Doctoral Thesis

Ecosystem processes in the dynamic Tagliamento river (NE-Italy)

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Publication Date: 2002

Permanent Link: https://doi.org/10.3929/ethz-a-004452244

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ECOSYSTEM PROCESSES IN THE DYNAMIC TAGLIAMENTO RIVER (NE-ITALY)

A dissertation submitted to the
SWISS FEDERAL INSTITUTE OF TECHNOLOGY ZURICH

for the degree of
DOCTOR OF NATURAL SCIENCES
presented by

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2002
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Summary:

Increasing human pressures have considerably altered most of the world's rivers, and many efforts are now under way to restore impacted rivers to more natural conditions. However, data about dynamic landscape-level processes in rivers under natural conditions are scarce. This thesis presents baseline data from the Tagliamento River in northeastern Italy, which is one of the least altered large rivers draining the European Alps. This river has only a low degree of regulation and a near-natural flood regime, providing a unique opportunity to study large-scale ecological and geomorphological processes under semi-natural conditions.

The relationships between water level, inundated area, shoreline length, and the complexity of the aquatic component of braided reaches were determined by delineating shorelines at different water levels with a differential global positioning system (dGPS)(Chapter 2). Geographic information systems (GIS) were used to convert spatial data into inundation maps. For most of the time shoreline length was high compared to that of regulated rivers (mean = 171 m ha⁻¹). Statistical relationships were established between water level and inundated area (linear) and water level and shoreline length (2nd-order polynomial). Using these relationships as simple models, the temporal availabilities of shorelines, inundated area, and exposed sediments were predicted. Channel braiding, hydrological connectivity, and the number of isolated waterbodies were related to water level, thus demonstrating changes in the spatial complexity of the aquatic environment which occur during expansion and contraction cycles.

Stability, turnover, and age distribution of aquatic and terrestrial habitats were determined using dGPS and GIS (Chapter 3). Following floods of different magnitudes, the habitats within the active zone of braided reaches were delineated. The effect of vegetated islands on the stability of aquatic habitats was investigated by comparing habitat turnover in two braided reaches, one with many islands and one with only few. In the absence of islands turnover rate of the combined aquatic area was up to 70% between survey dates (cumulative 90 % over 2.5 y period);
turnover in the presence of islands was lower (cumulative 75% over 2.5 y period). In general, aquatic habitats were very young (< 7 months) with particularly short half-lives for backwaters and pools (< 3 months). Despite the large changes in reach configuration, the relative habitat composition within the active corridor remained almost constant.

LW storage and distribution were investigated in braided reaches. Because the availability of LW on the flood plain was very large, we developed a sampling procedure to estimate the total LW storage (Chapter 4). A randomized block design was used to establish permanent plots at random positions that were localized in the field using GPS. Using this method we quantified the storage of LW on braided reaches on four occasions, spanning 1.5 years (Chapter 5). The results indicated LW storage to be larger than anticipated (100-150 t ha\(^{-1}\)) and relatively constant at a time-scale of several years, despite very high turnover rates for individual LW deposits (as much as 95% by a single flood). Analyses of LW persistence showed that the probability that a flood does not move a LW deposit depended on flood magnitude, and the volume and location of the deposit.

An attempt was made to determine the mass balance of coarse organic matter (LW + CPOM) in braided reaches (Chapter 6). LW storage was about 200 times higher than CPOM storage. The accumulation of CPOM in vegetated islands and riparian forest was due mainly to litter fall rather than retention of drifting material. The preliminary mass balance suggested that input of CPOM/LW from upstream was 1000 t y\(^{-1}\). Various improvements in the methodology are suggested. Future research should be conducted with large numbers of replicates over longer time spans and should include exposed sediments (dominant but unstable stratum), continuous transport measurements of CPOM, transport of LW, and belowground storage of LW.

The significance of the findings for river restoration is discussed. In attempting to restore rivers it is important to recognize the ecological significance of small flow pulses and abundant LW.
Zusammenfassung:


Zusammenhänge zwischen Wasserstand, überströmter Fläche, Uferlänge und Komplexität des aquatischen Lebensraumes wurden in Flussabschnitten mit verzweigten Gerinnen untersucht. Dazu wurde der Uferverlauf bei verschiedenen Wasserständen mit einem differentiellen globalen Positionierungssystem (dGPS) kartiert (Kapitel 2). Anhand dieser räumlichen Daten wurden mit geographischen Informationssystemen (GIS) Überströmungskarten erstellt. Im Vergleich zu regulierten Flüssen war die Uferlänge groß (Mittelwert = 171 m ha⁻¹). Statistische Beziehungen zwischen Wasserstand und überströmter Fläche einerseits (linear), sowie Uferlänge andererseits (Polynom zweiter Ordnung) wurden etabliert. Mit derartigen Modellbeziehungen wurde die Verfügbarkeit der Elemente Ufer, überströmte Fläche und exponierte Sedimente vorhergesagt. Das Ausmaß der Verflechtung von Gerinnen, die hydrologische Konnektivität und die Anzahl isolierter Tümpel waren vom Wasserstand abhängig. Dies zeigen die Veränderungen der räumlichen Komplexität bei Expansions- und Kontraktionszyklen.

Stabilität, Umlagerung, und Altersverteilung von aquatischen und terrestrischen Habitate wurden mit dGPS und GIS bestimmt (Kapitel 3). Alle Habitate in der aktiven Aue von Flussabschnitten mit verzweigten Gerinnen wurden nach Überströmungen verschiedener Stärke kartiert. Der Effekt von Vegetationsinseln auf die Stabilität der aquatischen Habitate wurde mit einem
Vergleich der Umlagerungsraten in je einem Flussabschnitt mit vielen und mit wenigen Vegetationsinseln untersucht. Im Flussabschnitt mit wenigen Inseln betrug die Umlagerungsrate der gesamten aquatischen Fläche bis zu 70% zwischen den Erhebungen (90% über den 2,5-jährigen Untersuchungszeitraum). Bei Vorhandensein von vielen Vegetationsinseln war die Umlagerungsrate geringer (75% über den 2,5-jährigen Untersuchungszeitraum). Die aquatischen Habitate waren im allgemeinen sehr jung (< 7 Monate), insbesondere Hinterwasser und Tümpel (< 3 Monate). Trotz der großen Veränderung in der Konfiguration der Flussabschnitte blieb die Zusammensetzung der Habitate in der aktiven Aue fast konstant.

Bestand und Verteilung von Totholz wurden in Flussabschnitten mit verzweigten Gerinnen untersucht. Da der Totholzbestand sehr hoch war wurde eine neue Methode zur Erhebung repräsentativen Stichproben entwickelt, bei der Dauerflächen an zufälligen Orten (GPS Lokalisierung) etabliert wurden (Kapitel 4). Mit dieser Methode wurde der Totholzbestand in vier Aufnahmen über einen Zeitraum von 1,5 Jahren quantifiziert (Kapitel 5). Die Resultate wiesen darauf hin, dass der Totholzbestand in frei fließenden Flüssen höher ist als erwartet (100-150 t ha⁻¹) und über mehrere Jahre konstant bleibt, obwohl einzelne Totholzablagerungen sehr hohe Umlagerungsraten aufweisen (bis zu 95% nach einem Hochwasserereignis). Analysen der Verweildauer dieser Ablagerungen zeigten, dass die Wahrscheinlichkeit, dass ein Hochwasserereignis eine Totholzablagerung nicht entfernt, von der Stärke der Überströmung, dem Volumen der Ablagerung und seiner Position in der Aue abhängig ist.

Eine vorläufige Erhebung der Massenbilanz von grobem organischen Material (GOM) und Totholz wurde in Flussabschnitten mit verzweigten Gerinnen durchgeführt (Kapitel 6). Der Totholzbestand war ungefähr 200 mal so groß wie der Bestand an GOM. Die Anhäufung von GOM in Vegetationsinseln und im Auwald konnte hauptsächlich auf Laubfall zurückgeführt werden und weniger auf den Rückhalt des im Wasser transportierten Materials. Die vorläufige Massenbilanz
belief sich auf einen Eintrag von 1000 t y⁻¹ GOM und Totholz aus flussaufwärts gelegenen Bereichen. Zukünftige Untersuchungen sollten viele Wiederholungen und lange Zeitspannen berücksichtigen, sowie Erhebungen an exponierten Sedimenten, kontinuierliche Messungen des Transportes von organischem Material und den unterirdischen Bestand umfassen.

Die Bedeutung dieser Ergebnisse für die Renaturierung von Flüssen wird diskutiert. Insbesondere wird empfohlen die ökologische Bedeutung von kleineren Überströmungen und großen Totholzbeständen bei Renaturierungsprojekten zu berücksichtigen.
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Riassunto:

L’impatto umano in continuo aumento ha considerevolmente alterato la maggior parte dei fiumi del mondo e molti sforzi vengono ora investiti per riportare questi fiumi in uno stato più naturale. Dati, che descrivono i processi a livello di paesaggi dinamici di fiumi in condizioni naturali sono però pochi numero. La presente tesi di laurea offre dati provenienti dal fiume Tagliamento nel nord-est dell’Italia, uno dei fiumi delle alpi europee meno alterati. Le sue acque sono solo poco regolate e il fiume presenta un regime delle piene quasi naturale, offrendo un’opportunità unica per studiare processi ecologici e geomorfologici su larga scala sotto condizioni seminaturali.

Sono state determinate le relazioni tra il livello dell’acqua, la superficie inondata, il perimetro dell’inondato e la complessità della componente acquatica dei corsi d’acqua intrecciati, delineando il contorno del sommerso a diversi livelli d’acqua con un sistema di posizionamento globale differenziale (differential global positioning system, dGPS)(capitolo 2). Sistemi di informazione geografica (GIS) (Geografic information system, GIS) sono stati usati per trasformare i dati registrati in cartine delle aree allagabili. Per la maggior parte del tempo, il perimetro dell’acqua era lungo, se confrontato con quello di fiumi regolati (media = 171 m ha \(^{-1}\)). Sono state stabilite relazioni statistiche tra il livello dell’acqua e la superficie sommersa (lineare) e tra il livello d’acqua e la lunghezza del perimetro dell’inondato (polinomiale di secondo grado). Usando queste relazioni come dei modelli semplici, sono stati predetti la disponibilità nel tempo della lunghezza del perimetro dell’allagato, l’area della superficie sommersa e quella del sedimento esposto. L’intrecciamento dei canali, la connettività idrologica e il numero di superfici d’acqua isolate, sono stati messi in relazione con il livello dell’acqua, dimostrando i cambiamenti nella complessità dell’ambiente acquatico, che avvengono duranti i cicli di espansione e contrazione.

Usando dGPS e GIS, sono stati determinati la stabilità, il rinnovo (turnover) e la distribuzione per età di habitat acquatici e terrestri (capitolo 3). In seguito a
piene di dimensioni differenti, sono stati delineati gli habitat all’interno della zona attiva dei canali intrecciati. Confrontando il rinnovo degli habitat all’interno di due tratti di fiume ad alveo ramificato, una con molti e una con pochissimi isolotti vegetali, è stato studiato l’effetto che queste isole vegetali hanno sulla stabilità di habitat acquatici. Con l’assenza di isole, tra un rilevamento e l’altro la rata di rinnovamento delle superfici acquatiche collegate ammontava a 70% (con un cumulativo di 90% per un periodo di osservazione di 2 anni e mezzo); il rinnovo in presenza di isole era più piccolo (cumulativo di 75% durante 2.5 anni). In generale gli habitat acquatici erano molto giovani (<7 mesi), e il tempo di dimezzamento dei rigurgiti e degli stagni (<3 mesi) era particolarmente corto. Nonostante i grandi cambiamenti nella configurazione dell’alveo, la relativa composizione degli habitat all’interno della zona attiva, è rimasta quasi costante.

L’accumulo e la distribuzione di GL (grossi pezzi di legno) sono stati studiati all’interno della zona a canali intrecciati. La quantità di GL all’interno dell’alveo era molto elevata. Per questa ragione abbiamo sviluppato un metodo di rilevamento per stimare l’accumulo totale di GL (capitolo 4). Un disegno a blocchi randomizzati è stato usato per determinare delle unità sperimentali permanenti, posizionate casualmente nell’alveo e localizzate usando il GPS. Usando questo metodo, in quattro occasioni sull’arco di un anno e mezzo, abbiamo quantificato l’accumulo di GL nel tratto di alveo ramificato (capitolo 5). L’accumulo di GL è risultato essere più grande del previsto (100-150 t ha⁻¹) e relativamente costante sull’arco di alcuni anni, nonostante il fatto che la rata di rinnovamento per singoli depositi di GL sia molto alta (95% causati da un’unica piena). Analizzando la persistenza del GL, abbiamo mostrato che la probabilità che un’inondazione non sposti un deposito di GL, dipende dalla dimensione dell’inondazione e dal volume e dalla posizione del deposito.

Abbiamo tentato di determinare il bilancio di materia organica grezza (GL + CPOM) all’interno della zona a canali intrecciati (capitolo 6). L’accumulo di GL era all’incirca 200 volte più grande dell’accumulo di CPOM. L’accumulo di CPOM
nelle isole vegetali e nella foresta riparia è dovuto soprattutto alla caduta di foglie, piuttosto che alla ritenzione di materiale galleggiante. Dal bilancio di materia preliminare risulta, che l'apporto di CPOM/GL da monte era di 1000 t all’anno. Nella tesi vengono proposte diverse possibilità di miglioramento del metodo. Studi futuri dovrebbero essere condotti con ampi margini, su un arco di tempo più lungo e dovrebbero comprendere i sedimenti esposti (lo strato dominante ma instabile), misurazioni continue del trasporto di CPOM e di GL, e l'accumulo sotterraneo di GL.

Futuri sforzi per riportare fiumi in uno stato più naturale dovrebbero tener conto dell’importanza ecologica delle piccole piene e di un abbondante presenza di GL.
CHAPTER 1

GENERAL INTRODUCTION
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General introduction:

River corridors worldwide are increasingly endangered ecosystems. In Europe human influences on river corridors date back 3000 years to the Neolithic age. By then large-scale deforestation for pastoral and agricultural purposes had resulted in increased erosion and sediment loads that in turn led to changes in channel form and stability (e.g. Bravard and Petts 1996). With the rise of technology, human pressures on river ecosystems have further increased through water abstraction (domestic, agricultural, and industrial), navigation, fisheries, hydroelectric power generation, and recreation (e.g. Petts et al. 1989, Dynesius and Nilsson 1994, Ward and Stanford 1995). These human pressures have profoundly affected the dynamic nature of river geomorphology. In the Alps, for example virtually all rivers were straightened or constrained during the 19th century (Whitton 1984, Petts et al. 1989), disconnecting main channels from their flood plains. Before regulation large floodplain rivers across the temperate forested biome were characterized by dynamic morphological processes, the presence of large wood (LW) and woody vegetated islands (Gurnell and Petts 2002).

In the context of rivers connectivity refers to exchanges of matter (e.g. water, sediments, and nutrients), energy (e.g. organic detritus), and organisms across riverine landscapes (Ward 1997). The level of connectivity is a fundamental property related to river ecosystem health (Pringle 2000). River corridors where connectivity exists along the longitudinal, lateral, vertical, and temporal dimensions of lotic ecosystems (Ward 1989) have become increasingly rare due to the above-mentioned human pressures (Bravard and Petts 1996). Remnants with high connectivity have remained mostly in headwater sections.

There are very few data about dynamic landscape level processes such as the effect of water level on the availability of floodplain habitats, the turnover rates, life expectancy and age-distribution of floodplain habitats, storage turnover and retention of LW and coarse particulate organic matter (CPOM >1mm).
Consequently, current concepts about river functioning have been advanced with very little empirical knowledge of natural systems (Edwards et al. 1999).

**Thesis goals and structure:**

Hydrology is the most important factor influencing the ecology of river flood plains (Table 1) (Tockner and Stanford 2002). Flow regimes influence expansion and contraction cycles of aquatic and terrestrial components of the flood plain, hydrological exchange, LW distribution and transport, and CPOM dynamics. Particularly important are the high flows which represent a natural physical disturbance along river corridors (Junk et al. 1989, Puckridge et al. 1998, Swanson et al. 1998, Tockner et al. 2000, Arscott et al. 2002), and which alter floodplain morphology at various spatial scales (Hughes and Rood 2001, Ward et al. 2002). Overbank flooding (flood pulses sensu Junk et al., 1989) may occur seasonally or unpredictably (Poff and Ward 1989). Rivers also experience frequent smaller water level fluctuations ("flow pulses" sensu Tockner et al., 2000) that occur well below bankfull discharge. Although not responsible for large morphological restructuring, these flow pulses are important for creating and maintaining habitat heterogeneity and influence various ecosystem processes (Benke et al. 2000, Tockner et al. 2000, Nat van der et al. 2002).

The research in this thesis contributes to our understanding of the effect of flow and flood pulses on floodplain habitat availability, turnover, life expectancy, and age distribution. Using a differential global positioning system (dGPS) inundated areas were delineated in two braided flood plains at various river stages. Using these data relationships between water level, inundated area, shoreline length, and system complexity were established. Simple statistical models were then developed to predict the availability of shorelines, inundated areas, and exposed sediments through time. Another investigation aimed at determining the effect of flood pulses on turnover rates and age distribution of aquatic and terrestrial habitats in the active zone of braided flood plains. For this purpose repeated dGPS
floodplain mappings were carried out at similar water levels following floods of different magnitude.

Table 1. The estimated relative importance of environmental factors that determine the properties of flood plains (empirical values, after Tockner and Stanford, 2002)

<table>
<thead>
<tr>
<th>Environmental factor</th>
<th>Relative importance (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hydrology</td>
<td>60</td>
</tr>
<tr>
<td>Fertility</td>
<td>&lt;10</td>
</tr>
<tr>
<td>Salinity</td>
<td>&lt;5</td>
</tr>
<tr>
<td>Disturbance</td>
<td>30</td>
</tr>
<tr>
<td>Competition</td>
<td>&lt;5</td>
</tr>
<tr>
<td>Grazing</td>
<td>&lt;5</td>
</tr>
<tr>
<td>Burial</td>
<td>&lt;5</td>
</tr>
</tbody>
</table>

LW influences geomorphological processes by trapping sediments and redirecting water flows (Hupp and Osterkamp 1996, Bendix and Hupp 2000, Tabacchi et al. 2000, Gurnell and Petts 2002). Propagule establishment or vegetative regeneration of living driftwood in the lee of LW deposits initiates the establishment of vegetated islands (Gurnell et al. 2001). Furthermore, LW influences hydrology, hydraulics, sedimentology, and morphology (e.g., Nakamura and Swanson 1993, Gurnell et al. 1995, Gurnell et al. 2002), as well as providing habitats for a large array of organisms (e.g. Benke et al. 1984, Fausch and Northcote 1992, Reich 1994, Steel et al. 1999, Mac Nally et al. 2001). LW has, however, been removed from most large rivers to facilitate navigation and hydropower generation. In order to obtain better understanding of the natural abundance and dynamic nature of LW in large rivers we investigated the influence of islands on LW storage, the stability of LW storage through time, species composition of LW, lateral LW input, and persistence of LW.
CPOM is a very important source of energy in river ecosystems. Longitudinal, lateral, and vertical exchange of CPOM connects different parts of a river to each other energetically (e.g. Wallace et al. 1999). The determination of mass balances of organic matter provides insight into the functioning of stream and river ecosystems (Wanner et al. 2002), taking account of storage, input, transport, retention, production, and decomposition. We conducted a preliminary investigation of the mass balance for CPOM and LW in braided flood plains, by establishing estimates of CPOM and LW storage, distribution, transport, production, and decomposition.

*An interdisciplinary project:*

The interdisciplinary ETH project "The role of island dynamics in the maintenance of biodiversity in an Alpine River system" which funded this dissertation was aimed at gathering baseline knowledge about a natural river ecosystem (Edwards et al. 1999). Such knowledge could be of considerable importance for the numerous river restoration projects currently underway or in development in Europe (Waal de et al. 1998). The thesis was conducted on the Tagliamento River in northeastern Italy. In the context of restoration it has been suggested that, due to a remarkably low degree of regulation (Petts et al. 2000) and a near-natural flood regime (Müller 1995, Gurnell et al. 2000) the river represents a reference condition for rivers draining the European Alps (Ward et al. 1999). It provides a unique opportunity to study dynamic large-scale ecological and geomorphological processes under natural conditions. The interdisciplinary project involved related and interdependent studies of: the ecology of woody pioneer species (Sophie Karrenberg), macroinvertebrates and habitat heterogeneity along a river corridor (David B. Arscott), dissolved organic matter (Edith Kaiser), invertebrate fauna of floodplain ponds (Ute Karaus), the ecology of shoreline zones (Achim Pätzold), and floodplain morphology/organic matter dynamics (this thesis).
References:


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Chapter 2

(With Andreas P. Schmidt, Klement Tockner, Peter J. Edwards and J.V. Ward)

(Ecosystems 5 (7), 2002)

And I said, "Hey gunner man, That's quicksand, that's quicksand that aint mud"

Have you thrown your senses to the war or did you lose them in the flood?

(Bruce Springsteen, Lost in the Flood, 1975)
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Inundation dynamics in braided flood plains
(Tagliamento River NE-Italy).

Abstract:

The relationships between water level, inundated area, and shoreline dynamics were investigated in a bar-braided and an island-braided flood plain of the Tagliamento River (NE-Italy). Ground based surveys with a differential GPS unit were used to delineate all aquatic-terrestrial interfaces (shorelines) in the active flood plain at different water levels. Despite complex inundation patterns a highly significant ($p<0.00001$) linear relationship between water level and arcsine square root of inundated area was found in both reaches ($y = 0.49x + 0.07$). A highly significant ($p<0.00009$) second-order polynomial relationship occurred between water level and shoreline length ($y = 87.83 - 65.85x^2 + 169.83x$).

Using these relationships as simple predictive models, we converted several years of water level data into predictions for degree of inundation and shoreline length. The plot of the simulated degree of inundation strongly resembled the actual hydrograph. Complete inundation of the active flood plains occurred 1-2 times per year; however, the degree of inundation at lower water levels was highly dynamic during most of the year. Simulated shoreline length averaged 171 m ha$^{-1}$ (13.6 km km$^{-1}$) with a maximum of 197 m ha$^{-1}$ (15.6 km km$^{-1}$) occurring during intermediate water levels. The corresponding values determined with GPS were somewhat higher, with an average value of 181 m ha$^{-1}$ (14.4 km km$^{-1}$) and a maximum of 214 m ha$^{-1}$ (16.3 km km$^{-1}$). During major flood events, actual shoreline length decreased to 28 m ha$^{-1}$ (2.1 km km$^{-1}$). Braiding index and upstream surface hydrologic connectivity were positively related to water level, whereas total area of isolated waterbodies was negatively related to water level. The number of nodes remained high most of the time during the two-year study period.
Introduction:

Flood plains are complex physical features bordering rivers in braided, meandering or anastomosing reaches. They are highly dynamic in natural state, because floods rework their morphology at various spatial scales (Hughes and Rood 2001, Ward et al. 2002b). Hydrology is the primary driving force for morphological and ecological processes in flood plains (Junk et al. 1989, Sparks et al. 1990, Bayley 1991, Spink et al. 1998, Tockner et al. 2000b).

Many of the early concepts describing river ecosystems such as the River Continuum Concept (RCC) (Vannote et al. 1980), the Nutrient Spiraling Concept (Newbold et al. 1982), or the Serial Discontinuity Concept (Ward and Stanford 1983) focussed on longitudinal patterns and processes. The Flood Pulse Concept (FPC) (Junk et al. 1989) explicitly incorporates lateral interactions between the channel and the flood plain and suggests that the flood pulse is the most important physical variable for biological processes in tropical (Junk et al. 1989, Bayley 1991) and temperate rivers (Tockner et al. 2000b). The focus given to the lateral dimension by the FPC stimulated many studies that examined the ecological implications of flooding in stream and river ecosystems (e.g. Sparks et al. 1990, Bayley 1991, Spink et al. 1998). Because of this increased awareness of river-floodplain interactions, existing theories were amended to incorporate lateral dimensions of river ecosystems (Ward and Stanford 1995, Fisher et al. 1998).

Overbank flooding (flood pulses) may occur seasonally or unpredictably (Poff and Ward 1989). In addition, rivers may experience many smaller water level fluctuations (flow pulses sensu Tockner et al. (2000b)) that occur well below bankfull discharge. Although not responsible for the main morphological restructuring of the ecosystem, these 'flow pulses', may be important for creating and maintaining habitat heterogeneity and ecosystem processes (e.g. Benke 2001). Both aquatic and terrestrial areas in the flood plain experience frequent cycles of expansion and contraction because of small water level fluctuations (Vries de 1995, Stanley et al. 1997, Tockner et al. 2000b). Contraction of aquatic habitats, for example, fragments
populations, reduces habitat availability, and alters ecosystem processes such as primary production and nutrient cycling (Stanley et al. 1997).

Expansion and contraction cycles control the availability of aquatic-terrestrial interfaces (shorelines). Shorelines are important habitats for a large spectrum of organisms such as fish (Wintersberger 1996, Schiemer et al. 2001), birds (Décamp et al. 1987, Reich 1994), and terrestrial invertebrates (Bonn et al. 2002). In larger rivers, shoreline habitats may serve as important flood refugia (Rempel et al. 1999). Additionally, they control the transfer of matter between aquatic and terrestrial zones (e.g. Naiman and Décamp 1997, Bardgett et al. 2001, Ward and Wiens 2001). With increased channelisation and regulation of rivers across the world (Petts et al. 1989), shoreline habitat is becoming increasingly rare (Pinay et al. 1990, Ward et al. 2002a). It is important to investigate the relationship between discharge and available shoreline habitat in the remaining, free-flowing rivers to gain a better understanding of the functioning of river systems in their natural state. This insight, in turn, is of importance for increasing the chances of success of river restoration projects (Benke et al. 2000). Cycles of expansion and contraction also determine the availability of several distinct aquatic habitats, such as backwaters, alluvial channels, and isolated pools. Increasing human pressure requires quantitative methods to predict how a river's form changes when discharge is altered.

Remote sensing studies have focused on the hydro-morphological dynamics of flood plains (e.g. Lesack and Melack 1995, Mertes 1997). Aerial photography has been used to analyze long-term changes in channel configuration of streams (Gilvear et al. 1995) and braided rivers (Warburton et al. 1993). On shorter time scales, aerial photographs have been used to analyze the relationship between discharge and channel change in a braided river (Mosley 1982). Mosley (1982) showed that increased discharge increased the availability of deep-water habitat without influencing the availability of shallow water habitat. Using aerial photography, Benke et al. (2000) established a linear relationship between discharge and the arcsine of inundated floodplain area for the subtropical Ogeechee River,
USA. Passive microwave observations have been used to establish a relationship between discharge and degree of floodplain inundation in tropical wetlands (Hamilton et al. 1996) and tropical rivers (Sippel et al. 1998). Global Positioning Systems (GPS) have also been used to acquire spatial data from flood plains. For instance, repeated transect measurements were used to develop a Digital Elevation Model (DEM) which was used to monitor channel change in a braided river in Scotland (Brasington et al. 2000).

We applied differential GPS to gather spatial data from two braided flood plains at various river stages. GPS allowed us to acquire data rapidly at any water level, independent of airplanes or satellites. The river system investigated was distinctly different from systems studied so far because it represents the last large free-flowing Alpine river in Europe (Müller 1995, Ward et al. 1999a). The goal of this study was to quantify the relationship between water level, inundated area, shoreline length, and system complexity in a bar-braided and an island-braided flood plain. Based on the relationship, a predictive model was generated to calculate the availability of floodplain habitats.

Study site:
The Tagliamento River is a 7th order gravel-bed river located in NE-Italy. It has a catchment area of 2580 km² and a total length of 172 kilometers. About 70 percent of its catchment area is located in the Southern Limestone Dolomite Alps. Average elevation of the catchment is 1159 m a.s.l, with a maximum of 2781 m a.s.l. (Ward et al. 1999a, Arscott et al. 2000). Through a sequence of constrained, braided, and meandering reaches, it flows from the Alps to the Adriatic Sea. The river has many vegetated islands, which cover approximately 9% of the entire active flood plain (Ward et al. 1999a). The active flood plain consists of three major landscape elements: water, exposed alluvial sediments, and vegetated islands. Along the Tagliamento, active floodplain width is up to 1.5 km. The active flood plain is fringed by continuous riparian woodland (Ward et al. 1999, Gurnell et al. 2000a).
Although the river has been subject to water abstraction and channelisation (lowermost 25 kilometers), it remains remarkably free of intensive management. Consequently, the Tagliamento still has an essentially pristine morphological character and a natural flood regime (Müller 1995, Ward et al. 1999a). The hydrology is characterized by a pluvio-nival regime with frequent flashy flood and flow pulses. Flood pulses generally occur in spring and autumn (Fig. 1B) and have

Fig. 1. (A) Location of the Tagliamento catchment, with insets showing the configuration of the bar-braided and island-braided reaches in May 2000. In the insets, black indicates floodplain forest and vegetated islands; dark gray indicates surface water; light gray indicates exposed alluvial gravel. (B) Stage height for the Tagliamento River at the San Pietro gauging station 1.5 km downstream of the island-braided reach.
an extremely low degree of predictability (Campolo et al. 1999). The average discharge at the location where the river leaves the Alps is approximately 90 m$^3$ s$^{-1}$, and the 2, 5, and 10 year floods are estimated to be 1100, 1600, and 2150 m$^3$ s$^{-1}$, respectively (Gurnell et al. 2000a). For a more detailed description of the catchment characteristics and hydrodynamics, see Ward et al. (1999a) and Gurnell et al. (2000a). This study was carried out in two morphologically distinct reaches; hereafter referred to as the bar-braided reach and the island-braided reach (reaches III and IV from Arscott et al. (2000)). The location of both reaches within the Tagliamento catchment is shown in Fig 1A. The bar-braided reach was located downstream of river kilometer 74. Under base-flow conditions, large areas of exposed sediments and multiple channels (Table 1) dominated this reach. Its active flood plain area was 103 hectares, reach length was 1.4 km, and active floodplain width was up to 1.0 km. The island-braided reach was located downstream of river kilometer 81 and upstream of a narrow bedrock constrained knick point at Pinzano (Fig 1A). Total surface area was 143 ha, length was 1.8 km, and the active flood plain width was 0.8 km. The reach included a complex channel network with ca. 20 vegetated islands. In addition to main- and secondary channels, many alluvial channels, backwaters, and isolated waterbodies occurred in both reaches. Exposed sediments in the form of lateral or mid-channel gravel bars, represented the largest proportion of landscape cover elements.

**Methods:**

Geographical data were collected using a Differential Global Positioning System (dGPS) rover unit (Trimble TCS1). All shorelines were delineated in situ by walking along them, carrying the dGPS receiver. Positions were recorded at 5-second intervals and later post corrected using base station data to obtain a precision of $\leq 0.3$ meter. The base station was located less than five km from both study reaches and remained stationary during all surveys to record variations in triangulation due to satellite movement.
Data were collected at different river stage heights following a major flood that started on 26 October 1999. Between 27 October and 15 November 1999, data were collected on five dates in the island-braided reach and on two dates in the bar-braided reach (Table 1). The dangerous and time-consuming task of mapping during flood conditions forced us to focus our attention on one reach at a time. Both reaches were mapped again in January 2000 during a period of lower water level (0.74 m). To obtain comparable data for the bar-braided reach we carried out two additional mapping sessions there in April and October 2000, when water levels reached values comparable to those for which we mapped the island-braided reach. Finally, in December 2001 we mapped the bar-braided reach at low flow (0.10 m). For three of the mapped water levels, the configuration and extent of inundation are shown in oblique photographs (Fig. 2).

The water levels covered by these surveys ranged between 2.35 m and 0.74 m at the gauging station of San Pietro (Fig. 1A) for the island-braided reach, and between

<table>
<thead>
<tr>
<th>Bar-braided reach</th>
<th>Island-braided reach</th>
</tr>
</thead>
<tbody>
<tr>
<td>Length (km)</td>
<td>1.5</td>
</tr>
<tr>
<td>Area active flood plain (ha)</td>
<td>103</td>
</tr>
<tr>
<td>Area water (%)</td>
<td>35</td>
</tr>
<tr>
<td>Area islands (%)</td>
<td>3.1</td>
</tr>
<tr>
<td>Area exposed sediments (%)</td>
<td>61.9</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>water level (m)</th>
<th>date</th>
<th>water level (m)</th>
<th>date</th>
</tr>
</thead>
<tbody>
<tr>
<td>0.15</td>
<td>10 Dec 01</td>
<td>0.74</td>
<td>10 Jan 00</td>
</tr>
<tr>
<td>0.74</td>
<td>09 Jan 00</td>
<td>1.02</td>
<td>12 Nov 99</td>
</tr>
<tr>
<td>1.00</td>
<td>15 Nov 99</td>
<td>1.23</td>
<td>01 Nov 99</td>
</tr>
<tr>
<td>1.20</td>
<td>03 Nov 99</td>
<td>1.50</td>
<td>28 Oct 99</td>
</tr>
<tr>
<td>1.46</td>
<td>04 Apr 00</td>
<td>1.65</td>
<td>27 Oct 99</td>
</tr>
<tr>
<td>1.65</td>
<td>15 Oct 00</td>
<td>2.35</td>
<td>26 Oct 99</td>
</tr>
</tbody>
</table>

Table 1. Morphological characteristics of a bar-braided and an island-braided reach of the Tagliamento River NE-Italy (at a water level of 0.95 m), and dates and water levels at which maps were made.
1.65 m and 0.15 m for the bar-braided reach. The mean water level at the San Pietro gauge is approximately 0.8 m. Due to the frequently changing and complex bed morphology, cross-sectional channel profiles change rapidly preventing a reliable estimation of discharge (see Campolo et al. 1999, Gurnell et al. 2000a). Consequently, water level rather than discharge was used in our analysis. Water level data were provided by the 'Direzione Regionale dell’ Ambiente' and the 'Autorita di Bacino dei Fiume Isonzo, Tagliamento, Livenza'.

![Fig. 2 Oblique photographs of the bar-braided (A) and the island-braided reach (B) taken at low, mean and high water levels. Arrows indicate the flow direction (Photos: C. Yoshimura and D. van der Nat).](image)

We used ARC-INFO software to generate maps and to determine complexity parameters from these maps. Sinuosity and braiding indices (Friend and Sinha 1993), number of nodes, and number and total area of floodplain ponds were determined. Nodes are ecotones within aquatic systems defined as the intersections between waterbody thalwegs (Arscott et al. 2000). Channel nodes are intersections
between channel thalwegs with both upstream and downstream connections. The total number of nodes also includes intersections with and within waterbodies with only upstream or only downstream connections. Upstream surface hydrological connectivity (SC) was measured from all maps following Malard et al. (2000), by dividing the area of channels fed by surface water (Ac) by the total area of channels with surface flow (A). A Spearman rank order correlation was performed to investigate which parameters best described the relationship between water level and complexity.

Additionally, we determined total inundated area and shoreline length from each map. An overlay function was used to combine all information into inundation maps, which showed the extent of inundation for a given water level for each flood plain. The largest vertical elevation differences found in topographic transects through the study reaches were about 2.5 m (see Gurnell et al. 2000b). Shoreline length is expressed in both meters per ha active flood plain and in kilometer per river-km. Inundated area is expressed as the proportion (%) of the active floodplain covered by water.

Relative inundation of the active flood plain was arcsine square root transformed and regressed against water level and reach in a linear model (Statistica 5.1 Statsoft Inc.). We assumed that at maximum water level shoreline length would equal the length of the border between the active flood plain and the adjacent riparian forest. This assumption made a second-order polynomial function the most logical regression curve. Inundated area was regressed in a linear model against water level squared, water level, and reach.

We used the observed relationship as a predictive model and converted 5 years of water level data from the San Pietro gauging station to a time series of relative inundation and shoreline length. Consequently, a duration curve for the degree of inundation and a frequency distribution of shoreline length were generated from the same data.
Results:

Sinuosity was low and constant (1.0 to 1.24) across water levels for both reaches (Table 2). The braiding index reached values as high as 6.87. The Spearman rank order correlation showed that both braiding index and upstream surface hydrologic connectivity showed a significant positive relationship with water level (Table 2). The total area covered by ponds was inversely related to water level. The total number of nodes was positively related to the shoreline length (Table 2). Channel nodes, sinuosity and number of ponds were not related to either water level or shoreline length (Table 2).

In the bar-braided reach more than 60 percent of the total active floodplain area was inundated at the highest mapped water level (1.65 m). During the falling limb of the hydrograph, the degree of inundation decreased monotonously to

Table 2. Complexity descriptors: sinuosity, braiding index, channel nodes, total nodes, number of ponds, total area of ponds, relative inundation, upstream surface hydrologic connectivity (SC), and shoreline length (SLL) for the bar-braided and island-braided reaches at all mapped water levels.

<table>
<thead>
<tr>
<th>Reach</th>
<th>Stage (m)</th>
<th>Sinuosity</th>
<th>Braid. index</th>
<th>Chan. nodes</th>
<th>Total nodes</th>
<th>no. pond</th>
<th>A (m²)</th>
<th>Inun (%)</th>
<th>SC</th>
<th>SLL (m ha⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bar-braided</td>
<td>0.15</td>
<td>1.24</td>
<td>2.17</td>
<td>24</td>
<td>33</td>
<td>12</td>
<td>14047</td>
<td>10.8</td>
<td>0.83</td>
<td>108.0</td>
</tr>
<tr>
<td></td>
<td>0.74</td>
<td>1.09</td>
<td>1.57</td>
<td>12</td>
<td>42</td>
<td>7</td>
<td>3834</td>
<td>12.6</td>
<td>0.43</td>
<td>145.6</td>
</tr>
<tr>
<td></td>
<td>1.00</td>
<td>1.11</td>
<td>5.40</td>
<td>64</td>
<td>99</td>
<td>21</td>
<td>6164</td>
<td>22.6</td>
<td>0.83</td>
<td>212.8</td>
</tr>
<tr>
<td></td>
<td>1.20</td>
<td>1.08</td>
<td>5.82</td>
<td>51</td>
<td>93</td>
<td>8</td>
<td>2404</td>
<td>29.5</td>
<td>0.91</td>
<td>214.0</td>
</tr>
<tr>
<td></td>
<td>1.46</td>
<td>1.08</td>
<td>4.99</td>
<td>24</td>
<td>60</td>
<td>9</td>
<td>3649</td>
<td>41.6</td>
<td>0.92</td>
<td>195.2</td>
</tr>
<tr>
<td></td>
<td>1.65</td>
<td>1.02</td>
<td>6.87</td>
<td>31</td>
<td>54</td>
<td>1</td>
<td>52</td>
<td>58.7</td>
<td>0.99</td>
<td>160.6</td>
</tr>
<tr>
<td>Island-braided</td>
<td>0.74</td>
<td>1.00</td>
<td>3.60</td>
<td>34</td>
<td>48</td>
<td>22</td>
<td>7542</td>
<td>24.3</td>
<td>0.84</td>
<td>178.4</td>
</tr>
<tr>
<td></td>
<td>1.02</td>
<td>1.00</td>
<td>4.41</td>
<td>50</td>
<td>83</td>
<td>33</td>
<td>12411</td>
<td>29.4</td>
<td>0.84</td>
<td>212.6</td>
</tr>
<tr>
<td></td>
<td>1.23</td>
<td>1.00</td>
<td>4.04</td>
<td>36</td>
<td>78</td>
<td>22</td>
<td>10491</td>
<td>35.2</td>
<td>0.92</td>
<td>194.5</td>
</tr>
<tr>
<td></td>
<td>1.50</td>
<td>1.00</td>
<td>4.94</td>
<td>41</td>
<td>96</td>
<td>22</td>
<td>1112</td>
<td>48.5</td>
<td>0.91</td>
<td>213.4</td>
</tr>
<tr>
<td></td>
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<td>1.00</td>
<td>4.93</td>
<td>16</td>
<td>46</td>
<td>11</td>
<td>2216</td>
<td>57.5</td>
<td>0.97</td>
<td>182.6</td>
</tr>
<tr>
<td></td>
<td>2.35</td>
<td>1.08</td>
<td>6.26</td>
<td>54</td>
<td>62</td>
<td>4</td>
<td>1010</td>
<td>78.2</td>
<td>1.00</td>
<td>157.8</td>
</tr>
</tbody>
</table>
about 10 percent at the lowest mapped water level (0.15 m). During the decline, the configuration of the channel network changed from being dominated by up- and downstream connected channels to a heterogeneous system of only downstream connected and isolated water bodies (Table 2, Fig. 3A). In the bar-braided reach, complete active floodplain inundation was estimated to occur at water levels ≥ 2.5 m.

Fig 3. Inundation patterns for a bar-braided (A) and an island-braided (B) reach of the Tagliamento River. Water levels correspond to stage height readings from the San Pietro gauging station.
Inundated area also decreased monotonously with receding water level in the island-braided flood plain. At the highest mapped water level (2.35m) almost 80 percent of the active flood plain was inundated. At this water level, only vegetated islands and very high gravel bars protruded above the water in the active flood plain. At the lowest mapped water level (0.74 m), 20% of the active flood plain was covered by water. The configuration of the flood plain consisted of a matrix of exposed sediments with numerous isolated water bodies and primarily downstream connected channels. Several upstream-connected channels remained, however, even at the lowest mapped water levels (Fig. 2 and Fig. 3B). In the island-braided reach, complete active floodplain inundation was estimated to occur at water levels ≥ 3.0 m.

### Table 3. Linear regression models for dependent variables (Arcsine square root of inundation and shoreline length) and independent variables (stage height, squared stage height and reach) for a bar- and an island-braided reach of the Tagliamento River. Beta and probability values for significant variables are highlighted bold.

<table>
<thead>
<tr>
<th>Arcsine square root relative inundation</th>
<th>Shoreline length (m ha⁻¹)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Beta</td>
</tr>
<tr>
<td>2 variables stage</td>
<td>0.97</td>
</tr>
<tr>
<td>reach</td>
<td>-0.02</td>
</tr>
<tr>
<td>3 variables stage² reach</td>
<td></td>
</tr>
<tr>
<td>stage</td>
<td></td>
</tr>
<tr>
<td>1 variable stage</td>
<td>0.96</td>
</tr>
<tr>
<td>regression</td>
<td>y = 0.49x + 0.07</td>
</tr>
<tr>
<td>p &lt; 0.0001, R²_adj = 0.910</td>
<td></td>
</tr>
</tbody>
</table>

Linear regression of the arcsine square root of inundation with water level and reach indicated that the effect of the factor reach was insignificant (p > 0.70 Table 3). We therefore excluded this factor from further analysis and combined the
inundation data for both reaches (Fig. 4A). A new linear regression model with water level as the only independent variable showed a significant effect of water level on degree of inundation ($p < 0.0001$). The linear regression model was highly significant ($p < 0.0001$, $R^2_{adj} = 0.91$, Table 3) and was described by the equation:

$$y = 0.49x + 0.07,$$

Fig. 4 (A) Linear regression of relative floodplain inundation (arcsine square root transformed) versus water level readings from the San Pietro gauging station. (B) Second order polynomial regression of shoreline length versus water level readings. Solid circles represent the bar-braided reach; open circles represent the island-braided reach.
Where y represents the arcsine square root of inundation and x represents water level (m).

Shoreline lengths ranged from 28 m ha\(^{-1}\) (2.1 km km\(^{-1}\)) to 214 m ha\(^{-1}\) (16.3 km km\(^{-1}\)) in the bar-braided reach. Shoreline length peaked at intermediate water levels between 1.0 and 1.2 m (Fig. 4B). Shoreline length in the island-braided reach showed a similar range. The minimum value of 37 m ha\(^{-1}\) (3.0 km km\(^{-1}\)) occurred at complete active floodplain inundation. The maximum of 212 m ha\(^{-1}\) (17.3 km km\(^{-1}\)) occurred at water levels of between 1.0 and 1.65 m (Fig. 4B).

Table 4. Shoreline length in km km\(^{-1}\) (around mean water level) for regulated rivers and free-flowing rivers. * Values extrapolated from braiding index. † Values for headwater flood plain.

<table>
<thead>
<tr>
<th>River</th>
<th>Region</th>
<th>Morphology</th>
<th>Shoreline length (km km(^{-1}))</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tagliamento</td>
<td>NE-Italy</td>
<td>free-flowing</td>
<td>14.7</td>
<td>this study</td>
</tr>
<tr>
<td>Tagliamento(^+)</td>
<td>NE-Italy</td>
<td>free-flowing</td>
<td>7.9</td>
<td>(Arscott et al. 2000)</td>
</tr>
<tr>
<td>Danube</td>
<td>Austria</td>
<td>semi-natural</td>
<td>8.5</td>
<td>(Tockner et al. 2000a)</td>
</tr>
<tr>
<td>Rhône</td>
<td>Switzerland</td>
<td>channelized</td>
<td>2.1</td>
<td>(Tockner et al. 2002b)</td>
</tr>
<tr>
<td>Piave</td>
<td>NE-Italy</td>
<td>regulated</td>
<td>~3(^*)</td>
<td>(Surian 1999)</td>
</tr>
<tr>
<td>Val Roseg</td>
<td>Switzerland</td>
<td>free-flowing</td>
<td>16.5</td>
<td>(Malard et al. 1999)</td>
</tr>
<tr>
<td>Willamette</td>
<td>Oregon, USA</td>
<td>prior regulation</td>
<td>&gt;10</td>
<td>cited in Amoros (1996)</td>
</tr>
<tr>
<td>Willamette</td>
<td>Oregon, USA</td>
<td>after regulation</td>
<td>&gt;2.6</td>
<td>cited in Amoros (1996)</td>
</tr>
<tr>
<td>Gearagh</td>
<td>Ireland</td>
<td>free-flowing</td>
<td>26(^*)</td>
<td>(Brown et al. 1995)</td>
</tr>
<tr>
<td>Waimakarii</td>
<td>New Zealand</td>
<td>free-flowing</td>
<td>8(^*)</td>
<td>cited in Brown (1995)</td>
</tr>
</tbody>
</table>

A linear regression model with water level, squared water level, and reach as independent variables, and shoreline length as the dependent variable indicated that the effect of reach was insignificant (\(p = 0.10\), Table 3). Hence, data for both reaches were combined and the regression was repeated without reach as an independent variable. Both water level and squared water level were found to be highly significant factors (\(p < 0.001\) and \(p < 0.0001\), respectively, Table 3). The
second order polynomial regression curve (Fig. 4B) was highly significant ($\phi < 0.00009, R^2_{adj} = 0.783, \text{Table 3}$) and was described by the following function:

$$y = 87.8 - 65.8x^2 + 169.8x,$$

Where $y$ represents shoreline length (m ha$^{-1}$) and $x$ represents water level (m).

Relationships between water level, inundated area, and shoreline length were used to convert water level data for the period between January 1996 and May 2000 into a time series for inundated area and shoreline length (compare Fig. 5 with Fig. 1B). Mapped water levels ranged from 0.15m to 2.35m and water level data for 1996-2000 ranged from 0.0m to 3.24m. Due to the linearity of the model, the simulated curve for inundated area strongly resembled the hydrograph (compare Fig 1A with Fig. 5A). Predicted shoreline lengths were high, averaging 171 m ha$^{-1}$ (12.8 km km$^{-1}$). Short-term (< 3 days each event) reductions in shoreline length occurred during major flood events. A duration curve of relative active floodplain inundation was used to illustrate inundation time during the 1996-2000 period (Fig. 6). The rate of increase in inundation was relatively stable during most of the time series with sharp increases during only about 10 percent of the time, when water levels were greater than 1.3 m (Fig. 6). A frequency distribution of simulated shoreline length is also plotted in Figure 6. Shoreline length was low at minimum and at maximum water levels. Maximum shoreline length occurred at a water level of approximately 1.3 m, which was exceeded 10% of the time. Simulated shoreline length was high (150-200 m ha$^{-1}$, 11-15 km km$^{-1}$ see also Fig. 5B) for most of the time (>90%).

**Discussion:**

The importance of the flood pulse for the ecology of floodplain ecosystems has been well documented (e.g. Junk et al. 1989, Bayley 1991, Tockner et al. 2000b). Information about inundation dynamics across a range of water levels is needed to better understand flow and flood dynamics as factors shaping flood plains and their ecology (Benke et al. 2000, Tockner et al. 2000b, Tockner et al. 2002b). The present
paper shows that availability of aquatic habitats and the ecologically important aquatic-terrestrial interface (Naiman and Décamps 1997, Ward et al. 1999b) are influenced by frequent minor water level fluctuations. This supports the hypothesis that in addition to the flood pulse, flow pulses below bankfull also constitute a major physical variable determining ecological processes in floodplain systems (Malard et al. 1999, Tockner et al. 2000b).

Fig. 5. Predicted inundated area (A) and predicted shoreline length (B) for the period January 1996 through May 2000, for a bar- and an island-braided reach of the Tagliamento River. Because the regressions for the bar-braided and the island-braided reaches were not significantly different, data from both reaches are combined here.
This study has shown complex inundation patterns to exist within the active flood plain of a large semi-natural Alpine river. Most of the time the aquatic component of the flood plain was a mosaic of braided channels (primary, secondary, and tertiary), alluvial channels, and isolated waterbodies. An increase in water level temporarily shifted this configuration towards a dominance by channels. Decreases in water level shifted the configuration toward isolated and downstream-connected waterbodies. The larger number of vegetated islands (Ward et al. 1999a) and large wood deposits (Gurnell et al. 2000b, Van der Nat et al. 2001) in the island-braided reach resulted in a higher abundance of isolated waterbodies such as scour holes (Table 2). These aquatic 'islands', temporary or permanent, are important habitats and refuges for aquatic invertebrates (Ute Karaus, personal communication, Homes et al. 1999), fishes (Galat et al. 1997, Galat et al. 1998), and amphibians (Morand and Joly 1995, Griffiths 1997, Tockner et al. 2002a).

Fig. 6. Duration curve (percentage of time at which value equaled or exceeded) for floodplain inundation and a frequency distribution of shoreline length in two braided flood plains of the Tagliamento River. Because the regressions for the bar-braided and the island-braided reaches were not significantly different, data from both reaches are combined here.
Although the bar- and island-braided reaches differed considerably in morphology, their inundation dynamics and shoreline lengths were similar. The sinuosity of the main channel in both reaches remained low and constant (Table 2). In complex braided systems, sinuosity is not as suited to describe shoreline complexity as it is in meandering or single channel systems. In the Danube downstream of Vienna, sinuosity was used to quantify shoreline complexity within the main stem (Schiemer et al. 2001). In our system, total shoreline length, the braiding index, upstream surface hydrologic connectivity (SC), and total area of isolated waterbodies best described the relationship between water level and complexity. The positive relationship between braiding index, SC, and water level indicates a potentially increasing complexity of the aquatic component of the active flood plain with rising water levels (up to a certain point). However, this increase is partly counteracted by a decrease in complexity due to a diminishing area of isolated waterbodies.

The water level above which the most rapid morphological changes to the flood plain occur in the Tagliamento River was estimated to be three times the average stage height (Arscott 2002). The average stage height during the period of simulation was 0.79m. According to our model considerable morphological changes would thus occur at inundations of 89% and more. Five such events have occurred during the period of simulation (Fig. 5A).

The linear relationship between water level and inundated area in this temperate unregulated river concurs with that found by Benke et al. (2000) for the subtropical Ogeechee River (SE USA). In contrast, in the hydrologically dynamic but morphologically regulated Danube flood plain of the Alluvial Zone National Park, Austria inundated area increased stepwise with increasing water level (Tockner et al. 2000a). Therefore, below mean flow, inundated area of the flood plain increased monotonously but slowly through seepage inflow. Above mean water level, overbank flow resulted in a sudden increase of the inundated area. The relationship between water level and inundation was also documented in tropical
rivers with predictable flooding regimes. For example, in a Pantanal wetland a linear relationship was found between monthly estimates of inundated area and the flood stages of the previous two months (Hamilton et al. 1996). Similar observations from the Amazon River revealed a second order polynomial relationship between monthly estimates of inundated area and river stage (Sippel et al. 1998).

The second-order polynomial relationship between water level and shoreline length was significant and similar to the unimodal pattern found by Tockner et al. (2000a) for the Danube (Austria). The maximum shoreline length found there (122 m ha\(^{-1}\), 8.5 km km\(^{-1}\)) was considerably lower than the maximum found in this study (213 m ha\(^{-1}\), 17.3 km km\(^{-1}\)).

Applying our predictive regressions from one year of data, to 5 years of water level data brings forward several complications. In a system that is as morphologically dynamic as the Tagliamento River, it is possible that channel bed elevations change frequently. Changes in bed elevation would result in a change in the stage-discharge relationship. However, there are indications that this relationship does not alter too dramatically in our system. Analysis of the hydrograph by Arscott et al. (2000) indicated that the yearly average stage height decreased 39 cm over the period of 1982 through 1999. A decline of 2 centimeters per year should not have too serious implications for the stage-discharge relationship over a five-year period. Secondly, the gauging station at San Pietro is located at a stable bedrock constriction, making changes in the stage-discharge relationship less likely to occur. Thirdly, large flood events dramatically change the configuration of the active flood plain in the Tagliamento. In our reaches as much as 45 percent of all aquatic habitats was turned over by a large flood pulse (Arscott 2002). However, at the same time the composition of landscape elements (e.g. water, gravel bars) in the active flood plain remains more or less the same. Flood pulses create as many new channels, alluvial channels, backwaters, and pools as they destroy. Had the flood pulses altered the stage-discharge relationship, maps
made at the same water level before and after a flood pulse would not have shown similar compositions.

Some water level values from the five-year simulation were outside the range of actually measured values. Because we have extrapolated in the predictive model, care has to be taken interpreting model outcomes. However, the simulation of inundated area in the study reaches did show a number of important trends. During 50 percent of the time inundated area was lower than 30 percent (Fig. 6). Consequently, during most of the year the active flood plain is a mosaic of aquatic patches within a matrix of exposed sediments. Complete or near-complete inundation of the active flood plain occurred 3-4 times per year; at these times all waterbodies were connected by surface-water flow, allowing mixing of populations that were isolated during much of the year. Animals such as young fish need these large inundations to disperse from their spawning grounds to other parts of the river (Fernandes 1997). Additionally, members of various fish families in temperate European and North American rivers require flowing water during some life stages (Galat and Zweimüller 2001).

Shoreline length remained high most of the time, in contrast to highly regulated rivers that have much lower shoreline lengths (Table 4). Minimum shoreline lengths, comparable to the values regulated rivers feature throughout the year, only occur during large spates. With the decreasing availability of free-flowing rivers, the shoreline interface has become an endangered fluvial landscape element (e.g. Pinay et al. 1990).

The fact that braided reaches of the Tagliamento River sustain an abundance of shoreline habitat throughout the year is of considerable ecological importance. Shorelines provide specific habitats for a variety of specialized biota (e.g. Salo et al. 1986, Décamps et al. 1987, Risser 1990, Reich 1994, Wintersberger 1996). High abundance of shoreline is important for sustaining riparian biodiversity (Risser 1990, Nilsson et al. 1997, Maiolini et al. 1998, Luken and Bezold 2000). It is also important for facilitating biotic and abiotic interactions between the aquatic and the
terrestrial part of the flood plain (Bardgett et al. 2001, Ward and Wiens 2001). The fact that these habitats are important to rare alluvial flora and fauna makes preservation of our remaining free-flowing rivers a significant task.

Acknowledgements:
The authors thank the subject editor Dr. Naiman and two anonymous reviewers for comments that resulted in considerable improvements to this paper. We also thank David B. Arscott, Michael T. Monaghan, and Sophie Karrenberg van der Nat for fruitful discussions and suggestions made to earlier versions of this manuscript. We are grateful to several people who helped with field logistics, especially Diego Nebuloni and Grazia Varra. Special thanks to Mr. Alberto Deana, Direzione Regionale dell' Ambiente, and Mr. Francesco Baruffi, Autorita di Bacino dei Fiume Isonzo, Tagliamento, Livenza, for providing hydrological data. Grant 0-20572-98 from the Forschungskommission of the ETH Zürich supported this research.

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...En aquel Imperio, el Arte de la Cartografía logró tal Perfección que el mapa de una sola Provincia ocupaba toda una Ciudad, y el mapa del Imperio, toda una Provincia. Con el tiempo, esos Mapas Desmesurados no satisfacieron y los Colegios de Cartógrafos levantaron un Mapa del Imperio, que tenía al tamaño del Imperio y coincidía puntualmente con él.

(Jorge Luis Borges, from Del rigor en la ciencia, 1960)
Habitat turnover in a braided river (Tagliamento, NE-Italy).

Abstract:

Turnover rates and age distribution of habitats were investigated in the active zone of a bar-braided and an island-braided reach of the Tagliamento River (NE-Italy). Between September 1999 and January 2002 6 habitat types were delineated with a differential global positioning system on 5 dates following floods of different magnitude. Overlay maps were employed to calculate age and relative turnover of habitats. We established exponential decay rates ($k$-values) for islands and major aquatic habitats. Turnover rates of all aquatic habitats combined were up to 70% between survey dates in the bar-braided reach, with a cumulative rate of nearly 90% over the 2.5-year period. Turnover values in the island-braided reach were lower with a cumulative turnover of 75% after 2.5 years. Significant exponential decay relationships were established for upstream and downstream connected channels (named channels), downstream connected, lotic channels (named alluvial channels), downstream-connected lentic backwaters (named backwaters), and pools in the bar-braided reach. Half-lives were particularly short for backwaters and pools. In the island-braided reach significant relationships existed for channels and alluvial channels. The half-lives of channels and alluvial channels increased with the presence of vegetated islands. Relative habitat composition within the active corridor remained almost constant, supporting the applicability of the shifting mosaic steady state model to the active zone of braided river ecosystems.

Introduction:

River corridors in their natural state are characterized by a diverse array of habitats including surface waters, the alluvial aquifers, riparian systems, and geomorphic features (Ward et al. 2002). This diversity of habitats is of key importance to a wide array of aquatic, amphibious, and terrestrial organisms, including aquatic macroinvertebrates (Homes et al. 1999, Benke 2001), fishes (Galat
et al. 1997, Galat et al. 1998), carabid beetles and spiders (Bonn et al. 2002), woody vegetation (Décamps 1996, Karrenberg et al. 2002), and amphibians (Morand and Joly 1995, Griffiths 1997, Tockner et al. submitted). In the developed world few rivers remain that still exhibit the natural dynamics of large-scale geomorphological and ecological processes, responsible for high habitat diversity. Because of the importance of intact river corridors to endangered species, understanding and preservation of these reference systems are significant future tasks.

In their natural state river corridors are complex physical features that border rivers in anastomosing, braided, and meandering reaches. Disturbances (sensu Pickett and White 1985) are important factors controlling community structure and ecosystem functions (e.g. Stanford and Ward 1983, Poff and Ward 1989, Sparks et al. 1990, Townsend et al. 1997). Flooding represents the major physical disturbance occurring along river corridors (Junk et al. 1989, Puckridge et al. 1998, Swanson et al. 1998, Tockner et al. 2000, Arscott et al. 2002). Floods of varying frequency and magnitude rework the morphology of flood plains and river channels at various spatial scales (Hughes and Rood 2001, Ward et al. 2002), making them highly dynamic. Overbank flooding (flood pulses sensu Junk et al., 1989) may occur seasonally or unpredictably (Poff and Ward 1989), but rivers also experience smaller, more frequent water level fluctuations ("flow pulses" sensu Tockner et al., 2000). Although not responsible for large morphological restructuring, these flow pulses are important for creating and maintaining habitat heterogeneity and for ecosystem processes (Benke et al. 2000, Tockner et al. 2000, Nat van der et al. 2002).

The effect of floods on river channel and floodplain structural features has been documented using remote sensing techniques (Mertes 2002). Aerial photography and historical maps have been used to study long-term changes of channel structure (e.g. Mosley 1982, Warburton et al. 1993, Gilvear et al. 1995, Zah et al. 2001, Poole et al. 2002). The recent availability of Global Positioning Systems (GPS) allows collection of precise spatial data that is cost-efficient, weather independent, and allows simultaneous collection of other data (Brasington et al. 2002).
2000, Arscott et al. 2002). Although flood dynamics have been well studied in river ecosystems little information is available on habitat turnover rates in natural floodplain rivers (but see Kollmann et al. 1998, Gurnell et al. 2001, Arscott et al. 2002).

The overall goal of the present investigation was to determine turnover rates of aquatic and terrestrial habitats in the active zone of near-natural braided river systems. The investigation was conducted in the Tagliamento River (NE-Italy), which provides the opportunity to study the effect of flooding on geomorphic features under relatively natural conditions (Ward et al. 1999). The following research questions are addressed: (1) How stable or unstable are terrestrial and aquatic habitats in the active zone of dynamic braided rivers? (2) What is the age-distribution of habitats in the active corridor? (3) Does habitat turnover occur continuously or only during large spates? (4) Does the presence of vegetated islands influence the turnover of aquatic habitats in the active zone? (5) Does the shifting mosaic steady state model (Bormann and Likens 1979) apply to the dynamic active zone of braided river systems?

Study Site:

The Tagliamento River in NE-Italy is a 7th order gravel-bed river with a catchment area of 2580 km² and a total length of 172 kilometers. Approximately 70% of its catchment is located in the Southern Alps (Fig. 1A). It flows to the Adriatic Sea traversing constrained (mainly single-thread), braided, and meandering reaches. The river's active zone consists of 3 major habitats: water, exposed riverine sediments, and vegetated islands, fringed by continuous riparian woodland (Ward et al. 1999, Gurnell et al. 2000). Active zone width is up to 1.5 km.

Despite water abstraction and channelisation of the lowermost 25 kilometers, the Tagliamento has a relatively low level of human intervention. Consequently, its flood regime and morphological character are essentially natural (Müller 1995, Ward et al. 1999). The hydrological regime is pluvio-nival with frequent flashy flood
Fig. 1. (A) Location of the Tagliamento catchment in Italy (I) near Austria (A) and Slovenia (SL), and the location of the bar-braided and island-braided reaches. (B) Stage height during the investigation period measured at the San Pietro gauging station 1.5 km downstream of the island-braided reach. Arrows indicate times of the mapping surveys.

and flow pulses (Fig. 1B). Floods can occur any time of year but are most likely in spring and autumn but their timing has an extremely low degree of predictability (Campolo et al. 1999). Precipitation in the catchment has a high degree of spatial variability (Tockner et al.). The average discharge at the location where the river enters the coastal plains is approximately 90 m$^3$ s$^{-1}$. The 2, 5, and 10-year floods have estimated magnitudes of 1100, 1600, and 2150 m$^3$ s$^{-1}$, respectively (Gurnell et
More detailed description of the catchment characteristics and hydrodynamics are available in Ward et al. (1999a) and Gurnell et al. (2000).

We conducted this investigation in 2 morphologically distinct reaches hereafter referred to as the bar-braided reach and the island-braided reach (reaches III and IV from Arscott et al. (2000)). Under base flow conditions large areas of exposed sediment and multiple channels dominated the bar-braided reach (downstream of river kilometer 74). The bar-braided reach included only 2 vegetated islands. The island-braided reach was located upstream of a narrow bedrock constriction near Pinzano (Fig. 1A). This reach included a complex channel network and ca. 20 vegetated islands. Both reaches included pools, backwaters, and actively-flowing channels, but exposed river-deposited sediments (mainly gravel with some sand) represented the largest proportion of landscape cover (Nat van der et al. 2002).

**Methods:**

During the 2.5-year investigation period both reaches were mapped 5 times (September 1999, May 2000, February 2001, May 2001, and December 2001/January 2002). All but one of the surveys were conducted within a relatively small range of river stage, measured 1.5 km downstream of the island-braided reach (Table 1) in order to minimize the effect of stage on habitat mapping (Nat van der et al. 2002). The mapping was generally conducted within 2 days for each reach, with the exception of the map for September 1999 in the island-braided reach, which was made on 3 nonconsecutive days subject to only small water levels fluctuations. The reported stage of 1.16m for September 1999 is the average stage recorded for the three survey days. All habitats and the banks of the active zone were delineated by walking along them carrying the receiver of a Trimble TCS1 differential Global Positioning System (dGPS). Positions were recorded at 5-second intervals and later post corrected using base station data to obtain a precision of ≤ 0.3 meter (see also Arscott et al. 2002, Nat van der et al. 2002). The base station
was located less than 5 km from both reaches and remained stationary during all surveys to record variations in triangulation due to satellite movement. The spatial data were converted to maps using Arc-Info 7.1 and Arc-View GIS 3.1 software (Environmental Systems Research Institute Inc., Redlands, California, USA).

Table 1: Water level (m), mapping date, and surface cover (%) of habitat types within the active zone of 2 reaches of the Tagliamento.

<table>
<thead>
<tr>
<th>Reach</th>
<th>Stage (m)</th>
<th>Date (mm-yy)</th>
<th>Channels</th>
<th>All. ian</th>
<th>Back waters</th>
<th>Tributary</th>
<th>Pools</th>
<th>Exp sed.</th>
<th>Veg. isl.</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bar-braided</td>
<td>1.00</td>
<td>09-99</td>
<td>30.3</td>
<td>4.1</td>
<td>0.3</td>
<td>0.3</td>
<td>61.9</td>
<td>3.1</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.98</td>
<td>05-00</td>
<td>20.1</td>
<td>1.6</td>
<td>0.3</td>
<td>1.8</td>
<td>73.1</td>
<td>3.0</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.65</td>
<td>02-01</td>
<td>26.3</td>
<td>2.4</td>
<td>0.2</td>
<td>0.2</td>
<td>68.7</td>
<td>2.2</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.73</td>
<td>05-01</td>
<td>26.1</td>
<td>2.5</td>
<td>0.3</td>
<td>0.3</td>
<td>68.5</td>
<td>2.2</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.15</td>
<td>12-01</td>
<td>7.9</td>
<td>1.6</td>
<td>0.1</td>
<td>1.3</td>
<td>86.9</td>
<td>2.2</td>
<td></td>
</tr>
<tr>
<td>Island-braided</td>
<td>1.16</td>
<td>09-99</td>
<td>32.2</td>
<td>5.8</td>
<td>0.4</td>
<td>0.4</td>
<td>50.4</td>
<td>10.6</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.98</td>
<td>05-00</td>
<td>24.8</td>
<td>2.4</td>
<td>0.5</td>
<td>0.4</td>
<td>61.6</td>
<td>9.8</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.65</td>
<td>02-01</td>
<td>24.2</td>
<td>3.0</td>
<td>0.3</td>
<td>0.2</td>
<td>64.0</td>
<td>7.5</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.70</td>
<td>05-01</td>
<td>22.6</td>
<td>4.0</td>
<td>0.1</td>
<td>0.2</td>
<td>65.2</td>
<td>6.9</td>
<td></td>
</tr>
<tr>
<td></td>
<td>0.0</td>
<td>12-01</td>
<td>11.2</td>
<td>5.3</td>
<td>0.2</td>
<td>0.3</td>
<td>74.0</td>
<td>7.9</td>
<td></td>
</tr>
</tbody>
</table>

Several habitats were distinguished: Channels were defined as all lotic waterbodies with upstream and downstream connection (excluding tributaries), alluvial channels were lotic waterbodies with only downstream connections to other channels. Backwaters also had only a downstream connection but exhibited a more lentic character. Pools were defined as disconnected waterbodies larger than 2 m². Vegetated islands were defined as areas of woody vegetation, with an maximum length of > 20m, where more than 50 percent of the trees were taller than 2 meters, and the projected surface cover of the vegetation was > 50 percent. The matrix containing all other habitats was exposed sediment. All habitats were treated as
polygons and the maps were queried to determine the areal extent of each habitat during all surveys.

The complexity parameters sinuosity, braiding-index (Friend and Sinha 1993), number of nodes, and number of pools were determined. Sinuosity (P) was defined as $L_{cmax}/L_R$, where $L_{cmax}$ is the mid-channel length of the widest channel through the reach, and $L_R$ is the overall, straight line, length of the reach (Friend and Sinha 1993). The braiding index (B) was defined as $L_{ctot}/L_{cmax}$, where $L_{ctot}$ is the sum of the lengths of all upstream connected channels within a reach. Nodes are ecotones within aquatic systems defined as the intersections between waterbody thalwegs (Arscott et al. 2000). They include intersections with and within waterbodies with only upstream or only downstream connections. Upstream surface hydrological connectivity (SC) was calculated from all maps following Malard et al. (2000), by dividing the area of channels fed by surface water ($A_c$) by the total area of channels with surface flow ($A$).

In order to establish the turnover of habitat types we used Arc-Info to make union overlay coverages of the obtained maps. To establish the degree of spatial overlap we combined each map with every possible subsequent map, resulting in 10 overlay maps from the 5 original maps for each reach. For all overlay maps we established for each polygon what habitat type it was in the first and in the subsequent map. In this way we were able to make confusion tables that showed which percentage of a certain habitat type was turned over into another. Secondly it showed which percentage of a habitat type did not change between mapping surveys. Riparian forest was excluded from these analyses.

The percentages of habitats that did not change were arcsine square root transformed (Zar 1999) and plotted against time. Non-linear estimation with an exponential decay function (Statistica 5.1, StatSoft, Tulsa, Oklahoma, USA) was used to determine the turnover rate and half-life of habitat types. The regression's y-axis intercept was forced through $\frac{1}{4}\pi$, the arcsine square root of 1. A union overlay map of all 5 maps was constructed using Arc-Info. Age distribution of
aquatic habitats was determined by establishing what proportion of the area of habitats in the last map was formed at which point in time.

Results:

The composition of individual habitats within each reach remained relatively constant throughout the investigation period (2.5 years). Exposed sediment was the dominant habitat in both reaches (Fig.2, and Table 1). The relative abundance of channels varied between 20 and 30 percent, except at low water levels when it was about 10 percent. Relative abundance of alluvial channels ranged from 1 to 5 percent. Backwaters and pools covered less than 2 percent of the active zone area. The cover of vegetated islands in the island-braided reach decreased from 10 to 7 percent during the investigation period.

Sinuosity was nearly constant throughout the investigation period (Table 2). Braiding-index values showed more variability, being lowest during mapping surveys at low water level (Table 2). The Braiding-index in the bar-braided reach was always higher than in the island-braided reach. Number of nodes varied with time, as did the number of pools. The number of pools was higher in the island-braided reach than in the bar-braided reach. Surface hydrologic connectivity was always very high (Table 2).

In the bar-braided reach aquatic habitat turnover between surveys varied through time (Fig. 3). The largest flood (stage 3.5m) turned over more than 70% of aquatic habitats (Table 3). The other bankfull flood (stage 2.35m) turned over 62.8 percent of aquatic habitats. Even the smallest flood (stage 1.5m) restructured almost 35 percent of the aquatic habitat area (Table 3). The cumulative turnover of aquatic habitats for the 2.5-year period was almost 90 percent (Table 3 and Fig. 3 bottom left). Turnover of the bar-braided reach's 2 vegetated islands was less pronounced. The largest flood (stage 3.5m) eroded large areas (7.3 ha) of riparian forest and islands (Fig. 3). Turnover of vegetated island habitat reached 27 percent for the largest flood and 29 percent in over the investigation period.
In the island-braided reach the turnover of aquatic habitats also varied with time and flood magnitude (Fig. 4). Turnover rates were lower than in the bar-braided reach. The 2 largest floods turned over 48 percent of aquatic habitat area (Table 3), whereas the smallest flood turned over 44 percent of aquatic habitat area. The other small flood turned over 22 percent of aquatic habitat area. The cumulative turnover for the 2.5-year period was more than 75 percent (Table 3 and Fig. 4 bottom left). The largest flood eroded 7.5 hectares of woody vegetation in the island-braided reach. Turnover for vegetated island habitat was higher in the island-braided reach than for the 2 islands in the bar-braided reach. Maximum turnover was 34.5 percent and the cumulative total turnover for the 2.5-year time-period was almost 40 percent (Table 3).

The confusion tables indicated that in both reaches the largest proportion of turnover was a shift of other habitats into exposed sediment and vice versa (Table 4). Exponential decay regressions of surviving habitat area in the bar-braided reach indicated significant relationships for 4 habitat types (Table 5), although the regressions for backwaters and pools were difficult to interpret. The regressions for exposed alluvium and vegetated islands were not significant. In the island-braided reach only channels and alluvial channels showed significant decay rates (Fig. 5,
Table 5). The turnover rate for both habitat types was slower than in the bar-braided reach.

Table 2: Water level (m), mapping date, and complexity descriptors within the active zone of 2 reaches of the Tagliamento. * surface hydrologic connectivity

<table>
<thead>
<tr>
<th>Reach</th>
<th>Stage (m)</th>
<th>Date</th>
<th>Sinuosity</th>
<th>Braiding index</th>
<th>Total nodes</th>
<th>Pools no.</th>
<th>SC*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Bar-braided</td>
<td>1.00</td>
<td>09-1999</td>
<td>1.08</td>
<td>3.66</td>
<td>50</td>
<td>12</td>
<td>0.88</td>
</tr>
<tr>
<td></td>
<td>0.98</td>
<td>05-2000</td>
<td>1.10</td>
<td>4.77</td>
<td>117</td>
<td>24</td>
<td>0.93</td>
</tr>
<tr>
<td>Island-braided</td>
<td>0.65</td>
<td>02-2001</td>
<td>1.04</td>
<td>5.56</td>
<td>54</td>
<td>4</td>
<td>0.91</td>
</tr>
<tr>
<td></td>
<td>0.73</td>
<td>05-2001</td>
<td>1.13</td>
<td>6.17</td>
<td>93</td>
<td>35</td>
<td>0.91</td>
</tr>
<tr>
<td></td>
<td>0.15</td>
<td>12-2001</td>
<td>1.24</td>
<td>2.17</td>
<td>33</td>
<td>12</td>
<td>0.83</td>
</tr>
<tr>
<td></td>
<td>1.16</td>
<td>09-1999</td>
<td>1.07</td>
<td>2.27</td>
<td>56</td>
<td>24</td>
<td>0.85</td>
</tr>
<tr>
<td></td>
<td>0.98</td>
<td>05-2000</td>
<td>1.04</td>
<td>4.42</td>
<td>113</td>
<td>44</td>
<td>0.91</td>
</tr>
<tr>
<td></td>
<td>0.65</td>
<td>02-2001</td>
<td>1.07</td>
<td>2.79</td>
<td>59</td>
<td>59</td>
<td>0.89</td>
</tr>
<tr>
<td></td>
<td>0.70</td>
<td>05-2001</td>
<td>1.00</td>
<td>3.64</td>
<td>64</td>
<td>86</td>
<td>0.85</td>
</tr>
<tr>
<td></td>
<td>0.00</td>
<td>01-20021</td>
<td>1.05</td>
<td>2.11</td>
<td>74</td>
<td>54</td>
<td>0.68</td>
</tr>
</tbody>
</table>

The age of habitat area indicated that channels had a relatively equal age distribution in both reaches (Fig. 6). The island-braided reach also contained some older patches of alluvial channel and pool habitat. Alluvial channels and pools in the bar-braided reach and backwaters in both reaches showed an age distribution that was strongly skewed to the left (Fig. 6), indicating that the majority of the habitat area was recently formed. Vegetated island area was mostly older than our first survey although some new patches did establish.
Fig. 3 Overlay maps resulting from dGPS data collection on 5 dates for the bar-braided reach, with turnover of aquatic (left column) and terrestrial (right column) habitats.
Fig. 4 Overlay maps resulting from dGPS data collection on 5 dates for the island-braided reach, with turnover of aquatic (left column) and terrestrial (right column) habitats.
Table 3: Turnover (%) of aquatic habitats and vegetated islands by floods of different magnitude between individual mapping dates, and cumulative rate over the 2.5-year period.

<table>
<thead>
<tr>
<th>Period</th>
<th>Flood stage (m)</th>
<th>Turnover aquatic habitats (%)</th>
<th>Turnover vegetated islands (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Bar-braided</td>
<td>Island-braided</td>
<td>Bar-braided</td>
</tr>
<tr>
<td>Sep99 → May00</td>
<td>2.35</td>
<td>62.8</td>
<td>48.1</td>
</tr>
<tr>
<td>May00 → Feb01</td>
<td>3.50</td>
<td>71.6</td>
<td>48.0</td>
</tr>
<tr>
<td>Feb01 → May01</td>
<td>1.60</td>
<td>34.9</td>
<td>21.6</td>
</tr>
<tr>
<td>May01 → Dec01/Jan02</td>
<td>1.50</td>
<td>68.1</td>
<td>43.7</td>
</tr>
<tr>
<td>Entire period</td>
<td>-</td>
<td>89.2</td>
<td>75.2</td>
</tr>
</tbody>
</table>

Table 4: An example of a confusion table for the island-braided reach, showing which proportion of habitat area in had been converted by one flood event. CH = channels, AC = alluvial channels, BW = backwaters, TRI = tributaries, PL = pools, GRV = bare alluvium, Isl = vegetated islands, and RF = riparian forest.

<table>
<thead>
<tr>
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<th>May 2000</th>
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<th>AC</th>
<th>BW</th>
<th>TRI</th>
<th>PL</th>
<th>GRV</th>
<th>ISL</th>
<th>RF</th>
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<td>93.6</td>
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</tbody>
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Fig. 5 Exponential decay regression curves for the arcsine square root of remaining habitat versus time (months). Only curves for habitat types having a significant regression are shown.

Discussion:

The active zones of natural river corridors are dynamic ecosystems that are very heterogeneous both in space and in time (Ward et al. 2002). Beside temperature (Arscott et al. 2001, Claret et al. 2002), flood and flow pulses are the primary mechanism driving the biotic community and the state of the physical environment in river corridor ecosystems (Richards et al. 2002) and connecting the river to the flood plain (Junk et al. 1989, Benke et al. 2000). By removing and
depositing Coarse Particulate Organic Material (CPOM) (Steiger et al. 2001), Large Woody Debris (LWD) (Gurnell et al. 2002, Nat van der et al. submitted) and sediments, flooding creates a mosaic of patches in space and time with different ecological succession trajectories (Amoros et al. 1996).

Table 5: $k$-values for the turnover of habitat types and the $P$ and $R^2$ values for exponential decay regression lines. Significant values are indicated in bold and were used to establish half-life ($t_{1/2}$) expectancies.

<table>
<thead>
<tr>
<th>Reach</th>
<th>Habitat type</th>
<th>$k$-value</th>
<th>$P$</th>
<th>$R^2$</th>
<th>$t_{1/2}$ (months)</th>
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<tbody>
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<td>Bar-braided</td>
<td>Channels</td>
<td>-0.28</td>
<td>&lt; 0.006</td>
<td>0.917</td>
<td>4.1</td>
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<td>Alluvial channels</td>
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<td>&lt; 0.003</td>
<td>0.959</td>
<td>≤ 7</td>
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<tr>
<td></td>
<td>Backwaters</td>
<td>-0.81</td>
<td>&lt; 0.001</td>
<td>0.998</td>
<td>≤ 7</td>
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<tr>
<td></td>
<td>Pools</td>
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<td>&lt; 0.03</td>
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<td>≤ 7</td>
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<td>&lt; 0.83</td>
<td>0.917</td>
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<tr>
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<td>-0.34</td>
<td>&lt; 0.06</td>
<td>0.751</td>
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<tr>
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<td>Exposed alluvium</td>
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<td>&lt; 0.11</td>
<td>0.730</td>
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<tr>
<td></td>
<td>Vegetated islands</td>
<td>-0.09</td>
<td>&lt; 0.11</td>
<td>0.777</td>
<td>-</td>
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</table>

Hydrological regimes of catchments vary with climate, location, geology, topography and vegetation cover. Proper description of a river's hydrological regime should include magnitude, duration, timing, and predictability of floods (Poff and Ward 1989, Puckridge et al. 1998). However, due to a limited rating curve for the Tagliamento (Campolo et al. 1999), precise measurements of hydrological variability could not be estimated (Arscott et al. 2002). Flood probabilities are highest in autumn and spring and flood seasons are predictable to some extent (Arscott et al. 2002). Floods with a peak discharge magnitude exceeding 2.5 times
the average flow occur more than 3 times per year, whilst smaller flow pulses occur with a much higher frequency.

We used stage data rather than discharge because discharge data were not available for the investigation period. A potential problem using stage data from a morphologically dynamic river, is that channel bed elevations change frequently; these changes could affect the stage-discharge relationship. The gauging station from which the stage data were obtained is located at a stable bedrock constriction, which helps to minimize instability in the stage-discharge relationship. However, there are indications that this relationship has altered slowly, with an apparent bed degradation of 2 cm per year over the last 20 years in this reach (Arscott et al. 2000). This should not have serious implications over the 2.5-year study period reported in this paper.

In general turnover of habitats in river corridors results from several processes. Flood pulses (Junk et al. 1989) can destroy, create, or shift habitats. Aquatic and terrestrial habitats also experience cycles of expansion and contraction and minor morphological changes due to flow pulses (Benke et al. 2000, Tockner et al. 2000, Nat van der et al. 2002). The pulses considered in this investigation were of different magnitudes and mapping occurred across a range of water levels of about 1 m. Hence, the turnover rates reported herein are the cumulative result of both flood and flow pulses.

Although delineation of all habitats occurred in the field, the classification occurred during the GIS analysis. As a result habitats have been defined based on shape and degree of connectivity, rather than on ecological traits. A further complication for the interpretation of our results is that we ignored habitats outside the active zone of the river corridor. Turnover of aquatic and terrestrial habitats was high in both reaches and habitats were generally very young. In just 2.5 years more than 75% of the aquatic area and 29% of vegetated islands were restructured. The smaller turnover of aquatic habitats in the island-braided reach can be attributed to the protective impact of islands on downstream habitats, reducing the
impact of floods, and also the association of islands with particular marginal habitats such as scour pools. Flood magnitude seemed to largely control the degree of turnover in both reaches, whereby time played a role as well. The 2nd smallest flood (1.6m stage) overturned less habitat area than the smallest (1.5m stage). This

Fig. 6 Aquatic habitat age distribution based on area for channels, alluvial channels, backwaters, pools, and vegetative islands for the active zones of the bar-braided and island-braided reaches in the Tagliamento River.
controversy probably results from the longer time-period between the lowest flood and the one preceding it. The effect that time since the last flood seems to have suggest that continuous cut and fill processes also play a part in the turnover.

Significant exponential decay regressions existed for channels, alluvial channels, backwaters, and pools in the bar-braided reach, and for channels and alluvial channels in the island-braided reach. The $k$-values were smaller (more negative) in the bar-braided reach than in the island-braided reach. This again points towards the increased stability of aquatic habitats in the presence of vegetated islands. Half-life ($t_{1/2}$) calculations for backwaters and pools were very difficult to interpret because almost none of these habitats survived floods. Since the shortest time-interval between surveys was 3 months, the significance of outcomes of less than a month are questionable. Empirical estimation of pool half-life also contradicts our observations. We argue that the half-life of both backwaters and pools is in any case less than 7 months, the approximate point where the regression lines parallel the x-axis (Fig. 5). When interpreting half-life values it is important to realize that they are area-based. Flow pulses may slightly shift the position of a habitat, without the returning observer noticing this. In such a shift new habitat area is created and some proportion of its old area is turned over into a different habitat (usually bare sediment or water). Habitat area is thus turned over, although the specific habitat still exists. The short life span of backwaters is most likely due to flow pulses and the continuous cut and fill processes that the channels, to which they are attached, experience.

In the past, vegetated islands characterized the river corridors of Europe's forested biome. These islands played an important part in developing and maintaining heterogeneous channel forms (Gurnell and Petts 2002). Floods and the characteristics of landforms that are shaped by floods affect riparian vegetation (Bendix and Hupp 2000, Karrenberg et al. 2002) in turn. In the Tagliamento River vegetated islands are still abundant and influence geomorphic processes. The understanding of island dynamics is in its infancy (but see Fetherston et al. 1995,
The exponential decay regressions for vegetated islands in the present study yielded no significant results. This was due to the short investigation period of only 2.5 years. Continued observation in the future will result in reliable half-life estimates for vegetated islands. Analysis of aerial photographs and tree ring analysis have indicated that vegetated islands in this system have a maximum life span of about 20 years and a half-life of about 8 years (Kollmann et al. 1998). The age of an island is inversely related to the risk of erosion. The erosion of vegetated areas in this investigation occurred in 2 ways; 1) continuous erosion of small areas of vegetation by channel migration, and 2) dramatic erosion during the largest flood (3.5m stage). Turnover of vegetated islands occurred mainly during the largest flood; the 2nd largest flood (2.35m stage) caused no dramatic erosion of vegetation, indicating the presence of a threshold for large-scale erosion. This supports the observation that flow thresholds are important factors influencing the geomorphic dynamics of river landscapes (Church 2002). Formation of vegetated islands and riparian forest occur by primary production and vegetation establishment often in association with large wood deposition (Gurnell et al. 2001), both causing new areas to meet the criteria we set for vegetated islands. The total area of vegetated islands in the island-braided reach decreased by 25 percent over just 2.5 years, an observation concurring with a model formulated by Gurnell and Petts (2002) that states that island dominated reaches go through cycles of island growth and decay.

Braiding-index values and sinuosity showed patterns very similar to those found by Arscott et al. (2002). The degree of braiding was always highest in the barbraided reach. At the lowest water levels channels became cut off at one end, which meant they were no longer incorporated into the braiding index (Friend and Sinha 1993). The high number of nodes at intermediate water levels supports the observation that system complexity is largest at these water levels (Nat van der et al. 2002). The high surface hydrologic connectivity (SC) in both reaches indicates a high potential for biotic and abiotic interaction between different parts of the active
zone, which are of great ecological importance (e.g. Tockner et al. 1999, Galat and Zweimüller 2001, Ward and Wiens 2001).

Turnover of habitats by erosive floods is a major way in which fluvial processes impact biodiversity (Tockner and Stanford 2002). The high degree of turnover and low stability of aquatic habitats documented herein could explain why benthic invertebrate taxa richness is relatively low (Arscott et al. in preparation). Most habitats being very young, may constrain habitation by species with perennial life cycles such as mollusks and dragon flies. The age distribution of aquatic habitat area (Fig. 6) indicates that despite massive restructuring some older patches of channels, alluvial channels, and pools remain. These surviving patches are likely important refugia for benthic macroinvertebrates during floods. These refugia play an important role in recolonizing newly formed habitats after the floods recede (Sedell et al. 1990, Robertson et al. 1995, Winterbottom et al. 1997, Rempel et al. 1999).

Habitats in the active zone will be restructured with a greater frequency than habitats in other parts of the river corridor, such as the adjacent riparian forest (Hughes 1997, Naiman et al. 1998). This will have implications for the age distribution of aquatic habitats in different parts of the active zone. Our results illustrate that in the active zone channels have a rather even age distribution over the 2.5-year investigation period, whereas the age distribution of alluvial channels, backwaters, and pools is strongly skewed to the left (Fig. 6). The age distribution of aquatic habitats in the riparian forest will probably be strongly skewed to the right (Kangas 1990, Hughes 1997). Hence, waterbodies in the riparian forest may serve as flood refugia (Sedell et al. 1990). The overall age distribution of aquatic habitats on the river corridor is the sum of the age of habitats in the active zone and the riparian forest. Considering the even age distribution for channels, the left-skewed distribution for other aquatic habitats in the active zone, and the hypothesized right-skewed distribution of habitats in the riparian forest, results in a relatively uniform distribution for the entire river and floodplain corridor (Fig. 7). The
availability of patches of different ages is important for maintaining river corridor biodiversity (Pearsons et al. 1992, Hughes 1997, Jordan et al. 1999).

From a landscape perspective riverine habitats within the active zone of the Tagliamento are in a state of dynamic equilibrium (sensu Huston 1994). Specifically Bormann and Likens' (1979) shifting mosaic steady state model applies to this system (Arscott et al. 2002). Our findings over a 2.5-year period are consistent with the findings of Arscott et al. (2002) that flood dynamics along the Tagliamento do not change aquatic habitat composition, but rather reconfigure the spatial environment leaving the same types and similar numbers of habitats available for re-colonization. Longer data collection is required to test whether the model is valid on other time-scales as well. When considering the constant composition of the riverine landscape it is important to realize that we compared maps made at similar water levels. Frequent and small flow pulses substantially alter the
availability of aquatic habitats in dynamic systems (Benke et al. 2000, Nat van der et al. 2002).

This study has indicated that floods are the major disturbances that determine habitat availability and stability in the active zone of temperate river landscapes. Turnover rates were very high and habitats generally very young. We also showed that vegetated islands play an important part for the stability of aquatic habitats. Future investigations of landform change are needed from additional temperate free-flowing rivers, on longer time-scales, including the active and more passive zones of the river and its flood plain. The outcomes of such investigations can help us better understand and preserve the few rivers within which dynamic large-scale geomorphological and ecological processes remain.

Acknowledgements:

We thank Dr. D.A.P. Hooftman for constructive comments on an earlier version of this manuscript. We are grateful to Andreas Schmidt and Silvio Blaser for their help in the field. Several people provided logistical support to our work in Friuli, especially Diego Nebuloni, Grazia Varra, and Claudio and Daniele di Scandiuzi. Sophie Karrenberg, Michael T. Monaghan, and David B. Arscott took part in many fruitful discussions concerning the work. Special thanks to Mr. Alberto Deana, Direzione Regionale dell' Ambiente, and Mr. Francesco Baruffi, Autorita di Bacino dei Fiume Isonzo, Tagliamento, Livenza, for providing hydrological data. Grant 0-20572-98 from the Forschungskommission of the ETH Zürich supported this research. Research by A. Gurnell was supported by grants GR9/3249 and NER/B/S/2000/00298 from the UK Natural Environment Research Council

References:


Osterkamp WR. 1998. Processes of fluvial island formation, with examples from Plum Creek, Colorado and Snake River, Idaho. Wetlands 18:530-545.


I see the ghosts of navigators but
they are lost
As the sail into the sunset they'll
count the cost

(Janick Gers, Bruce Dickinson, and Steve Harris, Ghost of the Navigator, 2000)
Quantification of large wood in large floodplain rivers: an area-based approach using differential GPS and GIS.

Introduction:

The importance of large wood (LW) in ecosystems has been well documented (Benke et al. 1984, Harmon et al. 1986, Steel et al. 1999). Although the occurrence of LW is largely restricted to streams, it once was a common feature in larger rivers as well (Maser and Sedell 1994). To investigate the importance of LW for ecosystem processes, such as organic matter retention and vegetation establishment, standing stocks must be determined. However, quantification of LW in dynamic and complex flood plains is difficult.

Thus far, three approaches have been used to quantify LW. One approach is to survey all deposits present in the study area (Bilby and Ward 1991). Although accurate and well suited for streams, this method is impractical to apply in large complex flood plains. A second approach is the 'line intersect' method (Warren and Olsen 1964) which estimates LW volume based on the diameter of LW encountered along a transect line. The complexity of large flood plains, however, makes accurate extrapolation from transects difficult. The third method is to survey LW in wide transects across the active corridor of large flood plains (Gurnell et al. 2000). Our objective was to develop a method that would allow insight into standing stocks, input, output, and fate of LW in large complex flood plains. Thus, a method that repeatedly subsampled a representative proportion of the total LW population was needed. In this paper, we present an area-based approach for quantifying LW and compare first results obtained with this method to results from the transect method of Gurnell et al. (2000).
Materials and Methods:

Study area:

The study sites were situated in the 7th order Tagliamento River (NE-Italy), which drains a 2850 km² catchment that flows from the southern limestone Alps to the Adriatic Sea. Its 172-km flowpath consists of a sequence of constrained, braided, and meandering reaches. In some reaches, vegetated islands cover as much as 20% of the total active corridor (Kollmann et al. 1998). Although not pristine, the Tagliamento remains remarkably free of intensive management (Gurnell et al. 2000), and consequently still has an essentially pristine morphological character (Ward et al. 1999).

This study focussed on two morphologically complex braided reaches in the middle section of the river: a bar-braided reach characterized by expansive areas of bare gravel and an island-braided reach characterized by the presence of many patches of woody vegetation (islands) on exposed sediments (Kollmann et al. 1998) and large quantities of LW (Gurnell et al. 2000). Arscott et al. (2000) provide a more detailed description of both reaches.

Survey design:
Using differential GPS (Global Positioning System) and GIS (Geographic Information System), spatial data were collected in both reaches and converted to maps. These maps were analyzed to determine the relative areas consisting of water, vegetated islands, and exposed sediments. To create a randomized block design, a grid (cell size 14.02 seconds of a degree) was superimposed over the digitized maps (Fig. 1A). Using GIS the coordinates (latitude / longitude) of the southwestern corner of each grid cell were established. Consequently, the products of six pairs of random numbers and cell sizes in lateral and longitudinal directions were added to the cells’ origins, creating six randomly located positions within each cell. We included all positions located in the active corridor and in a 30 m wide
strip of riparian forest (Piégay and Marston 1998), resulting in 69 and 94 positions for the bar- and island-braided reaches, respectively. In June 2000 all positions were located using a GPS unit with a horizontal precision of 0.3 m. This allowed for precise and repeatable location of the positions. At all positions, 100 m² circular plots were outlined in which the total standing stock of LW was estimated (Fig. 1B). All woody deposits with a length exceeding 1.0 m or a diameter exceeding 0.1 m were considered LW. Three kinds of deposits were distinguished; whole trees, jams, and logs. The dimensions of the smallest geometrical box that contained the deposit, were measured. These box volumes were converted to wood volumes using empirical conversion factors for trees, trunks, and jams (Thevenet et al. 1998). Volumes were converted to mass estimates using a conversion factor of 0.5 t m⁻³. To compensate for edge effects caused by deposits exceeding the plot limits, protruding deposits on the southern half of the plot were excluded from measurement.

In addition to a heterogeneous distribution of LW, the island-braided reach contained several very large jams. These jams likely represented a considerable proportion of the total standing stock of LW, but only a small proportion of the
total surface area of the study reach. All jams exceeding 10 m³ of volume were surveyed to compensate for underestimation of standing stocks, due to sample plots representing only 1% of the reaches total surface area.

**Results / Discussion:**

Comparison of the number of plots located on exposed sediments, islands, riparian forest, or in the water with relative areas of these strata from previously analyzed maps indicated that our subsample was representative of the entire LW population.

Standing stock of LW per hectare was 7 times higher in the island-braided reach than in the bar-braided reach (Table 1), reflecting the importance of vegetated islands for LW retention and input. The largest proportion of LW was associated with vegetated islands in the island-braided reach (Fig. 2A). Riparian forest also formed an important retention structure for LW. Most LW accumulated in the form of jams and single trees in the island-braided reach (Fig. 2B). The amount of LW stored in jams with a volume exceeding 10 m³ was about 5% of the total standing stock.

![Graph](image)

Fig. 2. (A) Standing stock (t ha⁻¹) of LW in different landscape elements of a bar-braided and an island-braided reach along the Tagliamento River, Italy. Abbreviations; RIF = riparian forest, ISL = vegetated island, GRV = exposed sediments, WTR = water. (B) Distribution of LW as jams, large jams, whole trees and trunks (see text). The + symbol indicates presence of LW, too small to show up as a bar.
Mass estimates exceeded those found using a transect approach in the summer of 1999 (Gurnell et al. 2000) by factors of 3 and 4.5 for the bar- and island-braided reaches, respectively (Table 1). There are several possible explanations for these differences. First, temporal variability in the abundance of LW, due to flood events since the first survey may have caused changes in standing stock and inputs of LW. Second, due to the heterogeneous spatial distribution of LW, extrapolation from a 100 m wide transects to a 2.5 km long corridor may considerably underestimate total wood volume. Only a small proportion of the difference between the two studies was explained by the inclusion of a 30 m wide strip of riparian forest in our survey that was omitted in the previous study.

Table 1. Morphological characteristics and estimated LW storage (t ha⁻¹) for a bar-braided and an island-braided reach along the Tagliamento River, Italy. Results of estimated wood volumes are compared to values from Gurnell et al. (2000).

<table>
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<th></th>
<th>Bar-braided</th>
<th>Island-braided</th>
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</thead>
<tbody>
<tr>
<td>Length (km)</td>
<td>1.5</td>
<td>2.5</td>
</tr>
<tr>
<td>Area of active corridor (ha)</td>
<td>103</td>
<td>143</td>
</tr>
<tr>
<td>Slope (m/km)</td>
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<td>1</td>
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<tr>
<td>Area water (%)</td>
<td>35</td>
<td>39.1</td>
</tr>
<tr>
<td>Area islands (%)</td>
<td>3.1</td>
<td>10.6</td>
</tr>
<tr>
<td>Area exposed sediments (%)</td>
<td>61.9</td>
<td>50.4</td>
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<tr>
<td>LW (t ha⁻¹)</td>
<td>Present Study</td>
<td>17</td>
</tr>
<tr>
<td></td>
<td>Gurnell et al. (2000)</td>
<td>6</td>
</tr>
</tbody>
</table>

Conclusions:

The area-based approach, with randomly distributed circular plots provided an accurate method to survey a representative subsample of the whole LW population in two large flood plains. Further it provided a feasible alternative to surveying all LW deposits or to extrapolating transect results to larger areas. The
repeatability of the method allows investigation of the fate and turnover of individual LW deposits.

References:


Maser C, Sedell JR. 1994. From the forest to the sea: The ecology of wood in streams, rivers, estuaries and oceans. St. Lucie Press, Delray Beach, Florida (USA).


Seite Leer / Blank leaf
We'd drive out of this valley down to
Where the fields were green
We'd go down to the river
And into the river we'd dive
Oh down to the river we'd ride

(Bruce Springsteen, The River, 1979)
Large wood dynamics of complex Alpine river flood plains

Abstract:

Despite a considerable amount of literature on large wood (LW) in freshwater ecosystems, its dynamic nature in larger rivers has hardly been investigated. The present study focussed on the mass and turnover of LW in braided flood plains of the Tagliamento River (NE-Italy), the last morphologically intact large river draining out of the Alps. LW masses and turnover were quantified by establishing 165 permanent plots (100 m² each) and then revisiting them after 4 floods of differing magnitude. The following hypotheses were tested: 1) Presence of vegetated islands increases LW densities, 2) masses of LW remain constant through time, 3) species composition of LW matches the species composition of woody plants on vegetated islands, 4) lateral erosion is the most significant source of LW, and 5) the probability that a LW deposit 'survives' a flood increases with the presence of islands, the size of the deposit, and its location within vegetation, but decreases with flood magnitude. During the investigation period, LW mass was high and constant in the island-braided reach, reaching values (100-150 t ha⁻¹) comparable to those reported for pristine mountain streams. In the bar-braided reach, LW mass was significantly lower (15-70 t ha⁻¹) and more variable. Although the total quantities of LW on the flood plain remained relatively constant, turnover rates of LW were very high (up to 95% during one major flood). An analysis of the species composition of LW showed that, while it was similar to that of woody plants on vegetated islands, at least 30% of the wood originated from upstream. Analysis of deviance from stepwise forward logistic regression models showed that the probability that a LW deposit survives a flood depends on flood magnitude, deposit volume and position.

Introduction:

Large wood (LW) plays an important role in river ecosystems by influencing hydrology, hydraulics, sedimentology, and morphology (e.g., Nakamura and

LW retains organic material and mineral sediments (Bilby and Likens 1980, Bilby 1981), thereby providing favorable conditions for germination of seeds and resprouting of vegetative propagules. Abundant floodplain pioneer species, such as Salicaceae, are well adapted to these conditions through their ability to resprout from vegetative fragments and their production of extraordinarily large numbers of seeds (Karrenberg et al. 2002a). Thus, LW dynamics is tied intimately to the dynamics of riparian vegetation (Kollmann et al. 1998, Gurnell et al. 2000a, Gurnell et al. 2001).

Once a feature of all large rivers, LW is abundant now only in undisturbed forested headwater sections (Dynesius and Nilsson 1994, Maser and Sedell 1994). For centuries, LW has been deliberately removed from river channels to facilitate navigation (Sedell and Froggatt 1984, Triska 1984, Petts et al. 1989). In addition, widespread forest clearance has reduced or cut off the input of wood to streams. Further, alterations of natural disturbance regimes by flow regulation has reduced the entrainment of woody vegetation and has truncated the downstream transport of LW. Consequently, most LW studies have focussed on small undisturbed streams (e.g., Bilby and Ward 1989, Bilby and Bisson 1998).

Several processes determine the abundance of LW in large floodplain rivers (Piégay et al. 1999). Input mechanisms include the transport from upstream
sections, erosion of lateral or instream vegetation, and excavation of buried deposits by scouring floods. Output mechanisms are decomposition, fragmentation, downstream losses, and firewood gathering by humans.

Past research has focussed mainly on effects of relatively immobile wood deposits on physical and biological processes, and only a few studies have investigated the dynamics of LW in rivers (e.g., Jacobson et al. 1999). The research reported here investigated the dynamics of LW abundance and the fate of individual LW deposits in braided river systems. We tested the following hypotheses: 1) presence of vegetated islands results in large LW mass in the active zone of natural braided rivers, 2) mass of LW in island-braided reaches remains approximately constant through time, 3) species composition of LW matches the species composition of woody plants on vegetated islands, 4) lateral erosion is the most significant source of LW, and 5) the probability that a LW deposit 'survives' a flood increases with the presence of islands, the size of the deposit, and its location in vegetation, but decreases with flood magnitude.

Study Site:

The Tagliamento is a 7th-order gravel-bed river located in NE-Italy (Fig. 1A), and has a catchment area of 2580 km² and a total length of 172 km. It is the last river arising in the Alps that still has a near-natural flooding regime. Although water is abstracted and the river is partly channelized in its lower reaches, the Tagliamento still has an essentially pristine morphological character (Müller 1995, Ward et al. 1999). Through a sequence of constrained, braided, and meandering reaches, it flows from the Italian Alps to the Adriatic Sea. The morphologically very dynamic "active zone" is up to 1.5 km wide, fringed by riparian woodland, and consists of 3 major landscape elements -water, bare alluvial sediments, and vegetated islands (Ward et al. 1999, Gurnell et al. 2000a). The main woody species in the upper reaches is Alnus incana. Populus nigra and 5 species of Salix dominate in the lower reaches (Lippert et al. 1995, Karrenberg et al. 2002b).
The river has a pluvio-nival flow regime with flash floods commonly occurring in spring and autumn. However, floods have only a low degree of predictability, and can occur at any time of year (Fig. 1B) (Campolo et al. 1999). As the river enters the coastal plain at Pinzano (Fig. 1A), it has an average discharge of approximately 90 m$^3$ s$^{-1}$. The 2, 5, and 10-yr floods are estimated to be 1100, 1600, and 2150 m$^3$ s$^{-1}$, respectively (Gurnell et al. 2000a). For a more detailed description of the river's catchment characteristics and hydrodynamics, see Ward et al. (1999).
and Gurnell et al. (2000a). Large floods frequently erode vegetated areas, making LW an abundant landscape element in the river corridor (Gurnell et al. 2000a, Gurnell et al. 2000b, Nat van der et al. 2002) (see Fig. 2B).

This study was conducted in 2 morphologically distinct reaches, hereafter referred to as the bar-braided reach and the island-braided reach (Fig 1A). The bar-braided reach, was located downstream of stream kilometer 74 (measured from the source); it was 1.5 km long and up to 1.0 km wide, and had an area of 105 ha. Under median-flow conditions, large areas of bare alluvium and multiple channels were the dominant landscape features in the bar-braided reach. The island-braided reach was upstream of a narrow bedrock gorge below kilometer 81. The 2.5-km reach had an active zone area of 146 ha and was 0.8 km wide. It included a complex network of channels and approximately 20 vegetated islands. In addition to main, secondary, and tertiary channels, many alluvial channels, backwaters, and isolated waterbodies occurred in both reaches (Arscott et al. 2000). Lateral or mid-channel gravel bars represented the largest proportion of landscape cover.

Methods:

The sampling scheme for LW was based on a notional grid set out over the floodplain. The cell size of the grid was 225x225m for the bar-braided reach and 300x300m for the island-braided reach (see Nat van der et al. 2002). Subsequently, the products of 6 pairs of random numbers and cell sizes were added to each cell’s origin, creating 6 randomly located positions within each cell (Fig. 2A). All positions that were not located in the active zone or in a 30-m wide strip of riparian forest bordering the active zone were excluded (Piégay and Marston 1998), resulting in 94 and 71 positions, respectively, for the bar-braided and island-braided reaches. We located the positions of each plot using a Trimble TCS1 GPS rover unit. Triangulation precision of < 0.3m allowed us to precisely relocate the positions repeatedly. At each position, a 100-m² permanent plot was outlined, in which the total LW mass was estimated (Fig. 2A).
We classified LW deposits as isolated trunks, wood accumulations, or intact shrubs/trees. The volume of LW deposits was estimated by determining the height, width and length of a rectangular form that enclosed the deposit as closely as possible (see Thevenet et al. 1998). Our definition of LW included all pieces with a minimal rectangular box width of 0.1 m or a minimal box length of 1.0 m. All deposits were marked with small aluminum labels to allow identification when revisiting the plot. This way we established turnover rates, which we defined as the relative number of LW deposits that were no longer there when revisiting the plots. We categorized the location of each deposit as water, exposed alluvium, vegetated islands, or riparian forest. Rectangular form volumes were converted to wood...
volumes using empirical conversion factors: 0.0013 for trees (van der Nat unpublished data), 0.82 for trunks, and 0.10 for accumulations (Thevenet et al. 1998). Volumes were converted to mass estimates using a conversion factor of $0.5 \times 10^3$ kg m$^{-3}$. To avoid overestimation deposits protruding from the southern half of the plot were excluded from analyses (Fig. 2A). We surveyed before and after flood events of different magnitudes in May 2000, February 2001, May 2001, and December 2001 (Table 1).

A significant proportion of LW on a flood plain occurs in a few large aggregates. We were concerned that if our sampling scheme by chance included none of these large aggregates we might underestimate the amount of LW. During the 1st survey, we located and measured all large deposits (geometrical box volume $>100$ m$^3$) in the active zone. This survey indicated that the LW found in large accumulations accounted for about 5 percent of the total volume (Nat van der et al. 2002). Consequently, future surveys of large accumulations located outside our designated plots were not thought to be necessary.

<table>
<thead>
<tr>
<th>Field survey</th>
<th>Previous flood</th>
<th>Flood magnitude (m)</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 2000</td>
<td>October 1999</td>
<td>2.35</td>
</tr>
<tr>
<td>February 2001</td>
<td>October 2000</td>
<td>3.5</td>
</tr>
<tr>
<td>May 2001</td>
<td>March 2001</td>
<td>1.6</td>
</tr>
<tr>
<td>December 2001</td>
<td>July 2001</td>
<td>1.5</td>
</tr>
</tbody>
</table>

Whenever we encountered new (untagged) deposits of LW, we took a small wood sample. In the case of debris accumulations, we sampled the LW piece that had probably initiated the accumulation (e.g., the largest or most decomposed piece in the accumulation). Wood samples were oven dried at 60° C for 1 week to avoid further decomposition before analysis. Subsequently, wood samples were identified
as far as possible based on the woody plant species, known to occur within the active zone of the flood plain (Karrenberg et al. 2002b). Species were identified from transverse, radial and tangential sections under a light microscope (Schweingruber 1990).

A Kruskal-Wallis H-test (Statistica 5.1, StatSoft Inc., Tulsa, Oklahoma, USA), with reach as the grouping variable and LW mass per plot as the dependent variable, was performed to determine whether the LW mass differed significantly between the reaches. Data for the bar-braided and island-braided reach were analyzed separately with a Friedman Anova (Statistica 5.1, StatSoft Inc., Tulsa, Oklahoma, USA) for differences between survey dates. LW deposit survival was analyzed with logistic regression models (R 1.4.1, R Development Core Team, 2002) with survival as the response variable and flood magnitude, reach, stratum, and LW type as categorical independent variables and the logarithm of LW volume as a continuous independent variable. A Chi-squared approximation of the difference in deviance between models with 1 independent variable and the null model was used to determine which variables explained most variance. Subsequently, we conducted an analysis of deviance for forward stepwise logistic regression models, starting with the variable causing the largest shift in deviance from the null-model. We set $P \leq 0.05$ as the criterion for inclusion of variables or interactions between variables into the model. Variables were added until no significant improvement of the model occurred.

Results:

On all survey dates the quantity of LW was significantly higher in the island-braided reach than in the bar-braided reach (Kruskal-Wallis H-test, $P = 0.0095$, Table 2A) (Fig. 3). LW mass in the island-braided reach did not differ significantly between dates (Friedman Anova $\chi^2(7,3) = 3.445, P < 0.32$, Table 2A), with a mean value of 120.9 t ha$^{-1}$ (minimum 102 t ha$^{-1}$, maximum of 157.5 t ha$^{-1}$). LW mass in the bar-braided reach showed larger variability with a mean value of 42.9 t ha$^{-1}$


Survey date influenced LW mass (Friedman Anova $\chi^2_{(9,3)} = 26.14, P < 0.0001$, Table 2A). Comparison between dates showed that LW mass in the bar-braided reach differed significantly on all dates (Table 2). For both reaches the largest densities were found during the survey following the largest flood in February 2001 (Fig. 3).

More than 60% of all observed wood belonged to the Salicaceae family (Table 3). Other frequent taxa were Gymnospermae (14%), Alnus incana (14%), Fraxinus excelsior (4%), and Hippophae rhamnoides L. (3.5%) (Table 3). The spatial distribution of LW across all plots was always very heterogeneous (Figs 4 and 5). The quantity of LW per plot ranged from 0 to >1000 t ha$^{-1}$; approximately 80% of plots had no LW. Although some plots contained about the same amount of LW in all surveys, LW mass in most plots changed considerably between sampling dates (Fig 5). Turnover of LW was up to 95% during the largest flood (stage height 3.5 m) and about 25% during the smallest flood (stage height 1.5 m). Nevertheless, total LW mass for the reaches always was within the same order of magnitude (Fig. 3). As much as 95% of the LW in the bar-braided reach was located on bare alluvial sediments (Table 4). In the island-braided reach, vegetated islands contained about 50% of all LW; bare gravel and riparian forest also held large proportions of the total LW mass (Table 4). Accumulations and downed trees were the predominant LW types, accounting for 75-95% of all deposits (Table 4).

Flood magnitude (Flm) explained 29% of the variance in wood survival probability in logistic regression models with single variables, much more than any of the other variables (Table 2B). The other variables were included in the model in the following order: logarithm of deposit volume (Lvol), LW type (Type), reach (Reach), and stratum (Stra). Inclusion of Lvol into the model significantly shifted deviance ($P=0.001$, Table 2B). Inclusion of an interaction term between Flm and Lvol did not improve the model. Inclusion of Type and Reach also did not significantly improve the model. Entry of Stra caused a significant shift in deviance ($P = 0.002$, Table 2B) but the inclusion of interactions between Stra and Flm or
Lwol did not. Consequently, flood magnitude, deposit volume (see also Fig. 6) and stratum described survival probability best. Despite the best fit, the model explained only 34.8% of the variance in the data.

Fig. 3 Lw mass (with standard errors) in tons per hectare for the bar-braided and island-braided reaches in May 2000, February 2001, May 2001, and December 2001. Black histograms represent values for the bar-braided reach (n = 94); gray histograms represent values for the island-braided reach (n = 71).
Table 2: (A) Kruskal-Wallis H-test for the effect of reach on LW mass and Friedman Anovas for the effect of survey date on LW mass. (B) Analysis of deviance for forward stepwise logistic regression models. The difference in deviance (Δ Dev) can be tested against a χ²-distribution. A significantly large value indicates that including the corresponding variable in the model gives a significant improvement in fit over the previous model. The Akaike Information Criterion (AIC) from single variable models determined the order of variable inclusion in the stepwise procedure.

Fm = flood magnitude, Lvol = the logarithm of wood deposit volume, Type = deposit type, Reach = reach, Stra = stratum.

### (A)

<table>
<thead>
<tr>
<th>Test</th>
<th>Variables</th>
<th>Test-value</th>
<th>Significance</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Kruskal-Wallis H-test</strong></td>
<td>dep. var.: LW mass, group. var: reach</td>
<td>$H_{(1,669)} = 6.73$</td>
<td>$P = 0.0095$</td>
</tr>
<tr>
<td><strong>Friedman Anovas</strong></td>
<td>all dates island braided reach</td>
<td>$\chi^2_{(7,1,3)} = 3.44$</td>
<td>$P &lt; 0.3283$</td>
</tr>
<tr>
<td></td>
<td>all dates bar-braided reach</td>
<td>$\chi^2_{(9,4,3)} = 26.14$</td>
<td>$P &lt; 0.001$</td>
</tr>
<tr>
<td>May 00 vs Feb 01</td>
<td>$\chi^2_{(9,4,1)} = 12.36$</td>
<td>$P &lt; 0.01$</td>
<td></td>
</tr>
<tr>
<td>May 00 vs May 01</td>
<td>$\chi^2_{(9,4,1)} = 3.92$</td>
<td>$P &lt; 0.28$</td>
<td></td>
</tr>
<tr>
<td>May 00 vs Dec 01</td>
<td>$\chi^2_{(9,4,1)} = 7.41$</td>
<td>$P &lt; 0.04$</td>
<td></td>
</tr>
<tr>
<td>Feb 01 vs May 01</td>
<td>$\chi^2_{(9,4,1)} = 14.29$</td>
<td>$P &lt; 0.002$</td>
<td></td>
</tr>
<tr>
<td>Feb 01 vs Dec 01</td>
<td>$\chi^2_{(9,4,1)} = 5.07$</td>
<td>$P &lt; 0.15$</td>
<td></td>
</tr>
<tr>
<td>May 01 vs Dec 01</td>
<td>$\chi^2_{(9,4,1)} = 3.46$</td>
<td>$P &lt; 0.04$</td>
<td></td>
</tr>
</tbody>
</table>

### (B)

<table>
<thead>
<tr>
<th>1 variable models</th>
<th>Δ Dev 0-model</th>
<th>Δ df</th>
<th>$P$</th>
<th>AIC</th>
<th>Var expl (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flm</td>
<td>151.8</td>
<td>3</td>
<td>&lt;0.001</td>
<td>362.9</td>
<td>29.8</td>
</tr>
<tr>
<td>Lvol</td>
<td>23.4</td>
<td>1</td>
<td>&lt;0.001</td>
<td>458.4</td>
<td>4.6</td>
</tr>
<tr>
<td>Type</td>
<td>23.0</td>
<td>3</td>
<td>&lt;0.001</td>
<td>491.8</td>
<td>4.5</td>
</tr>
<tr>
<td>Reach</td>
<td>20.5</td>
<td>2</td>
<td>&lt;0.001</td>
<td>492.3</td>
<td>4.0</td>
</tr>
<tr>
<td>Stra</td>
<td>14.6</td>
<td>4</td>
<td>&lt;0.001</td>
<td>494.2</td>
<td>2.9</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Stepwise model</th>
<th>Δ Dev simpl.</th>
<th>Δ df</th>
<th>$P$</th>
<th>AIC</th>
</tr>
</thead>
<tbody>
<tr>
<td>Flm + Lvol</td>
<td>10.3</td>
<td>1</td>
<td><strong>0.001</strong></td>
<td>354.6</td>
</tr>
<tr>
<td>Flm + Lvol + (Flm*Lvol)</td>
<td>0.63</td>
<td>2</td>
<td>0.733</td>
<td>358.0</td>
</tr>
<tr>
<td>Flm + Lvol + Type</td>
<td>4.95</td>
<td>2</td>
<td>0.084</td>
<td>353.7</td>
</tr>
<tr>
<td>Flm + Lvol + Reach</td>
<td>2.89</td>
<td>1</td>
<td>0.089</td>
<td>353.3</td>
</tr>
<tr>
<td>Flm + Lvol + Stra</td>
<td>14.8</td>
<td>3</td>
<td><strong>0.002</strong></td>
<td>345.8</td>
</tr>
<tr>
<td>Flm + Lvol + Stra + (Flm*Stra)</td>
<td>4.83</td>
<td>6</td>
<td>0.566</td>
<td>353.0</td>
</tr>
<tr>
<td>Flm + Lvol + Stra + (Lvol*Stra)</td>
<td>0.54</td>
<td>3</td>
<td>0.910</td>
<td>351.3</td>
</tr>
</tbody>
</table>
Table 3: Relative abundance (percentage of total dry mass) for woody species occurring on the active floodplain of the Tagliamento River between river kilometres 75 and 95 (from: Karrenberg 2002) and percentage of LW identified as belonging to those species. Brackets indicate the total percentage for species that could not be distinguished.

<table>
<thead>
<tr>
<th>Taxon</th>
<th>species</th>
<th>Relative abundance in live vegetation (% dry mass)</th>
<th>Relative abundance in wood samples (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Gymnospermae</td>
<td><em>Pinus sylvestris</em> L.</td>
<td>0.00</td>
<td>14.22</td>
</tr>
<tr>
<td></td>
<td><em>Juniperus communis</em> L.</td>
<td>0.00</td>
<td></td>
</tr>
<tr>
<td>Salicaceae</td>
<td><em>Salix alba</em> L.</td>
<td>1.15</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>S. daphnoides</em> Villars</td>
<td>3.28</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>S. purpurea</em> L.</td>
<td>8.19</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>S. triandra</em> L.</td>
<td>3.10</td>
<td>61.03</td>
</tr>
<tr>
<td></td>
<td><em>S. eleagnos</em> Scop.</td>
<td>30.81</td>
<td></td>
</tr>
<tr>
<td></td>
<td><em>Populus nigra</em> L.</td>
<td>39.31</td>
<td></td>
</tr>
<tr>
<td>Betulaceae</td>
<td><em>Alnus incana</em> (L.) Moench</td>
<td>2.78</td>
<td>14.22</td>
</tr>
<tr>
<td>Oleaceae</td>
<td><em>Fraxinus excelsior</em> L.</td>
<td>0.05</td>
<td>3.92</td>
</tr>
<tr>
<td>Eleagnaceae</td>
<td><em>Hippophae rhamnoides</em> L.</td>
<td>0.56</td>
<td>3.43</td>
</tr>
<tr>
<td>Ranunculaceae</td>
<td><em>Clematis spp.</em></td>
<td>0.00</td>
<td>1.72</td>
</tr>
<tr>
<td>Corylaceae</td>
<td><em>Corylus avellana</em> L.</td>
<td>0.00</td>
<td>0.49</td>
</tr>
<tr>
<td>Fagaceae</td>
<td><em>Quercus spp.</em></td>
<td>0.00</td>
<td>0.25</td>
</tr>
<tr>
<td>Fabaceae</td>
<td><em>Robinia pseudacacia</em> L.</td>
<td>0.05</td>
<td>0.49</td>
</tr>
<tr>
<td></td>
<td><em>Amorpha fruticosa</em> L.</td>
<td>8.50</td>
<td>0.25</td>
</tr>
</tbody>
</table>
Fig. 4 Spatial distribution of LW (t ha\(^{-1}\)) across the permanent plots in May 2000, February 2001, May 2001, and December 2001.
Table 4: Relative distribution (%) of LW mass over the strata gravel (GRV), riparian forest (RFO), vegetated islands (ISL), and water (WTR), and the distribution LW pieces across the types accumulation, tree, and trunk.

<table>
<thead>
<tr>
<th></th>
<th>Bar-braided</th>
<th>Island-braided</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>GRV</td>
<td>ISL</td>
</tr>
<tr>
<td>May 00</td>
<td>87.8</td>
<td>0.0</td>
</tr>
<tr>
<td>Feb 01</td>
<td>76.1</td>
<td>0.0</td>
</tr>
<tr>
<td>May 01</td>
<td>95.7</td>
<td>0.0</td>
</tr>
<tr>
<td>Dec 01</td>
<td>98.4</td>
<td>0.1</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th></th>
<th>Accumulation</th>
<th>Tree</th>
<th>Trunk</th>
<th>Accumulation</th>
<th>Tree</th>
<th>Trunk</th>
</tr>
</thead>
<tbody>
<tr>
<td>May 00</td>
<td>34.6</td>
<td>41.8</td>
<td>23.6</td>
<td>45.4</td>
<td>38.1</td>
<td>16.5</td>
</tr>
<tr>
<td>Feb 01</td>
<td>39.8</td>
<td>44.1</td>
<td>16.1</td>
<td>59.5</td>
<td>24.0</td>
<td>16.5</td>
</tr>
<tr>
<td>May 01</td>
<td>43.8</td>
<td>30.0</td>
<td>26.2</td>
<td>60.2</td>
<td>24.1</td>
<td>15.7</td>
</tr>
<tr>
<td>Dec 01</td>
<td>34.3</td>
<td>34.3</td>
<td>18.6</td>
<td>62.3</td>
<td>28.6</td>
<td>9.1</td>
</tr>
</tbody>
</table>

Discussion:

During recent decades, most investigations of LW have been conducted in relatively small headwater rivers rather than in large rivers (but see Piégay and Gurnell 1997, Piégay and Marston 1998). The present study therefore adds an important aspect to the understanding of LW in rivers, by focusing on the dynamics of LW (transport, turnover) in a large braided river system. The method applied (a high number of sampling plots selected by a random-block design) provides a statistically valid sampling of a representative proportion of a very large LW population. Because the method using GPS to locate samples is relatively rapid, the risk of a major flood occurring during the conduct of a survey is reduced. Its repeatability allows observations of the fate of individual deposits, and makes the procedure applicable for long-term ecological monitoring.

The size and abundance of pieces of LW and therefore their geomorphological and ecological importance purportedly decrease in a downstream direction (Swanson et al. 1982, Harmon et al. 1986, Bilby and Bisson 1998). This decrease in LW mass is inherent because the relatively constant input from lateral
forests is divided over an ever-increasing area. However, some part of this decrease may reflect increased human impacts downstream (Gurnell et al. 2002). Engineering of rivers, clearance of LW from river channels for navigation, and the removal of riparian forests have dramatically reduced the input of LW and the retention capacity of most rivers (Piégay and Gurnell 1997). Our results showed that in dynamic large rivers LW can be as abundant as in small streams. In the active zone and first 30-m of riparian forest of the island-braided flood plain of the Tagliamento, mean LW mass was 120 t ha\(^{-1}\), close to the mean value for 80 North American streams that included mainly headwater segments (232 t ha\(^{-1}\), Table 6; (Harmon et al. 1986).

![Fig. 5](image-url)

Fig. 5 Left: logarithm of LW mass across permanent plots with plots ranked from largest to smallest in May 2000, February 2001, May 2001, and December 2001. Right: logarithm of LW mass across permanent plots, with plots ranked according to LW mass in the previous survey. The gray shaded area indicates plots that contained LW but did not contain LW during the previous survey.
Fig. 6 Average volume (m$^3$) of LW deposits in the bar-braided and island-braided reaches that were retained or lost after floods of 1.48m stage height (A), 1.6m stage height (B), and 2.5m stage height (C).

Table 5: Processes and parameters that influence the large wood budget for the island-braided reach. NPP is Net Primary Production of living woody plants.

<table>
<thead>
<tr>
<th>Process</th>
<th>Derivation</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>NPP vegetated islands</td>
<td>11.7 t ha$^{-1}$ y$^{-1}$ * 13 ha</td>
<td>150 t y$^{-1}$</td>
</tr>
<tr>
<td>Biomass vegetated islands</td>
<td>217 t ha$^{-1}$ * 13 ha</td>
<td>2800 t</td>
</tr>
<tr>
<td>LW mass</td>
<td>102-157 t ha$^{-1}$ *146ha</td>
<td>17500 t</td>
</tr>
<tr>
<td>Erosion</td>
<td>7.4 ha y$^{-1}$ * 217 t ha$^{-1}$</td>
<td>1600 t y$^{-1}$</td>
</tr>
<tr>
<td>Decomposition</td>
<td>0.14-0.17 t t$^{-1}$ y$^{-1}$ * t</td>
<td>2700 t y$^{-1}$</td>
</tr>
<tr>
<td>Excavation / burial</td>
<td>unknown</td>
<td></td>
</tr>
<tr>
<td>Gathering by humans</td>
<td>unknown but small</td>
<td></td>
</tr>
</tbody>
</table>
A few estimates of LW biomass exist for near-pristine large rivers (Table 6). For example, LW mass in the 6th-order Ogeechee River (Georgia, USA) is about 65 t ha$^{-1}$ (Benke and Wallace (1990), measured as ash-free dry mass, which restricts direct comparison. In the Thomson River (Victoria, Australia), where only a small amount of debris was removed by human activities, LW volume was 172 m$^3$ ha$^{-1}$ (Gippel et al. 1996). The dominant riparian tree in this reach was *Eucalyptus camaldulensis*, which has a wood mass of 560 kg m$^{-3}$ (Hawkins 1987). This wood volume is equivalent to about 100 t ha$^{-1}$ of fresh wood.

During our investigation period, the amount of large wood was significantly higher in the island-braided reach than in the bar-braided reach. This observation is consistent with hypothesis 1. Since both reaches have a comparable morphology and

<table>
<thead>
<tr>
<th>River name</th>
<th>Drainage (km$^2$)</th>
<th>LW mass (t ha$^{-1}$)</th>
<th>Reference</th>
</tr>
</thead>
<tbody>
<tr>
<td>78 streams</td>
<td>0.02-67.5</td>
<td>232 (range 1-1800)</td>
<td>Harmon et al. (1986)</td>
</tr>
<tr>
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<td>1024</td>
<td>24</td>
<td>Lienkaemper in Harmon et al. (1986)</td>
</tr>
<tr>
<td>East Fork Salmon River</td>
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<td>1</td>
<td>Lienkaemper in Harmon et al. (1986)</td>
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<td>65</td>
<td>Wallace and Benke (1984)</td>
</tr>
<tr>
<td>Thomson River</td>
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<td>99</td>
<td>Gippel et al. (1996)</td>
</tr>
<tr>
<td>Tagliamento River</td>
<td>2100</td>
<td>27 *</td>
<td>Gurnell et al. (2000b)</td>
</tr>
<tr>
<td>Tagliamento River</td>
<td>2100</td>
<td>120 *</td>
<td>This study</td>
</tr>
</tbody>
</table>

slope, LW retention by and erosion of vegetated islands is likely to cause the difference in LW mass. The large flood (stage height 3.5 m) that occurred in the autumn of 2000 eroded 7.3 ha and 7.5 ha, respectively, of vegetated areas in the bar-braided and island-braided reaches. The estimated fresh woody biomass in vegetated islands was 217 t ha$^{-1}$ (van der Nat unpublished data). One single flood
thus caused LW inputs of approximately 1500 t into both reaches, representing about 33% and 9% of total LW mass in the bar-braided and island-braided reaches, respectively (Table 5). Although the input to both reaches through lateral erosion of riparian vegetation is similar, the portion of LW retained is significantly higher in the island-braided reach. Islands can be considered as surface-roughness structures in a river corridor that increase the retention of LW rather as boulders or riffles in small streams retain leaves and twigs (e.g., Ehrman and Lamberti 1992, Bretschko and Moser 1993, Webster et al. 1994).

Despite the relatively constant quantities of LW present on the flood plain during the investigation period, local turnover rates were extremely high (Figs 4 and 5). LW mass in the island-braided reach is more than 10 times higher than the yearly input calculated through erosion (Table 5). Analysis of aerial photographs revealed that relative surface cover of vegetated islands was nearly constant over the period 1984-1991. Half-life values of vegetated islands were about 8 years and maximum age about 20 years (Kollmann et al. 1998). Hence, on time-scales of a few years, a near-equilibrium between island formation and island erosion seems to exist. Had there been no near-equilibrium the areal extent of vegetated islands would have either decreased or increased. The near-equilibrium could arise from a positive feedback between LW mass and island formation, the LW serving to attenuate fluctuations in total island cover, at least over a short time-scale (see below).

Gurnell et al. (2000a) found that LW accumulates within distinct elevational bands along the Tagliamento. Similar observations were made for organic matter distributions on the River Severn, UK (Steiger et al. 2001). There is some evidence for a sorting of the LW according to size. This effect is demonstrated by our observation that the smallest flood moved 25 percent of LW deposits, primarily LW deposited close to the channel margin. Because the recurrence frequency for these minor floods is 5-6 times per year, residence time for such deposits is a few
months at most. The release of LW of different size classes at different times also contributes to the continuously changing spatial distribution of LW.

The fact that large-scale erosion of vegetated areas occurs only during large floods (recurrence frequency 2-3 years) and that LW transport already occurs during minor floods (recurrence frequency 5-6 times per year) suggests that the thresholds for large-scale erosion and LW transport are not the same. Although transport and erosion do not commence simultaneously, we hypothesize that both LW mass and biomass of vegetated islands remain relatively constant, over a time-scale of several years at the floodplain scale. LW mass fluctuates as a result of transport, decomposition, and erosion (Fig 7B). Living woody biomass on islands also fluctuates because of primary biomass production and erosion (Fig.7C). Gurnell and Petts (2002) hypothesized that major floods (e.g., a 10-year flood event) erode large parts of islands and reset the system from an island-braided to a bar-braided one. Following such a flood event, a positive feedback between LW deposition and vegetation establishment is expected that fosters island formation (see Fetherston et al. 1995, Kollmann et al. 1998, Montgomery and Buffington 1998, Osterkamp 1998, Edwards et al. 1999). Moreover, LW deposits may be the nuclei that trigger the formation of islands (see discussion in Gurnell et al. (2001)). Further, it is doubtful if the large-scale erosion of islands during major flood events promotes the development of islands in downstream sections, leading to an upstream-downstream shift in island formation. At a time scale of a few years, our results that the differences in LW mass between dates were not significant are consistent with hypothesis 2—that in island-braided reaches the mass of LW remains relatively constant.

The broad similarity (70%) in species composition of LW deposits and woody plants on vegetated islands suggests that a large portion of LW could originate from instream sources (islands and the lateral fringes of riparian forest). The observations could thus be consistent with hypothesis 3. The local species, however, also occur upstream of our reaches, making it impossible to establish if
the wood really was of local origin. However, a considerable proportion of the LW (30%) was composed of species absent from our reaches but typical for headwaters and tributaries (Karrenberg et al. 2002b), indicating a minimum import estimate of about 30%. Therefore, both lateral erosion and transport from upstream reaches contribute to LW. Hence, hypothesis 4 -that lateral erosion is the most significant source for LW- cannot be addressed with the present results. These observations concur with observations from the Queets River (Washington, USA) where hardwood species (*Alnus rubra*, *Populus trichocarpa*, and *Acer macrophyllum*) were better represented in the riparian forest than in instream LW, and conifers were better represented in LW than in the riparian forest (Hyatt and Naiman 2001). The abundance of conifer wood, however, could also have been caused by greater structural integrity of conifer wood, which makes it more resistant to fragmentation.

Fig 7 (A) Hypothetical hydrograph for a free-flowing river. Minor fluctuations in LW mass as a result of decomposition, transport and input through erosion (B), and minor fluctuations in living biomass of woody plants on vegetated islands due to net primary production and erosion (C), over a time-scale of 4 years.
According to our best fitting logistic regression model, the survival probability of a LW deposit depends on the magnitude of the flood, the size of the deposit, and its location. However, the best model only explained 35% of the variance in the data. It is clear that many factors influence turnover dynamics, and that they are only partly explained by the variables we measured. For example, the location of LW deposits above base-flow water level, their spatial arrangement, the presence of a lee at the downstream site of an island and their orientation relative to flow direction may also control deposit survival. Secondly, during our investigation period, the largest flood occurred first and the smallest one last. Therefore, deposits were considered for analysis only for the flood following their deposition, to avoid dependency in the data. Continued data collection in the future may provide a more random order of flood magnitudes, allowing the application of Poisson regression models.

Our mass estimates were higher than those derived by transect data from the same reaches along the Tagliamento (Gurnell et al. 2000b). These authors calculated LW densities for the bar- and island-braided reaches of 6 and 27 t ha\(^{-1}\), respectively. The consistently higher values during our surveys did not indicate large temporal variability. Several possible explanations for the observed differences remain. Firstly, the heterogeneous spatial distribution of LW may lead to an underestimation of LW densities when data from transects are extrapolated to an entire reach. This is particularly likely, given that the study by Gurnell et al. (2000b) was based on surveys along a single transect within each of the two reaches, and that the topographic survey requirements of a clear line of sight across the active zone would undoubtedly have given rise to an underestimation of island cover within the sampled area. Second, we included a 30-m band of riparian forest, whereas the survey by Gurnell et al., was conducted entirely within the active zone. Third, although we found little temporal variability in LW between surveys, the survey by Gurnell et al. (2000b) was conducted in August 1998, before the present study commenced. Finally, but probably of least importance, Gurnell et al. (2000b)
defined LW as being > 0.1 m in diameter and > 1 m in length, while we used > 0.1 m in diameter or > 1 m long as a criterion. However, when we omitted LW deposits that did not meet the criteria of Gurnell et al. (2000b) from our analysis, we found only a very minor reduction in total LW mass.


In order to better understand the influence of flood processes on LW dynamics under near-natural conditions further and longer investigations of LW input, output, and movement are required. Unfortunately, few rivers remain in the developed world in which the natural dynamics of large-scale geomorphic and ecological processes still prevail. The near-natural morphology and flow regime of the Tagliamento River provide unique opportunities to investigate the influence of flood processes on LW abundance and turnover under natural conditions.

Acknowledgements:

We thank associate editor Dr. F.J. Triska and two anonymous reviewers for constructive comments that helped to improve this manuscript. We are grateful to Professor Angela M. Gurnell for constructive comments on an earlier version of the manuscript. We also wish to thank: Andreas Schmidt, Mark Böhrer, Chihiro Yoshimura, Marianne Suter, Silvio Blaser, Michael Monaghan, Cécile Clarét, Rea Bonzi, Andreas Rotach, Daniel Schläpfer, Claudio Cruciat, and Simone Blaser for
assistance in the field. For constructive discussion concerning survey design, we thank Roland Brun, Chris Robinson, and David Arscott. For assistance with the statistical analysis, we thank Ruth Meili, and Sophie Karrenberg. Special thanks are due to several people who helped with field logistics, especially Diego Nebuloni, Grazia Varra, and Claudio Cruciat. Mr. Alberto Deana, Direzione Regionale dell' Ambiente, and Mr. Francesco Baruffi of the Autorita di Bacino dei Fiume Isonzo, Tagliamento, Livenza provided hydrological data. Grant 0-20572-98 from the Forschungskommission of the ETH Zürich supported this research.

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Abbe TB, Montgomery DR. 1996. Large woody debris jams, channel hydraulics and habitat formation in large rivers. Regulated Rivers: Research and Management 12:201–221.


Maser C, Sedell JR. 1994. From the forest to the sea: The ecology of wood in streams, rivers, estuaries and oceans. St. Lucie Press, Delray Beach, Florida (USA).


Osterkamp WR. 1998. Processes of fluvial island formation, with examples from Plum Creek, Colorado and Snake River, Idaho. Wetlands 18:530-545.


He placed it in his hands. Maybe his impatience heated his hand, or maybe it was the small size of the ice twig. Whatever it was, the small thing disappeared. Before they could look closely at it, it melted away and slipped through the cracks between his fingers. Father still held out his hand, but the Indians were staring at his finger stump.

(Paul Theroux, The mosquito coast, 1982)
Preliminary mass-balance for coarse particulate organic matter (CPOM) and large wood (LW) in an island-braided reach

Abstract:
An investigation of particulate organic matter (POM) dynamics was conducted in an island-braided reach of the free-flowing 7th-order Tagliamento River in NE-Italy. POM was divided into two categories: large wood (LW) and all other coarse particulate organic matter (CPOM). Distributions of LW and CPOM were measured in six distinct habitat strata across the reach's active zone and in a narrow band of adjacent riparian forest. Decomposition rates for wood and leaves of the dominant tree species were determined using wood cube bioassays and litterbag experiments. Discrete CPOM drift samples were taken on 18 dates to estimate input/output via fluvial transport. Aboveground biomass production by woody species on vegetated islands was determined by making repeated measurements of stem circumference at breast height. LW standing stock was very high, averaging 120 t ha⁻¹, two orders of magnitude higher than CPOM standing stock (0.6 t ha⁻¹). CPOM accumulation was highest in vegetated islands and the riparian forest. CPOM drift rates differed greatly among dates, but were not strongly related to discharge; however, there was no significant difference between input to and output from the reach on any date. With an average of 0.67 t t⁻¹ y⁻¹, leaf decomposition rates were relatively slow compared with published values. Decomposition of wood was difficult to interpret due to accidental losses; an estimate value of 0.15 t t⁻¹ ha⁻¹ was used in mass balance calculations. At 11.7 t ha⁻¹ y⁻¹, the mean annual biomass production was comparable to that of other forested wetlands. The goal of providing a POM mass balance was only partly successful due to severe sampling problems encountered in this highly dynamic river corridor. The preliminary mass balance indicates that the reach acts as a POM sink, accumulating ca. 1000 t y⁻¹. Recommendations are made for future research needs in order to improve estimates of organic matter mass balances on braided reaches.
Introduction:

In their natural state, rivers and their reaches are integrated ecosystems that exchange energy, organic matter, and organisms (Amoros and Roux 1988, Junk et al. 1989, Ward 1989, Tockner et al. 1999). Organic material is transported from the river to the flood plain as suspended sediments and nutrients, and from the flood plain to the river as organic detritus and algal biomass. The driving force for these interactions is lateral hydrologic connectivity between individual parts of the floodplain ecosystem (Tockner and Bretschko 1996, Ward 1997, Tockner et al. 1999, Malard et al. 2000).

Since aquatic ecosystems are strongly influenced by their organic matter supply (Wallace et al. 1999), the calculation of budgets provides insight into the functioning of stream and river ecosystems (Wanner et al. 2002). Such budgets should include investigations of standing stocks and input, as well as of processes such as transport and retention, production, and decomposition of organic matter; these various processes are often spatially separated within a flood plain. For example, primary production is highest on vegetated islands, while decomposition is typically more important in aquatic areas.

Standing stocks and inputs of CPOM have been intensively studied in streams and rivers. The vertical (Smock 1990) and lateral distribution of CPOM in streams has also been investigated (Bretschko 1990, Smock 1990, Bretschko and Moser 1993). The flood plains of larger rivers act as important storage and processing areas for CPOM, and there have been several studies of standing stocks and inputs of CPOM in large rivers. These include the Ogeechee in Georgia, USA (Cuffney 1988, Meyer et al. 1997), the Danube in Austria (Tockner and Bretschko 1996), and the Drôme River in France (Piégay et al. 1999). Transport and retention of CPOM have been investigated in both streams and rivers throughout the world (e.g. Cuffney 1988, Lieberman and Burke 1993, Wallace et al. 1995, Meyer et al. 1997).
Once a common feature of most rivers, large wood (LW) is abundant nowadays mainly in forested headwater sections (Dynesius and Nilsson 1994, Maser and Sedell 1994). Standing stocks, inputs, and outputs of LW have thus been predominantly investigated in small forested streams (e.g. Harmon et al. 1986, Maser and Sedell 1994, Nakamura and Swanson 1994). LW standing stocks have, however, also been investigated in a few remaining (semi-)natural rivers such as the Tagliamento River in NE-Italy (Gurnell et al. 2000a, Gurnell et al. 2000b), The Drôme River in France (Piégay et al. 1999), the Ogeechee River in Georgia, USA (Wallace and Benke 1984a, Benke and Wallace 1990), the Thomson River in Australia (Gippel et al. 1996) and the east Fork of the Salmon River (Lienkaemper cited in Harmon et al. 1986).

Direct investigations of transport and retention of LW are scarce, because transport mainly occurs during periods of high discharge and turbid water. Transport distances have been measured by marking trees in a subtropical floodplain swamp (Benke and Wallace 1990) and a large ephemeral river (Jacobson et al. 1999). LW transport or depletion rates can also be measured indirectly by comparing age and species distribution of instream LW with the adjacent riparian forest (Hyatt and Naiman 2001) or by estimating LW standing stock before and after floods (Van der Nat et al. 2003).

Annual biomass production in floodplain forests varies from 2 to 20 t ha⁻¹ y⁻¹ (e.g. Mitsch 1991, Conner 1994, Robertson et al. 1999). As much as 62 percent of the annual aboveground biomass production in floodplain forests can be leaf production (Megonigal et al. 1997). Mitsch (1991) found that aboveground biomass production in forested flood plains is highest at locations that frequently alternate between wet and dry phases. However, other investigations have found highest biomass production (sum off above and below ground) at poorly drained sites (Clawson et al. 2001).

This investigation presents a preliminary mass balances for CPOM and LW in an island-braided reach of a large semi-natural river. CPOM was defined as all
organic material larger than 1 mm that was not LW. The CPOM fraction mostly consists of leaves, twigs, fruits, and miscellaneous material such as seeds, roots and catkins. LW was defined as woody material fitting in a geometrical form longer than 1 m or wider than 0.1 m (Van der Nat et al. 2002b). The paper establishes values for standing stock, distribution, transport, production, and decomposition of CPOM and LW. We also discuss the practical difficulties of studying such ecosystem processes in a large dynamic river and provide recommendations for future research.

Study site:

The Tagliamento River is a 7th order gravel-bed river located in NE-Italy. The river is 172 km long, and drains a catchment of 2580 km². About 70 percent of its catchment area lies in the southern dolomite Alps. Average elevation of the catchment is 1159 m a.s.l, with a maximum of 2781 m a.s.l. (Ward et al. 1999, Arscott et al. 2000). The river flows from the Alps to the Adriatic Sea through a sequence of constrained, braided, and meandering reaches. The Tagliamento has many vegetated islands which cover approximately 9% of the entire active zone (Ward et al. 1999). The active zone consists of three major landscape elements: exposed sediments, water, and woody vegetated islands, fringed by continuous riparian woodland (Ward et al. 1999, Gurnell et al. 2000a).

The most important characteristic of the Tagliamento River is that, although the lowermost 25 km of its length have been channelized, most of the river is morphologically intact and largely unconstrained by engineering works. Consequently, the Tagliamento still has a near-natural flood regime (Ward et al. 1999). The hydrology is characterized by a pluvio-nival regime with frequent flashy flood and flow pulses. Flood pulses generally occur in spring and autumn and have an extremely low degree of predictability (Campolo et al. 1999). The average discharge at the location where the river leaves the Alps is approximately 90 m³ s⁻¹, and the 2, 5, and 10 year floods are estimated to be 1100, 1600, and 2150 m³ s⁻¹,
respectively (Gurnell et al. 2000a). For a more detailed description of the catchment characteristics and hydrodynamics, see Ward et al. (1999) and Gurnell et al. (2000a).

This study was carried out in an island-braided reach (Table 1), located in the middle section of the river, before it enters the coastal plain. For a more detailed description of the site see Arscott et al. (2000) and van der Nat et al. (2002a).

Table 1. Morphological characteristics of the island-braided reach.

<table>
<thead>
<tr>
<th>Metric</th>
<th>Value</th>
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<tbody>
<tr>
<td>Location of reach (river km)</td>
<td>80</td>
</tr>
<tr>
<td>Altitude (m a.s.l.)</td>
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<tr>
<td>Average slope (m m⁻¹)</td>
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<tr>
<td>Maximum width of active zone (km)</td>
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<tr>
<td>Floodplain length (km)</td>
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<tr>
<td>Area of active zone (ha)</td>
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</tr>
<tr>
<td>Average shoreline length (m ha⁻¹)</td>
<td>190</td>
</tr>
<tr>
<td>Area of vegetated islands (ha)</td>
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</table>

Methods:

CPOM distribution:

The sampling design divided the active zone and the first 30 m of the fringing riparian forest of the reach into six distinct habitat strata: riparian forest, vegetated islands, bare sediments, lotic waterbodies, lentic waterbodies, and shorelines. Collections were carried out in June 1999, January 2000, and May 2000. The CPOM samples collected in June 1999 were divided into three categories - leaves, twigs, and miscellaneous material. On exposed sediments and along shorelines a square frame (0.25 m²) was used to delineate the sampling area, and all CPOM was picked from the sampling area with forceps (10 replications). In the riparian forest and vegetated islands the sampling area was smaller (0.04 m²) because of the large amounts of CPOM (10 replicates). Aquatic sites were sampled with a Hess sampler (⌀ 30 cm) that was placed upon the bed surface. The top layer of the sediment was disturbed for 30 seconds. At the lentic sites current was
generated to drive CPOM into the net. All samples were oven-dried at 60 °C (usually a week), ignited at 540 °C for 4 hours, and then re-weighed so as to determine the ash free dry matter (AFDM).

The influence of the factors sampling date and habitat stratum on CPOM standing stock was tested using an ANOVA with CPOM ash free dry mass per square meter as dependent variable and date and stratum as independent variables (Statistica 5.1, StatSoft, Tulsa, Oklahoma, USA). One-way ANOVA procedures were used to test the influence of stratum and sampling dates. Tukey honest significant difference (HSD) post-hoc comparisons were used to calculate differences between habitat strata (Statistica 5.1, StatSoft, Tulsa, Oklahoma, USA). The average CPOM standing stock per area for all strata was converted to an estimate standing stock for the entire reach using the relative areal contribution of the habitat strata determined with GIS software.

**CPOM drift:**

Drift was determined by collecting material in nets with a diameter of 10 cm and a mesh of 1mm placed within a metal frame in the river for 5 minutes (4 replications). Flow velocities were measured at the inlet of the nets in order to calculate the volume of water that passed through them. Drift was measured upstream and downstream of the reach on 18 sampling dates between September 1999 and April 2000. Stage height was read from a gauging station located at the lower sampling site. Samples were oven-dried at 60 °C (1 week) and subsequently ignited at 540 °C for 4 hours (AFDM). The effects of location and sampling date/stage were determined using an ANOVA with CPOM drift as a dependent variable and date and location as independent variables (Statistica 5.1, StatSoft, Tulsa, Oklahoma, USA). Drift values were log-transformed and averaged in order to estimate average daily and annual transport rates.
LW distribution and standing stock:

LW was surveyed after flood events of different magnitudes in May 2000, February 2001, May 2001, and December 2001 (Van der Nat et al. 2003). The surveying scheme was based on a notional grid set out over the reaches. The grid had a cell size of 300*300m (see Van der Nat et al. 2002b). Subsequently, 6 randomly located positions were established within each cell. Only positions in the active zone or within a 30 m strip of riparian forest were included, resulting in 71 positions. The positions were located with a Trimble TCS1 GPS rover unit. Positioning precision of < 0.3m allowed precise relocation of the positions as necessary. At each position, a 100 m² permanent plot was set up in which total LW standing stock was estimated.

LW was classified as trunks, wood accumulations, or shrubs/trees. LW volume was estimated by determining the height, width and length of a notional rectangular box which enclosed the deposit as closely as possible (see Thevenet et al. 1998). The location of each deposit was classified as water, exposed sediments, vegetated island, or riparian forest. Box volumes were converted to wood volumes using empirically derived conversion factors: 0.0013 for trees (van der Nat unpublished data), 0.82 for trunks, and 0.10 for accumulations (Thevenet et al. 1998). Volumes were converted to mass estimates using a conversion factor of 0.5.10³ kg m⁻³.

Leaf decomposition:

Leaf litter from the three dominant riparian trees: Salix eleagnos, Populus nigra and Alnus incana (Karrenberg 2002) was collected within 1 day of abscission (Gessner et al. 1998) in the autumn of 1999. Because of phenological differences between species, leaves were air-dried after collection, until the last species, S. eleagnos had shed its leaves. Air-drying was continued until there was no further mass loss. Samples of leaves weighing 5 g were put into coarse mesh (10 mm) nylon litterbags. Ten samples were frozen immediately.
Litterbags were exposed in 5 lentic aquatic and 5 terrestrial (vegetated islands) sites across the reach. In early December 1999, 8 sets of 3 litterbags (one for each species) were attached to a brick (aquatic sites) or secured to the ground with a metal rod (terrestrial sites). Litterbags were retrieved from the aquatic sites after 1, 6, 18, 38, 68, 118, 147, and 175 days. Retrieval occurred from the terrestrial sites after 1, 6, 18, 38, 68, and 175 days (two planned terrestrial retrievals were omitted because of the slow rate of decomposition). Decomposition rates were calculated by establishing ash-free dry mass and exponential decay regressions with remaining mass versus exposure time (Statistica 5.1, StatSoft, Tulsa, Oklahoma, USA).

Subsamples of all samples were frozen immediately after retrieval and freeze-dried to determine dry mass. Subsamples of about 50 mg were crushed and extracted with alkaline methanol and cleaned using solid phase extraction cartridges (Gessner and Schmitt 1996). Ergosterol concentrations were measured with HPLC at a wavelength of 282 nm. The resultant ergosterol concentrations in mg g⁻¹ dry mass of leaves were converted to fungal biomass (Gessner and Schmitt 1996).

**Wood decomposition:**

This bioassay used wood cubes of *Salix alba, Populus nigra,* and *Alnus incana.* One meter samples of 4 trunks of each species were harvested, from the riparian forest and sawn into cubes (4x4x4 cm). These were air-dried to constant mass and ten cubes from each individual tree were used to establish a conversion factor for air-dried mass to oven-dried mass. After weighting, the remaining cubes were individually wrapped into a metal gauze (mesh size 5-mm). Five aquatic (lentic) and 5 terrestrial (vegetated islands) sites were chosen.

The cubes were set out in the study sites in July 1999. The gauze bags containing the cubes were randomly attached to 4 thick plastic-coated metal lines (length 4 m) and either submerged (aquatic sites) or secured to trees (terrestrial sites). Retrievals were planned for 3, 9, 12, and 24 months after initiation. Large
floods in September 1999 and November 2000, however, drastically reduced the scope of the experiment. The first flood destroyed 3 of the 5 aquatic sites; the latter eroded substantial areas of vegetation and damaged three of the terrestrial sites. The retrieval at $t = 3$ months was cancelled in order to leave enough cubes for later harvests. The first retrieval occurred in April 2000 after 9 months. At that moment 5 terrestrial and 2 aquatic sites remained. The retrieval planned for $t = 12$ months was also delayed with the aim of having enough cubes for harvests after 18 and 24 months. However, the flood of November 2000 and human interference made it necessary to terminate the experiment earlier than planned.

The cubes retrieved after 9 months were oven-dried at 60 °C until dry (which took more than 1 week) to establish mass loss. Relative remaining mass was log-transformed and plotted against time to obtain $k$-values and to compare annual decomposition rates for LW with literature values.

**Biomass production:**

In March 1999, 8 permanent 25 m$^2$ plots were established on vegetated islands. The stem circumference at breast height of all trees within these plots was measured with a tape measure. The procedure was repeated in March 2000 to establish annual biomass production by woody vegetation. Circumferences were converted to aboveground biomass estimates using a second-order polynomial relationship between radius and aboveground biomass of trees carrying foliage (Karrenberg et al. 2002). We thus estimated the biomass the trees would have had (including their foliage) in the previous summer. The difference between the biomass in both years was thus the annual production of leaves and wood by woody vegetation.
Results:

**CPOM distribution:**

CPOM standing stock across the different strata ranged from less than 1 g m$^{-2}$ to more than 1000 g m$^{-2}$. The ANOVA showed that there were significant effects of the factors date, and habitat stratum on CPOM dry mass (g m$^{-3}$) (Table 2A). However, the interaction between the two factors was also significant. The one-way ANOVA's with only habitat stratum as a factor indicated significant effects of stratum on all dates (Table 2B). The observed distribution and the significant differences were similar at all dates (Fig. 1). For all sample dates, the quantities of CPOM in riparian forest and island were much higher than in aquatic and non-vegetated terrestrial strata. Differences between these vegetated strata were only significant in January 2000 when the CPOM in islands was somewhat higher than in riparian forest (Fig. 1).

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**Table 2.** (A) Analysis of variance of the factors date and stratum influencing CPOM distribution in an island-braided reach of the Tagliamento River. (B) Analysis of variance for the factor stratum in June 1999, January 2000, and May 2000. Significant effects are indicated in bold.

**(A)**

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<td>5</td>
<td>1391981</td>
<td>19.2</td>
<td>&lt; 0.001</td>
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<tr>
<td>January 2000</td>
<td>stratum</td>
<td>5</td>
<td>232467</td>
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</tr>
<tr>
<td>May 2000</td>
<td>stratum</td>
<td>5</td>
<td>1729553</td>
<td>10.5</td>
<td>&lt; 0.001</td>
</tr>
</tbody>
</table>
Fig. 1: CPOM ash-free dry mass (g m², log-scale) for the strata riparian forest (Rf), vegetated islands (Isl), bare sediments (Grv), lotic waterbodies (Lot), lentic waterbodies (Len), and shorelines (Shr). Different letters indicate significant differences between the strata, no cross-comparisons between the different graphs can be made. Note different axes in January 2000. Division of CPOM ash free dry mass into the fractions leaves, twigs and miscellaneous material (fruits, roots, catkins etc.) in June 1999.

The non-vegetated and aquatic strata were not significantly different from each other in any sampling period. In the vegetated and aquatic strata CPOM was dominated by leaves, on exposed sediments by twigs and along shoreline by leaves and twigs (Fig. 1). The maximum standing stock per hectare of active zone calculated from the relative areal extent of the different strata recorded was about 0.8 t ha⁻¹, in May 2000. CPOM standing stock was lowest in January 2000, being 0.4 t ha⁻¹ (Fig. 2A).
Fig. 2: (A) Total CPOM standing stock (t ha⁻¹) for the island-braided reach in June 1999, January 2000, and May 2000. (B) Total LW standing stock (t ha⁻¹) in May 2000, February 2001, May 2001, and December 2001 (data from van der Nat et al. (2003)).

**CPOM drift:**

CPOM drift was highly variable between sites and dates (Fig. 3). The ANOVA with date and location as independent factors indicated that there was a significant effect of date but not of location (Table 3). The interaction between both factors was not significant. This indicates that input and output are not

Table 3. Analysis of variance for the factors date and location, influencing the drift of CPOM. Significant effects are indicated in bold.

<table>
<thead>
<tr>
<th>source</th>
<th>df</th>
<th>SS</th>
<th>F Ratio</th>
<th>P</th>
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<tr>
<td>date</td>
<td>17</td>
<td>1.843</td>
<td>41.5</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>location</td>
<td>1</td>
<td>0.020</td>
<td>0.4</td>
<td>0.50</td>
</tr>
<tr>
<td>date * location</td>
<td>17</td>
<td>0.046</td>
<td>1.0</td>
<td>0.44</td>
</tr>
<tr>
<td>error</td>
<td>108</td>
<td>0.044</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

significantly different and thus balanced. Highest drift values reached more than 3000 mg m⁻³ and occurred during a large spate in October 1999. The average CPOM drift was 21.0 mg m⁻³ (s.d. 1.6 to 280.9 mg m⁻³). Using the average discharge at our investigation site (90 m³ s⁻¹) annual CPOM transport was estimated to be about 60 t y⁻¹ with a range of 4.5 to 798 t y⁻¹ (±1 standard deviation).
Fig. 3: CPOM drift (mg m$^{-3}$) at a location upstream of the island-braided reach (grey circles) and a location downstream of the island-braided reach (black circles) from September 1999 through April 2000, with stage height (m) from a gauging station 2 km downstream of the island-braided reach.

Table 4. Distribution of LW (% of total standing stock) over the strata gravel (GRV), vegetated islands (ISL), riparian forest (RFO), and water (WTR) in the island-braided reach. n = number of plots per stratum.

<table>
<thead>
<tr>
<th>Date</th>
<th>GRV</th>
<th>ISL</th>
<th>RFO</th>
<th>WTR</th>
<th>Total LW (t ha$^{-1}$)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>%</td>
<td>%</td>
<td>%</td>
<td>%</td>
<td></td>
</tr>
<tr>
<td>05-00</td>
<td>34.1</td>
<td>47.7</td>
<td>18.2</td>
<td>11</td>
<td>119.6</td>
</tr>
<tr>
<td>02-01</td>
<td>25.4</td>
<td>52.5</td>
<td>19.7</td>
<td>7</td>
<td>156.9</td>
</tr>
<tr>
<td>05-01</td>
<td>67.5</td>
<td>0.3</td>
<td>25.8</td>
<td>6</td>
<td>102.5</td>
</tr>
<tr>
<td>12-01</td>
<td>56.2</td>
<td>34.5</td>
<td>9.2</td>
<td>6</td>
<td>104.5</td>
</tr>
</tbody>
</table>

LW distribution and standing stock:

LW was quite abundant with total standing stock ranging from 102 to 157 t ha$^{-1}$ (Table 4). Bare sediments held 34 to 68 % of the LW, being the dominant stratum. Vegetated islands contained as much as 53 of the LW although they represented less than 10 percent of the plots (Table 4). Riparian forest accounted
for about 10% of the plots and contained 9 to 26% of LW. Standing stock of LW in water never exceeded 6% although water accounted for about 25% of the plots.

Leaf decomposition:

Mass loss of *S. eleagnos* leaves could not be determined because many of the narrow leaves of this species were lost through the mesh bags, particularly in the aquatic sites. For *Alnus incana* and *Populus nigra*, the ANOVA, with location and species as independent variables and the natural logarithm of relative remaining mass as dependent variable, indicated significant effects of location and the interaction location x species (Table 5A). The two species showed opposite trends in their decomposition rates: *Alnus incana* leaves showed higher decomposition rates under terrestrial conditions (*k*-value = -0.0046 day⁻¹) than under aquatic (*k*-value = -0.0027 day⁻¹), (Fig. 4, Table 5B), while *Populus nigra* leaves decomposed more rapidly under aquatic (*k*-value = -0.0040 day⁻¹) than under terrestrial conditions (*k*-value = -0.0019 day⁻¹) (Fig. 4, Table 5B). All exponential decay regressions were significant (Table 5B).

Although not used for mass loss determination, the few remaining leaves of *S. eleagnos* were used in the determination of the fungal colonisation of decaying leaves. Highest fungal biomass values were found in the period between *t* = 38 days and *t* = 68 days for all species (Fig. 4C and D). Analysis of variance for the factors species and exposure indicated significant effects of both species and exposure under both terrestrial (Table 6A) and aquatic conditions (Table 6B).

Wood decomposition:

Mass loss of wood cubes was only determined after 9 months, because of losses due to floods. The mass loss was higher in all 3 species under aquatic than under terrestrial conditions (Table 7). Decay rates under aquatic conditions were highest for *P. nigra* and lowest for *S. alba*. Under terrestrial conditions the decay rate of *A. incana* was the highest (Table 7), and lowest for *S. alba*. 
Table 5. (A) Analysis of variance for the factors species and location influencing the decomposition of *Alnus incana* and *Populus nigra* leaves. Significant effects are indicated in bold. (B) Decomposition rates (*k*-values day$^{-1}$), adjusted $R^2$, and probabilities for exponential decay regressions of the mass loss of *Alnus incana* and *Populus nigra* leaves

(A)

<table>
<thead>
<tr>
<th>source</th>
<th>df</th>
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<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>location</td>
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<td>0.20507</td>
<td>17.8</td>
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<tr>
<td>species</td>
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<td>0.02744</td>
<td>2.3</td>
<td>0.13</td>
</tr>
<tr>
<td>location * species</td>
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<td>0.06516</td>
<td>5.5</td>
<td>&lt; 0.03</td>
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<tr>
<td>error</td>
<td>154</td>
<td>0.012</td>
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</tbody>
</table>

(B)

<table>
<thead>
<tr>
<th>location</th>
<th>species</th>
<th><em>k</em>-value (day$^{-1}$)</th>
<th>$R^2_{adj}$</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>aquatic</td>
<td><em>A. incana</em></td>
<td>-0.0027</td>
<td>0.83</td>
<td>0.0006</td>
</tr>
<tr>
<td>terrestrial</td>
<td><em>A. incana</em></td>
<td>-0.0046</td>
<td>0.88</td>
<td>0.0031</td>
</tr>
<tr>
<td>aquatic</td>
<td><em>P. nigra</em></td>
<td>-0.0040</td>
<td>0.82</td>
<td>0.0009</td>
</tr>
<tr>
<td>terrestrial</td>
<td><em>P. nigra</em></td>
<td>-0.0019</td>
<td>0.64</td>
<td>0.0245</td>
</tr>
</tbody>
</table>

**Biomass production:**

The average standing stock of biomass of woody vegetation across our 8 permanent plots rose from 182 t ha$^{-1}$ to 193 t ha$^{-1}$ (range 42 to 435 t ha$^{-1}$) between March 1999 and March 2000. Average annual biomass production by woody vegetation was thus 11.7 t ha$^{-1}$ y$^{-1}$. 
Table 6. (A) Analysis of variance for the factors species and exposure influencing the fungal biomass on *Alnus incana*, *Populus nigra*, and *Salix eleagnos* leaves under terrestrial (A) and aquatic (B) conditions. Significant effects are indicated in bold.

(A) terrestrial

<table>
<thead>
<tr>
<th>source</th>
<th>df</th>
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</thead>
<tbody>
<tr>
<td>species</td>
<td>2</td>
<td>150.05</td>
<td>3.2</td>
<td>&lt; 0.05</td>
</tr>
<tr>
<td>exposure</td>
<td>5</td>
<td>1083.59</td>
<td>23.2</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>species * exposure</td>
<td>10</td>
<td>87.58</td>
<td>1.92</td>
<td>0.06</td>
</tr>
<tr>
<td>error</td>
<td>69</td>
<td>46.80</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

(B) aquatic

<table>
<thead>
<tr>
<th>source</th>
<th>df</th>
<th>SS</th>
<th>F Ratio</th>
<th>P</th>
</tr>
</thead>
<tbody>
<tr>
<td>species</td>
<td>2</td>
<td>1365.25</td>
<td>9.780013</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>exposure</td>
<td>7</td>
<td>1304.57</td>
<td>9.345316</td>
<td>&lt; 0.001</td>
</tr>
<tr>
<td>species * exposure</td>
<td>14</td>
<td>176.727</td>
<td>1.265973</td>
<td>0.24</td>
</tr>
<tr>
<td>error</td>
<td>95</td>
<td>139.60</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 7: Annual decomposition rates (k-values y⁻¹) for the decomposition of *Salix alba*, *Populus nigra*, and *Alnus incana* wood under terrestrial and aquatic conditions.

<table>
<thead>
<tr>
<th></th>
<th><em>Salix alba</em></th>
<th><em>Populus nigra</em></th>
<th><em>Alnus incana</em></th>
</tr>
</thead>
<tbody>
<tr>
<td>aquatic</td>
<td>0.1607</td>
<td>0.1892</td>
<td>0.1710</td>
</tr>
<tr>
<td>terrestrial</td>
<td>0.1029</td>
<td>0.1504</td>
<td>0.1526</td>
</tr>
</tbody>
</table>
Fig. 4: Dry mass remaining of Alnus incana and Populus nigra leaves in litterbags at aquatic (A) and terrestrial (B) sites in an island-braided reach of the Tagliamento. Fungal biomass accumulation on leaves of A. incana, P. nigra, and Salix eleagnos under aquatic (C) and terrestrial (D) conditions. N.B. Salix eleagnos was not included in the mass loss graphs because its leaves were too small for the litter bag’s mesh.

Discussion:

Cummins et al. (1983) stated that stream ecology will not benefit from numerous short-term organic matter budget studies but needs determination of total stream ecosystem budgets at a few selected sites where long-term storage dynamics can be investigated. The present study represents a first step towards long-term investigation of the LW and CPOM mass balance of braided rivers. Below we discuss the practical difficulties that were experienced when determining a POM mass balance for a large dynamic river and their potential solutions for future investigations.

In many rivers the stored CPOM in floodplain soils can be an order of magnitude greater than in river sediments (Tockner and Bretschko 1996, Tockner et
In this investigation CPOM standing stock was higher in the vegetated strata than in the non-vegetated strata. However, there was no significant difference between exposed sediments, shorelines and the aquatic strata. CPOM standing stock in vegetated islands and riparian forest had values similar to those from other vegetated reaches (Cuffney 1988). The presence of vegetated islands strongly increases the standing stock of CPOM. The fact that all interactions between habitat stratum and date were significant made cross-comparison between dates difficult. LW standing stock was two orders of magnitude higher than CPOM standing stock (compare Fig. 2A and 2B); once again, the presence of islands is associated with a much higher storage capacity.

The following shortcomings of this investigation should be considered in designing future research. Firstly, only 3 surveys were conducted covering just under a complete calendar year. More frequent surveying might reveal significant difference between exposed sediments, shorelines, lentic, and lotic strata. Secondly, 10 replicate samples is scarcely sufficient to deal with the extreme patchiness of CPOM distribution (Cummins et al. 1983). Finally, for time reasons CPOM was separated into leaves, twigs and other material on only one occasion (June 1999), although a strong seasonality exists in the relative contribution of different components (e.g. Pozo et al. 1997).

No differences were found in the drift of CPOM into and out of the island-braided reach. Hence, the difference in CPOM standing stock cannot be explained by retention of particulate organic material by islands. Rather the difference must result from production of predominantly leaf litter in vegetated islands. The average CPOM transport rate of 0.021 mg l⁻¹ reported here is comparable to values reported for blackwater streams (Jones and Smock 1991) and large rivers (Angradi 1991). Again, there are lessons to be learnt from this study for future work. More frequent sampling is needed to make reliable annual estimates. Because transport of CPOM occurs mainly during large spates (Angradi 1991) CPOM transport should be measured as often as possible including large spates (Cuffney and Wallace 1988).
Thirdly, drifting and particularly floating CPOM is important for faunal dispersal in rivers (Hax and Golladay 1998). Future investigations should include fauna and distinguish between drifting and floating CPOM.

With an average LW standing stock of 120 t ha\(^{-1}\) for the island-braided reach, the Tagliamento exhibits a LW storage capacity similar to that for forested low-order streams in the Pacific Northwest (Harmon et al. 1986). Only a few other unregulated rivers are known to have similar LW standing stock (e.g. Wallace and Benke 1984b, Gippel et al. 1996), but most regulated large rivers have much less LW (Piégay et al. 1999). There is considerable scope for improving our understanding of the dynamics of LW in rivers. Possibilities include direct measurement of LW transport, for instance by radio-telemetry (a technique commonly used for studying animal movement) and investigation of the below-ground storage pool with ground penetrating radar (Naegeli et al. 1996). In particular, because processes influencing LW standing stock operate over time-scales of decades rather than years, long-term studies are needed (Gurnell et al. 2001, Gurnell et al. 2002).

The exponential decay values for *A. incana* and *P. nigra* leaves under aquatic conditions reported here are low when compared to values (e.g. Neiff and de Neiff 1990, Hieber and Gessner 2002). They are, however similar to values for the same species in the Garonne River in France (Chauvet 1987). Leaf litter decomposition rates depend on environmental factors such as water chemistry (Suberkropp and Chauvet 1995), climate (Gessner et al. 1998), flow velocity and flooding frequency (Chauvet 1988). A potential problem in this investigation is that all leaves were exposed under lentic conditions, whereas most published studies have been performed under lotic conditions, which facilitate a continuous supply of nutrients and oxygen for decomposers. Future work should include experiments under both lentic and lotic conditions. Decomposition rates of *A. incana* and *P. nigra* leaves under terrestrial conditions were slow when compared to values for the same species in the Garonne River in France (Chauvet 1987). However, our \(k\)-values
were at the high end of the range reported in the literature (e.g. Vaun Mc Arthur et al. 1994, Baker et al. 2001). Future study should attempt to establish decomposition rates on exposed sediments, which is the dominant but morphologically very unstable stratum. The role of macroinvertebrate community in leaf breakdown also requires investigation.

Decomposition of wood in aquatic environments has not been studied intensively (but see Melillo et al. 1983). Our decomposition rates of wood under aquatic conditions \(k = 0.18 \, \text{y}^{-1}\) are comparable to the decay rates of birch ice-cream sticks at lotic sites \(k = 0.15 \, \text{y}^{-1}\) (Sinsabaugh et al. 1992) but much lower than the decomposition of wood chips (Melillo et al. 1983). The decomposition rates of wood in vegetated islands in this study \(k\)-value = 0.10-0.15\) was higher than values reported for the terrestrial environment of reaches \(k\)-values 0.05-0.09\) in the literature (Sinsabaugh et al. 1993, Chueng and Brown 1995). However, our values were lower than those reported for LW decomposition in the Atchafalaya basin by Rice et al. (1997). They also appear rather low compared with our own casual observations of very rapid decay of fallen logs of trees such as *Populus nigra* on the reach. This may reflect a weakness of the bioassay using small cubes of prepared wood. We suggest that continued investigation of LW decomposition distinguish between the lentic and lotic parts of the aquatic environment. Furthermore exposed sediments should be included as investigation stratum since a large proportion of LW is situated there. Sampling campaigns should be set up with more samples to compensate for flood losses. The use of twigs and small branches across a range of sizes will probably better estimate the LW decomposition than the use of bio-essays. The role of the faunal community present on decomposing wood in reaches also needs investigation (Anderson et al. 1978).

Annual aboveground biomass production by woody vegetation of about 11 t ha\(^{-1}\) y\(^{-1}\) was very similar to values reported for dry and intermediately inundated plots from North American floodplain forests (Mitsch 1991, Megonigal et al. 1997). Mitsch (1991) found that aboveground biomass production in forested flood plains
is highest at locations that frequently alternate between wet and dry phases. However, other investigations found highest biomass production (sum off above and below ground) to occur at poorly drained sites (Clawson et al. 2001). Although we did not conduct this investigation along a wetness gradient, the high production and the fact that the vegetated islands are periodically inundated (recurrence frequency approximately 2 y⁻¹) support the observations made by Mitsch (1991). The relationships between circumference and aboveground biomass that we used were based on weightings of the aboveground parts of trees including their foliage. Hence, no separation into woody and leaf biomass could be made. Other research, however, has indicated that as much as 62 percent of the annual aboveground biomass production in floodplain forests can be leaf production (Megenigal et al. 1997).

LW standing stock in the island-braided reach was relatively constant at a time-scale of a few years (Van der Nat et al. 2003). LW also constitutes the bulk of the coarse organic matter standing stock (Fig. 2 and Table 8). Hence, we can presume that the total standing stock of POM is constant at this time-scale. Equilibrium in standing stock of POM can only exist when the total input of both equals their output. Input pathways are upstream input, erosion of lateral vegetation, primary production in vegetated islands, and excavation of buried material by scouring floods. For the island-braided reach primary production of biomass by trees and shrubs equals 150 t y⁻¹, input through erosion equals 1600 t y⁻¹ (Table 8). The upstream input is unknown but since the LW standing stock is more than 10 times larger than the input through erosion (Table 8, see also van der Nat et al. (2003)), it becomes evident that the island-braided reach is a sink for POM. Output pathways for POM are decomposition (2700 t y⁻¹ for LW, and 58 t y⁻¹ for CPOM), downstream output, burial of material by floods, and firewood gathering by humans. Since the reach is degrading at a slow rate (Arscott et al. 2000) we presume here that excavation and burial of LW and CPOM are small and in the same order of magnitude if not equal. Firewood gathering does occur but only at a
Table 8: Standing stocks, input pathways, and output pathways of CPOM and LW in an island-braided reach of the Tagliamento River.

<table>
<thead>
<tr>
<th></th>
<th>Derivation</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Standing stock</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LW standing stock</td>
<td>102-157 t ha(^{-1}) * 146 ha</td>
<td>17500 t</td>
</tr>
<tr>
<td>Biomass vegetated islands</td>
<td>217 t ha(^{-1}) * 13 ha</td>
<td>2800 t</td>
</tr>
<tr>
<td>CPOM standing stock</td>
<td>0.59 t ha(^{-1}) * 146 ha</td>
<td>86 t</td>
</tr>
<tr>
<td><strong>Input</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Erosion</td>
<td>7.4 ha y(^{-1}) * 217 t ha(^{-1})</td>
<td>1600 t y(^{-1})</td>
</tr>
<tr>
<td>NPP wood (islands / riparian forest)</td>
<td>0.35 * 11.7 t ha(^{-1}) y(^{-1}) * 15 ha</td>
<td>62 t y(^{-1})</td>
</tr>
<tr>
<td>NPP leaves (islands / riparian forest)</td>
<td>0.65 * 11.7 t ha(^{-1}) y(^{-1}) * 15 ha</td>
<td>115 t y(^{-1})</td>
</tr>
<tr>
<td>LW excavation by floods</td>
<td>Small</td>
<td>x</td>
</tr>
<tr>
<td>CPOM transport</td>
<td>(0.01-2.18 t day(^{-1})) * 365</td>
<td>60 t y(^{-1})</td>
</tr>
<tr>
<td><strong>Output</strong></td>
<td></td>
<td></td>
</tr>
<tr>
<td>LW decomposition</td>
<td>0.14-0.17 t t(^{-1}) y(^{-1}) * 17500 t</td>
<td>-2700 t y(^{-1})</td>
</tr>
<tr>
<td>CPOM decomposition</td>
<td>0.67 t t(^{-1}) y(^{-1}) * 86 t</td>
<td>-58 t y(^{-1})</td>
</tr>
<tr>
<td>LW burial by floods</td>
<td>Small</td>
<td>x</td>
</tr>
<tr>
<td>LW gathering</td>
<td>unknown but small</td>
<td></td>
</tr>
<tr>
<td>CPOM transport</td>
<td>(0.01-2.18 t day(^{-1})) * 365</td>
<td>-60 t y(^{-1})</td>
</tr>
</tbody>
</table>

\[ \Delta \text{upstream input - downstream output} \] \[ \pm 1000 \text{ t y}^{-1} \]

small scale (van der Nat, personal observation) and probably does not disturb the mass balance in any major way. Presuming that the sum of all inputs equals the sum of all outputs, upstream input must be approximately 1000 t y\(^{-1}\) larger than downstream output (Table 8), supporting the finding that the island-braided reach is a sink for organic material (Van der Nat et al. 2003). The finding is intriguing, but its significance for the functioning of the river as an ecosystem is far from clear. It could be that the presence of islands increases the capacity of the reach to trap organic matter, and therefore the island-braided reach functions as a sink. Alternatively, it may be that a sink effect occurs because of certain hydro-geomorphological characteristics of the river that promote the dumping of organic matter and sediment in this reach. On this interpretation, the establishment of
islands may be favoured here because of the tendency for organic matter and sediment to accumulate.

Understanding of the functional processes occurring in the few remaining free-flowing temperate rivers is crucial for preservation of existing near-natural rivers and restoration of regulated ones. Long-term investigations of the mass-balance of rivers and their flood plains are required for such understanding (Cummins et al. 1983). The results reported herein are important first steps toward that goal. Learning from the difficulties and shortcomings of these preliminary findings will enable us to tackle the difficult task of establishing POM mass balances for large braided rivers.

Acknowledgements:

We wish to thank Dr. Sophie Karrenberg, Luana Bottinelli, Sonja Ketterer, Dr. David, B. Arscott, Prof. Johannes Kollmann, Andreas Schmidt, and Edith Kaiser for their help with the fieldwork. We are grateful to several people who helped with field logistics, especially Diego Nebuloni, Grazia Varra, and Claudio Cruciat. We also want to thank Richard Illi for conducting the ergosterol analysis in the laboratory. Fruitful discussions with Dr. Mark Gessner and Prof. Keller Suberkropp greatly improved the design of the leaf decomposition experiment. Grant 0-20572-98 from the Forschungskommission of the ETH Zürich supported this research.

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GENERAL CONCLUSIONS AND RECOMMENDATIONS

But whether Joe knew how poor I was, and how my great expectations had all dissolved, like our own marsh mists before the sun, I could not understand.

(Charles Dickens, Great expectations, 1860-1861)
General conclusions and recommendations:

The research reported in this thesis contributes to our understanding of ecosystem processes in riverine landscapes. The discovery that small, frequent flow pulses (sensu Tockner et al. 2000) have an influence on the availability of shoreline and exposed sediment habitats (Chapter 2) requires consideration in future river restoration and management (see also Benke et al. 2000, Tockner et al. 2000). Both shorelines and exposed sediment have become increasingly rare in European rivers (Pinay et al. 1990, Ward et al. 2002), and their conservation and restoration is important for the continued survival of a wide array of floodplain species. At present, hydropower generation schemes are considered 'ecologically sound' as long as flood pulses (sensu Junk et al. 1989) are allowed to pass through the system. However, this research has found a linear relationship between water level and inundation, and a polynomial one between water level and shoreline length. These relationships are in strong contrast to relationships in regulated and channelized rivers where shoreline length is low and inundation increases in a stepwise fashion (Fig. 1). It is clear that small flow pulses have significant ecological implications and should be considered in river management schemes as well. Letting large flood pulses pass unhindered through the system is an important first step towards more natural rivers but it should not be the only one. If a natural or near-natural river is desired, restoring or mimicking a natural flow regime with both large and small water level fluctuations is required.

The turnover of aquatic habitats in braided reaches by flood pulses was surprisingly high (as much as 90% turnover in 2.5 y, Chapter 3) with active zone habitats being mostly very young (particularly the lentic ones, < 3 months). The dominance of very young habitats in the active zone of braided reaches contrasts strongly with the age distributions of floodplain habitats in regulated and channelized rivers, where older habitats dominate (Fig. 1). Our findings support earlier observations by Arscott et al. (2002) that the composition of the active zone of the flood plain is a "steady state shifting mosaic" (Bormann and Likens 1979).
Scouring floods destroy specific habitats but they create about the same area and number of new habitats for recolonization. The presence of vegetated islands in the active zone increases the stability of the aquatic habitats. The sheltering effect of islands is not dramatic (75% of aquatic area turned over in 2.5 years compared to 90% in the absence of islands) but the increase may have profound effects on biodiversity. The few additional aquatic habitats that survive a flood pulse in the presence of islands may act as refugia for many organisms. These refugia play an important role in recolonizing newly formed habitats after the floods recede (e.g. Sedell et al. 1990, Rempel et al. 1999). Thus, vegetated islands indirectly play a role in the maintenance of aquatic biodiversity in braided reaches. Vegetated islands have almost completely disappeared from the lower sections of rivers (Gurnell and Petts 2002). Their ecological importance and the fact that they result from the dynamic interaction between natural flow regimes and abundance of large wood (LW) needs to be considered in river restoration and management schemes.

Fig. 1: Hypothetical relationships between water level (WL), shoreline length (SLL), and degree of inundation and a habitat age distribution for bar-braided, island-braided, regulated, and chanellized reaches.

The approach for quantifying LW based on a rigorous subsampling of the entire LW population proved an effective way of assessing wood storage in a
morphologically complex river. The method takes only 2-3 days per reach (Chapter 4), which reduces the risk of interference by floods. Its repeatability allows observations of the fate of individual LW deposits, and makes the procedure applicable to long-term ecological monitoring. LW mass was surprisingly high (Chapter 5) in the presence of islands (100-150 t ha⁻¹). These values are comparable to the mean value reported for pristine mountain streams (Harmon et al. 1986) and contradict the hypothesis that the abundance and ecological importance of LW decreases with stream order (Swanson et al. 1982). Turnover of LW was very high during flood pulses (95%) but flow pulses also transported a considerable proportion (25%) of the LW deposits, another indication of the ecological importance of small water level fluctuations. The persistence of individual LW deposits depends on flood magnitude, and the size and location of the deposit within the reach. Despite the high turnover rate of LW, the total LW mass in the presence of islands was surprisingly constant. This constancy, and the fact that LW mass exceeds annual lateral input by as much tenfold, indicates that island-braided reaches function as sinks for LW. Within these sinks, flood and flow pulses spatially rearrange LW deposits rather than just transport them downstream. This observation is particularly important when we consider that LW is actively removed from many rivers to protect man-made structures such as roads and bridges downstream. In many cases, this active LW removal is probably not justified and may even be counter-productive; this is because LW is involved in the formation of vegetated islands (Gurnell et al. 2001), and these structures may cause the reach to act as a sink for LW. Future research on LW in braided reaches should be conducted over decades rather than years (Gurnell et al. 2001, Gurnell et al. 2002). The investigation of the transport of LW in large rivers is in its infancy (but see Benke and Wallace 1990, Jacobson et al. 1999) and much more remains to be done. Measurements of transport, for instance with telemetry (Bodie and Semlitsch 2000), and of the below-ground storage of LW with ground penetrating radar (Naegeli et al. 1996) are promising approaches for the future.
Stream ecology would benefit more from long-term studies of organic matter mass balances at a few selected sites, where long-term storage dynamics can be investigated, rather than from numerous short-term studies (Cummins et al. 1983). This thesis made the first steps toward a long-term POM (CPOM+LW) mass balance for the braided reaches of a large Alpine river (Chapter 6). The preliminary results show that CPOM is 200 times less abundant than LW and is primarily restricted to vegetated islands and the riparian forest. This accumulation is not an effect of retention capacities of vegetation but rather the result of litter fall on site. Since the storage of LW was constant at the time-scale of the present study, the POM mass balance suggests that 1000 t y⁻¹ is imported into the study site from upstream regions. This again indicates that the fear that LW poses a threat to downstream structures has little foundation.

The experience gained from this investigation teaches us that working in a highly dynamic environment requires certain adaptations of the usual experimental procedures. Due to potential massive losses by floods, large numbers of replicates are needed in any study that involves leaving samples on the flood plain. Future research should be conducted over long time-spans (particularly concerning LW), and include investigations of decomposition on exposed sediments (dominant but unstable stratum). Because of strong temporal variation, transport of CPOM should be measured continuously if we are to establish reliable annual transport rates.

The research in this thesis has provided insight into the extraordinary dynamic ecosystem processes in natural flood plains, and has raised many questions, which require future research. Although the Tagliamento represents a fine example of a near-natural Alpine river (Ward et al. 1999), generalisations considering large-scale ecosystem processes in near-natural rivers can only be made when more rivers are investigated. Nevertheless we propose that certain features of the Tagliamento - a natural flood regime with frequent flow and flood pulses, and
the abundance of LW - will prove to be of key importance in maintaining biodiversity in all natural rivers.

References:


Acknowledgements:

I wish to thank Prof. Peter J. Edwards, Prof. Angela M. Gurnell, and Dr. Klement Tockner for their support, supervision, encouragement, and constructive criticism during the last years. I also extend my gratitude toward Prof. J.V. Ward, who has been involved with the conducted research from the beginning. The standards these supervisors set have been a challenge for me to meet.

Great gratitude is due to my wife, Sophie Karrenberg, who has unconditionally supported me throughout the good phases and the bad phases that occur when writing a dissertation. Although writing a dissertation of her own she was always there for me. I also wish to thank my parents, whose early stimulation of my interests has resulted in my pursuit of intellectual challenges.

During the last 4 years my colleagues of the Tagliamento research program have been of great help in the field, the lab, and in many fruitful discussions. Sophie Karrenberg, Klement Tockner, Johannes Kollmann, Luana Bottinelli, Dave Arscott, Frederieke Moeslacher, Edith Kaiser, Cecile Clarét, Ute Karaus, and Achim Pätzold have all provided help at some point and have helped to make all the stays in Italy a joy.

I also thank the friendly population of Friuli-Venezia-Julia, in particularly Grazia Varra, Diego Nebuloni, Claudio Cruciat, and Claudio and Daniele di Scandiuizi who solved many of my logistical problems. Special thanks to Grazia Varra for her patience in teaching me some Italian and her understanding when I was unable to express myself.

My stay in Zürich had not been so enjoyable had it not been for living in the "Schwamikon" WG. Sharing a house with this extraordinary group of people helped make the last years an unforgettable experience. I am very thankful to Sophie Karrenberg, Mike Monaghan, Michael Steiner, Marie Pezze, Elke Büchel, and Anna Treydte for their companionship throughout the years.
Of all my colleagues at EAWAG I am most indebted to those who put up with sharing an office with me, especially during phases of stress. Monika Winder, Luana Bottinelli, and Ute Karaus experienced my dissertation first hand. Occasional small talk over the many cups of coffee and tea we have drunk over the years, afterwards allowed me to work concentrated for many hours each day.

During the years it has been a great pleasure to work at the EAWAG Limnology department. I wish to thank: David Arscott, Simone Blaser, Luana Bottinelli, Peter Burgherr, Diane Bürge, Hans-Rudi Bürgi, Nanna Büsing, Cecile Clarét, Christine Dambone, Michael Döring, Barbara Fassnacht, Andreas Frutiger, Mark Gessner, Massimiliano Gilli, Mäggi Hieber, Richard Illi, Regula Illi, Christa Jollidon, Edith Kaiser, Ute Karaus, Barbara Keller, Esther Keller, Sandra Lass, Hansueli Laubscher, Gabi Meier, Laurence Meunier, Mike Monaghan, Fredericke Moeslacher, Achim Pätzold, Daniel Pellanda, Franziska Pfister, Christiane Rapin, Chris Robinson, Andreas Schmidt, Diana Soldo, Piet Spaak, Barbara Sulzberger, Klement Tockner, Urs Uehlinger, Prof. J.V. Ward, Verena Wenzelides, Monika Winder, Chihiro Yoshimura, and Rainer Zah for being a wonderful set of colleagues.

For the translation of the summary into German and Italian I extend my gratitude toward Sophie Karrenberg and Silvio Blaser, respectively.
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