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Interactions between sediment transport, physical habitat and benthic communities in a mountainous river affected by natural and human disturbances

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Cover photo: The River Cinca during in-channel gravel mining, August 2014.

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TESI DOCTORAL

INTERACTIONS BETWEEN SEDIMENT TRANSPORT, PHYSICAL HABITAT AND BENTHIC COMMUNITIES IN A MOUNTAINOUS RIVER AFFECTED BY NATURAL AND HUMAN DISTURBANCES

María Béjar Maceiras

Memòria presentada per optar al grau de Doctor per la Universitat de Lleida

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Al ingeniero que me enseñó a hacer mapas con Paint.

Esta tesis habla de ríos y de montañas, pero está hecha por sus gentes: por aquellos que decidieron formar su familia entre montañas (gracias Ta, Pepe, Abuelo y Abuela, sabia decisión la vuestra), por aquellos que me enseñaron a amarlas (Mamá, Papá), por aquellos que decidieron vivir por y para sus ríos (Damià, Ramon, Chris, RIUS) y por todos aquellos amigos que día tras día han creado entre las montañas este enérgico imán. Pero sobre todo, esta tesis está hecha para todos aquellos nabateros, condesas del Sobrarbe y damas del lago que dieron su vida en el Cinca. "Si los hombres permanecen los dioses ya volverán".

Los mapas siguieron trayendo tu nombre
¡Quién puede olvidarse de ti!
tozales altivos y ríos crecidos
gritaban: "¡Seguimos aquí!".
Pero divididos y sin dar batalla
te fuimos perdiendo, país.

Tal vez a trocitos se te fue llevando
la gente que hicieron marchar,
o bajo las aguas de un negro pantano
reposas dormido y en paz,
igual que a los pies de Peña Montañesa
las ruinas de San Beturián.

¡Venid dioses que dormís debajo un dolmen!
¡Guerreros y Santos venid!:
hijos de la historia y de nuestras leyendas,
ayuda os pedimos ¡venid!.
¡Cruzad ya los puertos, viejos guerrilleros,
reconquistemos el país!

Manuel Domínguez. La Ronda de Boltaña Fragmento de "El Pais Perdido".

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Había oído muchas veces aquello de que la tesis no la hace solo una persona pero, como siempre, hasta que no te toca no ves lo que de verdad implica. Y una tesis no es solo campo, pero sí que es la parte más humana, una de la que más energía demanda y para mí una de las más gratificantes. Quiero agradecer enormemente la ayuda de cada una de las personas que ha pasado por el Cinca durante estos años, que han sido muchas (Figura 0. me he permitido hacer una figura extraoficial de la tesis). El Alto Cinca, un lugar maravilloso para trabajar, rodeado de esas preciosas montañas, pero también con esas temperaturas extremas y esas distancias inabarcables. Un lugar en el que todo RIUS de una manera u otra ha participado, gracias a aquellos que vinieron desde Barcelona y Girona para ayudar en las primeras campañas, a los que echaron manos y piedras durante la extracción, aquellos que ayudaron en las 30 mil instalaciones del Cinca y en especial a Ester, Efren y Manel que se tragaron todas las campañas de bichos. A los amigos que en plena campaña de extracción tuvieron que venir last minute a echar una mano y a mis padres que más de una vez me han acompañado con la excusa de comer luego en la plaza. A la gente de Ainsa que ha facilitado la vida en el campo y nos ha transmitido su pasión por el Cinca. Y en esta parte final, a Celso García y Sergi Sabater que como revisores externos han ayudado a mejorar la versión final de esta tesis. En definitiva a todo aquel que entre gravas y agua de deshielo ha aportado su granito de arena.

Y después del Cinca vienen las interminables horas de laboratorio y los largos días del 5B. Muchas gracias a Clara por ser como un ángel de la guarda con todos los papeleos y la burocracia. A todo el Departamento de Medio Ambiente por resolver las dudas y los problemas que he tenido durante estos años. A los becarios del departamento, estos cuatro años hubieran sido distintos sin las sobremesas en la cafetería y los jueves en el Lizarran. A

toda la gente de Aberdeen que me acogió en el Northern River Institute y en especial a Baptiste por cuidar de mi durante esos meses y por todas las charlas y cervezas compartiendo penas y glorias de la tesis. También a la gente que desde lejos siempre se ha preocupado de cómo iba el proceso incluso se han leído los artículos, desde Humbert Torres a Salamanca pasando por todos los que forman la familia elegida.

En total han sido 35 kilogramos de material tamizado (entre fino y carga de fondo), 200 muestras de drift + 430 muestras de benthos (a una media de 60 animales por muestra salen 37800 animales muestreados e identificados), 600 litros de muestras de agua filtradas, 3 cuadernos de notas, unos 8000 km de carretera Lleida-Ainsa-Lleida, unos 100 km andados por las gravas del Cinca y el 14 % de mi tiempo vivido dedicado a lo que pone dentro de este volumen. Espero que si alguien se lee aunque solo sea una parte de la tesis la pueda disfrutar tanto como yo lo he hecho durante estos años.

Figura 0.Porque una tesis nunca es cosa de uno, ni de dos, ni de diez.

CONTENTS

SUMMARY

This doctoral thesis presents new insights on the interactions between physical and biological processes driven by natural (i.e. tributaries and floods) and human disturbances (i.e. gravel mining and hydropeaking) in a montane river (the upper Cinca, Southern Pyrenees). A multi-spatial and temporal data acquisition approach was implemented during two hydrological years (October 2014 – September 2016). A number of complementary conventional and up-to-date techniques and methods were integrated to examine key physical and ecological processes in the river. Discharge, suspended sediment transport, river-bed and suspended sediment grain-size distributions, invertebrate drift and benthic invertebrate assemblages were monitored and sampled along a 9.8-km river reach. Metrics of habitat heterogeneity and bed surface complexity were derived from high resolution field data. The work addressed processes and interactions spanning instantaneous to annual temporal scales, and from grain to river reach scales. Results indicate that human impacts and natural dynamics influenced water and suspended sediment transport (dynamics and magnitudes). Water and sediment loads differed substantially between the three monitoring sections: suspended sediment transport patterns changed from a torrential regime in (a) the most upstream section, controlled mainly by floods, to (b) a middle reach highly controlled by hydropeaking and tributary inflows (annual water yield here increases four-fold compared to upstream), to (c) the most downstream section where sediment load doubled as a result of instream gravel mining and the supply from tributaries (i.e. increased sediment availability). Patterns indicate that geographic location and the magnitude of such processes can exert an additive or compensatory effect on runoff and sediment load over relatively short channel distances. In particular, the specific sediement yield of the Upper Cinca at Ainsa has been estimated at around 74 Mg/km^2/y. Notably, in-channel mining increased sediment transport locally during short periods when gravels are being extracted. Average concentrations during mining were similar to those observed during natural floods. Maximum concentrations around 6 g/l and loads in the order of 17 Mg/day were obtained at sections close to the mining. Transported sediments were coarser than the observed under natural flood conditions, thus sediment transport was highly controlled by grain-size and particles settle in the channel due to the low competence of the flows while mining. Fine sediment deposited in the channel increased sediment availability downstream the perturbed reach, generating off-site effects and deferred impacts downstream. The results show as these effects propagated for 1.5 km downstream. Invertebrate drift responded directly to the increase in sediment concentrations, with the duration in which concentrations are high and the size of the transported particles both influencing the numbers of animals drifting. The response to suspended sediment increases varied between taxonomic groups, with relationships positive for Ephemeroptera, Plecoptera and Trichoptera and negative for Chironomidae. Results from this work do not allow to define the ultimate causes of invertebrate drift, but these open a new research line for future works. Activities such as gravel mining or flow regulation that homogenise river bed habitat potentially affect benthic invertebrate diversity. To address this issue, different sampling techniques and methodologies were integrated to provide high resolution data that was used to characterise habitat structure at a range of spatial scales. Locations with low microhabitat heterogeneity and surface complexity generally had the fewest individuals and taxa. Although flow hydraulics and surface complexity changed markedly across scales (moving from 1 to 16 m^2), between-patch variation in macroinvertebrate community structure was best explained by patch scale (1 m^2) habitat characteristics; indicating that macroinvertebrates within the study sections of the Cinca were responding their local immediate habitat, rather than conditions across surrounding streambed areas. Overall, this thesis highlights how physical and biological processes interact at different scales in river systems affected by multiple disturbances. Identification of the key scales of interaction and the integration of sampling techniques that provide continuous and high density measurements are shown to provide important insights into species-habitat interactions and, in turn, a firm evidence base the assessment and sustainable management of fluvial systems.

Key words: suspended sediment transport, in-channel gravel mining; hydropeaking; water yield, sediment load,; tributary inflow; grain size distribution; sediment availability; benthic invertebrate assemblages; invertebrate drift; habitat diversity, bed surface complexity, River Cinca, Ebro basin.

RESUM

Aquesta tesi doctoral analitza les interaccions entre diversos processos físics i biològics que es donen a les lleres dels rius, el desencadenant de les quals està relacionat amb factors naturals (crescudes i entrada d'afluents) i activitats antròpiques (extraccions d'àrids i hidropuntes). La investigació s'ha dut a terme en un riu de muntanya, l'alt Cinca (sud dels Pirineus). Per assolir aquest objectiu s'han obtingut dades de camp seguint un disseny multiescalar tant a nivell temporal com espacial durant dos anys hidrològics (octubre 2014 - setembre 2016). L'estudi dels processos físics i biològics s'ha realitzat mitjançant la integració de mètodes i tècniques de mesura i presa de mostres convencionals i de darrera generació. Al llarg de la tesi s'han mesurat i mostrejat el cabal, el transport de sediments en suspensió, la distribució de la mida de les partícules de la llera i dels sediments en suspensió, la deriva dels macroinvertebrats i les comunitats bentòniques en diverses seccions del riu al llarg d'un tram de 9,8 km. Així mateix, s'han calculat índexs d'heterogeneïtat de l'hàbitat i complexitat superficial del llit del riu a partir de dades d'elevada densitat obtingudes durant campanyes de camp realitzades per aquest propòsit. Aquests processos i les seves interaccions s'han avaluat des d'una escala instantània fins anual, i des de l'escala de partícula a l'escala de tram. Els resultats mostren que tant els factors naturals com les activitats antròpiques modifiquen de manera notable el règim hídric i el transport de sediments, tant en variabilitat com en magnitud. L'aportació hídrica i la càrrega de sediments varien substancialment entre les seccions de control: (a) en el tram d'aigües amunt el règim líquid i sòlid és de tipus torrencial i està controlat principalment per les crescudes; (b) en el tram central la circulació d'aigua i de sediments està controlada tant pel règim d'hidropuntes com per l'entrada d'un afluent important (p.ex. l'aportació hídrica anual en aquesta secció s'incrementa quatre vegades en comparació amb la secció de aigües amunt); i (c) en el tram final la càrrega de sediments es duplica a causa de les activitats extractives i a la contribució d'afluents menors (i.e. increment de la disponibilitat de sediments). La càrrega sedimentària de l'alt Cinca a la sortida del tram d'estudi és de 72 Mg/km^2/any. El règim hidrosedimentari observat al Cinca mostra que tant la localització geogràfica dins de la conca com la magnitud dels processos exerceixen un efecte additiu o compensatori sobre l'aportació hídrica i la càrrega de sediment en distàncies (10^1-10^3 m) i en períodes de temps relativament curts (hores). Cal destacar, per exemple, l'efecte puntual però intens de les extraccions d'àrids, que incrementen localment el transport de sediment arribant a concentracions que habitualment només es registren durant crescudes (i.e. concentracions de fins a 6 g/l i càrregues totals superiors als 17 Mg/dia). A les zones d'extracció la mida del sediment transportat és major que en condicions naturals. L'escassa competència dels cabals durant les extraccions, condiciona el transport de sediments i facilita una ràpida sedimentació de les partícules a la llera, augmentant la seva disponibilitat per crescudes posteriors. Les observacions al Cinca han permès també constatar que els macroinvertebrats responen directament a l'increment de la càrrrega sòlida en suspensió aigües avall de les actuacions. La durada de les concentracions altes i la mida més gran de les partícules transportades influencien en el nombre de macroinvertebrats que entren a la deriva. Aquesta resposta varia en funció dels grups taxonòmics analitzats, amb respostes positives en els grups Ephemeroptera, Plecoptera i Trichoptera i respostes negatives en els Chironomidae. Els resultats de la tesi no permet definir les causes últimes d'aquesta deriva però obren una línia d'investigació per a treballs futurs. El paper de les extraccions d'àrids i les alteracions del cabal com a factors d'homogeneïtzació de l'hàbitat, i els seus efectes sobre la diversitat de les comunitats de macroinvertebrats han sigut també examinats. Aquests efectes s'han avaluat mitjançant la integració de mètodes i tècniques de mostreig que faciliten l'obtenció de dades d'elevada resolució; per exemple, la generació d'informació topogràfica d'alta densitat i de manera continua (espai) permet caracteritzar de manera detallada l'estructura de l'hàbitat físic d'una llera fluvial, i fer-ho a diferents escales espacials. En general, els resultats idiquen que els llocs amb una heterogeneïtat d'hàbitat i complexitat superficial menors són els que menys abundància i tàxons de macroinvertebrats presenten. D'altra banda, tot i que les condicions de l'hàbitat van canviar al llarg de les escales estudiades (des d'1 m^2 a 16 m^2), en conjunt, en el tram d'estudi del Cinca els macroinvertebrats mostrejats estan més influenciats per l'hàbitat local (i.e. escala d'1 m^2) que per les condicions físiques generals de l'entorn fluvial. Globalment aquesta tesi descriu diferents nivells d'interrelació entre processos físics i biològics fonamentals en sistemes fluvials desencadenats per diversos factors naturals i antròpics. La identificació de les escales d'interacció així com la integració i l'ús de noves metodologies i tècniques de mostreig han demostrat ser importants per a l'estudi de la relació espècie-hàbitat i constitueixen un nou exemple d'integració eco-geomorfològica que contribueix al coneixement dels sistemes fluvials i dóna suport a la gestió sostenible dels mateixos.

Paraules clau: Transport sediments en suspensió, extraccions d'àrids, hidropuntes, aportació hídrica, afluents, distribució granulomètrica, disponibilitat de sediment, deriva de macroinvertebrats, comunitats bentòniques, diversitat d'hàbitat, complexitat superficial del llit, riu Cinca, conca de l'Ebre.

Resumen

Esta tesis doctoral analiza las interacciones entre diversos procesos físicos y biológicos que tienen lugar en cauces fluviales, y cuyo desencadenante está relacionado con factores naturales (crecidas y entrada de afluentes) y actividades antrópicas (extracciones de áridos e hidropuntas). La investigación se ha desarrollado en un río de montaña, el alto Cinca (sur de los Pirineos). Con este objetivo se han obtenido datos de campo siguiendo un diseño multiescalar tanto a nivel temporal como espacial durante dos años hidrológicos (octubre 2014 - septiembre 2016). El estudio de los procesos físicos y biológicos se ha llevado a cabo mediante la integración de técnicas de medición y muestreo convencionales y de última generación. A lo largo de la tesis se han medido y muestreado el caudal, el transporte de sedimentos en suspensión, la distribución del tamaño de partículas del sedimento en suspensión y del lecho, la deriva de macroinvertebrados y las comunidades bentónicas en diversas secciones del río a lo largo de un tramo de 9,8 km. Asimismo se han calculado índices de heterogeneidad del hábitat y complejidad superficial del lecho obtenidos durante mediciones específicas realizadas a tal efecto. Dichos procesos y sus interacciones se han evaluado desde una escala instantánea hasta anual, y desde la escala de partícula a la escala de tramo. Los resultados muestran que tanto los factores naturales como las actividades antrópicas modifican de manera notable el régimen hídrico y el transporte de sedimentos, tanto en variabilidad como en magnitud. La aportación hídrica y la carga de sedimentos varían sustancialmente entre las secciones monitorizadas: (a) en la tramo de aguas arriba el régimen líquido y sólido es de tipo torrencial y está controlado principalmente por las crecidas; (b) en el tramo intermedio la circulación de agua y de sedimentos está sujeta a un régimen de hidropuntas y a la contribución de un importante afluente (p.ej. la aportación hídrica anual en esta sección se incrementa cuatro veces en comparación con la sección de aguas arriba); y (c) en el tramo final la carga de sedimentos se duplica debido a las actividades extractivas y a la contribución de afluentes menores (i.e. incremento de la disponibilidad de sedimentos). El régimen hidrosedimentario observado en el Cinca muestra que tanto la localización geográfica dentro de la cuenca como la magnitud de los procesos ejercen un efecto aditivo o compensatorio sobre la aportación hídrica y la carga de sedimento en distancias (10^1-10^3 m) y en períodos de tiempo relativamente cortos (horas). Cabe destacar, por ejemplo, el efecto puntual pero intenso de las extracciones de áridos, que incrementan localmente el transporte de sedimentos llegando a concentraciones que habitualmente solo se registran durante crecidas (i.e. concentraciones de hasta 6 g/l y cargas totales por encima de 17 Mg/día). En las zonas de extracción el tamaño de los sedimentos transportados es mayor que en condiciones naturales. La escasa competencia de los caudales durante las extracciones, condiciona el transporte de sedimentos y facilita una rápida sedimentación de las partículas en el lecho, aumentando su disponibilidad para crecidas posteriores. Las observaciones en el Cinca permiten también afirmar que los macroinvertebrados responden directamente al incremento de las concentraciones aguas abajo de las actuaciones. La duración de altas concentraciones de sedimentos y el mayor tamaño de las partículas transportadas influencian el número de macroinvertebrados que entran en deriva. Esta respuesta varía en función de los grupos taxonómicos analizados, con respuestas positivas en los grupos Ephemeroptera, Plecoptera y Trichoptera y respuestas negativas en los Chironomidae. Los resultados de la tesis no permiten definir las causas últimas de dicha deriva pero abren una línea de investigación para trabajos futuros. El papel de las extracciones de áridos y las alteraciones del caudal como factores de homogeneización del hábitat, y sus efectos sobre la diversidad de las comunidades de macroinvertebrados, han sido también examinados en esta tesis. Dichos efectos se han evaluado mediante la integración de métodos y técnicas de muestreo que proporcionan una alta resolución en la obtención de información en terreno; por ejemplo, la generación de información topográfica de alta resolución y en continuidad permite caracterizar la estructura del hábitat físico de un cauce a diferentes escalas espaciales. En general, los lugares con una heterogeneidad de hábitat y complejidad superficial menores fueron los que menos abundancia y taxones de macroinvertebrados presentaron. Por otro lado, aunque las condiciones del hábitat cambiaron a lo largo de las escalas estudiadas (desde 1 m^2 a 16 m^2), los resultados indican que, en conjunto, en el tramo de estudio del Cinca los macroinvertebrados muestreados están más influenciados por el hábitat local (i.e. escala de 1 m^2) que por las condiciones físicas generales del entorno fluvial. Globalmente esta tesis describe diferentes niveles de interrelación entre procesos físicos y biológicos fundamentales en sistemas fluviales desencadenados por diversos factores naturales y antrópicos. La identificación de las escalas de interacción así como el uso de nuevas técnicas de muestreo han demostrado ser relevantes para el estudio de la relación especie-hábitat y constituyen, por ello, un nuevo ejemplo de integración eco-geomorfológica que contribuye al conocimiento de los sistemas fluviales y apoya la gestión sostenible de los mismos.

Palabras clave: Transporte sedimentos en suspensión, extracciones de áridos, hidropuntas, aportación hídrica, afluentes, distribución granulométrica, disponibilidad de sedimento, deriva de macroinvertebrados, diversidad de hábitat, complejidad superficial del lecho, río Cinca, cuenca del Ebro.

CHAPTER 1

INTRODUCTION

1.1. BACKGROUND AND RATIONALE OF THE THESIS

1.1.1. Rivers: water, sediments and biota

River basins are organized as channel networks that enable the transfer of water and sediments from the headwater to downstream depositional zones. This transfer creates morphological patterns along their course. Graded rivers exist in a quasi-equilibrium state, adjusting flows and sediment loads (Lane et al., 1995). The balance consists of a dynamic equilibrium between impelling and resisting forces that generate erosional and depositional process in the fluvial system. Sediment acts as a resisting force and water flow acts as impelling force; thus, either excess flow or reduced sediment load drive the equilibrium to erosion and incision, whereas excess sediment or reduced flows tilt the balance towards deposition (Fryris and Brierley, 2013). Sediment transport is a continuos process through the basin and it is the result of climate, vegetation, litology and biological process in the basin. Physical and biological processes are inseparable in river systems, where an extraordinary diversity of benthic organisms comprising from the micro (i.e. unicellular organisms such as bacteria) to the macro scale (i.e. fishes, mammals) live and reproduce. River form and sediment create and maintain a variety of instream habitats that support biodiversity (Batalla and Vericat, 2013). Benthic organisms continuously experience forces of flowing water (i.e. Hart and Finelli, 1999), but are also influenced in a variety of ways by the structure of river channel, formed by sediment of different sizes, shapes, textures and geological composition (Hynes, 1970).

Among aquatic organisms, benthic macroinvertebrates have a key role in fluvial ecosystems. The term “macroinvertebrates” refers to aquatic invertebrate fauna larger than 500 µm. Generally this macroinvertebrate assemblage consists of many species from numerous phyla. Its diversity has been studied extensively over several decades, since: (a) macroinvertebrates are important in aquatic food chains, as they link organic matter resources (e.g. leaf litter, algae, detritus) and fishes; and (b) their ubiquity across many different types of aquatic habitats (e.g. Morse et al., 1980; Roy et al., 2003; Hose et al., 2004) makes them an important element of diversity. Some macroinvertebrates live in fast moving streams, consuming leaves and detritus that fall into the water. Others live in shallow waters, scraping algae off rocks or on the surfaces of large aquatic plants. Many are predators, or prey upon other macroinvertebrates, while others capture food that is drifting along in the current. Some live on or close to the bed surface, while others occur in the subsurface (‘hyporheic’) zone. Whatever their trophic or habitat preference, macroinvertebrates represent an important food source for fish and other predators such as birds (Wallace and Webster, 1996).

Despite the acquired knowledge of the ecology of macroinvertebrates, there is still an enormous unresolved questions and, consequently, benthic invertebrates continue to be a central part of stream ecological studies (e.g. Hauer et al., 2000; Boyero and Bailey 2001; Lamouroux et al., 2004). Macroinvertebrates invaded the freshwater environment many different times and in many different ways (Resh and Solem, 1996). The mechanisms developed to survive in river flow conditions (i.e. hydraulic forces) involve a variety of different approaches and morphological adaptations (e.g. gills or respiratory pigments to obtain oxygen from the atmosphere or hooks to adhere to the substrate). Their presence in freshwater environments is used to indicate water quality since invertebrates tolerate different stream conditions and levels of pollution. For example, most larvae of caddisflies, mayflies, and stoneflies cannot survive in polluted water and, consequently, streams with these macroinvertebrates are assumed to have good water quality (Clements et al., 2000; Hauer and Resh, 2007). The absence of these organisms in a water body, however, does not necessarily indicate that the water quality is poor since other factors, such as temperature, substrate and flow hydraulics (i.e. altogether representing the river's physical habitat), also play a role.

1.1.2. The quality of water bodies within the EU Water Framework Directive

In the year 2000, as a result of the European Commission's concern over the status of continental waters, the Water Framework Directive (2000/60/CE, hereafter WFD) was enacted. One of the main goals was to establish a framework to protect water and its associated aquatic ecosystems by requiring the Member States to achieve "good status" in all their water bodies. Good status is defined as the quality in terms of: (a) water chemistry, (b) river's hydromorphology and (c) biological community:

(a) <u>Good chemical status</u> is established in terms of all the quality standards for chemical substances at the European level (i.e. more than 40 pollutants of high concern across Europe). This will ensure at least a minimum chemical quality, particularly in relation to very toxic substances, across all European Union member states.

(b) <u>Good hydromorphological status</u> is defined by three properties: (i) hydrological regime, (ii) river continuity and (iii) connection to groundwater bodies. A high proportion of European water bodies were identified as being at risk because of alterations to their structural characteristics (i.e. their morphological characteristics) and the associated impacts on their water flow and level regimes (i.e. their hydrological characteristics). Numerous authors have developed and implemented indices and characterisation protocols (reviewed relatively recently by Fernández et al., 2011 and Belletti et al., 2015). Hydromorphological assessment was first seen to be synonymous with physical habitat surveys (i.e. river habitat index in Pardo et al., 2002; Habitat Condition Index in Oliveira and Cortes, 2005). Nevertheless, over the last few years a trend emerged to understand river functioning and evolution as a basis for interpreting current hydromorphological conditions. For instance the hydrogeomorphological assessment index (i.e. IHG; Ollero et al., 2011) evaluates not only channel quality (e.g. riverbed continuity, riverbank naturalness) but also functional quality (e.g. flow regime naturalness, sediment supply and mobility). Similarly, Rinaldi et al. (2015) developed a methodological framework (i.e. IDRAIM methodological structure) for the hydromorphological assessment of rivers based on past channel evolution, present river conditions, and future trends. This trends lead to a better understanding of physical process resulting in the hydromorphological status of the river system.

(c) The <u>good quality of the biological community</u> is based on fish, invertebrates and aquatic flora. However, the majority of the monitoring programs have been based on macroinvertebrates (Feio and Poquet, 2001; Dallas, 2012). Several metrics have been proposed for the assessment of the ecological status using benthic invertebrates. For instance, the River Invertebrate Prediction And Classification System (RIVPACS, Wright et al., 1984) and the MEDiterranean Prediction And Classification System (MEDPACS, Poquet et al., 2009) consist of statistical tools that follow the concept of the Reference Condition Approach (Reynoldson et al., 1997; Bailey et al., 2004). This reference community is predicted from a set of environmental features and compared with the community of the assessed site (i.e. test site). Thus, the quality of a test site is calculated as the deviation of its community from the reference condition. Multi-metric approaches have also been used in which scores of different assemblages (i.e. invertebrates, fish, macrophytes, algae) are integrated to reflect the good quality of the biological (Santucci et al., 2011). Clarke (2009) highlighted that, although different indices and classification systems are available, there is a lack of coherence between ecological indices and status classification systems. This lack is also shown between the three elements of good status (chemical, hydromorphological and biological aspects). In response to this need, new methods have been developed (e.g. the REstoring rivers FOR effective catchment Management project, REFORM) to provide a framework for improving the success of hydromorphological restoration programmes to reach targeted ecological status. Nevertheless the WFD highlights the importance of the process interaction (physical and biological) for river management.

1.1.3. Physical habitat heterogeneity

Exogenous factors control the occurrence and abundance of organisms through their influence on ecological niches (e.g. the presence of a suitable habitat). Within a catchment, geology and climate directly affect hydrologic patterns, and the movement and storage of inorganic and organic materials. Nutrients and the downstream transport of solutes are affected by channel substratum complexity and the interaction of ground and surface waters. Interactions between the stream channel, hyporheic zone, and riparian floodplains are important features in the structure and function of the stream corridor (e.g. Ward, 1997; Stanford et al., 2005). These and many other factors affect the physical habitat and therefore the distribution and abundance of stream macroinvertebrates at the catchment scale.

At the reach scale, channel morphology dictates the physical habitat variables which together represent the instream habitat conditions used by biota (e.g. Wood, 2008). The ability of channels to erode and deposit sediments, to form pools, bars and other units or channel forms, and the interaction of the flow regime with these variable forms helps support the high ecological diversity observed in fluvial systems (Leopold et al., 1964; Ward and Stanford, 1995; Poff et al., 1997). Hence, geomorphic characteristics and dynamics are key factors controlling physical habitat conditions which are crucial to maintaining stream ecological diversity.

Geomorphic dynamics (i.e. bed-material mobility, sediment transport, river forms) and the diversity of physical habitats are driven by hydrological perturbations that periodically alter habitats or sometimes create a new habitat template, to which the species will adapt. Directional changes in habitat (i.e. moving to a new/alternative quasi equilibrium) may eventually create a new ecological status. In natural systems, fluvial dynamics will be controlled by flood regime, specifically by flow competence (i.e. flows exceed channel disturbance thresholds), and sediment supply, availability and transport. Regimes are not dictated just at the reach-scale, are mainly influenced by processes operating at the catchment scale (e.g. runoff generation, sediment production, connectivity), whereas dynamics propagates downstream as well. Additionally, nowadays, a high proportion of water bodies are human impacted (i.e. changes in land use, gravel mining, hydropower schemes and large dams, channel embankments and canalisations, etc.). These human-induced impacts represent an additional non-natural stressor which modifies physical and ecological processes and dynamics (Reice et al., 1990; Wyzga et al., 2013). When human-induced impacts are located on mainstem reaches, tributaries play a key role in the downstream recovery (i.e. Pattinson et al., 2014; Rice, 2017)

Habitat heterogeneity affects a variety of ecological aspects, such as species interactions, dispersal and specific ecologic functions (Ziv, 1998). Positive correlation between species diversity and habitat heterogeneity are reported in a range of ecosystems (i.e. freshwater in Wheaton et al. (2004); marine in Carvalho and Barros (2017) and terrestrial in Tews et al. (2004)). Thus, habitat heterogeneity is considered essential for maintaining biodiversity. Under human-induced impacts typically complex patterns are often transformed towards simple channels alternating habitat heterogeneity in the fluvial system (Kondolf, 1997).

Within the current context of increasing pressures on water bodies, understanding geomorphic perturbations on rivers is an inter-disciplinary task that requires a comprehensive understanding of the interactions between physical and biological processes, their evolution and drivers. Examples of observational studies integrating ecology and fluvial geomorphology date back to at least the early 1970s (e.g. Hynes, 1970). Stream ecologists have long viewed fluvial geomorphology as a key determinant of benthic assemblages where biota have adapted life-cycles to hydrological regime (e.g. Downes et al., 1998; Buendia et al., 2013), but also where benthic communities influence physical habitat by bioturbation (reviewed in Statzner, 2012). Such interdisciplinary work is fundamental in human modified catchments in order to assess the physical processes leading to the functional and compositional impairment of stream ecosystems and, in turn, to provide the basis for river basin management projects (Stanford, 2006; Wohl et al., 2015).

Within this inter-disciplinary and multi-scale framework, the present thesis focuses on the study of the degree to which human activities may alter the transfer of water and sediments (especially suspended sediments), and the ecological implications both this and associated changes in habitat heterogeneity. Instream gravel abstraction (‘mining’) alters many aspects of river bed sediment characteristics and transport dynamics, as well as alters benthic habitat across a variety of spatial and temporal scales, yet its ecological impacts have been less well studied than, for instance, impoundment, flow regulation or channelisation. To help address this knowledge gap, this thesis deals with the physical and ecological implications of in-channel gravel mining, assessing impacts at different temporal and spatial scales (from hourly to yearly and from sub-metric to kilometric scale).

1.2. AIM, OBJECTIVES AND HYPOTHESIS

The main aim of the thesis is to provide new insights into the interactions between physical and biological processes driven by human and natural-based disturbances at multi spatial and temporal scales in montane fluvial systems. The following hypotheses that are tested in the thesis:

- **H1**. Natural dynamics and human-based impacts influence water and suspended sediment transport, altering flow regime and modifying sediment availability, supply and transfer.
- **H2**. In-channel gravel mining activities increase suspended sediment transport locally, but transport is tightly controlled by grain-size and particles settle quickly, increasing sediment availability downstream from perturbed river reaches.
- **H3**. Invertebrates respond directly to suspended sediment concentrations by entering the water column and drifting downstream, but the duration in which concentrations are high and the size of the transported particles will also influence on drift.
- **H4**. Physical habitat heterogeneity and surface complexity affect invertebrate diversity positively (and therefore by inference, reductions in these will reduce diversity). Moreover, it is hypothesised that the effects of heterogeneity transcend different scales: benthic assemblages do not respond only to their immediate (local) habitat but also are influenced by the physical properties at larger spatial scales.

Based on this aim and these hypotheses, this thesis addresses the following specific objectives:

- **O1**. Study suspended sediment transport dynamics in a river reach subjected to a) natural dynamics i.e. floods and inflow from tributaries, and b) human impacts i.e. in-channel gravel mining and hydropeaking.
- **O2**. Analyse how in-channel gravel mining activities affect suspended sediment transport during mining operations.
- **O3**. Assess whether invertebrate drift increases in response to the increases in suspended sediment associated with in-channel gravel mining.
- **O4**. Evaluate the degree in which habitat heterogeneity and river bed surface complexity controls the invertebrate community composition and diversity.

1.3. FRAMEWORK OF THE THESIS

Figure 1 shows the structure of the thesis with the different chapters (a total of 7), associated objectives, hypothesis and journals where papers have been published or are in the process of publication. The main objectives of the thesis form the focal chapters (from Chapter 3 to Chapter 6 inclusive), with each of these corresponding to a scientific paper. Following the regulation of the University of Lleida (*Article 28 of the Normativa acadèmica de doctorat de la Universitat de Lleida*), each of the focal chapters can be considered as a self-contained unit, with an introduction section outlining the specific research context and objectives of the chapter, the full details of the methods used, the results and discussion. The present chapter represents a general introduction of the thesis, with Chapter 2 then providing details of the study area and an overview of the approaches and methods used for respective chapters. The final chapter (Chapter 7) is a synthesis that integrates the results from each of the individual chapters/papers, presenting a comprehensive discussion that subsequently allows general conclusions do be drawn in relation to each of the objectives of the work. Finally, several annexes are included containing relevant information obtained during the thesis. Amongst them we present useful data to obtain flow from water depth and full details of the monitoring sections and sampling sections (Annex A and B). Finally, we also present two conference proceedings chapters (Annex C) that were undertaken in the background of the PhD and that provide a more extended description of the thesis framework.

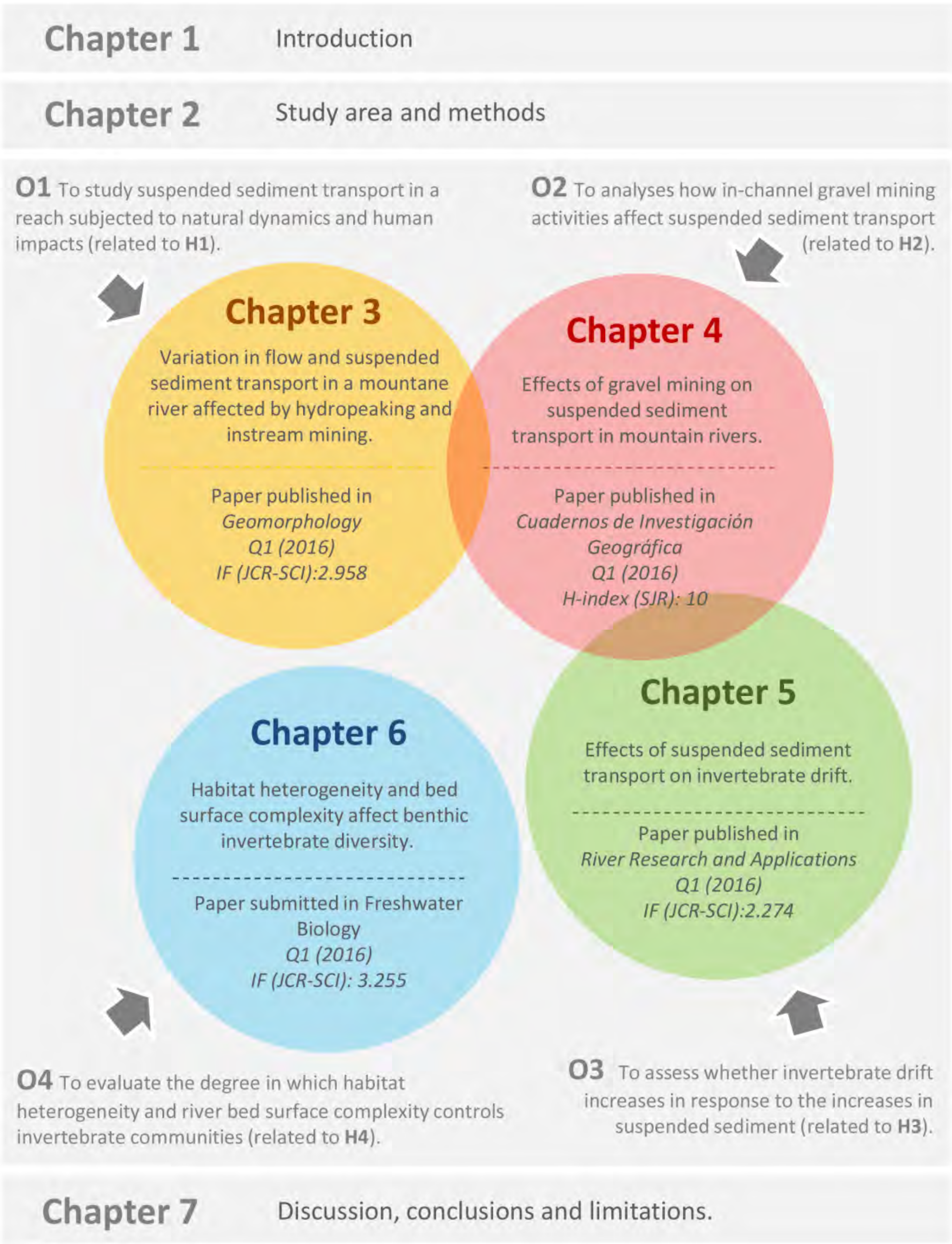

Figure 1. Structure of the thesis with the different chapters, associated objectives, hypothesis and journals where papers have been published or will be submitted

This thesis is directly linked to the research projects i) *MORPHSED: Morphosedimentary Dynamics In Human-Stressed Fluvial Systems: Coupling Channel Morphology And Ecological Diversity* (CGL2012-36394, www.morphsed.es) and ii) *MORPHPEAK: Morpho-Sedimentary Dynamics In Mountain Rivers Affected By Hydro-Peaking: Effects On Habitat And Implications For Management* (CGL2016-78874-R) funded by the Spanish Ministry of Economy and Competiveness and the European Regional Development Fund Scheme. The main aim of these projects was to analyse the morphosedimentary dynamics of a representative human-stressed fluvial system suffering major local physical habitat alterations (due to in-channel gravel mining and hydropeaking), their drivers and their impacts on the river's ecological integrity.

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CHAPTER 2

STUDY AREA AND METHODS

1. STUDY AREA

1.1. Location

The River Cinca is located in the southern part of the Central Pyrenees, in the NE of the Iberian Peninsula (Figure 1A). It is the second largest river flowing into the Ebro, draining an area of 9,740 km^2 (equivalent to 11 % of the Ebro basin area; Figure 1B). The study reach is located in the upper part of the Cinca (Figure 1C), where elevation ranges from 522 m a.s.l. in Ainsa to 3,355 m a.s.l. in the Monte Perdido, with 1/3 of the catchment area located above 2,000 m a.s.l. (Figure 1D). The catchment area of the study reach is 849 km^2.

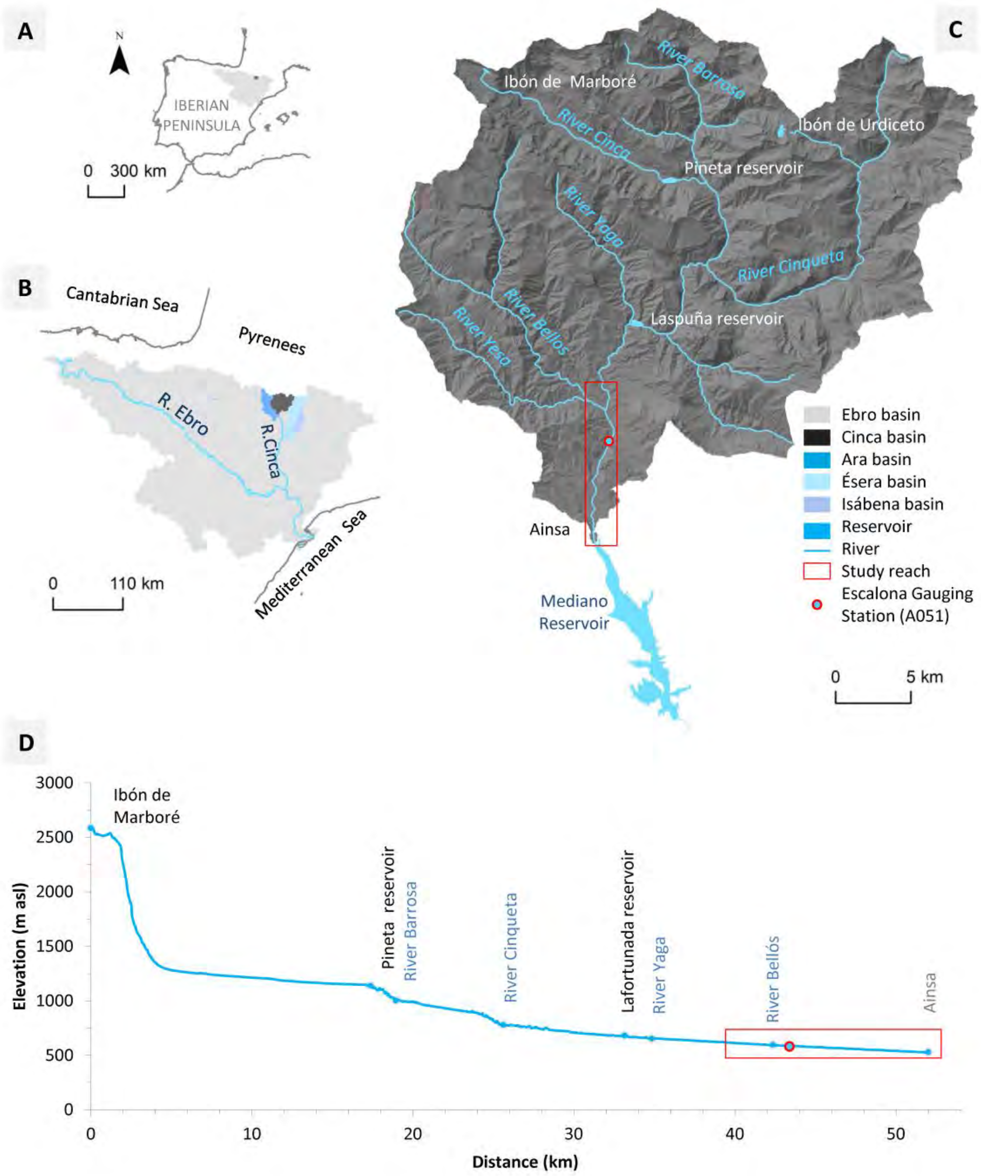


Figure 1. Location of the upper Cinca catchment within the Iberian Peninsula (A) and the Ebro basin (B). (C) The upper Cinca, indicating the location of the main streams, reservoirs, the official gauging station in the study reach (A051 Escalona) and the main ibones (i.e. Pyrenean glacier lakes). (D) Longitudinal profile of the upper River Cinca from the Ibón de Marboré to Ainsa.

1.2. Climate, hydrology and geology

The Cinca is located within the Continental Mediterranean climatic domain, which is characterized by a large contrast between winter and summer. Winter tends to be cold and dry, while summer can be warm and very stormy, with a high number of torrential rainfall events (Rubio, 1995). Annual precipitation ranges from 430 mm in the lower areas to almost 2960 mm in the summits (Figure 2A). Climatically, the study area can be divided into two main units: one in the southern part of the catchment, characterized as being dryer and warmer; and the other one to the north, with more frequent rainfall events and lower temperatures (Rubio, 1995). Buendia et al. (2016) examined trends in temperature and precipitation regimes in the Central Pyrenees, and reported a decrease in precipitation and upward trend in temperature over the last decades. Mean precipitation and temperature and their temporal variability were assessed in Ainsa (located in the dryer and warmer part) from the period 1962-2010 (data source: Aragon Statistics Institute, 2017). The annual precipitation in Ainsa averages 852 mm (coefficient of variation, CV = 0.2), with maximum mean monthly values of 95 and 98 mm in May and October, respectively (Figure 3). Minimum monthly mean values of 47.5 and 49.5 mm are observed in July and February, respectively. Note that the minimum monthly value is 10 mm in June, indicating frequent rainfall events in this month across the last decades. The basin shows a clear temperature gradient, so that the mean annual temperature varies between 5 °C in the north to 13.5 °C in the valley bottom (i.e. Ainsa, Figure 2B). Annually, mean temperature ranges between 9 and 16°C in Ainsa (for the period 1956-2010). The maximum values of temperature are reached in July and August (i.e. 27.8 and 27.0 °C, respectively) while minimum values are reached in February and January (i.e. -0.8 and -0.6 °C, respectively).

The hydrology of the basin is characterized by a rain-snowy regime with large intra-annual variability, leading to high variations in flow (Figure 4). There is a seasonal variation in the occurrence of floods, which typically occur in spring due to snowmelt and, especially, in the late summer and autumn as a consequence of localized thunderstorms. The mean annual discharge at the official Ebro Water Authority Escalona gauging station (only historical discharge records are available at A051 Escalona, data source: Automatic Hydrologic Information System-Ebro Water Authority, 2017) for the period of record 1959-2014 is 27 m^3/s (inter-annual CV = 0.38 and intra-annual CV = 0.42). Annual floods exceed 220 m^3/s and high magnitude floods (i.e. floods with a recurrence interval of 10 years) exceed 750 m^3/s. Minimum flows occur in summer and winter but the river never dries up. Mean annual runoff is 984 mm (CV = 0.4). Despite some missing data (i.e. missing data from 1992-2003) and the inherent variability, annual runoff shows a decreasing trend over the last decades in the upper Cinca and the nearby catchments (Figure 5A). This trend has been extensively reported in other Iberian catchments (e.g. Garcia-Ruiz et al., 2001; Gallart et al., 2011; Garcia-Ruiz et al., 2011; Lorenzo-Lacruz et al., 2012; Buendia et al., 2016). Nevertheless, the runoff reduction in the upper Cinca is higher than the ones in the nearby Ara (its main tributary), the Ésera and the Isábena. Both, the upper Cinca and its tributary the Ara show a higher reduction than the Ésera and its main tributary, the Isábena, suggesting different behaviours of these catchments (Figure 5B). Overall, afforestation (Gallart and Llorens, 2003; Beguería et al., 2006), abandonment of agricultural activities in mountain areas (Grove and Rackham, 2001), higher temperatures and lower precipitation (Milly et al., 2005) likely contribute to changes in runoff (Figure 5C).

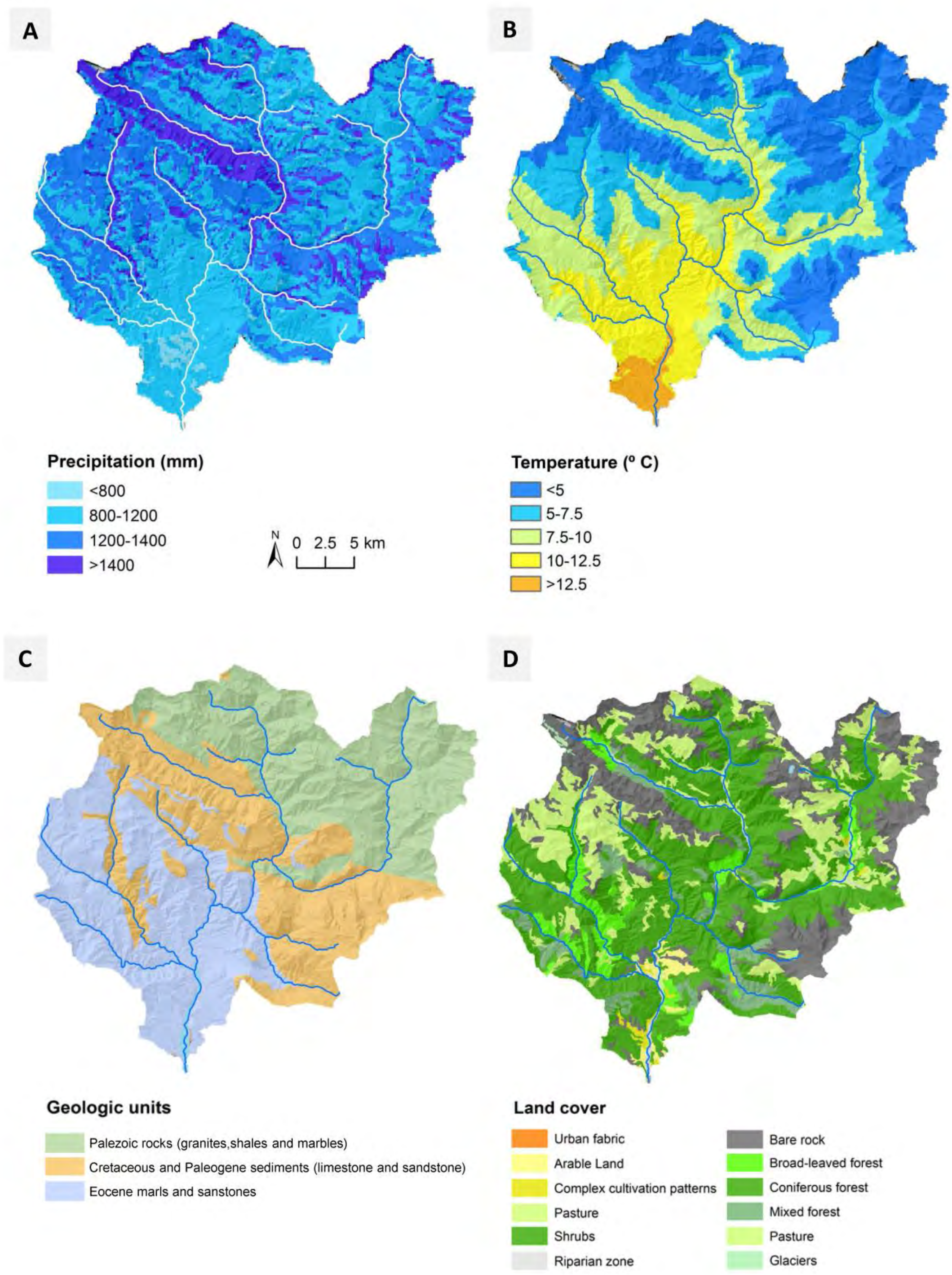


Figure 2. The upper Cinca catchment physical context: Mean annual precipitation (A) and mean annual temperature (B) for the period 1962-2010, data source: Ninyerola et al. (2005); (C) Main geological units, data source: Instituto Geológico y Minero de España and Bureau de Recherches Gèoligiques et Minières (2009); (D) Land cover in 2006, data source: European Environment Agency (2006).

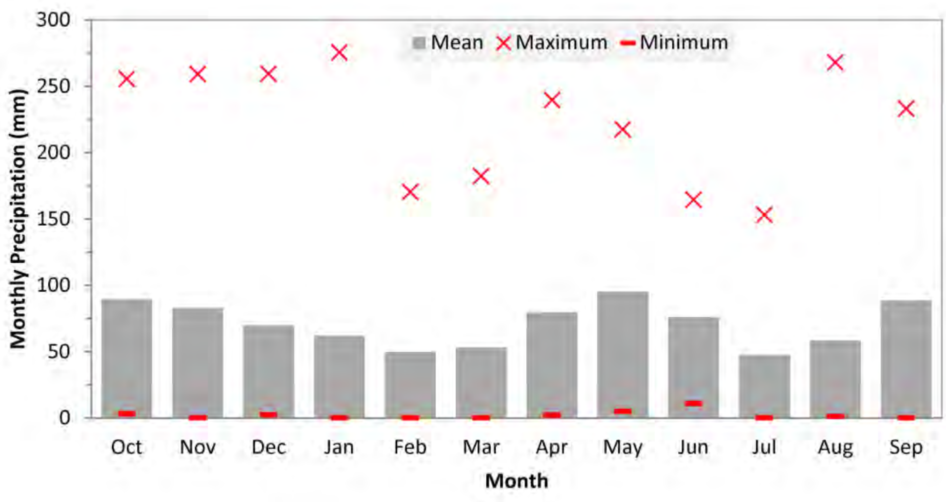


Figure 3. Mean monthly precipitation in Ainsa for the period 1962-2010. Maximum and minimum mean monthly values are represented by red crosses and red slashes respectively. Data source: Aragon Statistics Institute (2017).

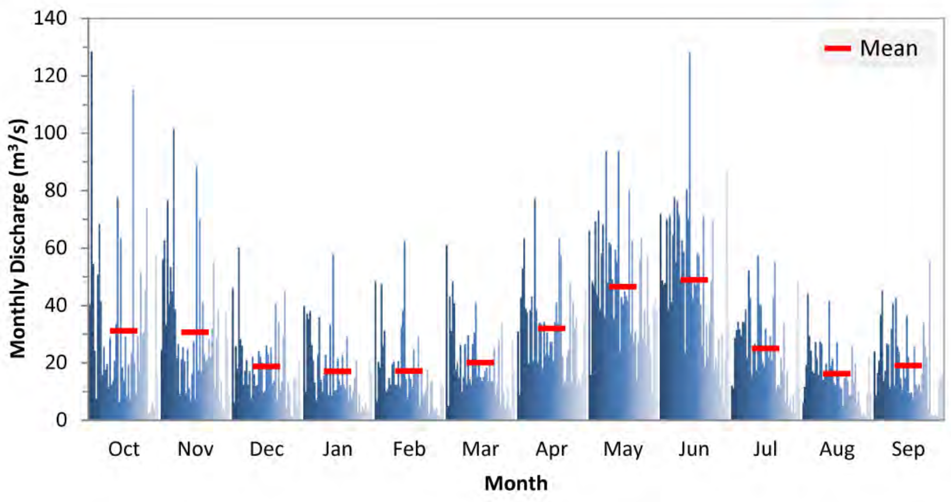


Figure 4. Annual mean monthly discharge in the upper Cinca (Escalona gauging station; A051) for the period 1959-2014, for each month annual values for the data period are shaded different densities, running from the oldest (darkest) to palest (most recent). Red horizontal lines represent mean monthly discharge for the entire period. Data source: Automatic Hydrologic Information System of the Ebro river Basin (2017).

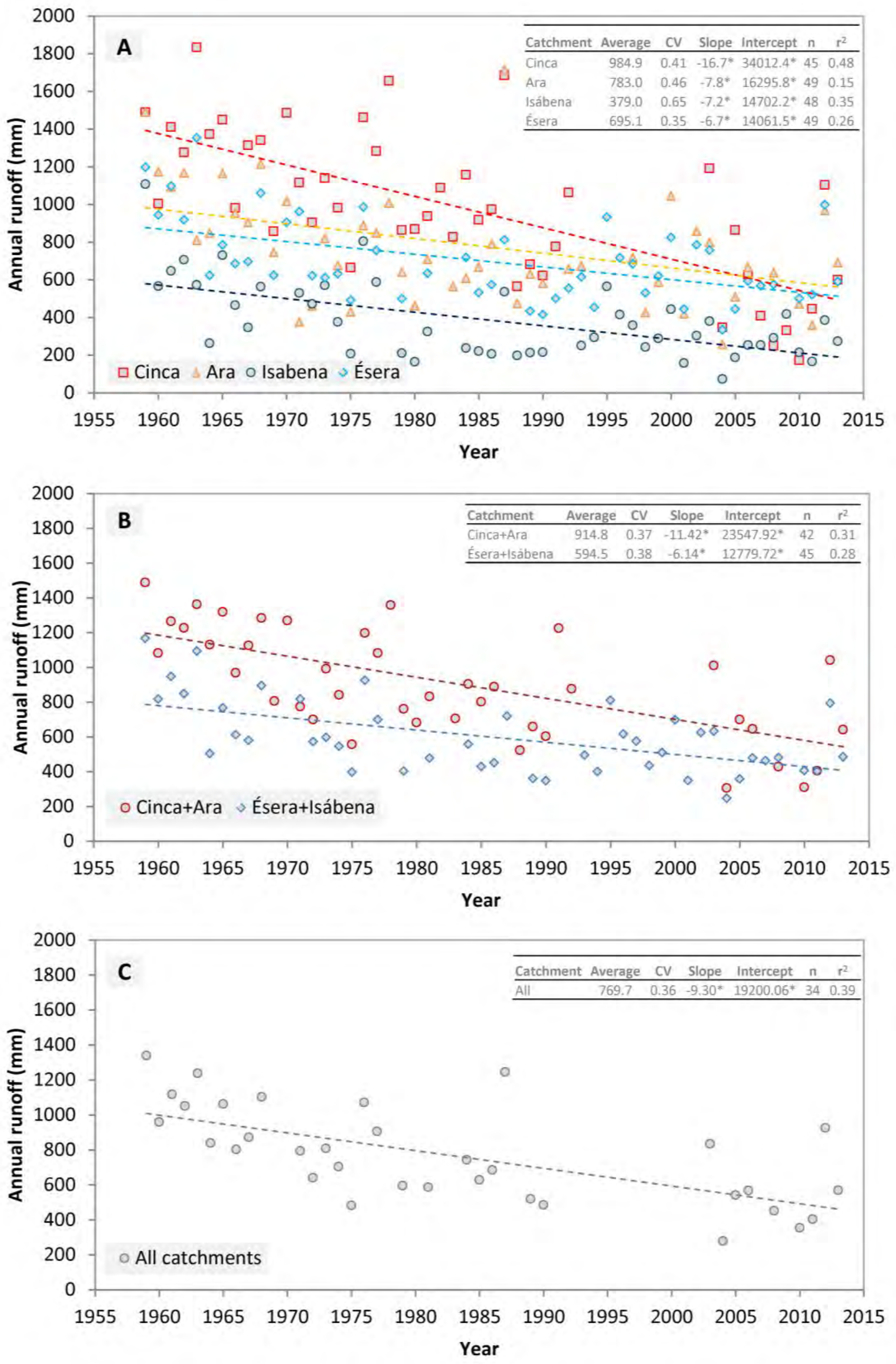


*Figure 5. Mean annual runoff for the period 1959-2014 in: (A) the upper Cinca (A051) and nearby catchments (i.e. the Ésera at the Graus gauging station (A013), the Isábena at the Capella gauging station (A047) and the Ara at the Boltaña gauging station (A040)); (B) the upper Cinca and Ésera catchments; and (C) in all catchments together. The inset tables summarise key statistics for each catchment. See location in Figure 1. Note * indicates p-value < 0.01. Data source: Centro de Estudios y Experimentación de Obras Públicas (2014).*

The geology of the basin encompasses several lithological units arranged in a N-S axis (Ríos et al., 1979): i) the axial zone of the Pyrenees (peaks above 3,000 m a.s.l), composed of Palaeozoic rocks, i.e. metamorphics (shales, marble) and plutonics (granites); ii) the Thrust sheets, which is a large over-thrusting fold of Cretaceous and Paleogene sediments composed mainly of limestone and sandstones; and iii) the Foreland basin (centre and outlet of the upper Cinca basin), which is formed on more erodible materials (mainly Eocene marls and sandstones) giving a relatively smooth relief (Figure 2C). During Quaternary times the fluvial drainage system, fed by glaciers in the headwaters, developed a terrace system in the Cinca valley (Lewis et al., 2009). Successive climate changes and glaciation-deglaciation phases controlled fluvial discharge and sediment availability, thus erosion rates (average estimated= 0.5 m/ 1000 years; Sancho et al., 2004), creating the stepped sequence of strath terraces. The oldest terrace in the upper Cinca (61 x 1000 years, with an error ± 4 x 1000 years, Lewis et al., 2009) is near the village of Ainsa and sits 172 m above the current river active channel.

1.3. Land cover and human activities

More than one third of the catchment area of the upper Cinca is occupied by forest (i.e. ≈40 %; Figure 2D). Forest in the lower part of the catchment (i.e. up to 800 m a.s.l.) is dominated by pines (*Pinus nigra salzmanii)* and oaks (*Quercus humilis* and *Q.subpyrenaica)*. The vegetation between 800 and 1800 m a.s.l. is characterised by *Fagus sylvatica, Abies alba* and *Pinus sylvestris*. Forest in the upper part of the catchment (i.e. 2200 m a.s.l.) is dominated by *Pinus uncinata* and *Betula pendula*. Pastures can only survive the climatic conditions up to 2200 m a.s.l. (where predominant genus is *Festuca sp.*), representing 19 % of the catchment area. During the second half of the 20th century, most cultivated fields and pastures on steep slopes were abandoned and a subsequent natural process of plant recolonization have occurred (e.g. Gallart and Llorens, 2003; Lasanta and Vicente-Serrano, 2007). Currently, arable lands occupy less than 2 % of the area. Other vegetation covers (i.e. transitional woodlands, shrub) and bare rocks comprise 17 and 22 % of the catchment, respectively.

The upper Cinca is a highly regulated river: small reservoirs are used for power generation in the headwaters, whereas the Grado and Mediano reservoirs, located at the outlet of the study area, impound water from the upper Cinca and its west tributary, the River Ara. The high range of elevations is exploited by a system of small reservoirs and tunnels for hydroelectric energy production. Reservoirs are located from 2,591 m a.s.l. in the Ibón of Marboré, to 650 m a.s.l at the Laspuña Reservoir (Figure 6A). During low flow conditions, the Laspuña Reservoir (in operation since 1965 with a water capacity of 0.35 hm^3) impounds water to be pumped to other small reservoirs to be used for hydropower generation. During high flow conditions, wicket gates are left open, allowing water and sediments to pass through the dam. This operation allows the dam not to affect water and sediment transfer in the river (Figure 6B). Nevertheless, the whole hydroelectric scheme affects the fluvial ecosystem. For instance De Jalon et al. (1988) reported a decrease in benthic abundance and diversity downstream the hydropower generation in the upper Cinca. Downstream from Ainsa, the Mediano Reservoir (399 hm^3, in operation since 1970) collects water and sediment from the upper River Cinca and the River Ara for hydropower purposes. Further downstream, there is the Grado Reservoir, built in 1966 (463 hm^3) for irrigation purposes and hydropower generation. The two together produce an Impoundment Ratio of 0.35 (IR calculated using flow data from Fraga, upstream of the confluence with the River Segre, as per Batalla et al., 2004), and represents the 11 % of the total reservoir capacity of the Ebro river basin.

The upper Cinca has a long history of in-channel gravel mining, with the most intense period between 1970 and 2014, at a rate of mean annual extraction ca. 50,000 m^3/year (Llena et al., 2016; note that this value is an estimate based on the data avilable at the files of the Ebro Water Authority). Nowadays, gravel is extracted during channel maintenance works (e.g. works to stabilize channel embankments for flood and prevention of lateral erosion), and so mining activities are often undertaken after high magnitude flood events (Figure 7). For instance, after the catastrophic 1982 flood (i.e. peak discharge, Q_c 1,085 m^3/s) large channel embankments were built in the lower part of the catchment (Figure 7).

In-channel gravel mining and other factors described above have modified the morpho-sedimentary equilibrium of the upper Cinca over the last decades. The resulting colonization after cultivated field and pasture abandonment (Figure 8) contributed to control both soil erosion and surface runoff, resulting in a stabilization of fluvial sedimentary structures (Garcia-Ruiz et al., 1996). Altogether, the upper Cinca has experienced a notable reduction of the active channel width (i.e. up to 50 % in the lower part of the catchment), a simplification in channel pattern (i.e. from braided to wandering), as well as an important channel incision (up to 7 m in places) over the last decades (Llena et al., 2016; Figure 9).

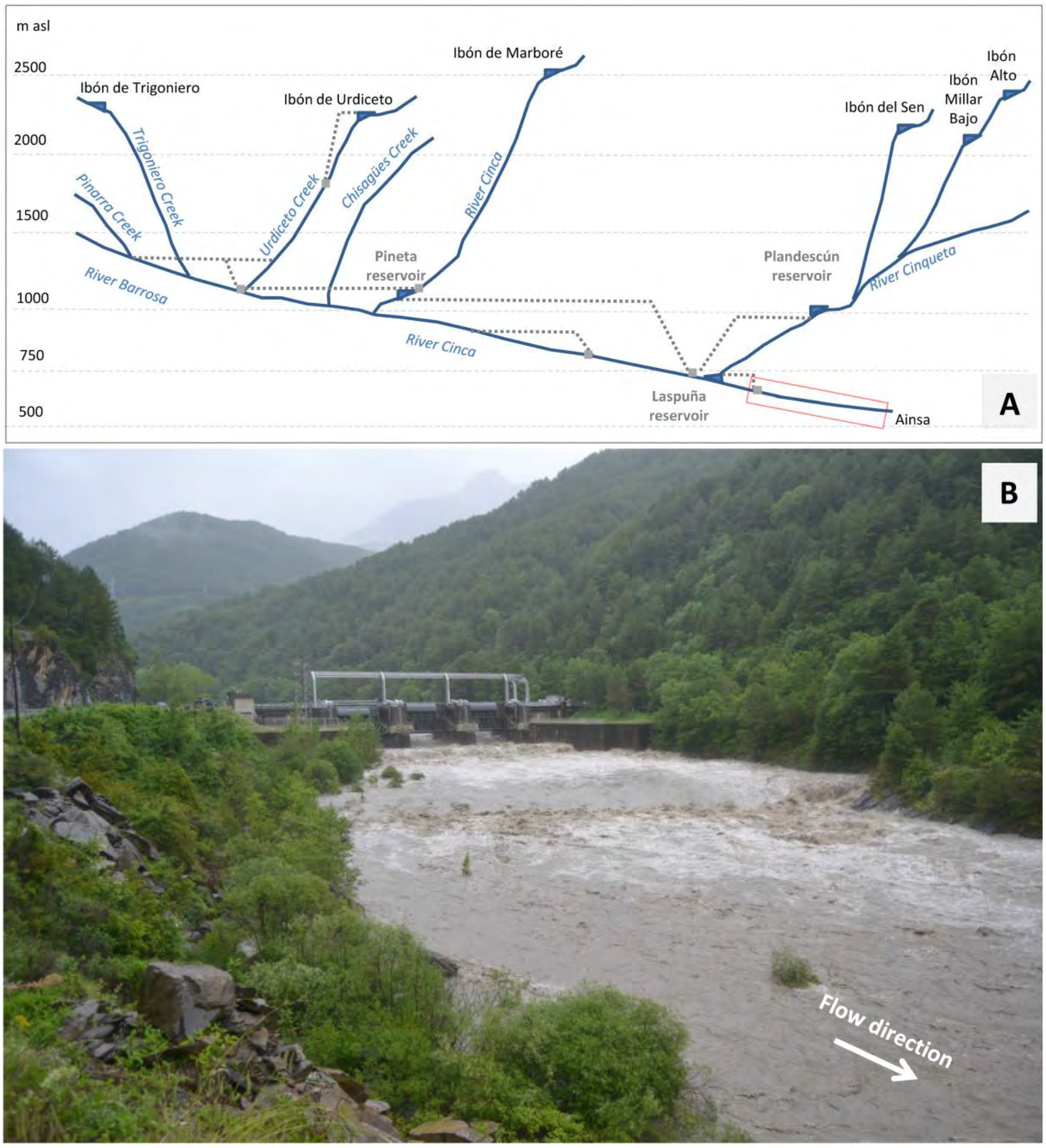


Figure 6. (A) Hydroelectric scheme across the longitudinal profile of the upper River Cinca (modified from J.A. Pérez del Pulgar, 1930). (B) A view of the River Cinca downstream from Laspuña Dam during a flood event in July 2014 (Photograph by Damià Vericat). Note that the wicket gates are left open, allowing water and sediments to pass through the dam.

Figure 7. In-channel gravel mining for flood and lateral erosion prevention at Escalona Village in March 2013 (Damià Vericat).

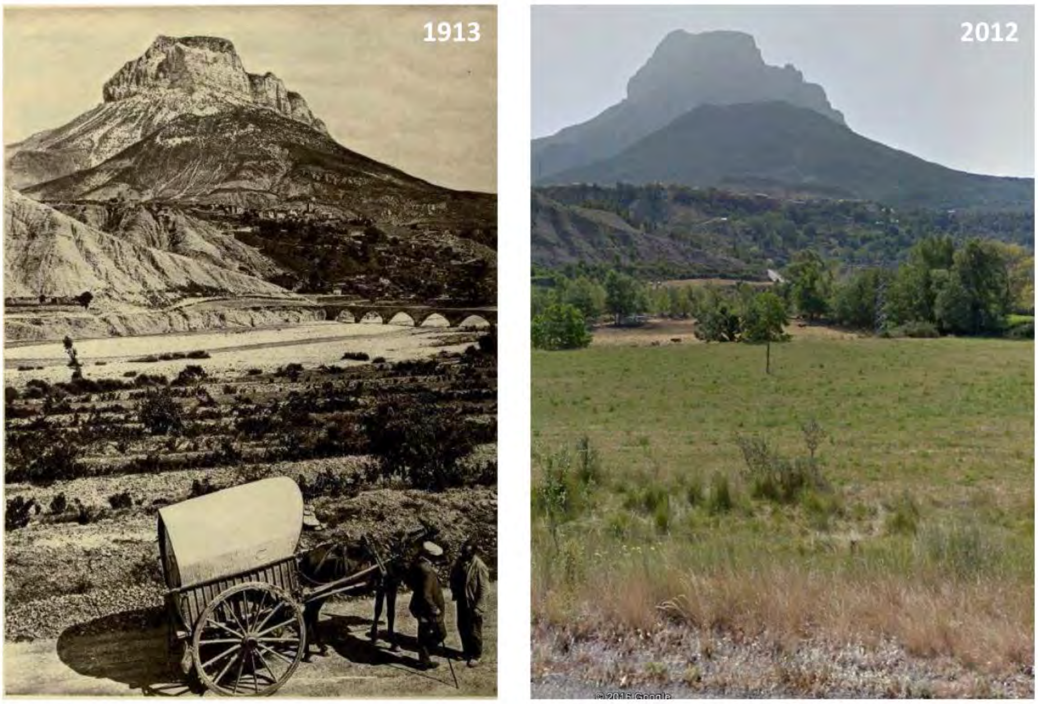


Figure 8. Upper River Cinca at Laspuña Bridge in 1913 and 2012. Data source: http://www.nabatiando.com/ and Google Maps (accessed on November 2017)

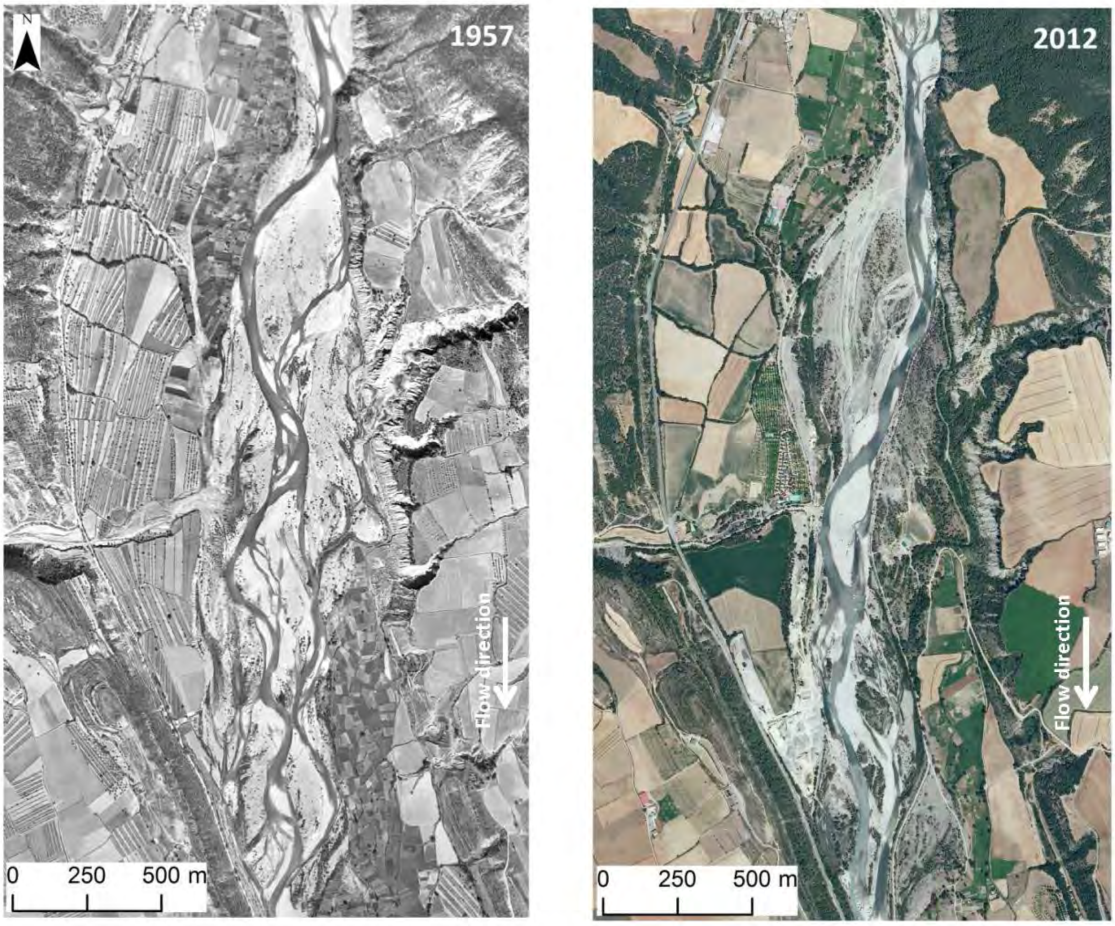


Figure 9. Aerial images from 1957 and 2012 of the upper Cinca near the village of Ainsa. Images illustrate the historical morphological changes experienced by the study reach see more details in Llena, 2016): channel pattern simplification, land cover changes and lateral disconnectivity are shown. Source: Llena et al. (2016).

1.4. River Ecology

The upper Cinca can be divided in two sub-reaches based on environmental conditions: (a) the high mountain reach from the headwaters to Escalona, and (b) the humid calcareous reach from Escalona to Ainsa. Reaches are classified based on altitude, lithology, conductivity, distance from headwater, slope, air temperature and discharge statistics. The lower reach is classified as humid calcareous mountain river based on its slightly higher water conductivity than the upper reach.

As stated in Chapter 1, the ecological status is classified by the biological community and the hydromorphological and chemical characteristics. In the case of the Upper Cinca biological indices were applied since 2000 by the Ebro Water Authorities. Two biological indices were applied: the IBMWP based on macroinvertebrates (Iberian Biological Monitoring Working Party, Alba-Tercedor et al., 2002) and the IPS diatoms index (CEMAGREF, 1982) at three monitoring sites (at the confluence of the River Cinqueta and the River Cinca, near Laspuña and near Escalona, see Figure 1). The most recent survey (conducted in 2005) provided a very good "ecological status" assessment.

Fish assemblages were surveyed in the lower part of the upper Cinca in 2013 between Ainsa and Laspuña as part of the MorphSed project (GESNA, 2014). Six species were found, being the brown trout *(Salmo trutta) the* dominant specie (Figure 10 A). Only early stages of development were found across the river reach, suggesting low densities of adults in the upper Cinca. Invertebrate community structure changes across the upper Cinca due to the high range of physical and climatic conditions. In the headwaters, assemblages are dominated by one or two taxa (i.e. Ephemeroptera and some Plecoptera) while further down a wider range of Ephemeroptera, Diptera, Tricoptera and Plecoptera occur, resulting in a more diverse benthic community (Orós, 2014; Figure 5 B and C).

Figure 10. Ecology in the upper Cinca. (A) Brown trout, Salmo trutta. Author: GESNA (2014). (B) Plecoptera, family Perlidae. (C) Diptera, family Simulidae.

2. SAMPLING DESIGN AND OVERVIEW OF METHODS

As far as we are aware, there are no previous studies focussing on the ecological implications of in-channel gravel mining in the upper Cinca, specifically on issues related to the implications of habitat heterogeneity and suspended sediment transport for benthic invertebrates. Therefore, in order to address the specific objectives of the Thesis (outlined above), a multi-spatial and multi-temporal approach was designed and implemented in a 9.8 km reach between Ainsa and Lapsuña during two hydrological years (October 2014 – September 2016). Figures 11,12,13,14 show the sequence of fieldwork campaigns and the study sites, together with the methodological workflow followed in each chapter of the Thesis. Full details of all these are presented in the respective Thesis chapters. An overview is provided below.

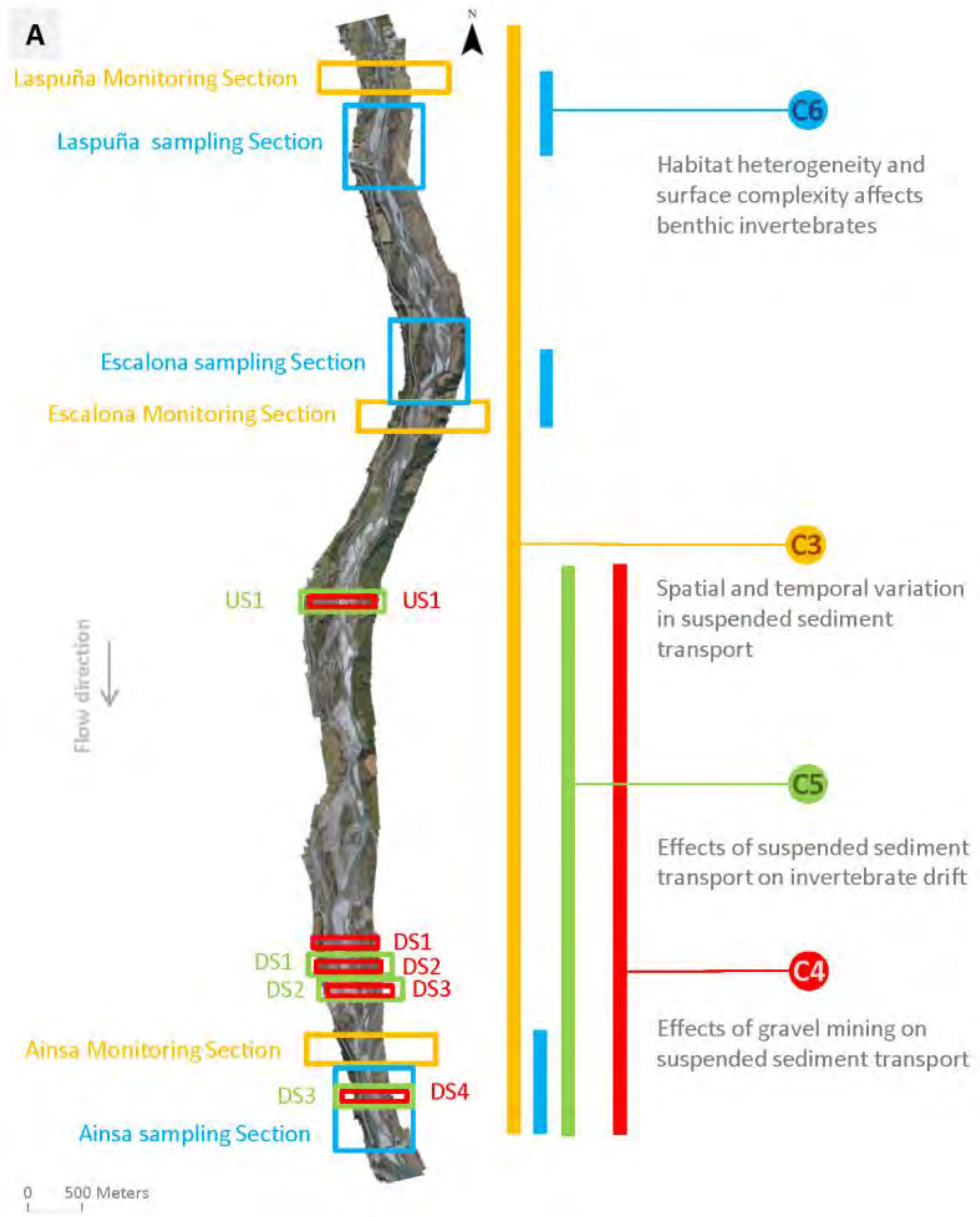
Discharge and suspended sediment transport (**Chapter 3**) were continuously monitored during two hydrological years at three monitoring sections (Figure 11), with the objective of estimating and analysing the water and the sediment loads along the selected river reach. The three sections were selected to cover a range of natural processes and human activities that could affect temporal and spatial variability of water and sediment yield (i.e. basin tributaries, hydropeaking and in-channel gravel mining). These sections were set up as part of MorphSed and are kept in the MorphPeak project. In the case of discharge, although the assessment of the relationships between water depth and flow (d/Q) were not fully developed in this PhD, flow data are used for several of the objectives of the thesis. The method used to obtain flow (discharge) from water depth measurements is summarised in figure A.1 (Annex A) and involved six interrelated tasks: (I) acquiring water depth data, (II) transforming water depth to water surface elevation (WSE), (III) flow gauging (Q), (IV) hydraulic modelling, (V) model performance assessment and (VI) data fusion, elaboration of WSE-Q relationships and discharge transformation. Further details are given figure A.1 in the annexes and also the methods section in Chapter 3.

For **Chapter 4**, discharge, suspended sediment transport and sediment grain size distribution were intensively measured during in-channel gravel mining operations in five monitoring sections along the study reach. Samples were taken in an upstream-downstream basis, in a section upstream from channel mining (i.e. taken as a reference section) and in four sections downstream (impact sections).

During channel mining operations, invertebrate drift responses to the increase in suspended sediment were examined in four monitoring sections (**Chapter 5**). As with Chapter 4, the same reference section (upstream) and three of the four impacted sections (downstream) were monitored.

Finally, in order to assess the potential ecological effects of reduced habitat heterogeneity caused by in-channel gravel mining, the importance physical habitat heterogeneity and surface complexity for invertebrate diversity was assessed (**Chapter 6**). This involved a field campaign within which parallel data on macroinvertebrates and habitat characteristics in 5 × 5 meters areas were collected. The workflow for this chapter is presented in Figure 14.

Annex B shows more details of all monitoring sections and sampling sections.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Ainsa Monitoring Section
DS3
DS4
Ainsa sampling Section
0
500 Meters
C6
Habitat heterogeneity and surface complexity affects benthic invertebrates
C3
Spatial and temporal variation in suspended sediment transport
C5
Effects of suspended sediment transport on invertebrate drift
C4
Effects of gravel mining on suspended sediment transport

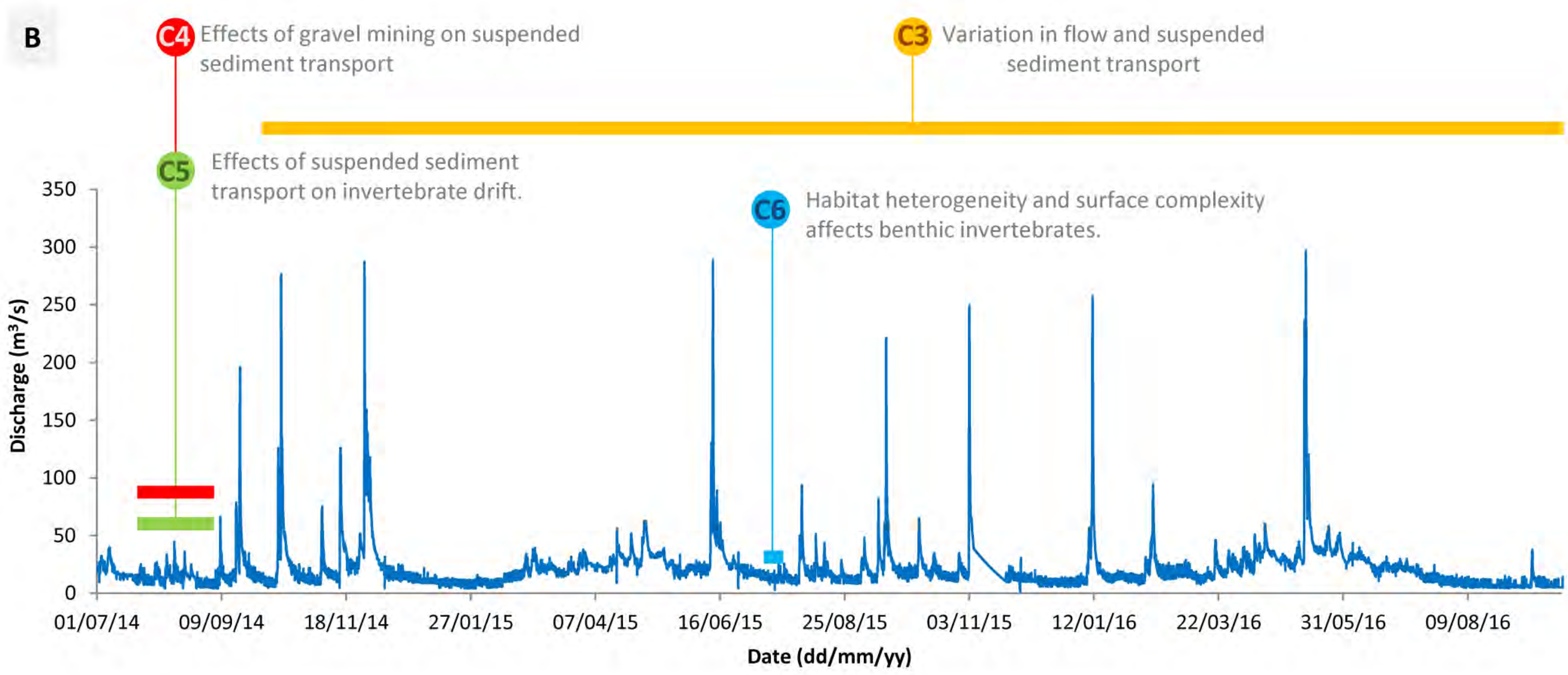

Figure 11. Spatial (A) and temporal (B) scales of analysis in relation to each of the Thesis chapters (see chapter 1 for a full description of this).

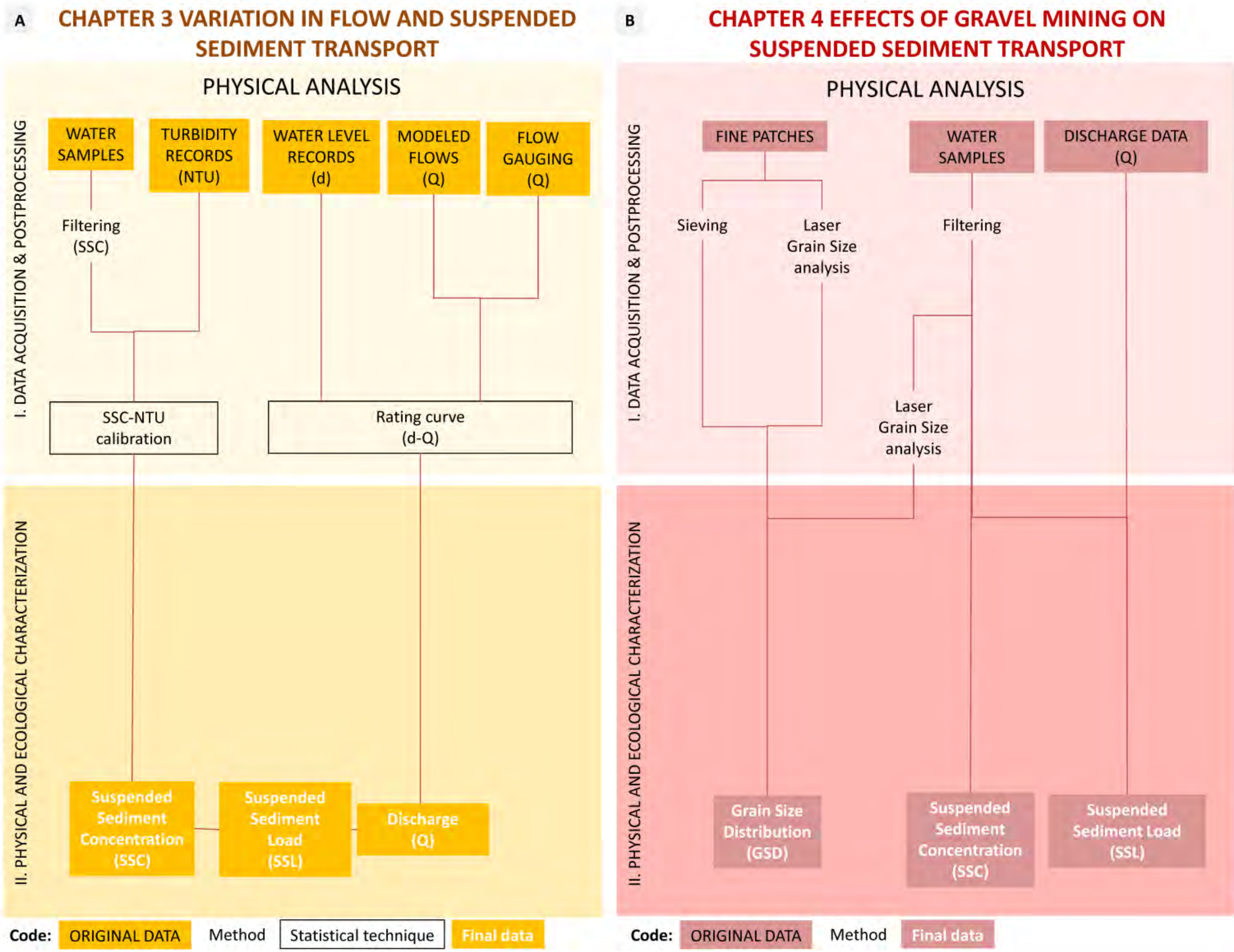

Figure 12. Schematic illustration of the Thesis Workflow to study: (A) The variability of flow and suspended sediment transport (i.e. Chapter 3) ; (B) The effects of in-channel gravel mining on suspended sediment transport (i.e. Chapter 4).

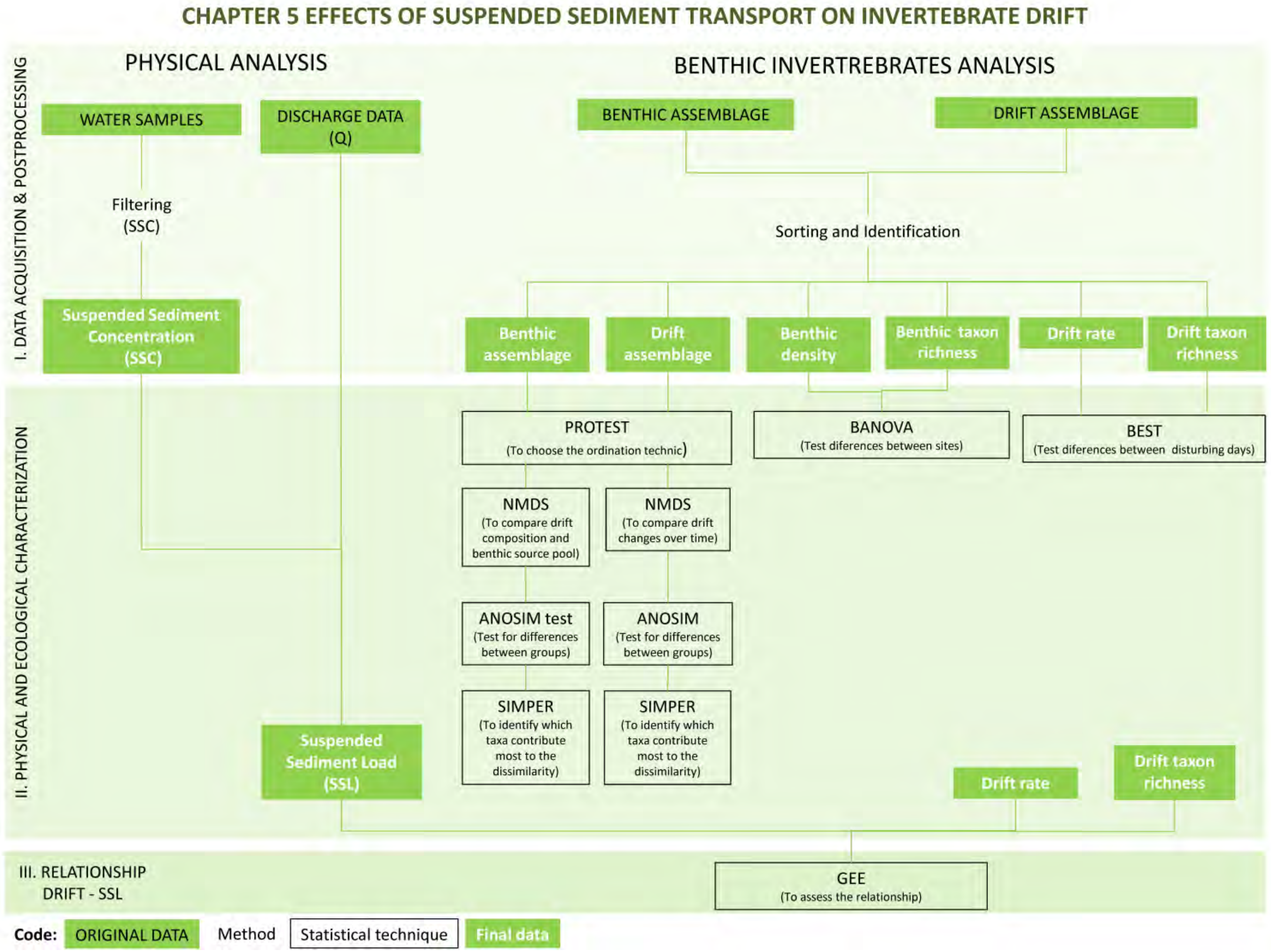

Figure 13. Schematic illustration of the Thesis Workflow to study the effects of suspended sediment transport on invertebrate drift (i.e. Chapter 5).

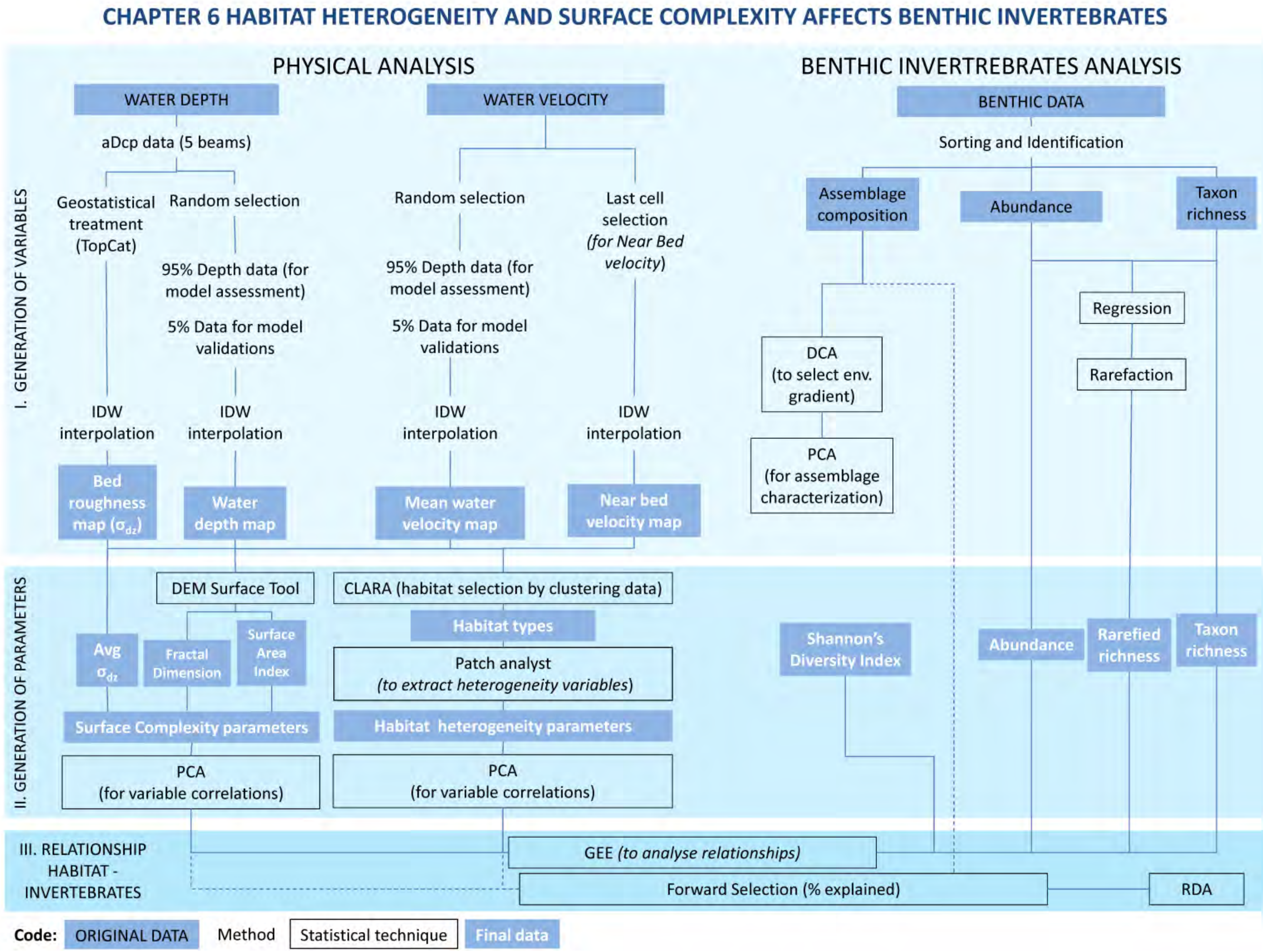

Figure 14. Schematic illustration of the Thesis Workflow to study how habitat heterogeneity and surface complexity affects benthic invertebrates (i.e. Chapter 6).

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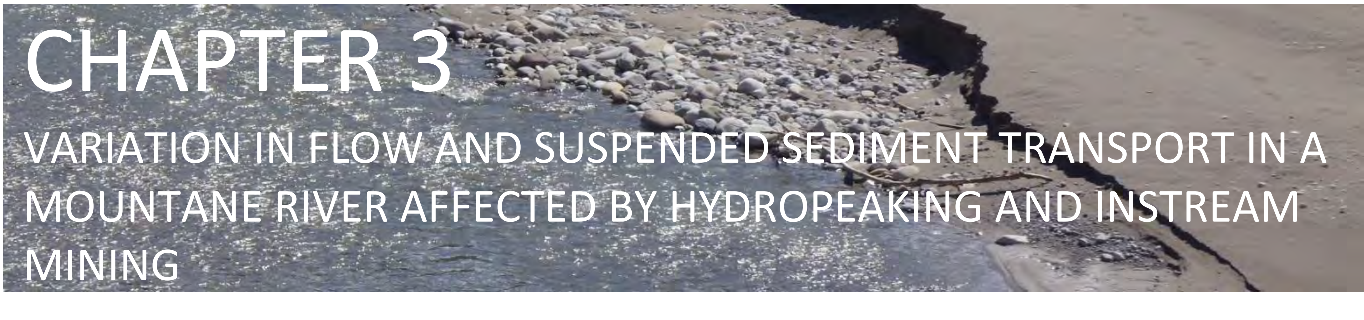

CHAPTER 3

VARIATION IN FLOW AND SUSPENDED SEDIMENT TRANSPORT IN A MOUNTANE RIVER AFFECTED BY HYDROPEAKING AND INSTREAM MINING

VARIATION IN FLOW AND SUSPENDED SEDIMENT TRANSPORT IN A MOUNTANE RIVER AFFECTED BY HYDROPEAKING AND INSTREAM MINING

This chapter contains the following accepted and already online published paper in the journal *Geomorphology.* JCR-SCI Impact Factor: 2.958. Category: Physical Geography; 1st Quartile.

Béjar, M., Vericat, D., Batalla, R.J., Gibbins, C.N., 2018. Variation in flow and suspended sediment transport in a mountainous river affected by hydropeaking and instream mining. DOI: 10.1016/j.geomorph.2018.03.001

ABSTRACT: The temporal and spatial variability of water and sediment loads of rivers is controlled by a suite of factors whose individual effects are often difficult to disentangle. While land use changes and localised human activities such as instream mining and hydropeaking alter water and sediment transfer, tributaries naturally contribute to discharge and sediment load of mainstem rivers, and so may help compensate upstream anthropogenic factors. The work presented here aimed to assess water and the sediment transfer in a river reach affected by gravel extraction and hydropeaking, set against a backdrop of changes to the supply of water and sediment from tributaries. Discharge and suspended sediment transport were monitored during two average hydrological years at three cross-sections along a 10-km reach of the upper River Cinca, in the Southern Pyrenees. Water and sediment loads differed substantially between the reaches. The upper reach showed a largely torrential discharge regime, controlled mainly by floods, and had high but variable water and sediment loads. The middle reach was influenced markedly by hydropeaking and tributary inflows, which increased its annual water yield four-fold. Suspended sediment load in this reach increased by only 25 % compared to upstream, indicating that dilution predominated. In the lowermost section, while discharge remained largely unaltered, sediment load increased appreciably as a result of changes to sediment availability due to instream mining and inputs from tributaries. At the reach scale, snowmelt and summer and autumn thunderstorms were responsible for most of the water yield, while flood flows determined the magnitude and transport of the sediment load. The study highlights that a combination of natural and human factors control the spatial and temporal transfer of water and sediment in river channels and that, depending on their geographic location and effect-size, can result in marked variability even over short downstream distances.

KEY WORDS: Suspended sediment transport, gravel mining, hydropeaking, River Cinca.

1. INTRODUCTION

The historical concept of natural river systems being divided into three zones according to dominant hydro-sedimentary processes (i.e. sediment production, transport and deposition i.e. Schumm, 1977) has been modified of late to account for catchment activities that alter basin and channel dynamics (De Castro et al., 2005; García-Ruiz et al., 2011). Reflecting these alterations, Preciso et al. (2012) introduced the reversed model concept where, mostly due to afforestation and soil conservation practices, catchment headwaters are no longer regarded as the main source of water and sediment. According to this model, in the alluvial plain disequilibrium between the input, transfer and export of sediments occurs as a result of human activities such as damming, water diversion, in-channel mining and channelization which alter the fluxes of water and sediment. Excess energy is dissipated at the expense of the river bed that, as a result, often becomes the main source of sediment. Ultimately, depositional zones such as deltas and estuaries receive less sediment; this, along with the reduced river discharge, leads to erosion becoming the prevailing geomorphic process, as observed in coastal areas worldwide (e.g. Dallas and Barnard, 2011; Randazzo et al., 2013).

Mountain areas are the most important zones for water generation, particularly in Mediterranean regions (e.g. García-Ruiz et al., 2011; López-Moreno et al., 2008a; Lana-Renault et al., 2010). Several studies indicate that precipitation in these regions has decreased and air temperature has increased in recent decades, leading to reduced runoff (Morán-Tejeda et al. 2010). Additionally, in many Mediterranean regions extensive areas of agricultural land on hillslopes were abandoned during the second half of the 20th century, resulting in increased forest cover i.e. re-afforestation (e.g. López-Moreno et al., 2002; Lasanta et al., 2006a; Buendia et al., 2016b). Such land use changes reduce runoff generation and, in parallel, sediments are not transferred to stream channels in the way they once were (e.g. Llorens et al., 1997; Gallart et al., 1997; Beguería et al., 2006; Buendia et al., 2016a; Herrero et al., 2017). As well as coarse material, changes in the suspended sediment load occur (e.g. Walling, 1991; Lorente et al., 2000). These changes influence basin sediment yields, with alterations especially evident in mountainous regions where erosional processes are usually higher. Such changes have been documented extensively in the Southern Alps (e.g. Liébault and Piegay, 2002), the Italian Alps and Apennines (e.g. Surian and Rinaldi, 2003), as well as in numerous other Mediterranean watersheds (e.g. Gallart and Llorens, 2003; Hooke, 2006).

Localised human activities also modify flow and sediment transport regimes (e.g. Walling, 2006; Florsheim et al., 2011) and may interact with or exacerbate the effects of land cover change. For instance, in-channel gravel mining modifies channel topography and river-bed structure, increasing the downstream transfer of sediment and reducing the ability of impacted reaches to retain particles, especially those that are transported in suspension (e.g. Lagasse et al., 1980; Rinaldi et al., 2005; Rovira et al., 2005). Hydropower schemes also alter river flow regimes and geomorphic characteristics (e.g. Zolezzi et al., 2009; Schmutz et al., 2015). The hydro-sedimentary significance of these alterations is determined by the availability and characteristics (e.g. size) of the river-bed sediments and, in the case of hydropower, the competence of the pulses (hydropeaks). Concerns about the ecological implications of such changes to river habitat have reported widely (e.g. Casas-Mulet et al., 2014; Bruno et al., 2016; Choi et al., 2017; Béjar et al., 2017).

When hydropower production or gravel mining activities are located on mainstem reaches, tributaries play a key role in the downstream recovery of water and sediment loads. For instance, Pattinson et al. (2014) described how downstream changes in the hydrograph and total water yield of the River Eden (UK) were affected by tributary inputs. Depending on mainstem versus tributary competence, tributaries also modify sediment transfer and may cause aggradation at confluence zones (e.g. Rice, 2017). Tributaries often become the main source of sediment, so end up controlling the total catchment sediment yield (e.g. López-Tarazón et al., 2010; Piqué et al., 2014); they are particularly important sediment sources downstream from dams (e.g. Petts, 1984).

This paper presents empirical data on the variation in flow and suspended sediment transport within a ca. 10-km river reach. The use of the term 'reach' follows Fryirs and Brierley (2012), and represents a section of river channel whose structure is relatively uniform and whose morphological attributes are primarily controlled by flow regime and sediment transport. Water and sediment transfer through study reach are influenced by hydropeaking, which alters the amount and timing of flow discharge; they are also influenced by gravel extraction which modifies channel geometry and river bed structure and, ultimately, controls in-channel sediment availability and fine-sediment supply. Several tributaries join river in the downstream part of the study reach, each supplying water and sediment. These tributaries drain land areas with contrasting sediment contributions - some are composed of highly erodible materials and other by forested landscapes. Thus, tributaries likely alter both sediment supply and transport capacity in ways that lead to adjustments downstream from mining-affected areas. The reach has also experienced long-term adjustments related to reductions in runoff and sediment supply from the catchment headwaters, due to profound and extended land use changes that have affected this mountainous region since the second half of the 20th century.

Within this context, our main hypothesis is that hydropeaking, in-channel gravel mining and tributaries influence flow and suspended sediment dynamics within the reach at a number of different spatial and temporal scales. We predict that the precise effects hydropeaking, mining and the tributaries will differ, but that together they alter the river's sediments budget and hence its fluvial equilibrium. Related to this, we also predict that, due to their relative positions, these three things produce a series of sharp re-adjustments in fluvial processes as the river progresses downstream. These hypothesis and predictions allow us to assess the extent to which natural (floods and tributaries) and anthropogenic (mining and hydropeaking) perturbations affect spatial and temporal dynamics of flows and suspended sediment transport in this montane catchment.

2. STUDY AREA

The River Cinca is the second main tributary of the Ebro (NE Iberian Peninsula), and drains an area of 9,740 km^2 (equivalent to 11 % of the Ebro catchment area). The study reach is located in the upper part of the basin and geologically is placed in the South Central Pyrenees (Figure 1B). The catchment area of the study reach is 849 km2. Elevation across the upper Cinca ranges from 522 to 3,355 m a.s.l., with 30 % of the catchment area located above 2,000 m a.s.l. More than one third of the catchment is occupied by forest, while bare rock areas and pastures represent approximately another third of the catchment; the remaining area consists of a variety of vegetation types (Corine Land Cover 2006 database, EEA, 2006; Figure 1B). Mean annual precipitation is 801 mm, measured in Ainsa at the outlet of catchment for the period 1981-2010 (Figure 1B) (Source: Spanish Meteorological Agency, AEMET). According to Verdú et al. (2006) for the nearby Isábena catchment, the rainfall gradient in the area is ca. 7 % for each 100 m rise in altitude.

The study reach is 9.8 km long, and extends from Laspuña Bridge (catchment area 596 km^2) to Ainsa (849 km^2). It is located upstream from the confluence with the River Ara and before the river enters the Mediano Reservoir (Figures 1B and 1C). Upstream from the Laspuña Bridge, the river is regulated by a small reservoir (Laspuña Dam), in operation since 1965 and with a capacity of 0.35 hm3. The dam is located 8.2 km upstream from the study reach and it is operated by wicket gates that impound water during low flow conditions. Water is then pumped to a series of underground reservoirs that are subsequently operated for hydropower generation. Gates of the dam are left open during high flows, allowing water and sediments to pass through it; hence, regulation only affects low flows. The compensation flow from the dam is 0.5 m^3/s. The reach downstream from the Laspuña Bridge is subjected to regular hydropeaks (i.e. sudden increases on river discharge as a consequence of turbine pulses that produce electricity). Maximum releases during hydropeaking reach 25 m^3/s (Acciona SA, private communication). On average, mean velocity and water depth increase 30 and 20 % respectively during hydropeaking. Although hydropeaks change channel hydraulics, they are not competent enough to entrain characteristic particle sizes of the river bed (for more details see Béjar at al., 2017).

Further downstream from Laspuña Bridge the reach has experienced significant morphological changes related to flood protection and channel maintenance works. These works commenced during the 1990s and include embankment, rip-rapping and the extraction of gravels from the channel to aid flood conveyance. As described by Rubio (1995) and recently with more detail by Llena (2015), the channel has changed from braided to wandering pattern, and as a result of embankment has become disconnected from its floodplain almost everywhere. Vegetation has encroached many on formerly active bars, while other areas have experienced channel incision. Mining activities are now reduced compared to historic levels, although gravels are still extracted occasionally (e.g. Béjar et al., 2017 and 2018), mainly under flood prevention works and to reduce aggradation in the main channel downstream from tributaries.

The Cinca has a nivo-pluvial hydrological regime. Mean discharge (hereafter Q) downstream from the Laspuña Bridge is 27 m^3/s, with mean water yield (hereafter WY) of 895 hm^3/y (i.e. 1123 mm/y) (measured for the period 1959-2013 at the Escalona gauging station, A051, Ebro Water Authority). Temporal differences in Q are marked (inter-annual CV = 0.38 and intra-annual CV = 0.42). Annual floods (at A051) attain >220 m^3/s, whereas the 10-yr recurrence interval floods are 740 m^3/s (based on Llena, 2015 who analysed the annual maximum series for the period 1951-2012). Mean daily maximum flow (Q_c) is 1085 m^3/s on 1982, while the maximum instantaneous discharge (Q_{ci}) estimated for the event ranged from 1583 and 1982 m^3/s (Llena et al., 2016 and Rubio, 1995; respectively). The active channel width in the study reach averages 200 m, while water depth at mean Q averages 0.3 m.

Channel maintenance works were performed twice in the reach during the study period: the first in March 2014, just upstream from the confluence with the River Bellos, and the second in August 2014 at the confluence of the Forcas, a small ephemeral tributary (see Figure 1C for locations). These works included gravel extraction and, consequently, they modified channel geometry and river bed structure, with effects on sediment availability during subsequent flood events (Béjar et al., 2017 and 2018).

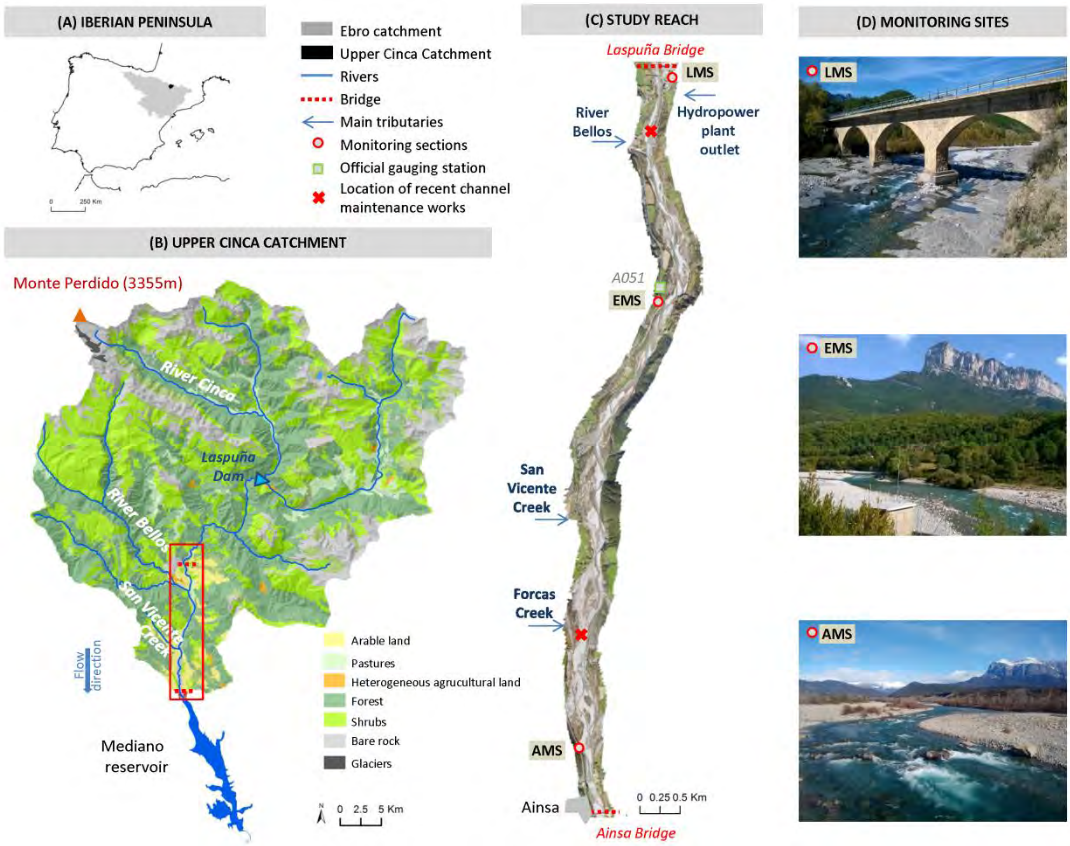

Figure 1. (A) Location of the Ebro and the upper Cinca catchment in the Iberian Peninsula. (B) Location of the study reach (red rectangle) in the upper Cinca Catchment. (C) Monitoring sites, main tributaries and channel maintenance works in the study reach. (D) Plates showing the three monitoring sections.

3. METHODS

3.1 Study sections

Discharge and suspended sediment transport were continuously monitored from October 2014 to October 2016 at three monitoring sections between the Laspuña Bridge and Ainsa (Figure 1C and D). The first section (i.e. Laspuña Monitoring Section, hereafter *LMS*) is located 100 m upstream from the outlet of the Laspuña hydropower station, where hydropeaks are generated. The catchment area at *LMS* is 596 km^2. The second section (Escalona Monitoring Section, *EMS*) is located 2.3 km downstream from the confluence of the River Bellos (193 km^2) and 3.3 km from *LMS*. *EMS* coincides with the location of the gauging station A051. Catchment area at *EMS* is 797 km^2, 201 km^2 larger than at *LMS*. The third section (Ainsa Monitoring Section, *AMS*) is located at the downstream end of the study reach. It is located near the village of Ainsa, 3.2 km downstream from the San Vicente confluence (10 km^2) and 6.5 km downstream from *EMS* (Figure 1D). Catchment area at *AMS* is 849 km^2, 52 km^2 larger than at *EMS*.

3.2. Flow discharge

Discharge was determined using water stage probes (measuring at 15-min intervals) at each of the monitoring sections. Water stage (hereafter *h*) was measured by means of: (i) a capacitive water stage sensor TruTrack® WT-HR at *LMS* and; (ii) Campbell® CS451 pressure transducer controlled by a Campbell® CR200X data-logger at *EMS* and *AMS*. The performance of the sensors was checked fortnightly. Values of h were transformed to Water Surface Elevation (hereafter *WSE*) values from the real elevation of the datum of each sensor and transducer. Elevations were obtained by means of a Leica® Viva GS15 RTK GPS system. *WSE* values were later converted to *Q* by applying site specific *WSE/Q* rating curves developed from direct discharge measurements and hydraulic modelling (Figure 2A). A similar procedure was followed recently by Lobera et al. (2016) and Piqué et al. (2017). Discharge was measured using an Acoustic Doppler Current profiler (ADCP; Sontek River Surveyor M9®) during both base flows and floods (gauged Q ranged between 1.9 and 122.4 m^3/s; Table 1). At *AMS*, a nearby bridge and the geometry of the section (wider and shallower) facilitated flow gauging during both low and high flows (Table 1). Higher flows at *LMS* and *EMS* could not be gauged accurately due to channel geometry and the presence of standing waves.

The Iber 2D® hydraulic model was used to improve discharge coverage for the *WSE/Q* ratings. Iber 2D® is a hydrodynamic model that solves the two-dimensional depth-averaged shallow water (2D St Venant) equations (Bladé et al., 2014). A 1×1 meter DEM of the study reach was imported to the model and a regular mesh was created. The DEM was obtained by means of Digital Photogrammetry applied to aerial photographs taken from a gyrocopter that flew between 200 and 250 meters above the ground (details in Vericat et al., 2016). Channel roughness was considered uniform for the whole channel; i.e. a Manning's value of 0.035 was used based on sediment size and sorting, and the presence of vegetation in some of bars. The model was run for each target *Q* (Table 1) until it reached a stable flow condition. Water Surface Elevation at the sensor locations was extracted for each of the modelled Q. A first assessment of model performance was made by comparing the registered and modelled WSE for the gauged Qs. For *LMS*, the absolute mean difference between observed and modelled WSE (i.e. residuals) was 0.05 m (N = 6), while the standard deviation (hereafter SD) of these residuals was also 0.05 m. For *EMS*, the absolute mean was 0.09 m and the SD 0.03 m (N = 4) while for *AMS* the absolute mean was 0.06 m with a SD 0.08 m (N = 4). These values are considered acceptable for producing *WSE/Q* relations for some higher discharges missing from the range of empirical ratings, especially given the complexity of the study river reach in terms of bed substrate, channel geometry and flow hydraulics. The modelled values obtained for EMS are in agreement with those reported by the Ebro Water Authorities for A051, located just 20 m upstream from the study reach (Figure 1). The main differences were observed during low flow conditions ($Q < 25 m^3/s$). The Root Mean Square Error (RMSE) estimated between our flow estimates and the data from the A051 was 9.9 m^3/s (for the whole range of flows). As channel geometry in the monitoring sections did not change significantly during the study period, a single *WSE/Q* rating curve per each section was used. Table 1 summarises the flows that were gauged and modelled for the three monitoring sections.

Table 1. Gauged and modelled flows to elaborate the site specific h/Q rating curves.

Monitoring Section	Range of gauged flows [1] (m^3/s)	Modelled flows[2] (m^3/s)
Laspuña (LMS)	1.9 – 13.4	2 – 800[2]
Escalona (EMS)	9.8 – 30.4	2 – 800[2]
Ainsa (AMS)	10.9 – 122.4	2 – 800[2]

[1] Discharge was gauged using an Acoustic Doppler Current profiler (Sontek River Surveyor M9®)
[2] Modelling was conducted by means of the 2D hydraulic model Iber 2D® (see methods for more details). A total of 14 steady discharges between 2 and 800 m^3/s were modelled: 2, 6, 10, 20, 40, 50, 70, 100, 150, 200, 250, 300, 400 and 800 m^3/s. Some of them were coincident with gauged flows. See text for more details and discussion.

3.3. Suspended sediment transport

Water turbidity (*NTU*) was measured continuously at the three monitoring sections. Initially turbidity was measured using McVann ANALITE® NEP-9530 probes (range 0-3000 *NTU*), with 15-minute average values registered by a Campbell® CR200X data-logger. The probes at *EMS* and *AMS* were replaced by Campbell® OBS300 probes (range 0-4000 NTU) in the second study year due to malfunctioning of the original ones.

Turbidity records were transformed to Suspended Sediment Concentration (*SSC*) using site-specific *NTU-SSC* rating curves (Figure 2B). For the short periods where turbidity exceeded the measuring range (during flood events), the maximum values observed SSC during the period of record were used. This is a conservative approach and provides a minimum estimation for these periods. This, together with the short duration of such out-of-range periods, is assumed to not significantly impact results. To produce the rating curves, 810 water samples were obtained across a flow range of 6 - 256 m^3/s (note that the maximum *Q* registered during the period was 296.5 m^3/s). Samples were collected both manually and automatically (by means of ISCO® 3700 samplers) in locations where turbidity probes were placed. Water samples were filtered using glass microfiber filters (Filter-Lab®, 0.0012 mm pore size), then dried and weighed to determine total concentrations (i.e. organic and inorganic fractions combined; in mg/l). Additionally, a total of 76 water samples were collected from bridges near the upstream and the downstream ends of the study section (34 samples in the Laspuña Bridge, near the *LMS*, and 42 samples in the Ainsa Bridge, near *AMS*, see Figure 1C for location details). For these, three vertical samples were collected during floods, with average of these used to represent cross-sectional *SSC* at a given time. They were also used to assess the spatial variability of *SSC* and evaluate the representativeness of the point *SSC* derived from the turbidity probes.

A statistical relation between NTU and SSC was established for each monitoring section. These used a linear regression (i.e. $SSC = a \times NTU$), with coefficients of determination (R^2) all significant and ranging from 0.89 to 0.94 (Figure 2B). The relationship between *NTU* and *SSC* did not change through the study period, indicating that physical properties of the suspended particles remained similar. Nevertheless, small differences were observable from event to event, probably indicating variations in the sediment characteristics (e.g. related to lithology; Soler et al., 2012). Moreover, a subset of 80 representative samples (i.e. selected to cover a range of *Q*, season and monitoring section) was analysed to determine the percentage of organic matter and mineral fractions. Following Tena et al. (2011) filters were burnt at 450ºC for 5 hours in the oven, and then re-weighed to determine the percentage organic and mineral content. Load data are presented for both total and mineral composition.

Relations between *Q* and *SSC* (i.e. rating curves) were established based on these procedures and the resulting data. In addition to these ratings, which represented the general (central) trends, quantile regression was used to model the upper and lower bounds of the relations. Quantile regression provides an alternative to ordinary least squares regression that does not assume homoscedasticity i.e. homogeneity of variance in values of X across the range of Y, and which allows the fit of models to the upper and lower limits of a response. The 90th and 10th quantiles (percentiles) were modelled for *SSC*, as these were the most extreme limits of the data for which quantiles could be validly calculated, given the sample size. Quantile regressions were run using the Quantreg Package (Koenker et al., 2017) in R® 3.4.0 (R Development Core team, 2017).

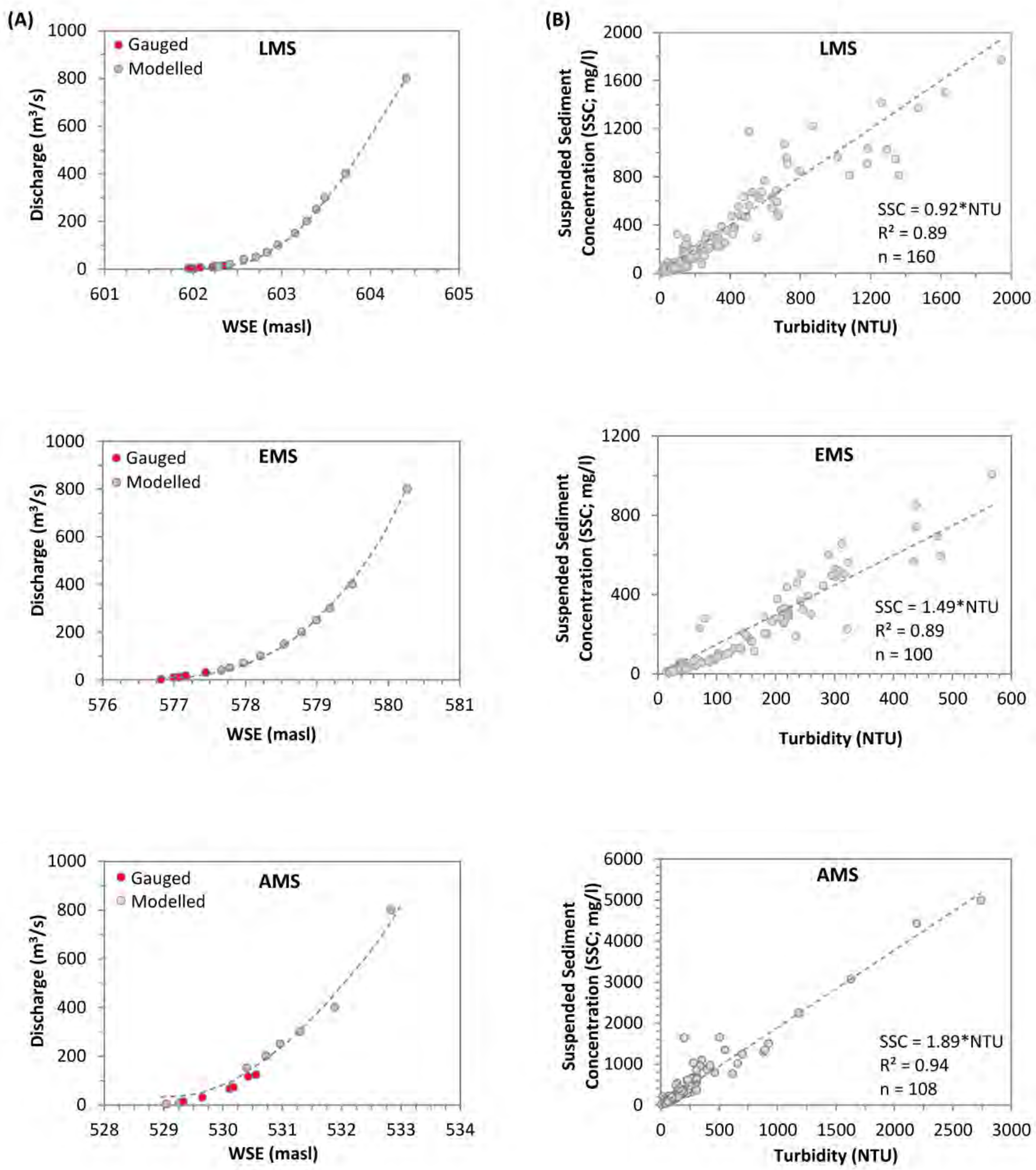

Figure 2. Rating curves between (A) Discharge (Q in m^3/s) and Water Surface Elevation (WSE in m a.s.l.) relationships for the three monitoring sections. Note that gauged and modelled data are indicated (see table 1 for more details). (B) Turbidity (NTU) and Suspended Sediment Concentration (SSC in mg/l) at the three monitoring sections, where LMS stand for Laspuña Monitoring Section, EMS means Escalona Monitoring Section, and AMS stands for Ainsa Monitoring Section. Note the scales of the SSC differ between curves.

3.4. Water and sediment load calculation

Suspended sediment load (*SSL*) was calculated at each monitoring section from the recorded and transformed 15-min *WSE* and *NTU* data. *SSL* was subsequently computed by multiplying *Q* and *SSC* at 15-min time steps.

Discharge was transformed to *WY* multiplying *Q* and time at 15-min time steps. Values were subsequently summed to obtain loads and yields at the different temporal scales and used to produce flow and sediment load frequency and duration curves. Finally, the Specific Sediment Yield (*SSY*) at each monitoring section was calculated by dividing the annual load by the catchment area. Table 2 shows all calculated variables, their abbreviation and units.

4. RESULTS

4.1. Hydrology

Long term data (1959-2013) registered at A051 was considered as the reference flow series to give a context for conditions observed during the study period. Annual average *WY* (at *EMS*) during the study period was 692 hm^3, almost 25 % lower than the long term value (895 hm^3). Mean *Q* was 22.2 m^3/s (CV = 0.85), compared to the long-term mean of 27 m^3/s. Both of the study years had *WY* values that sit between the 25^{th} and 75^{th} percentiles of the long term *WY* values. Therefore, following Santa and Yuste (2010) we consider our period as average when compared to the long term flow data. Nevertheless, *WY* was different between years, with 2014-2015 slightly wetter than 2015-2016 (Table 2) Figure 4A shows the flow duration curves for the two study years and compares them to the long-term ones. For high flows, the long-term curves plot above those for the study period, indicating that the magnitude of floods during 2014-2016 was lower than the historical data. This pattern was reversed for low flows, especially for the 2014-2015 hydrological year.

Figure 3 shows the hydrographs for the monitoring sections for the two hydrological years. Marked differences in Q were evident between the monitoring sections (Figure 4B and Table 2). Mean *Q* downstream from the Laspuña Dam (Figure 1) was 3.1 m^3/s, a value that equates to an annual *WY* of 96 hm^3 (2014-2016; source: Acciona, the company that operates the dam and hydropower station). Further downstream at *LMS*, a section not affected by hydropeaking (Figure 1C), mean Q was 5.4 m^3/s (CV= 2.2; Figure 3) and mean specific discharge (hereafter Q_S) was 0.009 $m^3/s\ km^2$. Annual *WY* at *LMS* averaged 170 hm^3. Maximum peak discharge (Q_{max}) ranged between 20 and 260 m^3/s (the latter occurring in June 2015, Figure 3). See Table 2 for more detailed flow statistics.

Further downstream, the flow at EMS showed the influence of both the hydropeaking and the River Bellos. River Bellos is the most important tributary, flowing into the Cinca 2.3 km upstream from the A051 gauging station and EMS (Figure 1C). The maximum outflow of the hydropower station is 25 m^3/s. According to the daily data supplied by Acciona, the annual volume used for hydropeaking is 432 hm^3 (16.7 m^3/s), with no marked differences between years (448 and 417 m^3/s, respectively). Mean annual *Q* at *EMS* was 21.9 m^3/s (CV= 1.2 i.e. around half than that observed at LMS), and QS was 0.027 $m^3/s\ km^2$, triple than the one in *LMS* (Table 2). In general, the flow duration curve (Figure 4B) indicates that flows at *EMS* were more evenly distributed through time than *LMS*.

Based on the mean annual *WY* of the hydropower station and that for *EMS*, the contribution from the River Bellos can be estimated to average around 90 hm^3/y (i.e. 466 mm/y). Seven flood events more were registered at *EMS* than at *LMS* (Table 2), reflecting localised rainfall events in the Bellos catchment. Qmax at *EMS* ranged from 33 to 296.5 m^3/s during the study period (Figure 3), with the latter having a recurrence interval of ca. 3 years (calculated from the series of annual maximum instantaneous flows estimated for the period 1959-1993, using the Gumbel Law).

The San Vicente is a small stream, but constitutes the main tributary entering the Cinca between *EMS* and *AMS* (Figure 1C). In contrast to *LMS*, here *Q* showed a similar pattern to that observed at *EMS* (Figure 3 and 4B). Mean Q was 24.9 m^3/s (CV=1.1 i.e. keeping the relatively low *Q* variability already observed at *EMS*) and Q_S was 0.029 $m^3/s\ km^2$. Mean *WY* was 788 hm^3, almost 100 hm3 more than *EMS* (Table 2). Q_{max} ranged between 37 and 362.6 m^3/s (the latter occurring in November 2014, Figure 3).

Flows throughout the study reach are characterized by a marked seasonality; for instance, floods typically occur in spring due to snowmelt, and in late summer and autumn as a consequence of localised thunderstorms (Figure 3). *LMS* is the section with the lowest monthly *Q* but has the highest CV (Figures 5A-C). Flows were least variable in winter, while spring and autumn were most variable (up to 300 %; Figures 5A-C). The monthly flow pattern changed downstream at *EMS* and *AMS*, where CVs were lower and Q more constant. At these lower sections, most variation was observed in winter (up to 150 %), probably related to the occurrence of isolated thunderstorms. *WY* also varied between years: it was more constant through the months in 2014-2015, while in 2016 one month (May) was responsible for at least 20 % of the total annual runoff at the three stations (Figure 6A-C). Figure 7A summarizes *WY* in the three sections while differences between sections are indicated in figure 7C (percentage differences)

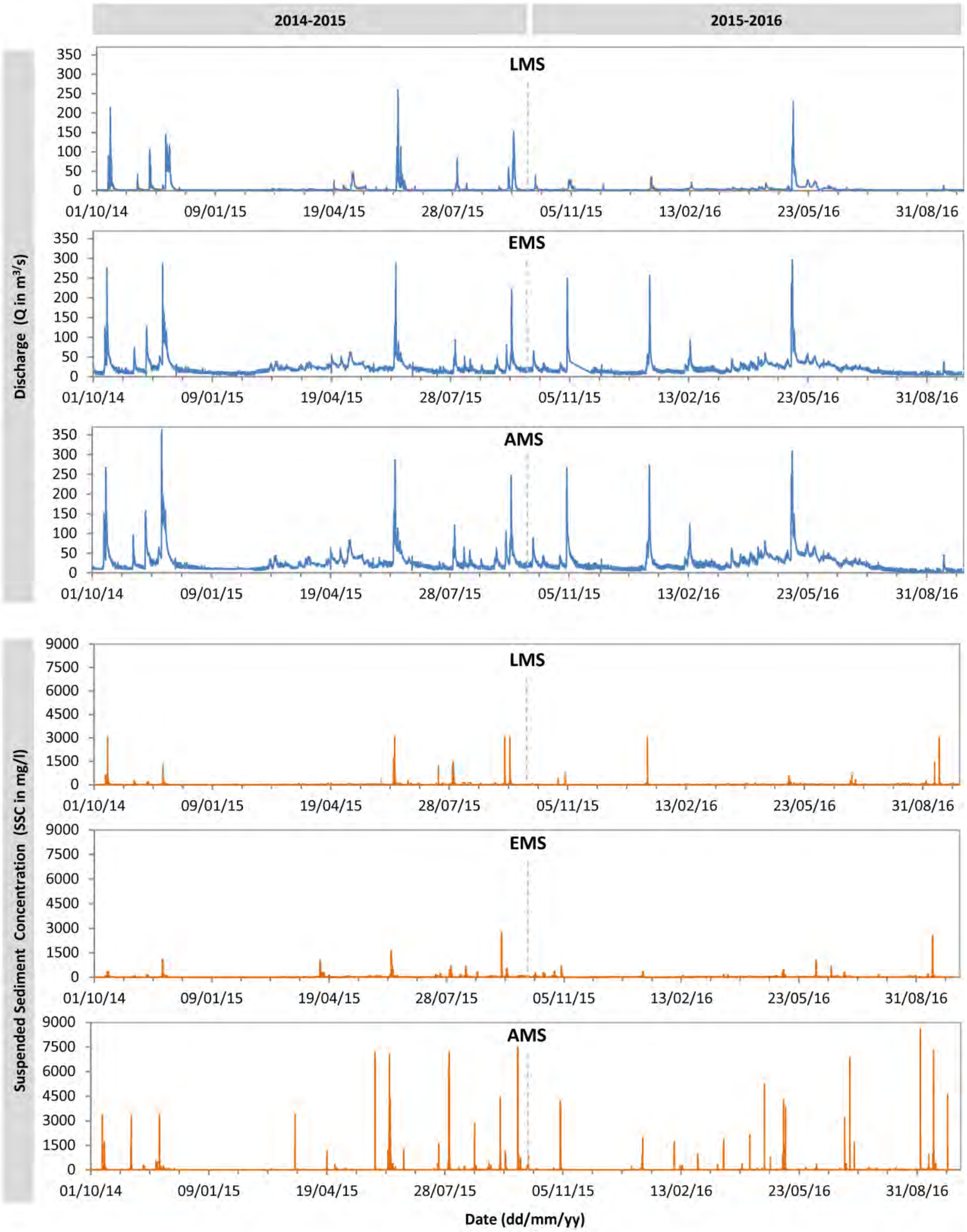

Figure 3. Discharge and suspended sediment concentration registered at each of the monitoring sections in the upper River Cinca during the study period. Note that LMS is located at the upstream end of the study reach, EMS is located 3.3 km downstream from LMS, and AMS is 9.8 km downstream from LMS and is located at the downstream end of the reach (see Figure 1).

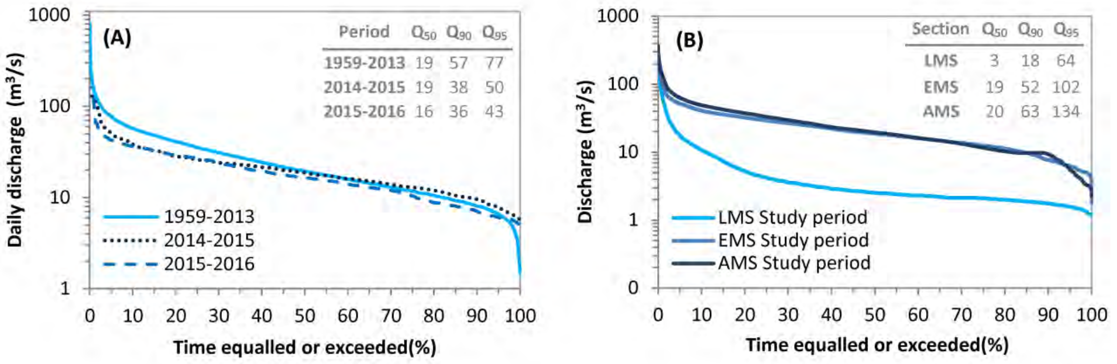

Figure 4. Flow duration curves (FDC) in the upper River Cinca. (A) FDC calculated for the two hydrological years of the study and for the period 1959-2013 at the A051 gauging station. (B) FDC calculated for the study period at the three monitoring sites. Note that y-axes are log transformed.

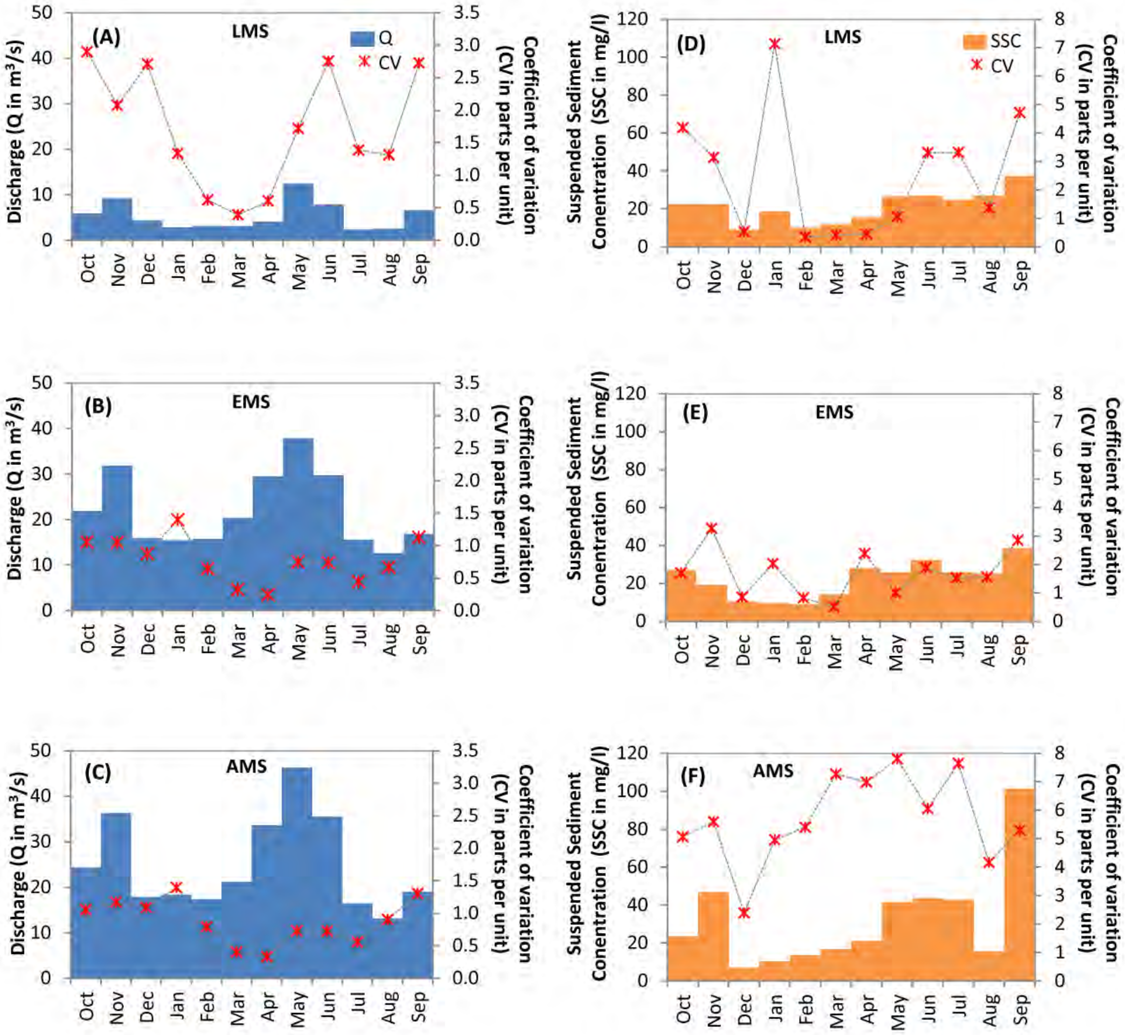

Figure 5. Mean monthly discharge (from A to C) and suspended sediment concentration (from D to F) for the period 2014-2016 at the three monitoring sections in the upper Cinca. The monthly coefficient of variation for each variable is also presented.

Table 2. Discharge and suspended sediment transport for the study period at the three monitoring sites in the upper Cinca River.

Monitoring Section and Catchment area	Year	Maximum discharge Q_{max} (m^3/s)	Mean discharge* Q_{mean} (m^3/s)	Water Yield *WY* (hm^3)	Mean Suspended Sediment Concentration* SSC_{mean} (mg/l)	Suspended Sediment Load *SSL* (t)	Suspended Sediment Yield *SSY* ($t/km^2/y$)	Flood events **
Laspuña (*LMS*) 596 km²	2014-15	260	6.1 (16.6)	192	23.3 (87.0)	27320	45.8	12
	2015-16	228	4.7 (9.5)	148	18.9 (75.6)	7544	12.6	11
	Average	*260*	*5.4 (13.6)*	*170*	*21.1 (81.6)*	*17432*	*29.2*	
Escalona (*EMS*) 797 km²	2014-15	288.7	23.0 (20.9)	726	23.3 (62.5)	36704	46.1	16
	2015-16	296.5	20.8 (19.8)	658	21.0 (36.2)	21507	27	14
	Average	*296.5*	*21.9 (20.4)*	*692*	*22.2 (51.1)*	*29106*	*36.5*	
Ainsa (*AMS*) 849 km²	2014-15	362.6	25.6 (25.9)	812	35.7 (255.7)	80462	94.9	16
	2015-16	309.2	24.2 (24.6)	763	28.0 (228.6)	45390	53.5	14
	Average	*362.6*	*24.9 (25.3)*	*788*	*31.9 (242.5)*	*62926*	*74.2*	

* Values between brackets represent the Standard Deviation (SD)
** Flood events defined as those exceeding 1.5x the average *Q* at the monitoring section

4.2. Suspended sediment transport

4.2.1. Data representativeness

The mean CV of the three replicates collected on each cross-section was around 10 %, indicating limited cross sectional spatial variability. Additionally, the mean ratio between the mean cross-section *SSC* and the *SSC* obtained from turbidity records are close to 1 in both sections. These results indicate that the SSC derived from at-a-point turbidity records can be considered fully representative of the mean cross-sectional *SSC*. It therefore appears evident that during flood events fines are well mixed in the water column. It is worth to mention that *SSC* can change over shorter time intervals than the frequencies in which loads are registered (i.e. Gitto et al., (2017)). While these short temporal fluctuations are not considered explicitly here, the high frequency of the records for the Cinca study reach (15-min), suggest that such variability would not significantly alter the final load computation.

The estimated *SSC* for the periods that the turbidity probes were out of range or saturated is more relevant in the lowermost section, *AMS*, where the probe was out of range 35 hours in total (0.16 % of the time; Figure 2B). Turbidity exceeded the measuring range of the probe during the 0.03 % of the time at *LMS* and it was never out of range at *EMS*. We used the *SSC* obtained from water samples to assess the magnitude and significance of potential errors related to periods with no data. Specifically, at *AMS* during the 31/07/2015 flood, missing SSC were filled with a value of 3062 mg/l, based on the maximum observed SSC during the period of record. For the same event SSC obtained with the automatic water sampler reached values up to of 17 g/l Then, if the values from the automatic sampler were used instead, the total load for the study period increases by only 450 t, equivalent to 0.35 % of the total load estimated by the approach we have followed.

The content of organic matter in the SSC samples was determined for a wide range of Q (i.e. between 13 to 258 m^3/s). The percentage of organic matter ranged from 1.5 to 14.4 %. The average content was very similar at the three monitoring sites (CV = 0.4). Additionally, no major differences between seasons and flows were observed (CV = 0.17). Consequently, taking into account this low variability, both spatially and temporally, we subtracted the organic matter from the *SSC* by using a single average percentage (5.3 %). Consequently, the reported *SSC* were corrected to only reflect the mineral content of the SSC.

4.2.2 Instantaneous concentrations and sediment load

At LMS the *SSC* averaged 21.1 mg/l, a relatively low value but one that was highly variable in time (CV = 3.9; Figure 3). Nonetheless, inter-annual variability of *SSC* was low (i.e. CV = 0.11; Table 2). Mean annual SSL was 17432 t, but with high inter-annual variability (Table 1), i.e. SSL in 2014-2015 was 3 times greater than in 2015-2016 (Figure 6). Mean annual *SSY* for the study period was 29.2 t/km^2.

Mean *SSC* at EMS was similar but less variable than at *LMS* (i.e. 22.2 mg/l; CV = 2.2), while instantaneous *SSCs* were generally less at *EMS* than observed upstream. Maximum *SSC* was 2706 mg/l. Mean annual *SSL* was 29,106 t (CV = 0.4; Table 2), almost 70 % higher than that estimated at *LMS*. This difference can be mainly attributed to the difference in runoff observed between the two sections (as described in section 4.1) and the increase in the contributing catchment area, and hence the extension of sediment sources. Annual loads were less variable (lower SDs; Table 2). The mean annual SSY during the study period was 36.5 t/km^2, although it differed markedly between the two years (being twice as high in 2014-2015 than in 2015-2016; Figure 6 and Table 2).

Figure 3 shows the increment in both the magnitude and variability of *SSC* in the lowermost section, *AMS*. Mean *SSC* here was 31.9 mg/l (CV= 6.7), with a maximum instantaneous concentration of 17 g/l (calculated from a water sample collected during a flood in June 2015). Mean annual SSL was 62,926 t, a value more than two-fold higher than *EMS* and 3.5 times that of LMS. Annual differences at *AMS* were substantial (56 % of difference; Table 2). Mean annual SSY was 74.2 t/km^2, more than twice for that estimated in the upstream sections. Differences in SSY illustrate the magnitude of variation in suspended sediment transport within a relatively short (ca. 10-km) study reach. Figure 7B summarizes the *SSL* in the three sections while differences (percentage difference) between sections are displayed in figure 7C.

Monthly patterns of *SSC* differed from those of *Q*, suggesting the key role that sediment production and availability play in the temporal distribution of *SSC* (Figures 5D-F). Monthly variability differed between the monitoring sections (Figures 5D-F and 6D-F). Spring and autumn produced the highest *SSL*, although there were marked differences between years; in particular, May 2015 contributed more than 30 % of the annual sediment load of the three monitoring sections. This pattern was also observed for *WY*. Minimum monthly *SSC* was observed in February and March (Figures 5D-F), whereas January 2016 yielded more than 10 % of the annual load at all the monitoring sections (Figure 6D-F).

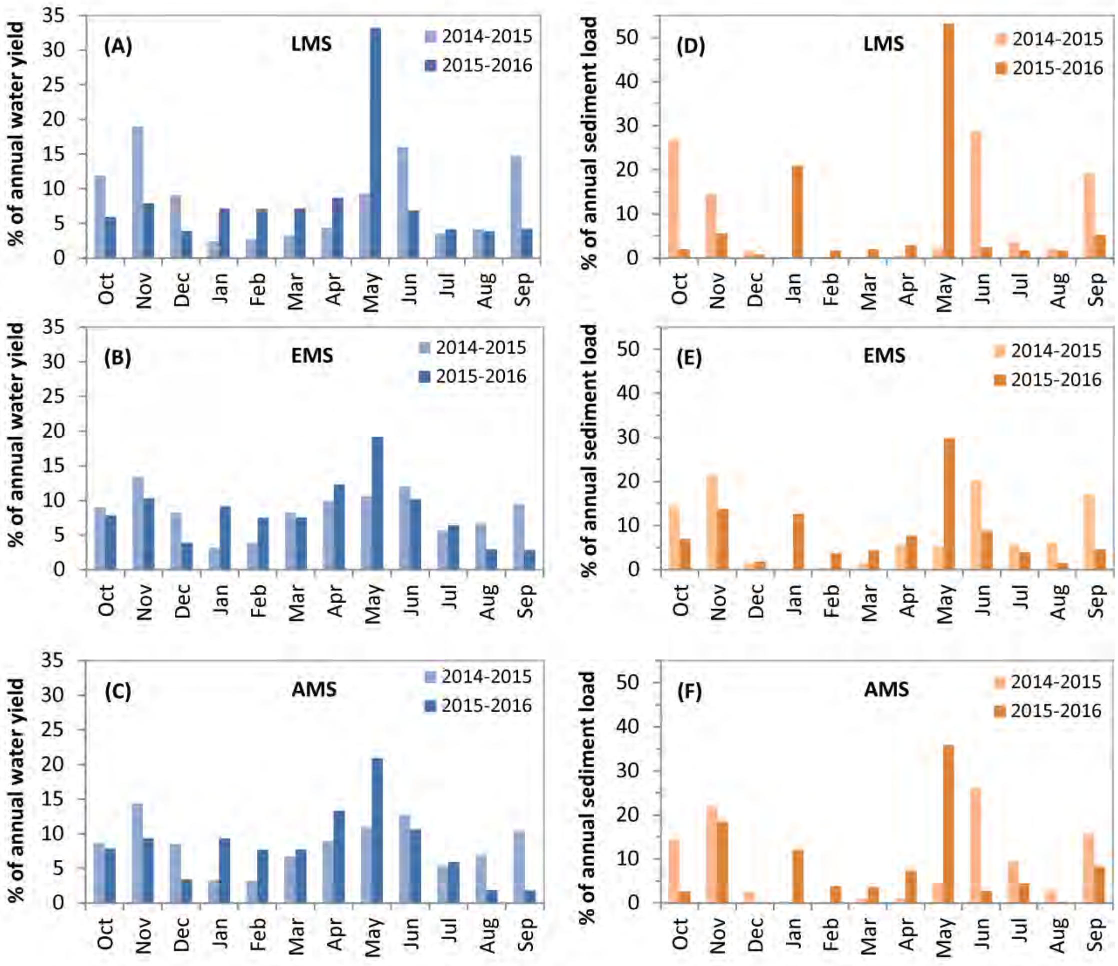

Figure 6. Monthly distribution (i.e. percentage over the total annual value) of the water yield (from A to C) and sediment load (form D to F) at the three monitoring sections for the two studied hydrological years.

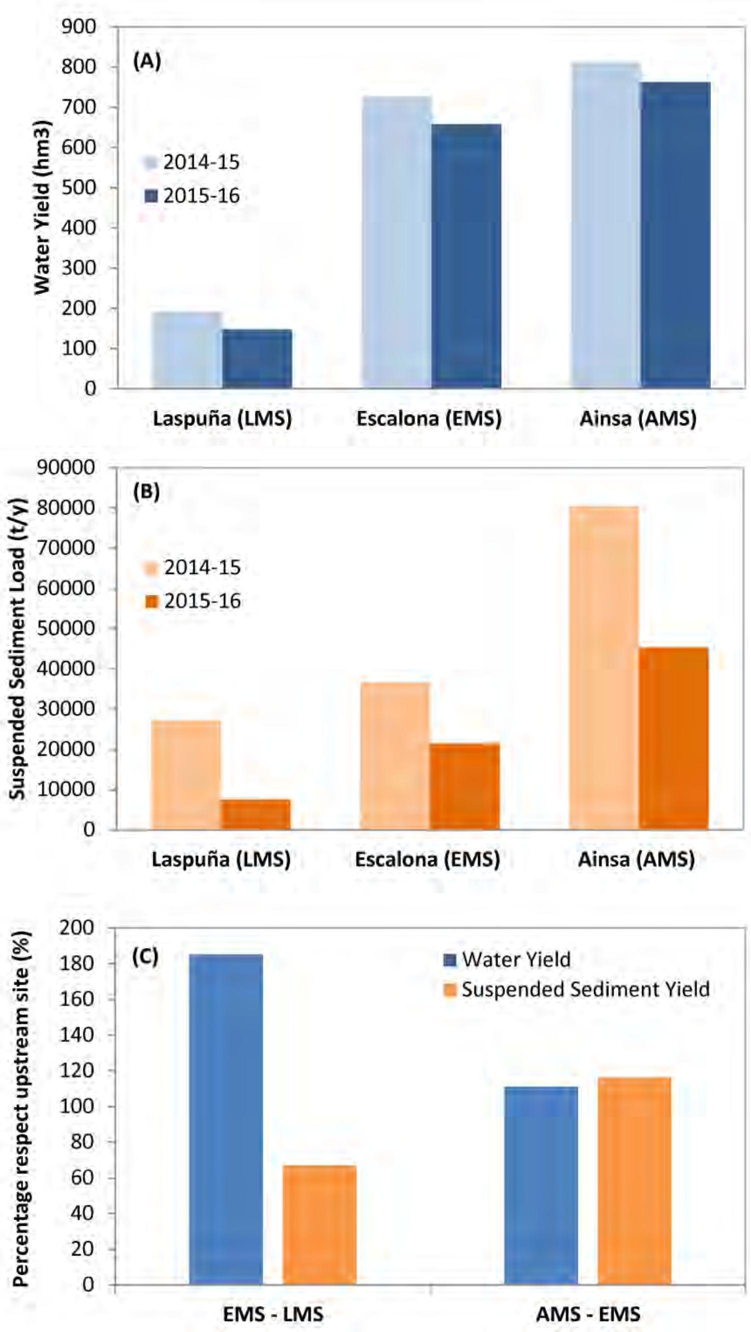

Figure 7. (A) Water yield and (B) suspended sediment load at the three monitoring sections during the two study years. (C) Differences between sections expressed as the percentage of the mean water yield or sediment load of a given section relative to the adjacent upstream section. Note that in C a value of 100 % means that the yield between consecutive sections is equal.

4.3 Relation between Q and SSC

Figure 8 shows the relationships between mean daily *Q* and *SSC* for the three monitoring sections. *Q* and *SSC* show positive relations, indicating that the magnitude of *SSC* is, in general, hydraulically dependent. Nevertheless, there is great scatter related to high variability of the observed concentrations for a given flow; for instance, in *AMS*, differences in *SSC* of up to two orders of magnitude (from 7 to 755 mg/l) can be observed for the same *Q* (16 m^3/s).

The quantile regressions fit to the 10 and 90 percentiles indicate that the slopes of the upper and lower bounds of the *Q-SSC* relations differ. For each section, the upper bound had a high gradient (e.g. β_1 = 3.86 for the *LMS* section) indicating that the maximum *SSC* increased markedly across the Q range. Conversely, the model fit to the lower bound had a much gentler gradient (e.g. 0.58 for the same *LMS* section), indicating that the minimum *SSC* did not increase as rapidly as discharge increased.

The scatter in the *Q-SSC* relationship was greatest at the downstream section, suggesting that *SSC* was less hydraulically dependent and influenced more by sediment availability and supply. At all three sections the variability of *SSC* decreased as *Q* increased (as indicated by the reduction in the scatter in the *Q-SSC* relationship). This indicates either that during floods sediment transport is more hydraulically dependent, regardless of the magnitude of sediment supply (or even when supply becomes limited), or that other sediment sources (e.g. bank erosion, bed subsurface) come into play during high flows due to flow competence; it could also be that both factors play a role.

It is worth noting that the scatter in the *Q-SSC* relationships differed seasonally (see different colours in Figure 8). For all the three sections the relationship in summer and autumn was the most variable, suggesting that a variety of sediment sources (badlands, catchment headwaters, tributaries) responding to localised rainfall episodes are most important in these seasons. In contrast, less variable, more hydraulically dependent sediment transport occurred during winter and spring, where sediment availability and supply is limited, and mostly depends on the amount of top-soil weathered sediments and the intensity and duration of snowmelt.

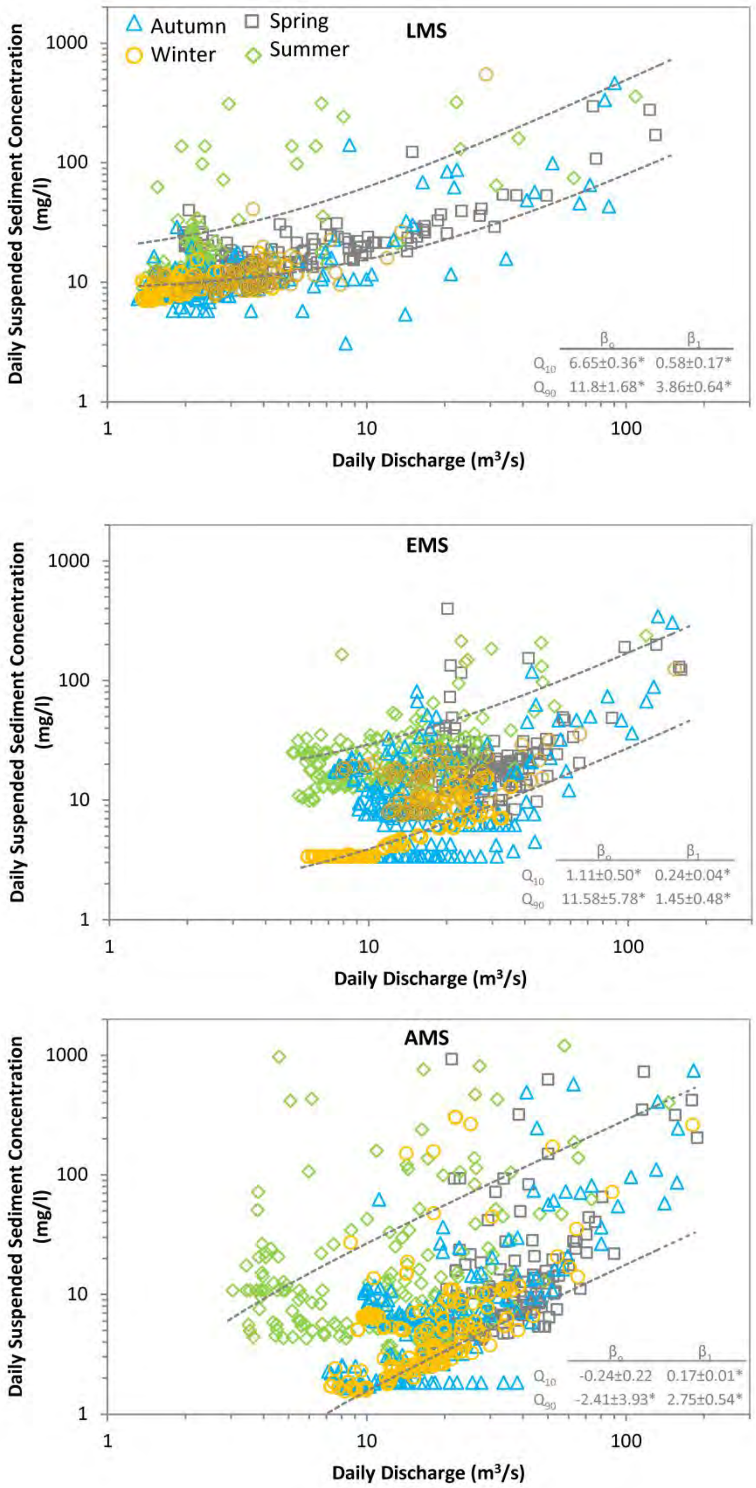

Figure 8. Scatter plots between mean daily discharge (Q) and mean daily suspended sediment concentration (SSC) at the three monitoring sites in the upper River Cinca for the sampling period 2013-2015. Note that data for each season is differentiated by colours (see key). Fitted lines represent the quantile regression models for the 90th (Q90) and 10th (Q10) percentiles. Inset shown estimates of regression coefficients (i.e. β0 = slope, β1 = intercept) for the quantile regression models ±Standard errors of the coefficients and significance (*). See text for more details.

4.4. Time concentration

Figure 9A shows the runoff and the suspended sediment frequency curves for each of the monitoring sections. While relatively frequent flows were responsible for the largest proportion of *WY* at *EMS* and *AMS* (e.g. 50 % of the WY was equalled or exceeded 30 % of time), higher flows were the ones that determined the magnitude of the total *WY* at *LMS* (i.e. the section not affected by hydropeaking); at *LMS* 50 % of the *WY* was equalled or exceeded 10 % of the time. In terms of *SSL*, floods were responsible for transporting most of the load: e.g. *Q* equalled or exceeded 2 % of time carried 73 % of the load at *LMS*, and the 48 and 53 % at *EMS* and *AMS*, respectively (Figure 9A). Interestingly, the *SSL* frequency curves at *EMS* and *AMS* differed more than those for *WY*. The curve at *EMS* was gentler, suggesting a supply-limited response of the river in comparison to *AMS*. At *AMS* the curve steepens, although less than at *LMS*. Note that *LMS* shows a flashier pattern, with floods playing an important role in both *WY* and *SSL*.

Figure 9B and 9C shows the cumulative *WY* and *SSL* during the study period and helps illustrating the influence of floods on water and sediment in the study section. At *LMS*, the *WY* curve has sharp changes in the slope (i.e. steps) during short periods of time (i.e. high *Q* events); while *EMS* and *AMS* show a more regular pattern (i.e. base-flows are higher and more sustained in time). Cumulative *SSL* curves for all sections showed marked steps, indicating that floods carry the majority of the sediment load. Differences may be again related to contrasting sediment sources and availability along the study reach during the study period.

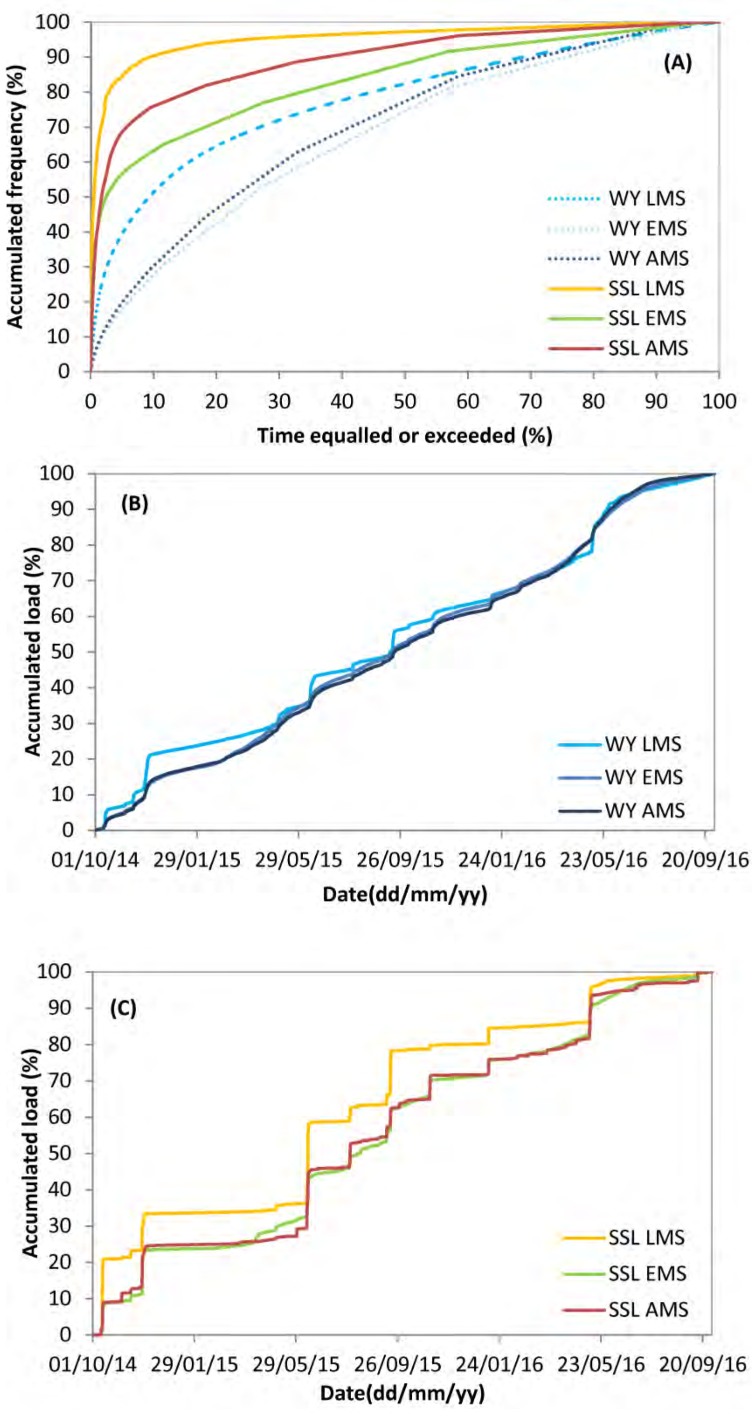

Figure 9. (A) Water Yield (WY) and Suspended Sediment Load (SSL) frequency curves at the three monitoring sections during the study period. (B) Cumulative WY and (C) SSL in the three monitoring sites along the study period

5. DISCUSSION

5.1. Key findings

This paper analyses the transfer of water and sediment through a 10-km wandering river reach affected by natural and anthropogenic factors. These factors - tributaries, hydropeaking and in-stream mining - affected *WY* and *SSL* in different ways and resulted in contrasting sub-reaches. Hydropeaking and inflow from tributaries increased water yield 4.6 times between the upstream and downstream ends of the study reach. The size of the contributing area increased 42 %, while mean annual *SSY* was 2.5 times greater at the downstream end of the reach. Sediment supply from tributaries and the increased sediment availability due to gravel mining appear to be the main factors contributing to the elevated SSY at the lowermost monitoring section. The spatial distribution of tributaries, hydropeaking and mining appear to exert a combination of additive and compensatory effects on water and sediment transfer over relatively short channel distances. In the following sections we discuss the water and sediment yield observed in the Cinca in relation to other catchments, and assess the main factors controlling these yields. Finally, we develop a preliminary sediment budget for the upper Cinca and discuss its wider implications, especially for reservoir siltation downstream.

5.2. Runoff and sediment yield in mountainous catchments

The upper River Cinca has a relatively high runoff and low *SSY* (927 mm/y and 74.2 t/km^2/y respectively) compared to other montane catchments in the Pyrenees. Differences in runoff between catchments are usually attributed to rainfall characteristics (e.g. intensity, amount, the presence and extent of snowpack), the processes and mechanisms by which runoff occurs, and catchment characteristics such as land use, soils and geology (Pilgrim et al., 1982). The River Ara, the main tributary entering the Cinca at Ainsa, just downstream from the lowermost measuring section), has an annual mean runoff of 750 mm/y (for the period 1951-2013; Centro de Estudios y Experimentación de Obras Públicas, Spanish Ministry of Public Works CEDEX; http://www.ceh-flumen64.cedex.es/anuarioaforos/default.asp). Rubio et al. (1995) pointed out that the runoff difference between these two catchments may be attributed to different catchment orientations, which determines vegetation cover and evapotranspiration. Other catchments located nearby also show less runoff than the Cinca. For instance, the River Ésera (894 km^2) has a mean runoff of 620 mm/y (1949-2010; Lobera et al., 2016), while the Isábena (445 km^2) has a mean value of 403 mm/y (1945-2008; Buendia et al., 2014). These differences may be attributed not only to precipitation but also to snowpack and groundwater transfers.

In contrast to runoff, SSY in the Cinca was lower than reported for these other catchments (e.g. 338 t/km^2/y in the Ésera reported by Lobera et al., 2016; and 544 t/km^2/y in the Isábena reported by López-Tarazón et al., 2012). The Ésera has a catchment area similar to the upper Cinca (at *AMS*), while the area of the Isábena is smaller. Both catchments, however, drain badland areas composed mainly of highly erodible Eocene marls that are the main sources of sediment (Francke et al., 2008a, b; López-Tarazón et al., 2009, 2010, 2011, 2012; Buendia et al., 2014). Although badlands are present in the upper Cinca, most of them have shown relatively low erosional responses compared to their counterparts, whereas others that are more active drain downstream from *AMS*, directly to the Mediano Reservoir, and so are not accounted for in this study. For example, Smith and Vericat (2015) reported annual mean net erosion rates of around 1.4 mm/y in a series of badlands in Pueyo de Araguás, 1 km downstream from *AMS* (see Figure 1 for location details), while Vericat et al. (2014) reported a mean erosion rate value of 6 cm/y and event-scale (i.e. rainfall event) net changes up to 1.8 cm in a small area of badlands in the Isábena catchment. Erosional rates in the order of 1 cm/y are also commonly found in the literature for other montane Mediterranean catchments (e.g. Clotet et al., 1988; Gallart et al., 2002; Nadal-Romero et al., 2006). These differences may be related to differences in methods, the size and specific physiographic and geological characteristics of the badlands, and the different time periods whose net changes were estimated. Overall though, net erosional rates in the upper Cinca are lower in magnitude, even when compared to those in Mediterranean mountain catchments with similar size (see for instance data presented in Nadal-Romero et al., 2011 or in an earlier review by De Vente and Poesen, 2005).

The mean *SSY* of the upper Cinca over the study period is very similar to the average *SSY* estimated by Batalla and Vericat (2011) using long-term reservoir sedimentation rates for catchments in the Iberian Massif where hydroclimatic and catchment characteristics are substantially different, and five times less than the estimates for catchments in the Pyrenean Range with similar characteristics. In addition, the *SSY* estimates for the Cinca plot in the lower bound of the values presented by de Vente et al. (2005) for Italian catchments with similar size, and within in the range of values for Mediterranean catchments in Israel (Inbar, 1992).

Spring and autumn were the seasons that contributed most to the total runoff and *SSY* in the upper Cinca. Runoff patterns were in agreement with those reported by Rubio (1995) for the same catchment. In mountainous catchments that contain badlands, the production, transfer and transport of sediments are controlled by processes on bare slopes. These processes are variable in time. Sediments deposited in the mainstem river exhaust at some point, and *SSCs* and as a result loads downstream are reduced (e.g. Lana-Renault et al., 2009; López-Tarazón et al., 2009; Gallart et al., 2013; Piqué et al., 2014; and Buendia et al., 2015). In such catchments sediment supply from badlands drives the annual sedimentary cycle, with sediment transport not hydraulically driven but controlled by the amount of sediment available in the river bed. In the upper Cinca, however, while badlands are present they are less extensive than in nearby catchments such as the Isábena, and erosional rates are significantly lower. Seasonal patterns of *SSL* and runoff were similar (i.e. with exceptions of April and November), indicating that sediment transport is mostly runoff-driven.

5.3. Factors affecting runoff and sediment yield

A combination of factors is responsible for the observed temporal patterns of runoff and sediment transport as well as their spatial variability through the study reach. The upper Cinca drains an extensively forested catchment. Forested areas influence runoff through interception, infiltration and evapotranspiration but they also control the production and transfer of sediment. In general, forested areas yield little sediment and so generate low *SSCs* (e.g. Gallart et al., 1998; Vericat and Batalla, 2010).

Average *SSY* of the Upper Cinca was low but both water and sediment transfers through the reach were highly variable. Reflecting this variability, three distinct sub-reaches can be identified:

(a) The sub-reach upstream from *LMS*. The character of this reach reflects both the natural behaviour of the catchment during high flows and the impact of the Laspuña Dam on the transfer of water and, to a lesser extent, *SSL*. Laspuña Dam (located 8.2 km upstream *LMS*) only affects the transfer of water and sediments during low flows since the wicket gates are fully opened during high flows. Therefore, in this uppermost sub-reach, the hydrological regime is more flashy (i.e. torrential pattern, steep curve in Figure 9A) and floods clearly influence the total *WY* (almost 45 % of the *WY* is provided by flows equalled or exceeded the 5 % of time). The transfer of sediment during low flows is negligible and it is during flood events (those equalled or exceed the 5 % of the time) when virtually all the sediment (i.e. 85 %) is transported (Figure 9A). This behaviour is found naturally in Mediterranean catchments (e.g. Batalla et al., 2005; Estrany et al., 2009; Pacheco et al., 2011; Piqué et al., 2017) but, in this sub-reach the Mediterranean character is not a natural hydro-sedimentary response but a consequence of the impoundment.

(b) Just 200 metres downstream from the lowermost point of *LMS*, *Q* and *SSL* are influenced by hydropeaking and tributary inflows. Hydropeaking is responsible for 62 % of the annual WY in the sub-reach immediately from *EMS*. The hydropower station has a maximum capacity of 25 m^3/s. This Q is equalled or exceeded the 30 % of time at *EMS* while only the 3 % of the time in *LMS* (Figure 4), indicating the impact of hydropeaking on the flow regime downstream from here. The role of hydropeaking modifying the magnitude of flows is also showed in the long-term FDC (Figure 4A).There is, however, no evidence that hydropeaking influences the sediment load here. This differs from Lobera et al. (2016) reported hydropeaking effects on SSL in the River Ésera, with increases in *SSC* up to one order of magnitude during flow releases. The difference between the Ésera and the Cinca can be attributed to the high competence of the hydropeaks in the Ésera as a result of factors such as channel geometry, river bed roughness and texture and sediment availability. Additionally, downstream from LMS and 2.3 km upstream from *EMS* there is the inflow of the River Bellos (Figure 1C). The Bellos contributes 13 % of the *WY* at *EMS* and accounts for the important increment of the runoff observed at this monitoring section (2.6 hm^3/km^2). The Bellos drains a limestone canyon where localised storms may affect the runoff regime. Thus, runoff generated in different parts of the upper Cinca at *EMS* is not always synchronous and this has important implications for mainstem flows (as per Pattinson et al., 2014). For instance, localised thunderstorms in the lower part of the Bellos are often not registered as increases in *Q* at *LMS* (i.e., the highest event registered during the study period had similar values at *EMS* (297 m^3/s) and *LMS* (260 m^3/s).

Although the sediment load increases from *LMS* to *EMS* (67 % i.e. + 58 t/km^2), the change is not as marked as that for runoff, indicating the supply limited condition of this reach combined with the dilution effect of the clear water released during hydropeaking. The *SSY* computed for the 200 km^2 increment in of the catchment area between *LMS* and *EMS* is still low compared to other catchments with similar characteristics. The main source of sediments in this reach is the channel itself together with the inflow from the Bellos (193 km^2, 96 % of the increment between *LMS* and *EMS*).

Despite its importance in this sub-reach, the role of the Bellos as a sediment source in the Cinca catchment as a whole should be considered moderate. While SSC reached up to 3 g/l several times at *LMS*, maximum *SSC* at *EMS* was around 2 g/l and it was registered just once during the study period. This difference indicates the dilution effect of hydropeaking and the Bellos, and that the sediment supply from the latter never reaches a sufficient level to alter the magnitude of *SSC*, hence *SSL*, further down at *EMS*.

(c) Finally, the main changes in the 6.5 km-long lowermost sub-reach (between *EMS* and *AMS*) are related to the river's sediment load. At *AMS* the catchment area increases 52 km^2 compared to *EMS* and the *WY* increases 14 % (i.e. 1.83 hm^3/km^2), but the sediment load increases 116 % (i.e. up to 650 t/km^2). While the *SSY* associated with this 52 km^2 of the catchment is very high, it still remains low compared to small catchments draining highly erodible badlands (e.g. 15,300 t/km^2 for the Araguás catchment, Nadal-Romero et al. 2007 and up to 2400 t/km^2 for the subcatchments in the Isábena, López-Tarazón et al., 2011). The ratio of increment for *WY* is smaller than the one computed at *EMS*, but is one order of magnitude larger for *SSL*. These values indicate that the new catchment area between *EMS* and *AMS* does not alter *WY* significantly but provides a large amount of sediment compared to sub-reaches further upstream. Changes in sediment availability here are related to the supply from small streams draining badlands, and the changes in river bed structure and in-channel sediment availability resulting from gravel extraction and associated channel maintenance works. The San Vicente Creek is the main tributary in this lowermost part of the study reach (Figure 1C). This ephemeral stream drains an area of badlands, which can be considered the main sediment source for the mainstem Cinca here. Other small tributaries (e.g. Forcas, Figure 1C) may also be relevant at certain times (i.e. floods), but have limited influence on the annual and long term sediment loads.

Two periods of channel maintenance occurred during the study period (see Figure 1) and another one (at the confluence of the San Vicente) was completed few months before the study commenced. Such works are usually performed under low flow conditions, to facilitate access of machinery to the channel and the extraction of gravels. Béjar et al. (2018) found that most of the sediments entrained and transported in suspension during these works were deposited within the 1.5-km downstream reach. Sediments were then readily available for remobilisation during subsequent competent events. Effects of gravel extraction and related channel operations on *SSL* have been reported to be variable in time and space (Kondolf, 1997). Additionally, the grain-size distribution of the bed, together with its structure (e.g. imbrication, armouring) is modified by mining, and in turn such changes affect in-channel sediment availability. No data on the volumes of fine sediment delivered to the study reach as a result of mining are available, so it is difficult to quantitatively assess the relative contribution of the different factors influencing loads observed during the present study period. Béjar et al. (2018) reported that the *SSL* associated with maintenance works (during stable low flows) was similar to that during flood events, while the *SSY* results for the lowermost sub-reach are in the line of many other reporting the effects of in-channel mining on sediment budgets (e.g. Lagasse et al., 1980 in the River Mississippi in 1968; Rovira et al., 2005 in the Mediterranean River Tordera in 1997; Zabaleta et al., 2016, in the Bay of Biscay).

The absolute flow and sediment transport values observed in the upper Cinca cannot be extrapolated directly to other catchments. Nonetheless, many of the natural and anthropogenic factors controlling flow and sediment transport in this catchment are evident in other montane areas, so the magnitude of changes in water and sediment yield between sub-reaches can be considered broadly indicative of the magnitude of hydrological and geomorphic change associated with such pressures.

5.4. The sediment budget of the upper Cinca and reservoir siltation

Downstream from the study reach, the Mediano Reservoir collects water and sediments from the upper River Cinca and the River Ara, as well as from a number of small tributaries that flow directly into it. Although there is much variation, the average *SSY* of the Cinca at the entrance of the reservoir is 75 $t/km^2/y$. Unpublished data suggest that the Ara may double the SSY of the Cinca at *AMS* (i.e. around 160 $t/km^2/y$). Moreover, according to the average sedimentation rate of the reservoir (Avendaño et al., 1997; Cobo, 2008; and Batalla and Vericat, 2011), the *SSL* is estimated at around 33,7000 t/y, yielding an annual SSY of 162 $t/km^2/y$. This value matches to the one estimated for the Ara but is significantly higher than that for the upper Cinca. Taking into account all these SSY values, it can be estimated that small tributaries flowing directly into the reservoir may yield ca. 550 $t/km^2/y$. This is a high value, similar for instance to the *SSY* of the River Isábena which contains extensive badland areas. According to Batalla and Vericat (2011) the SSY of rivers in the Ebro catchment range from 3 to more than 2000 $t/km^2/y$ (estimates based on reservoirs sedimentation data from CEDEX). Thus, the current *SSY* of the catchments draining into the Mediano Reservoir sits within the middle to lower range of that of the whole Ebro catchment.

The upper Cinca appears to be the system that contributes least to siltation of Mediano. This reservoir does not experience the major siltation reported for other reservoirs in the Pyrenees (e.g. Valero-Garcés et al., 1999). Its annual loss of capacity has been estimated at 0.07 %, compared to an average of 0.3 % for other reservoirs within the Ebro catchment. Both of these values are still lower than estimated for dams worldwide (between 0.5 and 1 % on average; Batalla and Vericat, 2011). However, it is worth emphasising that data reported here do not include the coarse fraction of sediment transported by the River Cinca. This this fraction may represent an appreciable proportion of the total sediment yield in highly dynamic mountainous fluvial systems so its influence on dam capacity warrants further study

6. FINAL REMARKS

This study aimed to assess how natural and anthropogenic factors influence suspended sediment dynamics within a 10 km-long reach of a mountain river. Results indicate that localised human activities and tributaries result in downstream discontinuities which produce a series of distinct river sub-reaches. The uppermost sub-reach was characterised by comparatively low sediment load and a torrential flow regime, with water and suspended sediment mostly transported during flood events. This character is typical of systems with a Mediterranean climate. Further downstream, hydropeaking and inflow from a major tributary increased water yield by almost 5 times over a distance of 3.5 km. Under the limited sediment supply conditions here, the increase in discharge exerted a dilution effect on the sediment load. In the most downstream sub-reach, gravel mining associated with channel maintenance works, together with the input from another tributaries, increased sediment load to up to 2.5 times compared to the upstream sections.

Quantitative data presented for the upper Cinca, a river representative of many Pyrenean catchments, provide insights into the influence of natural processes and anthropogenic pressures on fluvial dynamics. More broadly, the study indicates how natural and anthropogenic factors can interact in ways that, due to their specific spatial location and temporal variability, can alter runoff and sediment load over relatively short channel distances. Catchment characteristics (e.g. water supply, sediment availability, land cover) and human activities such as instream mining and hydropeaking exert complex effects, sometimes additive and sometimes compensatory. As these effects can result in marked downstream discontinuities, accurate delineation of river reaches is needed before more complex analysis of river processes are undertaken.

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CHAPTER 4

EFFECTS OF GRAVEL MINING ON SUSPENDED SEDIMENT TRANSPORT IN MOUNTAIN RIVERS

EFFECTS OF GRAVEL MINING ON SUSPENDED SEDIMENT TRANSPORT IN MOUNTAIN RIVERS (UPPER RIVER CINCA, CENTRAL PYRENEES)

This chapter contains the following accepted and already online published paper in the journal *Cuadernos de Investigación Geográfica. SJR* H Index (2016): 10. Category: Geography, Planning and Development; 1st Quartile.

Béjar, M., Vericat, D.,Nogales,I., Gallart, F. and Batalla, R. J.,2017. Effects of gravel mining on suspended sediment transport in mountain rivers (upper River Cinca, Central Pyrenees). *Cuadernos de Investigación Geográfica*. DOI: 10.18172/cig.3256

ABSTRACT: This paper examines the effects of *channel maintenance works* (including gravel mining) on the suspended sediment transport in the upper River Cinca (Central Pyrenees). Discharge, sediment transport and sediment grain-size distribution were measured, sampled and further determined in five monitoring sections along a 5-km river reach. Samples were taken at a section upstream from the mined area (i.e. reference section) and in four sections downstream (0, 200, 500 and 1500 meters). The results show that sediment concentrations downstream from the target site were up to one order of magnitude higher than in the reference section. Average concentrations during the impact were similar to those observed during floods, with maximum values attaining 6 g/l. Total load ranged from 2.2 to 17 Mg/day between sections; these values include the effects of the earth-moving works performed before the maintenance works started. Concentrations at the lowermost section of the study reach were similar to those observed at the upstream reference section, suggesting that most of the suspended material was deposited in the channel. The transported material was coarse than that sampled under reference conditions; nevertheless, largest particles settled quickly, suggesting a selective transport downstream. This study constitutes a first step towards a better understanding of local sediment dynamics in rivers affected by maintenance works and related activities, such as gravel mining, and, overall, supports the comprehensive assessment of the effects of human actions on channel morphodynamics and the ecological functioning of mountain fluvial systems.

KEY WORDS: channel maintenance works, gravel mining, suspended sediment transport, grain-size distribution, deposition, River Cinca.

1. INTRODUCCIÓN

Los ríos mantienen un balance sedimentario donde la cantidad de sedimento procedente de las zonas de erosión es equivalente a la cantidad de sedimento que reciben las zonas de sedimentación. Por tanto, la transferencia de sedimentos es un proceso continuo a lo largo de la cuenca y es el resultado de las características hidro-climáticas, litológicas, edafológicas y vegetales, así como de los procesos biofísicos que tienen lugar en la misma (generación de sedimentos, erosión y transporte). Sin embargo, las actividades humanas (i.e. presas y actuaciones en el cauce como la construcción de escolleras y las extracciones de áridos) y otros procesos como los cambios en los usos del suelo de la cuenca, alteran esta transferencia de sedimentos aguas abajo (Williams y Wolman, 1984; García-Anquela et al., 1985; Kondolf, 1994)

Las arenas y gravas transportadas por los ríos se utilizan para la construcción (i.e. áridos) y se extraen en muchos casos de los depósitos aluviales (terrazas, llanura de inundación) o directamente del cauce fluvial. Como consecuencia, las extracciones de áridos alteran la transferencia de sedimento aguas abajo generando un déficit de las fracciones de mayor tamaño (aquellas que forman la arquitectura básica del cauce) e incrementando proporcionalmente la disponibilidad de sedimento fino. Los principales efectos descritos en la literatura incluyen: la alteración de la geometría del canal, la disminución del nivel freático debido al sobredrenaje, el incremento del tamaño de partículas del lecho y el incremento de sedimento fino (p.ej. Kondolf, 1994). Se han realizado numerosos estudios de los efectos geomorfológicos de las extracciones de áridos (p.ej. Rinaldi et al., 2005; Rovira et al. 2005), pero son escasos los trabajos que han analizado específicamente el efecto de estas actuaciones en el transporte de sedimentos finos en suspensión (arcilla, limo e incluso arena; Brown et al., 1998). Durante las operaciones de extracción aumenta el material en suspensión debido a la erosión lateral al extraer material del canal (Warner et al., 1977, al paso de maquinaria por el cauce del río (Forshage y Carter, 1973) y a la remoción de materiales y exposición de sedimentos finos al romper la coraza del lecho (Lagasse et al., 1980). Los efectos no son localizados y se pueden extender en el espacio y en el tiempo (kilómetros aguas abajo e incluso aguas arriba, durante años y décadas) hasta que el río alcanza una nueva condición de equilibrio (Kondolf, 1997).

El incremento de la carga de sedimento fino comporta una notable perturbación de los ecosistemas acuáticos (Wood y Armitage, 1997). Este incremento puede afectar directamente a los organismos e indirectamente al hábitat físico (Jones et al., 2012). Las partículas en suspensión pueden colapsar las agallas de los organismos afectando a la eficiencia de respiración (Bilotta y Brazier, 2008). También pueden causar abrasión sobre los animales (Newcombe y Macdonald, 1991). Además, el sedimento en suspensión se sedimenta aguas abajo alterando las características del sustrato y por tanto la capacidad de refugio de los animales (Buendía et al., 2011), así como procesos esenciales para el ecosistema como la freza de salmónidos (Waters, 1995). La presencia de sedimento también puede alterar los niveles de oxígeno y desencadenar otros cambios en la composición química del agua (Pretty et al., 2006). Del mismo modo, el aumento de la turbidez puede desencadenar cambios físicos como la reducción de la penetración de luz, limitando el crecimiento algal (Parkhill y Gulliver, 2002). Por tanto, cuantificar el impacto de estas actuaciones en la dinámica hidro-sedimentaria es fundamental para entender la dinámica en los ríos afectados por actividades humanas y, en la medida de lo posible aportar información a los organismos gestores para que puedan anticiparse a los posibles efectos ambientales y diseñar e implementar las actuaciones de mitigación necesarias dentro de los programas de gestión de cuencas. En este contexto, el objetivo de este trabajo es analizar los efectos de una extracción de áridos sobre el transporte de sedimento en suspensión, evaluando los cambios de las concentraciones, las cargas sedimentarias y las características granulométricas durante la actuación, y determinando el alcance espacial de dicho impacto. El estudio se realiza en un tramo del alto Cinca (Pirineo Central), afectado históricamente y hasta la actualidad por distintos tipos de actuaciones en el cauce, singularmente extracciones de áridos. Cabe destacar que la extracción de áridos monitorizada en este estudio se realiza en el marco de una actuación en el cauce para la prevención de inundaciones.

2. ÁREA DE ESTUDIO

2.1 El tramo alto del Cinca

El río Cinca tiene un régimen nivo-pluvial y es uno de los mayores tributarios del Ebro. El tramo alto del río Cinca se sitúa en el noreste de la Península Ibérica en la vertiente sur de los Pirineos (Figura 1A). La cuenca del alto Cinca tiene una superficie de 848 km2 y un gradiente altitudinal que se extiende desde Monte Perdido (3355 m snm) hasta Ainsa (522 m snm; Figura 1B). La aportación media anual de agua es de 587 hm3, lo que supone un caudal medio en el tramo de estudio de 29 m^3/s (según datos en la estación de aforos A051 de Escalona entre 1959-2015; ver Figura 1C). El tramo está sujeto a hidropicos esporádicos que no exceden los 25 m^3/s (i.e. originados desde la central hidroeléctrica de Laspuña, 8 km aguas arriba del tramo de estudio). Estos hidropicos alteran la hidráulica del canal pero no son suficientemente competentes para arrastrar materiales del lecho. La magnitud de las crecidas anuales excede los 225 m^3/s. La pendiente media del tramo de estudio es 0,007 m/m, el ancho del canal activo promedia 200 m y la profundidad media es 0,4 m para caudales bajos (i.e. 10 m^3/s). En las últimas décadas la morfología del tramo ha experimentado una reducción en su complejidad, pasando de un patrón trenzado a un patrón sinuoso con canales más estables (Llena, 2015). Los sedimentos del lecho están poco clasificados (i.e. Índice de dispersión IF&W = 1,1; Folk y Ward 1957), el rango de partículas alcanza desde grandes bloques a gravas finas (> 2000 a < 8 mm). La mediana de la distribución granulométrica superficial (D50-s) es de 67 mm (i.e. río de transición entre gravas y cantos). La concentración media de sedimentos en suspensión para el periodo 2013-2015 es 0,09 g/l, alcanzando un máximo instantáneo de 17 g/l asociado a un caudal de 48 m^3/s durante la crecida del 31/07/2015. Se trata de una concentración similar a las obtenidas por Lobera et al. (2016a y 2016b) en la vecina cuenca del río Ésera.

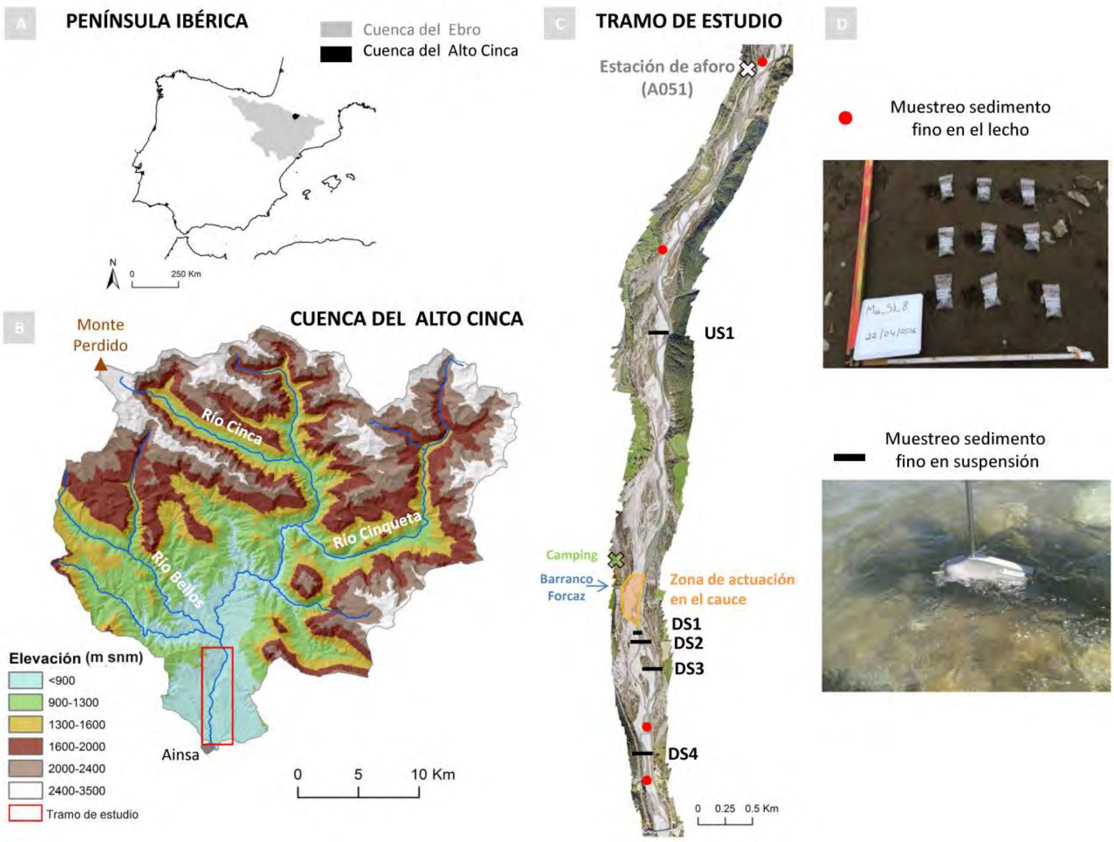

Figura 1. Área de estudio y métodos aplicados. (A) Localización de la cuenca del alto Cinca en la Península Ibérica. (B) Localización del tramo de estudio (rectángulo rojo) en la cuenca del alto Cinca. (C) Tramo de estudio y localización de las secciones y puntos de muestreo. (D) Detalle del muestreo de sedimento fino en el lecho y la adquisición de sedimento en suspensión mediante un toma muestras de integración en profundidad US DH48.

2.2. Extracciones de áridos

El alto Cinca ha experimentado extracciones de áridos desde mediados del siglo XX. En la actualidad las extracciones de áridos se enmarcan dentro actuaciones en el cauce activo para la prevención y mitigación de los efectos de las crecidas (i.e. defensas de márgenes, re-ubicación del cauce). La actuación estudiada tuvo lugar en agosto de 2014 en el término municipal de Labuerda. La zona de la actuación se localiza en la desembocadura de un afluente, barranco Forcaz donde, además, el cauce se encuentra próximo a un camping (Figura 2A). La figura 2 ilustra las distintas fases de la actuación en base a los objetivos de la misma: (a) alejar el cauce activo del camping y (b) extraer los sedimentos ubicados en una barra activa del cauce para disminuir la cota del cauce y reducir así el riesgo de inundación. La actuación se programó en tres fases: 1) una primera fase de preparación de un cauce alternativo (Figura 2B), en la que se extrajeron sedimentos para la creación del cauce alternativo; 2) una segunda fase donde se conectó el extremo superior del cauce alternativo con el cauce principal y se derivó el agua hacia el nuevo cauce (Figura 2C) y, finalmente, 3) una tercera fase en la que se extrajo material de la barra activa del cauce (Figura 2D). El aporte de sedimento fino se produjo sobre todo durante el primer día de la primera fase (día 6; Figura 3A) y durante el primer día de la segunda fase (día 18; Figura 3B). La tercera fase se llevó a cabo en cauce seco y por tanto no aportó sedimento fino de manera directa al cauce.

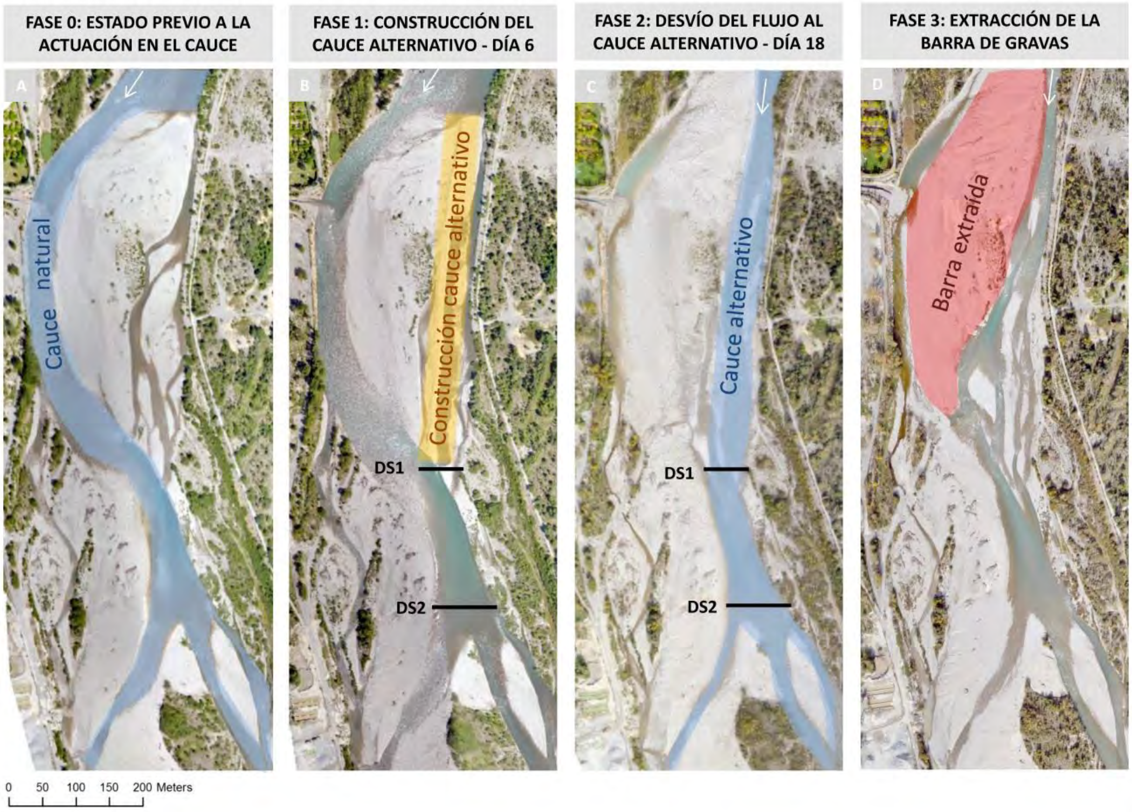

Figura 2. Fases de la actuación en el cauce analizada en un tramo del alto Cinca en agosto de 2014. (A) Fase 0: Estado previo de la actuación. (B) Fase 1: Construcción del cauce alternativo en el margen izquierdo. Esta fase implica la extracción de materiales para dicha construcción. (C) Fase 2: Apertura del cauce alternativo y desviación del flujo. (D) Fase 3: Extracción de áridos en la barra diagonal. Nota: La flecha blanca indica la dirección del flujo

Figura 3. Fotografías durante los trabajos de adecuación previos a la actuación en el cauce (extracción de áridos) en un tramo del alto Cinca en agosto de 2014. (A) Pluma de sedimento en suspensión durante la creación del cauce alternativo el día 6 de agosto, corresponde a la máxima concentración registrada. (B) Detalle de la erosión lateral durante la apertura del cauce alternativo el día 18 de agosto.

3. MÉTODOS

3.1. Caudal e hidráulica

El caudal se obtuvo a partir de los registros de la estación de aforos de Escalona (A051, Confederación Hidrográfica del Ebro) ubicada 1,5 km aguas arriba de la zona de actuación (Figura 1C). Un limnígrafo obtiene medidas cada quince minutos de nivel de agua (h) que posteriormente se transforman a caudal (Q) mediante una curva de gasto h/Q. El modelo hidráulico 2D Iber® ha sido utilizado en el tramo de estudio para caracterizar la hidráulica durante las actuaciones en el cauce. El modelo utiliza datos topográficos y ecuaciones de dinámica de fluidos para simular la lámina de agua en ríos y estuarios (para más detalles consultar Bladé et al., 2014). La parametrización del modelo se ha realizado usando un Modelo Digital del Terreno de 1 metro de resolución (ver Vericat et al., 2016) y un solo valor de rugosidad (n = 0,035) basado en los datos granulométricos del tramo). La modelización se ha hecho para un caudal constante 7 m^3/s con el objetivo de obtener valores de hidráulica característicos en el tramo.

3.2. Medición del transporte de sedimentos

El muestreo del sedimento en suspensión se llevó a cabo en cinco secciones del tramo de estudio, una sección aguas arriba (US1) y cuatro aguas abajo (DS1-4; Figura 1C). La sección aguas arriba se utilizó como control. La primera sección aguas abajo (DS1) se localizó inmediatamente aguas abajo de la zona alterada. Las secciones restantes se situaron a 0,2, 0,5 y 1,5 km de la zona de actuación. Durante los dos días en los que el cauce fue alterado (día 6 y día 18), se realizó un muestreo horario en las secciones más próximas a la actuación (DS1-DS3). El muestreo se llevó a cabo mediante un muestreador de integración en profundidad US DH48 (Figura 1D) en aquellas secciones donde las diferencias de turbidez en la sección eran evidentes (DS1-DS3). Cada muestra se compuso de tres sub-muestras tomadas en diferentes puntos a lo ancho de la sección. De este modo, las muestras representan las concentraciones medias en la sección ya que cada muestra integra la CSS (Concentración de Sólidos en Suspensión) en tres puntos representativos a lo ancho de la sección y cada punto integra la CSS en la columna de agua (i.e. vertical). En las secciones US1 y DS4, donde el transporte era más homogéneo, se muestreó únicamente en el centro de la sección con un intervalo de tiempo mayor entre muestras (3-4 horas). En total se obtuvieron 300 muestras de 1 litro. Las muestras se filtraron (filtros Filter-Lab, 1,2 µm tamaño de poro), se secaron y se pesaron en el laboratorio para determinar la concentración de sedimentos en suspensión (CSS en g/l).

3.3. Obtención de las granulometrías de los sedimentos finos

Las muestras retenidas en los filtros fueron re-suspendidas en agua para su posterior análisis por vía húmeda. La granulometría se determinó usando un analizador láser de tamaño de partículas (Malvern Mastersizer®/E) que permite analizar la distribución granulométrica de las fracciones entre 0,02 µm (arcillas) y 500 µm (arenas medias). Del total de las muestras obtenidas se caracterizaron setenta y dos (localizadas en DS1-DS3). El resto no se pudieron analizar debido a la falta de material suficiente para su detección.

Además, se tomaron muestras granulométricas del lecho en cuatro puntos a lo largo del tramo de estudio y en una sola ocasión (22 de abril de 2016). En total, se muestrearon dos puntos aguas arriba y dos puntos aguas abajo de la zona de actuación (Figura 1C). En cada punto se seleccionó un parche superficial de sedimento fino y se tomaron 10 réplicas del material (Figura 1D). En total se tomaron cuarenta muestras de aproximadamente 15 g cada una. Las muestras se secaron, tamizaron y pesaron. Las partículas de 2000 µm (arenas muy gruesas) a 500 µm (arenas medias) se tamizaron en una torre granulométrica de la marca Filtra®. Las partículas más finas de 500 µm se dispersaron en agua agitando durante seis horas y posteriormente se procesaron siguiendo el mismo procedimiento detallado anteriormente.

3.4 Análisis de datos

El cálculo de la carga sólida total (CST) se realizó para el intervalo de 09:00 a 19:00 horas ya que son las horas de trabajo en la zona de actuación. Para cada sección de muestreo, la carga total se calculó como la suma de las cargas de cada intervalo de muestra de agua (CST t=i). Para ello CST t=1 (t) fue calculado multiplicando la CSS t=1 (g/l) obtenida de cada muestra de integración en profundidad por el caudal circulante (Q t=1; l/s) por el intervalo de tiempo entre muestras (t2-t1, s).

Durante el día 6 y en el caso particular de DS1, se tuvieron que realizar cálculos proporcionales de caudal ya que la sección estaba inicialmente dividida en un brazo principal (i.e. el canal natural) y uno secundario (i.e. donde se construiría el cauce alternativo; ver Figura 2B). De este modo, durante el día 6 se estimó un caudal circulante en el brazo secundario (basado en observaciones de campo) del 10 % del caudal total. En esta misma sección el día 18, el caudal varió a medida que se desviaba el flujo hacia el cauce alternativo (Figura 2C). Por tanto, los cálculos se han realizado en base al caudal aproximado que se estimó en campo (observaciones de anchura, profundidad y velocidad superficial) para cada intervalo de muestreo. El caudal representaba el 10 % del total al inicio del muestreo, mientras que al finalizar el muestreo el caudal en el cauce alternativo representaba el 90 % del total.

Las distribuciones granulométricas de las muestras de sedimento fino del lecho se truncaron por ambos extremos elaborándolas para el intervalo 1,2-2000 µm. De este modo los valores estadísticos de las curvas son comparables con las distribuciones granulométricas del material en suspensión muestreado durante las actuaciones en el cauce.

4. RESULTADOS Y DISCUSIÓN

4.1. Hidrología e hidráulica

Durante el mes de los trabajos de adecuación y extracción (agosto de 2014) el caudal medio fue relativamente bajo y estable (igualado o excedido el 85 % del tiempo en la curva de frecuencia de caudal para el período 1959-2016, datos sin publicar), cercano a los 9 m^3/s (CV= 0,4, donde CV es coeficiente de variación). El caudal para todo el año hidrológico 2013-2014 osciló entre 5,2 m^3/s y 477 m^3/s (representado éste último valor un caudal pico de crecida con un periodo de retorno de 5 años según Llena, 2015). Los caudales durante los días de estudio (días 6 y 18) fueron 6,9 y 7,0 m^3/s respectivamente (para más detalles ver Béjar et al. 2017). Estos valores son igualados o superados un 93 % del tiempo. Los resultados preliminares (datos sin publicar) del modelo hidráulico 2D Iber® indican que la profundidad media del cauce en todo el tramo de estudio durante los días 6 y 18 fue de 0,3 m (CV = 0,7 m), y la velocidad media 0,7 m/s (CV = 0,6). Los caudales de esos días no produjeron ningún movimiento apreciable del material grueso del lecho.

4.2. Transporte de sedimentos

4.2.1. Dinámica temporal y espacial

La CSS en la sección de control (i.e. US1) osciló entre 2 y 50 mg/l, con un promedio de 20 mg/l (CV = 1,2) durante los días 6 y 18 (Tabla 1). Estos valores se consideran normales si se comparan con los promedios obtenidos en DS4 para los seis meses anteriores a la actuación (41 mg/l, datos sin publicar). En cambio, aguas abajo de la actuación la Css media fue un orden de magnitud mayor y mostró una mayor variabilidad (CSS = 0,2 g/l; CV = 3,8).

Durante el día 6 la CSS incrementa puntualmente en la sección más próxima a la actuación (DS1), alcanzándose un máximo de 5,8 g/l (Figura 4A; Tabla 1). Este incremento se debe probablemente a la liberación de materiales finos durante la construcción del cauce alternativo en el margen izquierdo (Figura 3A). En las siguientes secciones (DS2, DS3, DS4), las concentraciones fueron menores alcanzándose máximos de 0,07, 0,02, y 0,02 g/l respectivamente (Tabla 1). La disminución de casi dos órdenes de magnitud en las CSS entre DS1 y DS2 (a 200 m de distancia) indica una probable sedimentación de la mayoría del material movilizado durante las actuaciones llevadas a cabo en el cauce en ese día.

Durante el día 18 se observó una dinámica completamente diferente (Figura 4B). La CSS incrementó aguas abajo registrándose el máximo (1,30 g/l) en la sección situada a 0,5 km de la actuación (DS3). Nuestra hipótesis es que esta recarga aguas abajo durante el día 18 responde a la movilización y tránsito de los materiales en el lecho producto de la fuerte sedimentación observada el día 6, y en gran medida relacionada con la apertura del cauce alternativo que moviliza el sedimento disponible en el lecho debido a la concentración del flujo (Figura 2C y 3B). En la sección más alejada (DS4) la CSS máxima fue 0,003 g/l. El patrón en DS4 es similar al observado en el día 6, indicando que el sedimento movilizado no se mantiene en suspensión a lo largo de los 1,5 km del tramo de estudio. Otros autores han reportado el mismo efecto, con plumas de sedimento durante actuaciones en el cauce en condiciones de caudal estable a lo largo de los primeros 500 metros (p.ej. Bryce, 1977; Mori and Braceli, 2011).

Las CSS medias alcanzadas durante los días de estudio son similares a los valores registrados durante crecidas naturales (p.ej. la CSS media durante un evento el 02/08/14 fue de 0,4 g/l). Sin embargo, la CSS máxima durante el periodo de estudio fue el doble que la CSS máxima alcanzada en el mismo evento (i.e. 2,6 g/l). Si se comparan la Css máxima obtenidas durante las actuaciones en el cauce con las obtenidas en eventos de crecida en el cercano río Ésera, la CSS máxima es mucho menor (i.e. CSS máxima en verano 27,8 g/l, Lobera et al. 2016b). Es interesante señalar también que las altas concentraciones alcanzadas durante los días de actuación en el cauce alteraron por ejemplo la deriva de macroinvertebrados en el tramo de estudio (para más detalles consultar Béjar et al., 2017).

Tabla 1. Concentración de sedimento en suspensión (C_{SS}) analizado durante una extracción en el alto Cinca en agosto de 2014.

Día	Sección[1]	Concentración de Sedimentos en Suspensión (g/l)		
		C_{SS} media	C_{SS} máxima	C_{SS} mínima
06/08/2014	US1	0,004[2]	0,004	0,004
	DS1	0,419	5,806	0,014
	DS2	0,017	0,070	0,006
	DS3	0,009	0,022	0,006
	DS4	0,008	0,023	0,004
18/08/2014	US1	0,021	0,054	0,010
	DS1	0,181	0,610	0,002
	DS2	0,122	0,842	0,002
	DS3	0,158	1,341	0,005
	DS4	0,002	0,003	0,001

[1] Ver la figura 1 para localización de las secciones.
[2] Solo se tomó una muestra, por tanto solo se tiene un valor de C_{SS}.

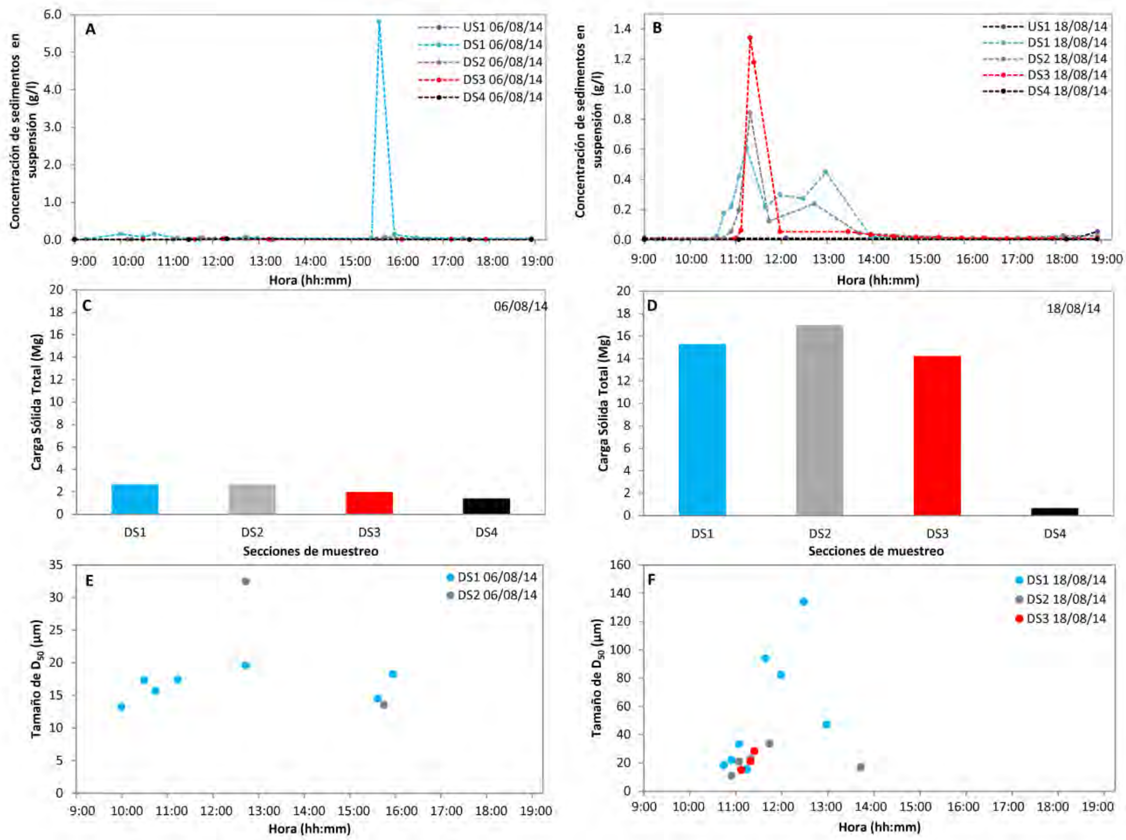

Figura 4. Transporte de sedimento en suspensión durante los trabajos asociados a la extracción de áridos en las secciones de muestreo localizadas aguas abajo de la extracción. (A) Concentración de sólidos en suspensión durante el día 6 de agosto. (B) Concentración de sólidos en suspensión durante el día 18 de agosto de 2014 (Nota: el eje vertical está modificado). (C) Carga sólida en suspensión durante el día 6 de agosto. (D) Carga sólida en suspensión durante el día 18 de agosto. (E) Tamaño medio de las partículas en suspensión durante el día 6 de agosto. (F) Tamaño medio de las partículas en suspensión durante el día 18 de agosto.

4.2.2 Carga sólida

La carga sólida (CST) diaria durante los dos días de muestreo varió entre 0,7 y 17 Mg. En la sección de referencia (US1) se registró una media de 1,8 Mg (CV = 0,7), un orden de magnitud inferior que el valor máximo observado aguas abajo de la actuación.

Durante el día 6 la CST media fue 2,2 Mg en las secciones situadas aguas abajo de la actuación (Figura 4C). El máximo (2,7 Mg) se registró en los primeros 0,2 km (DS1 y DS2). Sin embargo, es importante remarcar que durante este día el caudal circulante por DS1 representaba sólo un 10 % del caudal total (Figura 3A), hecho que demuestra el diferente grado de impacto de las actuaciones en el cauce en la carga sedimentaria en ambas secciones. Durante el día 18 la carga total media en las secciones DS1 a DS4 fue un orden de magnitud superior al estimado durante el día 6 (11,8 Mg; Figura 4D). La carga sólida circulante por las secciones situadas en los primeros 0,5 km (DS1-DS3) fue similar (CV = 0,1) y el máximo de carga se alcanzó en DS2 (17 Mg).

Es interesante señalar que mientras la CSS máxima se registró en DS1 el día 6 (sección 4.2.1) la CST máxima se observó en DS2 durante el día 18. En la Figura 4A se muestra como la CSS máxima el día 6 es un valor puntual precedido y sucedido por valores próximos a 0. Sin embargo, en la figura 4B se observa como las CSS durante el día 18 fueron de menor magnitud pero con una concentración media elevada y mantenida en el tiempo (hasta 4 horas). Este hecho tiene efectos en el balance sedimentario pero también en la vida acuática. Newcombe y Macdonald (1991) ya demostraron como la biota acuática responde tanto a los cambios puntuales en la CSS como a la duración de la exposición. En el caso del Cinca dichos efectos se describen en el trabajo recientemente publicado por Béjar et al. (2017).

La carga media durante los dos días de actuación en la sección más lejana de la actuación (DS4) fue de 1 Mg, valor similar al obtenido en la sección de referencia (US1) para el mismo periodo. Este hecho sugiere que la pluma de sedimento generada por las maquinas al extraer gravas no impactó a DS4, indicando que todo el material movilizado durante los días 6 y 18 se sedimentó en el cauce intermedio entre secciones, quedando como una fuente local de sedimento para próximas crecidas. La carga máxima registrada durante los días de estudio representa un 10 % de la carga alcanzada durante eventos naturales en la misma época del año. Por ejemplo, la carga movilizada durante el evento del 02/08/14 en condiciones similares de duración y caudal (10 horas, caudal medio 12 m^3/s) fue de 227 Mg.

4.3. Textura del sedimento en suspensión

El sedimento movilizado durante el día 6 en las secciones más próximas a la actuación presentó una granulometría limosa (D50-medio = 18 µm, con un rango entre 13 y 33 µm; Tabla 2). El D50 máximo se obtuvo en la sección DS2 durante un periodo de baja concentración (Figura 4E). Durante el día 18, la distribución del tamaño de partículas fue más variable en el tiempo y también entre secciones (Figura 4F). En la sección más próxima a la actuación (DS1) fue donde se observó una mayor heterogeneidad granulométrica (IF&W = 2,6; Tabla 1) a partir de la apertura del cauce alternativo, así como un incremento en el tamaño medio de partícula, pasando de limos a arenas finas a medida que transcurría la intervención en el cauce (Tabla 2). Estos resultados responden a los procesos de erosión y transporte observados tras la apertura del cauce alternativo (Figura 3B) que movilizaron materiales peor clasificados y más gruesos que los que normalmente se mueven en suspensión durante crecidas naturales. En las secciones más alejadas (DS2 y DS3) el rango del tamaño medio de partículas es similar al observado durante el día 6 (11-34 µm; Tabla 2). Además, en estas dos secciones se observa una tendencia a la homogeneización granulométrica con tan solo un valor de IF&W por encima del límite entre clasificación pobre y muy pobre (IF&W = 2,1). Globalmente los resultados indican un lavado selectivo de los materiales transportados aguas abajo (i.e. las partículas de mayor tamaño tienden a sedimentar en los primeros metros aguas abajo del impacto como consecuencia de la falta de caudales competentes para su transporte). Es importante destacar este efecto en la dinámica espacial de las concentraciones durante los dos pulsos del día 18 (Figura 4B) i.e. el primer pulso, con partículas más finas (Figura 4F), alcanza las secciones más alejadas y además se produce una movilización del sedimento depositado en los días previos, lo que condicionaría la magnitud de las Css; sin embargo, durante el segundo pulso, las partículas tienen un mayor tamaño y el incremento de CSS no alcanzó las secciones más alejadas debido al lavado selectivo (Figura 4F).

Las texturas del sedimento transportado durante las actuaciones en el cauce fueron ligeramente superiores a las texturas promedio obtenidas durante crecidas (p.ej. en una crecida de Junio 2015 el D50-medio = 13 µm; Nogales, 2016; Figura 5). Puntualmente, en la sección más próxima a la actuación, el tamaño medio transportado llegó a ser un orden de magnitud mayor que la media en crecidas (D50-máximo = 134 µm). El 80 % de las partículas en suspensión muestreadas durante las actuaciones pertenecen a fracciones limosas. Este porcentaje es similar al obtenido para las partículas transportadas en suspensión durante crecidas naturales (Figura 5). No obstante, es importante destacar que en la sección DS1 durante el día 18 el sedimento muestreado era más grueso, de modo que la fracción limosa solo llegó a representar el 64 % de la muestra. Julien (1998) evalúa el modo de transporte de sedimento en función del ratio entre la velocidad de sedimentación (Tabla 2) y la velocidad de corte (0,176 m/s, para los parámetros estimados en los modelos hidráulicos (pendiente media = 0,007 m/m y profundidad media = 0,4 m). Según el rango de partículas observado (D50 = 13 - 134 µm), todas las partículas se transportarían en suspensión. Sin embargo las partículas de mayor tamaño quedan muy cerca del límite de transporte mixto. Estos resultados indican que las actuaciones en el cauce causaron la movilización de sedimentos de mayores tamaños que en condiciones naturales pero, como se ha indicado, estos materiales transportados en suspensión fueron sedimentados en los primeros metros aguas abajo debido a la ausencia de caudales competentes.

Es interesante destacar que el sedimento movilizado durante los días de la actuación es, en general, de menor tamaño que el material de los parches de finos del lecho (Figura 5), aunque puntualmente, los tamaños transportados llegan a alcanzar valores similares a los del lecho (i.e. D95-lecho = 796 µm; D95-transportado = 745 µm durante el día 18 en DS1; Figura 5). Estas fracciones de mayor tamaño representan materiales que no son transportados frecuentemente. Por tanto, los trabajos en el cauce ocasionan puntualmente el transporte de materiales de mayor tamaño procedentes del lecho que no son transportados frecuentemente durante crecidas, donde los sedimentos transportados en suspensión son más finos y proceden mayoritariamente de la cuenca. Tanto la sedimentación selectiva aguas abajo como la movilización de materiales de mayor tamaño, aunque afecte sólo a pequeños tramos del cauce, altera la porosidad de lecho y puede causar modificaciones en el hábitat, afectando a las comunidades bentónicas (Lisle, 1989; Buendia et al., 2011; Jones et al., 2012).

Tabla 2. Descriptores sedimentológicos del material en suspensión analizado durante una extracción en el alto Cinca en agosto de 2014.

Día	*Sección*	*Hora*	*D_{50} (µm)*	*Velocidad Sedimentación[1] (cm/s)*	*Clasificación Wentworth*	*Índice de dispersión[2]*	*Grado de clasificación[2]*
06/08/14	DS1	10:00	13,2	0,06	Limo	1,43	Pobre
		10:30	17,3	0,09	Limo	1,50	Pobre
		10:45	15,6	0,08	Limo	1,53	Pobre
		11:15	17,4	0,10	Limo	1,43	Pobre
		12:45	19,6	0,12	Limo	1,63	Pobre
		15:40	14,5	0,07	Limo	1,49	Pobre
		16:00	18,2	0,10	Limo	1,75	Pobre
	DS2	12:45	32,5	0,33	Limo	1,81	Pobre
		15:48	13,5	0,06	Limo	1,44	Pobre
18/08/14	DS1	10:45	18,1	0,10	Limo	1,56	Pobre
		10:55	21,8	0,15	Limo	1,75	Pobre
		11:05	33,1	0,35	Limo	2,05	Muy pobre
		11:15	15,2	0,07	Limo	1,99	Muy pobre
		11:40	93,7	2,77	Arenas muy finas	2,53	Muy pobre
		12:00	82,2	2,13	Arenas muy finas	2,61	Muy pobre
		12:30	133,8	5,65	Arenas finas	2,63	Muy pobre
		13:00	46,9	0,69	Limo	2,63	Muy pobre
	DS2	10:55	10,7	0,04	Limo	1,39	Pobre
		11:05	20,7	0,13	Limo	1,48	Pobre
		11:20	22,6	0,16	Limo	1,70	Pobre
		11:45	33,5	0,35	Limo	2,05	Muy pobre
		13:45	16,7	0,09	Limo	1,85	Pobre
	DS3	11:08	14,9	0,07	Limo	1,85	Pobre
		11:20	21,0	0,14	Limo	1,57	Pobre
		11:25	28,3	0,25	Limo	1,51	Pobre

[1] Velocidad de sedimentación calculada mediante la ley de Stokes, tomando una temperatura del agua de 15°C
[2] Índice y grado de dispersión según Folk and Ward (1957)

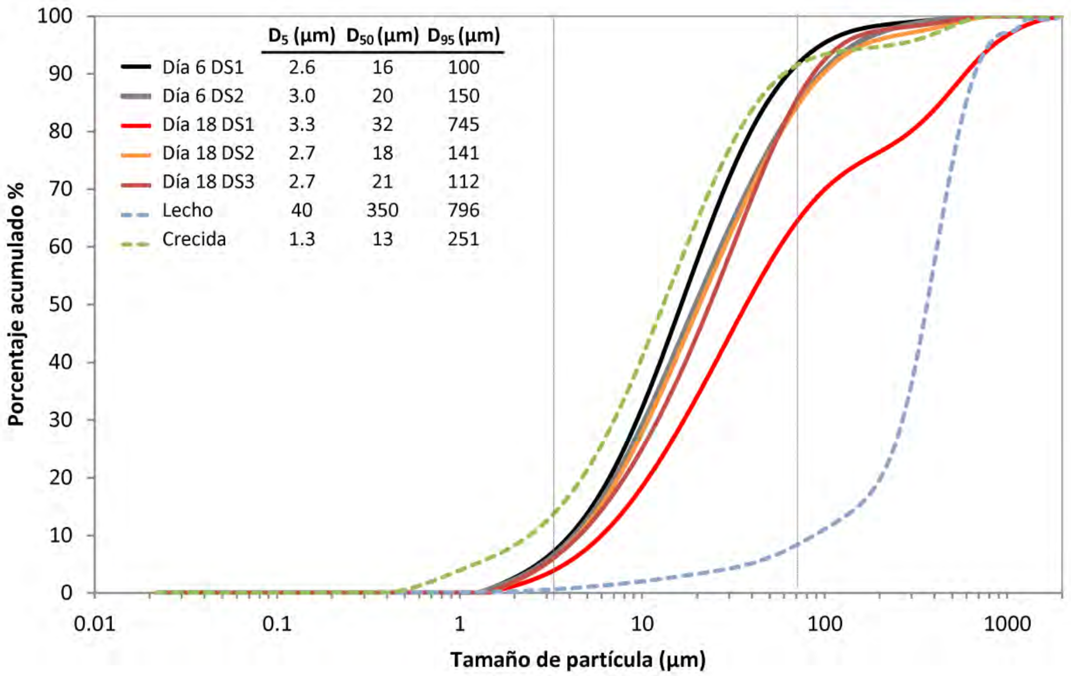

Figura 5. Distribución del tamaño de partículas en suspensión durante los trabajos asociados a la extracción de áridos en un tramo del alto Cinca en agosto 2014. Cabe destacar que se combinaron los datos de todas las muestras pertenecientes a la misma sección al mismo día para obtener las curvas. Las líneas de puntos muestran como referencia la distribución granulométrica de: (i) los parches de sedimento fino del lecho analizados, (ii) el sedimento fino en suspensión en una crecida en Junio 2015 muestreada en la sección DS4 (Nogales, 2016). Las líneas de puntos verticales delimitan las distintas fracciones en la escala Wentworth (arcilla <3,9 µm, limo < 63 µm, arena <2000 µm). En la tabla se muestran los percentiles característicos de cada distribución granulométrica.

5. CONSIDERACIONES FINALES

Este artículo analiza los efectos de una extracción de áridos realizada en el marco de una actuación en el cauce para prevenir inundaciones sobre el transporte del sedimento en suspensión en el río Cinca. Las concentraciones medias observadas durante los días de estudio fueron mayores que en la sección de referencia aguas arriba de la actuación. Los resultados muestran como la actuación en el cauce alteró la dinámica del sedimento, sobre todo en el tramo situado inmediatamente aguas abajo. Los efectos de las actuaciones en el cauce sobre las concentraciones de sedimento y la carga sólida del río ya no se observaban a 1,5 km aguas abajo. La máxima concentración de sedimentos fue similar a las obtenidas en crecidas naturales; por su parte, la carga sólida total transportada como consecuencia de los trabajos en el cauce fue notablemente más baja que en crecidas. Finalmente, cabe señalar que el tamaño de partícula movilizado por la extracción fue similar o puntualmente mayor que la media transportada durante crecidas.

Los resultados muestran como el incremento de la carga sedimentaria debido a los trabajos en el cauce se asemeja a la que se produce durante las crecidas naturales. Este aumento repentino de las concentraciones sin alteraciones sobre la hidráulica conlleva efectos sobre la biota y el hábitat fluvial (p.ej. Weeks et al., 2003). En este sentido, este trabajo muestra la alteración en la duración de la exposición a altas concentraciones, así como la alteración puntual en el tamaño de partícula transportada. Ambos factores son importantes ya que tienen efectos sobre la biota acuática (p.ej. deterioro de órganos respiratorios, abrasión, deriva involuntaria; Bilotta and Brazier, 2008; Béjar et al., 2017). A largo plazo las extracciones de áridos tienen también efectos en el balance sedimentario del tramo fluvial donde se localizan. Las actuaciones en el cauce suelen ejecutarse en época de caudales bajos para facilitar el acceso de las máquinas y aumentar la disponibilidad de material (i.e. cantidad de sedimento expuesto). Estos caudales no son competentes para transportar el sedimento que se moviliza en los trabajos de preparación del área y durante la extracción del material. Consecuentemente, el sedimento movilizado por la extracción se acumula en los primeros metros quedando disponible para crecidas posteriores. Los ríos de montaña no alterados por presas u otros impactos tienen ciclos naturales de sedimento (i.e. alternancia de periodos con cargas más altas y bajas en relación a los procesos de producción, almacenaje y transferencia de materiales en la cuenca). En estos ríos se alternan épocas de producción de sedimento en áreas fuentes (p.ej. meteorización de roca madre por procesos de hielo-deshielo en invierno y primavera), con periodos de transferencia del material al cauce fluvial por las lluvias y el deshielo en primavera y, finalmente, exportación del mismo por crecidas estivales y otoñales. Estos ciclos han estado descritos para cuencas vecinas a las del Cinca en los trabajos de López Tarazón et al. (2011), Piqué et al. (2014), Lobera et al. (2016b). Así, la generación y redistribución de sedimentos producida por las actuaciones en el cauce puede alterar este ciclo natural y algunas funciones ecosistémicas asociadas (como por ejemplo la freza de los peces o la deriva de invertebrados) en función de la época del año en la que se realicen.

Se ha demostrado que los efectos de las extracciones de áridos sostenidas en el tiempo pueden propagarse varios kilómetros aguas abajo (Kondolf, 1994), e incluso aguas arriba (erosión remontante). Por ejemplo, en cuencas donde la transferencia de sedimentos se interrumpe por azudes y presas, el sedimento movilizado por la extracción quedará retenido en estas estructuras, contribuyendo a reducir su vida útil (Gilbert, 1917; Kondolf, 1997). Este estudio presenta un análisis cuantitativo no habitual en estudios sobre impactos fluviales, y supone un primer paso para la comprensión de la dinámica del sedimento en ríos localmente afectados por actuaciones en el cauce, y su influencia en la dinámica morfosedimentaria y la integridad ecológica del río.

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CHAPTER 5

EFFECTS OF SUSPENDED SEDIMENT TRANSPORT ON INVERTEBRATE DRIFT

EFFECTS OF SUSPENDED SEDIMENT TRANSPORT ON INVERTEBRATE DRIFT

This chapter contains the following accepted and already published paper in the journal *River Research and Applications*. JCR-SCI Impact Factor-JCR (2016): 2.274. Category: Water Resources. 1st Quartile.

Béjar, M., Gibbins, C. N., Vericat, D., and Batalla, R. J. (2017) Effects of Suspended Sediment Transport on Invertebrate Drift. *River Research and Application,* 3, 1655-1666. DOI: 10.1002/rra.3146.

ABSTRACT: Invertebrate drift plays an important role in river ecosystems. Although drift has been studied extensively, the relative importance of the various factors that initiate drift during disturbances remains unclear. Instream gravel mining releases fine sediment and so provides an opportunity to assess the influence of suspended sediment on drift, without the confounding effects of hydraulic changes and bed-material entrainment associated with floods. This paper examines invertebrate drift responses to increases in suspended sediment during an episode of mining in a Pyrenean river. During short periods of mining activity, suspended sediment concentrations and thus suspended sediment loads (SSLs) increased one order of magnitude at downstream monitoring sections, with maxima similar to those observed during natural floods in the river. Maximum SSLs were recorded at the sections closest to the mining, with downstream transport patterns suggesting that the majority of suspended material was deposited within 1.5 km. Invertebrate drift rates, the number of taxa drifting and the taxonomic structure of the drift changed at sections close to the mining when suspended sediment concentrations and SSLs were high; such changes were not observed at the section 1.5 km downstream. There were significant relationships between SSL and drift, positive for some groups (Ephemeroptera, Plecoptera and Trichoptera) and negative for others (Chironomidae). Our work shows that increases in suspended sediment alone are sufficient to trigger changes in drift, although further studies are needed to elucidate the underlying mechanisms, and especially to explain the varying responses shown by different taxonomic groups.

KEY WORDS: invertebrate drift; suspended sediment transport; gravel mining; physical disturbance; gravel-bed river; River Cinca

1. INTRODUCTION

Invertebrate drift describes the downstream transport of organisms suspended within the water column (Brittain and Eikeland, 1988). The importance of drift in river and stream ecosystems worldwide has ensured that it has long been a subject of interest (Waters, 1972), especially in relation to the question of what causes animals to leave the bed (Naman et al., 2016). During high flows, shear stress increases, and once entrainment thresholds are exceeded; both fine and coarse sediments are mobilized. Invertebrates may be dislodged directly by high shear stress or entrained along with bed sediments. Conceptual models (Statzner et al., 1984) and empirical evidence (Gibbins et al., 2007a) both suggest that rates of drift entry correspond directly to rates of sediment entrainment. While there may be very limited drift entry prior to the bed becoming unstable (i.e. shear stress alone does not appear to be a major driver), once the bed becomes unstable, even very low bedload transport rates (marginal bedload transport as defined by Garcia et al., 1999) are enough to trigger mass drift (Gibbins et al., 2007b).

Drift is generally higher during periods when suspended sediment concentration (hereafter SSC) is high (e.g. Culp et al., 1986; Doeg and Milledge, 1991; Suren et al., 2005; Larsen and Ormerod, 2010). However, as high flows result simultaneously in increased shear stress, bedload transport and SSC, the importance of suspended material as a trigger for drift has been difficult to isolate from these other factors. Bond and Downes (2003) manipulated flow conditions and added fine sediment to channels to assess their relative impacts on benthic diversity and drift. They found strong interactive effects of flow increases and fine sediment, but the addition of sediment alone had little effect. Their work was conducted in artificial channels, but to our knowledge, there are no studies that have assessed drift responses to increases in SSC in the absence of other flow-related disturbance in field settings.

Instream sediment mining and associated operations can entrain fine sediment and increase SSCs for several kilometres downstream (Kondolf, 1994). Thus, reaches downstream from mining may experience temporary increases in SSC even though flows remain unchanged and the bed remains stable. Episodes of mining may therefore represent natural experiments that allow drift responses to suspended sediment to be quantified without the confounding effects of high flow and bedload. The aim of this paper is to assess invertebrate drift responses to increases in suspended sediment. Drift and suspended sediment transport were quantified to (i) assess the relationship between drift and suspended sediment, (ii) assess the relative propensities of different taxa to drift in response to increases in suspended sediment and (iii) determine the downstream extent of short-term changes in suspended sediment and drift related to mining. Our main hypothesis is that disturbances created by instream mining increase SSCs downstream and, consequently, increase invertebrate drift rates and alter the taxonomic composition of the drift. We further hypothesized that the magnitude of the changes in drift is related directly to the magnitude of SSC. These hypotheses were tested by monitoring drift and SSC before and during mining activity in a gravel-bed river; mining released fine sediment from the bed, but discharge remained unchanged, so hydraulic and bed mobility changes did not influence drift in the way they do during natural flood events.

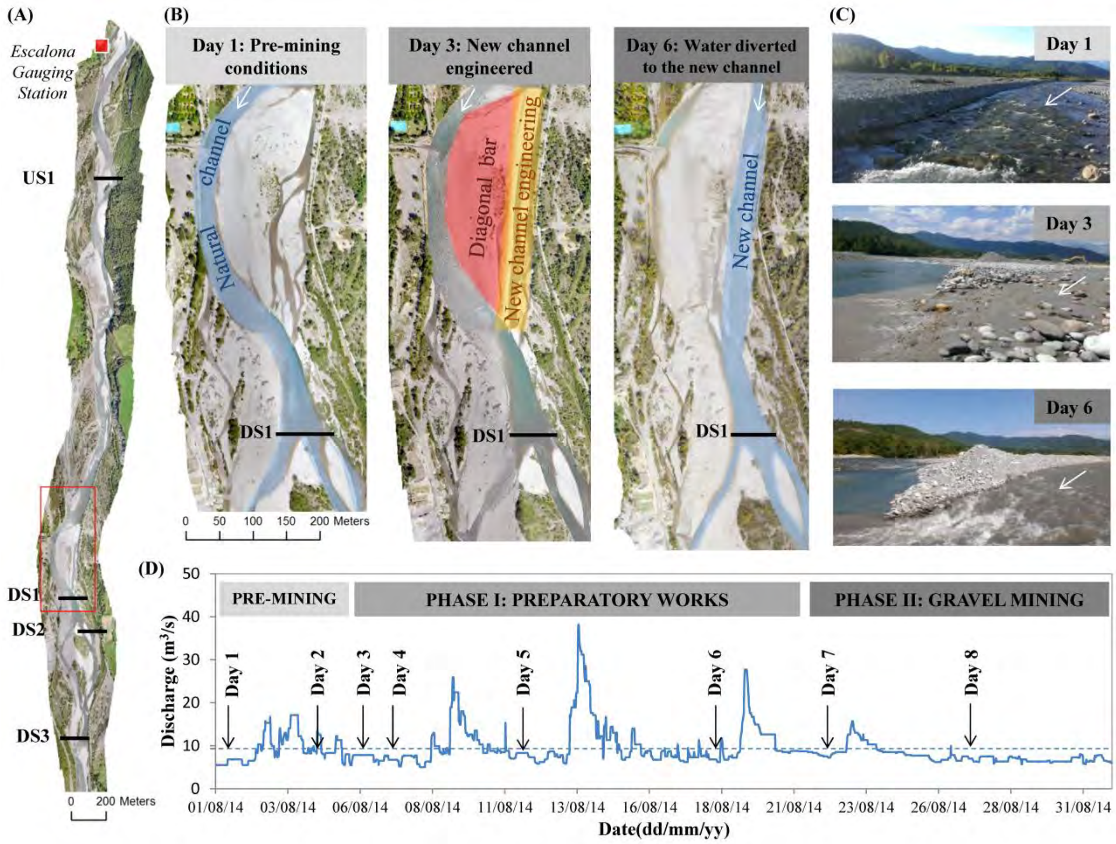

Figure 1. Location of the study reach in the River Cinca and sampling design. (A) Aerial photograph of the study reach indicating the sampling sections where invertebrate drift and water samples were taken and the Escalona gauging station. Red box highlights mined section, with US1 being the upstream monitoring section and DS1–3 downstream sections. (B) Aerial photographs of the mined area on different sampling occasions. Here, the pink shading highlights the bar that was mined, while images for phase I show days that fine sediment was being released from the new channel as a result of excavator activity. (C) Images of the downstream outlet of the engineered channel showing contrasting levels of turbidity on three sampling days. White arrows indicate flow direction. (D) Discharge in the River Cinca (Escalona gauging station) for the mining period, along with drift sampling occasion (days 1–8). Dotted line indicates average discharge for the whole period.

2. MATERIALS AND METHODS

2.1. Study area

The work was carried out in the upper part of the River Cinca, a 9740-km^2 tributary of the River Ebro, in the Southern Pyrenees (NE Iberian Peninsula, Figure 1A). Sampling was conducted in a 5-km reach upstream of the Mediano-Grado dam complex, at 569 m a.s.l. Mean annual discharge (Q, period 1959–2015) in the reach (measured at Escalona gauging station, Figure 1A) is 29 m^3/s, annual floods exceed 220 m^3/s and relatively high magnitude floods attain 740 m^3/s (i.e. floods with a recurrence interval of 10 years). Average active channel width is around 200 m, while water depth at mean Q averages 0.3 m. Sediments are very poorly sorted, ranging from sands to large boulders (i.e. 0.5 to 2000 mm). The median size of the surface sediments is 67 mm, so the upper Cinca can be considered a gravel-bed river, with a notable presence of cobbles and even coarser materials.

The upper Cinca has a long history of instream gravel mining. Nowadays, owing to environmental and legal constraints, gravel is removed purely for the purpose of channel maintenance (e.g. improving channel embankments for flood protection). Mining that took place in the study reach in summer 2014 provided an opportunity to assess changes in SSC and drift. In preparation for mining, during what we define as phase I, the channel was moved to the true left bank to allow access to the bar that was earmarked for gravel extraction (Figure 1B). This preparatory work was undertaken during 2 days (days 3 and 6, Figure 1B and C), with fine sediment released only on these days. Figure 1B–D shows the different phases of the operation. The mining took place during what we define as phase II (Figure 1D), but no fine sediment was released at this time because the whole area was isolated and dewatered.

2.2. Data acquisition

Overview. Figure 1D shows the timing of drift sampling in relation to mining operations and flow conditions. Q was low during the sampling period (mean of 9 m^3/s), with only a peak slightly higher than that of the mean annual flow. Flow varied from day to day, but it remained constant during the time that drift nets were deployed in the channel. Before the mining, sampling was conducted on two occasions (days 1 and 2), with four more during phase I (days 3, 4, 5 and 6), and another two during phase II (days 7 and 8). Drift was sampled at four cross sections, one upstream from the mined area (US1) and three downstream (DS1–3; Figure 1A) on each of the eight occasions. The upstream section acted as a reference site. Downstream sections were positioned 0.2, 0.4 and 1.5 km below the mined reach. None of the sections were located within the mined area. Water samples were collected at each cross section and later used to determine SSC. Drift and sediment samples were collected from all sections on all occasions, with the exception of days 1 and 2, when drift was not sampled at DS2.

Discharge and hydraulic conditions. Q was measured continuously by the Ebro Water Authorities at the Escalona gauging station. This station is located approximately 1 km upstream from the study reach (Figure 1A). No tributaries discharged into the Cinca between Escalona and the study reach during the sampling period. A two-dimensional hydraulic model (Iber; see www.iberaula.es) of the reach was built. The model uses topographic data and fluid dynamic equations to simulate free surface flow in rivers and estuaries (Bladé et al., 2014). The model was parameterized using a digital elevation model of 1-m resolution (see Vericat et al., 2016, for details) and a single value of roughness (0.035). Flow input was retained as being constant until the output stabilized at the same value. This model was used to obtain the distribution of water depths and velocities in the study reach for key Q's recorded during the monitoring period.

Suspended sediment. A US DH48 depth integrating water sampler was used in sections DS1 and DS2 where lateral differences in water turbidity were apparent during mining activity. Here, each water sample was obtained by integrating three sub-samples collected vertically at different points across each section. In contrast, the variation in turbidity across sections US1 and DS3 was not marked, so water samples were collected vertically in a single mid-channel point using a 1-l bottle. Samples were collected on an hourly basis on days when the excavator was working (days 3 and 6). During days when there was no excavator activity, a minimum of two water samples were taken at all four sections during the time that drift nets were deployed. Water samples were vacuum filtered (Filter-Lab, 0.0012-mm pore size), dried and weighed in order to determine SSC (in g/l). As the primary interest was whether an increase in SSC leads to increased drift, organic and inorganic fractions were not separated.

Newcombe and Macdonald (1991) found that characterizing suspended sediment using a concentration– duration model was useful for representing potential ecological impacts. Therefore, suspended sediment load (SSL) during the time that each drift net was deployed was used as a measure of concentration–duration. For a given net deployment, SSL was calculated by the sum of the loads of all water sampling intervals ($SSL_{t=i}$) within the occasion. For this, $SSL_{t=i}$ (g) was calculated by multiplying the $SSC_{t=i}$ (g/l) times the discharge ($Q_{t=i}$; l/s) times the total time (s) of the interval.

Invertebrates. Drift at each section (all riffles; 70–90 m wide) was sampled using standard nets (40 × 20-cm frame; net length 0.7 m; mesh 1000 µm). Five nets were positioned laterally across each section, spaced evenly to cover the width of the channel. Nets were placed on the bed, and because of their height relative to water depth, typically sampled ≅80 % of the water column. All sampling was conducted between 09:00 and 19:00 h (i.e. daylight hours) to avoid bias resulting from daytime/night-time drift oscillations. A pilot study (unpublished data) indicated that, given the daytime drift rates at low flow, a minimum of 4 h was needed to collect sufficient animals for robust statistical analysis. Thus, nets were left in place for 4 to 6 h on each of the eight sampling occasions. The mesh size helped minimize clogging of nets with fine particulate organic material over this sampling period, although it meant that we lost small animals. However, as we always compare relative patterns (before–during mining and between sites), this loss does not influence assessment of changes drift rates.

Benthic samples were collected at US1, DS1 and DS3 before the excavator entered the channel (days 1 and 2; Figure 1D). These samples were used to characterize the potential drift source pool immediately upstream from each cross section. Six samples were collected from each section on each day, spread proportionally between the morphological units present (pool, plane bed and riffle). Sampling was completed under base flow conditions using a surber sampler (300-µm mesh size, 0.215-m^2 sampling area). All animals were preserved in 95 % alcohol for subsequent analysis. In the laboratory, they were sorted and identified as much as possible, using Tachet et al. (2002), Hynes (1993) and Prat and Rieradevall (2014).

2.3. Statistical treatment

Taxa present only as one individual in a single sample were removed from the database to reduce the potential effect of rare species. Drift was expressed as drift rate, that is, number of animals or taxa per unit area per unit time ($n/m^2/h$) rather than density, as calculation of density can add noise that is likely to obscure patterns in the data (Downes, 2010). The reason for such noise links partly to how net filtration efficiency (i.e. net clogging) influences estimates of filtered water volume (Muehlbauer et al., 2016).

Bayesian analyses were applied to estimate the differences in the means between sections and sampling days. Bayesian analyses provide alternatives to classical frequentist approaches to statistical inference; in particular, they do not have to make approximation assumptions typical in null hypothesis significance testing. The inferences from Bayesian analyses provide richer and more informative insights than null hypothesis significance testing. Bayesian estimation supersedes the t-test (BEST) was used as an alternative to the t-test (Kruschke, 2013) using http://www.sumsar.net/best_online/, while Bayesian analysis of variance (BANOVA) was performed as an analogue to ANOVA using SPSS© 25.0.

Non-metric multidimensional scaling (NMDS; Kruskal, 1964) was used to compare drift composition with the benthic source pool and assess temporal changes in drift composition. Prior to the full analysis, a PROcrustean Randomization TEST (PROTEST; Jackson, 1995) was used to test whether the distribution of samples on the ordination plots was affected by the choice of the ordination technique. In PROTEST, results from NMDS were compared with those from a Principal Component Analysis (PCA), with matrices subjected to 999 randomizations to test whether sample positions differed between the two techniques. As they did not differ (m^2 = 0.47; the probability that m2 is smaller than expected by chance is <0.001), NMDS was applied knowing that interpretations about sample similarities were not simply a product of the use of this technique. Within the NMDS, Sorensen's index, modified by Bray and Curtis (1957), was used as the dissimilarity measure for assessment of drift changes over time, while Jaccard's similarity coefficient (Jaccard, 1912) was used to assess differences between the benthos and the drift. Jaccard's index was used for the latter as it uses only information on species composition (i.e. not their abundances) and so is appropriate when abundance differs markedly between samples (in our case, there were many more individuals in benthic than drift samples).

Stress was used to judge how well the NMDS ordinations summarized distances among the samples. If the stress value is smaller than 0.2, the two-dimensional plot is capable of representing well the rank dissimilarity values. Analysis of similarities (ANOSIM) was used as a *post hoc* test for differences between groups of samples on the NMDS. Where ANOSIM indicated a difference, Keppel's modified Bonferroni correction was applied to p values to allow for multiple comparisons between groups. The similarity of percentages procedure (Clarke and Warwick, 1994) was used to identify which taxa contributed most to the dissimilarity between groups. All of these analyses were run using the Vegan package (Oksanen et al., 2016) within R© 3.2.3 (R Development Core Team, 2015).

The relationship between drift and SSL was assessed using generalized estimating equations (GEEs). These were used because drift was sampled repeatedly across the same sections and so samples could not be considered truly independent. GEEs account for the lack of statistical independence between samples by adjusting regression coefficients and variance to avoid spurious correlations (Zorn, 2001; Larsen et al., 2009). GEEs were fit using SPSS© 17.0.

3. RESULTS

3.1. Flow conditions

Discharge for the hydrological year 2013–2014 ranged from 5.2 to 477 m^3/s (the latter representing a recurrence interval of 5 years). During the study period (5–31 August), Q was rather low (mean of 9.2 m^3/s, flow equalled or exceeded 85 % of the time in the flow duration curve, i.e. Q_{85}) and rather stable (standard deviation; SD = 4.1 m^3/s; Figure 1 D). Discharge during the eight sampling days ranged from 5.2 to 11.7 m^3/s (mean 7.5 m^3/s, i.e. Q_{93}), and SD was 0.58 m^3/s (Figure 1D). Day 6 registered the largest variation in Q (SD = 1.74 m^3/s). A flow event of 40 m^3/s (= Q_{20}) occurred on 13–14 August.

The outputs from the hydraulic model indicated that mean water depth in the study reach at the average Q prevailing during mining was 0.3 m (SD = 0.23 m) while mean velocity was 0.74 m/s (SD = 0.39 m/s). During the 13–14 August event (i.e. 40 m^3/s), mean water depth was predicted by the model to be 0.5 m (SD = 0.34 m), while mean velocity was predicted to be 1.18 m/s (SD = 0.58 m/s). It is important to note that, owing to the size of the sediments and the relatively low slope of the channel (average 0.7 %), 40 m^3/s was an insufficient Q to entrain bed material and so is not considered to result in geomorphic disturbance. No bedload was collected in any of the drift nets during sampling, indicating that only fine material was being transported.

3.2. Suspended sediment

Upstream of mining (at US1) SSC averaged 0.01 g/l (SD = 0.02 g/l) over the sampling period, and did not show any changes at the time that mining was taking place downstream (e.g. Figure 2). At the sections downstream from mining (DS1–DS3), mean SSC on the days that the excavator was not disturbing the channel was 0.01 g/l, with a maximum of 0.07 g/l. In contrast, on days 3 and 6 when the excavator was working, mean SSC at these downstream sections was almost one order of magnitude higher (mean 0.09 g/l) and reached a maximum of 1.34 g/l (Figure 2). On day 3, mean SSC at DS1 and DS2 was 0.07 and 0.02 g/l, respectively, suggesting deposition of sediment in the reach between the two sections. Further downstream (DS3), mean SSC on day 3 was 0.01 g/l, similar to that observed at US1. On day 6, mean and maximum SSCs at DS1 were 0.13 and 0.84 g/l, respectively.

Sediment load at US1 was relatively low but highly variable (mean of 1.52 tonnes, SD = 2.11 tonnes). This value was similar to that estimated at the downstream sections on the days when the excavator was not active (i.e. 1.2 tonnes). However, at the downstream sites, average sediment load on the days with excavator activity was seven times higher (i.e. 8.6 tonnes). Maximum loads were observed on day 6, when they were one order of magnitude greater than the average of the days when the excavator was not active (e.g. 16.5 tonnes at DS1 and 14.2 tonnes at DS2). Average sediment load at DS3 during phases I and II was 1.1 tonnes (SD = 1.02 tonnes), similar to the mean sediment load at US1. As with the SSC data, this suggests that the plume of fine sediment produced from the mining site was deposited along the river bed and did not reach the downstream end of the study area.

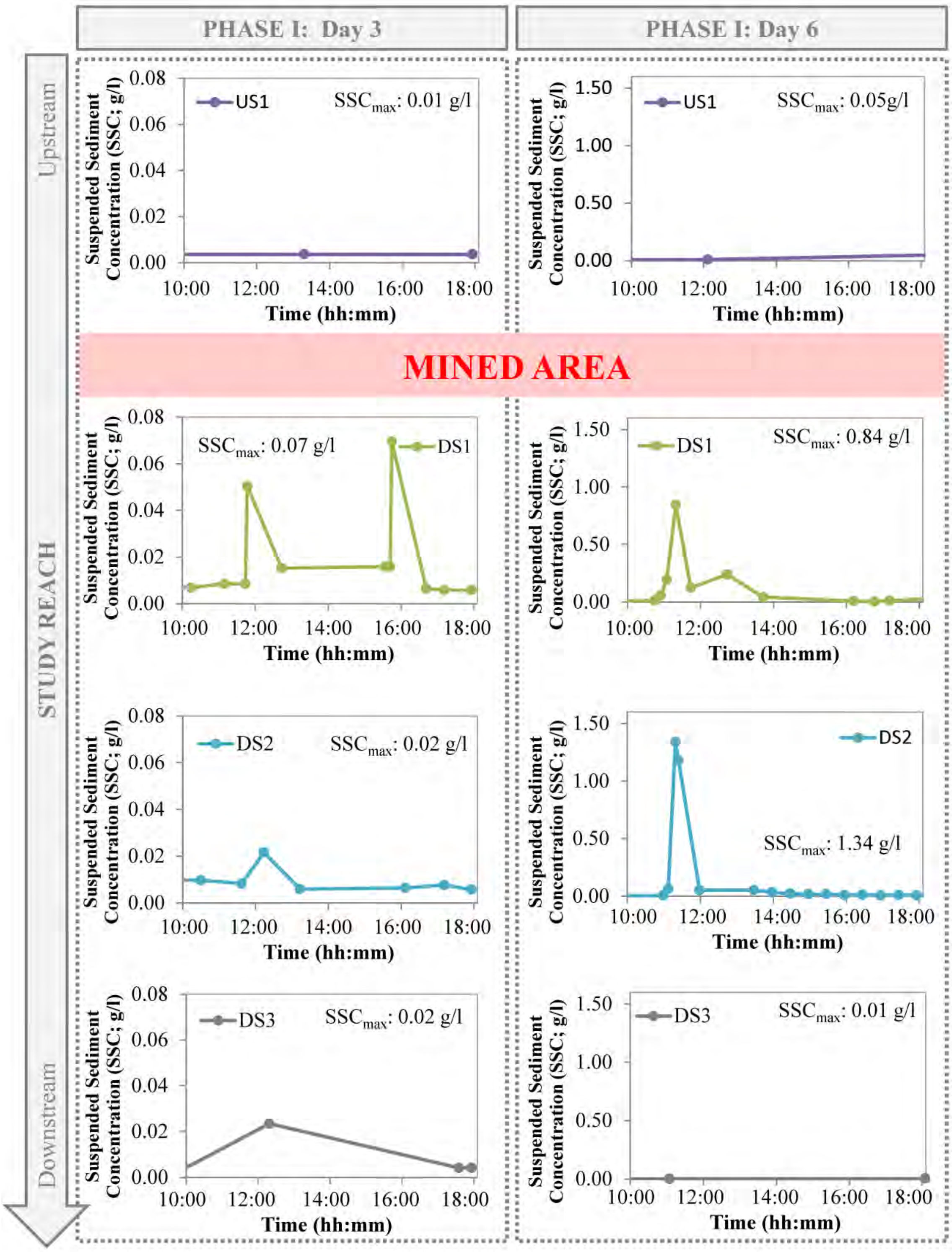

Figure 2. Sedigraphs at the 4 sampling sections at the start of the two mining phases (on days 3 and 6). Note that scale in day 3 differs from that in day 6. SSC_{max} refers to the maximum instantaneous SSC. See Figure 1 for the location of the sampling sections

3.3. Invertebrates

Benthic abundance averaged 538 ind/m^2 (SD = 217 ind/m^2) before mining operations, with strong evidence of differences between sections (Table I; BANOVA: BF = 2.42). Conversely, there was little evidence of differences in taxon richness between sections (BANOVA: BF = 0.16). The most abundant taxa were the mayflies *Ephemerella sp.* and *Baetis sp.*, the midge Orthocladiinae and the stonefly Leuctridae. Together, these taxa represented more than 75 % of the benthic assemblages.

Ephemeroptera, Plecoptera and Trichoptera (EPT), along with the Chironomidae, together accounted for an average of 90 % of drift. At US1 (Figure 3), drift rate ranged from 32 to 164 ind/m^2/h (average 72, SD = 38 ind/m^2/h). Here BEST indicated that there were no credible differences in either drift rate or the number of taxa drifting between days with and without excavator activity (Table II; drift rate BEST: % likelihood that the difference between the two groups is <0 - %$C_{<0}$- was 48.4%; taxon richness BEST: % $C_{<0}$ = 76.4 %). This differed from patterns at downstream sections. At DS1, drift rates and drift taxon richness were credibly higher on the days with excavator activity than without (BEST % $C_{<0}$ = 93.1 % and 97.3 %). At DS2, difference in drift rates between days with and without excavator activity remained uncertain (BEST % $C_{<0}$ = 75.6 %), although there was a credible difference in taxon richness (BEST % $C_{<0}$ = 99.8 %). The maximum drift rate recorded during the study period was 304 ind/m^2/h, observed at DS1 during day 6 in which mean SSC was seven times higher than at US1 (Figure 3). The maximum number of taxa recorded in the drift (13 taxa) was also on day 6 at DS1.

Prior to mining operations, the composition of the drift was different to that of the benthos (Figure 4, ANOSIM test of between group difference R: 0.85, $p < 0.001$). This suggests that, during daytime periods when there is no disturbance and drift can be considered as background, not all benthic taxa have the same propensity to drift.

Patterns of drift for individual taxa were varied and complex (Figure 5). For many taxa, there was a general trend of reduced drift over the study period. This trend was very clear and consistent at US1. At DS1 and DS2, several taxa showed an abrupt increase in drift on day 6; these increases were less apparent further downstream at DS3. Differences were evident for *Ephemerella sp.* and *Baetis sp.* (increases in drift) and Orthocladiinae (decreases).

At US1, there was no compositional change in drift between day 2 (just prior excavator activity) and day 3 (first day of activity; Figure 6A; ANOSIM test, R = 0.248, $p < 0.05$). However, immediately downstream of the mining site (i.e. at DS1), drift composition changed between these days (R = 0.448, $p < 0.05$); in contrast, such changes were not observed further downstream (DS2 or DS3: $p > 0.05$). Changes in drift composition also occurred at the time of the second period of excavator activity (between days 5 and 6; Figure 6B); that is, there were significant differences between days 5 and 6 at DS1 and DS2 (R = 0.648, 0.548; $p < 0.05$ respectively), while there were none upstream (US1: R = 0.056, $p > 0.05$) or further downstream (DS3: R = 0.04, $p > 0.05$). *Baetis sp.*, *Ephemerella sp*. and Orthocladiinae made up the majority of the changes observed at DS1 and DS2 (these taxa accounted for 80 % of the average dissimilarity at DS1 between days 5 and 6, and 56 % at DS2).

Table 1. Mean benthic invertebrate abundance and taxon richness (± Standard Deviation, SD) for the study reach on Day 1 and Day 2 (pre-mining period, figure 1).

Sampling section	Invertebrate density (ind/m^2)	Taxon Richness(taxa/m^2)
US1	482.6 (±223.0)	54.6 (±15.3)
DS1	433.3 (±226.0)	59.3 (±14.0)
DS3	700.8 (±271.7)	58.1 (±13.2)
Total	*538.9(±262.2)*	*57.3(±13.9)*

Table 2. Results of the BEST test (Bayesian analysis) comparing drift rates and taxon richness grouped by the level of disturbance: (i) no disturbance (i.e. no excavator activity; Day 1, Day 2, Day 4, Day 5, Day 7 and Day 8) and (ii) disturbance (i.e. excavator activity; Day 3 and Day 6). See figure 1 and the text for more details.

Sampling section	Drift rates				Taxon richness			
	DoM*	95 % HDI**		% $C_{<0}$ ***	DoM*	95 % HDI**		% $C_{<0}$ ***
US1	0.2	-42.0	38.5	48.4	-0.5	-1.8	0.8	76.4
DS1	-35.5	-88.7	14.9	93.1	-1.9	-3.7	0.0	97.3
DS2	-9.0	-36.4	24.4	75.6	-2.7	-4.3	-1.0	99.8
DS3	-2.7	-16.5	21.3	38.2	-1.0	-2.8	0.8	87.7

*DoM: Difference of Means

**95 % HDI: 95 % Highest Density Interval

** % $C_{<0}$: Credible that the difference between the two groups is <0

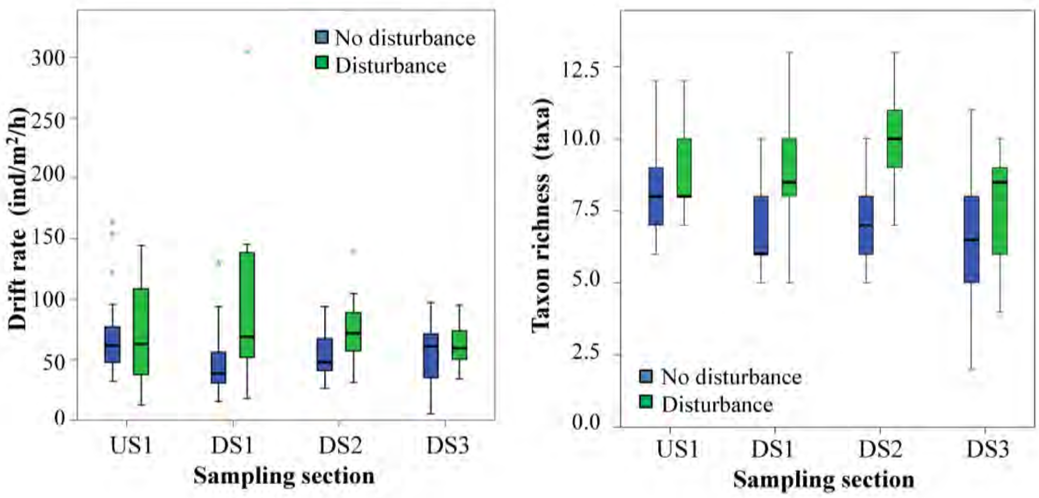

Figure 3. Boxplot of (A) drift rates and (B) taxon richness for the study reach grouped by the level of disturbance: (i) no disturbance (days 1, 2, 4, 5, 7 and 8) and (ii) disturbance (days 3 and 6). Note that the distribution of the sampling occasions during the study period is detailed in Figure 1. Boxes show median (horizontal bars) and interquartile ranges (rectangles) while stars are outliers

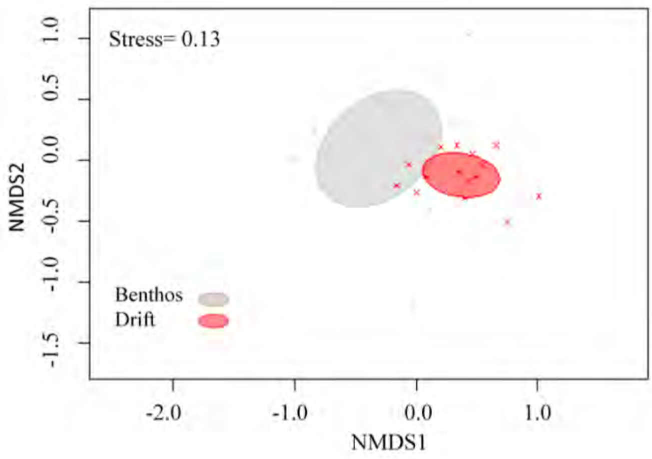

Figure 4. Non-metric multidimensional scaling (NMDS) ordination of benthic and drift invertebrates for all the sampling sections in the River Cinca during the pre-mining period (day 1). The figure shows the 95 % confidence ellipses constructed for all samples (i.e. 18 samples for benthic assemblages and 15 for drift). The ordination was produced using Jaccard's similarity values. The stress value (0.134) is low (less than 0.3), indicating this plot is capable of representing well the original Jaccard index values

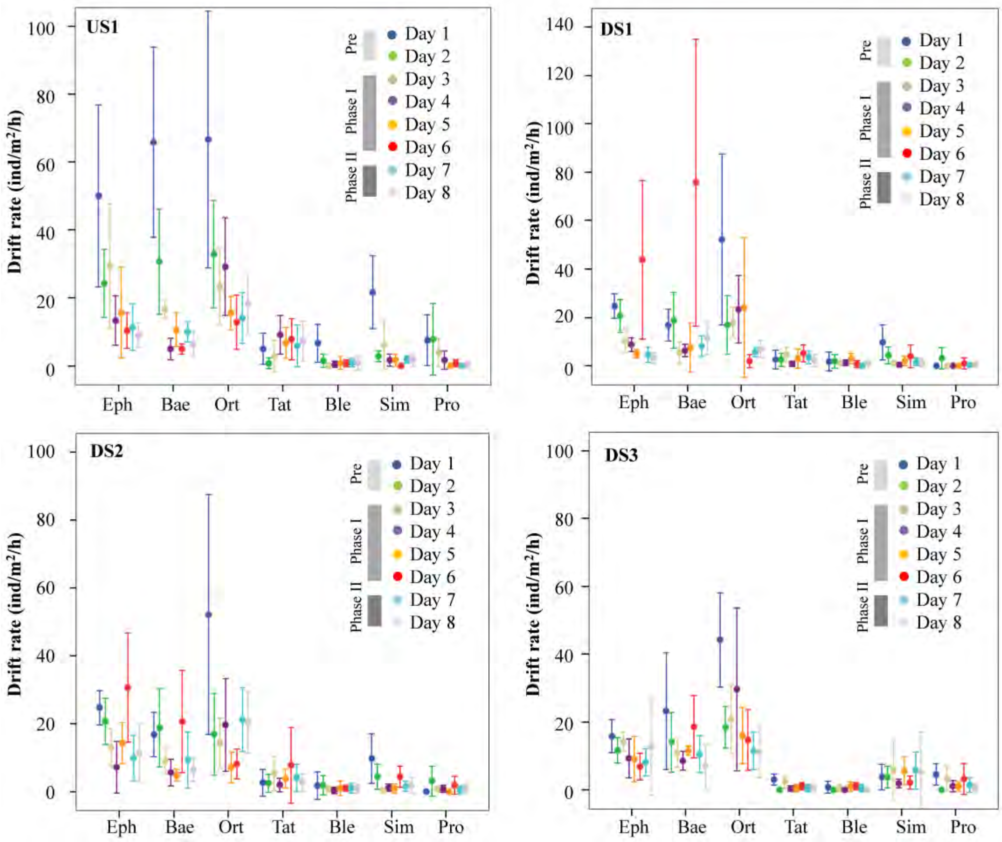

Figure 5. Drift rates of the most abundant taxa in each sampling section in the River Cinca for the study period. Error bars indicate one standard deviation above and below the mean. US, upstream section; DS, downstream sections. Sampling occasions are detailed in Figure 1. Note that scale in DS1 is different.

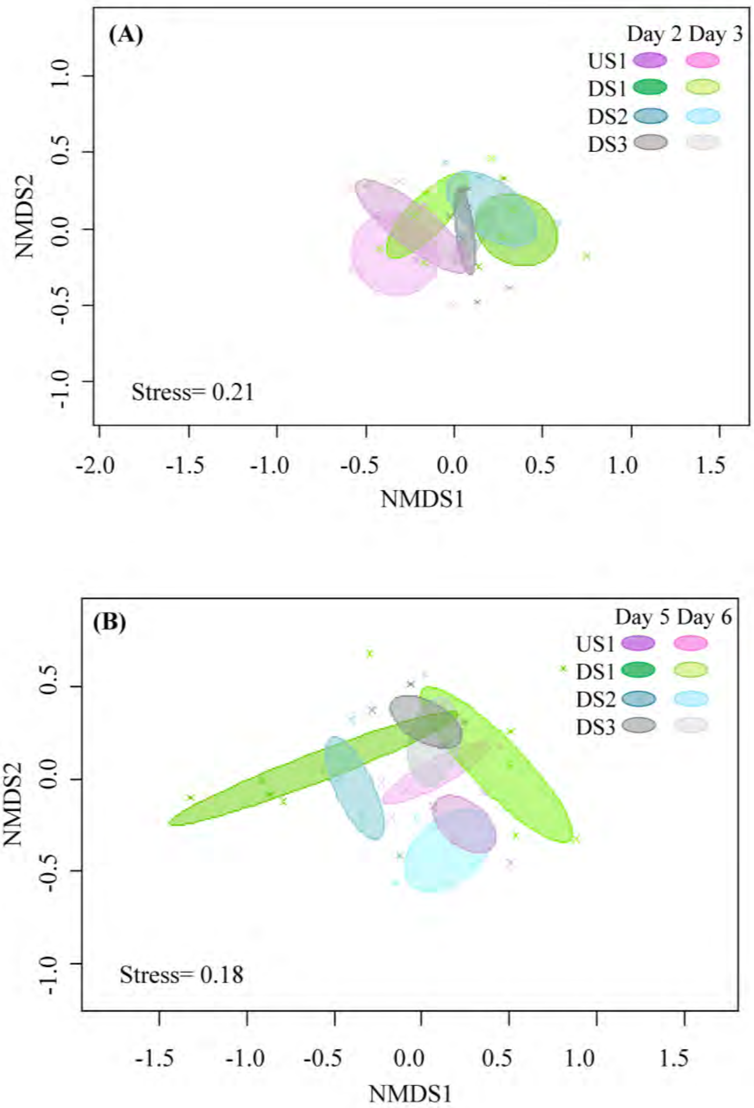

Figure 6. Non-metric multidimensional scaling ordination comparing invertebrate drift composition across two key periods at the study sections. Each diagram compares the drift the day before mining with drift during mining activity. The diagrams show 95 % confidence ellipses constructed for the five samples collected in each section (see section on Study area for more details). (A) Assemblages in sampling sections before (day 2) and during the start of phase I (day 3). (B) Assemblages in sampling sections before (day 5) and during the star of phase II (day 6)

3.4. Relations between drift and suspended sediment

There were significant positive relationships between SSL and both drift rate (goodness of fit = 69; Wald χ^2 = 38.50, p < 0.001; Figure 7A) and the number of taxa drifting (goodness of fit = 12; Wald χ^2 = 17.04, p < 0.001; Figure 7B). Thus, increases in suspended sediment were paralleled by increases in drift. The intercept of the model fit using GEEs was significantly greater than zero for drift rate (Wald χ^2 = 2381, p < 0.001) as well as taxon richness (Wald χ^2 = 1270, p < 0.001), indicating that drift can be expected in the absence of (detectable) suspended sediment transport. The absolute and relative abundance of EPT taxa in the drift was positively related to sediment load (absolute abundance: goodness of fit = 38; Wald $\chi^2 = 8.83$, $p < 0.003$; relative abundance: goodness of fit = 16; Wald $\chi^2 = 6.01$, $p < 0.05$). These relationships were negative for Chironomidae (absolute abundance: goodness of fit = 47; Wald $\chi^2 = 6.5$, $p < 0.01$; relative abundance: goodness of fit = 32; Wald $\chi^2 = 75.7$, $p < 0.001$). Patterns are depicted in Figure 7.

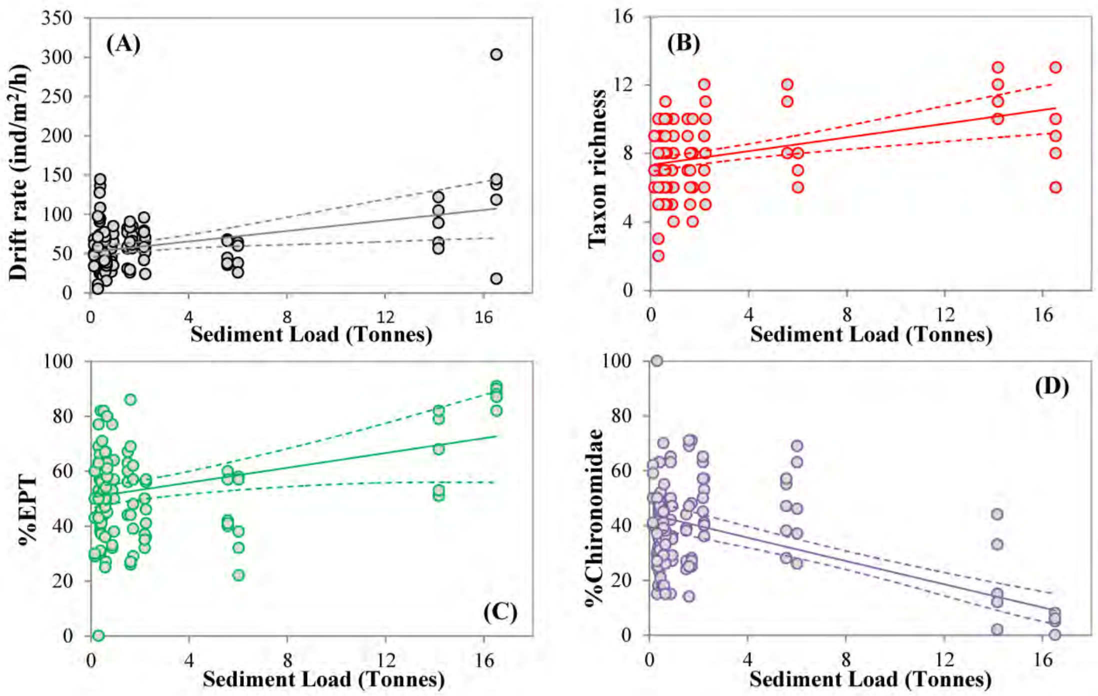

Figure 7. Response of invertebrate drift in the River Cinca to increasing sediment loads: (A) drift rate response to sediment load. (B) Taxon richness response to sediment load. (C) Relative abundance of Ephemeroptera, Plecoptera, Tricoptera taxa response to sediment load. (D) Relative abundance of Chironomidae taxa response to sediment load. The relationships were modelled using generalized estimating equations. Dotted lines show 95 % confidence limits. For brevity, only relative abundance is shown.

4. DISCUSSION

Suspended sediment transport increased notably in a section of the upper River Cinca as a result of short periods of in-channel mining that entrained fine material from the bed. Maximum sediment load was recorded at the monitoring section 200 m downstream of the mining site, with data from other sections suggesting that the majority of suspended material was deposited in the channel within 1.5 km. Rates and composition of the drift changed at sections close to mining on days when SSC was high, with the spatial patterns indicating that the drift response extended as far downstream as the sediment plume extended. In summary, our findings therefore support the hypothesis that increases in SSC alone can trigger an increase in drift. The increased drift rates occurred despite a general decrease in drift during the study period. Similar late summer decreases in drift, likely attributable to natural, life cycle-related changes in benthic abundance, have been reported for other Iberian streams (e.g. Rincón and Lobón- Cerviá, 1997).

Instream mining alters sediment availability and, consequently, may have geomorphic implications for downstream reaches (Kondolf, 1994; Surian et al., 2009). Mining normally takes place under low flow conditions that lack competence to transfer sediments downstream. When mining ends, this results in sediment deposition in the area immediately downstream, but subsequent competent events can remobilize and transport material greater distances. Thus, although mining effects initially are localized, in the longer term, they may extend further downstream. We are not aware of any studies that have characterized suspended sediment dynamics and/or ecological responses in downstream sections on days that gravel mining is actually taking place and fine sediment is being released.

In the Cinca, maximum SSCs associated with mining activity reached 5.8 g/l, similar to values observed during natural flood events in this and nearby mountain rivers (e.g. Lobera et al. [2016] reported 28 g/l in the Ésera). The key difference is that increases in SSC in the Cinca occurred in low flow conditions rather than in response to elevated discharge and associated bed disturbance (i.e. particle entrainment).

Interactions between suspended sediment transport, deposition and benthic invertebrate community structure have been studied by numerous authors (e.g. Suren et al., 2005; Larsen and Ormerod, 2010), but field studies linking either SSC or SSL to drift remain scarce by comparison (O'Hop and Wallace, 1983). The few studies published on these links have suggested that animals leave the bed in increasing numbers when suspended sediment increases (e.g. Ciborowski et al., 1977) and that certain taxa contribute disproportionately to increases in drift (e.g. *Baetis rhodani* and *Baetis muticus*; Larsen and Ormerod,2010). In the case of the River Cinca, we found significant relationships between SSC duration and both drift rate and the taxon richness in the drift, although there were some notably contrasting responses between different taxonomic groups. The increased drift of EPT taxa was expected, given published information on their sensitivity to fine sediment (Wood and Armitage, 1997). Unlike during natural floods when changes in drift relate to the complex interaction of shear forces and the entrainment of both coarse and fine sediment (Bond and Downes, 2003; Naman et al., 2016), in the Cinca, changes occurred when the only disturbance was the increase in fine sediment loads related to mining.

There are several mechanisms that might explain drift increases during periods of elevated SSC. One possible mechanism is that abrasion by fine material may dislodge animals from the bed (Jones et al., 2012). Another mechanism may relate to the subsequent deposition of fine material. Fine sediment deposited on the bed is problematic for invertebrates, for reasons related to its direct effects (e.g. clogging of gills) and its effects on their habitat, especially reducing porosity and movement of oxygenated water into and through the bed (reviewed by Wood and Armitage,1997). Buendia et al. (2013a, 2013b) found that spatial and temporal variations in fine sediment being deposited on the bed resulted in marked spatio-temporal variation in invertebrate assemblages. Thus, in the Cinca, animals entering the drift in response to increases in suspended sediment may be doing so to avoid the subsequent effects of this material once it settles. If this is the case, then drift may be an anticipatory mechanism.

However, given the low flow conditions and the fact that not all of the fine materials released from the mining operations reached the most downstream monitoring section (i.e. there was settlement within the study reach), something which we cannot discount is that the observed increases in drift were a direct response to the risk of smothering. Finally, the increased drift of some taxa may reflect a response to changes in light. Behavioural drift is well known to increase during periods of darkness (Flecker, 1992). Increases in suspended load result in more turbid water and limited penetration of light to the bed. Given the absence of hydraulic or bed disturbance in the Cinca, increased drift may therefore reflect a darker and hence safer environment in which to redistribute. Decreases in the drift of Chironomidae observed in the Cinca may reflect a different response. Chironomids are commonly found in the hyporheic zone, where conditions are often different to those of the benthic zone (Armitage et al., 1994; Gibbins et al., 2016). Their decreased drift may reflect a strategy of moving down within the hyporheic zone to avoid fine sediment effects (i.e. the hyporheic refuge hypothesis; Dole-Olivier, 2011). However, as our correlative data provide no direct insight into the causal mechanisms at play, this discussion of responses shown by the different taxonomic groups is speculative. Mechanisms could, however, be tested through carefully planned field and/or laboratory experiments.

The River Cinca has extensive parts of its catchment covered with highly erodible Eocene marls and sandstones, has a highly dynamic hydrological regime and has experienced intensive mining activity over the last century (e.g. Llena et al., 2016). Together, these disturbances produce repeated cycles of high suspended loads, periodic storage of fine sediments on the bed and their subsequent remobilization. As reported by Buendia et al. (2013a), in the long term, such cycles select for taxa with specific life history traits. Thus, the short-term drift responses observed in the Cinca may represent those of a benthic assemblage already shaped by a long history of fine sediment disturbances. Reponses in other rivers, where such disturbances are rare, may be quite different.

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CHAPTER 6

HABITAT HETEROGENEITY AND BED SURFACE COMPLEXITY AFFECT BENTHIC INVERTEBRATES

HABITAT HETEROGENEITY AND BED SURFACE COMPLEXITY AFFECT BENTHIC INVERTEBRATE DIVERSITY IN A GRAVEL-BED RIVER

This chapter contains the following submitted paper in the journal *Freshwater Biology.* JCR-SCI Impact Factor: 3.255. Category: Marine and Freshwater Biology; 1st Quartile.

Béjar, M., Gibbins, C.N., Vericat, D., Batalla, R.J., 2018. Habitat heterogeneity and bed surface complexity affect benthic invertebrate diversity in a gravel-bed river (to be submitted).

ABSTRACT: Maintaining or restoring physical habitat structure is a central tenet of sustainable river management, yet the ecological significance of habitat complexity and heterogeneity in fluvial systems has long remained equivocal. The lack of clear evidence partly reflects the problems of characterising habitat in ways that are ecologically meaningful, and at scales that are relevant to biota. This paper assesses the influence of habitat structure on macroinvertebrate assemblages in an upland gravel bed river. Using 0.1 m resolution data, heterogeneity and complexity in hydraulic conditions and bed topography were characterised at scales ranging from 1 to 25 m^2 centred on points from which macroinvertebrate samples were collected. Habitat data were collected using an Acoustic Doppler Current Profiler connected to a Real Time Kinematic Global Positioning System. Cluster analysis was applied to the hydraulic and topographic data to identify discrete habitat types. From these data, 13 metrics of habitat heterogeneity and bed surface complexity were calculated at two scales (1 m^2 and 16 m^2). Although turnover of invertebrate taxa between sampling points was rather limited, observed differences in diversity corresponded to a several of the habitat metrics. At the local scale (1 m^2), habitat diversity, patch size coefficient of variation, mean patch size and bed roughness had direct effects on invertebrate density, taxon richness and community structure. Flow hydraulics and surface complexity changed markedly across the scales (from 1 to 16 m^2), but characterising habitat at the larger scale did not improve the explanatory power of the models fit to data on invertebrate community composition. Our approach provided high resolution data on riverbed habitat structure at multiple spatial scales, and evidenced that habitat heterogeneity and complexity influence benthic invertebrates.

KEY WORDS: hydraulics; bed topography; habitat structure; spatial scales; invertebrate diversity; ADCP; gravel-bed river; River Cinca

1. INTRODUCTION

The habitat heterogeneity hypothesis (Simpson, 1949) remains central to much of applied ecology. This hypothesis holds that a physically diverse habitat provides a greater number of niches, with different ways of exploiting environmental resources positively influencing species composition and richness and allowing greater persistence of populations (Lancaster and Downes, 2010; Sueiro et al., 2011; Heino et al., 2015). Although the hypothesis is widely accepted, empirical tests in river systems have yielded contradictory results, and often failed to find evidence of expected relations (see review in Barnes et al., 2013). Despite the lack of empirical support, the principle of maximising habitat heterogeneity forms a key part of many aspects of river management and rehabilitation (FISRWG, 1998, Pollock et al., 2012).

Finding evidence of the effects of habitat structure has been complicated by a number of factors. First, uncertainty remains to the definition of some important concepts and metrics related to habitat heterogeneity (recently reviewed by Carvalho and Barros, 2017), and this may confound attempts to find statistical relations between habitat structure and biological communities (Robson and Barmuta, 1998; McCoy and Bell, 1991; Tews et al., 2004). Second, habitat heterogeneity arises from complex interactions among multiple causes that operate at many spatial scales (e.g. Li and Reynolds, 1995; Robson, 1996; Stevenson et al., 1996), so the influence of habitat structure on macroinvertebrates is likely to depend on the spatial scale examined (Downes and Jordan, 1993; Sanson et al., 1995; Robson and Chester, 1999). Thus, collecting data at the wrong spatial scale may mean that relationships are missed. Finally, despite recent technological advances, few field ecological studies have applied the potentially insightful approaches to geomorphic and hydraulic characterisation that have become commonplace in other fields.

Physical habitat provides shelter, food and dissolved oxygen for macroinvertebrates. The occurrence of these elements in a given location is driven by habitat structure, including flow hydraulics. Early studies of habitat structure involved visual assessment of bed conditions (Hastie et al., 2000; Moorkens, 2000) or point measurements of flow hydraulics (Palmer, 1992). Although increasing numbers of ecological studies have applied new data acquisition techniques (e.g. Acoustic Doppler Current Profilers, ADCPs) to acquire high density depth and velocity data that improve habitat characterization, most of these are, to our knowledge, based in artificial channels or flumes (e.g. Rice et al. 2008, Oldmeadow et al., 2010). In parallel with the availability of new measurement devices, data processing techniques also allow generation of high resolution maps of bed topography, roughness and flow hydraulics, further improving the representation of the physical conditions experienced by macroinvertebrates. For instance, ADCPs mounted on floating platforms and linked to Real Time Kinematic (RTK) GPS systems allow Doppler-scanning of river reaches and extraction of velocity fields (horizontally and vertically) down to 0.02 m cell sizes (e.g. Rennie et al., 2010; Guerrero and Lamberti, 2011). At the same time, water depth and bed elevation data from each of the ADCP beams can be obtained, data that allow extraction of high resolution bed topography. High resolution topographic data have revolutionized the way and scale in which landscapes can be characterised (see general review by Passalacqua et al., 2015; and a recent applied study to river restoration by Marteau et al., 2017). The combination of sensors and platforms allow multivariate and multi-scale data to be collected and interrogated, providing powerful insights into river macro-topography (i.e. bed elevation) and micro-topography (i.e. bed roughness; see Brasington et al., 2012 and Pearson et al., 2017). To our knowledge no studies have yet used these approaches to assess the influence of habitat structure on benthic invertebrate diversity.

In this paper we use ADCP-derived field data, interrogated using evolving data processing tools, to provide multi-scale insights into the physical habitat structure in a gravel-bed river. Metrics of habitat heterogeneity and complexity are then used to assess the degree to which habitat influences macroinvertebrate community structure and diversity, and the influence of the spatial scale on the strength and nature of the relationships. Our hypotheses are that: (a) high resolution habitat characterization will reveal the habitat-diversity relationships that previous studies have often failed to find, and (b) characterising habitat structure over a larger area (but at the same resolution) better explains local variation in invertebrate community structure than when it is characterised only in the immediate vicinity of the sampling point. Our work follows the general approach and statistical analyses set out by Barnes et al. (2013). These authors found no evidence to support the hypothesis that habitat heterogeneity increases macroinvertebrate diversity, and that the effects of bed surface complexity were weak. In essence our study attempts to see whether using different field data collection techniques to characterise bed topography, as well as incorporating hydraulic data to assess habitat heterogeneity, alters their conclusions about the influence of habitat structure on river invertebrates.

2. STUDY AREA AND METHODS

2.1. The upper River Cinca

The work was carried out in the upper part of the River Cinca (Southern Pyrenees), one of the main tributaries of the River Ebro in the NE Iberian Peninsula (Figure 1A). Sampling was conducted in a 9.8-km reach upstream from the village of Ainsa. The altitude at Ainsa is 570 m a.s.l. while the catchment area is 849 km^2. Mean annual discharge (Q, period 1959–2015) in the reach (measured at the A051 official gauging station located in the middle of the study reach) is 27 m^3/s, while the flows with a recurrence interval of 10 years attain 740 m^3/s (Llena, 2015). Average active channel width in the reach is ca. 200 m while water depth at mean Q averages 0.3 m. Sediments are poorly sorted, with a sorting index of 1.1 (as per Folk and Ward, 1957), ranging from sands to large boulders (i.e. 0.5 mm to 2 m *b*-axis). The median size of the surface sediments ranges from 54 to 76 mm, so based on the Wentworth scale the characteristic sediments in the reach are very coarse gravels.

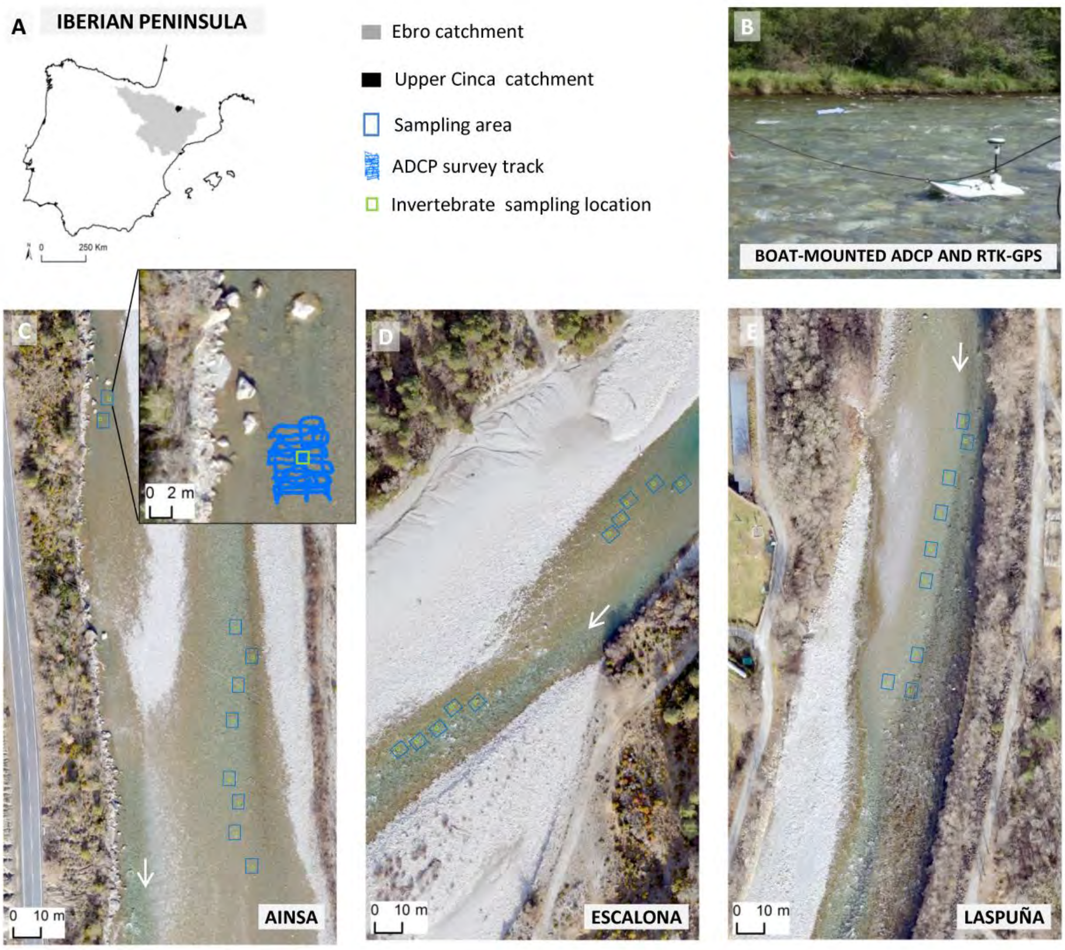


Figure 1. (A) Location of the upper River Cinca catchment in the Ebro basin and the Iberian Peninsula. (B) The ADCP mounted on a board and connected to a RTK-GPS system. (C) to (E) Aerial photos showing study sites and sampling points denoted by blue squares. The inset in C shows the invertebrate sampling area (square) along with the ADCP track around this. Arrows indicate flow direction.

2.2. Data acquisition

Invertebrate sampling and habitat characterisation were undertaken between 13th and 15th July 2015. Data were obtained in three river reaches (ca. 150-m long each) within a 9.8-km reach of the upper Cinca (Figure 1 C-E). A total of 30 areas (each 25 m^2, see Figures 1C-E) were sampled and characterised (i.e. 10 samples per section), with these chosen to cover the greatest possible variation in habitat. The 10 samples were spreadly taken between all morphological units present in each reach, with these split in proportion to the plan areas of respective units. Discharge and suspended sediment transport were recorded (15-minute data) at Ainsa (at the downstream end of the study reach) during the previous month in order to characterize conditions experienced by macroinvertebrates immediately prior to sampling (for more details see Béjar et al. 2017 and 2018); *Q* was also monitored during the sampling period.

Data collection was based on the need to acquire invertebrate samples and both hydraulic and topographic data for the same bed areas. Following Barnes et al. (2013) we collected data to compute metrics of habitat heterogeneity and bed surface complexity in each area. Habitat heterogeneity encompasses both the composition (i.e. number and relative abundance) and configuration (i.e. spatial arrangement) of distinct habitat types present in an area. Surface complexity represents the bed structure features (i.e. McCoy and Bell, 1991, Barnes et al., 2013; Carvalho and Barros, 2017). Figure 2 illustrates the steps followed for data collection and post-processing, to further assess the influence of habitat conditions on benthic invertebrates.

Water velocity and depth were measured using a Sontek® M9 River Surveyor ADCP, mounted on a floating boat and connected to a Leica® Viva GS15 RTK GPS system (Figure 1B). The RTK-GPS provides precise real-time kinematic position data to the ADCP (further details of precision below). Each sampling area (25 m^2) was surveyed by multiple zigzags with the ADCP (see the inset in Figure 1C). This approach (as opposed to single transects) allowed for maximum data coverage. Locations with water depths below 0.2 m were avoided due to the ADCP blanking distance. Although water depth data can be obtained for depths around 0.1 m (or even shallower), in general the acoustic signals do not travel a sufficient distance before reflection to obtain reliable velocity data. Conversely, areas approaching depths of 1 m were typically either too fast to sample safely (too fast and deep) or too deep and slow-flowing to properly operate the surber to sample macroinvertebrates (see details below). These difficulties reduced the range of sampled hydraulic conditions.

Immediately after using the ADCP, macroinvertebrates were sampled using a surber placed in a single location in the centre of each 25 m^2 area. The surber had a larger frame (0.5 m × 0.5 m) than the standard design, to allow a greater bed area (0.25 m^2) to be sampled. Net mesh size was 1000 µm. During sampling, the area within the base frame was systematically disturbed using a trowel, with larger sediments also rubbed by hand to dislodge invertebrates. We used a single larger frame positioned at one point in preference to collecting several smaller samples from around the 25 m^2 area; this was in case standing and moving around the bed while operating the ADCP disturbed animals prior to sampling. We did not collect invertebrate samples first because the bed disturbance would affect topographic and hydraulic characterisation. Barnes at al. (2013) also sampled invertebrates after habitat data had been collected. Invertebrates were retained in bottles and preserved with alcohol for later sorting and identification in the lab.

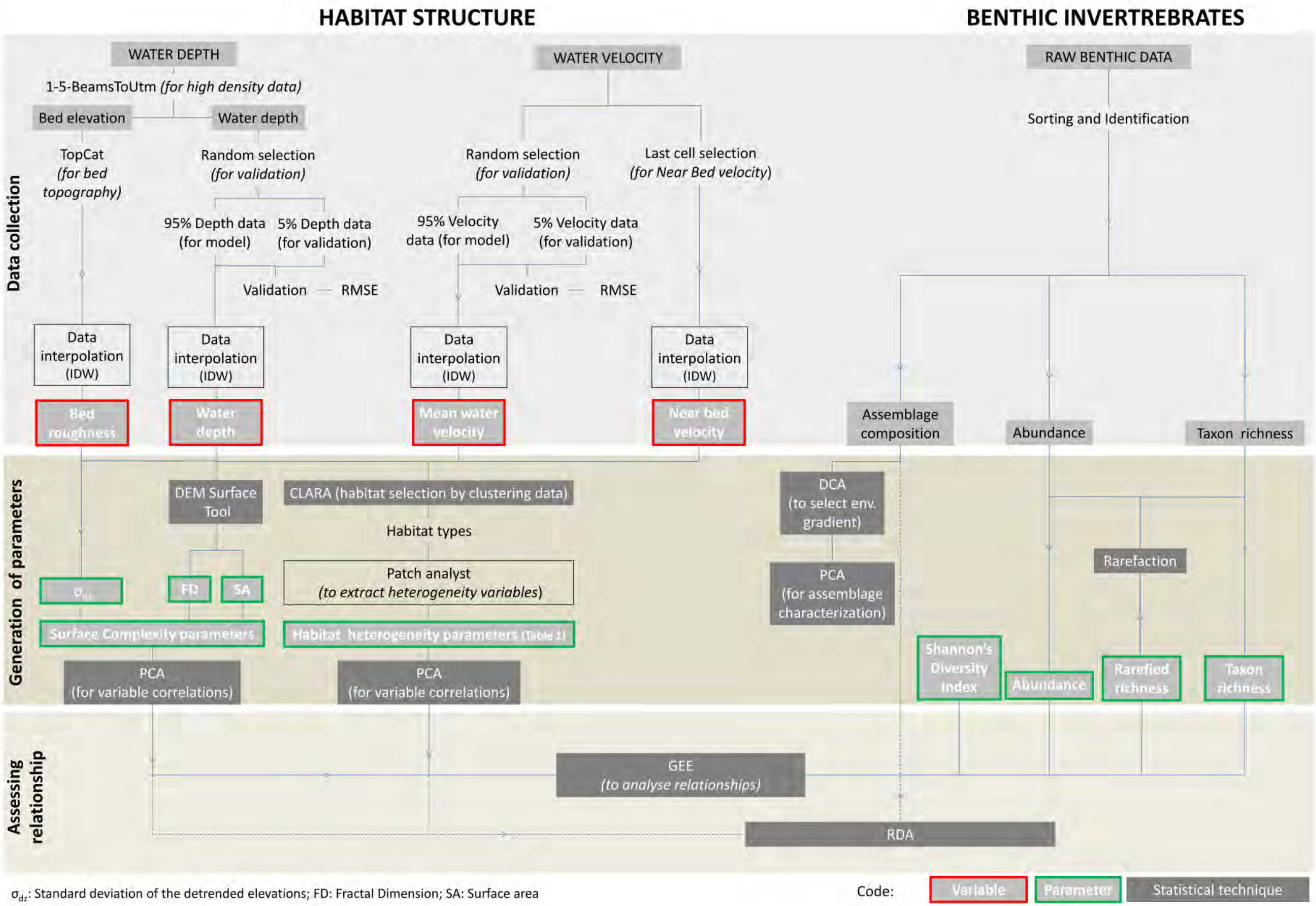


Figure 2. Schematic overview of data collection and analysis. The upper panel summarises data collection and the habitat and ecological variables resulting from this. The central panel represents the analyses undertaken to produce a range of metrics from the field data. The bottom panel shows the statistical approaches used to assess relations between habitat and ecological metrics.

2.3. Data post-processing

2.3.1. Extracting habitat variables

Habitat was characterised using water depth, mean column water velocity, near bed velocity and bed surface topography (Figure 2). Water depth and mean water velocity were directly extracted from the ADCP survey data, while near bed velocity and surface topography were computed using the raw data. Mean depth and velocity are widely used in river habitat characterisation, while several studies have now demonstrated that benthic invertebrate distribution corresponds well to near bed hydraulics and the refugia provided by bed topography (Lancaster and Belyea, 1997; Hart and Finelli, 1999; Rice et al., 2008). Figure 3 shows a visual example of how habitat variables were extracted; specific details are provided in the following paragraphs.

Water depth (d): The RTK-GPS streams the location of the vertical beam of the ADCP. Network-based RTK corrections (GPRS-based) are provided from the ARAGEA reference stations (Aragon Government). The 3d quality of these observations is less than 0.05 m. The ADCP has a total of four 25º -tilted dual beams (3.0 MHz/1.0 MHz) and a vertical 0.5 MHz beam. Data for each of these beams are registered every second. Depth accuracy is set at 1 % while resolution is 0.001 m. The combination of the data for all the beams make it possible to obtain not just the vertical depth at the point where the ADCP is located at a given time, but also four additional water depth values at a distance from the vertical beam that equates to half the vertical d. For instance, given the tilt of the dual beams (25º), when d = 0.5 m, the depth measurement points for the tilted beams would be displaced 0.233 m from the location of the vertical one. Therefore, each 1-second interval ADCP measurement is composed of five d measurements distributed in a sampling circle with a diameter similar to the vertical d. In the present study, the zigzag pattern meant that every 25 m^2 survey area contained approximately 500 measures of d and was complemented by 2000 more measures extracted from the dual beams. This resulted in total of 2500 measurements in each sampling area, equating to an average of 100 values/m^2.

Mean water velocity (V_m): The three velocity components (X, Y and Up) for each cell are measured by a single ADCP beam. The ADCP records a water velocity profile (i.e. from the water surface down to the river bed) with a resolution dictated by the cell size. The cell size, in turn, depends on the sampling frequency used and the water depth. In the present study cell sizes ranged between 0.02 and 0.1 m. The velocity profile is recorded every second, with V_m for each one calculated automatically. Each 25 m^2 sampling area was characterised by an average of 500 measurements of V_m, yielding a mean resolution of 20 values/m^2.

Near bed velocity (V_{nb}): In order to obtain a value of velocity near the bed, it was necessary to first remove all profiles for which the size of the cell was larger than 0.03 m (we consider this cell value integrates an area too large to be considered representative of the near bed velocity). The velocity value of the first cell above the bed cell was used to represent V_{nb}. As the ADCP blanking distance is around 0.05 m and the cell sizes are 0.02 m, the velocity estimates represent values approximately 0.07 m above the bed. On average, each sampling area was characterised by 400 measures of near bed velocity, equating to 16 values/m^2.

Bed roughness (σ_{dz}): The ADCP is linked to the RTK-GPS, providing georeferenced depth and hence bed elevation data. The multi-beam bed elevations deliver high resolution topographic data sets; on average, the circles covered by the tilting beams yielded a density of depth observations of around 100/m^2 across each 25 m^2 sampling area, allowing confident assessment of bed micro-topography as follows.

Micro-topography is considered as changes in elevation related to particle size variability, and therefore represents bed roughness. Based on this, following Brasington et al. (2012), the detrended standard deviation of the elevations (σ_{dz}) was used as an estimate of micro-topography. σ_{dz} is now used widely across the earth sciences as a roughness metric (Smith, 2014) and can be significantly correlated to bed grain size percentiles such as the median (D_{50}; Brasington et al 2012). Recently, Pearson et al. (2017) assessed the influence of the shape and form of the particles on roughness and the grain size relationships. They argued that these relationships are not universal, so care needs to be exercised when using roughness estimates to extract grainsize. The σ_{dz} is therefore used primarily a measure of bed surface complexity (for which see below). High values indicate a large variability in micro-topography (i.e. a rough or complex surface), while values close to 0 indicate a flat (relatively smooth) surface, with minimal micro-topography. To calculate σ_{dz} we followed the approach of Brasington et al. (2012) in using the TopCat® geostatistical toolkit (included in the Geomorphic Change Detection Software for ArcGIS®, see http://gcd.joewheaton.org, Wheaton et al., 2010). The plane containing all observations of elevation was decomposed into a set of non-overlapping square grid cells (in our case 0.5 × 0.5 m). A series of elevation statistics were calculated for the points in each grid cell. For each cell, a neighbourhood triangular tessellation based on the mean elevation of the surrounding cells was used to construct local planes representing the surface used to detrend the observations. Several statistics can be calculated by the software, one of which is standard deviation of these detrended elevations. This is considered the roughness (i.e. σ_{dz}). In this study, because of the zigzag pattern of the ADCP and the resulting resolution of the beam-derived bed elevation values, an estimate of σ_{dz} was produced for each 0.5 × 0.5 m grid cell across each 25 m^2 sampling area, with the density of points used to compute each σ_{dz} value being approx. 100/m^2. The grid cell size means that 100 σ_{dz} values were calculated for each 25 m^2 area.

2.3.2. Point data interpolation and accuracy

The density of observations differed between the hydraulic and topographic variables (from averages of 16 to 100 point values/m^2), mainly because of the way data were extracted from the ADCP, as explained above. Point data interpolation was performed in order to homogenise these densities and obtain continuous raster-based maps of each variable for each sampling area (see an example in Figure 3). Data interpolation allows estimation of values at un-sampled points using existing point observations, generating a regular grid of values for the whole sampling area (Erdogan, 2009). The three most commonly used interpolation methods (Inverse Distance Weighting, Krigging and Natural Neighbours) were tested on water velocity and depth data in one sampling area. Before interpolation, 5 % of the observations from the raw data were removed (randomly) and used as check points (Figure 2). A resolution of 0.10 m was used for the interpolation, such that each raster contained a total of 2500 cells across the 25 m^2 area. The interpolated values were then compared to their corresponding check points, using residuals to assess the accuracy of each interpolation method. The method that yielded the lowest residuals was the Inverse Distance Weighting (*IDW*), so this interpolation was applied to all variables for all the 25 m^2 areas.

The accuracy of final mean water velocity and depth raster data sets was also assessed for all of the areas. These two are considered the main primary variables, as near bed velocity is influenced by the blanked distance of the ADCP and also the cell size, while bed roughness is based on the detrended elevations and other errors may arise in this assessment. Root Mean Square Errors (RMSEs) were used to assess error in depth and mean velocity, with these calculated using interpolated and observed values for respective check points.

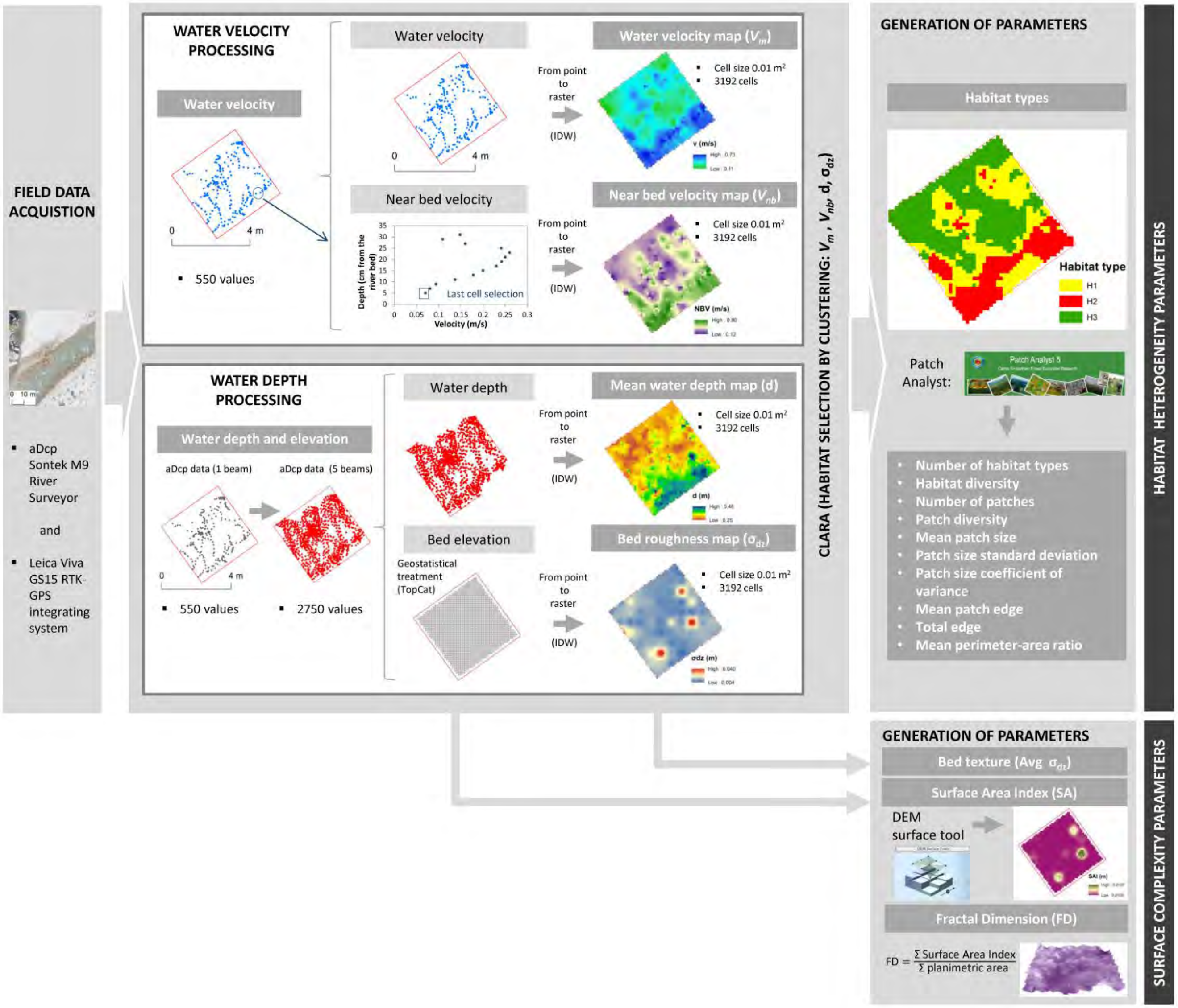


Figure 3. Example of data processing used for the generation of metrics representing habitat structure and the mapping of habitat types within survey areas. The diagrams on the left bottom show examples of 'smooth and 'complex' bed topographies based on complexity

2.3.3. Habitat Types, Habitat heterogeneity and Bed Surface Complexity

Habitat types

Habitat types were defined using information on mean water velocity, near bed velocity, water depth and bed roughness (Figure 2). A partitioning clustering method (Clustering LARge Applications, CLARA; Kaufman and Rousseeuw, 1990) was used to determine the habitat types. Generally, clustering is defined as the grouping of multivariate objects, such that objects within a cluster are as similar as possible, whereas objects from different clusters are as dissimilar as possible. The goal of a clustering analysis is therefore to find the clusters which minimize the sum of within-group dissimilarities. In the present case, all raster cells were transformed to points (n = 48,230), with each point having an associated value of mean water velocity, near bed velocity, water depth and bed roughness. These point values were then used in the clustering to identify distinct habitat types, and associate each point to a habitat.

A range of possible cluster numbers (i.e. number of distinct habitats) was tested, ranging from k=2 to k=12. Following Lechner et al. (2016) the optimum number of clusters was selected using average Silhouette width. This width (S) is defined as the mean similarity of each element to its own cluster, minus the mean similarity of the next most similar cluster. Partitioning clustering method was performed using the Cluster Package (Maechler et al., 2017) in R® 3.4.0 (R Development Core team, 2017).

Once the optimum number of clusters/habitats was selected, each data point was coded according to the cluster to which it belonged. As all points were georeferenced, code values were then used to produce a map showing the distribution of habitats across each 25 m^2 area. This process (undertaken using ESRI® ArcMap™ 9.3) is graphically illustrated in Figure 3. These maps provided the basis for assessment of habitat heterogeneity in the sampling areas.

Habitat structure: heterogeneity and surface complexity

Following Barnes et al. (2013) habitat structure was decomposed into its heterogeneity and its complexity, as follows:

Habitat heterogeneity: This was assessed using the spatial characteristics of the habitat types. The Patch Analyst Tool (version 4.2, Rempel et al., 2008) was used for this with the same set of metrics used by Barnes et al. (2013) calculated for the whole of each 25 m^2 area. These metrics are listed along with their abbreviations in table 1.

Surface complexity: Following Barnes et al. (2013), three measures of surface complexity were calculated for each surveyed area: a) the Surface Area (*SA*), representing the topographic surface area, b) a Fractal Dimension (*FD*) corresponding to an index of topographic convolutedness and c) bed surface complexity, as represented by the Bed Roughness. Calculation of these indices differed somewhat from Barnes et al. (2013) because of how topographic data were collected. The *SA* of a given cell corresponds to the area defined by the eight triangles that can be drawn using the elevation of the cell and the surrounding eight elevations. This gives a more appropriate estimate of the bed area available than the simple planimetric area, and it corresponds more closely to the area that is available for macroinvertebrates. Surface Area was calculated using the DEM Surface Tools for ArcGIS®

(Jenness, 2013). The *FD* was calculated as the ratio between the *SA* and the planimetric area. The planimetric area of a cell corresponds to the pixel resolution (i.e. 0.10 × 0.10 m = 0.01 m^2). A ratio equal to 1 would indicate a completely flat surface, while the ratio increases as topographic roughness increases. As detailed above, bed surface complexity was assessed by means of the standard deviation of the detrended elevations (σ_{dz}).

Table 1. Parameters describing habitat heterogeneity and surface complexity of sampling locations.

Parameters	Abbreviation	Description
Habitat heterogeneity		
Nº Habitat Types	Habtype	Number of habitat types present within the surveyed area
Habitat Diversity	Habdiver	Measure of relative habitat diversity (i.e. Shannon's diversity Index for habitat)
Number of Patches	Patches	Total number of patches
Patch Diversity	Patchdiver	Measure of relative patch diversity (i.e. Shannon's diversity Index for patches)
Mean Patch Size	MPS	Average patch size (m)
Patch Size Standard Deviation	PSSD	Standard deviation of patch areas (m)
Patch Size Coefficient of Variance	PSCoV	Coefficient of variation of patches
Mean Patch Edge	MPE	Average amount of edge per patch (i.e. TE/Patches)
Total Edge	TE	Sum of all perimeters of patches (m)
Mean Perimeter-Area Ratio	MPAR	Shape complexity (I.e. sum of each patches perimeter-area ratio/patches)
Surface complexity		
Surface Area	SA	Estimation of the topographic surface area (m)
Fractal Dimension	FD	Index of topographic roughness and convolutedness (i.e. SAI/ planimetric area)
Bed Roughness	σ_{dz}	Standard deviation of the detrended elevations (m)
Benthic invertebrates		
Total abundance	Abundance	Number of individuals
Taxonomic richness	Taxonomic richness	Number of taxa
Rarefied richness	Rarefied richness	Number of taxa standardized for overall abundance
Taxonomic diversity	SDI	Measure of relative taxon diversity (i.e. Shannon's diversity Index)

2.3.4. Spatial scales of interest

Each sampling area covered approximately 25 m^2. For the purpose of assessing relationships between habitat structure and invertebrates, the metrics of heterogeneity and surface complexity were calculated at two different scales within this area: at 1 m^2 and 16 m^2. The whole 25 m^2 area was not used because of the possibility of edge effects. Data at the edges of the area may be less accurate than the rest due to the nature of the *IDW* interpolation used (see section 2.3.2). Also, due to the zigzag pattern of the ADCP movement across each area, fewer observations were typically recorded around the edges (i.e. more central areas were crossed several times but the edge less so; see example in Figure 2).

To select the largest scale to be used, velocity data abstracted from 12 scales (from 0.2 to 25 m^2) were analysed to assess scale-related patterns of hydraulic heterogeneity. The assessment started at the centre of the 25 m^2 area (the point at which the invertebrate sample was collected), with data then abstracted for concentric circles around this; the first had a radius of 0.25 m (= 0.2 m^2), with circles enlarged to yield 0.25 m increments in radius. The Cumulative Frequency Distribution Curve (*FDC*) for velocity at each area was constructed (Figure 4B) and from this, a value of the data range computed. The ratio of velocity percentiles 95 and 5 (i.e. P_{95}/P_5) was used the measure velocity range at each scale. In general the range of velocity magnitude increased with area (Figure 4C). At the smallest scale, the ratio approached 1, indicating high slopes in the *FDC* (i.e. very well sorted, a small data range). As the scale increased, the ratio increased, indicating less sorting and a greater range of velocity values. The ratio reached a plateau at a scale of 16 m^2 (ratio = 1.45) and then remained more or less constant as the scale of analysis increased further. Thus, this simple analysis indicates that, in our case, sampling areas greater than 16 m^2 did not add additional hydraulic heterogeneity. This area was therefore chosen as the largest scale for the abstraction of variables used to represent habitat heterogeneity. The smallest scale chosen was 1 m^2, as it has been widely used as a local (detailed) scale in a number of ecological and geomorphic studies (i.e. Barnes et al., 2013 and Boyero et al., 2003).

In the results section, we use 50th and 95th and 5th percentile values (i.e. P_{95}, P_5 respectively), for the habitat variables rather than maximum or minimum absolute ones, to avoid effects of extreme or outlier values. The percentiles are considered representative of the *FDC*.

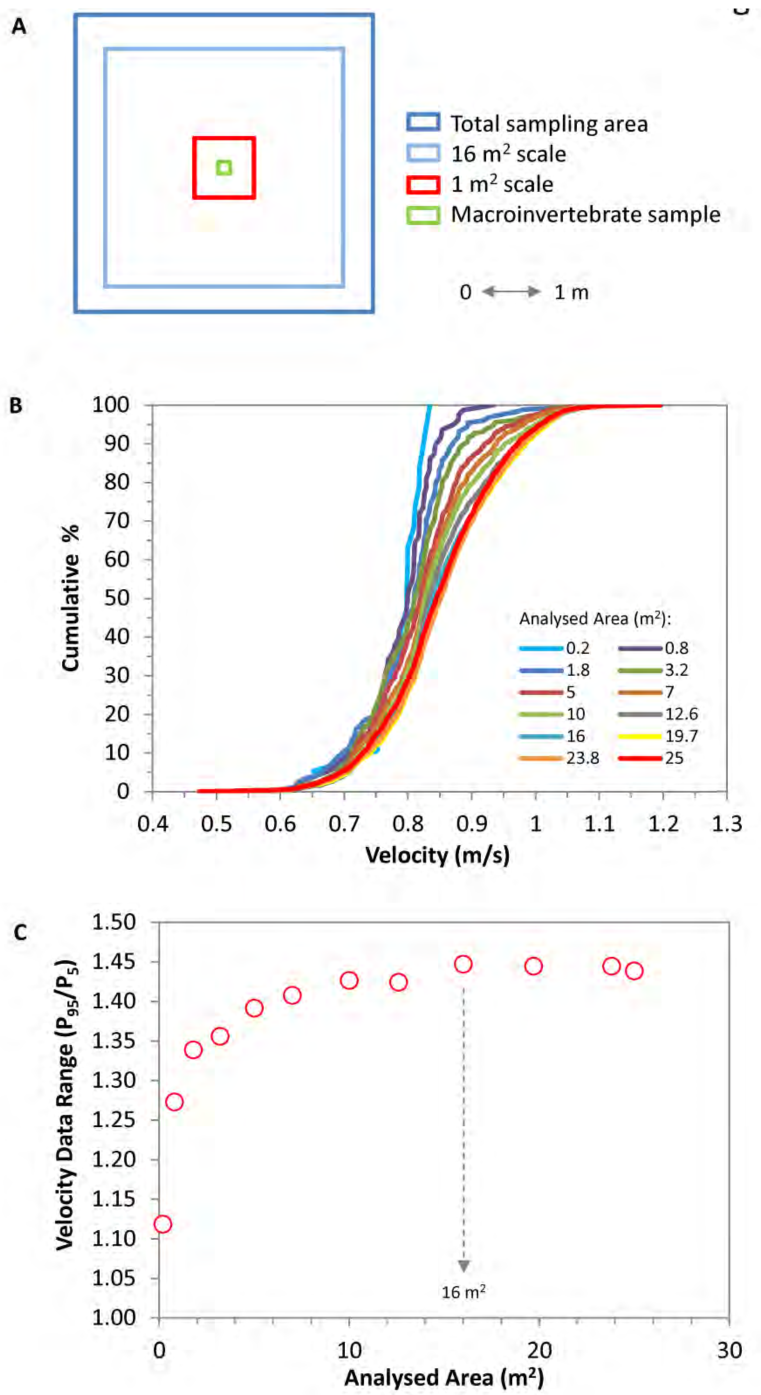


Figure 4. Habitat scales and flow characteristics across the analysed scale range. (A) Illustration of three main scales analysed for each sampling area: (i) Surber scale (0.25 m^2), (ii) small scale (1 m^2), (iii) large scale (16 m^2) and (iv) total sampling area (25 m^2). (B) Cumulative Frequency Distribution Curves (FDCs) for the velocity data extracted at multiple scales within one sampling area (further details in the text). (C) Velocity data range for the FDC in B, with data range defined as the ratio between the percentiles 95 and 5 (P_5 and P_{95} respectively).

2.3.5. Benthic invertebrates

Invertebrate samples were cleaned and animals sorted and identified as far as available keys allowed (Hynes, 1993; Tachet et al. 2002; Prat and Rieradevall, 2014). One sample was spoilt so the final analysis was based on 29 of the 30 samples collected.

Taxa present only as one individual in a single sample were removed to reduce the potential effect of rare species. Total abundance, taxonomic richness, rarefied richness and taxonomic diversity (using Shannon's diversity index) were calculated for each surber sample (Figure 2 and Table 1). Linear regression indicated that the number of taxa recorded in all samples was a function of the number of individuals. Thus, raw taxon richness was also rarefied to remove artefacts of greater abundance on taxonomic richness values (i.e. Sanders, 1968), allowing comparisons to be made between habitats where different numbers individuals were collected (Simberloff, 1978).

2.4. Statistical treatment of data

An overview of statistical analyses is provided in Figure 2. Principal Component Analysis (*PCA*) was used to help identify the main aspects of variation in habitat structure (Zuur et al., 2010). Assemblage composition was first analysed in a Detrended Correspondence Analysis (*DCA*) to determine which method of direct gradient analysis (i.e. linear or unimodal) was best suited to the data. The gradient length from this DCA was 1.69, indicating that linear methods were more appropriate (McCune and Mefford, 1995; Leps and Šmilauer, 2003).

Relations between habitat and macroinvertebrates were assessed using several complementary approaches. Prior to analysis, correlations between values of the habitat metrics were assessed using a *PCA*, with redundant variables removed. The choice of which one of each correlated pair to retain was based on *PCA* component loadings (higher one retained). Both the *PCA* and the *DCA* were run using the Vegan Package (Oksanen et al., 2011) in R® 3.4.0 (R Development Core team, 2017).

Relationships between macroinvertebrate diversity (e.g. taxon richness, rarefied richness, Shannon) and metrics of habitat heterogeneity and surface complexity were modelled using Generalized Estimating Equations (*GEE*s; Zorn, 2001; Larsen et al., 2009). The *GEE* approach treats data values as non-independent samples in the regression. As the invertebrate data collection points were nested within three river reaches (Figure 1), samples could not be treated as being independent; thus, *GEE* was an appropriate form of regression. *GEEs* were also used by Béjar et al. (2017) for analysis of invertebrate drift at the same sites. The regressions were obtained using SPSS® Statistics 17.0.

Finally, Redundancy Detrended Analysis (*RDA*; a linear direct gradient analysis) was used to assess the influence of habitat structure on invertebrate community composition. The heterogeneity and complexity metrics used for this are listed in table 1, with redundant ones removed as per the *PCA*. Through the forward selection, the *RDA* also permitted assessment of the relative importance of the individual heterogeneity and complexity metrics for explaining compositional differences between samples. The marginal effects values (λ_1) from the forward selection were used for this (ter Braak and Šmilauer, 2002; Leps and Šmilauer, 2003). *RDA* was run using CANOCO® 4.5.

The *PCA*, *GEE* and *RDA* used habitat data for the 1 m^2 areas around each surber sampling point; e.g. invertebrates collected using the 0.5 × 0.5 m surber were related to the 1 × 1 m area of bed surrounding the sampling point. Following these analyses, a second *RDA* was undertaken relating the same invertebrate data to the 4 × 4 m (16 m^2) area surrounding each sampling point. The explanatory power of this *RDA* was compared to the first, to assess whether characterising habitat at a larger scale improved insights into the controls on community composition.

3. RESULTS

3.1. Physical habitat characteristics

The raster-based maps had RMSEs ranging between 0.02 and 0.08 m for depth and 0.04 and 0.24 m/s for velocity. On average, these errors represent 7 % of the mean water depth and 18 % of the mean flow velocity.

At the 1 m^2 scale, water depth averaged 0.39 m, with P_5 and P_{95} being 0.28 and 0.65 m respectively (Table 2). Mean column water velocity ranged from 0.46 to 1.17 m/s, with a mean value of 0.75 m/s. Mean near bed velocity was 0.66 m/s, ranging from 0.40 to 1.10 m/s (P_5 and P_{95} respectively). Mean bed roughness (i.e. σ_{dz}) was 0.03 m, with P_{05} = 0.02 and P_{95} = 0.07 m. Values of bed roughness ranged from 0.02-0.65 m, with a mean of 0.03 (Table 2). The median surface particle size (i.e. D_{50}) estimated using this mean value and applying the σ_{dz}-D_{50} relationship presented by Brasington et al. (2012) was 66 mm. This is in accordance with the empirical value of 67 mm reported by Béjar et al. (2017) (based on the measurement of 14,800 particles across the whole study reach).

Table 2. Range of values for the habitat variables for the two main study scales. The 5th (P_5), 50th (P_{50}) and 95th (P_{95}) percentiles are used, while n_{ixi} indicates the number of cells in each sampling location at a specified scale of analysis.

Physical variables	Scale 1 m^2			Scale 16 m^2		
	P_{05}	P_{50}	P_{95}	P_{05}	P_{50}	P_{95}
Depth (m)	0.28	0.39	0.65	0.27	0.40	0.69
Velocity(m/s)	0.46	0.75	1.17	0.45	0.75	1.20
Near Bed Velocity (m/s)	0.40	0.66	1.1	0.38	0.68	1.11
Roughness (m)	0.02	0.03	0.07	0.02	0.03	0.09

n_{1x1}= 3036; n_{4x4}= 48230

Mean silhouette widths for clusters (*k* = 2-12) ranged from 0.25 to 0.47, with the highest value being for *k* = 2. This suggested that the optimum number of clusters (i.e. habitat types) was 2. This is a small number of habitat types, and using only 2 may have obscured ecological differences between sampling points. Average *S* for *k* = 3 was 0.46, very similar the value of 0.47 for *k* = 2. S dropped to 0.34 for *k* = 4, indicating appreciably reduced cluster coherence as cluster number increased beyond 3. Therefore, k = 3 was selected as the optimum number of clusters to use in subsequent analyses. Figure 3 shows an example of the spatial distribution of these three habitats in one of the 25 m^2 areas, while Figure 5 shows the *FDC* for the four variables used to define these habitats, together with their characteristics percentiles. Habitat type 2 represents deep environments, with relatively fast flows and large particle size (i.e. high roughness), while habitat type 3 represents shallow environments with slow velocities and relatively small particle sizes (the smallest roughness estimates present in sampling areas); finally, habitat type 1 represents *intermediate* environments.

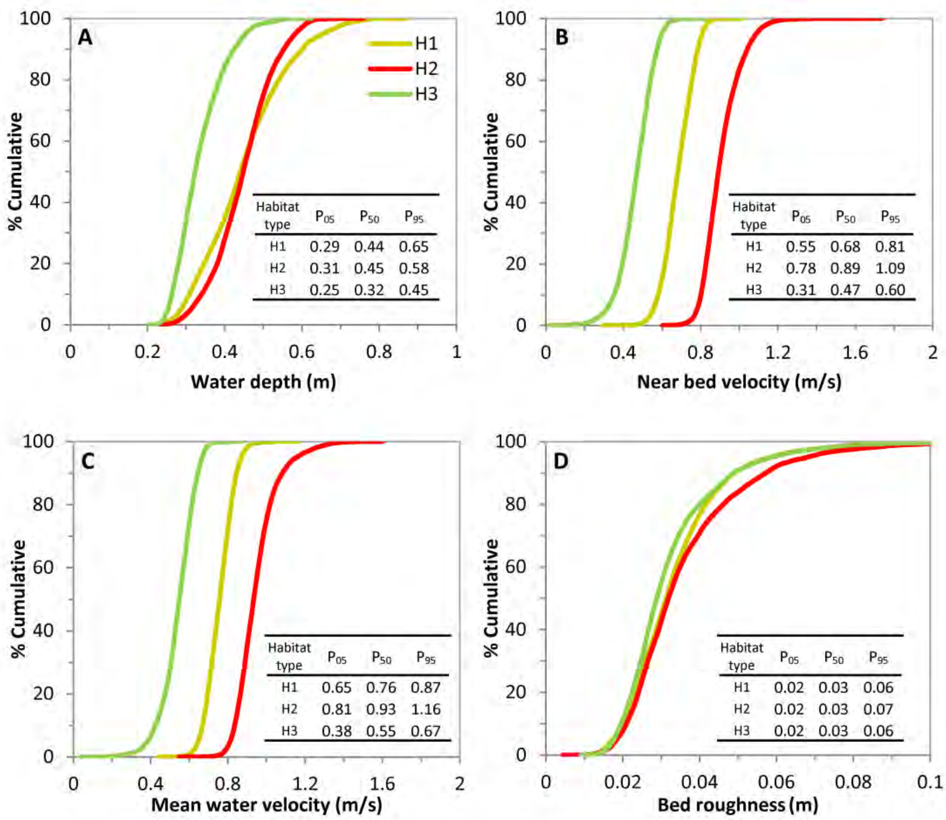


Figure 5. Cumulative Frequency Distribution Curves for each Habitat (H) type: for water depth, near bed velocity, mean water velocity, bed surface complexity. Inset tables show the 5th (P_5), 5th (P_{50}) and 95th (P_{95}) percentiles.

3.2. Habitat heterogeneity and complexity

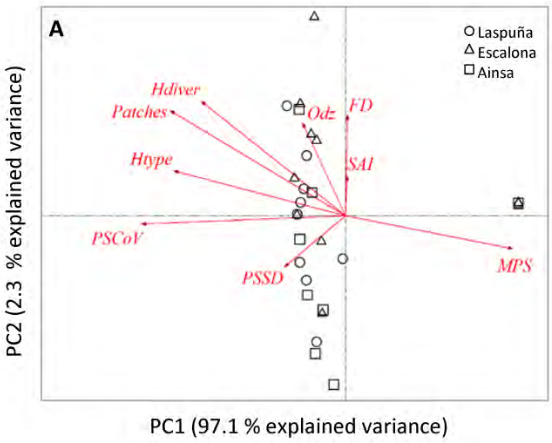
Habitat heterogeneity and surface complexity showed great variability at the 1 m^2 scale (Table 3). For instance, the number of patches varied between 1 and 10 per area, while Patch Size Coefficient of Variation ranged from 0 to 2.2 (where a zero value represents an area containing just one patch). Perimeter-area ratio ranged from 4 to 68 m/m^2. The metrics representing surface complexity also varied between sampling areas. For instance, the minimum Fractal Dimension was 1.01 indicating a smooth surface, while the maximum was 1.14 suggesting a convoluted or complex surface; what these values represent, in terms of the visual appearance of a patch of river bed, is illustrated in Figure 3.

Table 3. Range of values for habitat heterogeneity and surface complexity metrics for the two study scales. The minimum and the maximum values are used to characterize the obtained ranges.

Parameter	Scale 1 m^2		Scale 16 m^2	
	Minimum	Maximum	Minimum	Maximum
Habitat heterogeneity				
Habitat Types	1	3	2	3
Habitat Diversity	0.0	0.9	0.0	1.1
Number of Patches	1	10	5	47
Patch Diversity*	0.0	1.3	0.0	1.9
Mean Patch Size (m^2)	0.1	1.0	0.3	3.2
Patch Size Standard Deviation (m^2)	0.0	0.5	1.1	6.4
Patch Size Coefficient of Variance	0.0	2.2	2.0	5.6
Mean Patch Edge (m)*	1.4	4.0	1.9	5.6
Total Edge (m)*	4.0	16.1	19.0	133.4
Mean Perimeter Area Ratio (m/m^2)*	4.0	67.8	16.4	33.1
Surface complexity				
Surface Area (m^2)	1.0	1.1	14.0	16.1
Fractal Dimension	1.01	1.14	1.01	1.15
Bed Roughness (m)	0.019	0.049	0.019	0.057

* Metrics correlated with at least one other and so were not included in subsequent analyses.

The *PCA* indicated that several of the habitat heterogeneity metrics were correlated (Table 3; see symbols), so the redundant ones were removed. Those removed were Patch Diversity, Mean Patch Edge, Total Edge and Mean Perimeter-Area ratio. The first two axes of the final *PCA* captured 99 % of the total variance in the habitat heterogeneity and complexity metric values at the 1 m^2 scale (Figure 6A). The first dimension captured 97.1 % of the variance and represented a gradient of patch size and patch size coefficient of variation, along with the number of habitat types present in each area. Most of the samples were situated on the left of the plot (representing high values of habitat heterogeneity and small patch size), with a tight cluster to the right (samples with larger mean patch sizes and correspondingly low numbers of habitat types). The second axes of the *PCA* represented a gradient of surface complexity. The low percent variance for this axis (2.3 %) indicated that variation in complexity was less important in distinguishing between samples than heterogeneity.



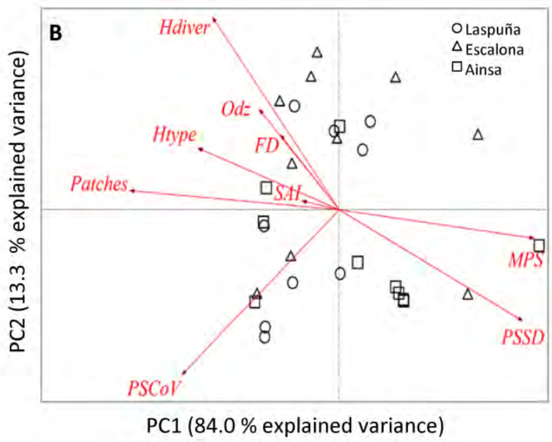


Figure 6. PCA ordination of the heterogeneity and surface complexity parameters of the 29 surveyed locations at (A) 1 m^2 and (B) 16 m^2. Symbols are used to group the samples by sampling site (Table 1 provides a full description of all abbreviations).

3.3. Relationship between habitat and macroinvertebrates

GEEs indicated that, for the 1 m^2 area data, there were significant relationships between some of the habitat heterogeneity metrics and measures of invertebrate diversity (Table 4 and Figure 7). Abundance was positively related to both Habitat Diversity and the Patch Size Coefficient of Variance (Habdiver: Wald $\chi^2 = 9.54$, $p < 0.002$; PSCoV: Wald $\chi^2 = 8.61$, $p < 0.003$) but was inversely related to Mean Patch Size (MPS: Wald $\chi^2 = 7.38$, $p < 0.007$). Benthic diversity (represented by Shannon's diversity index) was positively related to the Number of Patches and Mean Patch Size (Patches: Wald $\chi^2 = 7.17$, $p < 0.007$; MPS: Wald $\chi^2 = 4.53$, $p < 0.033$). In terms of metrics of surface complexity, taxon richness was positively related to bed roughness (σ_{dz}: Wald $\chi^2 = 3.98$, $p < 0.048$). No relationship was found between rarefied taxon richness and any of the habitat parameters.

Table 4. Invertebrate response shown by Generalised Estimating Equations (GEEs) fit to habitat heterogeneity and surface complexity data at 1 m^2. Values of Wald-Chi square test are shown.

Metrics	Abundance	Taxon Richness	Taxon richness rarefied	Shannon's diversity index
Habitat heterogeneity				
Habitat Types	1.18	2.00	1.45	3.62
Habitat Diversity	9.54 **	0.31	0.46	0.23
Number of Patches	3.72	1.66	1.81	7.17*
Mean Patch Size (m^2)	7.38*	1.35	1.87	4.53*
Patch Size Standard Deviation (m^2)	0.17	0.28	0.22	3.22
Patch Size Coefficient of Variance	8.61**	0.66	0.13	0.01
Mean Patch Edge (m)	2.68	0.33	1.5	0.26
Surface complexity				
Surface Area (m^2)	0.11	0.19	0.15	0.18
Fractal Dimension	0.01	0.00	0.02	0.03
Bed Roughness (m)	2.97	3.90**	2.1	3.00

** p-value <0.005 and * p-value <0.05

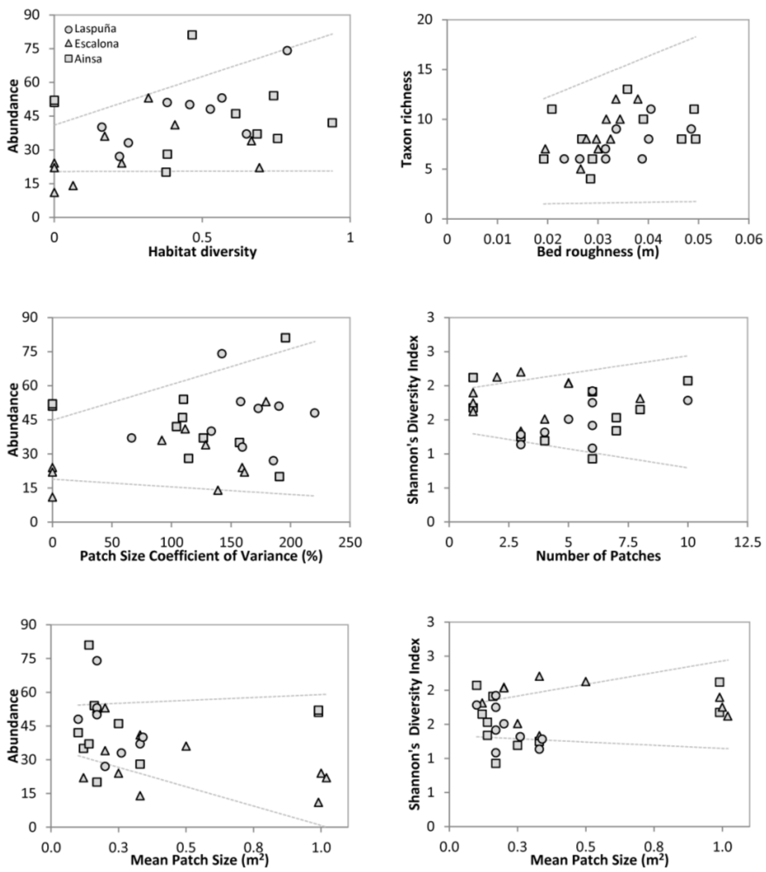


Figure 7. Scatterplots of the macroinvertebrates parameters and heterogeneity and complexity metrics. Only significant relationships (determined from GEE analysis) are shown. Dashed lines represent the 95 % Confidence Interval Bounds of the relationships. Symbols are used to group the samples by sampling site (see Figure 1 for location of the sites).

The *RDA* indicated that, using 2 dimensions, around 37 % of the total variation in community structure could be explained by habitat metrics calculated at the 1 m^2 scale (Figure 8A). The marginal effects values (Table 5) indicated that between-sample differences correlated significantly with Habitat Diversity and Patch Size Coefficient of Variation, (λ_1 = 0.1 and 0.09 respectively). The biplot showing the relation between individual taxa and habitat parameters is shown in Figure 8B (only the most abundant taxa are shown, to avoid clutter). Correspondence in the directions of the vectors suggested increased abundance of *Heptageniidae* in response to increasing Habitat Diversity and Number of Patches, as well as increasing abundance of *Baetidae* and *Rhitrogena* in locations with high Patch Size Coefficient of Variance and Patch Size Standard Deviation. *Ecdyonurus* was more abundant in locations with lower Habitat Diversity, but larger Mean Patch Size.

Table 5. Redundancy Detrended Analysis eigenvalues and related statistics for data at 1 m^2 scale and marginal effects resulting from the forward selection.

RDA	**Axes 1**	**Axes 2**	**Axes 3**	**Axes 4**
Eigenvalues(lambda)	0.287	0.088	0.056	0.03
Species-environment correlations	0.714	0.773	0.675	0.659
Cumulative % variance of species data	28.7	37.5	43.1	46.1
Cumulative % variance of species-env relation	60.5	79.1	90.9	97.2
Sum of all canonical eigenvalues				0.474
Marginal Effects resulting from the forward selection				
Parameter				**Lambda1**
Hdiver				0.1
MPS				0.1
PSCoV				0.09
Patches				0.08
σ_{dz}				0.06
FD				0.06
SA				0.05
PSSD				0.03
Htype				0.02

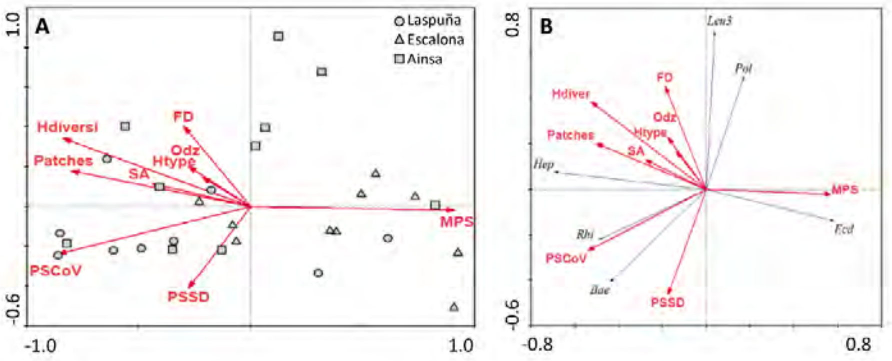


Figure 8. Redundancy Detrended Analysis (RDA) at 1 m² scale: (A) Biplot of the habitat metrics and invertebrate samples. (B) Biplot of the habitat metrics and invertebrate taxa. See Table 1 for abbreviations. Note that taxa are abbreviated: Baetidae (Bae), Rhitrogena (Rhi), Heptageniidae (Hep), Leuctridae (Leu3), Polycentropodidae (Pol), Ecdyonurus (Ecd).

3.4. The influence of spatial scale

Median values for bed roughness and hydraulic characteristics (d, V_m, V_{nb}) at the 16 m² scale were similar to those at 1 m² (Table 2). The ranges of the hydraulic variables increased by around 15 % at the larger scale compared to the smaller one, while bed roughness increased by up to 40 %, suggesting that characterization at larger scales captured a higher range of particle sizes than smaller scales.

In general, the ranges habitat heterogeneity and surface complexity metric values were highest at the 16 m² scale (Table 3). Increases in some metric values with scale are self-evident, as for instance in the case of the number of patches and total edge. The magnitude of the difference between 1 and 16 m² differed between metrics: some increased markedly (Patch Size Coefficient of Variation) but others remained more or less the same (Number of Habitat Types, Habitat Diversity Fractal Dimension). The patterns in the *PCA* for the parameters that represent habitat heterogeneity and complexity at 16 m² scale were similar to those obtained at 1 m² scale, Figure 6B).

The *RDA* ordination using the 16 m² habitat data was able to account for 30 % of the total variation in species data (Figure 9). This value is slightly lower than that obtained at 1 m² (Table 5). Habitat Diversity and Patch Size Standard Deviation had the highest values for λ_1 (0.11 and 0.11 respectively), but only Habitat Diversity contributed significantly to the overall model for this scale (Table 6). While the characteristics of some samples appeared to be determined primarily by habitat heterogeneity (i.e. they sat close to habitat heterogeneity vectors), others appeared to be influenced strongly by surface complexity. The *RDA* biplot showing the relation between individual taxa and habitat parameters is shown in Figure 9B (as with Figure 8B, only the most abundant taxa are shown). The abundance of Heptageniidae was positively correlated habitat diversity at 16 m², while *Caenis* and *Ecdyonurus* were more abundant in samples with lower Habitat Diversity at this scale.

Table 6. Redundancy Detrended Analysis eigenvalues and related statistics for the 16 m^2 scale and marginal effects resulting from the forward selection.

RDA	Axes 1	Axes 2	Axes 3	Axes 4
Eigenvalues(lambda)	0.236	0.075	0.04	0.02
Species-environment correlations	0.666	0.833	0.529	0.511
Cumulative % variance of species data	23.6	31.1	35.1	37.1
Cumulative % variance of species-env relation	61.1	80.5	90.8	95.9
Sum of all canonical eigenvalues				0.386
Marginal Effects resulting from the forward selection				
Parameter				**Lambda1**
Hdiver				0.11
PSSD				0.11
Htype				0.08
σ_{dz}				0.07
FD				0.07
MPS				0.06
Patches				0.05
SA				0.04
PSCoV				0.03

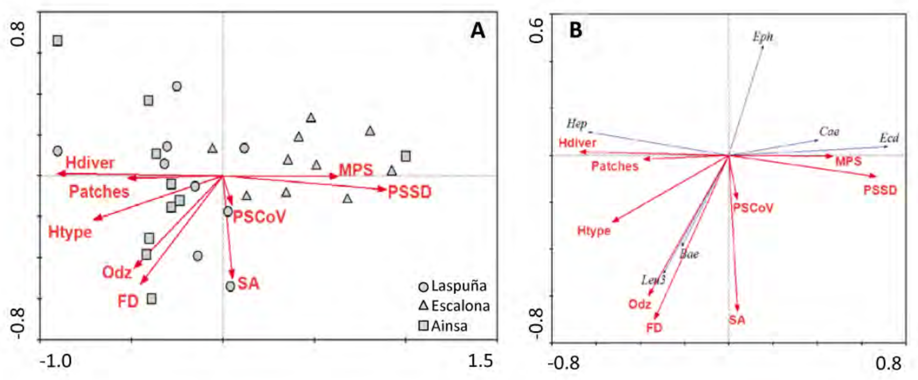


Figure 9. Redundancy Detrended Analysis (RDA) at 16 m^2 scale: (A) Biplot of the habitat metrics and invertebrate samples. (B) Biplot of the habitat metrics and invertebrate taxa. See Table 1 for abbreviations. Note that taxa are abbreviated: Ephemerella (Eph), Caenis (Cae), Ecdyonurus (Ecd),Baetidae (Bae),Leuctridae (Leu), Heptageniidae (Hep)

4. DISCUSSION

4.1 New insights into an old problem

The degree to which lotic invertebrates are affected by bed surface complexity and local hydraulics has remained an open question in aquatic ecology for decades. Barnes et al. (2013) provide a recent and insightful synopsis of the problems of trying to demonstrate links between habitat structure and benthic invertebrate diversity. As these authors pointed out, a central problem is the quantification of habitat structure. We took the same general approach as them, breaking down habitat structure into elements related both to its heterogeneity and its complexity. However, our work differed in three important ways. First, our assessment of habitat heterogeneity was based on habitats defined using both bed micro-topography (related to sediment grain sizes) and flow hydraulics. Second, our assessments of bed surface complexity were based on high resolution bed elevation data collected using an ADCP deployed over large areas (then used to compute bed roughness). Finally, we also assessed whether characterising habitat over a larger area (16 m^2) around the point sampled for invertebrates might improve understanding of habitat effects relative characterisations using only local (1 m^2) data. Results support the expectation that alternative approaches to habitat characterization (high resolution, integrating topographic and hydraulic data) are capable of revealing often hidden relations between habitat structure and invertebrate diversity. Consequently, the results provide further support for the habitat heterogeneity hypothesis postulated by Simpson (1949). However, results also show that characterising habitat structure over a larger area surrounding the point of interest does not necessarily provide improved insights into species-environment relations.

4.2. Habitat structure and benthic invertebrates

In the case of the gravel bed River Cinca, habitat diversity, patch size and patch size coefficient of variation influenced macroinvertebrate abundance and community structure in a positive direction. Conversely, abundance was negatively correlated with Mean Patch Size (*MPS)*. This negative correlation relates to the interaction between patch size and the number and types of patches present in the 1 m^2 sample areas i.e. areas with small mean size are those with a greater number and types of patch ($MPS = 1.0015 \times Patches^{-1.003}$; $R^2 = 0.99$; $p<0.001$), so the decreased abundance as patch size increases may reflect the associated reduction in habitat heterogeneity. Nevertheless, larger patches had greater taxonomic diversity, as indicated by Shannon. The *GEE* models suggested that patch size as well as the number of patches influences taxonomic diversity (taxon richness and Shannon Index models both significant). Bed surface complexity also appeared to influence taxon richness. Many authors have found higher species richness in freshwater environments with more complex habitat architecture (Jeffries, 1993; Downes et al., 1995), with numerous explanations for this having been postulated (e.g. rough beds may provide zones used by individuals as a refuge from disturbance or predators; Downes and Jordan, 1993).

As well as influencing invertebrate diversity (abundance and taxon richness), results from *RDA* indicate that habitat structure influenced community structure in the River Cinca. Variation in community structure between the sampled areas was related to habitat diversity and the patch size coefficient of variation. Similarly, Anderson et al. (2006) found that β-diversity in macroinvertebrates assemblages was positively related to environmental heterogeneity. Barnes et al. (2013) found correlations between surface complexity and macroinvertebrates (i.e. macroinvertebrate diversity positively correlated fractal dimension; taxonomic richness and abundance positively correlated fractal dimension and surface area). However they found no significant influence of the heterogeneity parameters. This highlights not only the value of high resolution habitat charactersation, but also that flow hydraulics (which were not included by Barnes et al., 2013) may be a key element of habitat diversity.

4.3. Scale issues

One of the reasons suggested as to why some studies have failed to find relations between habitat structure and diversity is because the influence of habitat varies with scale. On the one hand, experiments carried out at multiple scales may be able to identify the processes operating at different scales and their roles in generating patterns in habitat and ecological communities (Robson and Barmuta, 1998); on the other hand, multi-scale studies may also conflate different types of scale (Lechner et al., 2012) and this may impact both their ability to detect relations and to identify the scale at which interactions occurring. Few aquatic ecologists have explicitly distinguished between what are termed the *intrinsic, the observation* and the *analysis* scales (Wu and Li, 2006 and references therein). The scale at which an organism interacts with its environment is termed the *intrinsic scale*. Thus, the intrinsic scale is the scale at which the highest correlations between environmental and ecological data occur. However, the data used to investigate such correlations may be collected and analysed at different scales to the intrinsic scale. The *observational scale* refers to the locations and densities of measurement points. In turn, this differs from the *analysis scale*, which is usually coarser than the observational scale. In the present study, the intrinsic scale is the scale at which benthic invertebrates experience and interact with flow hydraulics and riverbed substrate, the observational scale is dictated by the ADCP observations, while the analysis scale is the scale at which habitats and benthic communities were compared (1 and 16 m^2).

What the intrinsic scale is for aquatic invertebrates is a difficult question to answer. The difficulty is related to the fact that dispersal kernels (the probability of dispersal from a parent) are poorly known, especially the movement distance accumulated over the entire larval stage (Lancaster et al., 2011). Larvae of a few mm in length are experiencing and respond to flow hydraulics at very small scales; e.g. flow forces causing an individual to alter its micro-position so as to be aligned with velocity vectors are likely akin to its body size, rather than to larger scale flow structures. Characterising habitat at these organism scales has long been a major challenge, although technological and methodological developments are permitting progressively finer scale characterisation (Buffin-Bellanger et al., 2006; Johnson and Rice, 2014), including measurement devices once constrained to laboratory settings now being deployed in the field (Cameron 2013; Biggs 2017). Thus, the gap between the observational scale and the fine scale at which animals experience flow is becoming smaller. However, this very small scale is not necessarily the most appropriate one to consider as being the intrinsic scale. Species-environment models typically use abundance as the ecological response variable, so the area over which abundance is assessed has to be larger than body size. For birds and mammals the territory size is frequently used (Suorsa et al., 2005), while for benthic invertebrates dispersal kernels are appropriate but poorly known. Additional complications arise when the focus is on how the environment influences diversity, because different species have very different kernels (e.g. sessile ones such as Simuliidae versus more mobile ones such as Ephemeroptera), whereas allometric relations also suggest that larger individuals should, in general, be more mobile than smaller ones (Peters, 1983; Englund and Hamback, 2004). Given that predators are larger than their prey, kernels may also differ between assemblages in different locations according to trophic structure (Holt, 1995). These differences suggest that identifying a single intrinsic scale that allows comparison of diversity between locations will be difficult. Such scale issues may help explain why demonstrating links between diversity and habitat structure has proved difficult.

Other scale issues concern characterisation of habitat and so relate to the observational scale. In the case of the Cinca, the observational scale for environmental data was dictated by the ADCP, which produced between 16 (nearbed velocity) and 100 (depth) points per square metre of bed. However, the IDW interpolation procedure applied to these data points then gave a higher and constant density of one point per 0.01 m^2 for all the variables; i.e. the observation scale of 0.01 m^2. Thus, the observational scale that fed into the analysis differed from the empirical observational scale and contained errors. These errors (RMSE 7 % and 18 % for depth and mean velocity respectively) may influence the ability to detect any species-environment relations that existed.

Two analysis scales were used in the Cinca, i.e. 1 and 16 m^2. The larger of these was determined from analysis of velocity data, which showed that measuring velocity over increasingly large areas does not necessarily increase the values of flow heterogeneity (defined as the ratio between the P_{95} and P_5 of the velocity distribution). An asymptote in velocity heterogeneity was reached at 16 m^2 and this was taken as an appropriate scale to describe hydraulic conditions. Habitat heterogeneity and complexity in the 16 m^2 areas around each invertebrate sampling point (surber area 0.25 m^2) were characterised, to assess the possibility that surrounding conditions may influence invertebrate community structure. The explanatory power of the model using measures of habitat structure at this larger scale was no better than that using only 1 m^2 data. This is consistent with the findings of Oldmeadow et al. (2010) who reported that invertebrates are not clearly more influenced by the physical properties of surrounding areas than by the characteristics of the patch itself. However, complexity parameters have more influence on benthic assemblages at the large scale than at local scale. Larger scales capture a higher range of particle sizes than local scales, creating a microhabitat architecture that may afford protection from flow and predators (Robson and Barmuta, 1998).

Dispersal by benthic invertebrates occurs chiefly by individuals either walking or crawling across the bed or by drifting downstream within the water column. Passive drift dispersal over large distances may override local movements, and so has been suggested as something which may obscure species-environment relations (Barnes et al., 2013). However, there is increasing evidence that the influence of drift dispersal has been overestimated, even for organisms long considered as iconic drifters (Lancaster and Downes, 2013) and that maternal behaviour may play a key role, at least for some taxonomic groups (Lancaster et al., 2011).

Several other factors may obscure the response of macroinvertebrates to habitat structure. We found a short ecological gradient in the upper Cinca (i.e. low β-diversity; variability in species composition among localities, as per Anderson et al., 2006). The low β-diversity during summer season, attributable to natural life cycle changes, has been previously reported in the upper Cinca and nearby catchments (see Béjar et al., 2017 and Buendia et al., 2013, respectively). First, this short gradient may confound attempts to relate turnover to environmental conditions in sampled patches. Second, species do not live in isolation, reacting only to the physical environment; rather, local changes in the spatial distribution and density, especially during non-disturbance periods, may reflect interspecific interactions (Peckarsky et al., 1980; Lancaster and Downes, 2010). Third, species may be able to tolerate a wide range of hydraulic and sedimentary conditions, and so occur in locations that are quite different; they may have very low abundance or be absent only from locations at the extreme ends of hydraulic and sedimentary gradients (Lancaster and Downes, 2010).

4.4. Constraints

Our work represents a snapshot of the physical habitat characteristics and invertebrate diversity in 29 sampling locations across a 9.8-km reach of the upper River Cinca. Physical habitats are dynamic, and vary temporally as a result of flow fluctuations and sedimentary dynamics. We characterised habitat and invertebrates on 2 days during summer low flow conditions ($Q = 12.6$ m^3/s). However, the month preceding this had a mean Q of 27 m^3/s, with a peak Q of 113.6 m^3/s (representing an ordinary event). Undertaking our study during or immediately following such high flows might result in different findings, as both habitat structure and invertebrate communities may have differed to those presented here. Indeed, it is possible that the rather low β-diversity of our samples data is indicative of assemblages impacted by and still recovering from flood disturbance.

While our approach to hydraulic and topographic characterisation has provided insights into species-environmental relations, some limitations should be acknowledged. First, the ADCP data were interpolated to provide a continuous raster data set of each variable across each sampling area. Interpolations were applied to data that that had different resolutions in each of the sampling areas, and the resolution of the variables differed (much higher for depth than velocity). Uneven data resolution influences the relative quality of point estimates following interpolation. Second, interpolation is itself thorny subject. None of the 3 interpolations methods tested consistently produced the most accurate result for all the habitat variables. The *IDW* method is known to represent topography well (see the review by Arun, 2013), but other interpolation methods (e.g. krigging) are commonly used for hydraulics (Kratzer et al., 2006). We elected to use the same method for both hydraulic and topographic data and, based on relatively low RMSEs for our data sets, we considered this acceptable.

4.5. River management

River management and restoration projects are frequently designed to increase habitat heterogeneity (Amoros, 2001). Several of the approaches used in the River Cinca for characterisation and typing could be valuable in such projects.

Traditionally habitat types have been identified visually by freshwater ecologists, and numerous protocols have been developed to aid habitat classification in the field (Bisson et al., 1981; Environmental Agency, 2003). The ADCP-based approach to data collection, combined with the use of CLARA and silhouette widths to define coherent habitat groupings, represents a systematic and quantitative method to define habitats and produce high resolution maps to show their distribution and properties within the sampling areas. If surveys are repeated over time, changes in habitat structure can be detected and quantified (e.g. alterations in heterogeneity or complexity). So-called 'objective-based' river management requires assessment of whether objectives have been met, and these approaches represent a robust means of determining whether habitat structure is changing in response to management interventions such assessment. While there are some practical limitations to the circumstances in which some of surveying technology can be used (e.g. use of ADCPs in shallow rivers due to blanking) future technological improvements may negate such issues and allow easier, more rapid and a wider scope for deployment.

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CHAPTER 7

DISCUSSION, CONCLUSIONS AND LIMITATIONS

1. DISCUSSION

This thesis has analysed the degree to which certain human activities, i.e. in-channel gravel mining and hydropeaking, affect the transfer of water and sediments through a 10-km river reach (see location details in Figure 1 of Chapter 1) and the ecological implications of this. Furthermore, the work examined the degree to which habitat heterogeneity and surface complexity determines ecological diversity in a representative section of the study reach. The work was carried out in the upper River Cinca, a regulated mountainous wandering gravel-bed river located in the Southern Pyrenees (Ebro basin).

Nowadays, gravels are mined from the channel of the upper Cinca mainly in association with channel maintenance works to prevent flooding, as well as for construction aggregate. Three in-channel gravel mining activities were carried out during the study period of the thesis. Mining is all regulated by the National Water Law approved by the Royal Legislative Decree 1/2001. Over the last 16 years the Ebro Water Authorities has authorized around 70 gravel extractions in a 20-km river reach of the upper Cinca (Figure 1 in Chapter 1), within which the study reach is located. These numbers highlight not only that mining remains a frequent human perturbation in the Cinca, as in many mountain rivers, but also reinforces the need to evaluate the potential effects on river morphology and ecology in order to propose recommendations to minimize the impacts, with the objective to guarantee the ecological functioning of such fluvial systems. Like the Cinca, many Pyrenean rivers are also subjected to hydropeaking. Hydropeaks modify the magnitude, duration and volume of flows in different ways, according to the particular production needs and operation procedures. The magnitudes of flow fluctuations are legally controlled by the Instruction for the Hydrological Planning approved by the National Order (ARM72656/2008). In the case of the upper Cinca, there is a complex hydropower system with small reservoirs, water transfer tunnels and hydropower plants (as shown in figure 6 of Chapter 2). In the study reach, the Laspuña hydropower plant generates regular hydropeaks (i.e. maximum releases may reach a value of 25 m^3/s) just one hundred meters downstream from the upstream section of the study reach. A summary of the main finding of this thesis and a discussion of the wider environmental implications for river systems are given in the next paragraphs. Note that each of the chapters of the thesis already includes a specific discussion, so the current chapter present more of an overall sysnthesis.

Catchment-scale processes, localised human impacts and reach-scale variability

Present physical catchment conditions (e.g. land cover, morphometry, drainage density, soils, geology) together with climatic forcing (e.g. rainfall intensity andvolume, air temperature) are widely accepted to have a major control on the catchment-scale processes that ultimately affect fluvial dynamics. But also past climatic conditions, geological processes and impacts of past human activities have a long-term effect on landscape forms and processes and, notably, on the associated flow and sediment fluxes in a given river reach (Brierley et al., 2009). Within this context, a catchment is composed by sub-catchments that may have different physical attributes, together controlling the production and transfer of water and sediment downstream. The results of this thesis highlight how tributaries and human impacts determine the spatial and temporal variation of suspended sediment transport and water yield in a 10-km length river reach of a large basin that is representative of the current hydro-sedimentary processes that take place in most rivers of this mountainous region.

The physical attributes of the tributaries control their role in terms of water and sediment supply to the mainstem Cinca. The River Bellos drains a limestone canyon and contributes importantly to the Cinca's water yield (13 % of the whole Upper Cinca catchment), whereas the San Vicente is an ephemeral creek that drains an area of badlands and supplies considerable amount of fine sediments to the Cinca (see Chapter 3 for more data and discussion on this). The sediment yield at the downstream end of the study reach increases around 200 % compared to the yield assessed at the upper part, an increment substantially driven by the sediment supply from the San Vicente Creek (no data is available to quantitatively assess the particular role of this tributary). Such differences are also observed in other mountain basins where sub-catchments supply large amounts of sediments that are disproportionate to their size (e.g. Gallart et al., 2005; Garcia Ruiz et al., 2008; López-Tarazón et al., 2012). Differences in sediment supply between sub-catchments can be characterised by the effective catchment area, i.e. the spatial area which directly contributes to or transports sediment along the fluvial network (Fryirs, 2013). The degree of connectivity of tributaries that supply sediments to mainstem rivers can also have an important effect on the sediment load. For instance, Marteau et al. (2017) found how a tributary reconnection have a major influence on fluvial dynamics in a supply limited main-stem in the Lake District (NW England). The reconnection of the ephemeral tributary resulted in only a small increase in the whole catchment size (i.e. 1.2 %), but its reconnection increased the mainstem sediment yield by 65%.

A substantial but variable percentage of the sediment delivered to the basin's mainsteam is sometimes deposited in the channel bed, owing to a reduction of the sediment transport capacity when the river flows downstream through the main valley. López-Tarazón et al. (2011) reported that channel bed storage equalled 5 % of the annual suspended sediment load delivered to the outlet of the nearby River Isábena (see location details in Figure 1 of Chapter 1). This accumulation varies as a function of the production of sediment in the source areas, its transfer to the channel network and the transport capacity of flood events (Buendia et al., 2013a; Piqué et al., 2014). In-channel mining activities also affect the amount of sediments stored in the bed, and this will vary depending on the frequency and magnitude of these extractive operations. Mining typically results in changes of fine sediment storage that, in turn, increases the transport of suspended sediment downstream, a phenomena that modifies the habitat for biota and may reduce biodiversity in the long term. This biodiversity reduction was proved to be a common characteristic of river channels structurally affected by large sediment loads (e.g. Dewston et al., 2007; Buendia et al., 2013a, 2013b, respectively). However, the frequency of the extractive operations does not affect fine sediment storage permanently. This may result in other biological responses.

Dams and hydropeaking alter river flow regimes (see for instance the studies by Batalla et al., 2004 and Zolezzi et al., 2009, respectively). In our case, the upper part of the study reach (upstream from the Laspuña hydropower plant) shows a highly variable pattern (i.e. the channel typically displays low flows but is affected periodically by flush floods when the gates of the Laspuña Dam are open), whereas a hundred meters downstream, hydropeaking increases discharges and the total water yield, producing a more constant flow regime (although there are hourly and daily fluctuations). The specific effects of these changes (from the natural) and differences (between sections) on the river ecosystem are yet to be comprehensively studied. Fluctuations in flow generated by hydropeaking may for instance alter sediment transport. Sediment deposited in the channel bed may be easily entrained and transported during large fluctuations in flow owing to the increase in the river's flow competence. Although the impact of flow regime on benthic assemblages was not addressed in this study, several authors reported that changes in flow velocity, wetted (cross-section) perimeter and temperature alter the physical habitat of fish and invertebrate species (e.g. Bruno et al., 2013; Rocaspana et al., 2016).

The results of this thesis indicate as the variability of natural (e.g. tributaries) and human factors (hydropeaking, gravel mining) define different river sub-reaches within river reaches. The combination of these factors potentially affects water and sediment fluxes, hence physical habitat and invertebrate assemblages. This will exert and additive or compensatory effect (as defined in Chapter 3). Therefore, the definition of river sub-reaches in altered rivers based on both, the physical attributes of sub-catchments and the localised human activities, is important to better understand the spatial variability of physical and ecological processes along river reaches; and, in turn, it should aid water authorities on the decision-making process of future human interventions, the assessment of impacts and the design of restoration actions in modified river systems.

Patterns of physical and ecological processes

Hydro-sedimentary and ecological processes in the upper Cinca show different patterns compared to nearby catchments. Overall, the upper Cinca has a relatively low sediment yield compared to neighbouring basins of similar size and physical conditions (Figure 1). However, and despite the low sediment load of the river, this work has highlighted how suspended sediment load can increase four-fold in a particular river reach owing to inputs from tributaries and human-related activities. In relation to ecology, De Jalón et al. (1988) reported that the invertebrate species richness was lower in the Cinca than in the unregulated River Ara (see location details in Figure 1 of Chapter 1). The difference was attributed to the suppression of thermophilous species by competitive disadvantage related to the hydropower plant located in the upper Cinca (Laspuña hydropower plant). Lower invertebrate densities were found in the Cinca compared to the nearby River Isábena (Buendia, 2013b). Moreover, and despite the low densities reported in this thesis, benthic assemblage changes from the upper sampling section to the downstream section of the study reach (see Chapter 6). The upper Cinca is a highly dynamic system where seasonal and annual fluxes vary significantly. Correspondingly, erosion, transport and deposition of sediments also contribute to physical habitat heterogeneity and surface complexity (i.e. creating new habitat templates). Additionally, it has been demonstrated that human disturbances in the reach play an important role, increasing fine sediment availability and bed structure (due to gravel mining), and changing the magnitude of flows (due to hydropeaking); under such conditions, macroinvertebrates present in the river have to be adapted and resilient to tolerate environmental variability. Similar conclusions were pointed out by, for instance, Robinson (2012) in relation to resilience to experimental floods. The present results show how in a relatively small geographical area (i.e. Southern Pyrenees) defined by similar characteristics (e.g. geology, soils, land cover, climate), hydro-sedimentary and ecological patterns between catchments differ substantially; this highlights the importance of processes operating at multiple spatial scales and the influence of human perturbations.

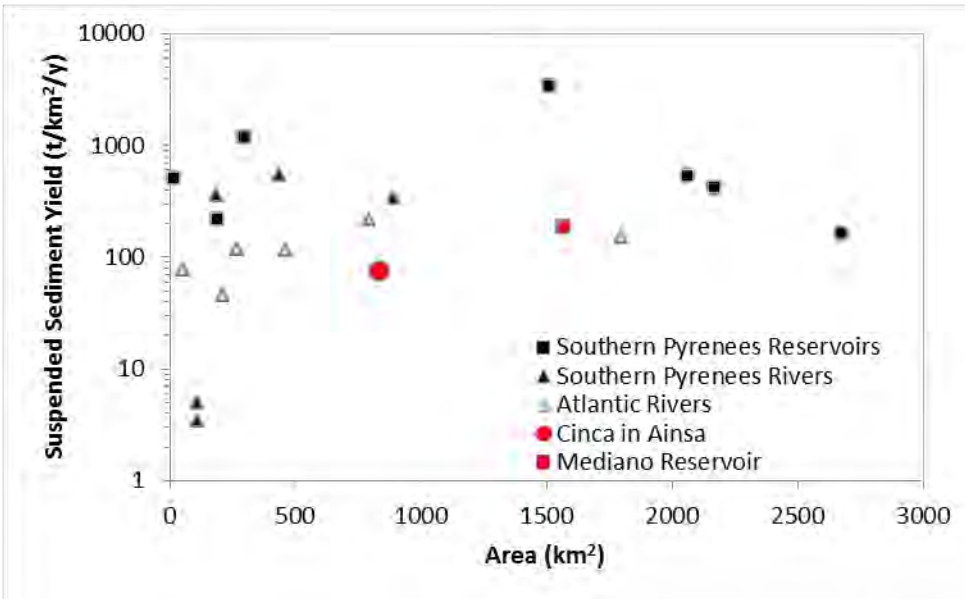

Figure 1. Suspended sediment yield in different Pyrenean catchments. Note the log scale on the vertical axis. The difference between values of Cinca in Ainsa and Mediano Reservoir is caused by input from the River Ara. Data source: Vericat and Batalla, 2010; Batalla and Vericat, 2011; Zabaleta et al., 2011; Lopez-Tarazón et al., 2012; Buendia, 2016; Lobera 2016; Tuset, 2016.

Off-site and deferred effects of gravel mining

In-channel mining triggers a series of physical and ecological process that interact at multiple spatial and temporal scales. During mining operations, alluvial forms (e.g. riffles, pools) are modified or removed, bed structure disturbed and, consequently, physical habitat is altered (Kondolf, 1997; Rinaldi et al., 2005). After mining, deeper and narrower channels (also more regular) are created, transforming complex channel forms and hydraulics into single-thread channels displaying more homogenous flow hydraulics, altogether reducing physical habitat diversity and bed surface complexity (Church et al., 2001; Rinaldi et al., 2005). As shown in Chapter 6, habitat diversity and complexity are important elements of habitat structure that support greater diversity; thus, reductions in complexity and diversity likely affect macroinvertebrate communities. While local (on-site) effects of in-channel mining on local physical habitat have been extensively studied (e.g. Erskine, 1997; Kondolf, 1994), studies focussed on the effects downstream from the mined area (off-site effects) are scarce (e.g. Mori et al., 2011). Mining operations normally entrain fine sediment under low flows, which lack competence to transfer this sediment downstream. Consequently, most of the released sediment is deposited in the channel increasing the availability of fine sediment and determining the dynamics and loads of the subsequent food events. These deferred impacts were analysed in Chapter 5.

Increasing sediment loads is one of the most important and persistent human disturbance on aquatic ecosystems globally (Wood and Armitage, 1997). Siltation affects physical habitat in the downstream reach by filling interstices and reducing gravel bed porosity and bed surface complexity (Garcia Molinos and Donohue, 2011; Larsen et al., 2009). Additionally, siltation may prompt indirect effects on macroinvertebrates such as decreasing food quality and availability (Suren, 2005; Jones et al., 2012). Sediment deposited in the channel downstream from mining areas is available for the next competent flow events (i.e. flows with the hydraulic forces required to entrain fines). Buendia et al. (2013b) reported that the annual sediment cycle determines macroinvertebrate assemblages, with only species with those traits conferring resistance and resilience able to persist. It is suggested that the deferred effect of in-channel mining on the annual sediment cycle can also affect macroinvertebrates that are adapted to a given (natural) annual sediment cycle. The magnitude of the deferred impacts will depend on the previous channel conditions (e.g. the amount of the sediment stored in the channel before the extraction). These conditions also control the impact of the mining on the suspended sediment concentrations subsequently, and the invertebrate responses. More sediment available in the channel generates higher suspended sediment concentration during periods when gravels are being mined. This thesis has shown that gravel mining not only increases concentrations and loads, but changes the sizes of the particles entrained (Chapter 4), potentially influencing bed porosity and bed surface complexity (Bilotta and Brazier, 2008). Altogether, as analysed in Chapter 5, has a direct impact on invertebrate drift.

The duration of exposure determines the impact

Traditionally, threshold concentrations for chemical pollutants have been applied in river management programs (Groffman et al., 2006). In the case of the suspended sediment, Doegg and Miller (1991) indicated that a threshold value would be a concentration that initiates macroinvertebrate responses. Nevertheless, the use of such absolute values may result in ineffective or inappropriate management decision since aquatic biota respond not only to the concentration of suspended sediments but also the duration of exposure (Newcombe and Macdonalds, 1991). Garcia Molinos and Donohue (2011) found that duration of individual perturbations can drive different responses of biotic assemblages. Similar outcomes are found during the mining activity monitored in the upper Cinca where duration of exposure to perturbation influenced macroinvertebrates response (see Chapter 5 for further details on this). While suspended sediment released by mining operations equates to 10 % of the average suspended sediment load registered during floods (as shown in Chapter 4), maximum observed concentrations were similar to those recorded in natural flood events.

The duration of high concentrations (or exposure) is considered to be a key factor to reduce the impact of sediment extraction on macroinvertebrates. Thresholds based on durations have already been applied for fish assemblages. For instance, Brignoli et al. (2017) applied a dose response model to establish the duration of suspended sediment concentration perturbation during sediment release operations in a river in the central Italian Alps. The model was based on the severity of effects on fish. Additional operational factors may have a direct impact on invertebrate responses to gravel mining. Floods typically follow a bell-shaped hydrograph (e.g. Lisle 1989), whereas the sediment response associated with mining activities tend to generate a ramp-shaped pattern (Krishnaswamy et al. 2006; Banas et al. 2008). Within this context, Garcia Molinos and Donohue (2011) reported that the shape of the sedigraph may also affect invertebrate response. Assessing the impact of hydropeaking on invertebrates in the Cinca was out of the scope of this thesis, but is worth future study, especially to understand interactions of flow modification and mining.

River management

Based on the European Water Framework Directive, the EU member states have to achieve 'good ecological status' in all their water bodies. Nowadays, Spain has some deficiencies in the implementation of monitoring programmes and in the definition of the specific environmental objectives (European Commission, 2015). In the last decades, many authors highlighted that river management programs may move from a reference-based to an objective-based strategy (Dufour and Piegay, 2009). Natural processes follow complex cycles and long-term trends, making it impossible to recover the former river states. Therefore, the goal is no longer to reach a fixed pattern based on previous state, but to achieve a combination of processes (e.g. flood regime, sediment entrainment or ecological succession) that are by definition highly variable, with partially unpredictable effects (Hughes et al., 2005; Thoms, 2006). For instance results in this thesis highlighted how suspended sediment realised during mining operations affect invertabrates. Some operational strategies (i.e. fine sediment retention in small dam downstream mined area as reported in ACA, 2007) or planning strategies (i.e. minig activities performed under relatively high flows to increase flow competence) would reduce the impact of suspended sediment on benthic invertebrates. Results also pointed out how physical habitat influence benthic invertebrates. As discussed above, in most of occasions channels become deeper and narrow after mining activities, transforming complex channel forms and hydraulics into single-thread channels displaying more homogenous hydraulics. These impacts reduce the number of physical habitats that ultimately have a direct effect on the ecological diversity. Therefore, special attention is needed after the extraction of the gravels to guaranty habitat heterogeneity and surface complexity to support threatened species (e.g. maintain secondary channels, artificial structures to create refuge and flow separation and convergence). Additionally, habitats are stage dependent since the spatial distribution and magnitude of water depth and velocity changes with the flow. Then, part of the active channels that are not wet during low flow conditions may become important (in terms of habitat) in relatively high flows when these areas are flooded. For instance, the head of gravel bars preserve river morphology during high flows. In order to reduce effects of mining on river morphology, it is recommended not to extract gravels from the head of gravel bars (ACA, 2007). This operational strategy will reduce effects of mining activities on physical habitat and benthic invertebrates. Within this context, Figure 2 shows a conceptual diagram integrating the main findings of this thesis to design human interventions in river channels to minimise impacts on river's functionality.

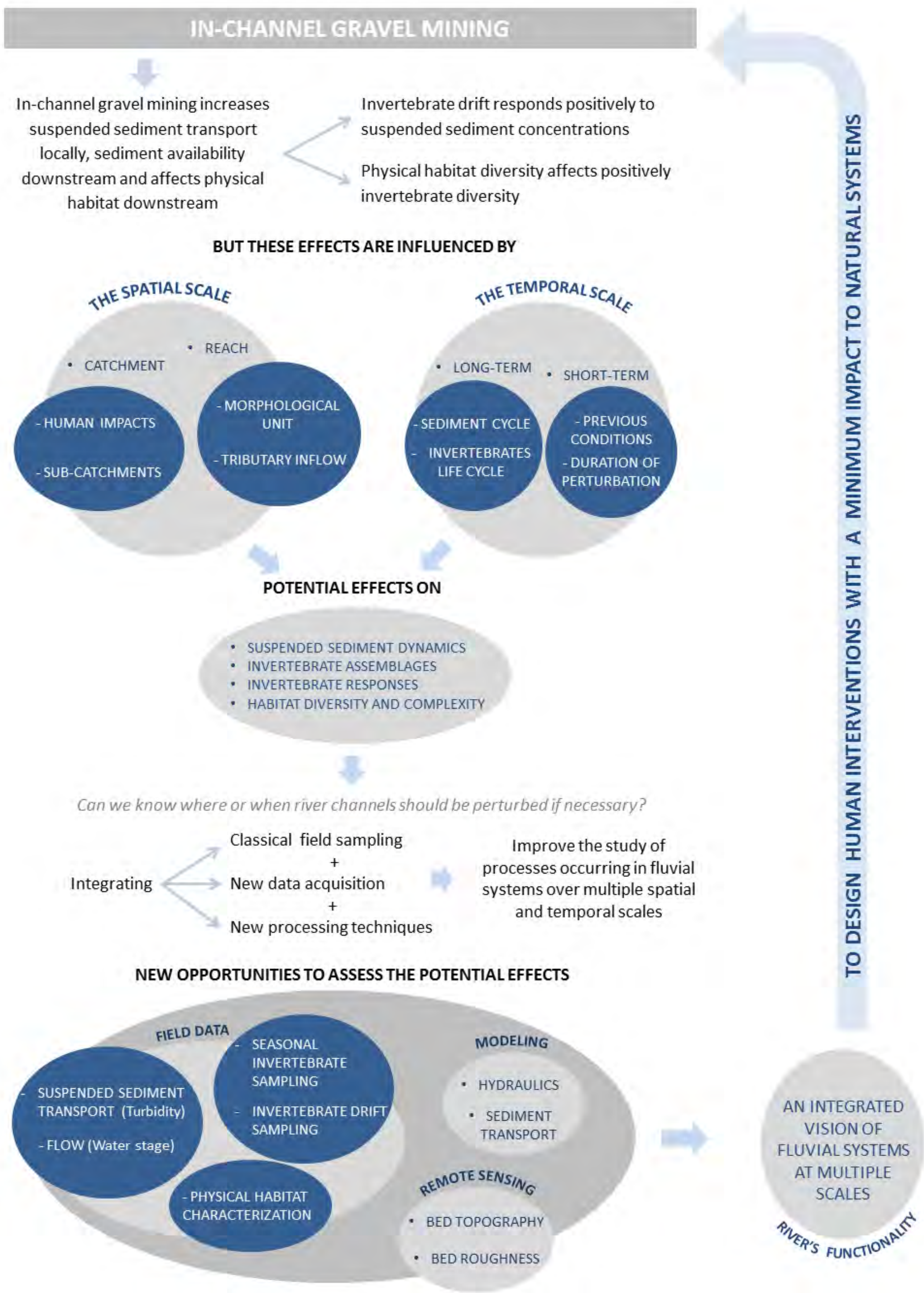

Figure 2. Conceptual scheme integrating the main findings of this thesis to aid designing human interventions in rivers with the minimum impact on river's functionality.

The challenge for scientist and managers is not only to understand natural processes in a context of altered environments, but to develop human interventions in harmony with natural systems (Rice et al., 2010). Natural processes and human interventions work at different scales, generating an extra challenge for scientist and managers. For instance, large temporal scales bring the benefits of understanding processes and changes over a range of timescales (e.g. flow anomalies resulting from climate change in Lamoroux, 2008; and the effects of land cover changes on water and sediment transfer in Buendia et al., 2016). But, as with this thesis, we now know that short temporal and spatial scales (i.e. river sub-reach, short term human activities) may be critical for the diagnosis of fluvial systems. Identification of the key spatial and temporal scales and their interaction throughout a river system is therefore essential for effective management. Within this context, new methods and technologies also have to play a key role in river management. For instance, new sampling and survey techniques (e.g. remote sensing surveys, acoustic Doppler current profilers) provide continuous and high density measurements that improve processes characterization of the river system over a range of spatial and temporal scales (i.e. Woodget et al., 2016). This integrated vision of the river system at multiple scales will enhance our knowledge of its functioning (e.g. hydrological, morphological and ecological processes) and be critical in a context of global change and, particularly, in relation water scarcity and changing river hydro-sedimentary regimes.

2. CONCLUSIONS

This thesis analysed the interactions amongst suspended sediment transport, physical habitat diversity and benthic communities in a mountainous river affected by natural and anthropogenic stressors. The main findings of the thesis corroborate fully the three initial hypothesis and partially the forth hypothesis listed in Chapter 1. The main conclusions can be drawn as follows:

- **C1**. Human-based impacts (gravel mining and hydropeaking) and natural dynamics (floods and inflow from tributaries) significantly alter water yield and suspended sediment transport (in accordance with Hypothesis 1 and Objective 1). The specific spatial variability of these factors exert an additive or compensatory effect on runoff and sediment load in relatively short channel distances, highlighting out the need for accurate specification of river sub-reaches.

- **C2**. Gravel mining increases suspended sediment transport locally. Sediment transport is highly controlled by grain-size and particles settle quickly (in accordance with Hypothesis 2 and Objective 2). Fine sediment deposited in the channel increases sediment availability downstream the perturbed river reach, generating off-site effects and deferred impacts downstream.

- **C3.** Invertebrate drift responds positively to suspended sediment concentrations without the confounding effects of hydraulic changes and bed-material entrainment associated with floods. The duration in which concentrations are high and the size of the transported particles also influence invertebrate drift. Responses vary between different taxonomic groups (in accordance with Hypothesis 3 and Objective 3).

- **C4**. Integrating new sampling techniques, technologies and methods allows extracting spatially distributed variables that improve physical habitat characterization. Physical habitat diversity affects invertebrate diversity positively; bed surface complexity also affects macroinvertebrates although to a lesser degree than habitat heterogeneity (partially in accordance with Hypothesis 4 and Objective 4). Contrary to our hypothesis 4, macroinvertebrates sampled from a small defined area (the 0.25 m^2 area, as per Chapter 6) are not significantly influenced by physical properties of areas outwith this (i.e. over spatial scales larger than $1m^2$), indicating that macroinvertebrates mainly respond to their immediate (or local) habitat.

3. LIMITATIONS OF THE THESIS

The limitations of this thesis need to be taken into account in order to be circumspect about the conclusions and better frame future works. Although specific discussions in terms of data accuracy and precision are presented in each chapter, the section below summarises the limitations based on two main elements regarding to the monitoring design and data acquisition.

Monitoring

Water and suspended sediment dynamics were analysed over two hydrological years. Due to the inter-annual variability of flow discharge, findings may not be directly extrapolated to other years or used to generate long-term predictions. Hydrologically, the two years are considered dry compared to the long-term average data (20 % less than the average discharge). Additionally, as discussed above, human-based perturbations in the reach add extra variability (i.e. sediment transport may be influenced by on the frequency of mining activities and the frequency of hydropeaking may also influence water yield). Thus, monitoring periods should be extended to the longest period possible. The experimental design during in-channel gravel mining does not include continuous sediment transport measures and does not quantify the amount of sediment transferred just downstream from the mined reach. Therefore, overall continuous records of sediment transport, but specifically some meters downstream from the mining activity, would improve the analysis of sediment dynamics in future work. In the present case the exact location of the section to be mined was not known in advance, so it was not possible to collect pre-disturbance data for this section. Such knowledge would be extremely valuable to help design future studies, and particularly to help test hypotheses about the causal processes at work.

Data acquisition

Suspended sediment transport was estimated based on turbidity records and water samples. Water samples obtained for a high range of concentrations were used to develop the calibration of the turbidity probes in each section. Soler et al. (2012) demonstrated different turbidity responses associated to the particle size of suspended sediments. Many mountain rivers have an annual sediment cycle where sediment sources alternate over the year (hence different lithology and particle sizes may be also expected). Within this context, a seasonal calibration may improve the calibrations of the turbidity probes. Figure 3 shows an example of the turbidity calibration and the trends for different floods registered in the Laspuña section. Additionally, automatic water samples were restricted to flood events where fines were well mixed in the water column and single point measurements represent well the load transported in the cross-section. The use of an integrating water sampler (and different verticals samples distributed along the cross-sections) may solve the non-well mixing of fines problem under low flows conditions. Finally, a fixed percentage value of organic content was subtracted from all the load estimates due to its low spatially and temporally variability. Nevertheless, the specific study of the organic content over the year may also be an aspect to consider in future studies.

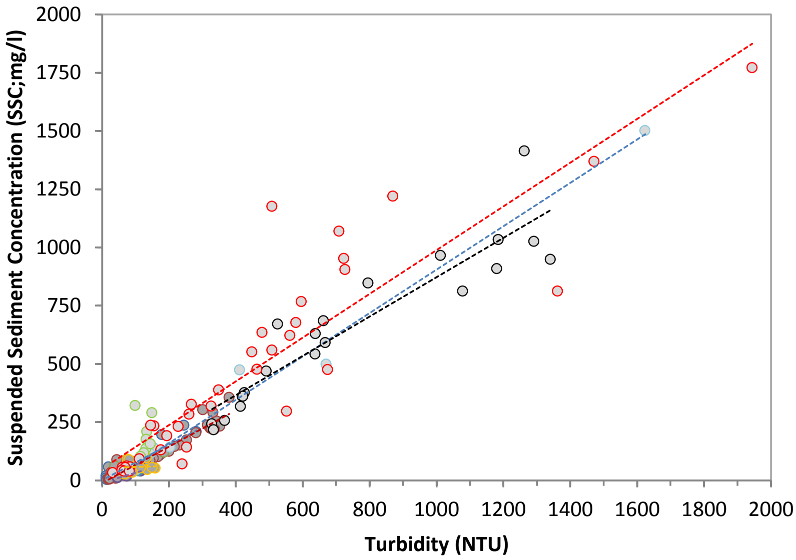

Figure 3. Rating curves between turbidity (NTU) and suspended sediment concentration (SSC) at the Laspuña monitoring section over the study period (2014-2016). Rating curves and dots are coloured to show different flood events.

Water samples were filtered to determine suspended sediment concentration. Some samples were re-suspended in order to be post-processed to obtain the grain size of the sediments, specifically for the analyses in chapter 4. In this way, part of the finer fraction may be retained in the filter while re-suspension took place. Separate an aliquot of the initial water sample may be a solution for further works.

Discharge was obtained from water depth records by means of water stage probes. Values of water depth were converted to discharge by applying site specific rating curves (i.e. relation between depth and discharge) developed from direct discharge measurements and hydraulic modelling. The method used to obtain discharge from water depth measurements is summarised in figure A.1 (Annex A). Water stage probes have a relatively good accuracy. However, the main error arises from the complexity of bed roughness, channel topography and flow hydraulics. The temporal variability of these key variables needs to be taken into account. Any change in topography, roughness or velocity in a given section for a given water depth will modify the corresponding rating curve and needs to be taken into account.

Invertebrates were sampled using nets with a 1000 µm mesh (larger than the standard 300 µm mesh). The selected pore size is the result of a compromise between net clogging and animal size. However, some animals may be lost during sampling. Additionally, surber method impedes sampling in deep areas (> 60 cm depth) and, consequently, benthic assemblages in such areas were not studied. These factors would need to be considered in future studies.

Acoustic Doppler Current Profiler (ADCP) systems offer lots opportunities to characterize physical habitat heterogeneity and surface complexity. Nonetheless, reliable velocity data for depths around 10-20 cm is limited, impeding the characterisation of physical habitats in shallow areas. Additionally, physical variables extracted from ADCP surveys have different densities and interpolation was performed in order to obtain continuous information for each variable. Interpolation simplifies variable distribution. Acquiring the same density of observations should be considered in order to obtain a better representation of physical habitat conditions in the river and experienced by invertebrates.

Finally, uncertainties were detected in data from official data sources. It is worth to mention that, as far as we know, discharge at the Escalona gauging station (A051) is assessed indirectly. Mainly, if changes on water stored in the Mediano Reservoir and the discharge of the River Ara are known, and assuming that the main contribution to Mediano is from the Cinca and the Ara together, the flow in the Cinca can be assessed as the difference between the change in volume in Mediano and the inflow from the Ara. Although this official discharge is in agreement with the estimates for medium to high discharges used in this thesis, official discharges tend to overestimate runoff for low flows. Periodic flow gauging may help in evaluating data quality and improve water resource estimations, particularly in mountain areas considered the most important zones for water generation in Mediterranean regions.

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ANNEX A

SUPPLEMENTARY MATERIALS

Annex A presents a figure summarizing the methodology designed and used to obtain flow (Q) from water depth measurements (d) in the three monitoring sections of the upper Cinca. Although the assessment of the d/Q relationships was not fully developed in this PhD, flow data are used for several of the objectives of the thesis. Flow data were obtained by means of six interrelated tasks: (I) acquiring water depth data, (II) Transforming water depth to water surface elevation (WSE), (III) flow gauging (Q), (IV) hydraulic modelling, (V) model performance assessment and (VI) data fusion, elaboration of WSE-Q relationships and discharge transformation.

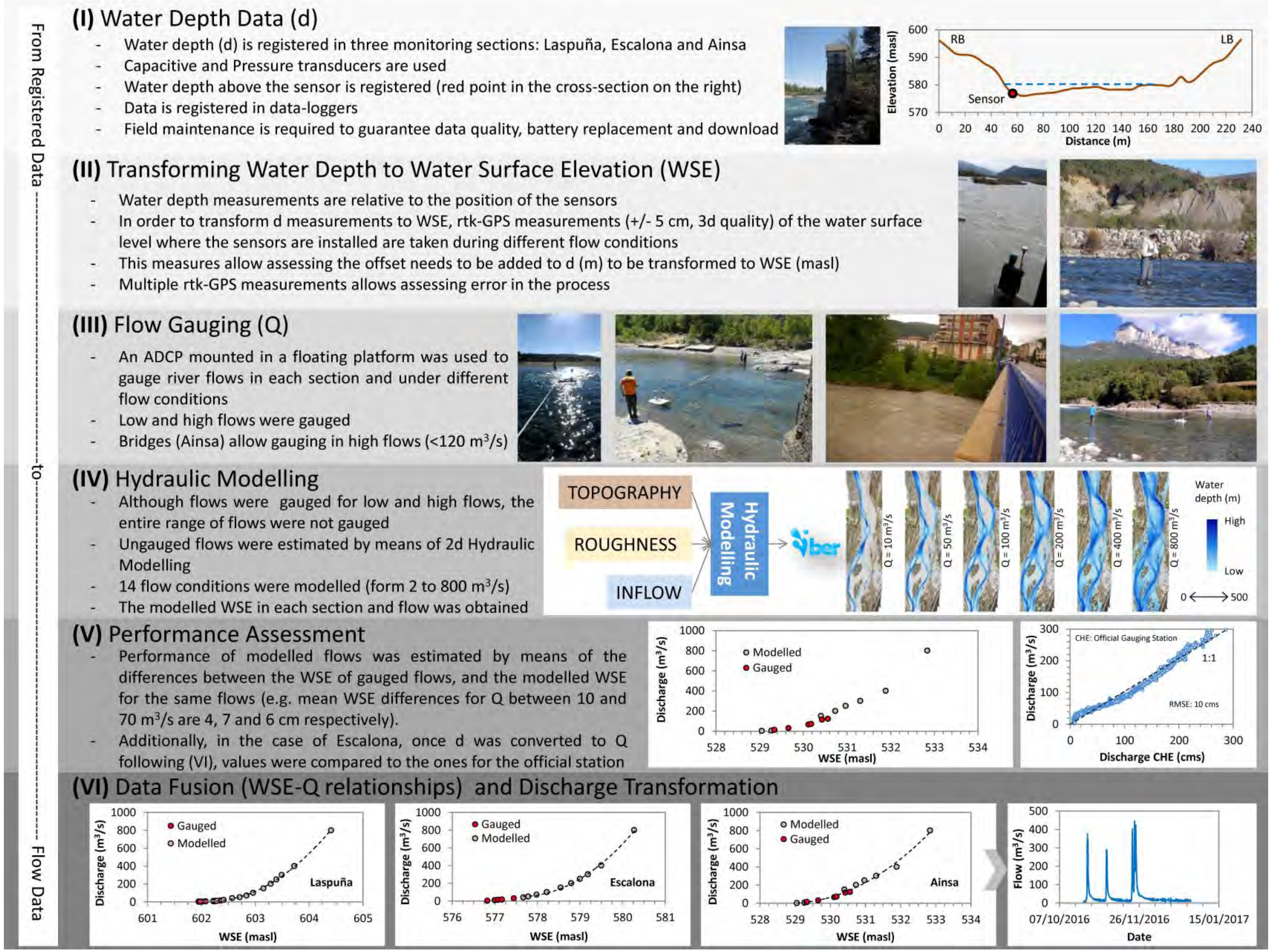


Figure A.1. Method used to obtain flow (Q) from water depth measurements (d) in the upper Cinca showing the six interrelated task.

ANNEX B

MONITORING SECTIONS AND SAMPLING SECTIONS

Annex B contains specific details of the three monitoring stations set up in this Thesis in the background of the research project MorphSed (see location in Figure 11 A, Chapter 1). Additionally, specific details of the sampling sections are provided in a summary sheet per station or section. It is worth to mention that, initially a total of three monitoring sections and seven sampling sections were selected. Finally, only data from the three monitoring sections and from three sampling sections (i.e. Laspuña, Escalona, Ainsa) were used in this Thesis. Even so, the summary sheet of the remind four section not used is provided to provide information for future projects if necessary.

B.1. MONITORING SECTIONS

SECCIÓN DE MONITOREO: LASPUÑA	
Coordenadas:	265692 m; 4709852 m (ETRS89-UTM31N)
Fecha instalación:	16/01/2014
Variables:	Turbidez, nivel de agua, temperatura, imágenes de la sección
Frecuencia:	Quinceminutales
Distribución del tamaño de partículas:	Sensor de turbidez modelo NEP-9530; Fabricante: Mc Vann Analite Datalogger CR800; Fabricante: Campbell Sensor de presión modelo WT-HR 1500; Fabricante: Trutrack Cámara Timelapse TLC200; Fabricante: Brinno
Alimentación:	Sistema de sensores : Bateria 12V Cámara timelapse: Pilas 1,5V AA

Fotografías:

Estación de monitoreo en las proximidades de Laspuña

Sensores de nivel y turbidez en sus respectivos sistemas de protección.

Sistema de almacenamiento de datos

Cámara Timelapse tomando imágenes periódicas de la sección monitorizada

SECCIÓN DE MONITOREO: ESCALONA	
Coordenadas:	265749 m; 4706734 m (ETRS89-UTM31N) (instalaciones de Acciona)
Fecha instalación:	14/02/2014
Variables:	Turbidez, nivel de agua, temperatura, imágenes de la sección
Frecuencia:	Quinceminutales
Distribución del tamaño de partículas:	Sensor de turbidez modelo OBS 300 (4000NTU); Fabricante: Campbell Sensor de nivel modelo CS451; Fabricante: Campbell Datalogger CR200X; Fabricante: Campbell Cámara Timelapse TLC200; Fabricante: Brinno
Alimentación:	Sistema de sensores : Bateria coche 12V Cámara timelapse: Pilas 1,5V AA
Fotografías:	

Estación de monitoreo en las proximidades de la estación A051 del SAIH-CHE

Sensores de nivel y turbidez anclados a la escala de referencia en la estación A051 del SAIH-CHE

Interior de la estación, sistema de almacenamiento de datos y alimentación.

Detalle del interior del tubo de protección de los sensores de turbidez, nivel y temperatura.

SECCIÓN DE MONITOREO: AINSA

Coordenadas:	264688 m; 4700609 m (ETRS89-UTM31N) (instalaciones de Acciona)
Fecha instalación:	06/03/2014
Variables:	Turbidez, nivel de agua, temperatura, imágenes de la sección
Frecuencia:	Quinceminutales
Distribución del tamaño de partículas:	Sensor de turbidez modelo OBS 300 (4000NTU); Fabricante: Campbell Sensor de nivel modelo CS451; Fabricante: Campbell Muestreador automático ISCO modelo 3700; Fabricante: Teledyne ISCO Actuador de nivel modelo 1640; Fabricante: Teledyne ISCO Modem GSM/GPRS; Fabricante: Campbell Datalogger CR200X; Fabricante: Campbell Cámara Timelapse TLC200; Fabricante: Brinno
Alimentación:	Sistema de sensores y telemetría: Placa solar + Regulador solar Blue solar 12/24V 10 Ah+ Bateria coche 12V Muestreador automático: Bateria 12V Cámara timelapse: Pilas 1,5V AA

Fotografías:

Estación de monitoreo en las proximidades de Ainsa y rio Cinca.

Interior de la estación y sistema de almacenamiento de datos .

Cámara timelapse tomando fotografías periódicas de la sección.

Sensor de turbidez y nivel de agua. Detalle subacuático.

B.2. MAIN SAMPLING SECTIONS

TRAMO DE ESTUDIO: LASPUÑA

Coordenadas: 265691 m; 4709670 m (ETRS89-UTM31N)

Fecha muestreos: 14/10/2013; 20/03/2014; 02/10/2014; 15/12/2014; 10/03/2015; 13/07/2015

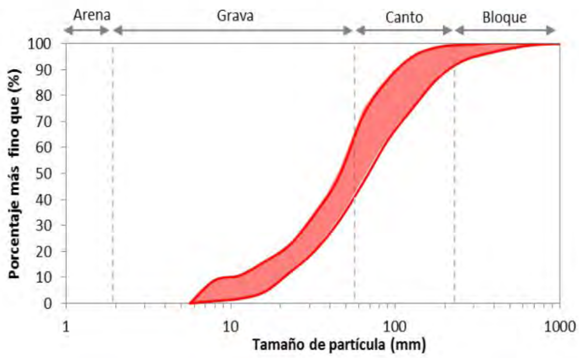
Variables: Nitratos, comunidades bentónicas, velocidad del agua, profundidad del agua, distribución granulométrica de las partículas superficiales del lecho

Frecuencia: Estacionales

Distribución del tamaño de partículas: Se presenta la distribución granulométrica obtenida durante las distintas campañas de obtención de datos desde 2013-2016 en el marco del Proyecto MorphSed

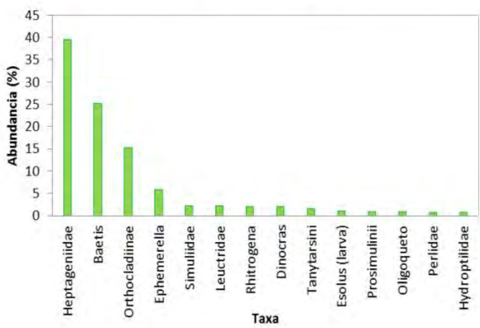
D_{50} = 53.7

D_{95} = 196.4



Unidades morfológicas muestreadas:

Comunidades bentónicas: Se presenta la distribución de taxones obtenidos en la campaña de campo de Julio 2015



Fotografías del área de estudio:

TRAMO DE ESTUDIO: ESCALONA

Coordenadas: 265928 m; 4706898 m (ETRS89-UTM31N)

Fecha muestreos: 14/10/2013; 20/03/2014; 02/10/2014; 15/12/2014; 10/03/2015; 13/07/2015

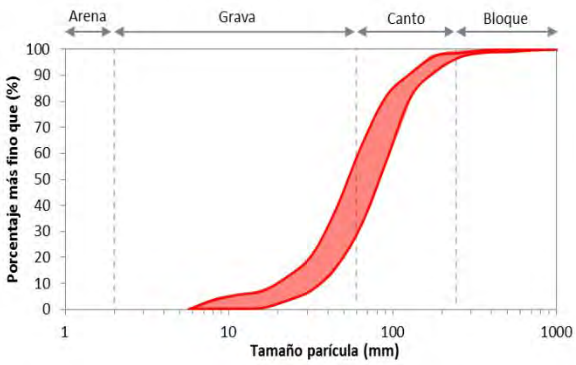
Variables: Nitratos, comunidades bentónicas, velocidad del agua, profundidad del agua, distribución granulométrica de las partículas superficiales del lecho

Frecuencia: Estacionales

Distribución del tamaño de partículas: Se presenta la distribución granulométrica obtenida durante las distintas campañas de obtención de datos desde 2013-2016 en el marco del Proyecto MorphSed

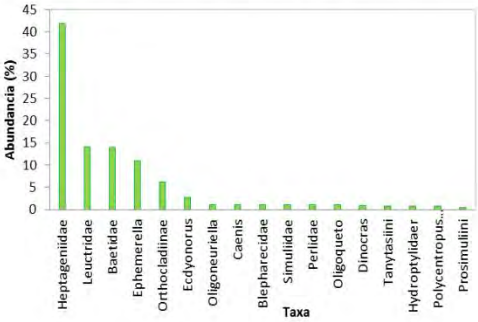
D_{50} = 63.74

D_{95} = 191.2



Unidades morfológicas muestreadas:

Comunidades bentónicas: Se presenta la distribución de taxones obtenidos en la campaña de campo de Julio 2015



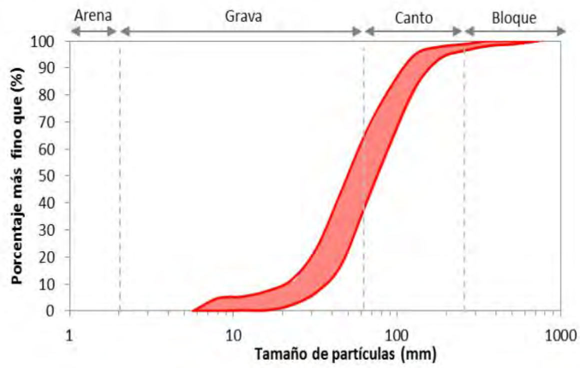
Fotografías del área de estudio:

TRAMO DE ESTUDIO: AINSA

Coordenadas:	264691 m; 4700624 m (ETRS89-UTM31N)
Fecha muestreos:	14/10/2013; 20/03/2014; 02/10/2014; 15/12/2014; 10/03/2015; 13/07/2015
Variables:	Nitratos, comunidades bentónicas, velocidad del agua, profundidad del agua, distribución granulométrica de las partículas superficiales del lecho
Frecuencia:	Estacionales
Distribución del tamaño de partículas:	Se presenta la distribución granulométrica obtenida durante las distintas campañas de obtención de datos desde 2013-2016 en el marco del Proyecto MorphSed

D_{50} = 60.9

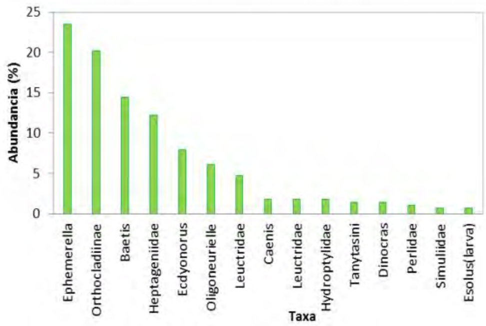
D_{95} = 168.9



Unidades morfológicas muestreadas:

Comunidades bentónicas

Se presenta la distribución de taxones obtenidos en la campaña de campo de Julio 2015



Fotografías del área de estudio:

B.3. ADDITIONAL SAMPLING SECTIONS

TRAMO ADICIONAL DE ESTUDIO: REFERENCIA

Coordenadas: 265404 m; 470531 m (ETRS89-UTM31N)

Fecha muestreos: 21/03/2014; 02/10/2014; 15/12/2014; 10/03/2015; 13/07/2015

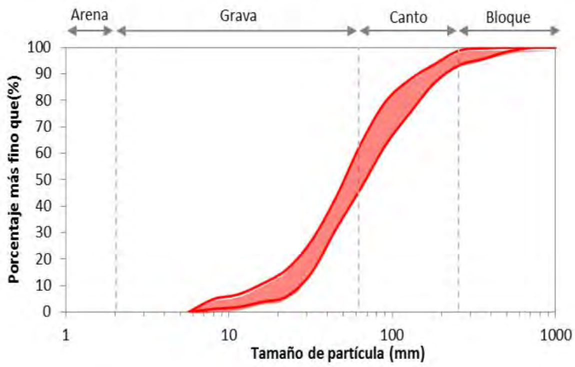
Variables: Nitratos, comunidades bentónicas, velocidad del agua, profundidad del agua, distribución granulométrica de las partículas superficiales del lecho

Frecuencia: Estacionales

Distribución del tamaño de partículas: Se presenta la distribución granulométrica obtenida durante las distintas campañas de obtención de datos desde 2013-2016 en el marco del Proyecto MorphSed

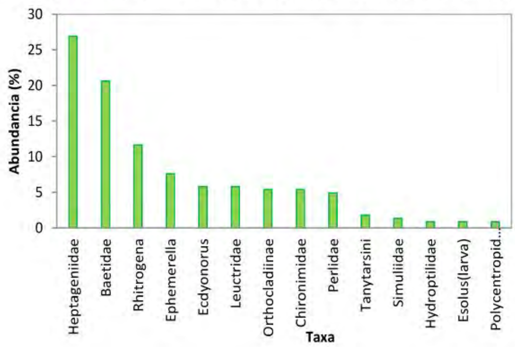
D_{50} = 58.2

D_{95} = 242.4

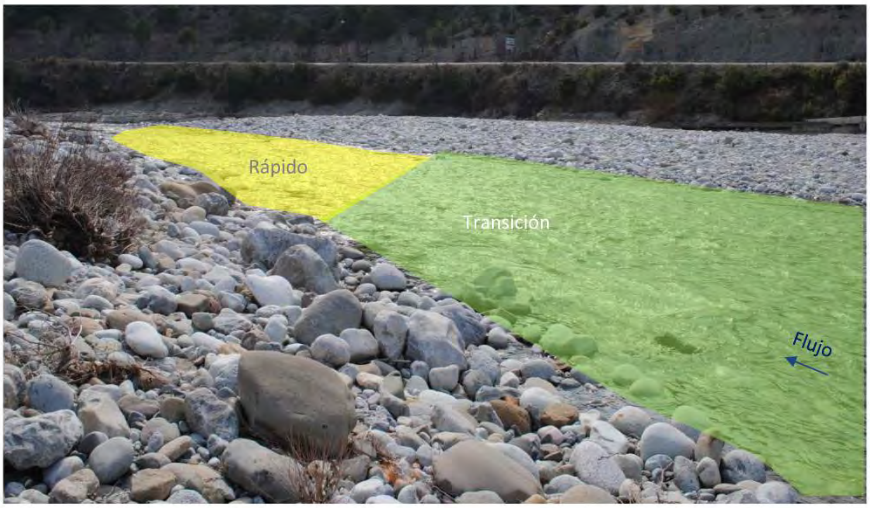


Unidades morfológicas muestreadas:

Comunidades bentónicas: Se presenta la distribución de taxones obtenidos en la campaña de campo de Julio 2015



Fotografías del área de estudio:



TRAMO ADICIONAL DE ESTUDIO: LASPUÑA 2

Coordenadas: 265586 m; 4709618 m (ETRS89-UTM31N)

Fecha muestreos: 20/03/2014; 02/10/2014; 15/12/2014; 10/03/2015; 13/07/2015

Variables: Nitratos, comunidades bentónicas, velocidad del agua, profundidad del agua, distribución granulométrica de las partículas superficiales del lecho

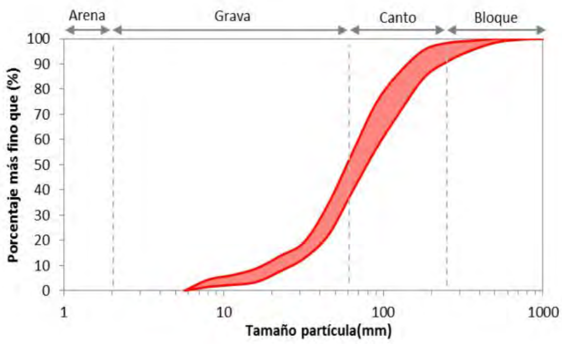
Frecuencia: Estacionales

Distribución del tamaño de partículas:

Se presenta la distribución granulométrica obtenida durante las distintas campañas de obtención de datos desde 2013-2016 en el marco del Proyecto MorphSed

D_{50} = 72.8

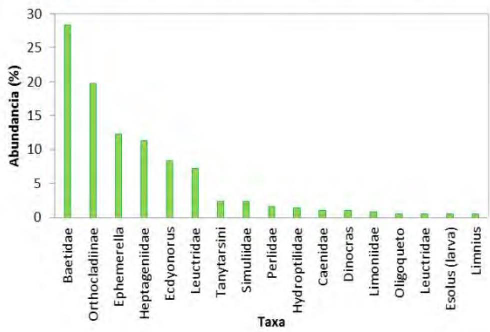
D_{95} = 253.1



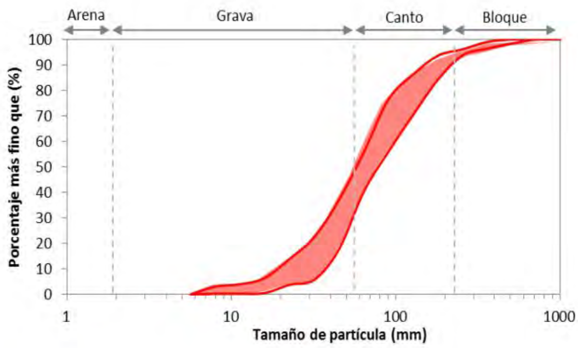
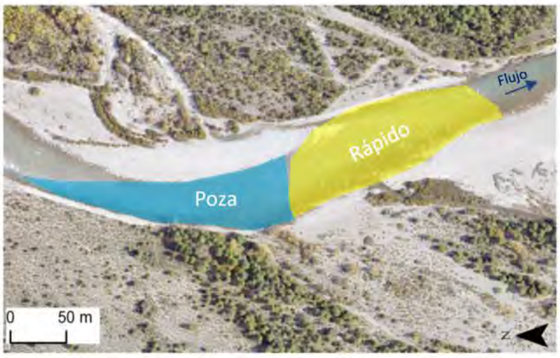
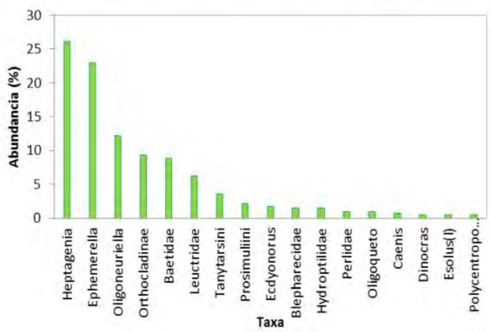
Unidades morfológicas muestreadas:

Comunidades bentónicas

Se presenta la distribución de taxones obtenidos en la campaña de campo de Julio 2015



Fotografías del área de estudio:

TRAMO ADICIONAL DE ESTUDIO: GRANJA LISA	
Coordenadas:	264992 m; 4705199 m (ETRS89-UTM31N)
Fecha muestreos:	14/10/2013; 20/03/2014; 30/07/14; 05/08/14; 02/10/2014; 15/12/2014; 10/03/2015; 13/07/2015
Variables:	Nitratos, comunidades bentónicas, velocidad del agua, profundidad del agua, distribución granulométrica de las partículas superficiales del lecho
Frecuencia:	Estacionales
Distribución del tamaño de partículas:	Se presenta la distribución granulométrica obtenida durante las distintas campañas de obtención de datos desde 2013-2016 en el marco del Proyecto MorphSed D_{50} = 67.6 D_{95} = 261.4 
Unidades morfológicas muestreadas:	
Comunidades bentónicas	Se presenta la distribución de taxones obtenidos en la campaña de campo de Julio 2015 

Fotografías del área de estudio:

TRAMO ADICIONAL DE ESTUDIO: CAMPING

Coordenadas: 264848 m; 4702576 m (ETRS89-UTM31N)

Fecha muestreos: 14/10/2013; 20/03/2014; 30/07/14; 05/08/14; 02/10/2014; 15/12/2014; 10/03/2015; 13/07/2015

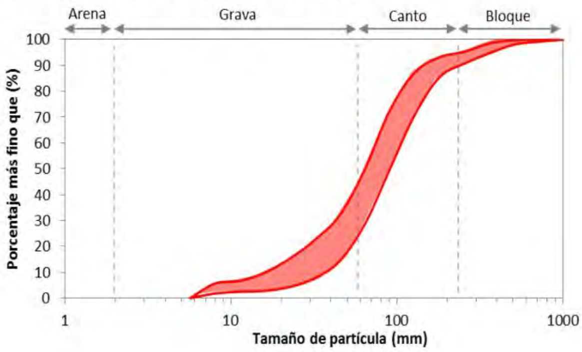
Variables: Nitratos, comunidades bentónicas, velocidad del agua, profundidad del agua, distribución granulométrica de las partículas superficiales del lecho

Frecuencia: Estacionales. Nota: Este tramo ha sido altamente modificado debido a las obras realizadas en el cauce referentes a la extracción de áridos estudiada en el Capítulo 4 y Capítulo 5 de la tesis.

Distribución del tamaño de partículas: Se presenta la distribución granulométrica obtenida durante las distintas campañas de obtención de datos desde 2013-2016 en el marco del Proyecto MorphSed

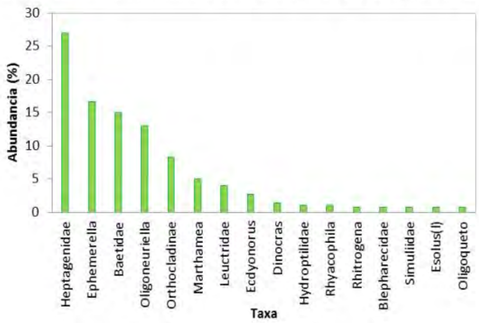
D_{50} = 75.0

D_{95} = 305.8

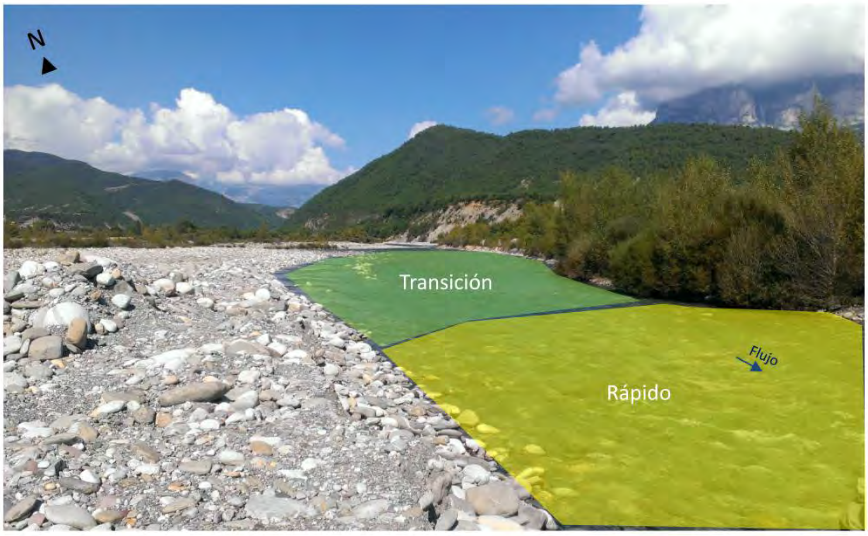


Unidades morfológicas muestreadas:

Comunidades bentónicas: Se presenta la distribución de taxones obtenidos en la campaña de campo de Julio 2015



Fotografías del área de estudio:



ANNEX C

OTHER PUBLICATIONS IN THE BACKGROUND OF THE THESIS

Annex C contains two conference proceeding papers (peer reviewed) prepared for the XIII Spanish Society for Geomorphology Meeting (Universidad de Extremadura, Cáceres, September 2014) and the XI International Symposium on Ecohydraulics (Melbourne, Australia, 7-12 February 2016). The papers provide complemnetary results obtained in the background of the thesis.

INTEGRATED METHODS TO STUDY THE IMPACT OF GRAVEL MINING ON BENTHIC INVERTEBRATE COMMUNITIES IN A HIGHLY DYNAMIC GRAVEL-BED RIVER

Integración de métodos para el estudio del impacto de las extracciones de áridos en las comunidades de macroinvertebrados en un río de gravas altamente dinámico.

This annex contains the following already published chapter in the conference proceeding Avances de la Geomorfología en España 2012-2014

Béjar, M.; Gibbins, C.N.; Vericat, D.; Batalla, R.J.; Buendía, C.; Lobera, G. (2014): Integrated methods to study the impact of gravel mining on benthic invertebrate communities in a highly dynamic gravel-bed river. In: Schnabel, S.; Gómez-Gutierrez, A. (eds): Avances de la Geomorfología en España 2012-2014. XIII Reunión Nacional de Geomorfología. ISBN: 978-84-617-1123-9, Universidad de Extremadura, Cáceres, 20-23.

ABSTRACT: This paper describes the integration of a set of methods implemented to assess the impacts of natural disturbance (i.e. flood) and human induced disturbance (i.e. gravel abstraction) on benthic invertebrate communities. Research is located in the upper River Cinca, a highly dynamic gravel-bed river that has experienced gravel mining activities since the middle of the last century. An experimental design integrated by four components is being implemented: (C1) topographic models are obtained by means of combining automated digital photogrammetry (SfM) and optical bathymetric models at the reach scale, and by acoustic Doppler current profiler (hereafter aDcp) local surveys at the local scale. Local characteristics of bed structure and hydraulics are also obtained by means of aDcp surveys; (C2) benthic invertebrates surber sampling are obtained to characterize the assemblages (composition, density, distribution, traits); (C3) flow and sediment transport are monitored to characterize competent events and understand how material is transferred through the study reach; (C4) statistical treatment of the data set to elaborate physical and ecological metrics. A total of two field campaigns have been completed. Our preliminary results highlight the importance of coupling geomorphological and ecological information to acquire a better knowledge of their interactions and to support management practices in fluvial systems.

KEY WORDS: disturbance, gravel mining, macroinvertebrates, geomorphology, metrics.

1. RATIONALE

The structure of alluvial river channels is formed of sediments that have experienced cycles of entrainment, transport and deposition (Church, 2006). Channel geometry and morphology result from the interaction between flow regime and hydraulics, bed sedimentology, and sediment availability, supply and transport (Leopold et al., 1964). Such interactions control the thresholds of channel deformation, which in turn influence the frequency and magnitude of physical disturbances.

Channel-bed disturbance is considered the main factor affecting the organization of riverine communities and contributes to key ecological processes. Geomorphic disturbances caused by competent events create a new habitat template, to which the species will adapt, eventually reaching a new ecological diversity status. Human-induced alterations such as gravel mining represent an additional, non-natural, stressor which modifies physical and ecological processes and dynamics

The temporal persistence of a particular community is determined by two main factors: (i) the resistance of the ecosystem to disturbance (i.e. channel condition, contemporary flood and sediment transport regimes, including sediment availability and supply) and (ii) their rate of recovery from a given perturbation event (i.e. distribution anddispersal ability of potential colonist; Webster et al., 1983).)

2. AIM AND OBJECTIVES

In this paper we present the hypothesis and the methods designed to analyze the impacts of gravel abstraction on benthic invertebrate communities in a 13 km reach of a highly dynamic gravel-bed river. The study reach is located in the upper River Cinca (Southern Pyrenees) an upland gravel-bed system historically and currently affected by periodical episodes of in-channel gravel mining. These methods have been developed in the background of MorphSed, a multidisciplinary research project funded by the Spanish Ministry. The experimental design of this work is based on two main hypothesis: (**H1**) invertebrates communities are fundamentally affected by gravel mining both due to physical disappearing of animals and due to changes in habitat conditions during and after gravel extraction; and (**H2**) recovery of pre-extractions communities will depend upon the degree of the morphosedimentary impacts and the occurrence of the subsequent competent events. According to these hypothesis, the specific objectives are: (O1) assess the spatial extent (upstream and downstream) of gravel mining impact on the hydraulic and sedimentary habitat of invertebrates; (O2) evaluate the spatial extent (upstream and downstream) of gravel mining impacts on the taxonomic and trait structure of invertebrate communities. This objective will include assessment of the long term (historical data) and immediate effects (own data). (O3) Assess the role of flood events in aiding recovery of invertebrate communities creating suitable habitat after mining activity; and (O4) determine the most suitable metrics for detecting and monitoring the impacts of gravel mining on benthic invertebrates.

3. RESULTS

In order to accomplish the objectives an experimental design composed of a methodology that integrates a total of four components was defined: topography, benthic invertebrates, discharge and sediment transport and statistical treatment (Figure 1). All methods were initially tested in the field. The data acquisition period started once the experimental design was re-defined and finished according field limitations and in relation to the specific objectives. Two complete field campaigns have been finalized. The work still in progress, field work is carried at the flood scale while other variables are continuously recorded. Data is being post-processed.

3.1. Topography and bed structure

Reach-scale topographic data are acquired by automatic aerial photogrammetry from high resolution aerial imagery. Point clouds obtained from this method will provide information of the channel topography and roughness (bed structure) and their evolution through the time and in relation to disturbance events (natural i.e. floods; and human-based i.e. mining). High resolution imagery is obtained by a conventional digital camera. Photos are obtained from an autogyro. These aerials are being post processed by means of Agisoft (Structure from Motion; see more details in Narciso et al. in this volume). At the patch scale (i.e. where macroinvertebrate samples are obtained), aDcp surveying is providing information of the local topography, its complexity (proxy of bed structure) and flow hydraulics. These data will be used to assess the spatial extent and magnitude of channel changes, and the impacts on the physical habitat of invertebrates.

3.2. Benthic invertebrates

Invertebrates are sampled in five reaches along the study reach in order to define assemblages and their characteristics (composition, density, distribution, traits). Reaches were selected according to their morphological characteristics and historical and future mining activities. Surber samples are being collected before and after channel disturbance. A total of 10 samples are obtained in each reach. Samples are weighted according the extension of the main morphological unites in each reach. Data will be used to assess both, temporal and spatial extent of channel disturbance impacts on the taxonomic and trait structure of communities.

3.3 Discharge and sediment transport

Discharge and sediment transport are monitored at three monitoring sections, upstream, middle (coincident with the official gauging station EA051) and downstream ends of the study site.

Flow data is registered continuously by means of water stage transducers (Druck 1730 – PDCR) and subsequently transformed to discharge by at-a-site rating curve. Discharge data is already available at EA051 but, .additional to these, data flow is being gauged at different flow conditions by means of a RiverSurveyor M9 boat mounted Doppler system. These gauges are used to build at-a-site rating curves.

Sediment transport is indirectly monitored by turbidity probes installed at the three monitoring sections (Analite NEP9350 and Endress+Hausser Turbimax W CUS41). Water samples are directly obtained at multiple flow conditions to calibrate the turbidity readings. These data will provide information regarding to the magnitude of competent events and the amount of transferred material through the study reach.

3.4 Statistical data treatment

Finally, a statistical treatment of the data sets will be applied in order to determine the most suitable metrics for detecting and monitoring the impacts of gravel mining. This data provides information about impacts and the recovery time: (i) on benthic invertebrates structure and (ii) on hydraulic and sedimentary habitat.

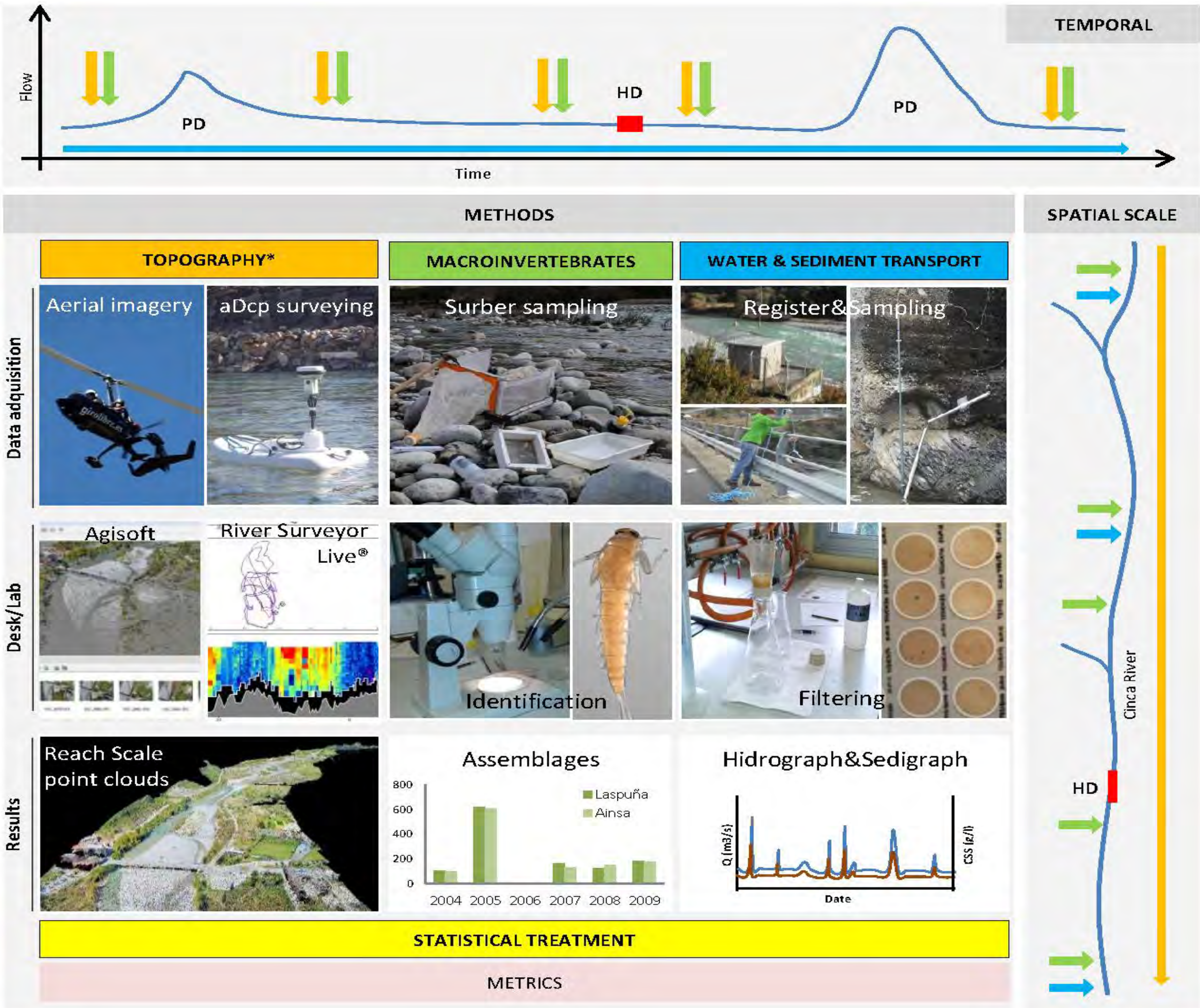


*Figure 1. The four methodological components implemented to analyze the impacts of gravel abstraction on benthic invertebrate communities in the upper River Cinca. Temporal and spatial scale are added to show where and when methods will be applied. Note that HD means Human induced disturbance; PD means Physical Disturbance. * See Naricso et al. in this volume for more details. Note that topography at the local scale is obtained by aDcp and also includes characteristics of bed structure and flow hydraulics.*

4. PRELIMINARY RESULTS

Two full field campaigns were carried out on October 2013 and March 2014. Here we present the preliminary results of the benthic invertebrate sampling that was carried out in October. A total of 50 invertebrate samples were obtained distributed in 5 different sections along the 9.8 km study reach. Water depths varied between 10 and 60 cm, Mean flow velocity at the sampling sites was 0.51 m/s, while median surface bed materials ranged between 50 and 76mm. The proportion of fine materials (materials finer than 8 mm) varied between 1 and 7 %. Invertebrate data shows the preliminary assemblage characterization. The most abundant taxon was Heptagenia (41-76 % of the total samples), following by Baetis (1-25 %) and Chironomidae (2-16 %) (Figure 2). Ephemeroptera, Plecoptera and Tricoptera order (EPT) dominates the assemblages range 58-70 %. The number of families is variable as shows Figure 2. Although these results are preliminary and based in a single field campaign, they show as, potentially, differences in number of families and assemblages could be attributed to the morph-sedimentary characteristics of the sites and, those, being directly influenced by historical gravel mining and all works conducted in the river channel (e.g. reduction of the active width by rip rap).

Data post-processing is in progress. These preliminary results provide a general idea about benefits to link geomorphological (contemporary and historical) and ecological processes. The methodology applied in this study and the conditions of the study site co (highly dynamic bed river) provides a unique opportunity to link morphodynamics, floods, impacts, ecological diversity (based on macroinvertebrates) and recovery. Results will be of a great interest for Water Managers in the context of the new regulations of the WFD.

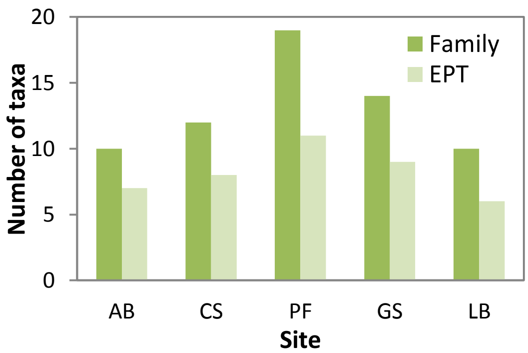


Figure 2. Preliminary assemblage characterization in 5 reaches along the study reach. Number of families and those that belongs to EPT orders (Ephemeroptera, Plecoptera and Trichoptera) are showed in the graph.

Acknowledgements

This research is carrying out within the framework of a research project funded by the Spanish Ministry of Economy and Competiveness (CGL2012-36394). The first author has a PhD grant funded by the University of Lleida. The third author is founded by a Ramon y Cajal Fellowship (RYC-2010-06264). Hydrological data were supplied by the Ebro Water Authorities. We thank all support provided by the Ebro Water Authorities, the interest of ENDESA in terms of the application of the methodology in other fluvial systems, and all logistics provided by Acciona.

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EFFECTS OF SUSPENDED SEDIMENT TRANSPORT ON INVERTEBRATE DRIFT IN A PIEDMONT RIVER CHANNEL: THE UPPER RIVER CINCA (SOUTH CENTRAL PYRENEES)

This annex contains the following already published chapter in the conference proceeding Proceedings of the 11th International Symposium on Ecohydraulics

Béjar M.; Gibbins C.; Vericat D.; Batalla R.J. (2016) Effects of suspended sediment transport on invertebrate drift in a piedmont river channel: The upper River Cinca (South Central Pyrenees) Paper 25943 in, Webb JA, Costelloe JF, Casas-Mulet R, Lyon JP, Stewardson MJ (eds.) Proceedings of the 11th International Symposium on Ecohydraulics. Melbourne, Australia, 7-12 February 2016. The University of Melbourne, ISBN: 978 0 7340 5339 8.

ABSTRACT: River channel disturbance is considered a key factor affecting benthic invertebrate community structure [1]. During disturbances, losses from the benthos occur as a result of involuntary drift. This involuntary drift, along with so-called voluntary or behavioural drift, plays an important role in the dispersal and population dynamics of invertebrates in rivers and streams worldwide [2]. Understanding how channel disturbance affects drift is therefore of great scientific and applied interest. In-channel gravel mining causes major bed disturbance, with consequences for channel morphology and sediment transport dynamics at different temporal and spatial scales. One notable point about the disturbance caused by gravel mining is that, unlike that associated with natural floods, high SSCs occur in the absence of marked flow changes. Thus, in-channel gravel mining provides an opportunity to separate the effects of suspended sediment concentrations (hereafter SSC) and flow change on invertebrate drift. Within this context, research described in this paper assessed changes in SSC and invertebrate drift associated with in-channel gravel mining in a piedmont river located in the Southern Pyrenees.

1. INTRODUCTION

River channel disturbance is considered a key factor affecting benthic invertebrate community structure [1]. During disturbances, losses from the benthos occur as a result of involuntary drift. This involuntary drift, along with so-called voluntary or behavioural drift, plays an important role in the dispersal and population dynamics of invertebrates in rivers and streams worldwide [2]. Understanding how channel disturbance affects drift is therefore of great scientific and applied interest. In-channel gravel mining causes major bed disturbance, with consequences for channel morphology and sediment transport dynamics at different temporal and spatial scales. One notable point about the disturbance caused by gravel mining is that, unlike that associated with natural floods, high SSCs occur in the absence of marked flow changes. Thus, in-channel gravel mining provides an opportunity to separate the effects of suspended sediment concentrations (hereafter SSC) and flow change on invertebrate drift. Within this context, research described in this paper assessed changes in SSC and invertebrate drift associated with in-channel gravel mining in a piedmont river located in the Southern Pyrenees.

2. STUDY AREA AND METHODS

The 5-km long study reach is located in the upper River Cinca (Ebro Basin; Figure 1A). The Cinca is characterized by frequent competent flood events. Mean annual discharge (1959-2015) is 29 m^3/s and annual floods exceed 223 m^3/s, while flows of 587 and 740 m^3/s have a recurrence of 5 and 10 years respectively. River bed sediments are very poorly sorted; the range of surface particle sizes encompasses large boulders to patches of sands and fine gravels (i.e. from 0.5 to 2000 mm). The mean SSC for the study period (2013-2015) was 0.09 g/l (turbidity data registered in a monitoring station near DS4 in Figure 1A). Maximum instantaneous SSC at this station was 17.4 g/l, observed during a flood event registered in summer 2015. The upper Cinca has a long history of in-channel gravel mining. Historically, some sections of the river have been riprapped, decreasing the width of the floodplain. Additionally, substantial land-use changes in the headwaters have taken place in the last century, notably land abandonment and the subsequent natural afforestation. All these impacts have resulted in changes in the river, slowly shifting from a braided to a wandering platform, while gravel bars have been vegetated and stabilized and the river channel now incises.

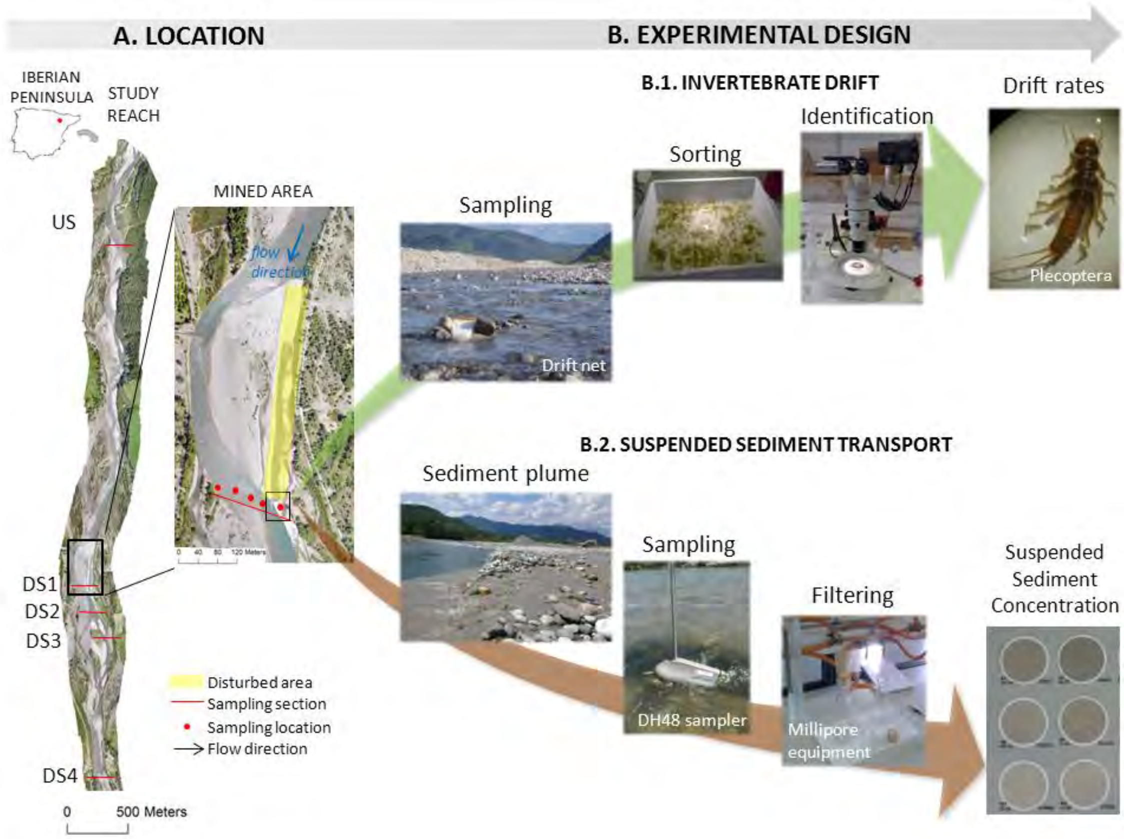

Figure 1. (A) Location of the upper River Cinca and the 5-km study reach. Gravel mining took place in August 2014 in the section highlighted in yellow. (B) Schematic representation of the workflow used to generate drift and suspended sediment data.

Fieldwork was carried out in August 2014. During this month, excavators periodically entered the channel to remove gravels. SSCs and macroinvertebrate drift were sampled on total of six days at sites upstream and downstream of the mined section (Figure 1A); during two of these days gravels were removed.

Invertebrate drift was sampled in riffles using standard drift nets (40×20 cm frame and a mesh of 1000 µm). Nets were left in place for a long enough period to collect sufficient animals for robust statistical analysis; sample time ranged from 240 to 420 minutes. On each day, five drift samples (positioned laterally across the full width of the channel) were collected in each of five sections (Figure 1A-B). The first section (US) was located 3.5 km upstream from the mining, with the remaining sections located at 5, 200, 400 and 1500 m downstream. This design allowed assessment of the magnitude of changes in drift to increases in SSC, and evaluation of the downstream extent of drift changes.

Drift samples were preserved in alcohol until being processed in the laboratory, where they were sorted and identified to genus level (Figure 1B). Identification was based on Tachet et al. [3]. Water samples were collected in each section on each drift sampling occasion using a USDH48 Depth-Integrating sampler and an automatic water sampler (ISCO® 3700). Water samples were filtered, dried and weighed in order to determine SSC (in g/l; Figure 1B).

Relations between SSC and drift rates were assessed using Generalised Estimating Equations (GEEs). GEEs are linear models that allow analysis of data that may not be truly independent because of serial correlation (e.g. repeat measurements at the same locations) or because samples are clustered (e.g. grouped within sites). GEEs were appropriate to our data because drift was repeatedly sampled at the same points across the same sections and because individual samples were geographically clustered by site/section. GEEs were applied to log transformed drift data. Goodness of fit was assessed using Quasi-likelihood values, with significance determined for Wald Chi2 values for the intercept and slope of the fit of drift rate to SSC.

3. RESULTS AND DISCUSSION

3.1. Suspended sediment transport

In total, 135 water samples were obtained during the sampling days. Discharge and SSC data are summarised in Figure 2A. Discharge was low (equalled or exceeded the 75 % of the time) and rather stable for a piedmont river like this, at around 7.5 m^3/s (i.e. with a coefficient of variation ±60 %).

SSCs in the section upstream of the mining ranged from 0.008 to 0.05 g/l; the mean was 0.011 (SD = 0.017 g/l). Downstream from the mining activity (sections DS1 to DS4), mean SSC on the days when mining was not taking place was 0.007 g/l, with a maximum concentration of 0.07 g/l. During mining, mean SSC was one order of magnitude higher than this (i.e. 0.19 g/l), while the maximum instantaneous concentration reached 5.8 g/l. Maximum values were recorded in the sections closest to the mining (i.e. DS1, the section highlighted in Figure 1A and shown in the photograph in 1B). Concentrations recorded during mining were similar to those recorded during natural floods: for instance, mean SSC during a flood in June 2015 was 0.88 g/l, while the maximum was 7.46 g/l.

3.2. Invertebrate drift and drift responses to increases in fine sediment transport

Patterns in the 150 drift samples collected during the study period are summarised in Figure 2B. Across all these samples, drift rate ranged from 5.3 to 150 animals/m^2/hour. The most abundant taxa in the drift were Baetis, Ephemerella and Orthocladinae, together accounting for a 67 % of all individuals recorded.

Upstream from the mining site (US section), drift rate ranged from 13 and 150 animals/m^2/hour; the latter was the highest rate recorded at any of the sites during the study period.

Average drift rate in the downstream sections during periods without mining was 44 animals/m^2/hour (SD= 21 animals/m^2/hour), while during mining this increased to 77 animals/m^2/hour (SD= 35 animals/m^2/hour). Maximum rate in these sections was 145 animals/m^2/hour, obtained during a period of mining when SSC was high (DS2, 200 m downstream from the mined area; SSC = 0.14 g/l).

GEEs indicated a significant positive relationship between SSC and drift rate (Goodness of fit 69; Wald Chi^2 38.50, $P < 0.001$). Thus, increases in SSC were paralleled by increases in drift. The intercept of the model fit using GEEs was significantly greater than zero (Wald Chi^2 63871, $P<0.001$); thus, drift can be expected in the Cinca in the absence of (detectable) suspended sediment transport. The spread of the drift values across the SSC gradient provide some potentially useful insights into the dynamics of drift. There was marked variability in drift at low SSC values, indicating that other factors are dictating variation in drift rates when suspended sediment is low or minimal. Such factors may include local differences in benthic densities, and hence the pool of available drifters, or invertebrate behavior. At higher SSCs, drift was not only generally higher (as indicated by the GEE model output) but was less variable. The slopes of the upper and lower bounds of the scatter to drift indicate that minimum drift expected for any given value of SSC increases at a faster rate than the maximum.

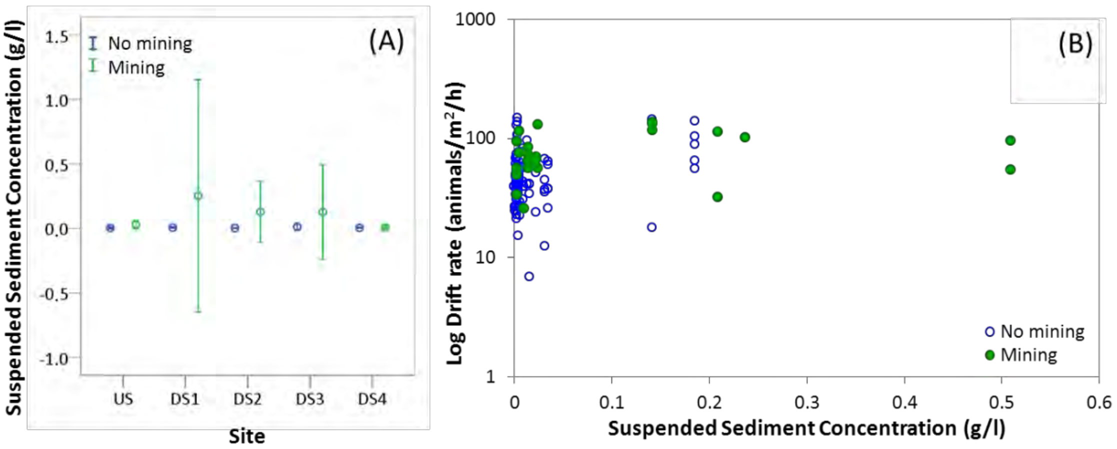

Figure 2. (A) Average suspended sediment concentrations recorded at the sections during August 2014. Error bars indicate one standard deviation above and below the average; (B). Invertebrate drift response to changes in suspended sediment.

4. CONCLUDING REMARKS

Results presented here are part of multidisciplinary research designed to assess the impacts of gravel mining on channel morphodynamics and macroinvertebrate communities. Data on bed grain size characteristics, hydraulic conditions, topography, bedload and suspended sediment transport, as well as benthic and drifting invertebrates are all being collected using a BACI [4] experimental design. Preliminary data presented here indicate that gravel mining results in marked increases in SSC, reaching values similar to those observed during natural flood events. Invertebrates apparently respond to these increases by leaving the bed and drifting downstream. Further comprehensive analysis of the full data set will allow us to more fully understand the physical and ecological disturbance effects of gravel mining and the conditions experienced by animals during mining.

Acknowledgments

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