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Can *Solidago gigantea* Impede the Establishment of a Riparian Forest Along a Restored River Section?

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Abstract: Riparian softwood forests support numerous ecological functions and high biodiversity. In the context of the LIFE+ Traisen project, a non-regulated new riverbed for the lower Traisen River ("New Traisen") was created within an artificially lowered floodplain corridor. Using vegetation monitoring from 2014 to 2021, we determined (i) the role of *Solidago gigantea* in the establishment of softwood forests, (ii) the habitat parameters (such as flooding height, fine substrate layer thickness, and vegetation cover) that impact the establishment and growth of woody plants, and (iii) the successional phase at which woody plants become established, as well as the potential creation of new germination habitats. During early succession, the softwood species, as light-tolerant pioneer species, colonized the open sites together with *S. gigantea* and subsequently established a floodplain softwood forest. Unexpectedly, we observed negative forest development only when the *S. gigantea* cover exceeded 90%. Neither the habitat parameters nor *S. gigantea* cover significantly impacted tree occurrence. However, we highlight the need for optimum habitat parameters for softwood forest development in early succession phases, ideally before *S. gigantea* forms dense, monospecific stands. Tailored monitoring strategies are needed to guide the succession of such semi-aquatic habitats toward the development of the desired habitat type.

Keywords: riparian softwood forest; invasive alien plants (IAPs); river restoration; habitat parameters; semi-aquatic habitats; successional phase; monitoring

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Citation: Becker, I.; Paul, M.; Egger, G. Can *Solidago gigantea* Impede the Establishment of a Riparian Forest Along a Restored River Section? *Water* **2024**, *16*, 3489. https://doi.org/10.3390/w16233489

Academic Editor: Rui Manuel Vitor Cortes

Received: 5 November 2024
Revised: 28 November 2024
Accepted: 30 November 2024
Published: 4 December 2024

1. Introduction

River ecosystems are highly efficient seed dispersal pathways because of their downstream connectivity [1,2]. In addition, open sand and gravel bars offer germination sites not only for native but also for alien vegetation [1]. This makes riparian ecosystems particularly prone to invasive alien (non-native) plants (IAPs). Alien plants are considered 'invasive' when they become dominant in their introduced range and change the biotic and abiotic conditions of their new environment [3]. This, in turn, can lead to the restructuring of native plant communities and to substantial changes in biodiversity [4,5], along with the modification of ecosystem functioning [6,7]. IAPs represent some of the major threats to local and global biodiversity [8,9]. In addition, vegetation reduces open river banks and stabilizes river channels, which can be accelerated and enhanced by IAP [10,11].

The invasibility of a riparian system is determined by several aspects, including the successional stage of the river section, the functional traits of the native and the invading species, and propagule pressure, defined as the rate of the spread of alien plant propagules into new areas [12–14]. Hydrochory (dispersal by water) is one of the most important dispersal pathways of IAPs in riparian ecosystems [2]. According to Hood and Naiman (2000), riparian areas are extensively colonized by IAPs, as plant competitive interactions among native species are constrained by hydrological disturbances and stress (such as floods and drought) [15]. However, such colonization is strongly dependent on the successional phase. In early and intermediate successional phases, invasive and native species have similar traits and strategies. Most invasive species are well-adapted to disturbed environments and therefore compete with the community of early and intermediate successional species [16]. In contrast, later successional phases are generally rarely invaded, as the ecological niches are already filled [16], and increased competition results in a reduced success of IAPs [17]. In floodplains, the late phase is commonly characterized by established softwood and hardwood riparian forests [18].

The ecological functioning of riparian forests is, to a large extent, dependent on flood pulses for primary productivity and biodiversity [19]. They provide various ecological functions and support a high plant diversity [15,17,20]. Over the past centuries, however, the natural floodplains along large rivers and their natural flood regimes have substantially been degraded by human activities [21]. At the European, national, and regional levels, floodplain habitats and species are highly threatened, and riparian habitats are largely lost because of the ongoing non-sustainable use of natural resources [22]. In Europe, riparian softwood forests of the early successional ecological phase are therefore a priority habitat type of the Natura 2000 directive. This habitat type (91E0*) includes a wide range of different forest communities in floodplains with high levels of oxygen-rich groundwater. The main riparian forest types are regularly flooded gallery forests of white willow (*Salix alba*) and black poplar (*Populus nigra*) on alluvial deposits of the vegetation community Salicion albae. The forests can be periodically inundated by annual floods of the river and well-drained and aerated during low-water periods [23].

Plants of the genera *Salix* and *Populus* are well-adapted to the variable conditions of floodplains as they produce numerous seeds that are dispersed over long distances via air and water. In addition, the species can spread vegetatively by penetrating new sand banks with their roots. They have also adapted to floods by bending branches and narrow leaves with low flow resistance or fast-growing roots that secure them in the ground and can reach groundwater on coarse gravel bars [24].

Riparian zones are characterized by high degrees of IAP colonization [4,25], either in terms of frequency [26] or absolute [27] and relative [28] species richness and cover. In restored riparian habitats across Europe, *Solidago gigantea* (Asteraceae) is a frequently occurring IAPs (Figure 1). It was introduced to Europe around 1758 and became naturalized in the middle of the 19th century [29]. Today, it is distributed across most of Europe, except for the southernmost and northernmost parts [30]. This species, a rhizomatous perennial herb and member of the North American goldenrod species complex, is highly invasive in Europe [31] and widespread in various ecosystems [31]. It occurs abundantly, forming dense stands in dry grasslands and steppes but also in riparian habitats and alluvial or humid grasslands [30]. This species easily colonizes disturbed habitats and establishes permanent, high-density populations [32,33], which makes it both a „passenger“ and a „driver“ of ecosystem change [34–37]. In its introduced range, it largely transforms the physicochemical and biological soil properties [38] and forms dense monospecific stands, thereby disturbing the structure of the local habitat [39]. Given its high competitive ability [40], rapid growth, and polyploidization [41], it displaces native plant communities by decreasing their species richness and diversity. Introduced (European) *S. gigantea* populations produce a larger number of shoots compared to native (American) populations, which may increase their ability to compete against established species in dense or nutrient-poor stands [42].

Figure 1. Dense *Solidago gigantea* stand at New Traisen (photo: I. Becker).

River restoration measures can facilitate *S. gigantea* invasion due to the creation of open germination sites with low inter-species competition [43]. Especially in the case of restoration measures that facilitate dynamic river development, the newly created large open sites are potential germination sites for native and invasive plants, making these areas prone to IAP colonization [44–46]. In areas where IAPs impede the development of riparian forests, management measures, such as reforestation or IAP management, are therefore advisable. Comparisons of recently restored sites with sites restored 6 or more years ago revealed a decreasing trend of IAP occurrence due to succession processes [47]. Despite the partially massive spread of *S. gigantea* along river systems, with potential impacts on riparian biodiversity, it is not known to influence channel morphology, probably due to its location on slightly higher sites in the floodplain.

One example for an *S. gigantea* invasion is the restored New Traisen River section in Lower Austria. The restoration project was implemented between 2014 and 2017 and is one of the largest river restoration projects in Austria, conducted in the frame of the LIFE+-project 'Habitat in the Estuary Section of the River Traisen' [48]. Since 2004, the area has been part of the Natura 2000 FFH site 16 "Tullnerfelder Donau-Auen" (www.noe.gv.at, accessed on 25 November 2024). The site covers a total area of 19,483 ha [23] and constitutes one of the largest connected riparian ecosystems in Austria [22]. The objectives of this project were the creation of valuable aquatic, semi-aquatic, and semi-terrestrial habitats with near-natural floodplain biocoenoses and an increase in biodiversity as well as the establishment of softwood riparian forest (FFH habitat type 91E0*) according to the Natura 2000 Directive. To this end, a new river corridor was created as a typical river section of a lowland meandering river. This river type typically shows only slight changes in the extension of the aquatic zones due to low sediment input and less pronounced hydro- and morphodynamics. In the terrestrial areas of the restored section, numerous open sites for native but also invasive species germination were formed. The restoration measures were accompanied by comprehensive vegetation monitoring for over 8 years, from 2014 to 2021 [49]. Although reforestation measures were initially planned for the New Traisen corridor, the rapid germination of native tree species on the newly created banks permitted heavier reliance on natural rejuvenation, albeit at different germination densities. Browsing of the young twigs by Sika deer was intense during the first years, and over the years, inter- and intraspecific competition with *S. gigantea* strongly increased. At the New Traisen, *S. gigantea* was already present in the surroundings of the construction area, increasing the invasion risk. Although the main goal of the restoration project was to establish a riparian softwood forest, initially, it was unclear whether this habitat type could be reached via natural vegetation succession alone or whether management actions, such as tree planting, should be considered.

In this context, the overall aim of this study is to understand riparian softwood forest establishment along a restored river section against the background of colonization by the IAP *S. gigantea*. To achieve this aim, we analyzed the vegetation monitoring data for the New Traisen River and addressed the following research questions:

(1) Which habitat parameters promote or inhibit the establishment and growth of woody plants in riparian areas?
(2) What is the role of the IAP *S. gigantea* in the establishment of riparian softwood forests?
(3) At which successional phase do woody plants in riparian areas become established? Does the dynamic nature of the newly created riverbed provide new germination habitats for riparian softwood species?

Based on our findings, we provide recommendations for the development of adequate river restoration measures in floodplains.

2. Material and Methods

Study Site

The study area is located along the Danube River in the downstream section of the Traisen River, Lower Austria, in the municipality of Zwentendorf an der Donau. The average elevation is 170 m above sea level. The nearby municipality of Tulln reaches an annual average temperature of 9.6 °C and an annual average precipitation of 616 mm [50]. The Danube River was regulated in the 19th century to reduce flooding and gain land area for agriculture. In 1973, with the construction of the Danube hydroelectric power plant in Altenwörth, the river mouth of the Traisen River was relocated.

The resulting lack of flood dynamics restricted the development of typical floodplains, although numerous populations of threatened species still occur in the area [51]. In the context of the LIFE+ Traisen project, a non-regulated new riverbed for the lower Traisen River (“New Traisen”) was created within a 9.5-km-long artificially lowered floodplain corridor with a width of up to 300 m. In the lowered corridor, meandering of the river course following its natural dynamics is enabled. The mean discharge (MQ) of New Traisen is 14 m^3/s. The flood flow rates are 100 m^3/s at HQ1, 140 m^3/s for the bankfull discharge, and 800 m^3/s at HQ100, with the highest discharges occurring in summer [52]. Flood discharges higher than HQ1 are split into the New Traisen river channel and the former artificial channel.

The project was implemented in three construction phases between 2014 and 2017 [49] (Figure 2). To minimize IAP introduction to the site, during the restoration process, the topsoil layer potentially contaminated with IAP seeds was removed and deposited in the lower soil layers [43]. Currently, the economic use of the area is dominated by the sylvicultural use of the poplar plantations and by the hunting of the large Sika deer populations [53]. The dominant vegetation types are poplar and ash forests.

In the context of comprehensive vegetation monitoring from 2014 to 2021, the succession development of the riparian habitats in the lowered river corridor was documented and analyzed regarding the potential impacts of the habitat parameters on floodplain forest development. Due to the consecutive construction works in the three construction sections, the site ages vary among the different sections, with differences in phenological aspects (Table 1).

During the vegetation monitoring period, the native softwood riparian forest tree species *Salix alba*, *Populus nigra*, and *Populus canescens* always germinated and established themselves in the first years after construction (Figure 3). Whilst stand density remained at a similar level, partly due to intra-specific competition, tree height increased, reaching a level of approximately 120 cm in the last monitoring years.

Simultaneously, the occurrence of *S. gigantea* was documented in the area (Figure 4). During the first monitoring in 2014, no *S. gigantea* was recorded in the first construction section. However, in the other sections, it germinated from the beginning, in particular in construction Section 3, where its population expanded rapidly. Consequently, we anticipated that *S. gigantea* would have a negative impact on riparian forest development.

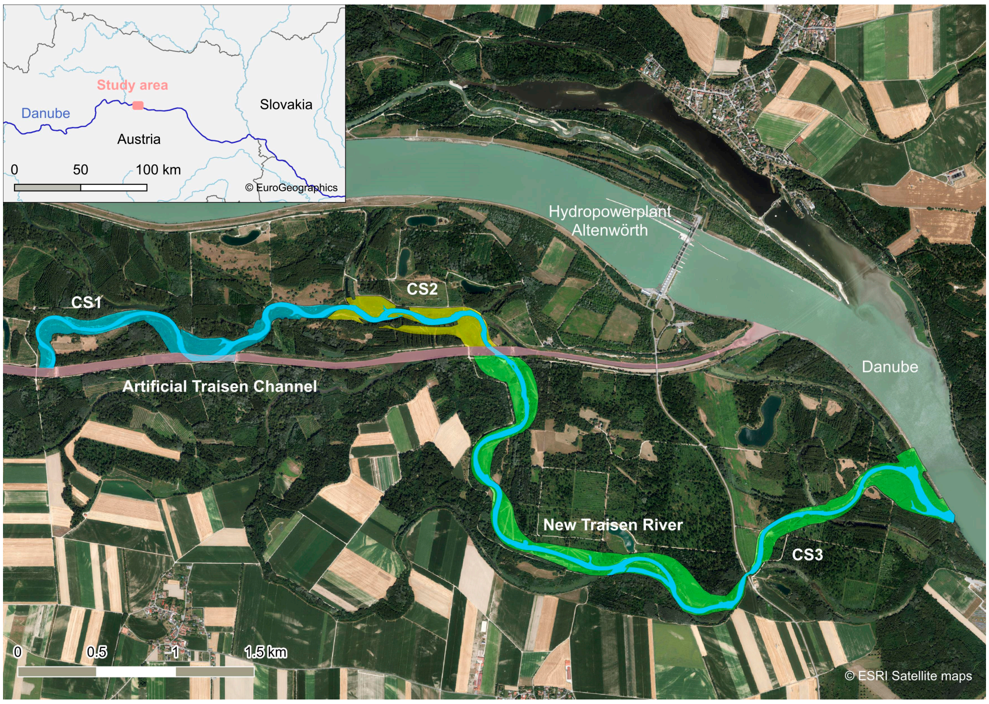
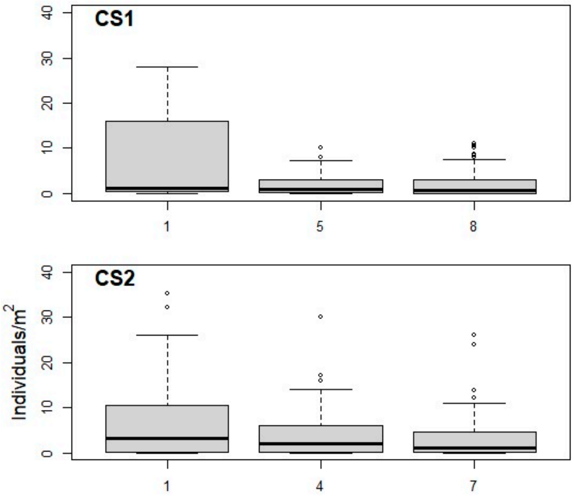


Figure 2. The lower Traisen River before its confluence with the Danube River downstream of the hydropower plant Altenwörth, approximately 50 km upstream of Vienna (Austria). Around the former artificial river channel, a lowered river corridor was created in three construction sections (colored sections), with an initial river course as the 'New Traisen' (basemaps: ESRI maps).

Table 1. Site ages (in years) in the three construction sections (CSs) per vegetation monitoring year; n/a: not applicable.

Year	Date of Vegetation Monitoring	CS1	CS2	CS3
End of construction works		June 2014	April 2015	December 2016
2014	November 2014	1	n/a	n/a
2015	August 2015	2	1	n/a
2017	June and July 2017	4	3	1
2018	June 2018	5	4	2
2021	July 2021	8	7	5



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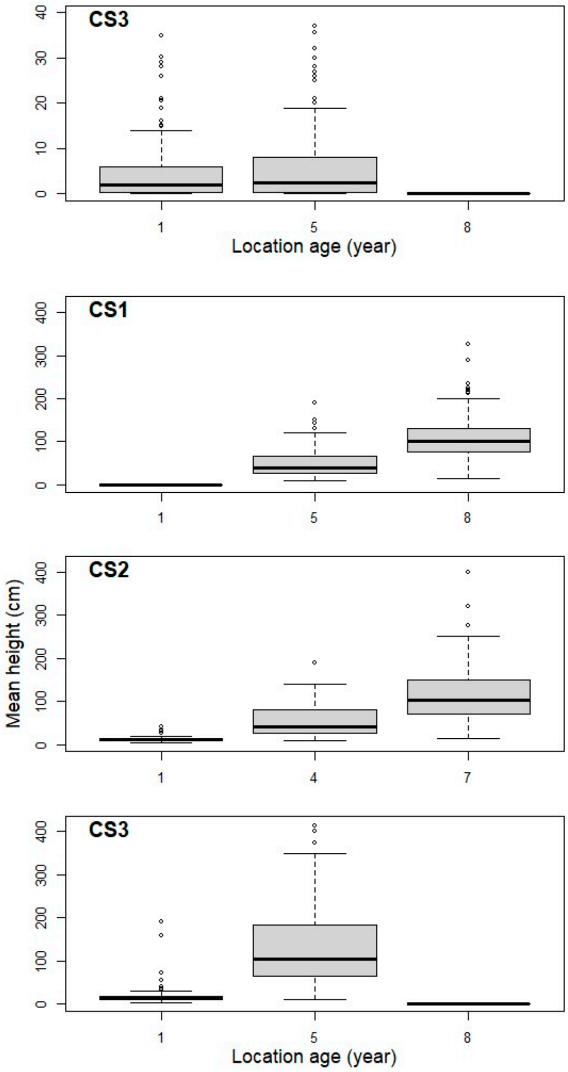
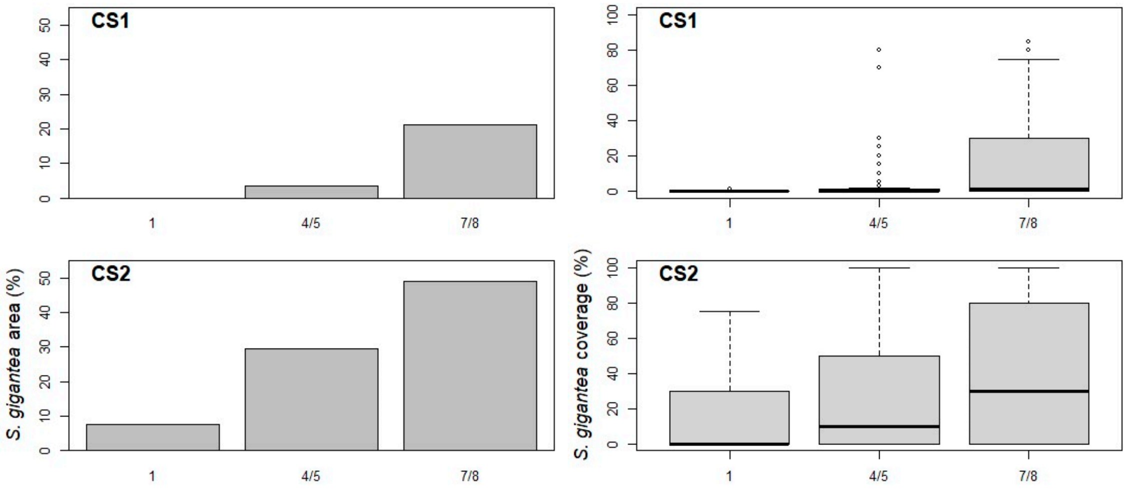


Figure 3. Development of the tree species across the three construction sections (CSs), focusing on tree densities and tree heights for all investigated site ages (dependent on monitoring years). For the location of the CSs, see Figure 2.



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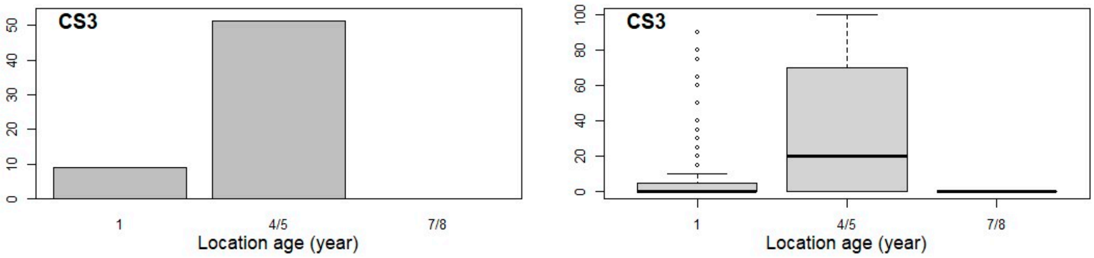


Figure 4. Development of *Solidago gigantea* across all three construction sections (CSs), focusing on habitats with *S. gigantea* and *S. gigantea* coverage within these habitats for three investigated site ages. For the location of the CS, see Figure 2.

3. Data Collection and Analysis

3.1. Habitat Parameters Determining the Establishment of Riparian Softwood Forests

To determine the habitat parameters driving tree growth and establishment, we tested the importance of several parameters. We visually estimated substrate class cover (in % for the following classes: silt, <0.063 mm, sand, 0.063–2.0 mm, gravel, 2.0–63 mm, stones, 63–200 mm, and blocks, >200 mm), measured fine substrate layer thickness (in cm) using a folding meter stick, and determined flooding height (in cm) by measuring the height of alluvial deposited material. Regarding the vegetation parameters, we classified the vegetation type, visually estimated the total vegetation cover (in %, including trees and *S. gigantea*), measured the mean vegetation height (in cm, including trees and *S. gigantea*), visually estimated the *S. gigantea* cover (in %), classified browsing among the three tree species (especially by Sika deer, divided into classes of 1 = low, 2 = medium, 3 = strong, 4 = very strong), and classified the current soil moisture level (completely inundated, wet, alternating wet, moist, alternating moist, fresh, moderately fresh, moderately dry, alternating dry, dry).

Via Spearman's correlation matrix, we determined the most relevant habitat parameters for further analysis, with little or no autocorrelation (correlation matrix see Table S1 in Supplementary Materials). The relevant numeric habitat parameters were applied in a principal component analysis (PCA) using the data of the last monitoring year (2021), with site ages of 5 years (CS3), 7 years (CS2), and 8 years (CS1). The habitat parameters were fine substrate cover, fine substrate layer thickness, flooding height, vegetation cover, *S. gigantea* cover, and browsing, which were combined with the mean height of each individual tree species. The PCA was performed with scaled data to compensate for differences in the scaling of the variables.

In the next step, a generalized linear model (glm) was created using the selected habitat parameter and tree occurrence from 2021. Since the dependent variable was binary (tree occurrence: yes or no), we used a glm function with the binomial family. Different model versions were tested and compared by the odds ratios of the omnibus test and the Akaike information criterion (AIC) values, together with McFadden pseudo-R^2 and their p-values. All calculations were carried out in R (version 4.2.2) using the glm function from the stats package. For the most relevant habitat parameters, scatterplots were created.

3.2. Relevance of *S. gigantea* Occurrence for Softwood Forest Establishment

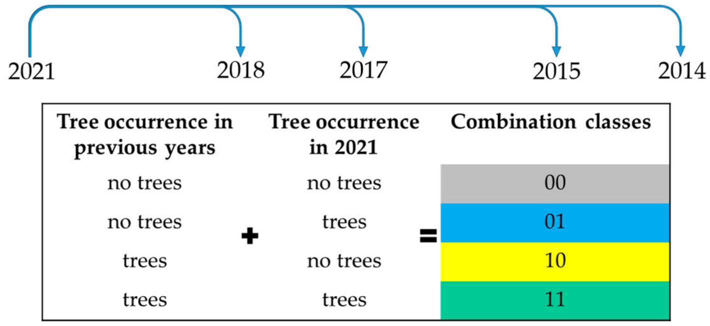
The three construction sections differed in terms of *S. gigantea* cover (Figure 4), and we investigated the impact of *S. gigantea on* tree occurrence in more detail. Specifically, using data from the last 2 monitoring years (2018 and 2021), we compared *S. gigantea* cover with tree species development. Tree species performance was estimated using a combination of change in tree height (cm) and change in tree density (individuals/m^2) (Table 2) and subsequently related to the different *S. gigantea* cover classes.

Table 2. Definition of the tree development classes considering increase or decrease of mean tree height and tree density per vegetation plot. The development classes are as follows: (1) high increase in height and density, (2) low increase in height and/or density, (3) low decrease in height and low increase in density, and (4) decrease in height and density.

		Change in Mean Tree Height (cm)				
		<−5	**−5–0**	**>0–5**	**>5–50**	**>50**
Change in tree density (individuals/m^2)	**<−0.2**	4	4	2	2	1
	−0.2–0	4	4	2	1	1
	0	4	4	2	1	1
	>0–0.2	3	3	2	1	1
	>0.2	3	3	1	1	1

3.3. Establishment of Woody Plants Along Succession

We identified the time of emergence of the trees in the areas where they were growing at the time of the last monitoring campaign in 2021 to understand at which successional phase they had germinated. To this end, we created stream graphs for each construction section showing tree occurrence in 2021 compared to that of each monitored year before (Figure 5). In each year, tree occurrence or absence was intersected with the occurrence in 2021, leading to the following four combination options: (i) no tree occurrence in 2021 or the previous monitoring year (00), (ii) tree occurrence in 2021 but not in the previous monitoring year (01), (iii) no tree occurrence in 2021 but present in the previous monitoring year (10), and (iv) tree occurrence both in 2021 and in the previous monitoring year (11).



Tree occurrence in previous years		**Tree occurrence in 2021**		**Combination classes**
no trees		no trees		00
no trees	+	trees	=	01
trees		no trees		10
trees		trees		11

Figure 5. Workflow of the analysis on the establishment of woody plants along succession, comparing tree occurrence or absence in 2021 with the previous years.

To estimate the river dynamics of the newly created river section, we compared the spatial extent and position of the river course between the end of the construction works and 2021 for each section. Additionally, we compared the vegetation types of the last two monitoring records (2018 to 2021) to investigate whether the river dynamics create new aquatic habitats and new germination sites for the tree target species, and whether *S. gigantea* stands are destroyed by these dynamics.

4. Results

4.1. Habitat Parameters Determining the Establishment of Riparian Softwood Forests

With the relevant six numeric habitat parameters identified via Spearman´s correlation matrix, we performed a PCA using the data of 2021 (Figure 6). The first two principal components explained 42% of the total variance (PC1: 22.9% and PC2: 19.1%), which was considered a sufficient basis for the visualization of the data structure (Table 3). The loadings for PC1 revealed that the vegetation parameters described by the mean tree species heights (loadings: mean *P. canescens* height: 0.529, mean *P. nigra* height: 0.412, and mean *S. alba* height: 0.313), together with the vegetation cover (loading: 0.398) and the *S. gigantea* cover (loading: 0.432), had similar shares on this PC. In contrast, PC2 was strongly negatively influenced by fine substrate cover (loading: −0.572) and fine substrate layer thickness (loading: −0.564).

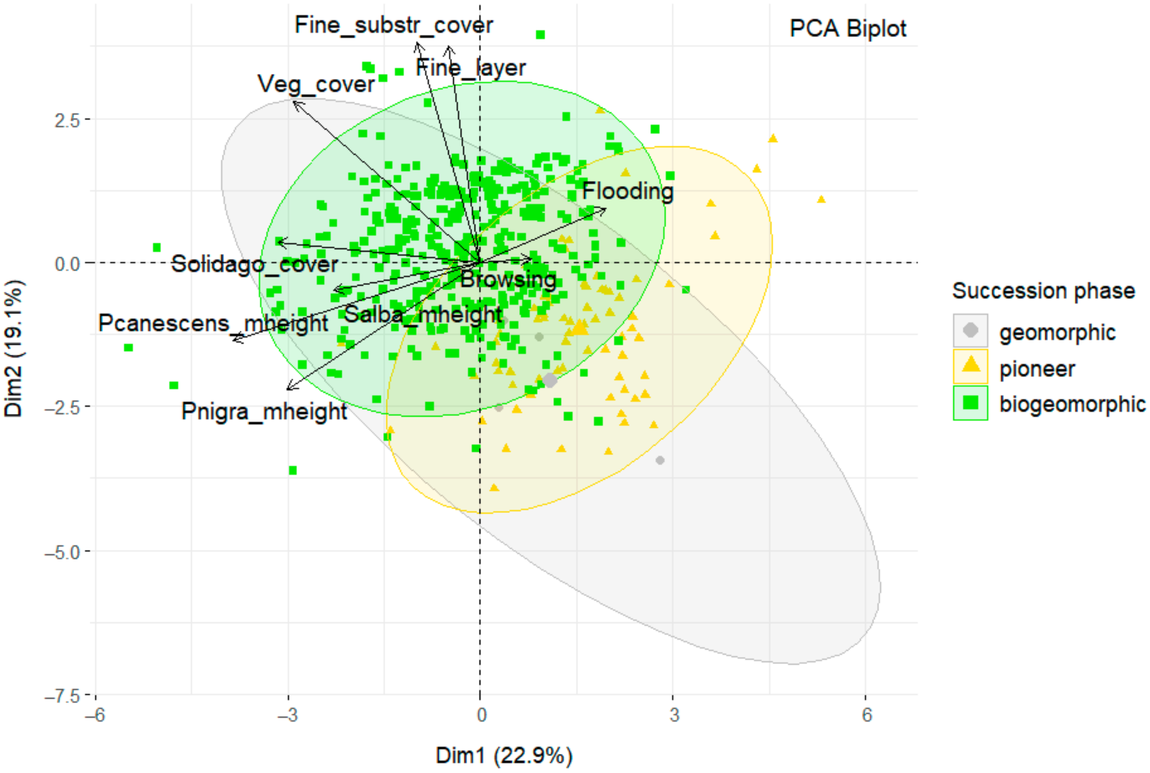


Figure 6. PCA using the species height and habitat parameters of the last monitoring year, 2021. For the definition of the succession phases, see Corenblit et al. [54,55].

Table 3. Loadings of the principal component analysis (PCA) variables showing the direction and strength of the influence on the first two principal components. On the bottom of the table are the standard deviations as measures of the dispersion of the data along each PC and the proportion of variance, indicating the percentage of the total variance in the data, as explained by each PC.

PCA Parameter	Parameter	PC1	PC2
Salba_mheight	Mean height of *Salix alba* (cm)	0.31286	0.07159
Pnigra_mheight	Mean height of *Populus nigra* (cm)	0.41219	0.33115
Pcanescens_mheight	Mean height of *Populus canescens* (cm)	0.52915	0.20254
Fine_layer	Fine substrate layer thickness (cm)	0.06945	−0.56422
Fine_substr_cover	Fine substrate cover (%)	0.13455	−0.57231
Flooding	Flooding height (cm)	−0.26840	−0.14140
Veg_cover	Vegetation cover (%)	0.39803	−0.41900
Solidago_cover	*Solidago gigantea* cover (%)	0.43223	−0.05141
Browsing	Browsing intensity (in classes)	0.10954	−0.01063
Standard deviation		1.436	1.3128
Proportion of Variance		0.229	0.1915
Cumulative Proportion		0.229	0.4204

The mean heights of the three tested tree species behaved in a similar pattern, as indicated by similar directions and sizes of the vector arrows in the PCA. Both browsing and flood height had negative impacts on mean tree height (vector arrows point in the opposite direction of mean tree height), whereas fine substrate cover and layer thickness, as well as vegetation cover, seemed to have no effect on mean tree height (the vector arrows perpendicular to mean tree height). The vector arrow of the *S. gigantea* cover was similar to that of mean tree height (the small angle between arrows).

To determine the importance of the habitat parameters in more detail, we adapted a glm with logistic regression, again using the data from 2021. However, instead of mean tree height, we used tree occurrence as the dependent variable to obtain more universal results and to mask the effect of browsing on tree height.

The best model performance was reached using an additive model with the habitat parameters *S. gigantea* cover, fine substrate cover, and flood height as independent variables (Table 4). All three parameters had a *p*-value of <0.05 and can therefore be considered as highly significant. To select the best model, we used the AIC, obtaining the following model: tree occurrence in 2021~*Solidago gigantea* cover + flood height + fine substrate cover.

Table 4. Model summary with the importance values of the tested variables.

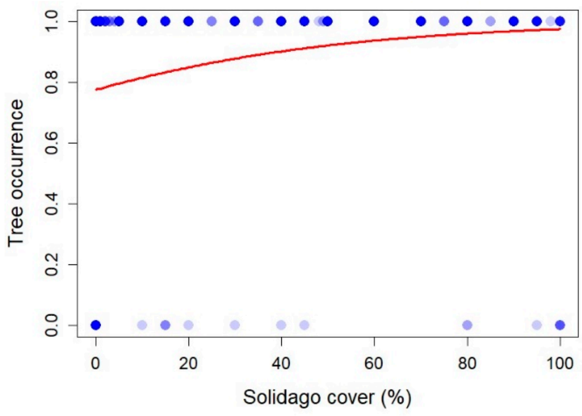
	Estimate	Std. Error	z Value (Wald Test)	*p*-Value (Pr(>|z|))
(Intercept)	−0.155940	0.304933	−0.511	0.609
Solidago gigantea cover	0.026147	0.005853	4.467	7.93×10^{-6}
Fine substrate cover	0.01868	0.003792	4.929	8.28×10^{-7}
Flood height	−0.070832	0.012590	−5.626	1.84×10^{-8}

The omnibus test revealed a good model performance, with a *p*-value of 0 and a McFadden Pseudo-R^2 of 0.2321, which is an acceptable result (McFadden 1974, Table 5). However, the odds ratios (ORs) of the three variables were close to 1, indicating small effect sizes. Hereby, the *S. gigantea* cover (1.026) and the fine substrate cover (1.019) showed slightly positive effects on tree occurrence, whereas flood height had a slightly negative impact (0.932).

Table 5. Results of the omnibus test revealing the odds ratios and lower and upper confidence intervals of each variable.

	Odds Ratios	2.5%	97.5%
(Intercept)	0.856	0.469	1.562
Solidago gigantea cover	1.026	1.016	1.039
Fine substrate cover	1.019	1.011	1.027
Flood height	0.932	0.907	0.954

The scatterplots in Figure 7 show the probability of tree occurrence, depending on the values of the three relevant habitat parameters. Tree occurrence slightly increased with increasing *S. gigantea* cover and fine substrate cover, whereas it considerably decreased with increasing flood height.



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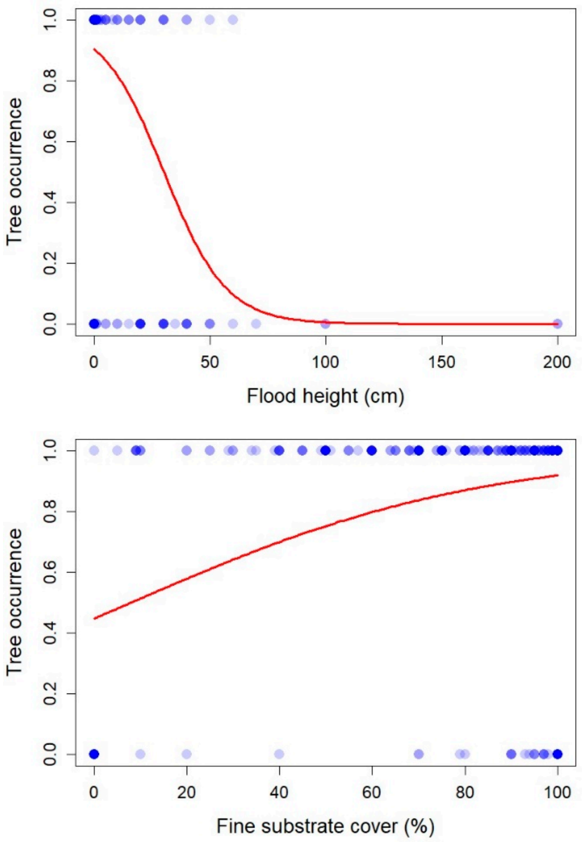

Figure 7. The most relevant habitat parameters influencing tree occurrence in the study area are *Solidago gigantea* cover, flood height, and fine substrate cover. The red line represents the estimated probability of the tree occurrence as a function of the individual habitat parameter. The blue dots represent individual plots, and the darker color represents a cluster of recording points.

Overall, all analyses revealed high significance levels but only small effect sizes and patterns regarding the impacts of the habitat parameters on tree occurrence and tree height. This indicates that no parameter played a dominant role in shaping tree establishment.

4.2. Relevance of *S. gigantea* Occurrence for Softwood Forest Establishment

Figure 8 shows the impact of the *S. gigantea* cover on tree development from 2018 to 2021. Although the sites in CS3 were 2 or 3 years younger than those in the other two construction sections, the *S. gigantea* cover in most plots in CS3 was high, mostly exceeding 80% (see also Figure 4). Irrespective of the construction section, we observed an increase in tree height and stand density between 2018 and 2021, with class 1 being the dominant development class. Decreasing tree height and density, represented by development classes 3 and 4, were highest in CS1 for the sites with no *S. gigantea* cover and, albeit at lower extents, for the sites with an *S. gigantea* cover of 10–19% and 40–49%, respectively. In CS3, negative tree development was observed for the sites with an *S. gigantea* cover of more than 80%, and particularly more than 90%. The plots in CS2 occupied an intermediate position between the other two construction sections, where the sites with negative tree development showed not distinct pattern in *S. gigantea* cover.

Our observations support the assumption that a high *S. gigantea* cover can have a negative influence on the development of woody plants. However, in our study, this only applied to sites with an extremely high *S. gigantea* density. Furthermore, other parameters and processes can also have a negative influence on tree growth, as can be seen from the negative tree development despite low *S. gigantea cover*.

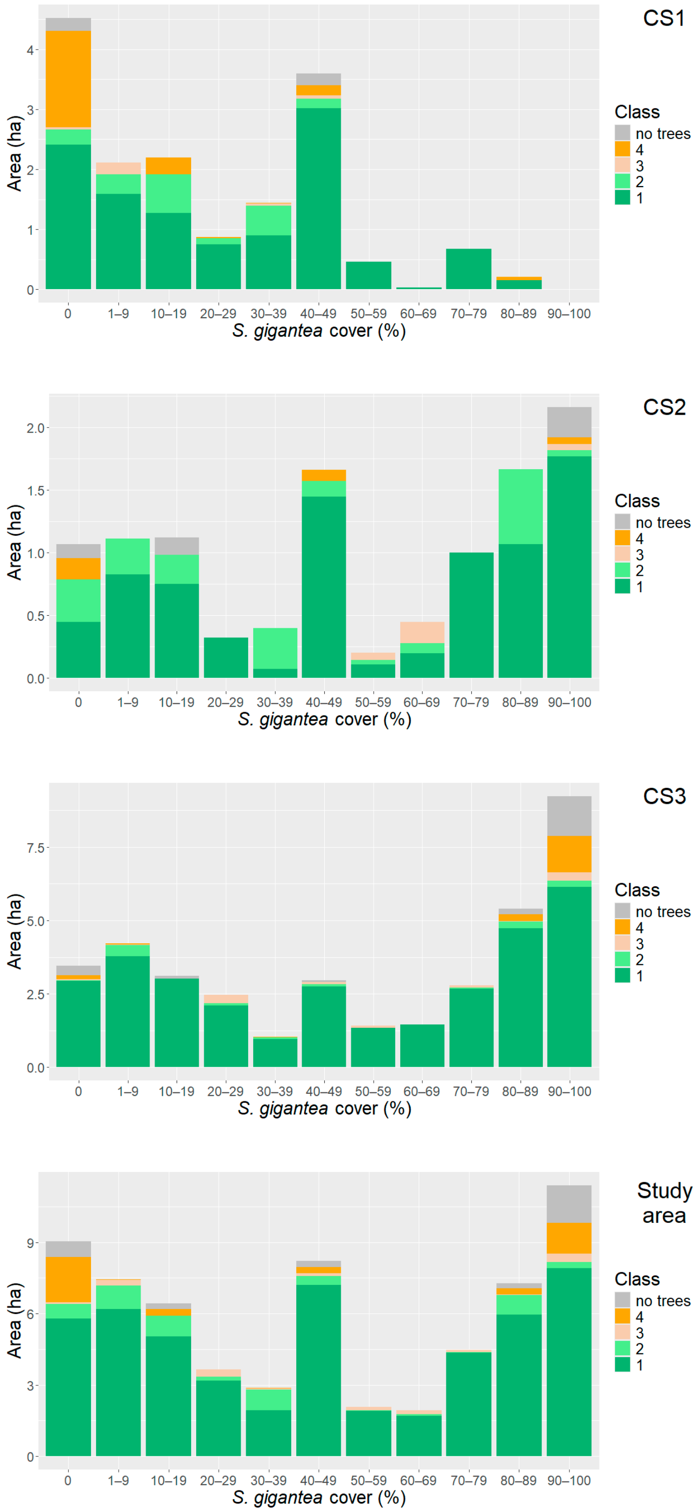

Figure 8. Tree development in 2021 compared to 2018 in relation to *Solidago gigantea* cover in 10% increments, for the individual construction sections (CSs) and the total study area. The tree development classes were determined by comparison of tree height (cm) and density (individuals/m^2) in 2021 and 2018, leading to a negative development when height and/or density decreased or to a positive development with increasing height and/or density. The tree development classes are as follows: (1) high increase in height and density (dark green), (2) low increase in height and/or density (light green), (3) low decrease in height and low increase in density (light orange), and (4) decrease in height and density (orange).

4.3. Establishment of Woody Plants Along Succession

Since neither the habitat parameters nor the *S. gigantea* cover had a significant impact on tree occurrence on the New Traisen river corridor, we investigated tree species occurrence over the monitoring years. For this, we compared tree occurrence in 2021 with that of each previous monitoring year (Figure 9).

Based on the results, the area with no tree occurrence across all years, compared to that in 2021 or with trees that vanished over the monitoring period, was low. The area without tree occurrence in the previous monitoring years but with tree occurrence in 2021 (combination 01 in Figure 9) occupied an intermediate position but was low and decreased toward the end of the monitoring period. Most of the study area was covered by trees from the beginning until the end of the monitoring period (combination 11 in Figure 9).

In all construction sections, the river course migration between the end of the construction works and the last monitored record in 2021 was low. The edge of the river course differed in the cut banks, were it tended to erode by several decimeters (mean: 44 cm in 5 to 8 years; in some meander curves, sedimentation with 1.2 m on average was observed). In the point bars, not only sedimentation was observed (mean: 1.2 m in 5 to 8 years), but also erosion or water level rise occurred from the initially created river course, with 4.3 m on average. Most often, reed canary grass reeds with willows or poplars (396 m^2, corresponding to 26% of the eroded vegetation types), willow–poplar pioneer shrubs (239 m^2, corresponding to 16%), and forbs with willow or poplars (175 m^2, corresponding to 11%) were eroded between 2018 and the last monitoring in 2021 (Table 6). During this period, approximately 88 m^2 of *S. gigantea* stands were destroyed by the river dynamics.

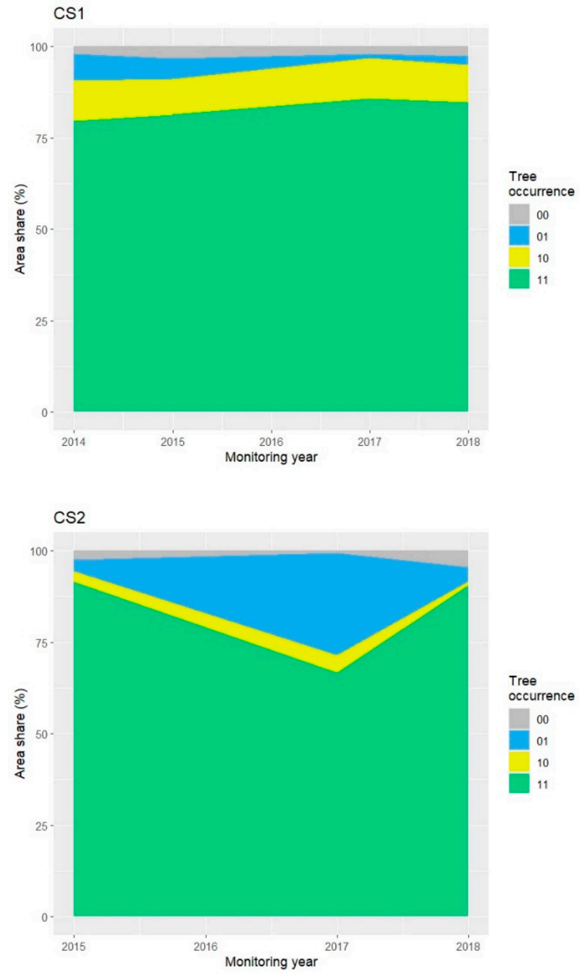

Figure 9. *Cont.*

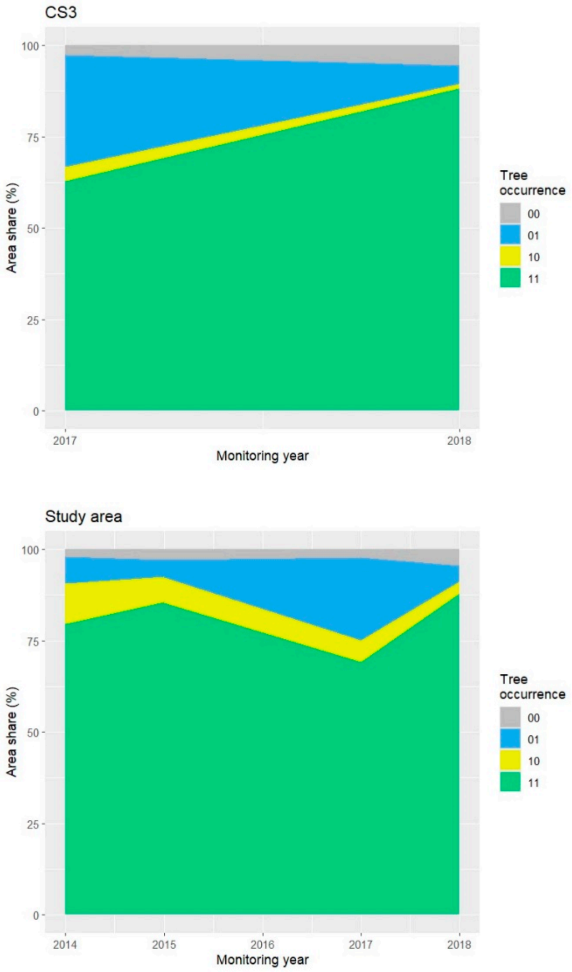

Figure 9. Tree occurrence in 2021 compared to that in the previous monitoring years (with occurrence = 1, no occurrence = 0, resulting in four possible combinations) for the whole study area and for the three individual construction sections (CSs) separately.

Table 6. Destroyed vegetation types between 2018 and 2021 by river dynamics (especially side erosion). Area shares showing the percentages of eroded areas between 2018 and 2021.

Vegetation Type	Pioneer Native Herb Stand	Pioneer Shrub Stand	Native Forb Stand	Willow/Poplar Shrub Stand	*S. gigantea* Stand	Sum
Area (m^2)	55.1	269.9	1061.5	47.3	87.4	1521.3
Area share (%)	3.6	17.7	69.8	3.1	5.7	100.0

5. Discussion

5.1. Factors That Promote or Inhibit the Establishment of Riparian Softwood Forests

Numerous factors can influence plant reproduction [56], as well as seedling emergence and growth [57,58], and the interaction of these factors is difficult to quantify. In our study, although the initial habitat parameters largely controlled tree development, tree occurrence in the medium term did not differ among the sites. Therefore, to determine the factors that drive tree establishment, we used the data from the year 2021. Based on the PCA results, flood height negatively impacted tree height. Likewise, several authors reported that floods are major disturbances that greatly influence riparian plant communities [59–61]. Forest dynamics in floodplains are largely influenced by floods [62], although these dynamics are, to a certain extent, reversible and driven by allogenic processes [63].

Our finding is also in line with the assumption that softwood tree seedlings mainly respond to flooding and light [64]. As floodplain environments are characterized by frequent flooding and shade, these two stressors limit the success of shade-intolerant species. As tolerance to flood increases with tree age [65], flooding is an important aspect limiting the growth and development of tree seedlings in riparian forests [66,67]. Plants generally respond to flooding by injury, inhibited seed germination, changes in vegetative and reproductive growth, altered anatomy, and the promotion of early senescence and mortality [65]. To acquire a certain tolerance to flooding, the seedlings need to grow taller than the mean flood level, as partially submerged seedlings have a lower mortality than totally submerged ones [68]. For all selected habitat parameters used in the glm to relate the site conditions with tree occurrence, we observed significant correlations, albeit with only small effect sizes, indicating that none of the parameters played a main role in shaping tree establishment.

Flooding may also induce fine substrate mobilization. Although *Salix* sp. is somewhat tolerant to substrate erosion because of its deep root system [62], other tree species may have been impacted by flooding-induced substrate erosion. In our study, according to the glm results, higher levels of fine substrate cover facilitated tree growth and development. We assume that this effect is a result of the nutrients attached to the fine sediment particles and the improved water-holding capacity, which is especially important during early growth [54,69,70].

The development of a softwood forest has various impacts on the aquatic habitats. It is connected, e.g., in terms of nutrient input and cycling, physical parameters such as runoff timing and quantity, erosion and sedimentation, or energy inputs such as light, heat, and tree products [71]. This is particularly the case for young forests [72]. Although the impacts of early successional forests on aquatic-terrestrial linkages are far from being understood, previous studies reported a protective role of riparian buffers in aquatic systems [73–75]. This points to the importance of intact floodplain forests in river functioning.

Floodplain forests are generally highly susceptible to invasion by IAP [76], especially considering their characteristic hydro- and morphodynamics in combination with human activities [76–78]. Surprisingly, in our study, the results of the glm indicate that a higher *S. gigantea* cover resulted in a slightly higher probability of tree occurrence. As *S. gigantea* cover differed across the sites, we investigated the impacts of this species on riparian forest development in more detail.

5.2. The Role of the IAP S. gigantea in the Establishment of Riparian Softwood Forests

Species of the genus *Solidago*, native to North America and Canada, are common IAPs in Europe, where they negatively impact plant species richness [39,79] and decrease functional richness. As they are generally taller than the coexisting species, they have an advantage of increased resource capture and use, particularly sunlight [80,81], which likely impedes the establishment of floodplain forests. In the first monitoring in 2014, in Section 1, no *S. gigantea* was recorded, whereas in the other sections, it germinated immediately. Especially in Section 3, the population expanded quickly, which translates into a high invasion degree. The invasion degree, which refers to the status of invasion, indicates the extent to which a community has already been invaded [82]. It is generally quantified as the total number of invasive species or their proportion of the total species richness in the invaded ecosystem [83]. The phenotypic traits of invasive species can change according to the invasion stage [84–86]. In a study on *S. canadensis*, which is extensively distributed across the Yangtze River Basin in China [87], Ren et al. (2022) observed a significant shift in the response of phenotypic traits at an invasion degree from 53% to 68% [82]. This supports the concept of the “threshold of potential concern” (TPC), namely that beyond this stage, the species can reduce some barriers that impede their invasion [88]. We therefore expected that an *S. gigantea* cover above 60% would considerably impact tree occurrence. However, the common assumption that the occurrence of *Solidago* sp. impedes the establishment of young forests does not always hold true. Contrary to our expectations, in terms of tree development from 2018 to 2021 (with the sites being 5 to 8 years old), we observed negative development only when the *S. gigantea* cover exceeded 80% and, especially, 90%. In another study on the same sections of the Traisen River, *S. gigantea* was among the 15 most dominant species and had formed dense populations prior to the beginning of the construction work; after construction, the species still occurred on some sites [43]. However, the cover percentage of IAPs in areas planted with target species, such as *Populus nigra*, *P. alba*, and *Salix alba*, was significantly lower than that in areas that were not planted [43]. Moreover, the occurrence of IAPs did not limit the reestablishment of native pioneer species.

In a study by Ren et al. (2022), *S. canadensis* showed a decrease in competitiveness with an increased invasion level [82]. This is in line with the findings of Pysek et al. (2020), who suggest that only in the early stages of invasion, the high competitiveness of invasive species is a potent driver [89]. In these stages, high competitiveness may compensate for a lower abundance in the community [82]. In line with this, in a study by Adomako et al. (2019), *S. canadensis* growth was impeded at a higher density or diversity of the co-occurring plant species [90], and Wang et al. (2019) observed that the allelopathic effects of invasive *S. canadensis* weakened with a greater degree of invasion [91]. In our study area, only few sites showed no tree occurrence, and a high *S. gigantea* cover was generally accompanied by the successful establishment of riparian softwood forests. A decline in softwood forests was mostly observed for sites without *S. gigantea* growth, indicating that *S. canadensis* did not have a direct negative impact on tree establishment and growth.

Van Oorschot et al. (2017), in their study on IAPs in river systems, observed that *F. japonica* could facilitate the establishment of Salicaceae species by reducing morphodynamic pressure inside and around vegetation patches [92]. However, such facilitation was only given when *F. japonica* acted as an eco-engineer that actively modified the environment, resulting in positive feedback on the establishment of native species. This is in line with the conceptual model of Corenblit et al. (2014), stating that non-dominant IAP species can act as eco-engineers by, e.g., trapping sediment and changing hydrodynamic and morphodynamic processes, thereby influencing biogeomorphic succession [55]. Although the facilitation of native species by invaders has rarely been reported, it might frequently occur, at least in some ecosystems. For example, in forest ecosystems, non-native initial colonizers that act as pioneers can facilitate the growth of a variety of other species [93]. In our case, *S. gigantea*, as a co-engineering species, may have facilitated the growth and development of softwood tree species, at least when the native species were not completely outcompeted and when the *S. gigantea* cover was below a certain threshold (80%).

Based on our findings, the *S. gigantea* stands were not threatened by river dynamics. In the last monitoring period between 2018 and 2021, only 88 m^2 (5.8%) of the *S. gigantea* stands were destroyed. Studies directly linking *Solidago* spp. to riverbank stabilization are limited, and the available research primarily focuses on the ecological effects of *Solidago* spp. invasions, which can indirectly influence riverbank stability. For example, Chinese authors examined the invasion of coastal shelterbelt forests by *S. canadensis* and reported that this species led to decreased understory plant richness and community stability, along with alterations in soil properties. These changes can destabilize the understory plant community, potentially impacting riverbank ecosystems [94].

5.3. Establishment of Softwood Tree Species in the Course of the Successional Process

Each successional phase is characterized by specific ecosystem properties. In the geomorphic phase, which is highly dynamic and productive, there is an enormous diversity of physical configurations, reflecting the regional and local geological and geomorphic settings. In the subsequent pioneer phase, in which vegetation establishment starts, there are no feedback mechanisms between vegetation and river hydromorphology, whereas in the following biogeomorphic phase, organisms and geomorphic components (such as surface matter, energy fluxes, landforms, and soils) strongly interact [54]. However, biogeomorphic ecosystems maintain their integrity under stressful conditions within specific domains of stability. Once the system changes into the ecological phase, it is characterized by the stabilization of its established geomorphic properties and stable biotic interactions, impeding feedback mechanisms. Depending on the interplay of species features and site conditions, we expected that individual IAPs can invade river ecosystems at different successional phases, thereby altering the successional patterns. For example, in our previous studies, we observed that *Fallopia japonica* invades open sand and gravel banks along the Schwechat River in Austria and the Allier River in France [92] during the early successional phase. This species is adapted to highly disturbed site conditions and can colonize a wide range of habitats [92]. In floodplains, it can become dominant, as it occupies similar recruitment sites as native species [95]. Further, *Salix fragilis* colonizes gravel banks along Patagonian rivers [96], including during the pioneer phase. The Danube River section in Machland, Austria, is characterized by a decrease in pioneer vegetation as a result of terrestrialization and habitat fragmentation, with an almost total replacement of the native vegetation by *Impatiens glandulifera* and *S. gigantea* [97].

Softwood forest trees are mainly pioneer species with specific traits and strategies. As their seeds require open sites for germination, the establishment of softwood riparian forests needs to occur during the early successional phase. However, because early succession is not limited by competition for resources, various species can coexist as long as they can cope with disturbance [28]. This highlights that both native and invasive non-native species can colonize floodplains during the early successional phase, with high inter-species competition in the following phases. In our study, softwood tree species germinated on the newly created open sites and formed a floodplain forest via subsequent vegetative or generative spread, with low germination rates. We observed low lateral erosion and, simultaneously, only low sedimentation on the point bars, which would have created new germination habitats for the target species. Although the development of softwood plant species was not prevented by *S. gigantea*, germination was impeded because of the lack of new germination habitats. We therefore assume that the future regeneration of the softwood riparian forest stands is severely hampered.

We observed almost no germination of hardwood tree species (such as *Ulmus laevis*, *Fraxinus* spp., *Quercus robur*, or *Carpinus betulus*), which should have germinated during the 8 years of monitoring [24]. Although we would have expected to find the occasional hardwood tree seedling, these species tend to germinate in the later successional phases under a denser canopy and are less flood-tolerant [98]. In line with these findings, in a study on various hardwood tree species (including *C. betulus*, *U. minor*, and *Q. robur*) in a floodplain in North Italy, seedling establishment was mainly determined by factors such as seed source availability, soil humidity, and light availability [76]. Although the main dispersal mode of *C. betulus* is wind dispersion, the authors observed that the presence of mother trees in the canopy had a significant impact on seedling occurrence. This might also be the case for *Fraxinus* spp., which has a medium dispersal distance of only 52 m [99]. In contrast, seeds of the zoochorous species *Q. robur* can be dispersed over long distances [100] but are more light-demanding [101]. Although our findings cannot be exhaustively explained by these aspects, we suspect that in our sites, which were characterized by dense *S. gigantea* growth, hardwood species could not germinate because of the unfavorable light and soil moisture conditions. Although mother trees were present in the surrounding areas, facilitating seed flight, the few patches not colonized by *S. gigantea* were probably too wet for the germination of hardwood tree species. This is in line with the findings of Terwei et al. (2013), who reported that *Q. robur* cannot establish under a closed canopy [76]. Such suppression by other species during later successional phases was also found for other forest types, such as oak forests in northern Italy [101]. A recent review of the impacts of IAPs on forest regeneration provides evidence that *S. gigantea*, along with several other IAPs, negatively influences the growth and development of *Q. robur* and *C. betulus* [102]. In contrast, the softwood species, as light-tolerant pioneer species, colonized the sites in the initial phases, together with *S. gigantea*, and subsequently established a floodplain softwood forest. This highlights the need for optimum habitat parameters in the initial phase, ideally before *S. gigantea* forms dense, monospecific stands, for the development of the habitat type 91E0*.

Deer browsing, as an important biotic filter in riparian forests, can substantially decrease tree seedling growth [103]. Whilst these authors, using a forest gap model, concluded that deer browsing has distinct impacts on riparian forest succession patterns but not on the final forest composition, others [104,105] found that browsing can alter the long-term composition of European forests. Similarly, in a study on sites along the Rhine River, southwestern Germany, browsing considerably impeded the rejuvenation of hardwood forests, albeit in a species-dependent manner [106]. Although browsing can increase the rate of forest succession [103], in our study, browsing by Sika deer only slightly impacted the long-term development of the softwood forest.

5.4. Recommendations for the Sustainable Management of Restored River Sections

For sustainable river restoration, measures that aim to increase the area and ecological quality of aquatic to terrestrial habitats, the size of the restoration measure area and the long-term conservation of the area through the acquisition of land by nature conservation organizations or authorities are crucial [107].

Evidently, the Natura 2000 habitat type 91E0* could be successfully re-established in the study area, despite the occurrence of *S. gigantea* and browsing. At the time of this study, the sites were in the medium successional phase, and we assume that in the long term, *S. gigantea* will be shaded out by the dense softwood forests. Based on our monitoring results, we predict that softwood riparian forests with *Populus* spp. (on higher and drier sites) and *Salix* spp. (on lower and wetter sites) will dominate the sites in the next two decades, with a gradual increase in hardwood species along natural succession. As pioneer sites will only remain on small areas, to facilitate forest rejuvenation, it is important to restore the original river dynamics to create new potential germination sites for the pioneer species. This is also crucial against the background of the dense *S. gigantea* cover on potential germination sites, highlighting the importance of allowing dynamic processes with hydro- and morphodynamics to preserve the habitat conditions for adapted species of conservation value. Morphodynamic processes, such as bank erosion, where the riverbank is washed out and trees are uprooted, can create new habitats for target species [108]. This requires that, during any restoration measures, the riverbanks not be stabilized [109].

To avoid the massive impacts of IAPs on softwood establishment, we emphasize the importance of suitable conditions for target species germination immediately after the completion of a restoration measure to create open sites. In particular, the time of seed distribution should be taken into account, depending on the target species. In the case of softwood tree species, this is generally in May. This ensures that the target species have a competitive advantage after germination as generally, *S. gigantea* seeds are not distributed before August. Further, when introducing topsoil material, it is crucial to ensure that it is not contaminated with IAP seeds.

Our study is limited by the nature of this restoration project, as we could only analyze construction sections that were finalized at different times. Consequently, our study areas differed in site age, with different monitoring periods and phenological aspects (Table 1). Therefore, the beginning of vegetation development was not only determined by the completion of the construction measures but also by the time of subsequent flooding and sedimentation of a fine substrate layer in the respective season.

Although in this study we focused on the occurrence of *S. gigantea* in a restored floodplain, along with the development of a specific habitat type, our findings can be transferred to other areas with similar conditions and IAP occurrences. Since *S. gigantea* is widespread in Europe [30], competition with native species occurs in several areas and different ecosystems. Furthermore, our study area is a meandering river section shortly before flowing into the Danube River, and the site conditions are transferable over many parts of Europe. Before the restoration measure, the river course and the floodplain areas were hardly laterally connected, which is a common problem in European rivers [110]. Due to the artificial lowering of the river corridor, the connection to the groundwater level and the flooding of the floodplain increased, which is an innovative approach worth transferring to other river sections.

6. Conclusions

Riparian ecosystems are highly susceptible to plant invasions. Therefore, understanding the specific requirements of the native tree species forming riparian forests, especially in terms of germination and establishment, can facilitate the development of efficient strategies to conserve these valuable habitats. The aim of the LIFE+ Traisen project was to create a new river corridor with valuable aquatic, semi-aquatic, and semi-terrestrial habitats. During early succession, riparian softwood species successfully colonized the floodplain, albeit together with the invasive species *Solidago gigantea*. A young riparian softwood forest could be established after only 8 years. Contrary to our expectations, we observed negative softwood tree development only when the *S. gigantea* cover exceeded 90%. This highlights the need for optimum habitat parameters during early succession, ideally before *S. gigantea* forms large monospecific populations, to facilitate the establishment of riparian softwood forest species. Further, neither flood height nor fine substrate cover played a dominant role in shaping tree establishment.

Large-scale river restoration projects, such as that presented here, are still rare, limiting our experience with such projects. This study is exceptional in that it provides the results of comprehensive vegetation monitoring along a newly created river section, starting immediately after the restoration measures and lasting for a period of 8 years. In this way, softwood forest development along succession could be monitored. Although the implementation of a monitoring regime is crucial to obtain positive outcomes, monitoring is often under-budgeted or neglected. Our monitoring results highlight the importance of long-term (at least 10 years) post-construction monitoring, especially in the context of changing environmental conditions and the ubiquitous occurrence of invasive species. This can help guide the succession toward the development of the desired habitat type.

Supplementary Materials: The following supporting information can be downloaded at https://www.mdpi.com/article/10.3390/w16233489/s1. Table S1: Spearman correlation matrix of the habitat parameters recorded in 2021.

Author Contributions: Conceptualization, I.B., G.E. and M.P.; methodology, I.B. and G.E.; software, I.B. and G.E.; validation, I.B. and G.E.; formal analysis, I.B. and G.E.; investigation, I.B. and G.E.; resources, G.E.; data curation, G.E. and I.B.; writing—original draft preparation, M.P., I.B. and G.E.; writing—review and editing, M.P., I.B. and G.E.; visualization, I.B. and M.P.; supervision, G.E.; project administration, G.E.; funding acquisition, G.E. and I.B. All authors have read and agreed to the published version of the manuscript.

Funding: This work is based on the results of a monitoring project financed by VERBUND Hydro Power GmbH. It was financially supported by the Schweisfurth Foundation within the framework of a doctoral scholarship awarded to I. Becker.

Data Availability Statement: The data presented in this study are available upon reasonable request from the corresponding author, as they were collected as part of commissioned work.

Acknowledgments: We thank the two anonymous reviewers for their valuable feedback, which has greatly improved the quality of our paper.

Conflicts of Interest: The authors declare no conflicts of interest.

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