Geomorphology 214 (2014) 157–167

Contents lists available at ScienceDirect

Geomorphology

journal homepage: www.elsevier.com/locate/geomorph

The interplay of sedimentation and carbon accretion in riparian forests

Isaak Rieger [a,c,*], Friederike Lang [b], Ingo Kowarik [a,c], Arne Cierjacks [a,c,d]

[a] Department of Ecology, Ecosystem Science/Plant Ecology, Technische Universität Berlin, Rothenburgstraße 12, 12165 Berlin, Germany
[b] Chair of Soil Ecology, Institute of Forest Sciences, University of Freiburg, Bertoldstraße 17, 79085 Freiburg i. Br., Germany
[c] Berlin-Brandenburg Institute of Advanced Biodiversity Research (BBIB), 14195 Berlin, Germany
[d] Biocenter Klein Flottbek, Biodiversity of Useful Plants, University of Hamburg, Ohnhorststraße 18, 22609 Hamburg, Germany

ARTICLE INFO

Article history:
Received 13 August 2013
Received in revised form 24 January 2014
Accepted 28 January 2014
Available online 12 February 2014

Keywords:
Carbon accretion rate
Carbon stock
Dendrogeomorphology
Dike
Floodplain forest
Sedimentation rate

ABSTRACT

Sediment trapping and organic carbon (OC) accretion in soil are crucial ecosystem services of floodplain forests. However, interactions between the two processes have scarcely been analyzed at the ecosystem level. This study aimed at quantifying OC accretion parameters (CAP, including sedimentation rate, OC concentration, OC accretion) over roughly the last 50 years on both sides of a dike in a Danubian floodplain forest in Austria. Additionally, we determined soil OC stocks (0–100 cm in depth) and modeled both CAP and OC stocks in relation to environmental parameters. Overall, mean sedimentation rate and OC accretion of the riparian forest were 0.8 cm y^{-1} and 3.3 t OC ha^{-1} y^{-1} and significantly higher in flooded riparian forest (FRF; 1.0 cm y^{-1} and 4.1 t OC ha^{-1} y^{-1}) than in diked riparian forest (DRF; 0.3 cm y^{-1} and 1.5 t OC ha^{-1} y^{-1}). In contrast, mean OC concentration (0.05 t OC m^{-3}) and OC stocks (238 t OC ha^{-1}) were significantly higher in the DRF than in FRF (0.05 vs. 0.04 t OC m^{-3} and 286 vs. 201 t OC ha^{-1}). Modeling revealed tree species, fluctuation of groundwater table, and the distance to the river as valuable indicators for OC accretion rate. The OC concentration and distance to the river were positively and sedimentation negatively correlated with OC stock. The dike was consistently ruled out as a significant predictor variable. Consequently, differences among FRF and DRF seem to be related rather to longer term processes during the last centuries than directly to the dike. Our findings highlight the relevance of sediment quality (i.e., OC concentration) for building up long-term soil OC stocks, whereas sediment quantity is the main driver of recent OC accretion rates.

1. Introduction

Wetlands fulfill various ecosystem services (Mitra et al., 2005) of which sediment trapping and organic carbon (OC) sequestration have been repeatedly highlighted (Phillips, 1989; Cavalcanti and Lockaby, 2005; Samaritani et al., 2011). Although wetlands cover only about 5–8% of the Earth's land area (Mitsch et al., 2012), they account for 20–30% of the total C stored in the Earth's upper surface layer (Bridgham et al., 2006; Mitsch et al., 2012). In contrast to an array of northern peatlands (Gorham, 1991; Turunen et al., 2002; Bridgham et al., 2008) and tropical coastal wetlands (Chmura et al., 2003; Suratman, 2008; Howe et al., 2009; Mitsch et al., 2012), OC dynamics of periodically flooded temperate riparian forests are understudied (but see Cierjacks et al., 2010, 2011; Mitsch et al., 2012). Soils and aboveground biomass comprise together more than 90% of the overall OC stocks in temperate forests, while other OC pools such as fine roots and woody debris are less relevant (Cierjacks et al., 2010; Rieger et al., 2013). In soil, sedimentation of organic matter and on-site OC input from highly productive riparian forests are expected to be the main drivers of OC accretion. However, the dynamic processes of sedimentation and their links to the riparian OC cycle are not well understood.

In many river systems, dike constructions and other human interventions have substantially changed the flow regime — and presumably sedimentation processes and carbon storage capacity (Phillips, 2003; Reid and Dunne, 2003; Howe et al., 2009). This may explain why sedimentation regimes have mostly been studied in small rivers where water discharges average 11.7 up to 250 m^3 s^{-1} (Asselman and Middelkoop, 1995; Heimann and Roell, 2000; Cabezas and Comín, 2010) or watersheds smaller than 4000 km^2 (Walling and He, 1998; Craft and Casey, 2000; Noe and Hupp, 2005, 2009; Samaritani et al., 2011), where natural sedimentation processes are likely to persist. In contrast, sedimentation rates and related OC accretion have been considered less frequently in major floodplains, such as for the Rhine and Meuse (Asselman and Middelkoop, 1995), Garonne (Steiger and Gurnell, 2002), and the Ebro rivers (Cabezas and Comín, 2010). Many existing studies provide a quantification of sedimentation (e.g., Asselman and Middelkoop, 1995; Heimann and Roell, 2000; Hupp and

Abbreviations: BRT, boosted regression trees; DRF, diked riparian forest; FRF, flooded riparian forest; OC, organic carbon; TRF, total riparian forest.

* Corresponding author. Tel.: +49 3031471502; fax: +49 3031471355.
E-mail addresses: isaak.rieger@tu-berlin.de, isaakrieger@outlook.com (I. Rieger), fritzi.lang@bodenkunde.uni-freiburg.de (F. Lang), kowarik@tu-berlin.de (I. Kowarik), arne.cierjacks@tu-berlin.de, arne.cierjacks@uni-hamburg.de (A. Cierjacks).

http://dx.doi.org/10.1016/j.geomorph.2014.01.023

Bornette, 2003; Noe and Hupp, 2009) or OC concentrations (Asselman and Middelkoop, 1995; Steiger and Gurnell, 2002; Hupp et al., 2008; Samaritani et al., 2011) but focused on isolated aspects such as spatial gradients (i.e., Pizzuto, 1987; Walling and He, 1997, 1998; Petts et al., 2000; Hupp et al., 2008), the trophic state of a river (McTammany et al., 2003), or the particle size distribution of sediments (Walling and He, 1998; Petts et al., 2000; Steiger and Gurnell, 2002). Important ecosystem services such as carbon sequestration in soil have not been taken into account comprehensively. Furthermore, the results of these studies are based on single flooding events that preclude general conclusions on the long-term sedimentation regime. The mechanisms of sedimentation and OC accretion at the ecosystem level in riparian forests have not been disentangled thus far. To overcome these limitations, we aimed to quantify and model each aspect of recent OC accretion. Moreover, OC stock in the upper soil (0–1 m) was analyzed to test if recent OC accretion translates to long-term carbon storage in soil. Organic carbon accretion is a function of sediment increment per year (sedimentation rate) and the OC concentration of the deposited allochthonous and/or autochthonous material along with soil development (Zehetner et al., 2009). We therefore refer henceforth to these three parameters, i.e., carbon accretion, sedimentation rate, and OC concentration, as carbon accretion parameters (CAP).

We determined CAP by applying dendrogeomorphical methods (Hupp and Bazemore, 1993; Hupp and Bornette, 2003) in a riparian forest of a large European river in the Donau-Auen National Park, Austria. Dendrogeomorphical methods link tree age from increment cores to the depth of burial of the stem base since the germination of the studied tree. This approach ensures that sedimentation processes can be assessed over longer periods of time (i.e., tree age) as opposed to short-term measurements using artificial grass mats, (fire) clay and feldspar pads, or markers (e.g., Asselman and Middelkoop, 1995; Steiger and Gurnell, 2002; Noe and Hupp, 2005, 2009; Olde Venterink et al., 2006; Hupp et al., 2008). An additional survey of long-term, presumably stabilized, OC stocks (0–100 cm) allows us to consider the potential link between CAP and OC stocks at the ecosystem level.

In particular, we aimed (i) to quantify mean CAP since germination of the tree and OC stock of the top meter of soil in total riparian forest (TRF), and in the same area divided into flooded (FRF) and diked riparian forest (DRF); (ii) to analyze which environmental parameters (spatial gradients, hydrology, forest stand structure, sedimentation, and dike presence) influence CAP and soil OC stock, and (iii) to assess if OC accretion during approximately the last 50 years translates to long-term OC stocks in the top meter of soil in the study area.

2. Methods and materials

2.1. Study area

The study area is adjacent to the Danube River which at 2875 km is the second largest in Europe (BMU, 2003) and part of one of the largest near-natural riparian systems remaining in central Europe, the Donau-Auen National Park in Austria (Fig. 1). About 65% (6045 ha) of the total park area is covered by forests with the remainder in open grasslands and waterways. The Danube River water level fluctuates within a range of up to 7 m as a response to meltwater and precipitation in the upstream watershed (Drescher and Fraissl, 2006). River embankments have been fixed using riprap, and since the end of the nineteenth century, the Marchfeld dike has divided the area into a section where surface flooding occurs ca. every 5 years (C. Baumgartner, Donau-Auen National Park, Austria, personal communication, 2012) and a section that is protected from flooding by the dike. Data were collected from both river banks of the section between the villages of Schönau (48°8′ N., 16°36′ E., river kilometer 1910) and near Hainburg (48°1′ N., 16°88′ E., river kilometer 1889), including areas of both flooded and diked floodplain forests (Fig. 1). Soils are fluvisols (calcaric, eutric) and gleysols (haplic, calcaric) (Sali-Bazze, 1981; Cierjacks et al., 2010). The nearest climate station (Schwechat, 48°7′ N., 16°34′ E.; 184 m asl) reports an annual mean temperature of 9.8 °C and a mean annual precipitation of 533 mm (1948–2008; Zentralanstalt für Meteorologie und Geodynamik, 2002). The area is part of the upper reaches with a fast flow regime (rhithral character) and was classified as the furcation type and order 9 (Wimmer and Moog, 1994). The main channel is ca. 350 m wide, has a mean annual discharge of 1950 $m^3\ s^{-1}$, and a slope of 0.045%. The velocity of the surface water ranges from 1.9 to 2.2 $m\ s^{-1}$ (Tockner et al., 1998).

2.2. Study design

To cover a wide range of environmental gradients in space and time, the study area was stratified into three lateral zones (flooded riparian

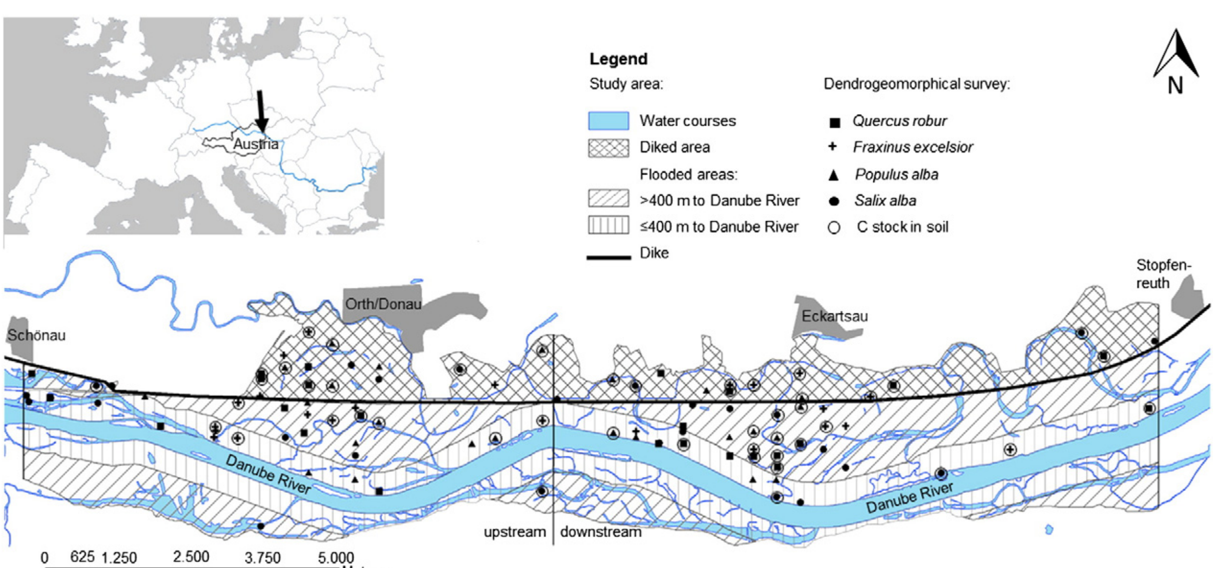

Fig. 1. Study area within the Donau-Auen National Park and study design (tree symbols refer to individually sampled trees).

forest south of the Marchfeld dike <400 m and >400 m from the main channel, and diked riparian forest north of the dike). All three lateral zones were further divided into an upstream and a downstream part (east and west of the village Orth/Donau, respectively; Fig. 1). In each of the resulting six zones, we randomly selected four sample canopy trees (diameter at breast height (DBH) >10 cm) out of dominant tree species of the three representative forest types. In the hardwood forest, *Quercus robur* and *Fraxinus excelsior* were chosen; *Populus alba* was the focus species in cottonwood forest, and *Salix alba* in softwood forest. Because any of the selected tree species reflect specific microsite conditions, this approach allows us to address the same representative habitats on both sides of the dike, which differ exclusively in their surface inundation regimes. To detect possible temporal variations in CAP, we sampled the same amount of younger and older trees, indicated by a DBH of <25 or >25 cm, respectively. Tree selection was performed using the National Park's forest inventory database applied on a 100 × 400 m grid. This approach resulted in 6 zones × 4 tree species × 4 replicates = 96 sample trees, which were studied by dendrogeomorphical methods to determine CAP. Additionally, the OC stock of the soil (0–1 m in depth) was studied exemplarily at 46 of the 96 sample sites to study its relation to OC accretion rate.

2.3. Dendrogeomorphical methods

We used the depth of the tree base below or above ground level as a measure of sedimentation or erosion, respectively, during a tree's life-span and related it to its age as determined by tree ring counts (Hupp and Bornette, 2003). For tree ring analysis, we used an increment borer (SUUNTO©) to extract an increment core measuring up to 40 cm in length and 0.5 cm in diameter from the eastward trunk side of each sample tree at 130 cm height above the tree base. All cores were prepared according to standard dendrochronological methods (Schweingruber, 1989) by using a scalpel blade (Swann-Morton©, non-sterile, carbon steel, type 23) to ensure a surface that could easily be analyzed (free of abrasive dust and smooth). In the case of questionable tree rings, we took a second core to detect missing or wedging rings. The number of tree rings was counted from bark to pith to determine tree age if the pith was hit by coring. If the pith was not included in the core, the number of missing tree rings was estimated by using a curve template that allowed for the determination of the position of the inner rings in relation to the center. Tree rings down to a precision of 0.01 mm were counted with the help of a measuring table (ANIOL 1986, type 180) and a binocular stereo microscope (ZEISS©). Tree rings were dated using the software TSAP Professional© 4.67.

Over the life-span of floodplain trees, beginning at germination, deposited sediment buries the trunk of a tree, while erosion leads to excavation of the stem base. At all of the studied sites we observed net sedimentation over the life-span of the trees. Because initial roots at the tree base of the tree seedling are a distinctive marker of the original ground surface (Hupp and Bornette, 2003), we excavated the initial root zone, defined as the root collar of each sample tree, with a spade and measured the distance from the present ground surface (exposed trunk) to determine the depth of accreted sediment. Subsequently, the depth of the accreted sediment was divided by tree age to calculate the sedimentation rate. Because this approach includes possible erosion events, the reported values on sedimentation rate refer to net sedimentation. The OC accretion was determined based on the upper 5 cm of sediment deposits that were sampled adjacent to each tree using a soil sampling ring (height 40.5 mm, diameter 56 mm). This layer represents the youngest layer of sediments with the shortest period of autochthonous OC input by soil development, which thereby is most representative for OC accretion. Moreover, the majority of study trees showed a sediment layer of at least 5 cm in depth even on DRF sites; at these sites, additional soil layers above stem bases presumably derive from autochthonous OC input. Therefore, our approach enabled a consistent sediment sampling at all study trees. To determine OC accretion since germination of the tree, we multiplied OC content of the sediment in 5 cm depth (based on OC concentration as percentage and bulk density) by sedimentation rate. For deeper sediment layers, we assumed a constant OC accretion over time. Despite the fact that different sediment layers may differ in OC concentration and bulk density normally increases with soil depth, the sampling method is expected to lead to representative results as the median of sediment depth was at 20.0 cm and usually covers a single soil horizon also at softwood sites (median of the upper soil horizon's depth 16.0 cm). At typical hardwood sites, we have to expect a large contribution of autochthonous OC to total OC content and consequently the formation of an Ah horizon that may add to OC accretion by sedimentation. Extraordinary high sediment deposits (>60 cm, n = 10) were composed of sandy, more or less homogenous sediments without visible signs of soil diagenesis. Soil samples were dried to a constant weight at 105 °C, and density was calculated as weight per volume. Based on the mean tree age, the derived sedimentation rates cover an average time span of 49.5 years (standard error = 2.8 years).

2.4. Soil organic carbon

To determine carbon accretion rate by sedimentation, the OC concentration of sediment samples (5 cm) was measured in a CN analyzer (Elementar). Air-dried soil samples were sieved (<2 mm) and homogenized by milling (Retsch MM2®). Because of the high inorganic carbon (carbonate) ratio, we performed dual temperature combustion. A 2-g subsample was transferred to a muffle oven (Heraeus M110®) and burned for 2 h at 220 °C and 440 °C, respectively, and for 6 h at 550 °C within a single run to eliminate OC. About 17 mg of the muffled sample and about 12 mg of the unburned soil were transferred to the CN analyzer. The OC is completely oxidized to CO_2 at low temperatures (550 °C), while inorganic C is combusted at high temperatures (1000 °C) only (Bisutti et al., 2007). Organic C concentration was calculated as the difference between total C percentage and inorganic C percentage. We further considered the bulk density to derive the volume-related OC concentration in deposited sediments in t C m^{-3}. The OC accretion rate was calculated as the product of OC concentration and sedimentation rate. Because this method does not allow for the differentiation of allochthonous OC and autochthonous OC in the topsoil, our OC accretion rate comprises both possible sources.

To determine long-term soil OC stocks, one soil core of the upper meter of soil (soil OC stock) was extracted with an auger (Pürckhauer) at a distance of 2 m from 46 of the 96 sample trees. Horizons were distinguished in the field by differences in color and soil texture based on the finger testing method (Ad-hoc AG Boden, 2005). Depth, thickness, and soil texture were recorded and a series of soil samples from depths of 5, 10, 30, 50, 80, and 100 cm were taken. The corresponding bulk densities were derived from the most similar soil horizon of reference profiles (data from Cierjacks et al., 2010) in terms of soil texture, depth, thickness, and aboveground vegetation. From each core, the series of soil samples was mixed according to the relative thicknesses and bulk densities of the corresponding soil horizons. The resulting homogenized composite sample was analyzed using dual combustion and CN analyzer as described for sediment samples. Soil OC stock was quantified based on OC concentration and bulk density estimates. As an approximation of the maximum age of the soil OC stock, we calculated the quotient of 100 cm soil depth (cm) and the site-based observed sedimentation rate (cm y^{-1}) of the corresponding plot.

2.5. Spatial and hydrological parameters

The location of each sample tree was recorded in February and March 2010 using a handheld differential Trimble® GeoXH™ 2005 series. Position data were later corrected with Rinex data (station: Leopoldsau) to reduce the deviation from the real position in the field <50 cm. We used the NEAR tool (ArcInfo) in ArcGIS 9.2 to assess the lateral (distances to Danube River and the nearest side channel) and longitudinal gradients (distance to the western limit of the study area).

Altitude above mean water level of the Danube River was calculated by subtracting the altitude above the sea level (asl) of the mean water level at the corresponding river kilometer (shortest distances of each sample tree to Danube River using the NEAR tool, data on water level provided by via donau-Österreichische Wasserstraßen-Gesellschaft mbH) from the altitude asl of the sample tree position taken from a digital elevation model provided by via donau-Österreichische Wasserstraßen-Gesellschaft mbH.

Groundwater parameters were calculated based on a groundwater model provided by the Vienna University of Technology. The model provides the altitude of mean and low groundwater tables for each site in the study area. We used the Extract Values to Points tool (ArcInfo) in ArcGIS 9.2 to calculate depths to the mean and low groundwater levels as the differences in the digital elevation model and the groundwater model. The magnitude of fluctuation in the groundwater level was calculated as the difference between the mean and low groundwater levels.

2.6. Forest stand structure parameters

Forest stand structure in the vicinity of the sample trees was analyzed by angle count sampling with the sample tree as a center using a Dendrometer II (see Kramer and Akça, 2002; Rieger et al., 2013). For each tree selected by angle count sampling, we determined tree species, DBH, and tree height (BLUME-LEISS BL 6, trigonometrically) and calculated stem number, basal area per hectare, and mean stand height and DBH. Covers of tree, shrub, and herb layers were estimated visually.

Mean diameter and length of lying woody debris (>10 cm in diameter) were measured in a sampling circle of 16 m in diameter around the central sample tree. The resulting volume values were transformed to biomass assuming a mean wood density of 0.6 t m^{-3} (Chave et al., 2006). Annual carbon increment of the sample tree was calculated as total C stock divided by tree age. Biomass of the sample tree was determined using allometric equations compiled by Zianis et al. (2005) (Table 1). For *S. alba*, we multiplied the allometric equation for stem volume by 0.5321 to derive stem biomass as described by Lehtonen et al. (2004). Resulting biomass value of woody debris and sample trees was multiplied by 0.5 to estimate OC stocks (Giese et al., 2003).

2.7. Statistics

All continuous variables were tested for outliers, normality (Shapiro–Wilk test), homogeneity of variance (Fligner test), and collinearity (VIF values) (Fox, 2008; Zuur et al., 2010). Because of the violation of normality and homogeneity, differences in the depth of accreted sediment, bulk density, soil age, and CAP response variables, namely between flooded vs. diked riparian forests, were tested using generalized linear models (GLM). For bulk density and OC stock in soil, we used the Welch two-sample *t*-test. Possible correlations among soil parameters such as bulk density, sedimentation rate, OC concentration, CAP, and OC stocks were tested using Pearson's product-moment correlation.

To analyze the impact of environmental parameters on CAP and OC stock in soil, we applied a boosted regression trees (BRT) analysis, which is a technique that combines a regression tree approach (De'ath and Fabricius, 2000) and boosting to improve model accuracy (Schapire, 2002). As the relative importance of predictors and the fitted functions for each variable are computed, interpretation of the results is straightforward (Elith et al., 2008). Furthermore, BRT analysis allows for combining nominal and metric-scaled data as well as for detecting interactions among predictor variables. The set of environmental parameters used for the analysis of sedimentation rate, topsoil OC concentration and accretion, and OC stock in soil included (i) spatial parameters, i.e., distances to main channel, nearest side channel, upstream limit of study area, and diking (waterside vs. landside); (ii) hydrological parameters, i.e., distances to mean water level and mean/low groundwater level, and magnitude of fluctuations in groundwater table; (iii) forest stand parameters, i.e., dominant tree species, number of tree species, stem number, basal area per hectare, lying deadwood as well as coverage of herb, shrub, and tree layers; and (iv) soil parameters, i.e., particle size, bulk density, sand content, sedimentation rate, and OC concentration. For each individual model calculated, we excluded environmental parameters that directly served as determinants for the response variable (e.g., for modeling OC accretion by sedimentation we excluded OC concentration in the topsoil, bulk density, and sedimentation rate). For modeling soil OC stocks, we excluded the mean OC concentration of the entire soil profile because of the direct statistical link of this variable to soil OC stock. Instead, we used the OC concentration of the topsoil that was measured independently from soil OC stocks and characterizes the recent sedimentation regime. All calculations were performed with R version 2.10.0 using the packages "AED", "brt", "car", "gbm", "gtools", "mass", and "vegan" (R Development Core Team, 2012).

3. Results

3.1. CAP and soil OC stock in flooded and diked forests

Overall sedimentation rate was 0.8 cm y^{-1} with significantly lower rates for DRF compared to FRF (Table 2). The depths of accreted sediment in the total floodplain forest ranged from 0 to 140 cm since germination of the sample trees with a median of 20 cm (75% of all depths were ≤32.0 cm). Bulk density in 5 cm depth of the accreted sediment was almost normally distributed ($P = 0.02$), ranged from 0.51 to 1.27 g cm^{-3} with a mean (S.E.) of 0.96 (0.02) g cm^{-3}. Depth of accreted sediment and bulk density showed significantly higher values in FRF compared to DRF ($P < 0.001$ in both cases). In contrast, OC concentration was significantly higher in sediments of DRF than of FRF. The resulting OC accretion of DRF was significantly lower than that of FRF. The OC stocks of the first meter of soil were significantly higher in DRF

Table 1

Allometric equations for biomass calculations (abbreviations: AB = aboveground biomass, ABW = aboveground woody biomass, SV = stem volume, DBH = diameter at breast height, H = height, Z = Zianis et al., 2005, App. = Appendix).

Species	Equation	Units of					Source
		AB	ABW	SV	DBH	H	
Quercus robur	$\ln(AB) = -0.883 + 2.14 \times \ln(DBH)$	[kg]			[cm]		Z, App. A, #600
Fraxinus excelsior	$\ln(ABW) = -2.4598 + 2.4882 \times \ln(DBH)$		[kg]		[cm]		Z, App. A, #134
Populus species	$AB = 0.0519 \times DBH^{2.545}$	[kg]			[cm]		Z, App. A, #514
Salix alba	$SV = -1.8683 + 0.2146 \times DBH^2 + 0.0128 \times DBH^2 \times H^2 + 0.0138 \times H^2 \times DBH - 0.0631 \times H$			[dm^3]	[cm]	[m]	Z, App. C, #218; we used equation for *Salix caprea*

Table 2
Carbon accretion parameters (CAP) and organic carbon (OC) stock in soil in flooded, diked, and total riparian forest (means and standard errors); superscript letters indicate significant differences ($P < 0.05$) among flooded and diked riparian forest (generalized linear models for CAP and soil age, and Welch two sample *t*-test for soil OC stocks).

Riparian forest	Flooded	Diked	Total
Sedimentation rate [cm y^{-1}] (n = 96)	1.0 ± 0.1^b	0.3 ± 0.0^a	0.8 ± 0.1
OC concentration in topsoil (5 cm) [t C m^{-3}] (n = 96)	0.044 ± 0.001^a	0.050 ± 0.003^b	0.046 ± 0.001
OC accretion in topsoil [t C ha^{-1} y^{-1}] (n = 96)	4.1 ± 0.7^b	1.5 ± 0.2^a	3.3 ± 0.5
Soil OC stock (0–1 m in depth) [t C ha^{-1}] (n = 46)	201.2 ± 13.2^a	286.0 ± 26.8^b	238.0 ± 15.0
Estimated soil age (0–1 m in depth) [years]	242.3 ± 43.2^a	360.5 ± 54.2^a	293.7 ± 34.7

compared to FRF, and soil age was marginally but significantly higher at DRF sites (Table 2).

3.2. Environmental drivers of CAP and soil OC stocks

Surprisingly, in none of the BRT models was the presence of the dike selected as an influential variable, despite the differences found in DRF and FRF. The CAP and soil OC stocks could consistently be predicted by other more influential environmental parameters. Moreover, we did not detect any interactions among predictor variables within the final models shown here.

The BRT model on sedimentation rate explained 27% of the total variance (see Table 3); and tree species, the magnitude of fluctuation in the groundwater table, the distances to the Danube River and the nearest side channel, and the DBH of the sample tree were identified as the five most indicative variables (Fig. 2). Above average rates of sedimentation were found under *S. alba* and in areas where the groundwater fluctuated >1.4 m. Moreover, higher values could be expected within 600 m of the Danube River and within 100 m of the nearest side channel. A high rate of sedimentation was also indicated for trees with a DBH <30 cm.

Modeled OC concentration of the topsoil was associated with the rate of sedimentation, the distance to the Danube River, lying deadwood, depth of the low groundwater level, C increment of the sample tree, and sand content (Fig. 3). The predictive power of the BRT model was 19% (Table 3). The OC concentration increased with lower sedimentation rates (i.e., highest rates were found at sites with sedimentation rates below 0.5 cm y^{-1}) and with greater distance to the Danube River — in particular at a threshold of 1400 m. Above average OC concentrations of the topsoil material were also associated with small amounts of lying deadwood (<25 t OC ha^{-1}), shallower low groundwater levels (<3.5 m), increased annual C increment of the sample tree (>12 kg C y^{-1}), and sand contents of the topsoil horizon below ~10%.

The BRT model for OC accretion rate explained 24% of the variation in the dependent variable (Table 3). The four most indicative variables were magnitude of groundwater fluctuation, tree species, distance to the nearest side channel, and distance to the Danube River. Above average OC accretion rates were found at sites with a groundwater fluctuation >1.4 m, those dominated by *S. alba*, and those within <100 m of the nearest side channel and within around 600 m of the Danube River (Fig. 4).

Our analysis of soil OC stock detected three sedimentation parameters (OC concentration, sedimentation rate, and bulk density) and the distance to the Danube River as the most indicative variables, explaining a total of 33% of the variation in the dependent variable. The fitted functions point to above average OC stocks in soil at sites characterized by a high OC concentration in the deposited sediment (>5%) or by a low sedimentation rate (<0.5 cm y^{-1}) and bulk density (<0.9 g cm^{-3}). Moreover, OC stock in soil was higher at sites >1400 m from the Danube River (Fig. 5).

4. Discussion

4.1. CAP and soil OC stock in flooded and diked forests

This study allows the mean CAP of approximately the last 50 years to be estimated by combining dendrogeomorphical methods with the analysis of OC concentration and to relate these to environmental variables and soil OC stocks.

Our study revealed a higher sedimentation rate and OC concentration (0.8 cm y^{-1} and 0.05 t C m^{-3}, respectively) than many previous studies that relied mainly on short-term observations. Mean sedimentation rates range from 0.17 to 0.54 cm y^{-1} in alluvial coastal floodplains in the U.S. (Hupp and Bornette, 2003) and were estimated as 0.18 cm y^{-1} in forested floodplains of seven rivers within the Chesapeake Bay watershed (Noe and Hupp, 2009). Our sediment deposits (derived from sedimentation rate and bulk density of the deposited sediment) were almost eight times higher than the sediment deposition from the River Waal — the main branch of the Rhine River (Asselman and Middelkoop, 1995) or about twice as high as from the Ebro River

Table 3
Documentation of all calculated boosted regression tree models (OC = organic carbon).

Dependent variable	Sedimentation rate	OC concentration in topsoil	OC accretion rate	Soil OC stock
Distribution	Poisson	Gaussian	Poisson	Poisson
Bag fraction	0.5	0.5	0.5	0.6
Number of trees	10,000	10,000	10,000	10,000
Mean total deviance	7.873	0	3.635	37.871
Mean residual deviance	5.72	0	2.759	25.452
Explained deviance	27%	19%	24%	33%
Estimated CV deviance	6.619	0	3.185	32.173
Estimated CV deviance (se)	1.6	0	1.196	11.329
Training data correlation	0.615	0.578	0.526	0.637
CV correlation	0.376	0.376	0.47	0.545
CV correlation (se)	0.125	0.099	0.07	0.07

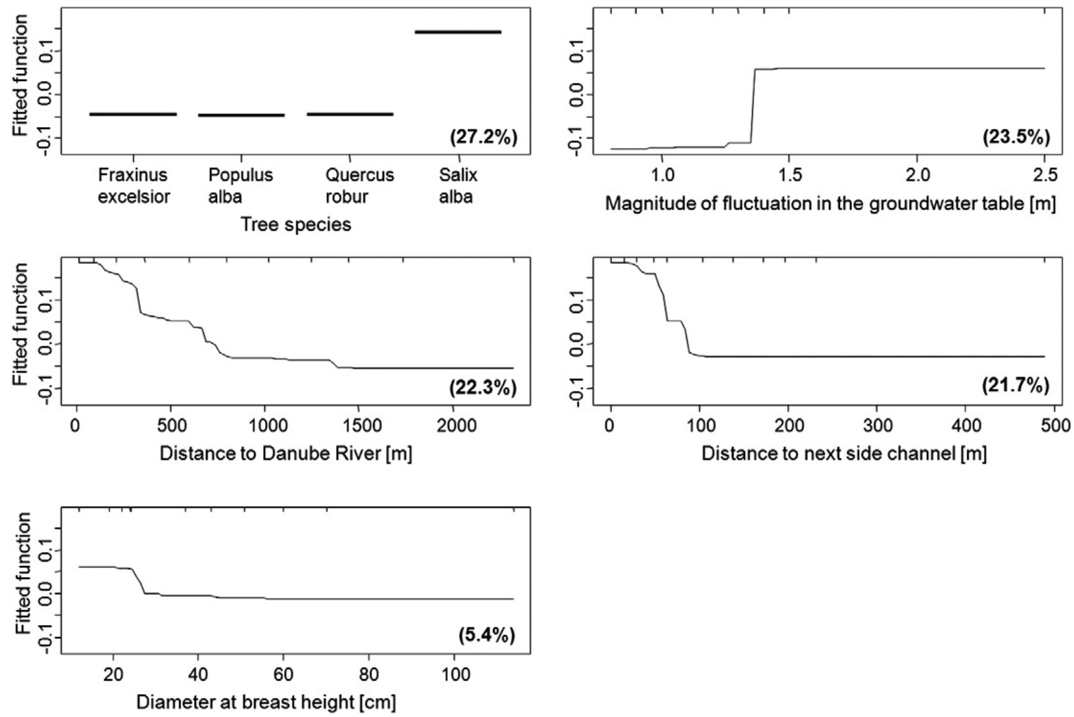

Fig. 2. Partial response plots of the five most indicative variables in the model for sedimentation rate (bold values in parentheses indicate the relative influence of the explained variance in the boosted regression tree model).

(Cabezas and Comín, 2010). While our study area is part of the upper reaches of the Danube River, Asselman and Middelkoop (1995) investigated sediment depositions in the lower part of the Rhine River in the Netherlands. As fine sediments prevail close to the delta of the river, coarse sediments as observed in our study may not be present at the study site in the Netherlands. For an uneven-aged riparian forest in northern Missouri, Heimann and Roell (2000) reported similar values (1.04 cm y^{-1}) as in our study, considering a time span of 27 years. Higher sedimentation rates for long-term observations point to the relevance of extreme sedimentation events that occur rarely and are

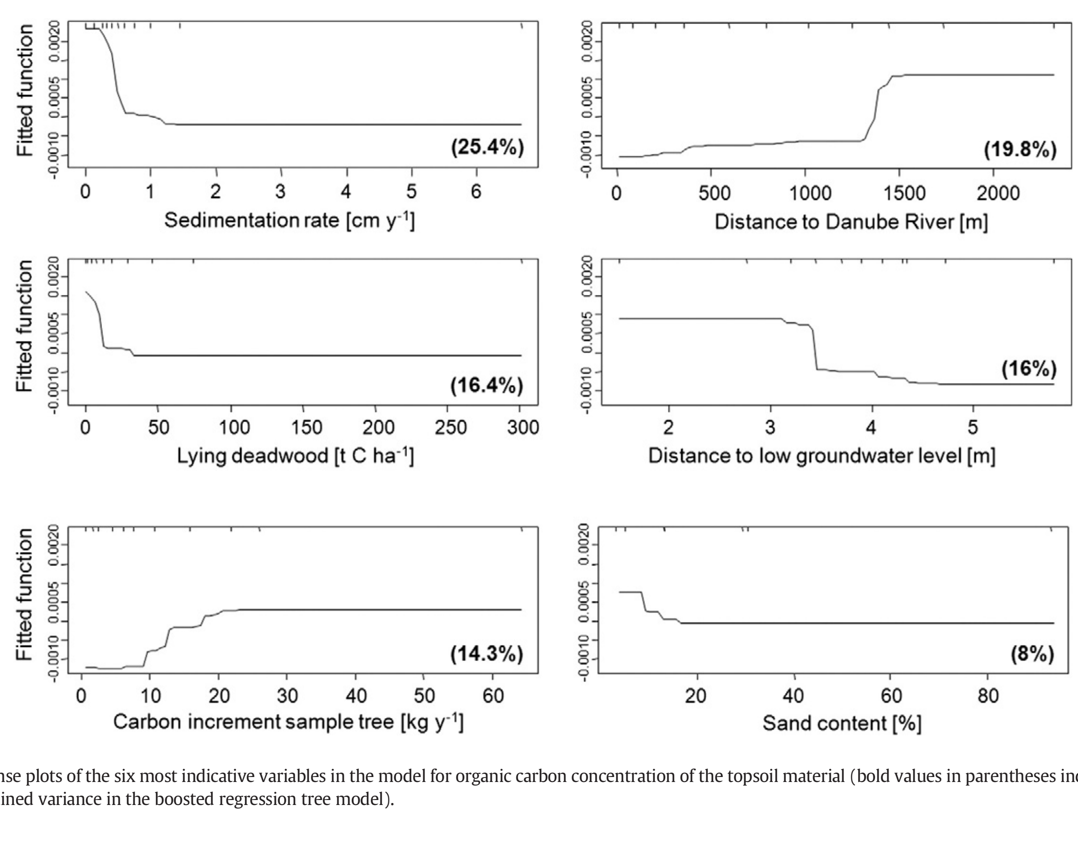

Fig. 3. Partial response plots of the six most indicative variables in the model for organic carbon concentration of the topsoil material (bold values in parentheses indicate the relative influence of the explained variance in the boosted regression tree model).

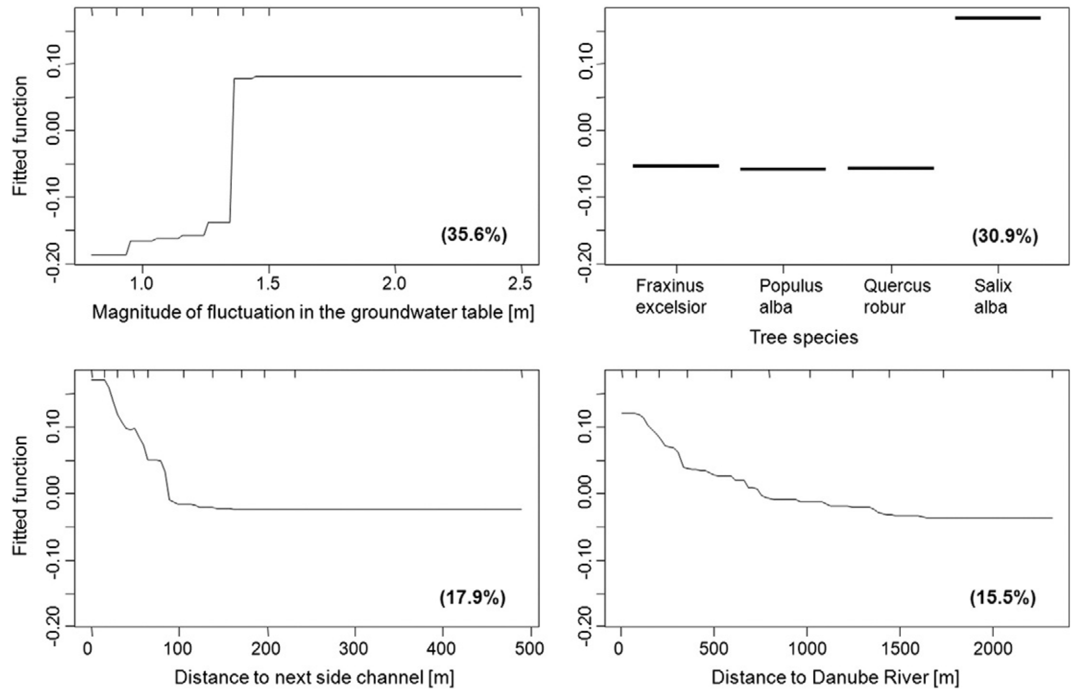

Fig. 4. Partial response plots of the four most indicative variables in the model for organic carbon accretion (bold values in parentheses indicate the relative influence of the explained variance in the boosted regression tree model).

therefore probably not captured by short-term studies. In our study region, sediment deposition of about 1 m during a single flood event has been observed (Christian Fraissl, Donau-Auen National Park, personal communication, 2012). Despite possible methodological constraints in identifying the initial root zone and determining tree age, the magnitude of our data seems rather robust as the timescale of the measurements — at about 50 years — is quite long and should compensate for those errors.

Erosion seems to play a minor role in our study area. A pronounced erosion of sediment deposition is often related to channel migration (Reid and Dunne, 2003), which is widely excluded in the study area by riprap-fixed shores. Hupp (2000) also found net sediment deposition as opposed to erosion for most of the bottomland floodplains. This is also supported by personal field observations, where only 6% of all studied plots showed minor signs of erosion after flooding events (3 June 2010, 15 January 2011).

Mean OC concentration of the topsoil material of the total riparian forest (0.046 t C m^{-3}) was higher than in other studies (Cabezas and Comín, 2010; Samaritani et al., 2011). Also Graf et al. (2007) found only 2.2% OC in the topsoil for forested sites in the same study area, but

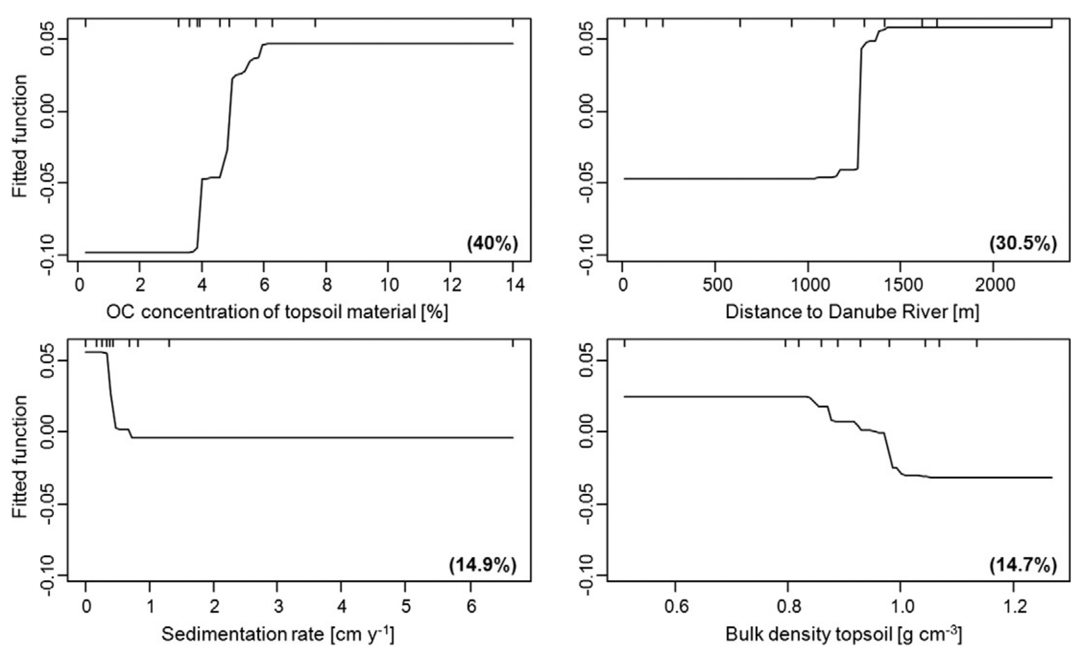

Fig. 5. Partial response plots of the four most indicative variables in the model for organic carbon stock in soil (bold values in parentheses indicate the relative influence of the explained variance in the boosted regression tree model).

the authors report only limited data for forest sites (n = 3) one of which is moreover located on an island characterized by coarse sediments and extremely low OC concentration. In contrast, Craft and Casey (2000) found slightly higher OC concentrations (0.053 t C ha^{-1}, derived from reported OC and bulk density) for the upper 30 cm soil of floodplain forest in Georgia. Surprisingly, overall variation in the volume-related OC concentration of the topsoil material was remarkably low (Table 2), although the OC concentration on a mass basis ranged from 0.2 to 14.0% (mean = 4.8%, standard error = 0.19%). This variation was apparently offset through the inverse relationships of OC concentration and bulk density (Pearson's $r = -0.549$, $P < 0.001$) as well as OC concentration and sedimentation rate (Pearson's $r = -0.261$, $P < 0.001$). The lower OC concentration values in other studies can presumably be attributed to smaller watersheds (lower organic material input), smaller lateral gradients (max. 200 m), and the inclusion of a broader range of functional riparian zones, e.g., gravel/coarse sediment bars, compared to our study area. Organic carbon concentration in the topsoil from allochthonous input (i.e., sedimentation) mainly depends on the amount of OC transported by the river and subsequent deposition during surface flooding. The river continuum concept (Vannote et al., 1980) provides a conceptual model to explain the OC supply to large rivers from upstream vegetation and catchment soils as main carbon sources and its downstream transport (Olley, 2002). The size of the watershed along with associated land use and climate is considered to be the main factor that drives the amount of organic material transported in the channel. Transported organic matter generally increases downstream (Minshall et al., 1992). In addition, sediment retention capacity of floodplains is positively correlated with basin size (Phillips, 1989), and lower flow velocities in huge basins lead to increased deposition of sediments (Pinay et al., 1992), which may explain the higher OC concentrations in our study. Higher OC concentrations may also be caused by the inclusion of older sediments. Most of the study sites (87%), tended to accumulate humus in the topsoil, which implies autochthonous OC input into the deposited sediment (5 cm) from fine roots (Rieger et al., 2013; data from additional 48 permanent study plots in the same area) or litterfall compared to sites closer to the river (Cabezas et al., 2009). The OC accretion rate found in our study (3.3 t C ha^{-1} y^{-1}) is within the range of 0.02 to 4.8 t C ha^{-1} y^{-1} (wetland studies from all over the world reviewed by Mitsch et al., 2012) but is only exceeded in a *Cyperus* wetland in Uganda (Saunders et al., 2007). Accordingly, our results are about five times higher than the mean OC accretion rate of forested plots from the Ebro River (Cabezas and Comín, 2010). In contrast to our study, Cabezas and Comín (2010) studied OC accretion rates based on individual flooding events, which may occur several times a year. In these cases, annual OC accretion rates may have been underestimated owing to the exclusion of extreme sedimentation events. This assumption is supported by a reported long-term OC accretion rate for forests (period: 1963–2007) in the same study area that was only twofold lower compared to our results (Cabezas et al., 2009). Our study revealed OC accretion rates at least three times higher compared to floodplain forests in the U.S. (Noe and Hupp, 2009), five rivers in the UK (Walling and He, 1998), and sections of the Rhine and Danube in Germany (Hoffmann et al., 2009). This may again be attributed to the fact that alluvial wetlands like the Danubian floodplain with a large basin have a greater capacity of trapping sediment than smaller river basins (Phillips, 1989). For instance, the subwatershed in our study (104,000 km^2) was clearly larger than that of the Ebro River (85,362 km^2). Zehetner et al. (2009) reported carbon accretion rates of at least 1 t C ha^{-1} y^{-1}, but their data refer to autochthonous net carbon accretion in contrast to our data, which include allochthonous and autochthonous OC.

The OC stock of the top meter of soil for the TRF (238 t C ha^{-1}) was higher compared to 177 t C ha^{-1} determined by Cierjacks et al. (2011) in the Donau-Auen National Park. However, our study considered a broader lateral gradient that included the diked riparian forest and therefore more carbon-rich hardwood forest soils (Cierjacks et al., 2010). Still, both studies show considerably higher soil OC stocks compared to other forests in Germany (50–150 t C ha^{-1}; Hofmann and Anders, 1996; Wiesmeier et al., 2012).

The CAP and soil OC stock showed clear differences between DRF and FRF. As expected, sedimentation rate and OC accretion rate were higher in FRF than in DRF (the fact that net sedimentation was detectable in DRF may be a consequence of OC input from litter fall). In contrast, OC concentration was significantly lower in FRF. This implies that the sedimentation rate is more important for OC accretion rate than the OC concentration of the topsoil material. Similarly, OC stock in soil was lower in FRF, which indicates that long-term OC stocks of the top meter soil are instead related to sediment quality and the stabilization of OC rather than to absolute sediment amount. To shed light on these mechanisms, we modeled CAP and soil OC stocks based on environmental variables.

4.2. Environmental drivers of CAP and soil OC stocks

Despite pronounced differences in CAP and soil OC stock between flooded and diked forests, dike presence was consistently ruled out as a predictor variable in the BRT analyses. All CAP and soil OC stocks were significantly related to the distance to the Danube River and both dike presence and distance to Danube River reflect the lateral gradient of the hydrosystem; dike presence seems to become less important when modeled together with the distance variable. In accordance, the calculated mean distance of the dike to the Danube River was around 1150 m with sample trees within FRF showing a mean distance of 441 m from the main channel, compared to 1458 m for sample trees of DRF. As the dike separates floodplain sections with naturally pronounced differences in sedimentation regime, our data indicate differences; but the underlying driver is the lateral gradient rather than the dike specifically. The relative influence of the dike presence was low when included into the model alone, at 4% for the OC stock zero model and <1% for the BRT models of CAP (data not shown). Notwithstanding extraordinarily high OC accretion rates at sites near the river, soil OC stocks were lower here (Table 2). Consequently, the observed discrepancy points instead to other aspects of long-term stabilization processes and autochthonous soil OC sequestration. Furthermore, it seems possible that OC accumulation from autochthonous sources is periodically interrupted by sedimentation at sites near the river, whereas at sites farther from the river greater time periods are available for autochthonous OC sequestration.

Sedimentation rate and OC accretion rate mainly responded to the same indicator tree species and the same environmental parameters, such as hydrological and lateral gradients. This supports the assumption that OC accretion is driven more by sediment amount than by the OC concentration.

A significantly higher sedimentation rate was found at sites dominated by *S. alba* (Fig. 2). This corresponds to this species' excellent adaptation to highly dynamic riverine habitats: it tolerates destructive flooding regimes because of its soft timber (Jackson and Attwood, 1996), vegetative reproduction (Barsoum, 2002), formation of adventitious roots (Koop, 1987; Timoney and Argus, 2006), and abundant germination on site with new sediment deposits (Karrenberg and Suter, 2003).

Sedimentation rate was also related to forest stand parameters (Fig. 2). A mean DBH of <30 cm was associated with above average sedimentation rates, probably due to greater tree mortality in highly dynamic river sections (Hupp and Bornette, 2003). Stand age and mean tree diameter are generally lower at high sedimentation sites (Everson and Boucher, 1998). The occurrence of young *S. alba* trees with a DBH <30 cm proved to be a valuable indicator for sedimentation rate. This corresponds to the prevalence of such trees in direct proximity of the main and side channels where contemporary sedimentation is highest. In contrast, thicker trees of *S. alba* with a DBH >30 cm, which usually dominate other sites such as historic river beds, probably point to high sedimentation in the past.

The OC concentration of the topsoil responded inversely to sedimentation rate (Fig. 3). This pattern results from the deposition of coarse, sandy sediments at high sedimentation rates (Asselman and Middelkoop, 1995; Samaritani et al., 2011). High sedimentation rates (>2 cm y^{-1}) are associated with low concentrations of soil organic matter (Hupp et al., 2008), which is particularly true for carbonate-rich, young alluvial soils at locations close to the river (Guenat et al., 1999).

Still, this lateral pattern of OC concentration in the topsoil was modified in the complex spatial mosaic of floodplains as indicated by the other predictor variables selected in BRT analysis (i.e., depth of low groundwater level, lying woody debris, OC increment in sample tree, sand content). Sites with shallower groundwater levels <3.5 m below ground were frequently found at greater distances from the Danube River in areas with rather low sedimentation rates (Rieger et al., 2013). Moreover, shallower groundwater tables are typical for sites such as (ephemeral) ponds and swales that mark the course of former riverbeds (Hupp, 2000). Consequently, the distance to mean groundwater level and elevation above mean water level of the Danube River were significantly correlated (Pearson's $r = 0.927$, $P < 0.001$, data not shown). At these depressions, i.e., ponds and swales, extraordinarily high fractions of suspended silty and clayey carbon-rich sediment can be deposited (He and Walling, 1998). Accordingly, a low sand content relates to high carbon concentration in the sediment (Hoffmann et al., 2009; Cabezas and Comín, 2010). Large amounts of lying deadwood are most commonly associated with softwood riparian forests that usually show increased tree mortality and high translocation rates of uprooted stems (Cierjacks et al., 2011). In parallel, stand opening by canopy dieback and continuous mechanical stress may be expected to lead to rather young stands with lower annual OC increments compared to hardwood forests.

An above average OC accretion rate was attributed to a fluctuation in the groundwater table of >1.5 m in BRT analysis (Fig. 4), and these sites were most commonly found in direct proximity of the Danube River (Rieger et al., 2013). While previous studies reported the highest sediment deposition rates at low elevation levels such as ponds (Olde Venterink et al., 2006; Hupp et al., 2008), we found extraordinarily high values (up to 6.7 cm y^{-1}) at higher elevation sites close to the main and side channels. The corresponding sediments consisted mainly of fine sand. On the other hand, less pronounced fluctuations in the groundwater table commonly indicate geomorphologic depressions (e.g., historic riverbeds) or hardwood floodplain forest at greater distances from the Danube River with a higher proportion of fine sediments (Sali-Bazze, 1981; Walling and He, 1998; Cierjacks et al., 2011).

Previous studies modeled soil OC stocks based on ground data (Cierjacks et al., 2011) or remote sensing (Suchenwirth et al., 2012) and found a correlation of soil OC stock with spatial (lateral, longitudinal, vertical gradients) and forest stand parameters (Cierjacks et al., 2011). A low model prediction ($r^2 = 0.22$) indicates that important environmental variables have not been considered by Cierjacks et al. (2011). Including hydrological and soil parameters into the model led to a clearly higher predictive power (explained variance = 33%), most likely because soil OC stocks are more influenced by water saturation than by productivity (Bernal and Mitsch, 2008). In accordance with Cierjacks et al. (2011), soil OC stock was linked to the lateral gradient with higher stocks at greater distances from the Danube River. Interestingly, the distributions of soil OC stocks were mainly predicted by soil and sedimentation parameters (e.g., OC concentration, sedimentation rate, bulk density) with an inverse relationship of OC accretion rate and resulting long-term soil OC stocks (Pearson's $r = -0.302$, $P = 0.04$).

Higher OC accretion rates are not translated to increased OC stocks. This observation indicates that mineralization and stabilization processes within affect OC stocks. Mineralization of OC and the subsequent release of CO_2 and CH_4 to the atmosphere rely on microbial activity (Lavelle et al., 1993). Microorganisms consume an average of 45% of the leached dissolved OC concentration in sediment (Baker et al., 2000), but this accounts for only 1% of the total OC in soil (Kayranli et al., 2010). Because the soils in our study area are mostly aerated, mineralization and CO_2 production are assumed to be the prevailing processes; whereas methane emissions are less relevant and compensated for by net uptake during the year (Samaritani et al., 2011). However, OC loss from mineralization depends on OC stabilization. In temperate soils, von Lützow et al. (2008) differentiated active, intermediate, and passive OC pools, which are characterized by an increasing degree of stabilization. Organic C stabilization in the passive pool is associated with the occlusion of organic matter in clay microstructures. Up to 75% of soil OC can be classified as a passive pool containing old and stable OC (von Lützow and Kögel-Knabner, 2009) that is protected from decomposition for a period of at least 100 years (von Lützow et al., 2008). Seemingly likely this kind of stabilization process is more important for hardwood sites, characterized by higher clay content along with a lower OC accretion rate compared to softwood forests, which overall leads to an inverse relationship of OC accretion rate and resulting long-term soil OC stocks.

5. Conclusions

This study showed that OC accretion and sedimentation rates are closely linked as each of them responded similarly to roughly the same environmental parameters. We thus conclude that OC accretion rate is influenced mainly by sedimentation rate, whereas sediment quality (i.e., OC concentration) seems less important. In contrast, soil OC stocks of the top meter are negatively associated with sediment quantity but respond positively to sediment quality with a high concentration of OC stabilized in silty and clayey sediments that is related with slow short-term OC accretion. The latter implies that OC input from high sedimentation rates cannot be considered sustainable, presumably because of the short residence times of these OC pools, which are not converted to soil OC stocks in the long term. Moreover, it is likely that — at sites farther away from the Danube River — the gradual shift in OC input from allochthonous OC deposited by sedimentation to autochthonous OC at sites with lower sedimentation rates becomes influential as does the presence of undisturbed biomass production, e.g., fine roots (Rieger et al., 2013) and litterfall. The higher estimated soil age at sites far from the river in DRF supports this assumption. Overall, the rather low explained variance in the presented models indicates that the high complexity of riparian ecosystems precludes a high accuracy in the prediction of their C dynamics, as equally shown by Suchenwirth et al. (2013). However, our data provides evidence that, owing to the low frequency of pronounced flooding events with high erosion rates within the Danube riparian forest, most parts of the study area may function as efficient OC sinks, which highlights the importance of this ecosystem service of riverine forests.

Acknowledgments

This study was funded by the Deutsche Forschungsgemeinschaft (DFG, CI 175/1). We thank Torben Lübbe for helping with the field work, Christian Fraissl and Christian Baumgartner (Donau-Auen National Park administration) for their technical and administrative support, Alfred Paul Blaschke from the TU Vienna for providing a groundwater model, the via donau-Österreichische Wasserstraßen-Gesellschaft mbH for providing the digital elevation model, and Kelaine Radvin for improving our English. Two anonymous reviewers and Richard A. Marston helped to improve a former version of this manuscript.

References

Ad-hoc-Arbeitsgruppe Boden, 2005. Bodenkundliche Kartieranleitung, 5. Verbesserte und erweiterte Auflage. Bundesanstalt für Geowissenschaften und Rohstoffe in Zusammenarbeit mit den Staatlichen Geologischen Diensten, Hannover.

Asselman, N.E.M., Middelkoop, H., 1995. Floodplain sedimentation: quantities, patterns and processes. Earth Surf. Process. Landf. 20, 481–499.

Baker, M.A., Valett, H.M., Dahm, C.N., 2000. Organic carbon supply and metabolism in a shallow groundwater ecosystem. Ecol. 81, 3133–3148.

Barsoum, N., 2002. Relative contributions of sexual and asexual regeneration strategies in *Populus nigra* and *Salix alba* during the first years of establishment on a braided gravel bed river. Evol. Ecol. 15, 255–279.

Bernal, B., Mitsch, W., 2008. A comparison of soil carbon pools and profiles in wetlands in Costa Rica and Ohio. Ecol. Eng. 34, 311–323.

Bisutti, I., Hilke, I., Schumacher, J., Raessler, M., 2007. A novel single-run dual temperature combustion (SRDTC) method for the determination of organic, in-organic and total carbon in soil samples. Talanta 71, 521–528.

BMU, 2003. Hydrologischer Atlas von Deutschland (HAD). Bundesministerium für Umwelt, Naturschutz und Reaktorsicherheit, Bonn/Berlin.

Bridgham, S.D., Megonigal, J.P., Keller, J.K., Bliss, N.B., Trettin, C., 2006. The carbon balance of North American wetlands. Wetlands 26, 889–916.

Bridgham, S.D., Pastor, J., Dewey, B., Weltzin, J.F., Updegraff, K., 2008. Rapid carbon response of peatlands to climate change. Ecol. 89, 3041–3048.

Cabezas, A., Comín, F.A., 2010. Carbon and nitrogen accretion in the topsoil of the middle Ebro River floodplains (NE Spain): implications for their ecological restoration. Ecol. Eng. 36, 640–652.

Cabezas, A., Comín, F.A., Walling, D.E., 2009. Changing patterns of organic carbon and nitrogen accretion on the middle Ebro floodplain (NE Spain). Ecol. Eng. 35, 1547–1558.

Cavalcanti, G.G., Lockaby, B.G., 2005. Effects of sediment deposition on fine root dynamics in riparian forests. Soil Sci. Soc. Am. J. 69, 729–737.

Chave, J., Muller-Landau, H.C., Baker, T.R., Easdale, T.A., ter Steege, H., Webb, C.O., 2006. Regional and phylogenetic variation of wood density across 2456 neotropical tree species. Ecol. Appl. 16, 2356–2367.

Chmura, G.L., Anisfeld, S.C., Cahoon, D.R., Lynch, J.C., 2003. Global carbon sequestration in tidal, saline wetland soils. Glob. Biogeochem. Cycles 17, 1111. http://dx.doi.org/10.1029/2002GB001917.

Cierjacks, A., Kleinschmit, B., Babinsky, M., Kleinschroth, F., Markert, A., Menzel, M., Ziechmann, U., Schiller, T., Graf, M., Lang, F., 2010. Carbon stocks of soil and vegetation on Danubian floodplains. J. Plant Nutr. Soil Sci. 173, 644–653.

Cierjacks, A., Kleinschmit, B., Kowarik, I., Graf, M., Lang, F., 2011. Organic matter distribution in floodplains can be predicted using spatial and vegetation structure data. River Res. Appl. 27, 1048–1057.

Craft, C.B., Casey, W.P., 2000. Sediment and nutrient accumulation in floodplain and depressional freshwater wetlands of Georgia, USA. Wetlands 20, 323–332.

De'ath, G., Fabricius, K.E., 2000. Classification and regression trees: a powerful yet simple technique for ecological data analysis. Ecol. 81, 3178–3192.

Drescher, A., Fraissl, C., 2006. Dealpine und demontane Auen im südöstlichen Mitteleuropa. Struktur und Standortsbedingungen von Auenwäldern des östlichen Mitteleuropa am Beispiel der Donau östlich von Wien. WSG Baden-Württemberg 10, 53–68.

Elith, J., Leathwick, J.R., Hastie, T., 2008. A working guide to boosted regression trees. J. Anim. Ecol. 77, 802–813.

Everson, D.A., Boucher, D.H., 1998. Tree species-richness and topographic complexity along the riparian edge of the Potomac River. For. Ecol. Manag. 109, 305–314.

Fox, J., 2008. Applied Regression Analysis and Generalized Linear Models, 2nd edn. Sage Publ., Los Angeles.

Giese, L.A.B., Aust, W.M., Kolka, R.K., Trettin, C.C., 2003. Biomass and carbon pools of disturbed riparian forests. For. Ecol. Manag. 180, 493–508.

Gorham, E., 1991. Northern peatlands: role in the carbon cycle and probable responses to climatic warming. Ecol. Appl. 1, 182–195.

Graf, M., Lair, G.J., Zehetner, F., Gerzabek, M.H., 2007. Geochemical fractions of copper in soil chronosequences of selected European floodplains. Environ. Pollut. 148, 788–796.

Guenat, C., Bureau, F., Weber, G., Toutain, F., 1999. Initial stages of soil formation in a riparian zone: importance of biological agents and lithogenic inheritance in the development of the soil structure. Eur. J. Soil Biol. 35, 153–161.

He, Q., Walling, D.E., 1998. An investigation of the spatial variability of the grain size composition of floodplain sediments. Hydrol. Process. 12, 1079–1094.

Heimann, D.C., Roell, M.J., 2000. Sediment loads and accumulation in a small riparian wetland system in northern Missouri. Wetlands 20, 219–231.

Hoffmann, T., Glatzel, S., Dikau, R., 2009. A carbon storage perspective on alluvial sediment storage in the Rhine catchment. Geomorphology 108, 127–137.

Hofmann, G., Anders, S., 1996. Waldökosysteme als Quellen und Senken für Kohlenstoff – Fallstudie ostdeutsche Länder. Beitr. für Forstwirtsch. und Landschaftsökologie 30, 9–16.

Howe, A.J., Rodríguez, J.F., Saco, P.M., 2009. Surface evolution and carbon sequestration in disturbed and undisturbed wetland soils of the Hunter estuary, southeast Australia. Estuar. Coast. Shelf Sci. 84, 75–83.

Hupp, C.R., 2000. Hydrology, geomorphology and vegetation of coastal plain rivers in the south-eastern USA. Hydrol. Process. 14, 2991–3010.

Hupp, C.R., Bazemore, D.E., 1993. Temporal and spatial patterns of wetland sedimentation, west Tennessee. J. Hydrol. 141, 179–193.

Hupp, C.R., Bornette, G., 2003. Vegetation as a tool in the interpretation of fluvial geomorphic processes and landforms in humid temperate areas. In: Kondolf, G.M., Piégay, H. (Eds.), Tools in Fluvial Geomorphology. Wiley, Chichester, UK, pp. 269–288.

Hupp, C.R., Demas, C.R., Kroes, D.E., Day, R.H., Doyle, T.W., 2008. Recent sedimentation patterns within the central Atchafalaya basin Louisiana. Wetlands 28, 125–140.

Jackson, M.B., Attwood, P.A., 1996. Roots of willow (*Salix viminalis* L.) show marked tolerance to oxygen shortage in flooded soils and in solution culture. Plant Soil 187, 37–45.

Karrenberg, S., Suter, M., 2003. Phenotypic trade-offs in the sexual reproduction of Salicaceae from flood plains. Am. J. Bot. 90, 749–754.

Kayranli, B., Scholz, M., Mustafa, A., Hedmark, Å., 2010. Carbon storage and fluxes within freshwater wetlands: a critical review. Wetlands 30, 111–124.

Koop, H., 1987. Vegetative reproduction of trees in some European natural forests. Vegetatio 72, 103–110.

Kramer, H., Akça, A. (Eds.), 2002. Leitfaden zur Waldmeßlehre, 4. Aufl. Sauerländer's Verlag, Frankfurt am Main.

Lavelle, P., Blanchart, E., Martin, A., Martin, S., Spain, A., Toutain, F., Barois, I., Schaefer, R., 1993. A hierarchical model for decomposition in terrestrial ecosystems: application to soils of the humid tropics. Biotropica 25, 130–150.

Lehtonen, A., Mäkipää, R., Heikkinen, J., Sievänen, R., Liski, J., 2004. Biomass expansion factors (BEFs) for scots pine, Norway spruce and birch according to stand age for boreal forests. For. Ecol. Manag. 188, 211–224.

McTammany, M.E., Webster, J.R., Benfield, E.F., Neatrour, M.A., 2003. Longitudinal patterns of metabolism in a southern Appalachian river. J. N. Am. Benthol. Soc. 22, 359–370.

Minshall, G.W., Petersen, R.C., Bott, T.L., Cushing, C.E., Cummins, K.W., Vannote, R.L., Sedell, J.R., 1992. Stream ecosystem dynamics of the Salmon River, Idaho: an 8th-order system. J. N. Am. Benthol. Soc. 11, 111–137.

Mitra, S., Wassmann, R., Vlek, P.L.G., 2005. An appraisal of global wetland area and its organic carbon stock. Curr. Sci. 88, 25–35.

Mitsch, W.J., Bernal, B., Nahlik, A.M., Mander, Ü., Zhang, L., Anderson, C.J., Jørgensen, S.E., Brix, H., 2012. Wetlands, carbon, and climate change. Landsc. Ecol. 28, 583–597.

Noe, G.B., Hupp, C.R., 2005. Carbon, nitrogen, and phosphorus accumulation in floodplains of Atlantic coastal plain rivers USA. Ecol. Appl. 15, 1178–1190.

Noe, G.B., Hupp, C.R., 2009. Retention of riverine sediment and nutrient loads by coastal plain floodplains. Ecosystems 12, 728–746.

Olde Venterink, H., Vermaat, J.E., Pronk, M., Wiegman, F., van der Lee, G.E.M., van den Hoorn, M.W., Higler, L.W.G.B., Verhoeven, J.T.A., 2006. Importance of sediment deposition and denitrification for nutrient retention in floodplain wetlands. Appl. Veg. Sci. 9, 163–174.

Olley, J.M., 2002. Organic carbon supply to a large lowland river and implications for aquatic ecosystems. The Structure, Function and Management Implications of Fluvial Sedimentary Systems. Proceedings of an International Symposium Held at Alice Springs, Australia. IAHS Publ. No. 276 (Available at: http://ks360352.kimsufi.com/redbooks/a276/iahs_276_027.pdf, accessed 25 October 2013).

Petts, G.E., Gurnell, A.M., Gerrard, A.J., Hannah, D.M., Hansford, B., Morrissey, I.I., Edwards, P.J., Kollmann, J., Ward, J.V., Tockner, K., Smith, B.P.G., 2000. Longitudinal variations in exposed riverine sediments: a context for the ecology of Fiume Tagliamento, Italy. Aquat. Conserv. 10, 249–266.

Phillips, J.D., 1989. Fluvial sediment storage in wetlands. Water Resour. Bull. 25, 867–873.

Phillips, J., 2003. Alluvial storage and the long-term stability of sediment yields. Basin Res. 15, 153–163.

Pinay, G., Fabre, A., Vervier, P., Gazelle, F., 1992. Control of C, N, P distribution in soils of riparian forests. Landsc. Ecol. 6, 121–132.

Pizzuto, J.E., 1987. Sediment diffusion during overbank flows. Sedimentology 34, 301–317.

R Development Core Team, 2012. R: A Language and Environment for Statistical Computing. R Foundation for Statistical Computing, Vienna, Austria. (Available at: http://www.R-project.org, accessed 11 September 2012).

Reid, L.M., Dunne, T., 2003. Sediment budgets as an organizing framework in fluvial geomorphology. In: Kondolf, G.M., Piégay, H. (Eds.), Tools in Fluvial Geomorphology. Wiley, Chichester, UK, pp. 463–500.

Rieger, I., Lang, F., Kleinschmit, B., Kowarik, I., Cierjacks, A., 2013. Fine root and aboveground carbon stocks in riparian forests: the roles of diking and environmental gradients. Plant Soil 370, 497–509.

Sali-Bazze, M., 1981. Entwicklung und Altersstellung von Böden der Donau- und Marchauen im Gebiet von Orth an der Donau und Marchegg/Baumgarten. (Ph.D. Dissertation) Universität für Bodenkultur in Wien, Austria.

Samaritani, E., Shrestha, J., Fournier, B., Frossard, E., Gillet, F., Guenat, C., Niklaus, P.A., Pasquale, N., Tockner, K., Mitchell, E.A.D., Luster, J., 2011. Heterogeneity of soil carbon pools and fluxes in a channelized and a restored floodplain section (Thur River, Switzerland). Hydrol. Earth Syst. Sci. 15, 1757–1769.

Saunders, M.J., Jones, M.B., Kansiime, F., 2007. Carbon and water cycles in tropical papyrus wetlands. Wetl. Ecol. Manag. 15, 489–498.

Schapire, R., 2002. The boosting approach to machine learning — an overview. In: Denison, D.D., Hansen, M.H., Holmes, C., Mallick, B., Yu, B. (Eds.), MSRI Workshop on Nonlinear Estimation and Classification. Springer-Verlag, New York.

Schweingruber, F.H., 1989. Tree Rings. Basics and Applications of Dendrochronology. Kluwer Acad. Publ, Dordrecht, Netherlands.

Steiger, J., Gurnell, A.M., 2002. Spatial hydrogeomorphological influences on sediment and nutrient deposition in riparian zones: observations from the Garonne River, France. Geomorphology 49, 1–23.

Suchenwirth, L., Förster, M., Cierjacks, A., Lang, F., Kleinschmit, B., 2012. Knowledge-based classification of remote sensing data for the estimation of below- and above-ground organic carbon stocks in riparian forests. Wetl. Ecol. Manag. 20, 151–163.

Suchenwirth, L., Förster, M., Lang, F., Kleinschmit, B., 2013. Estimation and mapping of carbon stocks in riparian forests by using a machine learning approach with multiple geodata. PFG Photogrammetr. Fernerkund. Geoinformation 4, 333–349.

Suratman, M.N., 2008. Carbon sequestration potential of mangroves in southeast Asia. In: Bravo, F., LeMay, V., Jandl, R., von Gadow, K. (Eds.), Managing Forest Ecosystems: The Challenge of Climate Change. Springer, Netherlands, pp. 297–315.

Timoney, K.P., Argus, G., 2006. Willows, water regime, and recent cover change in the Peace–Athabasca Delta. Ecoscience 13, 308–317.

Tockner, K., Schiemer, F., Ward, J.V., 1998. Conservation by restoration: the management concept for a river-floodplain system on the Danube River in Austria. Aquat. Conserv. 8, 71–86.

Turunen, J., Tomppo, E., Tolonen, K., Reinikainen, A., 2002. Estimating carbon accumulation rates of undrained mires in Finland — application to boreal and subarctic regions. The Holocene 12, 69–80.

Vannote, R.L., Minshall, G.W., Cummins, K.W., Sedell, J.R., Cushing, C.E., 1980. The river continuum concept. Can. J. Fish. Aquat. Sci. 37, 130–137.

von Lützow, M., Kögel-Knabner, I., 2009. Temperature sensitivity of soil organic matter decomposition — what do we know? Biol. Fertil. Soils 46, 1–15.

von Lützow, M., Kögel-Knabner, I., Ludwig, B., Matzner, E., Flessa, H., Ekschmitt, K., Guggenberger, G., Marschner, B., Kalbitz, K., 2008. Stabilization mechanism of organic matter in four temperate soils: development and application of a conceptual model. J. Plant Nutr. Soil Sci. 171, 111–124.

Walling, D.E., He, Q., 1997. Investigating spatial patterns of overbank sedimentation on river floodplains. Water Air Soil Pollut. 99, 9–20.

Walling, D.E., He, Q., 1998. The spatial variability of overbank sedimentation on river floodplains. Geomorphology 24, 209–223.

Wiesmeier, M., Spörlein, P., Geuß, U., Hangen, E., Haug, S., Reischl, A., Schilling, B., von Lützow, M., Kögel-Knabner, I., 2012. Soil organic carbon stocks in southeast Germany (Bavaria) as affected by land use, soil type and sampling depth. Glob. Chang. Biol. 18, 2233–2245.

Wimmer, R., Moog, O., 1994. Flußordnungszahlen österreichischer Fließgewässer. Umweltbundesamt (Monographien/Umweltbundesamt, 51), Wien.

Zehetner, F., Lair, G.J., Gerzabek, M.H., 2009. Rapid carbon accretion and organic matter pool stabilization in riverine floodplain soils. Global Biogeochem. Cycles 23, GB4004. http://dx.doi.org/10.1029/2009GB003481.

Zentralanstalt für Meteorologie und Geodynamik, 2002. Klimadaten von Österreich 1971–2000. Available at http://www.zamg.ac.at/fix/klima/oe7100/klima2000/klimadaten_oesterreich_1971_frame1.htm (accessed 25 November 2011).

Zianis, D., Muukkonen, P., Mäkipää, R., Mencuccini, M., 2005. Biomass and stem volume equations for tree species in Europe. Silva Fenn. Monogr. 4, 1–63 (Available at: http://www.metla.fi/silvafennica/full/smf/smf004.pdf, accessed 15 September 2011).

Zuur, A.F., Ieno, E.N., Elphick, C.S., 2010. A protocol for data exploration to avoid common statistical problems. Methods Ecol. Evol. 1, 3–14.