sub-basin of the Pesquería River where 15 sampling sites where selected (Figure 2) using digital maps from the INEGI, which were obtained from the Geographic Information system GAIA (INEGI, 2015).

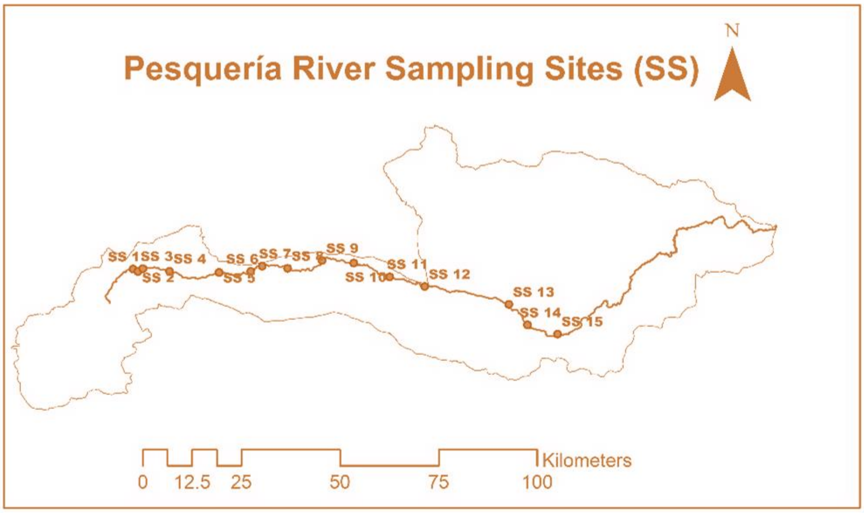


Figure 2 Pesquería River sampling sites

A transect of a maximum of 150 meters was estimated for each sampling site, taking into consideration the riparian zone and the width of the channel. For each sampling site, the QBR-RNMX was applied using the field sheet (Appendix A), containing the adapted version of the index. Aerial photographs obtained via drone were used to evaluate the level of riparian coverage (vegetation coverage and connectivity between the adjacent forest ecosystem and the riparian forest), the structure of the coverage (tree coverage and concentration of halophytes), and any modifications to the waterway. Images were taken at two heights in order to show waters above and below the source, and to create an orthomosaic map with various images of the stretch studied (Figure 3).

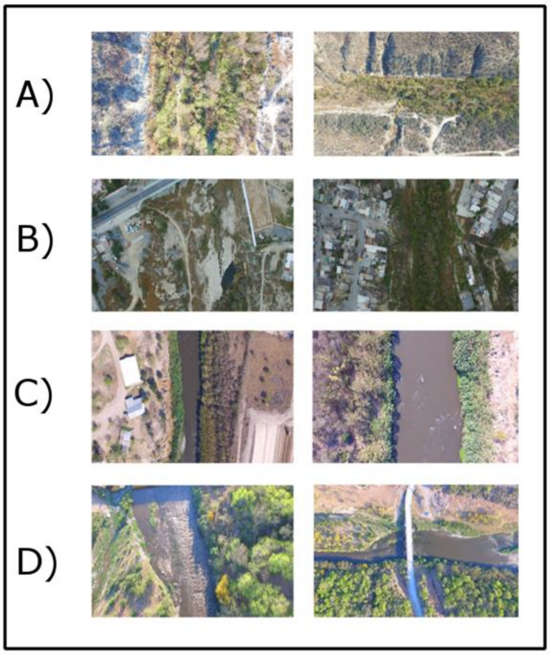


Figure 3 Aerial pictures taken with the drone. a) Reference condition zone (sampling site 1); b) Industrial and urban zone (sampling site 4); c) Agricultural impacted zone (sampling site 12); Non-impacted agricultural zone (sampling site 15)

In order to evaluate the third category, the most abundant species of vegetation and the geomorphological type corresponding to each sampling site were determined *in situ*. For this study halophytes and shrubs that grow between the riparian zone and the channel were recorded. They increase the value of the index as they offer a habitat for many species. Statistical analyses were conducted using the open source software R (R Core Team, 2016) with R Studio 1.0.153 Data (Racine, J. 2011). Pearson´s correlation was used to analyse the relationship between the QBR-RNMX and the physicochemical parameters.

QBR-RNMX Index

When evaluating riverbank forests using the QBR index proposed by Munné et al. (1998; 2003), there should be a focus on the essential aspects of riverbank vegetation according to four different categories (Jáimez-Cuellar, et al. 2002). The adaptation of the QBR index for use along the Pesquería River, for which we propose the name QBR-RNMX (QBR- Ríos del Norte de

México [Northern Rivers of Mexico]), included the modification of several categories based on previous adaptations of the QBR to other rivers (Acosta et al. 2009; Munné et al. 2003; Munné et al., dirs., 2006).

For the first category, the percentages for evaluating riverbank vegetation coverage were modified (Table 1). This does not affect the evaluation, since the aridity gradient in this climate reduces the natural density of the arboreal layer and prevents species from colonizing (Tabacchi et al. 1996). Furthermore, the salinity of the systems found here usually limits the survival of many potential species (Suárez & Vidal-Abarca, 2000).

Table 1 Differences in the evaluation criteria between QBR index and QBR index for Ephemeral rivers to assess the total riparian cover.

Score	QBR INDEX (Munné *et al.* 2003)	Score	QBR INDEX Ephemeral rivers (Munné et al., dirs., 2006)
25	**> 80 % of riparian cover**	**25**	**> 50 % of riparian cover**
10	**50-80 % of riparian cover**	**10**	**30-50 % de of riparian cover**
5	**10-50 % of riparian cover**	**5**	**10-30 % of riparian cover**
0	**< 10 % of riparian cover**	**0**	**< 10 % of riparian cover**

For the second section of the criteria, the expression regarding “tree coverage less than 50% and other shrub coverage between 10 and 25%” was adapted as follows: “tree coverage less than 25% and other shrub coverage between 10 and 25%”, considering the vegetation characteristics of the river, and grading them in the same manner (Vidal- Abarca *et al.* 2004).

The section that required the most modifications was section three, which was adapted to consider the geomorphology and number of autochthonous and allochthonous species present. In this section, the possible definitions for the geomorphological type of the riparian zone were reduced from three to two, offering only the following typologies: headwaters, and middle stretches. As a tributary of the San Juan River, the Pesquería River does not fit the characteristics of the third type, which as defined by Munné *et al.* (2003) represents the final stretch of a river. The number of autochthonous and allochthonous species was evaluated during the characterization in order to set boundaries between quality classes in this category. Within this section, an extra option was removed, which reads as follows: “if there exists any continuity in the community throughout the river (between 75% and 50% of the riparian zone)". This was because during the investigation no area was found to demonstrate this characteristic, and further, it is considered to be within the extra positive aspects gradable for each section. Finally, for the fourth section, one of the "extra" negative values was imported from the third section, which evaluates the presence of waste in the stretch of river studied; an extra negative marking point was added for the presence of permanent waste in the river that is difficult to remove (Acosta, et al. 2009).

RESULTS

River characterization

The characterization of the Pesquería River showed morphological characteristics consistent with loamy basins, and an elevated concentration of salt in the water, as well as vegetation and fauna typical of semiarid zones. The distribution of the land use and vegetation obtained for the sub-basin was: scrubland (61%), mesquite-huizachal (16%), woodland (2.3%), thicket (0.65%), agricultural, livestock and forestry use (18.7%), major towns (1.34%), and areas without vegetation (0.04%).

Riparian zone vegetation

A total of 14 species of riparian trees and shrubs were recorded for all transects, covering every sampling site. Six of the species were present at the majority of the sites, while 2 of the 14 were recorded at a single site. The vegetation inventory revealed seven native species. Four are considered representative of the region while three are widely distributed across Mexico. The rest were identified as invasive species (Appendix B). Some of the common species are not specific to the riparian area and are widely distributed in the basin, while the species associated exclusively with the riparian areas (e.g. *Salix nigra*) were not the most abundant in many cases.

River water characteristics and the QBR-RNMX

The physicochemical characterization is an indicator of the potential anthropogenic pressures. It also shows the high natural salt content of the river water (see the total of dissolved solids and the conductivity at site 1, which is close to reference conditions). The physiochemical parameters used in the characterization are presented in Table 2. The physicochemical results show a close relationship with those obtained using the QBR-RNMX index.

Table 2 Physicochemical parameters in the Pesquería River

Sampling sites	pH	Conductivity (μS /cm)	Salinity (ppt)	TSS (ppt)	TDS (ppm)	Cl^- (mg/L)	SO^{-2}_4 (mg/L)	DO (mg/L)	BOD (mg/L)
SS 1	**6.21**	**1907**	**0.92**	**1**	**894.33**	**547.30**	**500.00**	**8.90**	**8.40**
SS 2	**6.97**	**1683**	**0.81**	**4**	**792.33**	**323.27**	**386.00**	**8.84**	**13.30**
SS 3	**7.02**	**1707**	**0.92**	**2**	**886.67**	**180.83**	**450.00**	**8.60**	**9.25**
SS 4	**7.38**	**1663**	**0.91**	**123**	**880.33**	**430.88**	**775.00**	**8.70**	**45.64**
SS 5	**8.5**	**10170**	**3.85**	**259**	**3473.00**	**779.57**	**838.33**	**5.92**	**19.00**
SS 6	**6.16**	**4100**	**3.57**	**19**	**3230.00**	**1251.53**	**220.00**	**7.13**	**9.50**
SS 7	**6.59**	**3720**	**2.54**	**12**	**2337.00**	**809.71**	**1527.00**	**7.19**	**9.90**
SS 8	**7.09**	**3810**	**2.41**	**13**	**2200.00**	**583.70**	**1800.00**	**7.82**	**13.30**
SS 9	**6.45**	**2269**	**2.36**	**8**	**3220.00**	**564.69**	**200.00**	**6.67**	**15.50**
SS 10	**6.77**	**2122**	**1.06**	**4**	**1000.00**	**226.49**	**250.00**	**5.61**	**9.10**
SS 11	**6.2**	**2160**	**1.09**	**5**	**1000.00**	**221.83**	**600.00**	**6.81**	**17.30**
SS 12	**6.7**	**1897**	**0.86**	**20**	**828.67**	**253.55**	**558.00**	**6.20**	**15.60**
SS 13	**6.13**	**2123**	**1.01**	**94**	**966.00**	**177.36**	**320.00**	**6.78**	**22.50**
SS 14	**6.96**	**2283**	**1.08**	**41**	**1010.00**	**314.90**	**380.00**	**6.28**	**19.50**
SS 15	**7.08**	**2637**	**1.13**	**46**	**1120.00**	**657.30**	**1074.00**	**5.58**	**18.70**

Note. Total Suspended Solids (TSS), Total Dissolved Solids (TDS), Chlorides (Cl^-), Sulfates (SO^{-2}_4), Dissolved Oxygen (DO), Biochemical Oxygen Demand (BOD).

The results of the Pearson correlation are presented in Table 3; this table indicates the correlation between each evaluation category of the QBR-RNMX with the physicochemical parameters obtained during the characterization of the Pesquería River. The results indicate that categories I, II and IV display a negative correlation with most physicochemical parameters. Only category III (largely influenced by the presence of invasive species) displayed no significant correlations.

Table 3 Pearson correlation coefficients and significance between QBR-RNMX categories and physicochemical parameters in the Pesquería River

QBR-RNMX	pH	Conductivity	Salinity	TSS	TDS	SO^{-2}_4	Na^+	Cl^-	DO	BOD
I BLOCK	-0.28	**-0.67****	**-0.69****	-0.31	**-0.69****	-0.11	**-0.70****	-0.48	-0.02	-0.07
II BLOCK	-0.33	**-0.54***	**-0.57***	-0.5	-0.60*	0.05	**-0.59***	**-0.56***	0.28	-0.27
III BLOCK	-0.14	-0.1	-0.12	-0.21	-0.15	0.39	-0.14	-0.05	0.23	-0.14
IV BLOCK	-0.2	-0.47	**-0.51***	-0.31	**-0.55***	0.18	**-0.54***	-0.49	0.09	-0.21

Note. Significant correlations are in bold. BOD: biochemical oxygen demand; DO: dissolved oxygen; TDS: total dissolved solids; TSS: total suspended solids**.** * $p < 0.05$; **: $p < 0.01$; ***: $p < 0.001$

Assessing riparian vegetation quality using THE QBR-RNMX

A total of 15 sampling sites were evaluated where the average value of the QBR-RNMX index for the Pesquería River was 59, which indicates the presence of major disturbances throughout the basin. Only 6% of the sites sampled were of excellent quality and without disturbances (sampling site 1, QBR-RNMX = 100), while 27% were good quality but showed some disturbances (sampling sites 2, 13, 14 and 15, QBR-RNMX = 90, 85, 80, 85), 34% were of intermediate quality with significant disturbances (sampling sites 3, 7, 8, 11 and 12, QBR-RNMX = 65, 65, 70, 65 y 70). 13% of the stations showed significant disturbances and were of poor quality (sampling sites 9 and 10, QBR-RNMX = 30, 40). Finally, 20% (sampling sites 4, 5, 6, QBR-RNMX = 15, 15 y 5) of the sites evaluated showed evidence of extreme disturbances and had very poor-quality riparian vegetation (Figure 4 & 5).

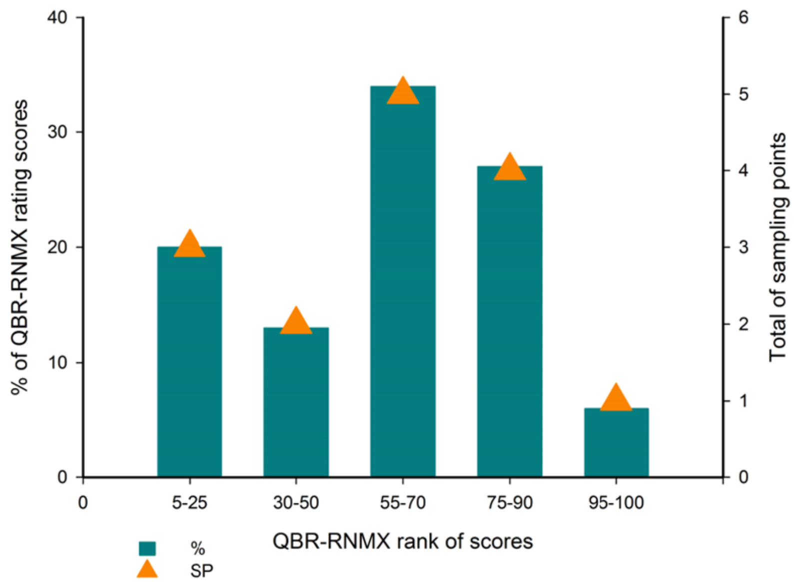


FIGURE 4 Sampling sites in each of the five categories of QBR-RNMX. Quality increases form left to right

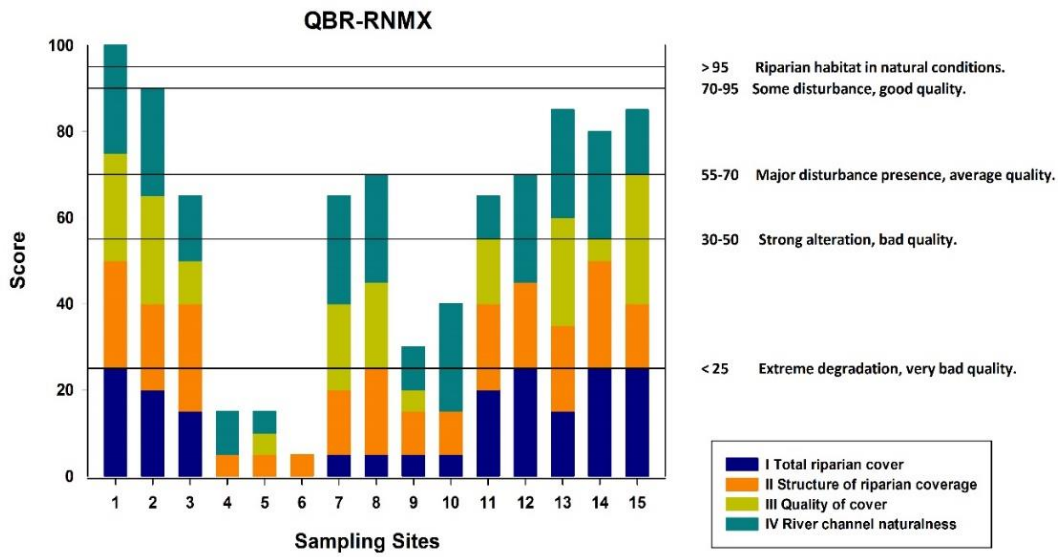


Figure 5 Final score obtained for each sampling site and the value for each of the four categories in which the index is divided. The limits between quality classes are shown in the figure.

DISCUSSION

The quality of riparian vegetation along the Pesquería River as measured using the QBR-RNMX

The use of the QBR methodology to establish the environmental health of the riparian areas of the Pesquería has been demonstrated to be useful. Different levels of degradation have been detected. Previous adaptations of the QBR index explain that scores > 95 could be considered as reference condition sites (Kazoglou et al. 2010). The only site in our study with these conditions is located at the Pesquería River's headwaters. Nevertheless, the increase of human activities is related to the lower index scores (Vannote *et al*. 1980). More than 80 km of the main reach of the Pesquería River crosses the city of Monterrey (Mexico), and the city's edifices and the urban area is located close to the river. Another factor that negatively impacted the QBR-RNMX values is the use of the river as a disposal site for waste by local people.

The agricultural practices on the city's outskirts produces lower pressures on the riparian ecosystem than the urban areas, and the majority of sites sampled in this area gave values of moderate quality. The agricultural areas did show reduced quality values mostly in block I; this is due to the reduced level of connectivity between the riparian zone and the naturally occurring adjacent vegetation (Rodríguez-Téllez *et al.* 2016[b]). In our study, the majority of the sampling sites in the agricultural area had farmland only in one bank of the river, whereas the other side usually still had vegetation. In each case, the agricultural area was normally found to be at some distance from the river itself; far enough to produce any relevant impact upon the QBR-RNMX scores. Therefore, the major reductions in the final scores in the index are due to both the lack of vegetation coverage and the connectivity to other ecosystems.

Transverse structures such as roads and bridges also contribute to the modification of the natural river channel and to soil erosion. They also act as routes of invasion for exotic species and as barriers that alter the dispersion patterns of native species (Smith & Armesto, 2002). In this study, we have tried to ensure that the role of such structures was minimal. Since they were used to access most sampling sites, the stretch evaluated was usually located 50 metres either side of these structures, although the presence of several invasive species cannot be discounted. As invasive species are one of the most important problems in Mexican riparian forests, we chose to examine this aspect in detail.

The role of invasive species

In recent decades, human activities have played an important role in transforming landscapes by reducing and changing the natural vegetation cover (Décamps et al., 1988). The replacement and eradication of native riparian communities by non-native ones leads to the simplification of the structural heterogeneity (Rodríguez & Herrera, 1993). Alien species reduce the diversity and abundance of native species, changing the structure and function of ecosystems (Lowe et al., 2000). Nearly 50% of the tree and shrub taxa found were invasive (6 of 14) and this is one of the more important features that prevent the riparian area from having very good or good conditions. The most predominant invasive species found throughout the investigation were *Arundo donax* (giant cane), *Ricinus communis* (castor bean) and *Tamarix aphylla* (salt cedar).

In Mexico, giant cane has proven to be one of the most difficult plants to eradicate. This species represents a substantial threat to ecosystems since it can drain rivers and streams (Boose & Hoolt, 1999). It may transform habitats by displacing native species, thus diminishing the levels of diversity and modifying the structure and composition of species and increasing the risk of bushfires (Contreras-Arquieta, 2007). The castor bean is often present in zones where there has been a previous anthropogenic or natural disturbance such as at the edge of roads, human settlements, and degraded riparian zones (Scarpa & Guerci, 1982). The salt cedar invades natural vegetation and is also capable of making soil more saline as it concentrates salt in its roots (Whaley, O. Q. *et al.* 2010). Its abundance near the Pesquería is in part due to the naturally high salt content of the basin. These three invasive species have historically caused most damage to Mexican water resources. This is due to the large quantities of water that they consume which, in turn, increases the salinity of the water by concentration and increases hydrological stress in semiarid regions (IMTA, 2007).

Interestingly the presence of invasive species does not correlate with the QBR index. This is because at some sites where the QBR score is low (sites 3, 4, 5, 7) there are few invasive species, while some sites with higher QBR scores host numerous invasive species. Again, the poor vegetation cover appears to be the most important reason for a low QBR-RNMX score, and not the displacement of native species by invaders.

The importance of substrate composition and river water salinity in the QBR values.

The characteristics of the banks of Pesquería River are typical of a loamy basin. The granulometric composition of the river bed ranged from pebbles found at its edges, to sand and clay found in the centre of the river channel, but in the banks the loamy natural substrate predominates which is rich in gypsum (calcium sulphate) and halite (sodium chloride). The composition of the sediments and the level of salinity are two of the main factors influencing the riparian vegetation and the river aquatic ecosystems (Moreno et al., 1996; Suárez & Vidal-Abarca, 2000; Vidal- Abarca *et al.* 2004). Moreover, it explains why the number of species of trees and shrubs is so low (only eight natural and six invaders see Appendix B) compared with other basins where the vegetation might include more typical riparian trees such as alders. In fact, of the eight native riparian plants, many of them are trees (such as Mesquite) that are present across the entire basin. Trees are abundant because of their adaptation to the aridity of the climate and the high salt content of the soil. Showing a significant negative relationship, the Pearson correlation between evaluation categories I, II and IV and the physicochemical parameters (Conductivity, Salinity, TDS, Cl^-, Na^+), reinforces the importance of salt in the composition and structure of the riparian vegetation. Even though the Pesquería River has naturally high levels of salinity, the increase in salinity levels in areas of the river located close to the city demonstrates the anthropogenic impact of urbanization. This is mainly due to the discharge of clandestine waste from industry and from water treatment plants.

It is interesting to note that salinity increases at site 4 not only due to the influx of salt from industrial waste, but also due to the low river level (in the dry period) caused in part by the withdrawal of water from the river to supply Monterrey. As can be seen in table 1, the salt value declines form site 10 downstream due to the inputs from sewage plants that have lower concentrations of salt than sites upstream. The high conductivity of the water explains the relatively poor biodiversity of the riparian plants on the banks. Simultaneously, it provides an example of the utility of the QBR as a quality index that can describe the degradation of riparian areas even in these unfavourable conditions for plants.

The use of drones as an ecological tool

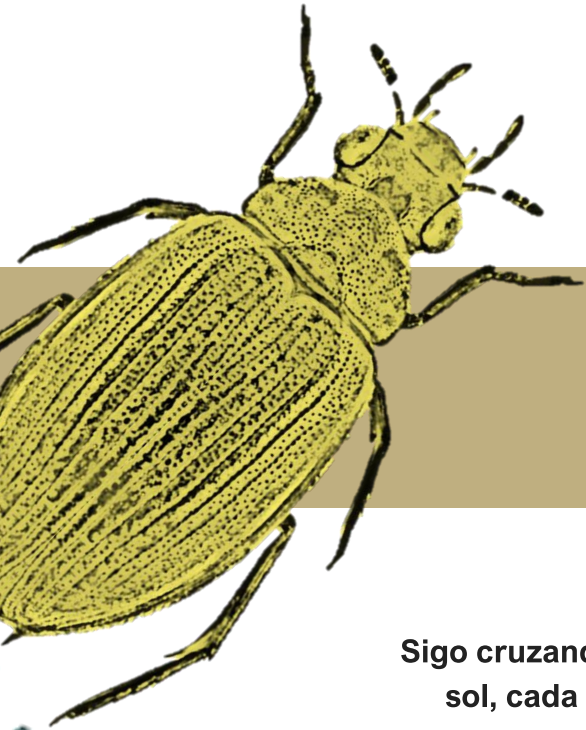
Drones have great potential for monitoring ecosystem dynamics as they provide a low-cost and low-impact solution for environmental managers working in a variety of settings (Ivosevic et al. 2015; Zhang et al.2016).The use of a drone in this investigation proved to be extremely practical for the evaluation of anthropogenic impacts on the riparian zone, and allowed us to clearly establish the composition, connectivity, distribution and amount (%) of vegetation coverage in both the agricultural and urban areas. It is worth adding that the use of the drone was key in determining the geomorphological type of the Pesquería River which is necessary in the evaluation of the third category of the QBR-RNMX (Quality of the cover), (Appendix A).

CONCLUSIONS AND CHALLENGES

This is the first study to adapt the QBR index for the evaluation of riparian vegetation quality in north-eastern Mexico, and the first attempt to evaluate the riparian vegetation of the Pesquería River. The assessment of riparian forest quality using the QBR-RNMX alongside other biological indicators and physicochemical parameters provides an overall picture of the general health of the Pesquería River. We recommend the application of the QBR-RNMX index for the management and development of ecological policies in Mexico.

As anthropogenic pressures and invasive species have the biggest impact on the Pesquería River’s riparian vegetation, we encourage an annual quality measurement using the QBR-RNMX index. A vegetation inventory must also be completed during the restoration process, and a plan to eradicate invasive species needs to be put in place. Conservation plans that allow for the preservation and reestablishment of the riparian forests must be drawn up urgently. We also suggest that the river management strategy be multidisciplinary so that managers from different fields can collaborate in the restoration of the river and its habitats. Further, this work is a clear example of the use of the QBR-RNMX index as a cost-benefit tool to determine the quality of a riparian forest. The index could be straightforwardly applied elsewhere in Mexico, and in other countries with similar morphological, ecological and hydrological characteristics.

We have demonstrated that the main anthropogenic impacts are related to the sparse vegetation cover in the riparian area. When considerable vegetation cover is present, invasive species are dominant. The rehabilitation measures should therefore prevent further degradation of the riparian forests and enhance restoration efforts. The challenge that remains is how to use this information to preserve and/ or restore the banks of the Pesquería River. Yet these measures will not be effective if the local population is not well informed about how the riparian forest supports the ecosystem, and consequently, their lives. The Government should therefore provide social and economic programs that engage local people in restoration activities. This will prevent further degradation and ensure that the rehabilitation measures are sustainable. Finally, we also encourage the use of drones for conducting more accurate surveys of the characteristics of riparian area.

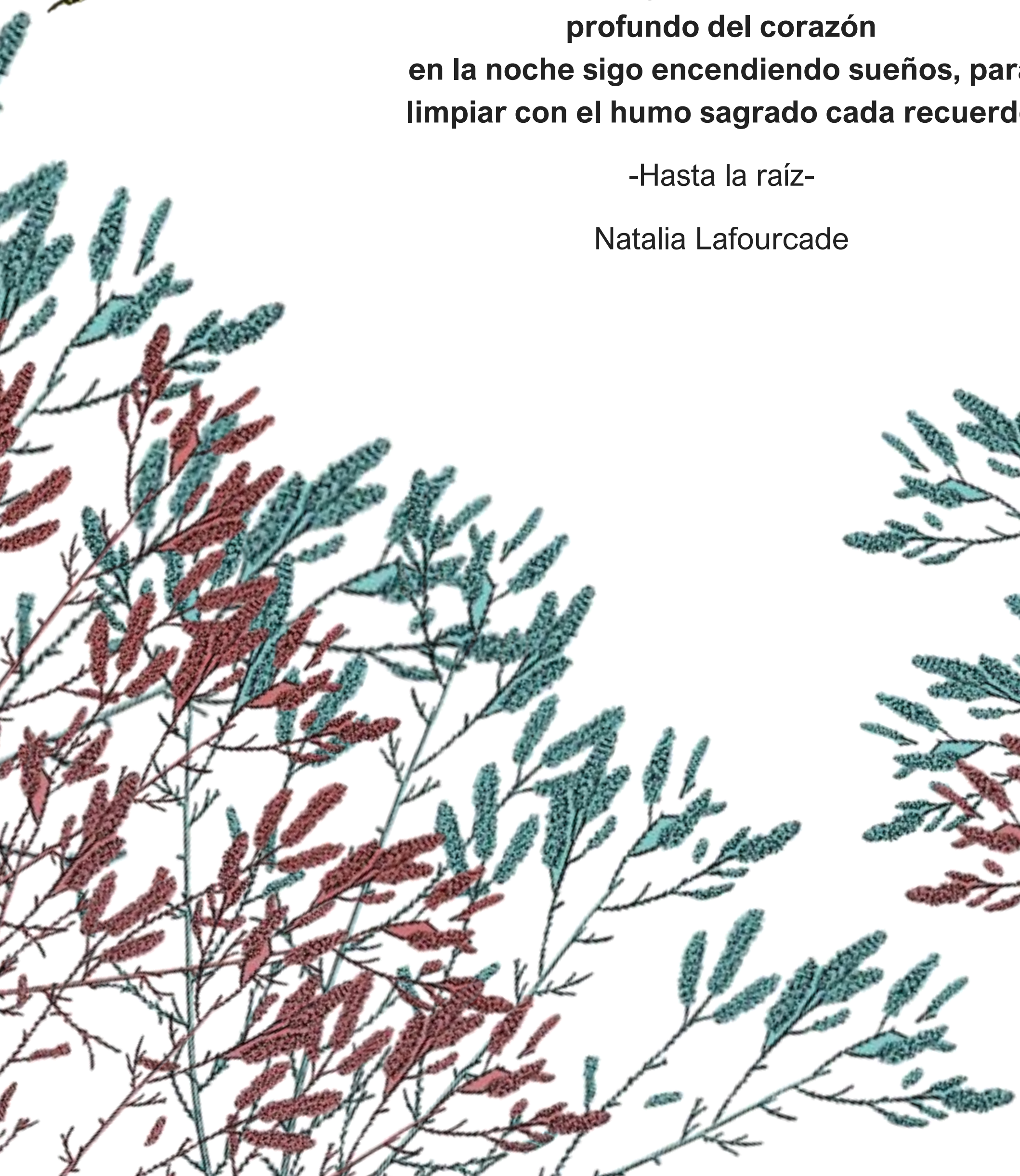


CHAPTER 3

Sigo cruzando ríos andando, selvas amando el sol, cada día sigo sacando espinas, de lo profundo del corazón
en la noche sigo encendiendo sueños, para limpiar con el humo sagrado cada recuerdo

-Hasta la raíz-

Natalia Lafourcade



The influence of riparian corridor land use on the Pesquería River's macroinvertebrate community (N.E. Mexico)

The Earth's freshwater ecosystems are currently under threat, particularly in developing countries. In Mexico, intensive land use and inadequate monitoring policies have resulted in the severe degradation of the country's freshwater ecosystems. This study assesses how the macroinvertebrate communities in the Pesquería River, located in Northeastern Mexico, are affected by riparian land use, in order to determine their potential use as bioindicators to evaluate the macroinvertebrate integrity of the Pesquería River. First, we characterized the land use cover in the riparian channel. Second, we sampled 16 sites for benthic macroinvertebrates along the main channel during the wet and dry seasons. Third, we evaluated the influence of the riparian channel land use on the macroinvertebrate community using 42 different biological metrics. The land use characterization depicted a riparian channel mainly influenced by agricultural and urban land use. Eighty-one invertebrate taxa were identified during the study. Permutational analysis of the variance analysis confirmed significant differences across the different land use classes and the macroinvertebrate community composition while no differences were found between seasons. The indicator species analysis revealed 31 representative taxa for natural land use, 1 for urban, and 4 for agricultural land use. Our modelling analysis showed that 28 of the 42 biological metrics tested responded significantly to land use disturbances, confirming the impact of land use changes on the Pesquería River's macroinvertebrate communities and suggesting that these metrics may have a use as bioindicators. Finally, this study may provide significant biological information for further studies in similar conditions.

INTRODUCTION

Nowadays, the Earth's ecosystems and biodiversity are changing at an accelerated rate due to population growth and human activities (Burdon et al., 2019; Price et al., 2019; Sala et al., 2000). Livestock and agriculture occupy more than a third of the world’s land surface and globally nearly 75% of freshwater is used for crop and cattle production (IPBES, 2019). Additionally, the total number of urban areas has more than doubled in recent years. It is projected that by 2050 there will be 25 million kilometers of paved roads (IPBES, 2019). The increase in agricultural and urban land use also has led to the fragmentation of the natural lands, an increment of nutrients load due to fertilization, and altered landscape hydrology (Krynak & Yates, 2018; Malmqvist & Rundle, 2002; McLaughlin, Kaplan, & Cohen, 2014; Walsh et al., 2005).

The scientific community has recognized that anthropogenic activities at landscape level gravely affect the natural conditions of freshwater ecosystems (IPBES, 2019). Moreover, several studies have shown that the transformation of natural areas into agricultural and/or urban areas can influence in-stream habitats, and affect the structure and composition of aquatic ecosystems (Death, Baillie, & Fransen, 2003; Stepenuck, Crunkilton, & Wang, 2002; Thompson & Townsend, 2004). It is well known that forest loss results in a low level of assimilation of nonpoint source pollutants in riparian ecosystems (Lowrance, Leonard, & Sheridan, 1985). Agricultural practices, for example, introduce nutrients into streams, which can damage the benthic habitat (Allan, 2004; Krynak & Yates, 2018). Further, the increase in the number of impervious surfaces and wastewater treatment plants due to urbanization results in altered peak flows and the introduction of nutrients and pollutants that alter a riparian channel’s shape and water quality (Hollis, 1975; Pizzuto, Hession, & McBride, 2000; Walsh et al., 2005).

Biomonitoring is the use of biological indicators to evaluate the health of the environment (Gerhardt, 2000). Biological indicators are crucial for assessing the condition of freshwater ecosystems due to their sensitivity to low levels of anthropogenic stress (Rosenberg & Resh, 1993). They are also suitable for testing the effects of pollution in experimental studies (Bonada et al., 2006). Benthic macroinvertebrate communities are the organisms most commonly used as a biomonitoring tool (Bonada et al., 2006; Prat et al., 2009). They can indicate a specific ecosystem’s health by highlighting changes in the diversity and species composition (Allan, 2004; Prat et al., 2009; Villamarín et al., 2013). Furthermore, some species have long life cycles, which can be used to trace the effects of pollution over longer periods, and their sensitivity to land use changes have made them a useful tool for biomonitoring programs (Barbour et al., 1996; Bonada et al., 2006; Fierro, Arismendi et al., 2018; Prat et al., 2009).

Biological metrics are parameters that are calculated to represent some of the aspects of the structure and function of the biological community in a given ecosystem (Barbour & Yoder, 2000; Bonada et al., 2006). Biological metrics vary according to anthropogenic impacts. The term "Multimetric indexes" has come to be used to refer to the use of combined biological metrics, which can offer a more detailed picture of the response to anthropogenic impacts (Resh, Norris, & Barbour, 1995). Multimetric indexes that assess the macroinvertebrate benthic community are considered one of the most effective methods for evaluating the biological condition of freshwater ecosystems (Fierro et al., 2018; Karr & Chu, 1997; Villamarín et al., 2013). The first phase of multimetric index development consists of selecting a set of biological metrics that show a response to multiple stressors. For instance, it is well documented that rivers impacted by agricultural land use present a loss in their riparian vegetation. This increases the availability of nutrients, which results in greater algal and periphyton production, and also changes the benthic assemblage composition (Allan, 2004; Heino, 2013). In addition, the increase in agricultural land use has also been change benthic communities dominated by shredders to one of grazers (Delong & Brusven, 1998; Krynak & Yates, 2018; Rabení, Doisy, & Zweig, 2005). Further, rivers affected by urban land use present an increase in the amount and variety of pollutants in runoff, plus an erratic hydrology due to the increased impervious surface areas, and an increment increase in water temperatures due to the loss of riparian vegetation. Heightened imperviousness in river areas increase the proportions of collectors-gatherers, while proportions of filterers, scrapers, and shredders decrease (Allan, 2004; Krynak & Yates, 2018; Stepenuck et al., 2002). Macroinvertebrate functional feeding groups exposed to impervious surfaces show a decrease in the assemblage of scrapers because of the low availability of food (i.e. periphyton). However, the availability of fine particulate food resources increases the presence of collectors/gatherers in the stream (Stepenuck et al., 2002). The different stress factors affecting freshwater ecosystems concurrently result in different biological responses, which also vary in each ecoregion. The importance of the creation and intercalibration of different indexes for each major ecoregion is, therefore, paramount (Herman & Nejadhashemi, 2015; Mathuriau et al., 2011).

Since 1974, the Mexican National Water Commission (CONAGUA) has monitored freshwater ecosystems using only physicochemical parameters. The development of biological metrics using macroinvertebrate communities and other biological groups is thus still needed to better assess the ecological status of Mexican freshwater ecosystems. This lack of evaluation can be explained by a combination of different factors: (i) the lack of taxonomic and ecological information regarding several key benthic macroinvertebrate taxa, (ii) the difficulty of finding riparian ecosystems without evidence of anthropogenic pressures that can act as reference sites (Mathuriau et al., 2011), and (iii) the fact that biomonitoring techniques are not included in the approved water evaluation methods outlined by the Mexican government (Arriaga-Cabrera et al., 2000). In light of this situation, the scientific community in Mexico has developed studies using benthic macroinvertebrates as bioindicators (Mathuriau et al., 2011). However, just a few have applied a multimetric approach in the evaluation of the ecological state of Mexican rivers (Ruiz-Picos et al., 2017; Ruiz-Picos et al., 2016; Weigel et al., 2002). Regarding the geographical coverage of these studies, just the central part of the country has been evaluated; there are no studies reported of ecological evaluation using benthic macroinvertebrates from the North and South of Mexico (Mathuriau et al., 2011). According to the Mexican Priority Hydrological Regions (PHR) program (Arriaga-Cabrera et al., 2000), the Pesquería River Sub-basin (PRSB) is a priority case for ecological evaluation given the high level of environmental degradation that has been caused by anthropogenic activity in the surrounding region.

Within this context, the overarching goal of this study was to test how the macroinvertebrate communities in the PRSB are affected by different land use categories, and to assess their potential use as bioindicators to evaluate the PRSB's ecological status. In order to do so, we first characterized the land use of the Pesquería River's riparian buffer to establish the natural and anthropogenic influences on the river. Second, we determined the response of the macroinvertebrate communities to different land uses by examining their community assemblages and a set of biological metrics. Next, the macroinvertebrate communities in the PRSB were evaluated using the said biological information. Finally, we discuss the potential use of the most significant biological metrics as tools for evaluating the ecological status of the Pesquería River, and their possible applications in other ecohydrological contexts.

MATERIALS AND METHODS

Study area: The Pesquería River

This study was conducted in the Pesquería River (PRSB), located in Northeastern Mexico. The river flows through the states of Coahuila and Nuevo León (24-Bravo-Conchos Hydrological region). The PRSB has a catchment area of 5,255.56 km^2, and its main course is 288.22 km in length. The mainstem flows through the Monterrey city metropolitan area. The PRSB's annual average flow is 2.04 m^3/s (Castro-López, Guerra-Cobián, & Prat, 2019). Its mean elevation is 542 masl with an average gradient of 0.4%. The climate is semi-arid (García, 2004), with an average temperature of 20°-24°C (Fierro, 2015). The PRSB has a total annual rainfall of between 400-700 mm. The wet season is from May to October, while remaining months, November to April, are dry (INEGI, 2017). The predominant vegetation is sub-montane scrub with Mesquite vegetation typical of sandy deserts, plus halophilic vegetation (Arriaga-Cabrera et al., 2000). Citrus production, livestock, aquaculture and rainfed agriculture are the main economic activities along the PRSB (Arriaga-Cabrera et al., 2000; Torres-Barajas et al., 2018). The natural water conditions of the PRSB present characteristics consistent with loamy basins and an elevated concentration of salt in the water (mesohaline habitats), as well as vegetation and fauna typical of semiarid zones (Cañedo-Argüelles et al., 2013; Castro-López et al., 2019; Williams, 1987). In order to corroborate this, we measured salinity and conductivity at each sampling site.

Riparian buffer land use and sampling site characterization

We used the Geographical information from the digital map database of the INEGI (National Institute of Statistics and Geography of Mexico) to obtain the main land use data for the PRSB, considering its entire drainage area (INEGI, 2016, 2017). Using the protocol for the assessment of the hydro-morphological quality of rivers (HIDRI protocol, (Munné et al., 2006), land use cover in the PRSB area was grouped into three categories (Figure 1): natural, urban, and agricultural, in order to evaluate the quality of the riparian channel; Munné et al., 2006).

Next, 16 sampling sites (each 100 m long) were strategically selected along 180 km of the PRSB's main channel, which included the headwaters of the river and sites before or after human settlements or agricultural areas. Then, the land use proportion for the riparian area of each sampling site was calculated, taking into consideration a 50m (width) buffer zone along both sides of the channel upstream of each site (Figure 1) (Broadmeadow & Nisbet, 2004; Munné et al., 2006; Valera et al., 2019).

Furthermore, we measured the distance from headwaters (DFH) of each sampling site to see if the macroinvertebrate community was influenced by the land use or by the longitudinal natural variability of river conditions (i.e. the river continuum concept; Tomanova et al., 2007; Vannote et al., 1980).

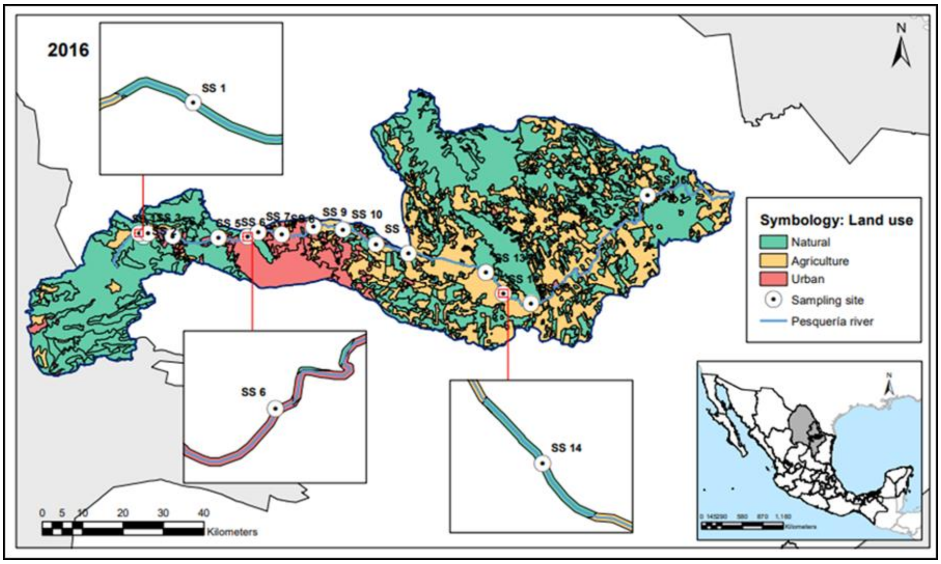


Figure 1 Illustration of the riparian buffer preparation.

The resulting riparian zone of each sampling site was classified as natural, agricultural or urban according to the following criteria (Munné et al., 2006): (i) Natural when more than 75% of the land use cover was unaltered, thus the sum of urban and agricultural land use does not exceed 25%; (ii) Agricultural when more than 50% of the land was used for farming, and (iii) and Urban when more than 50% of the land was used for urban areas/human settlement (Appendix C/Table S1). Arc Map 10.1 was used for this analysis (ESRI, 2012). To corroborate the land use typology previously proposed for each sampling site and to detect punctual impacts (i.e. wastewater sewages, industrial sewages, human settlements) aerial photographs were obtained via drone (Appendix C/Figure S1).

Macroinvertebrate sampling, taxonomic identification and biological metrics

We took samples of the benthic macroinvertebrate community at each sampling site during the wet season (August 2015) and dry season (February 2016). The multi-habitat kick-net sampling method was used (MAGRAMA, 2013). A D-frame net (0.5 m wide with 250-μm mesh) was used to collect a 1L sample per site, combining 20 kick samples (~0.5m2/ sample), which were taken in proportion with the microhabitat types (i.e. hard substrate, woody debris, aquatic macrophytes, vegetated bank margins and sand/ other fine sediment) present in each 100 m reach (Barbour & Yoder, 2000). The samples were preserved in 10% formaldehyde and transported to the laboratory (MAGRAMA, 2013). Once in the laboratory, we counted the macroinvertebrates and identified them at the lowest possible taxonomic level (Dominguez & Fernandez, 2009; Lanza-Espino & Pulido, 2000; Merritt et al., 2008; Tachet, 2010). Taxonomic resolution was primarily to the genus level, with some taxa identified to species level. Some Diptera and Hirudinae were identified to family level (e.g. Chironomidae, Erpobdebilldae), while Oligochaeta, Ostracoda, Cladocera, and Copepoda were identified at higher taxonomic levels.

We considered 42 candidate metrics generally used in studies of macroinvertebrate responses to anthropogenic pressures in Mexico and elsewhere (Edegbene et al., 2019; P. Fierro et al., 2018; Lu, et al., 2019; Macedo et al., 2016; Oliveira et al., 2019; Villamarín et al., 2013; Weigel et al., 2002; see Table 1). These metrics represented a range of structural and functional macroinvertebrate community characteristics, including diversity, species composition and trophic structure. Metrics of tolerance of pollution were not considered given the lack of precise taxonomic information for this region (Mathuriau et al., 2011). All these metrics were grouped into four different categories that relate to different attributes of the macroinvertebrate assemblages (Merritt et al., 2008): taxonomic richness, taxonomic composition, diversity indexes, and functional feeding groups' (FFG) metrics (Table 1). Regarding the FFG analysis, an "FFG x sampling sites" array was created (Lanza-Espino & Pulido, 2000; Merritt et al., 2008; Poff et al., 2006; Ramírez & Gutiérrez-Fonseca, 2014; Tachet, 2010; Tomanova, Goitia, & Helešic, 2006), where taxa were substituted by the FFG they belong. Then, the number of different taxa (richness) and the relative abundance (percentage) of each FFG was calculated for each site (Merritt et al., 2008; Poff et al., 2006; Ramírez & Gutiérrez-Fonseca, 2014; Tachet, 2010; Tomanova et al., 2006).

Table 1 Biological metrics categories and their expected response to anthropogenic pressures (Edegbene et al., 2019; P. Fierro et al., 2018; Lu et al., 2019; Macedo et al., 2016; Oliveira et al., 2019; Villamarín et al., 2013).

BIOLOGICAL METRICS	Expected response to anthropogenic activities
Category I "Taxonomic richness "	
Richness	Decrease
Rarified Richness	Decrease
Richness OCH (odonata + coleoptera + heteroptera)	Decrease
Richness EPT (ephemeroptera + plecoptera + trichoptera)	Decrease
Richness EPT / (richness EPT + richness OCH)	Decrease
Richness Ephemeroptera	Decrease
Richness Trichoptera	Decrease
Richness Odonata	Decrease
Richness Coleoptera	Decrease
Richness Diptera	Decrease
Richness Diptera wihout chironomidae	Decrease
Richness Gasteropoda	Decrease
Richness non-insect taxa (amphipoda + copepoda + ostracoda + gateropoda + oligochaeta + hirudinea)	Decrease
Category II "Richness composition"	
% OCH (odonata + coleoptera + heteroptera)	Decrease
% EPT (ephemeroptera + plecoptera + trichoptera)	Decrease
% EPT / (% EPT + % OCH)	Decrease
% Ephemeroptera	Decrease
% Trichoptera	Decrease
% Odonata	Decrease
% Coleoptera	Decrease
% Diptera	Increase
% Diptera wihout chironomidae	Decrease
% Gasteropoda	Decrease
% Non-insect taxa (amphipoda + copepoda + ostracoda + gateropoda + oligochaeta + hirudinea)	Increase
% Baetidae / % EPT	Decrease
% Baetidae / % Ephemeroptera	Decrease
% Hydropsychidae / % EPT	Decrease
% Hydropsychidae / % Ephemeroptera	Decrease
% Chironomidae	Increase
% Oligochaeta	Increase
% Chironomidae + % Oligochaeta	Increase
Category III "Diversity indexes"	
Shannon's Diversity using base e	Decrease
Simpson's Diversity	Decrease
Pielou's Evenness	Decrease
Category IV "Functional feeding groups"	
Richness Collectors-Gatherers	Decrease
% Collectors-Gatherers	Increase
Richness Predators	Decrease
% Predators	Decrease
Richness Herbivores	Decrease
% Herbivores	Decrease
Richness Collectors-Filterers	Decrease
% Collectors-Filterers	Decrease

Data Analysis

We began by exploring the aquatic macroinvertebrate community structure in relation to land use categories (agricultural, natural and urban). To do so, we performed a non-metric multidimensional scaling (NMDS) based on square root relative abundances and Bray-Curtis distance. Then, following the sqrt-transformation of the macroinvertebrate relative abundance data, we used permutational multivariate analysis of variance (PERMANOVA, ‘adonis2’ function in R) on the Bray-Curtis distance matrix in order to test differences in macroinvertebrate community composition across types of land use and different seasons (Anderson, 2001; Oksanen J et al., 2012). To define the indicator taxa for each land use category, we used the indicator species analysis (IndVal) established by Dufrene and Legendre (Dufrêne & Legendre, 1997). This analysis generates an indicator value index (IV) for each taxon and land use category. The indicator calculation is based on specifity and fidelity. To perform these tests, we used the packages vegan, labdsv and ade 4 in R (Oksanen J et al., 2012; Roberts, 2019; Thioulouse et al., 2018).

To evaluate the response of the 42 biological metrics to land use we followed the protocol outlined by Feld et al., 2016. First, we checked any outliers in the data, the variable distributions (skewness) and the assumption of normality (Bartlett and Shapiro test). For the variables that did not fulfil the assumptions of normality and homoscedasticity, we transformed the original data using a square root transformation (Appendix D/Table S1). Second, to quantify the effects and significance of land use, season, DFH and their interactions with the biological metrics, we fit linear mixed models (LMMs) and linear models (LMs). Next, from these models, we selected the final model for each biological metric, choosing those with the greatest explanatory capacity and parsimony (i.e. lower Akaike Information Criterion values; (Akaike et al., 1973). For each LMM, we considered land use, season, DFH and their interactions as fixed effects and the sampling site as random effects. We validated the final model by visually checking any residuals for normality and homoscedasticity (Zuur et al., 2009). When the null hypothesis had been rejected, in order to explore the differences between each land use category we performed post hoc Tukey pairwise comparisons using the ‘multcomp’ package in R. To calculate the determination coefficients, we followed the methodology proposed by Nakagawa and Schielzeth, 2013.

Since one of our aims was to detect macroinvertebrate-based metrics that can potentially be used as bioindicators for land use disturbances, we selected the metrics that better responded to the different types of land use in the PRSB. Therefore, we selected those biological metrics in which the models' (LMM´s or LM´s) response to land use presented significant results ($p < 0.05$). Within all metrics that varied significantly according to land use disturbances, we highlighted those with higher determination coefficients and the most ecological relevance for the PRSB (Gutiérrez-Cánovas & Escribano-Ávila, 2019). All the statistical analyses were performed using the R statistical software (R Core Team., 2016), version 3.4.1, with the significance level set at $p < 0.05$ for all tests (Racine, 2012). The datasets used in this study are available in Appendix D/Table S2.

RESULTS

Land use Characterization in the PR.

Georeferenced tools and aerial photographs corroborated that the typology of land use for each sampling site was in line with our previous land use classification (natural, urban & agricultural). These tools also showed small strips of riparian forest/vegetation in impacted areas (Appendix C/Table S2). We also located the discharge of six wastewater treatment plants (WWTP), three water stabilization ponds (WSP), clandestine garbage dumps (CD) and the presence of salt from indirect industrial waste (Appendix C/Table S3). The average salinity value for natural land use was 0.90 (± 0.02) ppt, while it was 2.50 (± 0.54) ppt for urban land use and 1.13 (± 0.06) ppt for agricultural land use. With regards conductivity, the average value for natural land use was 1758 (± 44) µS/cm, 5272 (± 1058) µS/cm for urban land use and 2187 (± 116) µS/cm for agricultural land use. In terms of riparian zone land use, three sampling sites were classified as natural (sites 1 to 3), six as urban (sampling sites 4 to 9), and seven as agricultural land use (sites 10 to 16; Appendix C/Table S3).

Aquatic macroinvertebrate community structure and land use

Eighty-one different macroinvertebrate taxa were identified in the survey. Taxa richness of the PRSB sites ranged between 5 and 38 taxa. The average taxa richness (mean ± SEM values) for natural sites was 35.2 (± 1.04), while it was 16 (± 2.06) for urban sites and 11 (± 1.23) for agricultural sites (Appendix D/Table S2). Biological communities are grouped on the NMDS bidimensional ordination according to the land use classification (Figure 3).

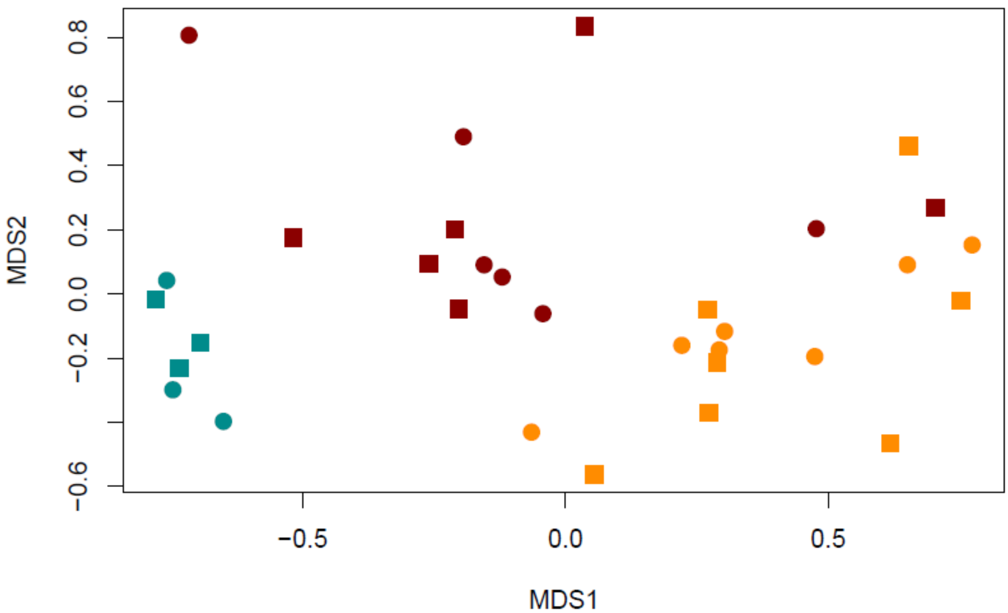


Figure 3 The Pesquería River non-metric multidimensional scaling (NMDS) based on square root relative abundances and Bray-Curtis distance. (NMDS stress value = 0.13). All the sampling sites represented by "square" are from the wet season sampling campaign and "circle" is used for the dry season sampling campaign. Each land use class is represented, with natural in blue, urban in dark red and agricultural in orange.

The PERMANOVA analysis confirmed that there were significant differences among macroinvertebrate communities for each land use category (Adonis, $F = 8.63$, $R2 = 0.38$, $p = 0.001$), while no differences were found between seasons (Adonis, $F = 0.95$, $R2 = 0.02$, $p = 0.43$). According to the IndVal analysis, 36 of the 81 taxa were indicators of one of the three classes of land use: 31 were indicators of natural land use, one taxon of urban land use, and four taxa of agricultural land use (Table 2).

Table 2 Results of the indicator species analysis (IndVal), maximum IV significance (IV is the individual value), associated to land use class for each species (NAT for Natural, URB for Urban and AGR for agricultural land use)

ORDER	TAXA	CLASS	IV	p-val
Odonata	*Ophiogomphus*	**NAT**	1	**0.001**
Trichoptera	*Oxyethira*	**NAT**	1	**0.001**
Hemiptera	*Ambrysus*	**NAT**	1	**0.001**
Ephemeroptera	*Baetis*	**NAT**	0.9	**0.001**
Diptera	*Atrichpogon*	**NAT**	0.8	**0.001**
Coleoptera	*Macrelmis*	**NAT**	0.8	**0.001**
Hemiptera	*Rhagovelia*	**NAT**	0.8	**0.001**
Trichoptera	*Agryalea*	**NAT**	0.7	**0.007**
Trichoptera	*Chimarra*	**NAT**	0.7	**0.002**
Coleoptera	*Psephenus*	**NAT**	0.7	**0.002**
Ephemeroptera	*Farrodes*	**NAT**	0.7	**0.004**
Ephemeroptera	*Tricorythodes*	**NAT**	0.7	**0.001**
Trichoptera	*Mayatrichia*	**NAT**	0.7	**0.002**
Megaloptera	*Corydalus*	**NAT**	0.7	**0.001**
Diptera	*Limoniia*	**NAT**	0.7	**0.002**
Diptera	*Stratiomys*	**NAT**	0.7	**0.001**
Diptera	*Ceratopogon*	**NAT**	0.6	**0.011**
Odonata	*Nehalennia*	**NAT**	0.6	**0.028**
Lepidoptera	*Petrophila*	**NAT**	0.6	**0.009**
Ephemeroptera	*Caenis*	**NAT**	0.6	**0.002**
Gasteropoda	*M. tuberculata*	**NAT**	0.5	**0.025**
Ephemeroptera	*Camelobaetis*	**NAT**	0.5	**0.003**
Coleoptera	*Cymbiodita*	**NAT**	0.5	**0.007**
Diptera	*Hemerodromia*	**NAT**	0.5	**0.008**
Odonata	*Macrothemis*	**NAT**	0.5	**0.007**
Coleoptera	*Lutrochus*	**NAT**	0.5	**0.013**
Ephemeroptera	*Callibaetis*	**NAT**	0.5	**0.007**
Trichoptera	*Leucotrichia*	**NAT**	0.4	**0.012**
Coleoptera	*Paracymus*	**NAT**	0.3	**0.035**
Diptera	*A. fransiscanus*	**NAT**	0.3	**0.039**
Diptera	*Euparhypus*	**NAT**	0.3	**0.023**

ORDER	TAXA CLASS	IV	p-val	ORDER
Coleoptera	*Berosus*	**URB**	0.8	**0.001**
Amphipoda	*Hyallela azteca*	**AGR**	0.7	**0.003**
Hirudinea	*Erpobdellidae*	**AGR**	0.6	**0.002**
Diptera	*Chironomus*	**AGR**	0.5	**0.012**
Odonata	*Ischnura*	**AGR**	0.4	**0.029**

Biological metrics and land use

Forty-two biological metrics were tested in this study. Of the final models (LMMs and LMs), 39 models had land use as the only explanatory variable (fixed effects), two models presented both land use and DFH as fixed factors (% EPT and % collectors-filterers) and just one model (richness collectors-gatherers) responded to land use, DFH and showed an interaction between land use and season (see Appendix D/Table S2). Thus, we discarded those models that responded significantly to DFH, keeping only those with biological metrics that responded to the different categories of land use (i.e. 28 metrics).

Within the metrics of taxonomic richness, richness (LMM, R2c=0.7, R2m=0.94, p<0.001) decreased in areas of urban and agricultural land use (55% and 68% respectively). The EPT richness was dominated by Trichoptera and Ephemeroptera richness given the lack of Plecoptera taxa in all Pesquería River sampling sites. The EPT richness index (LMM, R2c=0.7, R2m=0.93, p<0.001) showed a decrease in Urban (79%) and Agricultural (87%) land use in comparison with Natural land use. For OCH richness (LMM, R2c=0.66, R2m=0.89, p=0.0015) we found the same pattern, there was a 54% reduction in areas of urban land use and 76% in agricultural areas when compared with natural land use areas. Finally, for Diptera richness, (LMM, R2c=0.62, R2m=0.94, p=0.003) the same pattern was observed. For Chironomidae plus Oligochaete abundance (LM, R2ad=0.34, p=0.007), a metric of taxonomic composition, we found the opposite pattern: in the sites with urban or agricultural land use, there was an increase in abundance of 126% at urban sites and 137% at agricultural sites. All of the diversity indexes (third category) were significant, and show decreases in the presence of agricultural and urban land use. For instance, the Shannon Index base e (hereafter Shannon Index; LM, R2ad =0.66, p<0.001) was 49% lower for urban land use areas and 60% lower for agricultural land use areas when compared with natural land use. In the 1st, 2nd and 3rd biological metric categories' post hoc Tukey pairwise comparisons (Tukey HSD, p<0.05), we observed significant differences between natural and anthropogenic (urban and agricultural) land use covers. Regarding the functional feeding metrics, sites with agricultural and urban land uses presented a higher percentage of collector-gatherers (LM, R2ad =0.42, p=0.0011), compared with natural land use sites (Tukey HSD, p<0.05). Predator richness (LMM, R2c=0.57, R2m=0.85, p<0.001) decreased by 50% in sites with urban land use and by 68% in areas of agricultural land use when compared with natural land use sites (Tukey HSD, p<0.05). Predator abundance showed a similar trend (LM, R2ad=0.24, p=0.006), presenting significant differences between natural and agricultural land use (Tukey HSD, p<0.05). Herbivore species also decreased in richness (LMM, R2c=0.59, R2m=0.88, p=0.007) and in relative abundance (LM, R2ad =0.64, p<0.001) wherever anthropogenic land use had increased. Herbivore richness presented a reduction of 60% at urban sites and 83% at agricultural sites when compared to natural sites. The relative abundance of herbivores decreased by 83% in the presence of urban land use and 97% in the presence of agricultural land use when compared with nautral land use, being the only biological metric that differed across the three types of land use (Tukey HSD, p<0.05).

DISCUSSION

Our study revealed how macroinvertebrate communities within the Pesquería river mainstem respond to different land uses along the riparian channel. It showed not only how the health of the river is affected by human land use (both agricultural and urban) but also that macroinvertebrate communities may have a promising application as bioindicators to be used in future monitoring programs for semi-arid rivers in the North of Mexico. As has been documented in most human-degraded ecosystems (Hillebrand, Bennett, & Cadotte, 2008; Johnson, 2013), the ecological communities in the PRSB seem to be decreasing in richness, diversity and evenness, with anthropogenic pressures affecting macroinvertebrate communities the most. Our measurements of salinity and conductivity showed that the PRSB headwaters show evidence of "primary and secondary salinization" in freshwater (Castro-López et al., 2019). In addition, urbanization and agricultural practices have also influenced the structure of the macroinvertebrate assemblage.

The macroinvertebrate communities found in the Pesquería River are characteristic of mesohaline habitats (Rutherford & Kefford, 2005; Velasco et al., 2006; Zinchenko & Golovatyuk, 2013). Despite the salt present at the headwaters of the PR, IndVal analysis revealed a large number of taxa that were indicators of the natural land use typology, which at reference sites are related to higher richness values (Fierro et al., 2017). Some of these taxa have been used as bioindicators of "healthy" reference sites sites (e.g. Philopotamidae, Leptophlebidae) in previous studies (Acosta et al., 2009; Junqueira & Campo, 1998; Roldán, G., 1999). The IndVal analysis also revealed that only Berosus sp. was characteristic of urban land use. Previous studies in Mexico reported Berosus sp. as an indicator of streams polluted with an excess of organic matter, also Berosus sp. is reported to be tolerant to salinity and total dissolved solids (Santiago-Fragoso & Sandoval-Manrique, 2001). For agricultural land use, all IndVal representative taxa have been reported as bioindicators of habitats typically impacted by this land use typology. This study is consistent with previous studies that found that Zygoptera and Hirudinea were the most abundant taxa in sites impacted by agricultural land use. The presence of Ischnura sp. in farming ditches, for example, has often been reported (Twisk, Noordervliet, & ter Keurs, 2000). Additionally, high abundances of Erpobdellidae were reported at sites influenced by agricultural land use (Koperski, 2005). Furthermore, high densities of chironomids and amphipods were found, which are characteristic of agricultural sites (Burcher & Benfield, 2006).

The biological metrics used in this study present similarities with the metrics used to develop multimetric indexes around the world (Edegbene et al., 2019; P. Fierro et al., 2018; Lu et al., 2019; Macedo et al., 2016; Oliveira et al., 2019; Villamarín et al., 2013; Weigel et al., 2002). We found that 28 of the 42 metrics tested responded significantly to land use disturbances and therefore can be potentially used as bioindicators. These metrics provided information about richness, abundance, diversity and the trophic structure of the PRSB's benthic macroinvertebrate community. For Taxonomic Richness metrics, Richness, EPT richness, OCH richness and Diptera richness were determined to be the metrics that explained the anthropological land use impacts on the PR. These metrics have the advantage of being simple to calculate and are suitable for all ecological systems, being particularly useful metrics for biological degradation (Roy et al., 2003). High richness values have been associated with the pristine conditions commonly found at river headwaters, while low values reflect anthropogenic impacts but also responses to natural saline systems (Cañedo-Argüelles et al., 2013; Delong & Brusven, 1998; Kefford et al., 2011). Other metrics, such as EPT richness, OCH richness and Diptera richness, take into account the tolerance of sensitive groups to anthropogenic disturbances and natural saline ecosystems (e.g. Plecoptera, Cañedo-Argüelles et al., 2013).

The EPT richness in our study is only composed of Ephemeroptera and Trichoptera species that are tolerant of salinity. It is well known that Ephemeroptera species are halophobes due to their lack of physiological tolerance to salt (Gallardo-Mayenco, 1994; Short, Black, & Birge, 1991). However, other authors have reported that Leptophlebiidae, Baetidae and Caenidae families are all tolerant of saline, this last taxon being the most tolerant of the three (Kay et al., 2005). Interestingly, Leptophlebiidae and Caenidae, both significant indicators of natural land use according to the IndVal analysis, have previously been used as biological indicators. Leptophlebiidae has received a high score on biological indexes, while Caenidae has received an average score (Acosta et al., 2009; Naranjo López et al., 2005; Roldán, G., 1999). For Trichoptera, it is known that increased levels of salt and chlorides directly affect species richness and the abundance and biomass of caddisfly larvae (Kholmogorova, N.V., 2009). Several studies around the world have demonstrated that some species of Trichoptera can live in saline waters (Kay et al., 2001; Piscart et al., 2011; Short et al., 1991). In addition, in our study, the Indval analysis revealed that Philopotamidae, which is considered an indicator of good ecological quality, was indicative of natural land use (Acosta et al., 2009; Figueroa et al., 2007; Naranjo López et al., 2005; Roldán, G., 1999). It is well documented that Plecoptera species require relatively undisturbed conditions, and they are therefore important bioindicators for freshwater quality (Stewart, 2009). The use of Plecoptera as a bioindicator is not particularly useful in saline streams or ponds, since it is normally absent. Even if this order contains tolerant species, be they rare or endemic, Plecoptera overall usually presents a low species richness in saline water bodies (Abellán et al., 2005; Kefford et al., 2011; Moreno, et al., 1996; Rutherford & Kefford, 2005; Sánchez-Fernández et al., 2006). The lack of Plecoptera in this study is probably due to the high salt concentration in the PRSB (Castro-López et al., 2019; Kefford et al., 2011; Kefford et al., 2016; Rutherford & Kefford, 2005).

Despite the natural concentrations of salt in the PRSB, many Odonata, Coleoptera and Heteroptera species are adapted to saline waters (Kefford, Papas, & Nugegoda, 2003). As we expected, OCH richness decreased in the presence of anthropogenic land use. IndVal analysis revealed species representative of OCH richness, which can be used as biological indicators in freshwater. For example, Gomphidae, which is representative of natural land use according to our IndVal analysis, has been reported as tolerant of saline water, as well as being a good biological indicator of high water quality. Conversely, Ischnura sp., which also has a high tolerance for salinity, was an indicator of agricultural land use, in line with previous studies (Ministerio del Ambiente y Energía, 2006; Velasco et al., 2006). Coleoptera presented similar circumstances. Pshephenidae was representative of natural land use according to the IndVal analysis, and has been previously used as bioindicator for good quality riparian ecosystems (Roldán, G., 1999). Heteroptera, Rhagovelia sp. and Ambrysus sp. were the only representative taxa present in our IndVal analysis for natural land use. Additionally, both have been reported as tolerant of saline environments (Davis, 1996; Kay et al., 2001; Piscart et al., 2011; Rutherford & Kefford, 2005). According to previous studies both taxa families have also been scored as tolerant of pollution (Alba Tercedor et al., 2002; Alvarez & P, 1983; Ríos-Touma et al., 2014; Roldán, G., 1999).

Previous research has indicated that Diptera richness decreases in response to anthropological land use (Fierro et al., 2018). Our IndVal analysis confirmed this observation: seven Diptera taxa were detected as indicators of natural land use, none were identified for urban land use and only one was found for agricultural land use. Nonetheless, several studies have reported that the abundance of some Diptera species increases when anthropogenic land use begins (Villamarín et al., 2013). This is directly linked to our second biological metric category, "Richness composition". For this category, the metrics selected were Chironomidae plus Oligochaete abundance, which is the percentage of Chironomidae plus the percentage of Oligochaete, both being representative of agricultural land use in our IndVal analysis. Several authors have described an overall increase in the relative abundance of Chironomidae and Oligochaeta in the presence of urban and agricultural land use (Dyer & Wang, 2002; Freeman & Schorr, 2004; Paul & Meyer, 2001). The PRSB sampling sites that recorded the most abundant taxa for Chironomidae and Oligochaetes (sampling sites 4, 9, 10, & 12) are all affected by the discharge from the WWTPs. It is well known that WWTPs in urban areas have "unhealthy" effects on benthic communities (Dyer & Wang, 2002). In addition, this study found that macroinvertebrate integrity is typically damaged in sites located immediately downstream of WWTPs.

For the diversity metrics, the Shannon Index (Shannon, 1948) was chosen since it is considered to be a robust tool that takes into account both evenness and species richness. The higher the values of the Shannon Index, the greater the diversity and the healthier the ecosystem (Serrano-Balderas, et al., 2016; Shannon, 1948). In our case, the Shannon Index showed healthy sites at the PRSB headwaters (sampling sites 1-3), while also singling out the most degraded sites (Appendix C/Figure S2). The most impacted sites for urban land use are located after the WWTP and WSP discharges, as well as after the possible indirect discharges from industry (sampling sites 4 & 5), and after the discharge of the city's principal WWTP (sampling site 9). Meanwhile, sites impacted by agricultural land use were located after the local WWTP and WSP (10,11 & 12 sampling sites; Appendix C/Figure S2).

Previous studies have linked macroinvertebrate-feeding ecology with environmental conditions, suggesting that the FFG's metrics detect stressors from multiple spatial scales (Cummins, 1974; Kerans & Karr, 1994; Wallace & Webster, 1996; Weigel et al., 2002). In addition, it has been observed that anthropogenic activities alter the availability of food sources, producing important variations in the distribution and relative abundance of FFG (Merritt et al., 2008; Serrano-Balderas et al., 2016; Weigel et al., 2002). The FFG metrics selected for the PRSB were collector-gatherers (percentage), predators (richness & percentage), and herbivores (richness & percentage). An increase in the percentage of collector-gatherers is well documented in areas of anthropogenic land use (Fierro et al., 2018). In our particular case, agricultural land use sampling sites showed the highest number of collector-gatherers. IndVal analysis for the PRSB revealed that collector-gatherer taxa like Chironomidae and Hyalella azteca sp. were representative of agricultural land use. Additionally, previous studies have remarked that the high availability of nutrients in streams impacted by agricultural land use leads to greater algal production and moss growth, which in turn provides a suitable habitat for both taxa (Delong & Brusven, 1998; Krynak & Yates, 2018).

Consequently, the increase in the number of collector-gatherers alters the functional composition of the benthic macroinvertebrate community, reducing the richness and abundance of other functional groups (predators, herbivores, etc.). Nevertheless, evidence of the abundance of predators in disturbed areas remains inconclusive. While some authors have found a high density of predators in forested areas, some have observed that predators occurred more often in urban areas, whereas others have found that predators have similar distributions across different land use typologies (Fu et al., 2016; Hepp & Santos, 2009; Miserendino & Masi, 2010; Miserendino & Pizzolon, 2004). In our case, the riparian forest/vegetation "strips" in sampling sites within urban land use areas served as buffers where predators can be found. Conversely, urban sites without these riparian forest /vegetation strips and sites highly impacted by WWTPs presented similar levels of abundance to agricultural land use sites with the same disturbances (Appendix C/Figure S2).

Finally, but importantly, herbivore abundance was the only biological metric in our study that presented significant differences across the three different typologies of land use. The percentage of herbivores is high at sites with natural land use in the PRSB area, but it declines drastically at urban sites, and almost disappears at sites with agricultural land use. Previous studies have shown that the abundance of herbivores was positively correlated with periphyton cover and the main substrate diameter, these aspects all being characteristic of undisturbed waters (Alvarez & P, 1983). Moreover, we found that distance from headwaters was only significant for 3 of the 42 metrics tested, demonstrating that the variability of macroinvertebrate communities in the PRSB observed in this study is linked to land use disturbances and not to the natural longitudinal variability of the river conditions (i.e. the river continuum concept). However, we should clarify that the current study was limited by the synergistic effect of the accumulated impacts caused by urban land use, especially between urban sites and agricultural sites. All the metrics selected highlighted the differences between natural and anthropic land uses (urban and agricultural) but just one was able to distinguish the differences between natural, urban and agricultural land use.

CONCLUSIONS AND CHALLENGES

The lack of watershed planning, management and restoration activities in Mexico have allowed unregulated anthropic activities to severely affect the ecohydrological health of the country's rivers. The biological metrics selected strongly support the notion that the PRSB ecosystem has been disturbed by the changes observed in the macroinvertebrate communities in response to the different riparian land uses. The PRSB diagnosis is in line with “urban river syndrome”, and it has the trademark characteristics of a river influenced by agricultural land use (“Agro-urban rivers”). The high rate of urbanization makes it particularly hard to find “healthy" reference sites in this region. Headwater streams in highly urbanized zones are therefore under constant threat. In spite of this, we were able to locate several riparian forest "strips" which help to mitigate the anthropogenic pressures affecting PRSB macroinvertebrate biodiversity. These small areas offer a great starting point for operations to recover the river’s ecological health. We also believe that the riparian zone border limit should be extended beyond the limits established in the "1992 Mexican National Water Law” (CONAGUA, 1992). The biological macroinvertebrate-based metrics selected for use in this paper are a promising tool that can be used as a first step in the construction of a Multimetric index for this river, and can hopefully be used as bioindicators of land use pressures in other semi-arid rivers in the North of Mexico. Finally, the PRSB is a clear example of how anthropogenic activities, a lack of regulation and a failure to safeguard river environments are threatening Mexican riparian ecosystems. Our findings have significant implications for watershed management and for the restoration of the PR, this being the first study that looks to propose biological metrics that reveal the influence of land use on the river ecosystem. Regardless of the advantages that biological metrics present, further work to improve biomonitoring efficiency in Mexico is needed. The development of taxonomic keys for Mexican fauna is urgently necessary in order to build suitable indexes for specific ecoregions.

It's in the water, it's in the story of where you came from

Your sons and daughters in all their glory it's gonna shape 'em

And when they pledge and come together, and start arising

Just drink the water where you came from, where you came from

-Radioactive-

Kings of Leon

GENERAL DISCUSSION & CONCLUSIONS

The overarching goal of this PhD thesis was to evaluate the ecological status of the Pesquería River and assess the effects of landscape degradation along its riparian channel. We focused on developing different Ecohydrological management tools for diagnosing the biological quality and the ecological integrity of the river as an ecosystem.

The present discussion aims to provide a general perspective on the current status of the Pesquería River through the different management tools interconnected by the riparian channel ecosystem. We will discuss the dynamics of the land use cover/change processes (Chapter 1), the role of the riparian vegetation in the evaluation of the Pesquería River's ecosystem health (Chapter 2), and the influence of the land use cover changes on the macroinvertebrate community (Chapter 3). Finally, we discuss the current status of Pesquería River and Mexican freshwater ecosystems and future challenges to their safeguarding.

LAND USE DYNAMICS IN THE PESQUERÍA RIVER BASIN AND CHANGES IN SPATIAL PATTERNS ALONG THE RIPARIAN CHANNEL.

Rapid, intense and continuous changes in land-use processes have changed the riparian landscape of the Pesquería River in the last few decades (1976-2016). Previous studies have evaluated changes in land use dynamics in river basins (Torres Barajas et al., 2018; Vela & Lozano, 2010; Vela & Lozano, 2007), but currently there is no research that evaluates these dynamics in the context of the riparian channel. In light of this, our research in **Chapter 1** contributed to the development of a method for the evaluation of land-use cover/change dynamics, the rate of change and the probability of permanence/change for the Pesquería River's riparian channel. The methodology developed in **Chapter 1** was based on the methodology for evaluating the "naturalness" (quality) of a riparian channel presented in the protocol for the assessment of the hydro-morphological quality of rivers (HIDRI protocol; Munné et al., 2006) and on the evaluation of land use cover/change dynamics detailed in (Castelán Vega et al., 2007; Zhang et al., 2017; Zhou, Wu, & Peng, 2012).

Our assessment showed that the ecological condition of the riparian channel has been considerably impacted by changes in land use over time. This was particularly perceivable in the high level of degradation of the riparian forest (natural vegetation). Within this context, we were able to describe how the Pesquería River's riparian forest has mainly become grassland and agricultural zones, whereas the secondary forest has been transformed into urban zones. Our results echo those of other studies, which have indicated the low probabilities of permanence for riparian forests and secondary forests in the riparian channel at both local and regional scales (Pérez-Miranda et al., 2012; Torres Barajas et al., 2018). Given that urban and agricultural settlements are well defined in the Pesquería River, both categories presented a high probability of permanence in the riparian zone.

Additionally, our findings are consistent with other research that has found that the loss pattern of shrublands is related to an increase in the area of grasslands and agricultural lands (Pérez-Miranda et al., 2012; Torres-Barajas et al., 2018). According to Vela & Lozano (2010), an increase of 10% in grassland area and 5% in agricultural land area in the northeast region of Mexico (where the Pesquería River is located) is expected by 2030. Population growth increases pressure on natural areas, which in turn degrades riparian ecosystems by converting the riparian forest into secondary forest. While the number of urban areas increased in size during the period studied, the number of agricultural areas did not compared with other land uses. This has most likely been caused by high levels of rural to urban immigration, which have led to the abandonment of agricultural land and its subsequent transformation into secondary vegetation (Díaz, 2003; García-Ruiz et al., 2013). This factor may explain why the secondary forest presented a high conversion rate, compared to agricultural land use. Nevertheless, these anthropogenic activities have led to the degradation of almost 39% of the riparian forest (natural vegetation) and 4% of the secondary forest (secondary vegetation).

Revegetation processes along the Pesquería River channel were divided into two stages: firstly, from anthropogenic land use to secondary forest (3%), due the lack of agricultural practices and the rural emigration trend in semi-arid regions. During the second stage, the revegetation rate was 1.2 % from anthropogenic land use (mainly grassland) that became into riparian forest, in line with prior studies undertaken in Mexico (Jaimes et al., 2003). Urban areas presented the lowest rate of revegetation. Although processes of revegetation are extremely uncommon in urban areas, a high number of flash flood events in recent years in the Monterrey metropolitan area have contributed to an increased rate of revegetation in the region (Aguilar-Barajas, et al., 2019; Fierro, 2015). These flash flood events have led to a decrease in anthropogenic land use along the banks of the Pesquería (i.e. urbanization, human settlements) and consequently have fostered processes of revegetation (Aguilar-Barajas et al., 2015; Aguilar-Barajas et al., 2019). Additionally, anthropogenic activities and natural disturbances (e.g. flash floods) generate changes in habitat distribution, assisting the spread and establishment of invasive species, reducing biodiversity, and hindering shrub restoration processes in semiarid areas. These changes damage the overall quality of the riparian forest (Bhattarai & Cronin, 2014; Calderon-Aguilera et al., 2012; Newton et al., 2009; Porensky et al., 2014). The presence of invasive species that negatively impact the Pesquería River's riparian forest quality was corroborated in our **second Chapter**.

Furthermore, the results in **Chapter 1** highlight the importance of correct legislation regarding riparian forest width. While previous studies have underscored the importance of riparian vegetation buffers, the 1992 "Mexican National Water Law" considers this zone to be a federal area and not an ecosystem (Almada et al., 2019; Chua, Wilson, Vink, & Flint, 2019; CONAGUA, 1992; Luke et al., 2019; Mendoza-Cariño et al., 2014). Thus, the implementation of sustainable management policies is hindered by the legislative gap generated by inadequate interpretation and application of this normative.

RIPARIAN FOREST QUALITY

One of the most useful tools used in the evaluation of riparian forest quality is the QBR index (Munné et al., 2003). This index has been successfully adapted around the world (Acosta et al., 2009; Carrasco et al., 2014; Colwell & Hix, 2008; Palma, Figueroa, & Ruiz, 2009; Sirombra & Mesa, 2012). **Chapter** 2 describes how the QBR methodology was adapted specifically for the semi-arid conditions present in the north of Mexico (QBR-RNMX). Moreover, this survey highlighted the importance of the characterization of the river's "natural conditions", i.e. the compilation of a vegetal inventory and water physiochemical characterization.

The results gained from evaluating riparian forest quality using the QBR-RNMX showed varying levels of riparian vegetation degradation. More than 80 km of the main reach of the Pesquería River crosses through the city of Monterrey, and is therefore subject to high levels of anthropogenic pressure. However, our results did highlight the presence of riparian strips that serve as buffers and lessen the impact of urban areas upon the ecosystem. Additionally, our results showed that the agricultural practices on the city´s outskirts put less pressure on the riparian ecosystem than the urban areas, and that the majority of the sites sampled in this area had farmland on only one bank of the river. In most cases, the opposite bank still had vegetation and riparian buffer strips. Furthermore, we were also able to locate sampling sites in reference conditions at the headwaters of the Pesquería.

As previously mentioned in **Chapter 1**, invasive species are one of the most pressing problems for Mexican riparian ecosystems. We decided to examine this aspect in detail in **Chapter 2**. The Pesquería River vegetal inventory revealed that nearly 50% of the tree and shrub taxa were invasive. This is one of the most prohibitive factors for improving riparian zone conditions. Additionally, throughout the investigation we detected that *Arundo donax* (giant cane), *Ricinus communis* (castor bean), and *Tamarix aphylla* (salt cedar) were the predominant species in the Pesquería River basin. These three invasive species have historically caused considerable damage to Mexican water resources. This is due to the fact that they consume large quantities of water which, in turn, increases the salinity of the water by concentration and increases hydrological stress in semiarid regions (I.M.T.A., 2007).

Another important aspect to emphasize was the correlation of the substrate composition and water salinity with the QBR-RNMX values. As previous studies have remarked, these aspects are two of the main factors influencing riparian vegetation and aquatic river ecosystems (Moreno et al., 1996; Suárez & Vidal-Abarca, 2000; Vidal- Abarca et al. 2004). The natural characteristics of the substrate plus the anthropogenic impacts resulting from urbanization (e.g. the dumping of industrial waste and discharge from wastewater treatment plants) increase the level of salinity in the river, which most likely explains the low number of tree and shrub species. These results are enhanced by a significant negative relationship, a Pearson´s correlation between evaluation categories I, II and IV (Total riparian cover, cover structure and "naturalness" of the riparian channel) and physicochemical parameters (Conductivity, Salinity, Total Dissolved Solids, Cl^-, Na^+), which highlight the importance of salt in the composition and structure of the riparian vegetation. Moreover, the high conductivity level of the water explains the relatively poor biodiversity of the riparian vegetation on the banks. Concurrently, it provides an example of how the QBR can accurately describe the degradation of riparian areas even in conditions that do not favour riparian vegetation.

Our QBR-RNMX adaptation also pioneered the use of drones for the evaluation of the riparian forest quality. Previous studies have pointed to the potential use of drones in ecological monitoring. Drone technology can provide a low-cost and low-impact solution for environmental managers working in a variety of settings (Ivosevic et al., 2015; Zhang et al., 2016). In our particular case, drone technology allowed us to establish riparian forest composition, river connectivity, distribution and the amount (%) of vegetation coverage along the Pesquería River. Moreover, drone technology was key in determining the geomorphological type of the Pesquería River which is necessary for the quality cover assessment category of the QBR-RNMX. Finally, the use of this technology proved to be extremely practical for the evaluation of anthropogenic impacts on the riparian zone.

RIPARIAN CHANNEL LAND USE AND ITS INFLUENCE ON THE MACROINVERTEBRATE COMMUNITY

Benthic macroinvertebrate communities are the organisms most commonly used as biomonitoring tools (Bonada et al., 2006; Prat, et al., 2009). Nevertheless, few Mexican freshwater ecosystems have been evaluated using these bioindicators, particularly in the northern and southern areas of the country. Thus**, Chapter 3** investigated how macroinvertebrate communities in the Pesquería River are affected by different land use categories, and assessed their potential use as bioindicators to evaluate the Pesquería River's ecological status. As has been observed in highly degraded ecosystems (Hillebrand, Bennet & Cadotte 2008; Johnson, 2013) ecological communities in the Pesquería River seem to be decreasing in richness, diversity and their distribution is becoming less even, with anthropogenic pressures directly affecting macroinvertebrate communities the most.

The land use characterization yielded three categories: natural land use, urban land use, and agricultural land use. Further, six wastewater treatment plants, three water stabilization ponds, several clandestine garbage dumps and the presence of salt from indirect industrial waste disposal in the Pesquería River were detected using drone technology. Aerial photographs obtained via drone revealed the presence of riparian buffer strips at urban and agricultural sampling sites.

The results in **Chapter 3** also showed that macroinvertebrate communities have a potentially promising usage as bioindicators in future monitoring programs for semi-arid rivers in the north of Mexico. The macroinvertebrate composition of the natural land use cover was mostly influenced by the natural salinization of the Pesquería River previously corroborated in **Chapter two**. Additionally, urbanization and agricultural practices have also influenced the structure of the macroinvertebrate communities. The macroinvertebrate composition was characteristic of mesohaline habitats (Rutherford & Kefford, 2005; Velasco et al., 2006; Zinchenko & Golovatyuk, 2013). Despite the salt present at the headwaters of the Pesquería River, our indicator species analysis (IndVal) revealed a high number of taxa representative of the natural land use typology (31 taxa), which at sites with reference conditions are representative of high richness values (Fierro et al., 2017). This analysis reported only one taxon representative of urban land use, whereas four taxa representative of agricultural land use were found.

The biological indicators used in this study present similarities with the metrics used to develop multimetric indexes (Edegbene et al., 2019; P. Fierro et al., 2018; Lu et al., 2019; Macedo et al., 2016; Oliveira et al., 2019; Villamarín et al., 2013; Weigel et al., 2002). We found that 28 of the 42 metrics tested showed a significant response to land use disturbances and therefore can potentially be used as bioindicators. We then highlighted the metrics (11) with the highest determination coefficients and the most ecological relevance for the Pesquería River. These eleven metrics provided important information about richness, abundance, diversity, and the trophic structure of the Pesquería River benthic macroinvertebrate community.

Finally, **Chapter 3** concluded that the health of the Pesquería River's ecosystem is directly affected by the different land use covers (i.e. natural, urban and agricultural). Still, we were able to locate several riparian forest "strips" which help to mitigate the anthropogenic pressures affecting the Pesquería River's macroinvertebrate biodiversity. These small areas offer a great starting point for operations to recover the river's ecological health. Moreover, we found that distance from headwaters was only significant for 3 of the 42 metrics tested, demonstrating that the variability of macroinvertebrate communities in the Pesquería River observed in this study is linked to land use disturbances and not to the natural longitudinal variability of the river conditions (i.e., the river continuum concept; Vannote et al.,1980). However, we should clarify that the current study was limited by the synergistic effect of the accumulated impacts caused by urban land use, especially between urban and agricultural sites. All of the metrics selected highlighted the differences between natural and anthropic (urban and agricultural) land uses, but just one metric (i.e. herbivores' percentage) was able to distinguish the differences between natural, urban, and agricultural land use.

CIUTATS AGERMANADES, RIUS AGERMANATS? THE LLOBREGAT AND THE PESQUERÍA RIVERS

This PhD thesis provides strong evidence that the declining ecological status of the Pesquería River has led to changes in the landscape of its riparian channel. The methodologies used in each chapter of this thesis proved to be useful and gave us new, reliable tools for the evaluation of the Pesquería River's ecological conditions. Furthermore, the research methodologies employed were mainly based on the Catalonian research and ecohydrological management tools that have been developed over the last four decades. The success of the diagnosis, monitoring and restoration programs implemented for Catalonian rivers, especially those targeting the Llobregat River, inspired us to adapt and create the different tools described in this PhD thesis. Interestingly, the Llobregat and the Pesquería had something of a shared history before this project even began, that we briefly summarize in the following paragraphs.

The term "*agermanat*" is a Catalan word meaning, "twinned". A foreign policy innovation, it was proposed in the 70´s, to create international links between Catalonia and other cities around the world. Since 1977, Barcelona (N.E. Spain) and Monterrey (N.E. Mexico) have been "*ciutats agermanade*s", or twinned cities (Ajuntament de Barcelona, 2019). Both cities share cultural, urban, industrial and agricultural characteristics, and furthermore, both have rivers flowing through their metropolitan areas that are subject to anthropogenic pressures. In spite of their similarities, the ecohydrological management responses in both cases have been dramatically different. The Llobregat river management programs provide an important example of how to improve ecosystem health, whereas a lack of research and management in Mexico has led to a decline in the ecological status of the Pesquería River.

As previously mentioned, the Llobregat River water quality, and damage to its ecosystem have been monitored for a considerable period by the water authorities (Catalan Water Agency) and research centres in Catalonia (Freshwater Ecology Hydrology and Management research group, University of Barcelona). The river has been severely impacted by anthropogenic activities for hundreds of years. The mid-section of the river flows through an important concentration of industrial sites, agricultural land and urban areas of Catalonia, while its headwaters are mostly affected by hydropower water diversion and low connectivity because of the presence of a large number of weirs (Prat & Ward, 1992; Sabater & Muñoz, 2014). While the evaluation of biological water quality from 1994 to 2010 evidenced a continued improvement in the quality of the Llobregat´s headwaters, the mid- and low-sections of the river have continually been classified as poor quality. In order to describe this data visually, we selected three different quality monitoring sites that have been monitored over the last few decades.

IBMWP scores and species richness data is available from 1994 onwards, whereas QBR scores have been recorded since 1998, when the index was created (Figure 1).

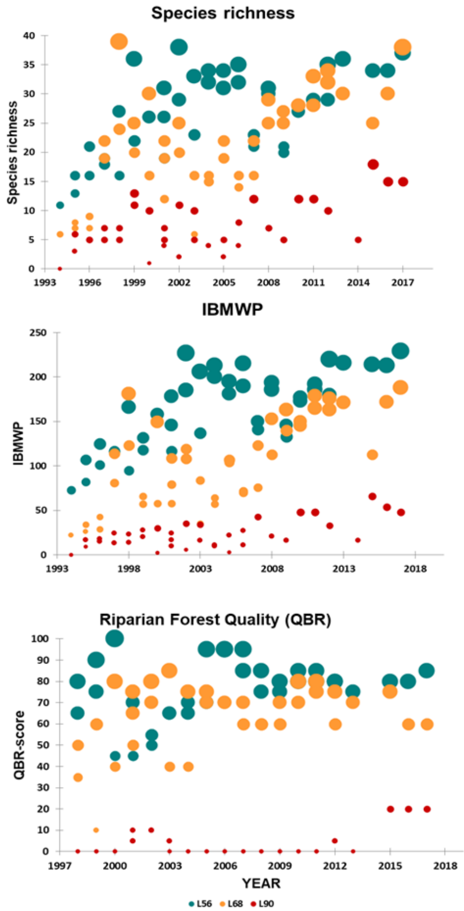


Figure 1 Graph showing macroinvertebrate species richness (S), biological quality (IBMWP), and riparian forest quality (QBR) for three sampling sites with different ecological statuses (good, fair, poor) over a period of 30 years for benthic macroinvertebrates, and 20 years for riparian forest quality. The headwaters of the Llobregat River are shown in blue (sampling station L56), the mid-section in orange (sampling station L68) and the lower section in red (sampling station L90).

The extensive and continued study of the Llobregat River offers a key example of the remarkable ecohydrological management of Catalonian Rivers (Munné et al., 2012).

The lack of environmental data for several sites makes impossible to complete the process with the BMWP index, for this reason we provide a preliminary version of the BMWP that may be applied to the Pesquería River. We have design a version of the BMWP to Pesquería River following the recommendations of Prat & Munné (2014).

1. We selected the three sampling sites located at heaadwaters (i.e. six samples), as reference stations.
2. We adapted the scores of the families of macroinvertebrates to the conditions of the Pesquería River following previous studies (Acosta et al., 2009; Ríos-Touma, Acosta, & Prat, 2014)
3. With the six samples obtained from the reference sites, we calculated the limit between very good and good ecological status, using the 25 % percentile of the variance (i.e. score: 121).
4. Of the resulting value, we substract the lowest score found in the Pesquería River through the BMWP index (i.e. 121-3: 118).
5. We divided the previous value by 4 (i.e.118/4) to define the limits between good moderate, moderate-poor and poor-very poor (following the WFD recommendations).
6. The quality values of each site, were calculated

The current ecological status of the Pesquería River is comparable to that of the Llobregat River around 40 years ago and is represented in Figure 2. Most of the sites are in very poor or poor conditions, which reflects the bad ecological status of the Pesquería River.

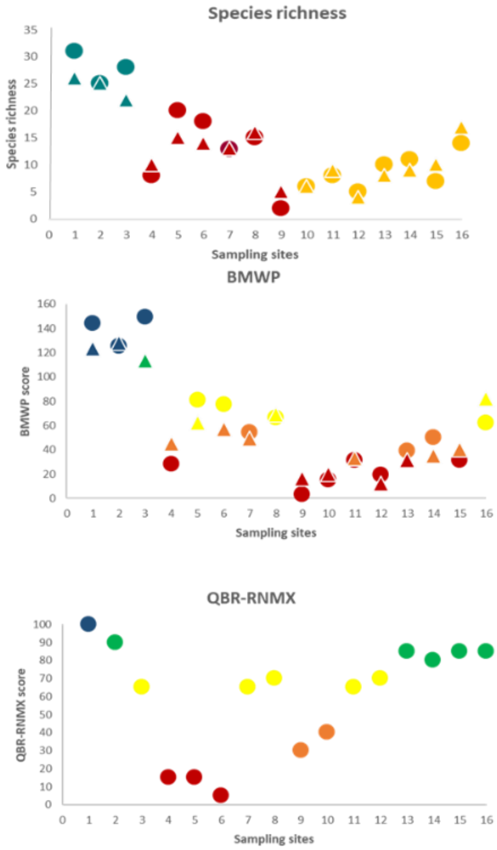


Figure 2 Graph showing macroinvertebrate species richness, biological quality (IBMWP), and riparian forest quality (QBR). Circles represent the sampling campaign conducted in August 2015, and triangles represent the sampling campaign conducted in February 2016. For species richness sites of good ecological quality (natural land use) are shown in blue, whereas sampling sites in red represent urban land use, and sites in orange represent agricultural land use. Both red and orange are indicative of poor ecological quality for the different scores. BMWP values of ecological quality are represented in colours (Blue-very good, green-good, yellow-moderate, orange-bad and in red-very bad). BMWP scores were obtained through an adapted version to Pesquería River of this index. QBR-RNMX values of riparian forest quality are in colours (Blue-natural conditions, green good, yellow-average, orange-bad and in red very bad conditions.

The results of our study show a river that has been impacted considerably by the different anthropogenic land-use processes recorded. By comparing the Pesquería with the Llobregat, we are able to emphasise the efforts need for a Mexican national water framework normative, similar to WFD, where the ecological quality of the river should be the main objective of riparian ecosystems recovery.

Although the Pesquería is currently in a poor ecological condition, cases such as that of the Llobregat demonstrate that ecological rehabilitation and recuperation of riparian ecosystems is indeed possible under the correct management conditions.

GENERAL CONCLUSIONS

The main conclusions of this PhD thesis are as follows:

- Anthropogenic land use can alter the riparian channel landscape, which, in turn, can affect biological quality and the ecological integrity of the river as an ecosystem.
- The Pesquería river is currently in poor ecological health, and our assessment evidences the considerable impact of the different types of anthropogenic land use present throughout the riparian channel.
- The ecohydrological management tools created and adapted for this thesis were found to be useful and appropriate for the evaluation of the ecological conditions of the Pesquería River.
- The continued lack of land use planning and inadequate monitoring polices in Mexico have contributed to the severe degradation of Mexican freshwater ecosystems, particularly that of the Pesquería River.

We present now the main results and conclusions of each chapter:

Chapter 1

Dynamics of land use cover/change and their influence on riparian channel ecosystem health: the case of the Pesquería River (N.E. Mexico).

- The principal land use cover/change processes in the Pesquería River basin have mainly converted the riparian forest into grassland

and agricultural zones, whereas the secondary forest has been transformed into urban zones.

- Significantly, revegetation processes were observed in the area studied.
- Revegetation was probably able to take place due the devastation of the areas of anthropogenic land use caused by the flash flood events that affected the Pesquería River occurred during the studied period (1976-2016).
- Anthropogenic activities and natural disturbances (e.g. flash floods) generate changes in habitat distribution, which assist the spread and establishment of invasive species and reduce the biodiversity of freshwater ecosystems.
- The riparian channel ought to be considered an ecosystem under the “Mexican National Water Law”. Current legislation understands it as a federal area and not as an ecosystem, meaning that it is not afforded the same level of environmental protection.
- A buffer area at least 50 meters in width should be defined and legally implemented in order to protect the riparian channel ecosystem under Mexican environmental law.
- Land use cover/change dynamics at riparian channel level should be considered an important evaluation tool for Mexican freshwater ecosystems given the precise information about land use generated by this analysis.

Chapter 2

The role of riparian vegetation in the evaluation of ecosystem health. The case of semiarid conditions in Northern Mexico

- Anthropogenic pressures and invasive species have had a considerable impact on the Pesquería River´s riparian vegetation.
- Anthropogenic impacts have caused vegetation cover to be sparse throughout most of the riparian area. In areas where the riparian vegetation cover was abundant, invasive species were found to be dominant.
- The QBR-RNMX was able to identify the main anthropogenic pressures occurring in the Pesquería River basin, were related with the urbanization and agricultural practices.
- The QBR-RNMX could still accurately describe the degradation of riparian areas even in locations where conditions were unfavourable for plants.

Chapter 3

The influence of riparian corridor land use on the Pesquería River's macroinvertebrate community (N.E. Mexico)

- The results of the indicator value analysis revealed different taxa for each typology of land use. These taxa can be used as bioindicators.
- The biological metrics used strongly support the notion that the Pesquería River ecosystem has been disturbed by the changes observed in the macroinvertebrate communities in response to the different riparian land uses.
- The Pesquería River's diagnosis is in line with "urban river syndrome", and it has the trademark characteristics of a river influenced by agricultural land use ("Agro-urban rivers").
- Several riparian forest "strips" were located, which help to mitigate the anthropogenic pressures affecting the Pesquería River's macroinvertebrate biodiversity. These small areas offer a great starting point for operations to recover the river's ecological health.
- The biological macroinvertebrate-based metrics selected for use in this chapter are a promising tool that can be used as a first step in the construction of a Multimetric index for this river, and can hopefully be used as bioindicators of land use pressures in other semi-arid rivers in the north of Mexico
- This study was limited by the synergistic effect of the accumulated impacts of urban and agricultural land use.

Furthermore, we propose a macroinvertebrate-based index (BMWP-system) for the evaluation of the ecological status of the river. However, more studies, and especially the relationship of this index with the river pressures are necessary to establish an index for the Pesquería River in the future.

Ce n'est pas du sang qui coule dans nos veines
C'est la rivière de notre enfance.

- La rivière de notre enfance -

Michel Sardou

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Oh, the river, oh the river, it's running free

And oh, the joy, oh the joy it brings to me

But I know it'll have to drown me

Before it can breathe easy

And I've seen it in the flights of birds

I've seen it in you the entrails of the animals

The blood running through, but in order to get to the heart

I think sometimes you'll have to cut through, but you can't

We will carry

We will carry you there

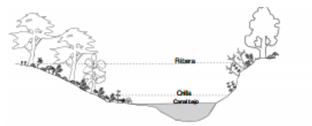
Just keep following

The heartlines on your hand

-Heartlines-

Florence + the Machine

SUPPLEMENTARY INFORMATION



APPENDIX A

Station	
Observant	
Date	

QBR-RNMX INDEX

Field data sheet

Riparian Forest Quality in rivers of the North of México.

Score of each part cannot be negative or exceed 25

TOTAL RIPARIAN COVER **Part 1 score**

Score			
1A	**25**	**> 50 % of riparian cover**	
1B	**10**	**30-50 % of riparian cover**	
1C	**5**	**10-30 % of riparian cover**	
1D	**0**	**< 10 % de of riparian cover**	
+10		if connectivity between the riparian forest and the woodland is total	
+5		if the connectivity is higher than 50%	
-5		connectivity between 25 and 50%	
-10		connectivity lower than 25%	

COVER STRUCTURE **Part 2 score**

Score						
	1A	**1B**	**1C**	**1D**		
2A	**25**	**10**	**5**	**0**	**75 % of tree cover**	
2B	**10**	**5**	**0**	**0**	**50-75 % of tree cover or 25-50 % tree cover but 25 % covered by shrubs**	
2C	**5**	**0**	**0**	**0**	**tree cover lower than 25% but shrub cover at least between 10 and 25 %**	
2D	**0**	**0**	**0**	**0**	**absence of trees ***	
+ 10 + 5 + 5					at least 50 % of the channel has helophytes or shrubs if 25-50 % of the channel has helophytes or shrubs if trees and shrubs are in the same patches	
- 5 - 5 - 10					if trees are regularly distributed but shrubland is > 50 % if trees and shrubs are distributed in separate patches, without continuity trees distributed regularly, and shrubland < 50 %	

***COVER QUALITY** (the geomorphological type should be first determined*)* **Part 3 score**

Score		*Tipo1*	*Tipo2*	
25	number of native tree species:	> 1	> 2	
10	number of native tree species:	1	2	
5	number of native tree species:	-	1	
0	absence of native trees			
+ 10 + 5 + 5	if the tree community is continuous along the river and covers at least 75% of the edge riparian area the tree community is nearly continuous and cover at least 50% of the riparian area when the number of shrub species is:	> 2	> 3	
- 5 - 5 - 10 - 10	if there are some man-made buildings in the riparian area is there is some isolated species of non-native trees** presence of communities of non-native trees presence of garbage			

NATURALNESS OF THE RIVER CHANNEL **Part 4 score**

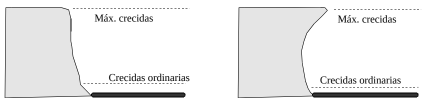
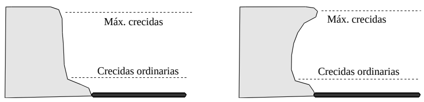
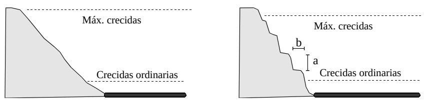
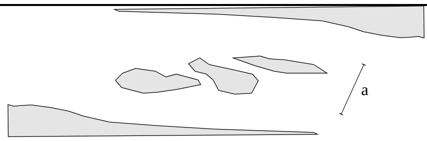
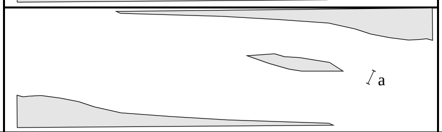
Score		
25	**unmodified river channel**	
10	**fluvial terraces modified, constraining the river channel**	
5	**channel modified by discontinuous rigid structures along the margins**	
0	**totally channelized river**	
- 10 - 10 -5 -10	river bed with rigid structures (e.g wells) transverse structures into the channel (e.g weirs) presence of garbage if there is a permanent landfill in the section studied	

Final score (sum of level scores)	

*** Trees with bushy size and the shrubs with arboreal size (height higher than 1.5 m) are considered.**

**** Type of the riparian habitat (to be applied at part 3, cover quality part). The score is obtained by addition of the scores assigned to left and right river margins according to their slope. This value can be modified if islands or hard substrata are present.**

****Add the slope type to the right and left of the bank and add or subtract according to the other two sections.**

		Score	
Slope and form of the riparian zone		**Left**	**Right**
Very steep, vertical or even concave (slope > 75º) margins are not expected to be exceeded by large floods.		**4**	**4**
Similar to previous category but with a bankfull which differentiates the ordinary flooding zone from the main channel.		**3**	**3**
Slope of the margins between 45 and 75 º, with or without steps. Slope is the angle subtended by the line between the top of the riparian area and the edge of the ordinary flooding of the river. (a > b)		**2**	**2**
Slope between 20 and 45 º, with or without steps. (a < b)		**1**	**1**
Presence of one or several islands in the river			
Width of all the islands "a" > 5 m.		- 2	
Width of all islands "a" < 5 m.		- 1	

Percentage of hard substrata that can made impossible the presence of plants with roots

> 80 %	Not applicable
60 - 80 %	+ 3
30 - 60 %	+ 2
20 - 30 %	+ 1

Total Score	

Geomorphological type according to the total score

> 6	**Tipo 1**	**Closed riparian habitats. Riparian forest, if present, reduced to a small strip. Headwaters.**
3 to 5	**Tipo 2**	**Headwaters or midland riparian habitats. Forest may be large and originally in gallery.**

**** Allochthonous trees species in the study area***: Acacia rigidula, Arundo donax, Chaptalia nutans, Cyperus alternifolius, Nicotina glauca, Ricinus communis, Tamarix aphylla, Typha ssp.*

APPENDIX B

Vegetation inventory recorded and used to evaluate the riparian vegetation quality of the Pesquería River.

Scientific name	Common name	PG†	Sampilng points
Acacia farnesiana	Huizache	NRR	2,4,5,6,7,8,9,11,13,15
Acacia rigidula	Chaparro prieto	NWDM	2
Arundo donax	Carrizo gigante	I	2,8,9,10,11,12,14,15
Baccharis salicifolia	Azumiate o Jara	NWDM	1,2,3,4,5,7,13,14,15
Cercidium texanum	Palo Verde	NRR	5,9,11,13
Chaptalia nutans	Agacha cabeza	I	6,8,9
Cyperus alternifolius	Paragüita	I	1,2,7,10,12
Nicotina glauca	Tabachín	I	3
Pluchea carolinensis	Santa María	NWDM	1,2,3,4,5,7,13,14,15
Prosopis glandulosa	Mezquite	NRR	2,3,6,7,8,9,11,13,15
Ricinus communis	Higuerilla	I	5,6,7,8,9,10,11,12,15
Salix nigra	Sauce de río	NRR	1,3,4,5,8,9,11
Tamarix aphylla	Pinabete	I	3,4,5,8,9,14
Thypa Latifolia	Junco, Tule	I	1,2,3,6,7

PG†= Phytogeographic Origin (N= Native Representative of the Region; NWD= Native Widely Distributed in Mexico; I=Invasive).

APPENDIX C

TABLE S1. Land use classification criteria applied on this study.

Type of land use [46,51]	Natural land use cover or anthropogenically influenced cover	Selected land use criteria for this study [44]
Microphyllous desert shrubland	Natural	Natural land use
Rosetophyllous desert shrubland	Natural	Natural land use
Tamaulipan thornscrub	Natural	Natural land use
Xerophitic scrubland mesquite	Natural	Natural land use
Piedmont scrub	Natural	Natural land use
Secondary vegetation of Tamaulipan thornscrub	Anthropic	Classification of land use will depend on the type of anthropic pressure closest to this type of land use. (Urban/ Agricultural).
Secondary vegetation of Piedmont scrub	Anthropic	Classification of land use will depend on the type of anthropic pressure closest to this type of land use. (Urban/ Agricultural).
Secondary vegetation of Microphyllous desert shrubland	Anthropic	Classification of land use will depend on the type of anthropic pressure closest to this type of land use. (Urban/ Agricultural).
Permanent grassland	Anthropic	Agricultural land use
Induced grassland	Anthropic	Agricultural land use
Anual rainfed agriculture	Anthropic	Agricultural land use
Anual irrigated agriculture	Anthropic	Agricultural land use
Urban zone	Anthropic	Urban land use
Human settlements	Anthropic	Urban land use
Non-vegetated area	Anthropic	Urban land use

Figure S1. Pesquería River Sub-Basin drone aerial photographs. Natural land use photographs correspond to sampling site 1, urban land use photographs to sampling site 4 and agricultural land use photographs to sampling site 15.

TABLE S2 Land use classification and percentage of each site

Sampling Site	SS-Name-reference	X-Location	Y-Location	DBSS-KM	D-KM	Channel width (m)	LUC	Nat %	Urb %	Agr %	Description of Land use in each sampling site.
SS1	García	334092.63	2855097.19	0.0	0.0	18	Nat	100	0	0.0	Natural Land use
SS2	García	335270	2854277	1.4	1.4	17.17	Nat	100	0	0.0	Natural Land use
SS3	García	336345	2855062	2.1	3.5	19.21	Nat	73	27	0.0	Natural land use and starts the urbanization on the left bank of the river.
SS4	García	342461.64	2854345.01	7.1	10.7	18.01	Urb	0	100	0	Urbanization in the left and right bank of the river.
SS5	García	353772.06	2853947.15	13.0	23.7	18.88	Urb	0	100	0	Urbanization human settlements in left and right bank of the river.
SS6	García	361061.29	2854164	8.2	31.9	19.26	Urb	0	100	0	Urbanization and human settlements left and right bank of the river. Small strips of riparian forest/vegetation less than 10 m width of the sampled area.
SS7	Escobedo	363688.43	2855392.36	3.8	35.7	19.33	Urb	0	100	0	Urbanization and human settlements left and right bank of the river. Small strips of riparian forest/vegetation less than 10 m width of the sampled area.
SS8	Escobedo	369493	2854813	6.6	42.3	17.1	Urb	0	100	0	Urbanization and human settlements left and right bank of the river. Small strips of riparian forest/vegetation less than 15 m width of the sampled area.
SS9	Santa Rosa	377346.01	2856678.33	10.5	52.8	20.4	Urb	0	100	0	Urbanization and human settlements left and right bank of the river.
SS10	Agua fría	384632.56	2855949.88	8.9	61.7	18.83	Agr	0	0	100	Agricultural fields
SS11	Pesquería	393388.76	2852351.96	10.0	71.7	21.27	Agr	39.7	0	60.3	Agricultural fields. Riparian forest/ vegetation is present in one of the river banks.
SS12	Hipódromo	400753.38	2849990.38	9.6	81.3	19.85	Agr	0	0	100	Agricultural fields.
SS13	Pesquería	420004.01	2845289.61	20.6	101.9	29.7	Agr	0	0	100	Agricultural fields.
SS14	San Isidro	424212.44	2840113.73	7.6	109.5	20.8	Agr	0	0	100	Agricultural fields. Small strips of riparian vegetation.
SS15	El Refugio	431131.22	2837805.68	8.4	117.9	32.2	Agr	48	0	52	Agricultural fields. Riparian forest/ vegetation is fully present in one of the river banks.
SS16	San Agustín	463589.31	2865321.59	60.0	177.9	27.34	Agr	0	0	100	Agricultural fields. Small strips of riparian vegetation.

CODE	Meaning
Nat	Natural
Urb	Urban Zones
Agr	Agriculture
SS	Sampling Site
DBSS	Distance between sampling sites
DF-SS1-SS16	Distance from site 1-16
LUC	Land use classification

TABLE S3 Sampling site impacts classification

Sampling Site	LUC	Impacts	Distance from the impact to the site
SS1	Nat	No impact present	Natural land use
SS2	Nat	No impact present	Natural land use
SS3	Nat	WSP	The water discharge is located 200 meters before reaching the next sampling site (ss4).
SS4	Urb	WWTP / Urbanization/ B) Indirect from Industry	Urbanization and human settlements left and right bank of the river.
SS5	Urb	Urbanization	The discharge comes from the WWTP located 12.8 km upstream / The indirect discharge from industry comes from the 9.36 km upstream from the sampling site.
SS6	Urb	CD/ HS	Clandestine dump is 0.10 km upstream from the sampling site. Human settlements are 0.15km upstream from the SS.
SS7	Urb	Urbanization	Urbanization and human settlements left and right bank of the river.
SS8	Urb	Urbanization	Urbanization and human settlements left and right bank of the river.
SS9	Urb	A) WWTP	The discharge comes from 8.45 km upstream from the sampling site 8.
SS10	Agr	A) WWTP/ B) WWTP	The discharge comes from A) 6.87 km / B) 3 km upstream from the sampling site 9.
SS11	Agr	Agriculture	Agricultural fields
SS12	Agr	WWTP	The discharge comes from 3 km upstream from the sampling site 11.
SS13	Agr	WTTP/ Agriculture	The discharge comes from 15.7 km upstream from the sampling site 12 /Agricultural fields
SS14	Agr	Agriculture	Agricultural fields
SS15	Agr	WSP	The discharge comes from 44 km upstream from the sampling site.
SS16	Agr	Agriculture	Agricultural fields

CODE	Meaning
WSP	Water Stabilization Pond
WWTP	Wastewater Treatment Plant
CD	Clandestine Dump
HS	Human settlements

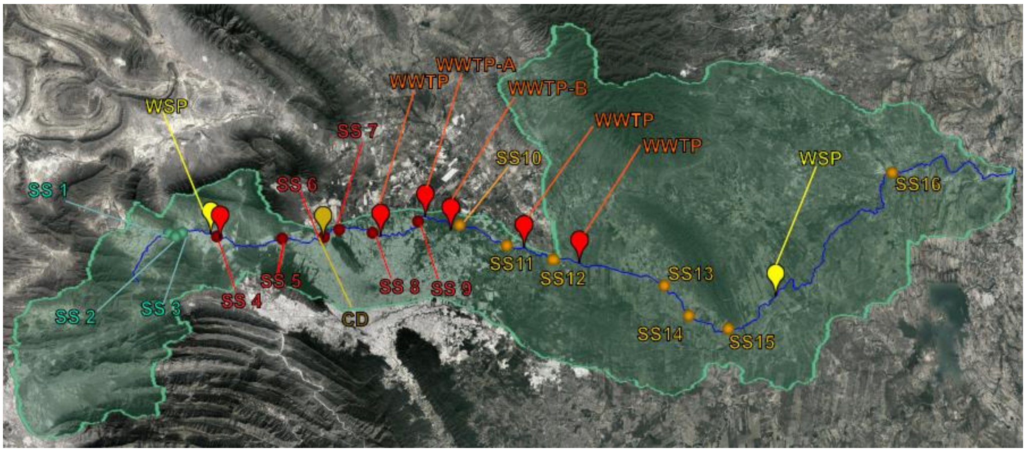


Figure S2 Pesquería River Sub-Basin drone aerial photographs

CODE	Meaning
WSP	Water Stabilization Pond
WWTP	Wastewater Treatment Plant
CD	Clandestine Dump
SS	Sampling Sites

APPENDIX D

TABLE S1 Metrics data

mean ± sem values

BIOLOGICAL METRICS REPRESENTATIVE OF LAND USE							TukeyHDS			NATURAL		URBAN		AGRICULTURAL	
CATEGORY I "TAXONOMIC RICHNESS"	ML/MML	TRNS	R2c	R2m	R2ad	p-values	N	U	A	mean	± sem	mean	± sem	mean	± sem
Richness	LMM	ST	0.7	0.94		<0.001	a	b	b	35.17	1.05	15.83	2.07	11.14	1.23
Rarified Richness	LMM	ST	0.77	0.95		<0.001	a	b	b	33.45	1.08	13.49	1.69	9.39	1.07
Richness OCH (Odonata + Coleoptera + Heteroptera)	LMM	ST	0.66	0.89		<0.001	a	b	b	12.33	0.84	5.67	0.80	3.00	0.53
Richness EPT (Ephemeroptera + Plecoptera + Trichoptera)	LMM	SQRT	0.54	0.93		<0.001	a	b	b	9.67	0.61	2.00	0.55	1.29	0.40
Richness EPT / (richness EPT + richness OCH)	LMM	ST	0.25	0.9		0.090	a	a	a	0.44	0.03	0.20	0.04	0.19	0.05
Richness Ephemeroptera	LMM	SQRT	0.6	0.95		0.001	a	b	b	4.33	0.49	0.42	0.34	0.36	0.25
Richness Trichoptera	LMM	SQRT	0.45	0.91		0.001	a	b	b	5.33	0.42	1.58	0.31	0.93	0.25
Richness Odonata	LMM	ST	0.27	0.65		0.045	a	ab	b	5.00	0.63	3.33	0.43	2.29	0.44
Richness Coleoptera	LMM	SQRT	0.52	0.82		0.002	a	a	b	5.00	0.97	1.92	0.38	0.50	0.25
Richness Diptera	LMM	SQRT	0.62	0.87		<0.001	a	b	b	7.17	0.48	3.33	0.48	2.21	0.15
Richness Diptera wihout Chironomidae	LMM	SQRT	0.48	0.86		<0.001	a	b	b	6.17	0.48	2.42	0.48	1.21	0.15
Richness Gasteropoda	LMM	SQRT	0.1	0.66		0.350	a	a	a	1.00	0.37	1.33	0.36	0.43	0.17
Richness Non-insect taxa (Amphipoda + Copepoda + Ostracoda + Gateropoda + Oligochaeta + Hirudinea)	LMM	SQRT	0.001	0.53		0.980	a	a	a	4.17	0.40	4.25	0.57	4.29	0.40
CATEGORY III "RICHNESS COMPOSITION"							N	U	A	MEAN	SEM	MEAN	SEM	MEAN	SEM
% OCH (Odonata + Coleoptera + Heteroptera)	LM	SQRT			0.41	<0.001	a	a	b	0.08	0.02	0.05	0.02	0.01	0.00
% EPT (Ephemeroptera + Plecoptera + Trichoptera)	LM	SQRT			0.64	<0.001	a	b	c	0.27	0.02	0.08	0.03	0.03	0.02
% EPT / (% EPT + % OCH)	LMM	ST	0.16	0.75		0.219	a	a	a	0.78	0.03	0.44	0.11	0.35	0.11
% Ephemeroptera	LM	SQRT			0.56	0.000	a	b	b	0.16	0.04	0.02	0.02	0.01	0.00
% Trichoptera	LMM	SQRT	0.28	0.87		0.640	a	ab	b	0.11	0.02	0.05	0.02	0.02	0.01
% Odonata	LM	SQRT			0.36	<0.001	a	a	b	0.03	0.00	0.02	0.01	0.00	0.00
% Coleoptera	LM	SQRT			0.37	<0.001	a	a	b	0.04	0.01	0.04	0.02	0.00	0.00
% Diptera	LM	SQRT			0.11	0.060	a	a	a	0.31	0.03	0.44	0.07	0.55	0.05
% Diptera wihout Chironomidae	LM	ST			0.17	0.025	a	b	b	0.09	0.03	0.02	0.01	0.03	0.01
% Gasteropoda	LM	SQRT			0.16	0.027	a	ab	b	0.04	0.03	0.01	0.01	0.00	0.00
% Non-insect taxa (Amphipoda + Copepoda + Ostracoda + Gateropoda + Oligochaeta + Hirudinea)	LM	ST			-0.024	0.540	a	a	a	0.33	0.03	0.43	0.06	0.41	0.05
% Baetidae / % EPT	LMM	ST	0.51	0.84		0.001	a	b	b	0.31	0.09	0.02	0.01	0.03	0.02
% Baetidae / % Ephemeroptera	LMM	ST	0.31	0.77		0.037	a	b	b	0.65	0.13	0.09	0.08	0.12	0.08
% Hydropsychidae / % EPT	LMM	SQRT	0.056	0.36		0.690	a	a	a	0.19	0.11	0.37	0.12	0.39	0.11
% Hydropsychidae / % Ephemeroptera	LMM	SQRT	0.69	0.007		0.993	a	a	a	0.40	0.11	0.40	0.11	0.42	0.12
% Chironomidae	LM	ST			0.19	0.016	a	b	b	0.22	0.03	0.42	0.08	0.53	0.04
% Oligochaeta	LM	SQRT			0.05	0.170	a	a	a	0.05	0.02	0.14	0.05	0.21	0.06
% Chironomidae + % Oligochaeta	LM	ST			0.34	0.001	a	b	b	0.27	0.02	0.55	0.09	0.73	0.05
CATEGORY I "DIVERSITY INDEXES"							N	U	A	MEAN	SEM	MEAN	SEM	MEAN	SEM
Shannon's Diversity using base e	LM	ST			0.66	<0.001	a	b	b	2.52	0.09	1.29	0.15	1.01	0.07
Simpson's Diversity	LM	SQRT			0.53	<0.001	b	a	a	0.13	0.02	0.41	0.06	0.47	0.03
Pielou's Evenness	LM	ST			0.46	<0.001	a	b	b	0.71	0.03	0.47	0.04	0.44	0.02
CATEGORY IV "FUNCTIONAL FEEDING GROUPS"							N	U	A	MEAN	SEM	MEAN	SEM	MEAN	SEM
Richness Collectors-Gatherers	LM	SQRT			0.8	<0.001	a	b	c	10.00	0.26	4.50	0.47	4.43	0.31
% Collectors-Gatherers	LM	SQRT			0.42	<0.001	a	b	b	0.68	0.04	0.86	0.04	0.93	0.02
Richness Predators	LMM	ST	0.57	0.85		<0.001	a	b	b	12.67	0.92	6.33	0.87	4.00	0.64
% Predators	LM	SQRT			0.24	0.006	a	ab	b	0.09	0.02	0.07	0.03	0.02	0.01
Richness Herbivores	LMM	ST	0.59	0.88		0.001	a	b	b	8.83	0.40	3.50	0.69	1.50	0.60
% Herbivores	LM	SQRT			0.64	<0.001	a	b	c	0.15	0.04	0.03	0.01	0.00	0.00
Richness Collectors-Filterers	LMM	SQRT	0.18	0.83		0.180	a	a	a	2.83	0.31	1.25	0.37	1.21	0.24
% Collectors-Filterers	LM	SQRT			0.31	0.002	a	b	c	0.07	0.03	0.04	0.02	0.04	0.02

Codes	
Linear Mixed Model	LMM
Linear Model	LM
Transformation	TRNS
Without transformation	ST
Square Root Transformation	SQRT
Coefficient of determination conditional for Linear Mixed Model	R2c
Coefficient of determination marginal for Linear Mixed Model	R2m
Coefficient of determination adjusted for Linear model	R2ad

TABLE S2 Discarded metrics. Models that responded to the distance from headwaters (HDF)

```
% EPT
Call:
lm(formula = EPTsq ~ impact + distance, data = dat)

Residuals:
     Min       1Q   Median       3Q      Max
-0.27465 -0.04805  0.00093  0.04183  0.32565

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)  0.5149693  0.0524569   9.817 1.44e-10 ***
impactB     -0.4112627  0.0701392  -5.864 2.64e-06 ***
impactC     -0.7565623  0.1110580  -6.812 2.12e-07 ***
distance     0.0032376  0.0009033   3.584  0.00127 **
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.1284 on 28 degrees of freedom
Multiple R-squared:  0.6801,   Adjusted R-squared:  0.6458
F-statistic: 19.84 on 3 and 28 DF,  p-value: 4.268e-07
```

```
Collector-Filterers %
Lm (formula = CFXsq ~ impact + distance, data = dat)

Residuals:
     Min       1Q   Median       3Q      Max
-0.21572 -0.05795 -0.01728  0.02553  0.26906

Coefficients:
              Estimate Std. Error t value Pr(>|t|)
(Intercept)  0.2323180  0.0523485   4.438 0.000129 ***
impactB     -0.2023677  0.0699943  -2.891 0.007339 **
impactC     -0.4650233  0.1108285  -4.196 0.000248 ***
distance     0.0035183  0.0009015   3.903 0.000545 ***
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 0.1282 on 28 degrees of freedom
Multiple R-squared:  0.3958,   Adjusted R-squared:  0.3311
F-statistic: 6.114 on 3 and 28 DF,  p-value: 0.002471
```

```
Collector-gatherers richnnes
Call:
lm(formula = CGR ~ impact * time + distance, data = dat)

Residuals:
    Min      1Q  Median      3Q     Max
-2.5920 -0.3676 -0.0525  0.4470  3.1686

Coefficients:
               Estimate Std. Error t value Pr(>|t|)
(Intercept)   10.969912   1.432033   7.660 5.13e-08 ***
impactB       -5.575059   1.770632  -3.149  0.00421 **
impactC      -10.583983   1.885806  -5.612 7.71e-06 ***
time          -0.666667   0.905661  -0.736  0.46851
distance       0.018422   0.007801   2.361  0.02631 *
impactB:time  -0.333333   1.109204  -0.301  0.76627
impactC:time   2.095238   1.082472   1.936  0.06430 .
---
Signif. codes:  0 '***' 0.001 '**' 0.01 '*' 0.05 '.' 0.1 ' ' 1

Residual standard error: 1.109 on 25 degrees of freedom
Multiple R-squared:  0.8447,   Adjusted R-squared:  0.8074
F-statistic: 22.66 on 6 and 25 DF,  p-value: 5.586e-09
```

APPENDIX E

This Appendix provides the original publications of Chapter 2 and 3.

The role of riparian vegetation in the evaluation of ecosystem health: The case of semiarid conditions in Northern Mexico.

Daniel Castro-López, Víctor Guerra-Cobián & Narcís Prat.

River Research and Applications 2019; 35:48–59

DOI: 10.1002/rra.3383

Impact factor (2018): 1.954; (Q1) Environmental Science; (Q1) Water Science and Technology.

The Influence of Riparian Corridor Land Use on the Pesquería River's Macroinvertebrate Community (N.E. Mexico)

Daniel Castro-López, Pablo Rodríguez-Lozano, Rebeca Arias-Real, Víctor Guerra-Cobián and Narcís Prat

Water 2019, 11(9), 1930.

This paper is part of a Special Issue about Human-Induced Changes to Aquatic Communities: Monitoring and Ecological Restoration.

DOI: 10.3390/w11091930

Impact factor (2018): 2.66; (Q1) Water Science and Technology; (Q2) Aquatic Science.