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„Analysis of Hydroscape Degradation after the Strong
Disconnection of the Danube Floodplain, Lobau post 1938“

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Abstract

Since the regulation of the Danube in 1875 ongoing terrestrialization, a sinking groundwater table and afforestation have altered the former hydroscape in the Lobau to a high degree. Important aquatic and semiaquatic habitats for protected species have virtually disappeared. A multi temporal analysis based on aerial pictures from 1938 to 2004 was performed to quantify the amounts and distributions of wetland- and water body losses over time. Hydrological data were included to investigate the relationships between area losses and commonly assumed causes for hydroscape degradation.

It was found, that changes were more pronounced in wetlands than water bodies. In main channels the dominance of wetland- versus water loss increased with the degree of disconnection, whereas side channels showed stronger effects in both wetlands and water bodies. The highest wetland losses were correspondingly found in the isolated side channels of the mid and lower Lobau and in all the strongly disconnected channels of the upper Lobau. In recent times afforestation seems to have accelerated there. Small area, the complexity of area shape and deepening of the Danube were connected to increased wetland loss in side channels. The analyses also showed that afforestation may have been hindered by more frequent high floods. For water bodies, expanses of main channels were less reduced with the degree of disconnection. Water expanse loss only occurred in the lower Lobau. Allogenic sediment intake is therefore assumed to have caused a higher loss of water bodies than the aggradation of autogenic sediments. Side channels showed the largest reduction in water expanse, but this seemed to have occurred in specific time intervals. Besides terrestrialization, the aspect of ongoing afforestation, the decay of biodiversity in floodplain forests and the added negative effect of inadequate flow conditions on afforestation would therefore be strong pro-arguments for reconnection measures in floodplain rehabilitation programs.

Introduction

“Former floodplains” of rivers are rare and highly biodiverse ecosystems, that have been endangered due to their disconnection and progressive degeneration. Particularly, in Europe and North America, these last remnants of floodplains have been decoupled from the rivers by human intervention. Initially, they had served as refuges for a wide number of protected plant and animal species. However, ongoing terrestrialization, dewatering of the areas and natural afforestation have produced a continuous degeneration of these important habitats. Since the end of the 19th century, construction of flood banks and the channelization of the riverbeds have cut off most floodplains from their rivers, in the interest of flood protection and navigation. Human intervention in these areas and its consequence for these habitats has not stopped. Requirements for energy supply as well as more efficient navigation and flood protection have lead to more intensive human intervention in these complex systems (Tockner & Stanford 2002). On the other hand, the advantages of floodplains as retention areas for flood protection have been recognized in the past decade. Their reintegration into riverscapes has thus gained new importance (Fiselier & Oosterberg 2004). Together with the development of increased ecological awareness and the fear of “climate change” these anthropogenic manipulations have placed a new emphasis on the need to understand causes and effects associated with these human interventions and floodplain degeneration.

The natural hydrological interaction or so-called connectivity, between rivers and floodplains is indispensable for the floodplain maintenance and the ecological functioning of both. After regulation, connectivity is usually, highly reduced. Natural floodplains are spatial and temporal dynamic complex mosaics of terrestrial, semi-aquatic and aquatic habitats distinctly influenced by the river in different stages of succession. Lateral connectivity to the river is required to allow fluctuating flow and floods into side arms that results in dynamic habitat regeneration and rejuvenation related to erosion and aggradation of transported sediments. Lateral and vertical connectivity, via groundwater, create fluctuating water levels that can produce a wide range of ecotones between aquatic and terrestrial habitats. River water transports large amounts of particulate and dissolved matter from catchment areas. It is high in allogenic sediments and nutrients, generated in or transported from the surroundings. Floodplains are therefore highly productive ecosystems with internal processes enhanced by river water (Amoros & Bornette 2002).

The construction of flood banks has primarily restricted lateral connectivity. So-called “former floodplains” were the result. Floods and their erosion-aggradation effects no longer

renewed and rejuvenated the disconnected water bodies. Instead the remaining connection provided nutrient input from the river water vertically, via groundwater, and when present also laterally via remnant surface connectivity. The remnant lateral connectivity may also stimulate more sediment input than output. As a result, the natural steady state equilibrium of destruction and (re)generation in these highly productive habitats would be or has been reduced to a unidirectional, enhanced ecological succession (Schiemer 1995).

With regard to the phenomena associated with floodplain degeneration, the aggradation of fine sediment produces terrestrialization. If no flushing follows, water body expanses are reduced in the long run and replaced by semi-aquatic wetlands. Amoros (1987) distinguished two kinds of terrestrialization, allogenic and autogenic. Allogenic terrestrialization occurs mainly in dead-end arms, when a remnant surface connection to the river is still available. It is mainly based on the aggradation of sandy-silt fine sediments, imported by river water. Its sedimentation takes place in areas and times with decelerated current. If output is lower than input, aggradation occurs. Autogenic terrestrialization happens primarily in more isolated backwaters. It is mainly caused by the aggradation of organic fine sediment produced by internal processes: The production of organic matter by primary producers (macrophytes and algae) and its degradation to detritus. With better light conditions due to a decreasing water depth, the littoral vegetation can progress into the center of the water body, and reinforce further aggradation. The progression of the littoral vegetation begins with submergent hydrophytes, followed by emergent (swimming) hydrophytes and ends with helophytes that tolerate permanent inundation and dry terrestrial conditions (Lüderitz 2009). Characteristic genera in this progression are *Myriophyllum*, *Nupha* and *Phragmites*. Allogenic nutrient input from connectivity to river and local terrestrial input -mainly litter fall- from the surroundings enhance internal processes and therefore autogenic terrestrialization (Schiemer 1995).

Thus, the process of water body terrestrialization reflects the location within the former floodplain, the geomorphology, the trophic state and the vegetation. The location defines the effect of the river via ground and/or surface water. Geomorphic characteristics define the morphology of the channel bed that changes with ongoing terrestrialization. Shallow water bodies generally terrestrialize faster, than deep ones. The trophic state determines the productivity and degradation, smaller or strongly terrestrialized water bodies are often eutroph or hypertroph, expressing a high productivity. Possible anoxic conditions can further re-suspend nutrients and slow down degradation processes. Terrestrialization is subsequently enhanced. The type of vegetation is mainly determined by the degree of

connectivity (Lüderitz 2009). In isolated water bodies with low current, macrophytes can grow being the dominant primary producers, whereas benthic algae and further phytoplankton follow with the degree of connectivity (Schiemer 2006).

After terrestrial ground has developed, afforestation can take place. It is the succession of floodplain forest causing the loss of semi-aquatic habitats. Limits for the afforestation are mainly water level fluctuations and floods causing inundation. Pioneer species therefore have to be adapted against anoxia and physical damage during floods (Schnitzler 1997). The vertical and lateral disconnections enhance forest development so that less adapted species can become established since they are then more competitive. Another type of afforestation is the alluvial island succession. It is an immediate effect of river erosion and rejuvenation and is therefore a predominant phenomenon found in former floodplains, after their disconnection. *Salix purpurea* is an example of a pioneer species, which is tolerant of water fluctuations and inundation. As the ground stabilizes and soil develops other than more competitive species of soft wood forest follow (Ward et al. 2002).

In former floodplains the increased ground elevation from aggradation can be intensified by sinking, groundwater tables. Dewatering accelerates terrestrialization and afforestation. Channelization of the riverbed, results in a concentration of erosive forces that deepen the riverbed and lower the groundwater table. When the river aquifer lowers, higher amplitudes of fluctuation are required to affect water bodies in former floodplains. Frequencies of flooding are, however, also lowered (Schiemer et al. 1999). Drainage measurements for land reclamation are another example for human intervention with the same consequences (Tockner & Stanford 2002).

Schiemer (1999) has schematically described the long-term shift of habitats for disconnected floodplains after regulation of the Danube. Directly after regulation, an increase of one-sided connections (plesio- and parapotamon) and isolated aquatic habitats (palaeopotamon) are found in the cut-off of former connected side arms. The original semi-aquatic wetlands in the outer borders of the floodplain declines drastically after regulation. Subsequently, during the transition from plesio-/para- to palaeopotamon due to ongoing terrestrialization and fragmentation, wetland area can increase again. In a latter phase, the decline of all habitats follows and afforestation takes its course. The consequences for species that are dependent on flowing or fluctuating conditions in aquatic and semi-aquatic habitats of floodplains are horrendous. Rheophilic fishes and mollusks dependent on flowing conditions are replaced by stagnophilic species, typical for lentic conditions. Today, especially macrophytes have become highly endangered species due to the strong loss of wetlands directly after regulation. The Lobau therefore serves as a refuge for them and in

addition also to many species endangered due to a depauperated landscape, where land reclamation eliminated most wetlands, not only floodplains. Unfortunately this refuge is being exposed to decay (Schiemer 1995).

In order to document and analyze long-term changes due to terrestrialization, dewatering and afforestation after regulation, historical data on a landscape scale with a high temporal resolution are needed. Short term, detailed ecological studies of floodplains came into fashion for small areas in the 1970's. As snapshots, they were not adequate to document ecological progression. However, it has been assumed that aerial pictures that provide sequential snapshots of whole landscapes might be a good database for multi temporal analyses. Aerial surveys have been taken since the 1930s and with the development of aviation. Since then these have been done regularly in many countries, providing large-scale historical database for analyses (Albertz 2007).

On the basis of the ecological problems described above, the need for documentation, and the development of computational techniques, we decided to apply the aerial photography database in a multi temporal landscape analysis of the Lobau. It is a former Danube floodplain in Vienna, Austria, that continues to exhibit a high biodiversity even though it has been disconnected from the Danube since 1875. The degradations of aquatic and semiaquatic habitats are known to have continued since then (Schiemer 1995, 1999). For this reason, the Lobau has been integrated into several conservation and rehabilitation programs over the past 30 years. In 1992 it was even declared as part of the Alluvial Zone National Park Donau-Auen. To maintain this refuge for a wide range of endangered species, measures have been undertaken to restrict further degeneration. A water enhancement scheme (WES) for the upper Lobau was planned in the late 1980s and finally implemented in 2002 (Weigelhofer et al. sub.). General reconnection measures are still being planned but have not been implemented (Weigelhofer et al. 2008).

Alterations in floodplain areas and habitat shifts in the Lobau, that occurred after the regulation of the Danube have been documented on the basis of photographic material by Doppler (1991) and additionally on historical maps by Eberstaller-Fleischanderl & Hohensinner (2004). The post disconnection trends that were actually common knowledge, were quantified in these studies, using databases with short or lower temporal resolutions. Besides these studies, other "snapshot" studies have been done to explore the recent conditions and processes immediately involved in the current degeneration, again with the disadvantages of their temporal frame. For this reason we began a multi temporal analysis with a higher temporal resolution by examining aerial photographs taken between 1938 and 2005. The main objectives were to document spatial and temporal

changes in both, semi-aquatic and aquatic habitats, to compare them with hydrological data and examine differences between assumed autogenic and allogenic dominated terrestrialized areas.

The specific questions addressed in this study were:

How has the terrestrialization of water bodies developed compared to the afforestation of wetlands?

Do losses in wetlands and water bodies, dominated by autogenic or allogenic terrestrialization differ?

How do the losses of water bodies and wetlands compare in side and main channels?

In wetlands dominated by autogenic terrestrialization, with allogenic nutrient input: Have losses been more influenced by combined effects?

Were the commonly assumed causal factors for degeneration from other studies statistically associated with the area losses in wetlands and water bodies in the Lobau?

And, Do these factors support allogenic and/or autogenic processes in the corresponding subdivisions, or was dewatering of the area mainly due to a deepening of the groundwater table?

In order to address these questions, spatial analyses were performed on main and side channels of 3 sectors in the Lobau that represented a hydrological gradient of surface and groundwater, connectivity and geomorphologic area types. Wetlands and water bodies were investigated separately. Hydrological and geomorphological data were included to assess their relationships with area losses. A further subdivision was made into individual homogeneous wetlands in both side and main channels within the three sectors to examine localized trends. It was assumed that enhanced terrestrialization resulted directly in enhanced afforestation in the long run. The more connected a subdivision was, superficially or with nutrient rich groundwater, the higher the area loss that would have taken place. A dominance of hydrological factors was assumed to support the idea of either dominant allogenic or enhanced autogenic terrestrialization by nutrient input from river water. Dominating geomorphic factors were taken to support autogenic terrestrialization enhanced by terrestrial litter fall.

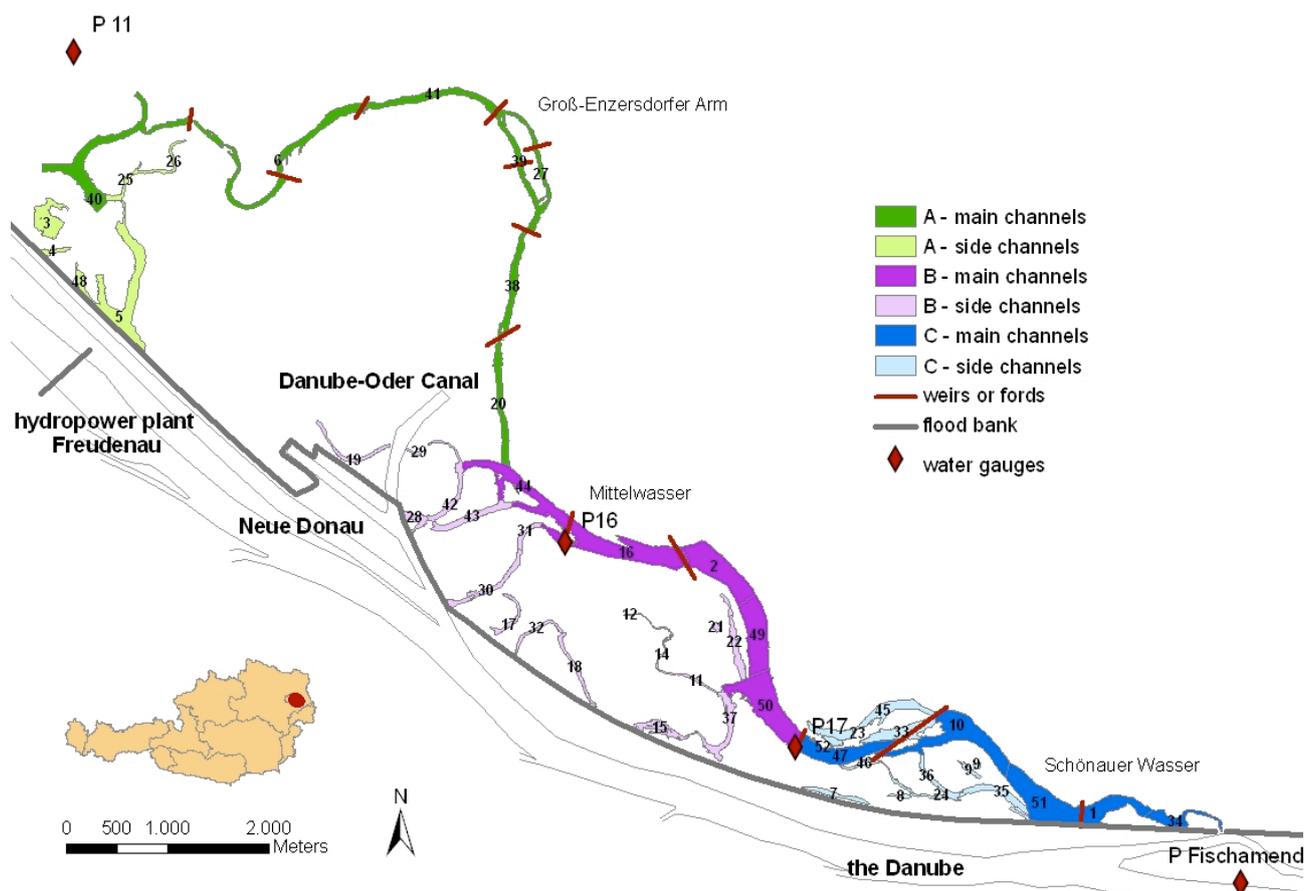


Figure 1: A schematic map of the recent Lobau. Wetlands and water bodies investigated over the study were merged. Human implementations are: flood banks (grey lines), weirs (red lines) and water gauges (rhombus). The subdivisions used in the analyses are expressed in colors. Sector A (green), B (violet), C (blue), main channels (dark), side channels (light) and subareas (numbered).

Methods

The Study Area: History and Hydrological Characteristics

The study was performed on aerial images of the Lobau, a former dynamic floodplain of the left Danube riverbank. The study was focused on the unsettled area in the eastern parts of the former floodplain. An overview of the area and its structures is in Figure 1. The Lobau as a whole is about 20 km long and is located in a northeastern municipality of Vienna, Austria. To the northeast it borders the alluvial plain of Marchfeld, which is of importance due to its groundwater contribution. The upstream western part of the Lobau has an altitude of 156 m that slopes down to 150 m at the downstream eastern end.

After the river regulation in 1875, the area was almost completely disconnected by a flood bank. Disconnection caused severe alterations in the hydrological and geomorphological characteristics, as a result of terrestrialization, dewatering and afforestation. The strongest morphological alterations took place directly after regulation (Ebertstaller-Fleischanderl &

Hohensinner 2004), since the shortening of the channelized Danube caused a stronger river current and with it a lowered groundwater table (Schratt-Ehrendorfer 2011). An equilibrium has not been established yet and so the process is continuing.

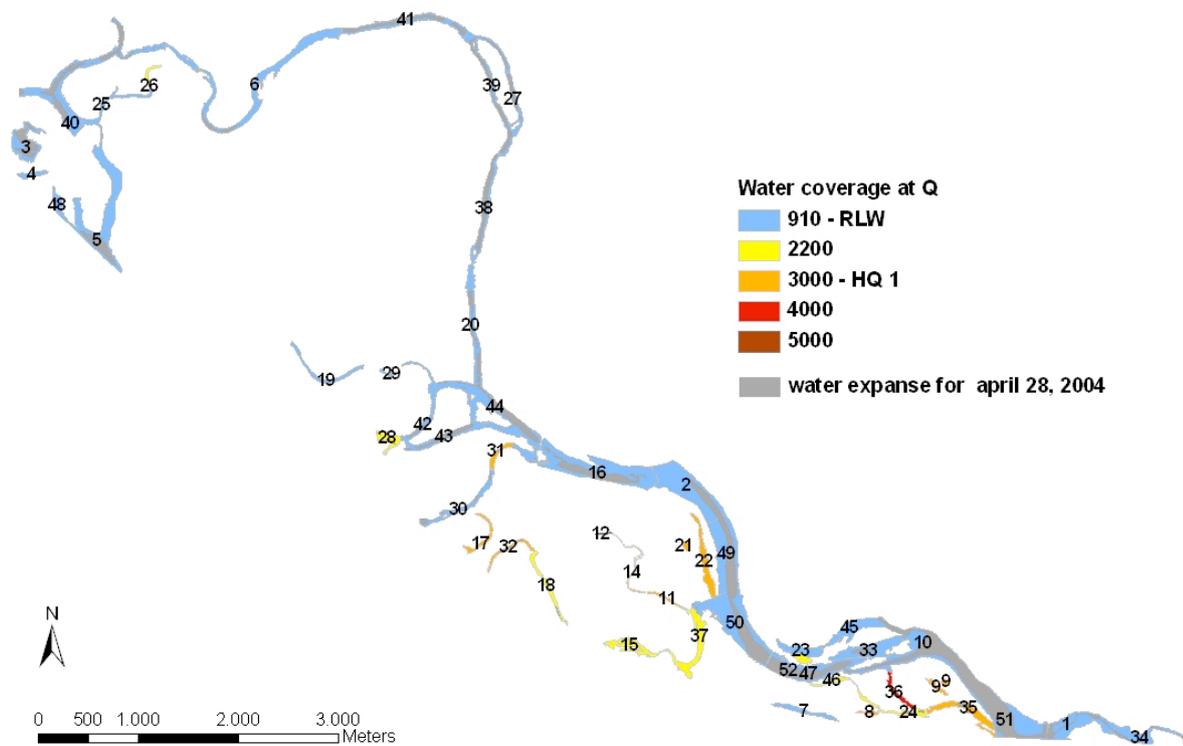
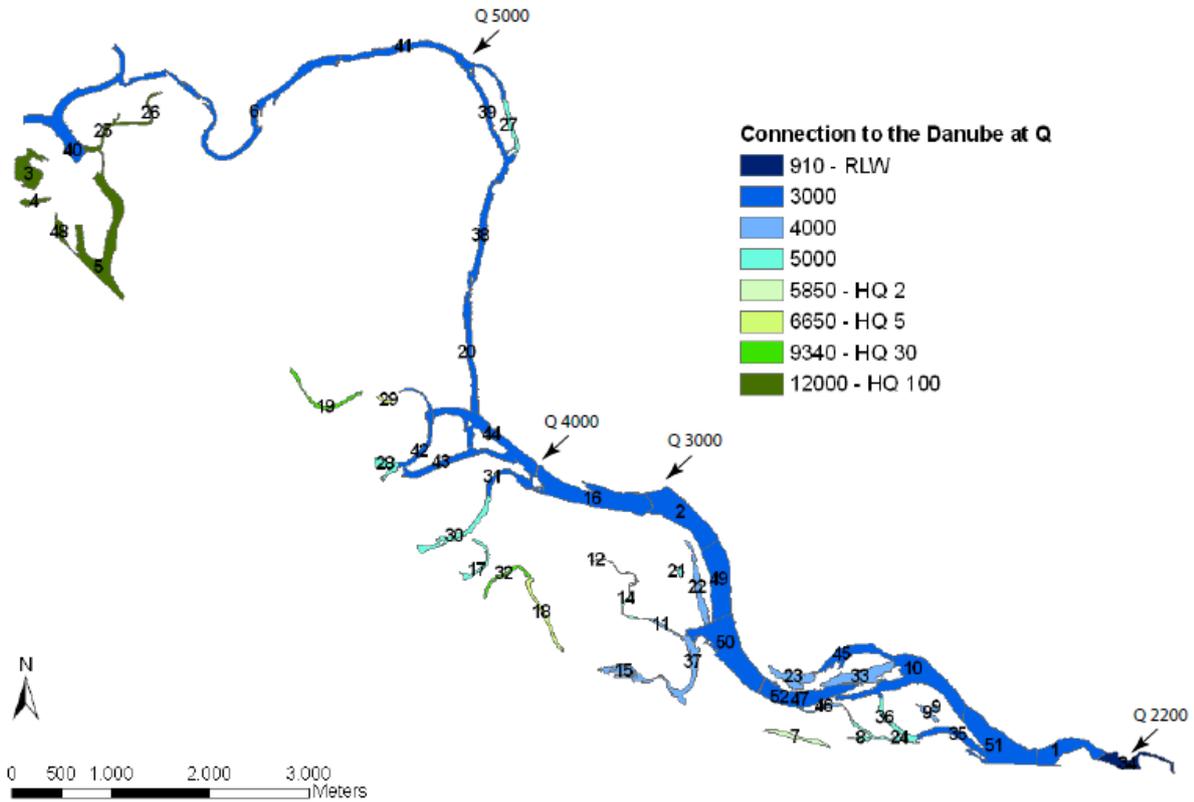
After regulation, the Lobau was mainly connected to the river via groundwater. A downstream disruption in the flood bank (the inlet) was left to drain the floodplain. This has also served as a back flowing connection from the river and hence is the only surface link between the Danube and the Lobau (Weigelhofer et al. sub.). The former erosive and scouring forces of through flow and floods that produced habitat rejuvenation and regeneration have been completely lost. In contrast, unidirectional terrestrialization and forest succession processes have continued undisrupted (Ebertstaller-Fleischanderl & Hohensinner 2004). The superficial connectivity present through the inlet has been assumed to enhance terrestrialization processes at the lower end by backflow input of the nutrient and sediment rich water (e.g. Jelem 1972). The distribution of superficial connection at different discharges of the Danube is shown in Figure 2 above. The thresholds for backflow shown in the figure are a detailed indication of the input. Indicators for the Lobau's disconnection are the large populations of macrophytes in the water bodies. (Schiemer et al. 2006)

In addition to the flood banks, the channelization of the Danube, done during river regulation has enhanced the disconnection of the Lobau by lowering the groundwater table and producing a loss of superficial water. The frequency of connection and the maximum amplitude of fluctuating water have decreased (Schiemer et al. 1999). In Figure 3 the deepening of the riverbed has been documented, by plotting the water levels at regulated low water over time (a standard value for navigation published by the Danube Commission). Accordingly, between 1938 and 2005 there was a change of 0.94 m. Since 1875 it has deepened approximately 1.5 m, between 1.5 and 2 cm per year (Schiemer et al. 1999). Finally, in Figure 4 the decreased frequencies of floods are shown expressing the biannual maximal discharge – HQ 2.

Another aspect enhancing disconnection and fragmentation of individual water bodies has been the ongoing aggradation of fine sediment caused by autogenic terrestrialization that caused further clogging from the groundwater aquifer (Schiemer et al. 1999). Smaller, isolated channels were more prone to these effects due to their high autogenic terrestrialization relative to bigger basins. A high individuality concerning their fine sediment layers has been documented in the lower and mid Lobau by Reckendorfer & Hein (2000, 2004).

After river regulation, human activity continued to interfere with the system. When the problem of dewatering was recognized, weirs were built to retain the water inside the floodplain, which resulted in a break down of the former channels into individual basins. This also enhanced terrestrialization (Imhof et al. 1992). The sediment load of river water has been deposited in front of the weirs closest to the inlet during inflow (Reckendorfer & Hein 2000, 2004). Thereafter, the retention of the nutrient rich river water during outflow has also enhanced internal processes and terrestrialization (Schiemer 1995, Schiemer et al. 2006). The section of the Danube Oder Canal built from 1938 to 1943 produced a hydrological barrier for groundwater between the upper and lower Lobau. In 1966 the first groundwater withdrawal stations for drinking water were opened in the lower Lobau (Jelem 1972). They are still used today (Schiemer 1995).

Between 1972 and 1984 a flood relief channel, the Neue Donau, was built parallel to the left bank of the Danube. For the upper Lobau this created an additional barrier against the groundwater connection to the river (Weigelhofer et al. sub.). During the 1970^s there was also a period of low precipitation that produced further water level decreases in the upper Lobau (Imhof et al. 1992). This can be seen in Figure 5 where the mean annual water levels of the four water gauges have been plotted. P11 located in the upper Lobau can be taken as a marker for the Marchfeld groundwater table. In 1997, the hydropower plant Freudenau began operation in the Danube. On one hand, with this impoundment, a stabilization or even increase of the groundwater table occurred in the upper Lobau. On the other hand, the impoundment reduced fluctuations in groundwater (Figure 5) (Weigelhofer et al. sub.). The ongoing terrestrialization, afforestation and water loss has left clear marks on the outward appearance of the Lobau post river regulation. Many former channels have become fragmented and even astatic water bodies, i.e. with temporary water cover. Figure 2 below illustrates the spatial distribution of water bodies today and the discharge of the Danube, at which they are covered with water.



Figures 2: Above: Distribution of superficial connection to the Danube with thresholds of backflow at discharges of the Danube. **Below:** Distribution of water coverage at minimum discharge of the Danube and water expanses of the composite picture of 2004. Both figures are based on the ground- and surface water model and geodataset. (see Methods)

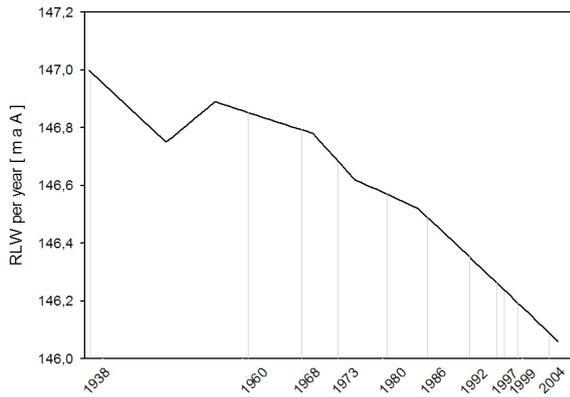


Figure 3: The deepening of the Danube's riverbed during the study expressed by means of the regulated low water (RLW) levels (level of the lowest water table permitted for shipping). Grey lines indicate the years represented by composite pictures.

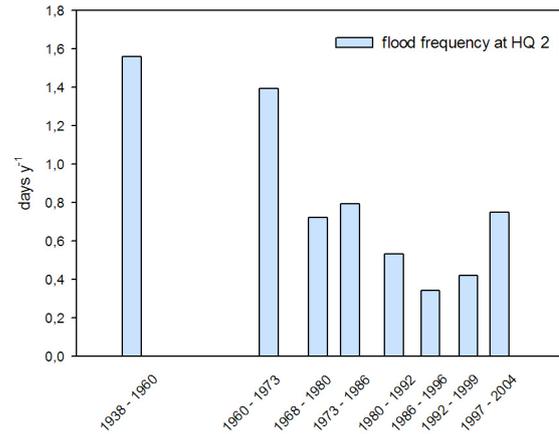


Figure 4: The average number of days per year with Danube discharges above HQ 2 for each time interval used in the analysis. HQ 2: biannual maximum discharge of the Danube river.

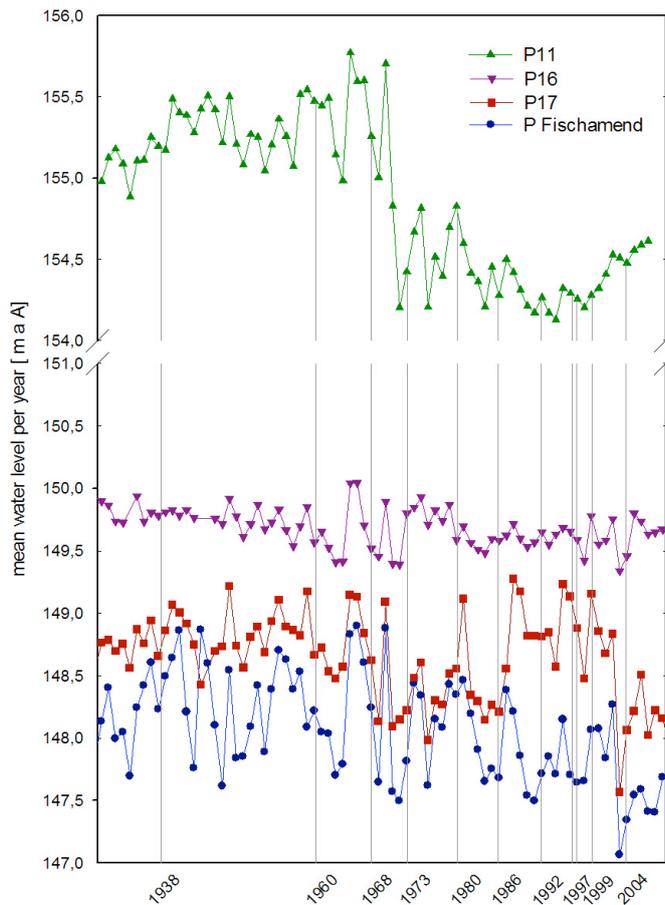


Figure 5: Annual means of water gauge measurements. Water gauge P11 represents water levels of Sector A, P16 and P17 Sector B and P Fischamend located in the Danube Sector C. The years with composite pictures are indicated with grey lines.

Resources

Photographic material

In this study, either aerial pictures or, in more recent years, orthophotos were sought out as a basis for the multi-temporal landscape analysis of the Lobau between 1938 and 2004. An initial problem was that in some of the available photographic material found, small parts of the proposed study area were missing. The specific solutions to this problem will be described later. In short, composite pictures from different years were constructed and analyses were adjusted to compensate for the calendar differences. The materials found for the analyses are listed in Table 1. The pictures on dates that lacked areas and were composed of composite photographs are grouped by background in the table. From 1960 onwards the material was obtained from the Austrian Bundesamt für Eich- und Vermessungswesen. The orthophotos after 1997 were released by the Donau-Auen National Park as part of the Biotoptypenkartierung. The 1938 photos were found as large grey-scaled photos in the Wiener Stadt Archiv. During scanning at 600 dpi they lost quite a bit of resolution. Unfortunately, there were large differences in both the techniques used to photograph the areas and consequently the quality of their reproduction. The range was from grey-scaled pictures with low resolution to color infrared with high resolution. There was some data from 1944 and 1945 however the resolution was so low, that they were omitted from analysis on the basis of their incompatibility.

Date	flight altitude [m] / ground resolution [m]	Type	quality
1938 Oct/Nov		aerial photos – B/W	low
1944 Sep-Dec	?	aerial photos - B/W	very low
1945 Apr 01	?	aerial photos - B/W	very low
1960 Aug 17	1350	aerial photos - B/W	good
1968 Jun 14		aerial photos - B/W	good
1973 Aug 06	3200	aerial photos - B/W	good
1973 Oct 07	3200	aerial photos - B/W	low
1980 Aug 01	3400	aerial photos - CIR	very good
1986 May 06	6500	aerial photos - B/W	good
1992 May 25	4800	aerial photos - B/W	good
1992 Jun 16	0.5	orthophoto - B/W	good
1996 Oct 04	0.5	orthophoto - B/W	good
1997 May 15	0.5	orthophoto - CIR	very good
1999 Jun 20	0.25	orthophoto - colored	very good
2000 Jun 10	0.25	orthophoto - colored	very good
2004 Apr 28	0.25	orthophoto - colored	very good
2005 Jun 24	0.25	orthophoto - colored	very good

Table 1: Photographic material used in the study. The date of shooting, the flight altitude for aerial pictures, the ground resolution for orthophotos, the type of image as well as the quality are shown. Pictures used in a composite image are grouped by background, shaded or blank. B/W: Black and White, CIR: Color infrared

Geographic data

Digital terrain model (DTM) from 2006. The model was obtained from the project Optima Lobau (Weigelhofer et al. 2008). It was used for calculations of certain predictors, as a basis to equate water expanses at different water levels and as reference in classifying water bodies.

Field mapping data of biotopes. They were released by the Donau-Auen National Park after the census Biotoptypenkartierung 1997 (Burger & Dogan-Bacher 1997). The data were used as a reference for classification of wetlands and water bodies and to distinguish helo- from hydrophytic vegetation.

Groundwater-surface water model: This model was generated for the project Optima Lobau, where a hydraulic ground- and a surface water model were coupled. As a two dimensional numeric model, it was calibrated for a recent situation (in 2006, Weigelhofer et al. 2008). It projects the spatial distribution of connected and unconnected surface water expanses in the Lobau at different discharges of the Danube: $910\text{m}^3\text{s}^{-1}$, $2200\text{m}^3\text{s}^{-1}$, $3000\text{m}^3\text{s}^{-1}$, $4000\text{m}^3\text{s}^{-1}$ and $5000\text{m}^3\text{s}^{-1}$. Using this model it was possible to classify the subareas (the smallest unit of subdivision used in this study) into five groups of superficial connectivity and water coverage.

Geodataset of water expanses: This dataset also contains assessments of connected water expanses of the Lobau but calculated for higher discharges of the Danube of $5300\text{m}^3\text{s}^{-1}$ (HQ1 -annual), $5850\text{m}^3\text{s}^{-1}$ (HQ2 -biennial), $6650\text{m}^3\text{s}^{-1}$ (HQ5 -quinquennial), $7300\text{m}^3\text{s}^{-1}$ (HQ10 -decennial), $9350\text{m}^3\text{s}^{-1}$ (HQ30), $10340\text{m}^3\text{s}^{-1}$ (HQ100 -centennial). It was used for the same calculations as the groundwater – surface water model.

Hydrographic data:

Water levels and discharges at water gauge P Fischamend, daily from 1893 – 2009, were obtained from the Via Donau. The calculations of discharge were based on, or better calculated from, the Danube's riverbed at 1956.

Water levels at the water gauges P11, P16 and P17 located in the Lobau were provided from the Gruppe Wasser. They were available over the whole time of investigation from 1916, 1923 or 1924 onwards.

Dataset Generation

The data used to analyze the changes of semi-aquatic and aquatic habitats were primarily based on areal analyses of the photographs combined with the dates and time intervals between shooting. In addition the other data resources were used to generate predictors for the statistical analysis. Both groups of data were combined in the geographic information system (GIS) software ArcView 9.2 of ESRI. GIS software principally provided a system for the generation, analyses, management and presentation of digital spatial data. The data collection was done in the following steps.

a) Preprocessing of aerial photos

Before areas could be defined and classified, the digital aerial pictures (from 1938 to May 2nd 1992) had to be georeferenced and rectified. It was then possible to overlay composite pictures from different years with other geodata for comparison to perform correct calculations. During georeferencing the spatial location of data was defined. To do this, the data (pictures) were aligned on a map coordinate system, a flat projected coordinate system based on a spherical geographic coordinates that defined the curved surface of the earth. A rectification was necessary, because the aerial photographs depicted the earth's surface in a central projection that caused distortions of metrics (e.g. areas and shapes) on the topographic relief. Due to the fact that the pictures were never taken from the same horizontal, vertical and inclined position, the distortions of the pictures were not comparable in a multi temporal analysis. During rectification the central projection was changed into an orthogonal one with a uniform scale in every photo (Albertz 2007). Noticeably, the distortion of vertical objects like trees located on the topographic relief could not be corrected in this way.

In GIS, georeferencing and rectification were performed simultaneously by identifying ground control points (GCP) of previously spatially referenced and rectified pictures. The raster coordinates of the GCP's of the distorted picture were then translated into the referenced map coordinates. The geometric transformation could then be defined by 2nd order polynomials. A resampling had to be done afterwards to allocate the pixel values to the corrected coordinate system. A cubic convolution algorithm was used since it integrates a high number of neighbored pixel values and leads to the best results (Albertz 2007). As an indication of the preciseness of georeferencing, the root mean square error (RMSE) was calculated, based on the residuals between the old and new values of the rectified coordinates. In contrast to aerial pictures, orthophotos were already preprocessed by orthorectification, a method including the topographic relief on the basis of a digital terrain model and parameters of the camera used. In general, orthorectification is the most

precise method to adjust for distortions, since it takes the slopes and elevations of the landscape directly into account. However, Rocchini showed, that for relatively flat terrains, a polynomial rectification of second order lead to the same quality of correction, when 20 GCP's were employed (Rocchini, Di Rita 2005).

For this study, the initial references were the composite orthophotos of 2004 and 2005. Because identification of these reference points became more difficult as one went into the past, it was necessary to use rectified pictures to accurately locate objects and reference points in older datasets. For instance 1968 was used for 1938 and 1960. Thirty GCP's were defined in each picture. This was the suggested minimum by Hughes et al. (2005). For the 1960 photograph, only around 10 GCP's were found per picture, due to the low altitude flown during photography. The GCP's chosen were always dispersed over a picture with preferential placement near water bodies. The RMSE was approximately 1 m, except for the 1938 photos where RMSE was 5 m. The 1938 photos had actually been rectified beforehand with an unknown method. However, after georeferencing they were rectified again because some extreme distortions were present. Hard GCP's preferably used were edges of walls, towers of overhead power lines and hectometers along the shorelines of the Danube. In older pictures the bases of trees near water bodies had to be used, since many of the later constructions did not exist.

For this study all geodata including photographic material were projected with the Transverse Mercator projection to the projected coordinate system of MGI_Austria_GK_East, based on the GCS_MGI geographic coordinate system.

b) Image classification of wetlands and water bodies

To compare the time series of pictures quantitatively, specific areas had to be defined that could be measured on screen by visual interpretation. This method was preferable to automatic classification, since the photos had a high spatial resolution and single objects were easily recognized. In contrast, the *spectral* resolution was low, especially in black-and-white reproductions. Moreover there were extreme differences in quality even within one composite picture, so the same classes of different images can be identified more easily visually than automatically. Therefore, visual interpretation can be seen as a necessary evil that is always shaded by subjectivity (Albertz 2007).

Two classes of habitats were used in the analyses: Wetlands and water bodies. Wetlands were defined semi-aquatic and aquatic areas in and around existing or former channels with a less than 25 % cover ratio of wood (including shrubs). The boundaries of the floodplain forest, loose woods or shrubs, were taken as the outer boundaries of wetland areas. Woodland boundaries were prone to some error: Tree bases were often covered by

their crowns and had to be estimated. Shade and/or distortion were present. In areas where pictures overlapped, two perspectives were used for better interpretation. Using the standard for field surveys based on aerial pictures for the Federal Republic of Germany (Eigner & Tschach 1995) shrubs and woods typical for terrestrialized areas were not identifiable and were therefore not classified separately.

Compared to wetlands, water bodies were defined as the visible open surface water surrounded by a boundary of the channel bed. In areas where the shoreline was covered with helophytic vegetation, the edge of the vegetation was used as the water body boundary. It should be noted that the helophytic border did not vary with changes in water levels. This is in contrast to the water boundary of the channel bed. Details and consequences of this for the analyses have been presented in the results. Since hydrophytic vegetation was a temporally and seasonally limited water cover, and their presence varied from one time point to the next, this type of vegetation was classified as part of the water body. There were, however, difficulties in distinguishing helo- from hydrophytic vegetation. These were a source of error in the defined boundaries.

A classification key was produced in advance for the composite pictures of every available year to ensure objectivity during classification (Albertz 2007). It contained characteristic cutouts of pictures and descriptions of color, texture and physiognomy (shape and texture) of the classified objects. The field mapping data of biotopes (see materials) served as reference, since it involved photogrammetric (3-D) interpretation of color-infrared orthophotos, which assured a much higher accuracy of interpretation than the performed 2-D interpretation used in this study. This was true especially for types of vegetation as mentioned above. In cases of low recognizability, the picture quality was improved by changing the contrast and/or intensity of the pixel values with ArcView 9.2 Effects toolbar. The classification occurred simultaneously with the definition of boundaries. The most recent picture was classified first, with the later years following chronologically. Classification was performed with the editor toolbar on a scale of 1:1500 m. In cases of abrupt loss of woodlands, the wetland boundary of the previous time point for that section of the boundary was taken, since forestry measures were assumed. The result was a series of spatially referenced digital maps containing polygons of the wetlands- and water body areas for each composite picture.

c) Field validation and attribution of dates

After classification every year was checked and corrected for misinterpretations, missing classes or areas and overlapping polygons. A comparison was done to check for plausibility among sequential data points. Moreover the correct dates of photography had

to be assigned for the corresponding wetland and water body areas. This was done by coupling the maps of every year with prepared geodata including the time boundaries and exact dates as attributes with the Intercept function of the Analysis tools.

d) Division into subareas, sectors and main and side channels

The next step before analysis was to define units of wetlands, including water bodies that could be considered as individual systems on the smallest scale for this study, the subareas. They were further classified as part of the large-scale system in the Lobau, the sectors and as main and side channels within the sectors.

A **subarea** was defined as a wetland of a former arm of the Danube that was surrounded by forest or separated from the former continuum by artificial weirs or natural fords. It is therefore hydrologically and geomorphologically distinct and when present contains individual water bodies. The DTM and the road network of the city of Vienna and Lower Austria were used as reference. In the interest of homogeneity the other geodata were used to check for further differences in hydrology. Longer continuous channels were subdivided into equal parts because of their size. Subareas that could be assigned to two different dates of composite pictures were assigned to the date of the biggest part. With these definitions of the subareas, the corresponding wetlands and water expanses were analyzed. In total 52 homogenic subareas were identified.

The sectors were defined as broad zones, summarizing subareas with a specific distance to the inlet. Hence each sector had a distinct interaction with the Danube by superficial connectivity and its aquifer and was distinctly influenced by the Marchfeld aquifer. A dominant type of terrestrialization (allogenic, autogenic influenced by the river and litter fall, autogenic influenced by litter fall) was assumed to be present in the main channels.

Sector A was defined as all subareas northwest of the Danube-Oder Canal, the so-called Upper Lobau. It represents the most disconnected part of the Lobau where water bodies are fed principally by groundwater (e.g. Jelem 1972). Due to its location and the Neue Donau after 1984 it is most influenced by the Marchfeld groundwater aquifer and thus less influenced by the nutrient rich Danube aquifer compared to the other sectors. It is known to be drier than the mid and lower Lobau (Imhof 1992). Terrestrialization and afforestation are assumed to be less enhanced by the main river, instead human activity in terms of management measures (excavations) and recreational use (fishing, bathing) have been more pronounced here than in Sectors B and C (Imhof 1992; Weigelhofer et al. sub.). This area has been influenced most by the implementations of the Neue Donau and the hydropower plant. Subarea 19 was chosen to be a part of Sector B, since in 1938 it was still a connected part to the former side arm of subarea 29 and 42.

Sector B as the upper part of the so-called Lower Lobau was a transition zone between A and C. For the sake of clarity here it is called the mid Lobau. The main channels were found to be influenced by backflowing, superficial, nutrient input from the Danube albeit to lesser extent than subareas of Sector C (Hein 2000). Groundwater table fluctuations here correspond more to the Danube water level than in A (Imhof 1992). Terrestrialization processes here, are presumably autogenic processes, being influenced in main channels by nutrients from the river water. Allogenic sediments were found to be negligible in the first subareas of the main channels (Reckendorfer & Hein 2000, 2004). The sector contained all adjacent subareas up to the Gänshaufentraverse –the weir between subarea 50 and 52.

Sector C was the most connected sector of the Lobau to the Danube and therefore most influenced by the effects of riverine backflow. After 1984 the groundwater fluctuations corresponded to the Danube water levels, due to the missing influence of the flood relief basin (Imhof et al. 1992). Terrestrialization processes in this sector were enhanced by riverine water and allogenic sediment intake from the Danube (Reckendorfer & Hein 2000, 2004). Nonetheless, it has to be pointed out, that also autogenic processes may occur here, but more slowly. The gradual transition from allogenic to autogenic dominated terrestrialization is broken down by weirs and ending right before Sector B (Reckendorfer & Hein 2000, 2004). However, water retention is assumed to be shortest in this sector. The sector contained all subareas from Gänshaufentraverse until the inlet.

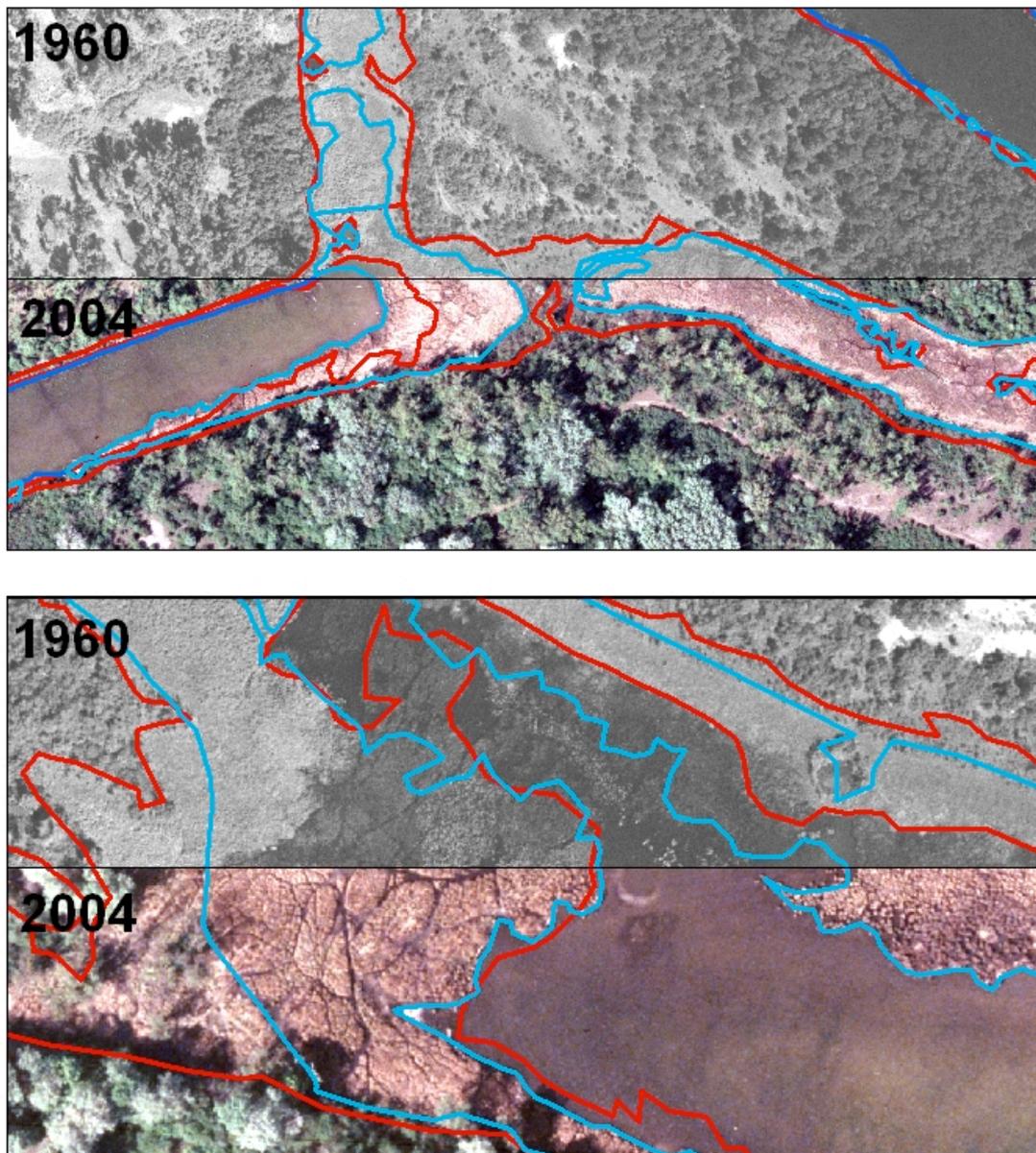
Within each sector, two types of water bodies were differentiated. One, the **Main channels** were defined as the subareas following the former main arm of the Danube. A superficial connection to the Danube was assumed in the study, however, with a decreasing effect of intake by riverine water -depending on the distance from the inlet. The other type was the side channels. **Side channels** are the smaller and more isolated former side arms, located next to the main side arms. Many subareas of side channels include astatic water bodies and are probably shallower than main channels. Side channels were assumed to exhibit more pronounced terrestrialization than main channels.

Data Analyses

Correcting water expanses for water level dependency – problems of detectability

Since water expanses of water bodies depend on the actual water level, patterns of the change of water expanses are more difficult to detect over time. For this reason the most obvious first step of analysis was to compare subareas at dates of similar water level. Since no patterns were found, the next approach was to equalize the measured water expanses to values corresponding to the same water level. Unfortunately there were several causes of problems connected with these analyses. To begin with, measurements of water levels were used as an indication of the hydrological situation for the water bodies of the whole sector on the day of shooting. This was caused by the fact, that water level measurements were only available from four gauges over the whole study. Only two of them were located in the area of investigation. The majority of water bodies were individual basins created by fords and traverses, connected only at certain water levels. A principal groundwater connection could be assumed, that caused a change in water levels observed in one basin, accompanied by changes in water levels in neighboring basins, but with a delay in time. Particularly side channels were assumed to show a higher degree in vertical disconnection due to relatively thick layers of fine sediment that have accumulated over time. The previous content of water in a basin before refilling was important as well. A period of low water before a flood event filled up the more disconnected basins with a longer delay, than would had happened after a previous high water period. Thus, these gauges did not reliably reflect the levels in the different subareas. On top of this, the data for the shooting dates were not always available and had to be estimated by the water level of the closest day or interpolated.

Finally, even if the data had been available, accurate changes in the boundaries of water bodies were only possible on bare shoreline. Where helophytic vegetation covered the actual shoreline, water expanses were independent of water levels and reflected the expansion and contraction of this vegetation. As these shores represent only a fraction of water body edges, an analysis was deemed to be too inaccurate. In addition, the classification of water bodies was prone to errors, since the boundary between open water and helophytic vegetation was hard to distinguish from hydrophytic vegetation. This was especially true during the growing season. Different types of photographic material, with distinct colors and qualities made this even more difficult. Figure 6 shows these problems on the basis of the photographic data of two areas taken in 1960 and 2004. For narrow side channels the major source of error in detecting the boundaries of water bodies, were the shades or distortion of the trees, which surrounded each wetland area.



Figures 6: Cutouts of photographic material from 1960 and 2004 Showing difficulties in detecting water expanses and changes in area loss. Lines: Boundaries found 1960 (red) and 2004 (blue) surrounding water bodies and wetlands. **Above:** Bare shoreline and helophytic vegetation surrounding detectable water expanse in Sector B. Strongest “water loss” by expanding helophytes. Fragmentation of former arm by afforestation. **Below:** Different outward appearance of helophytes in Sector C. 1960 harder to distinguish from hydrophytic vegetation. Contracting helophytic vegetation.

Nonetheless, for the sake of completeness, a partial analysis of the dataset with correction for the water levels was done. The idea was to exclude the measurable variation in water expanses caused by differences in the water tables. Two areas were taken into focus. One represented main channels. It was the lower section of Sector B where the water gauge P17 accurately measured the water levels of subareas 50, 49 and 2 up to the next weir. The other, representing side channels, was the upper part of Sector B above the weir where the water levels of P16 were relevant (subareas 19, 28, 29, 30, 31, 42, 43). On the

basis of the 2006 DTM theoretical water expanses were calculated for different water levels. This was performed with the ArcView Spatial Analyst – Less Than Equal tool. A function was calculated representing the relationship between theoretical water expanses and water levels. In order to compare the water expanses over time, the mean water level of the gauge for the dates of shooting was entered into the function and a new value for area was calculated. The corrected areas of water expanses were then plotted according to date, to analyzed for temporal variation.

Spatial analyses

To perform the spatial and statistical analysis, appropriate parameters were calculated on the basis of the generated areas and the other geo- and hydrographic data. To illustrate the developments of areas in the Lobau, basic statistics were used. Initially the subareas of subdivisions (sectors, main –side channel) were summed and the percent areas relative to an initial starting date were calculated. In a second step the subareas of each sector were examined according to type (main and side channels).

For wetlands the data of 1938 were used as reference year, since the boundaries of woods were distinctively recognizable. For Sector A, 1968 had to be used as reference. Between 1938 and 1960 human activity in forestry and dredging in the subareas 3, 5, 27, 38, 39 and 41 was obvious and had changed the landscape. 1960 could not be used for Sector A, because the photos lacked subareas 3, 4 and 40. For water bodies, in all sectors, the years 1938, 1960 and 1973 were not used. In 1938 and 1973 the photographic quality was low and water expanses were not recognizable. In 1960 relatively extreme high water levels were measured in all water gauges used, so that it was not comparable with the other years. For the same reason 1980 could not be used in Sector A, even though extreme water levels were only measured in A. In Sector B and C the composite picture of 2004 was not used because of the extremely low water levels.

Statistical tests and calculations were performed with the software R 2.12.1. To test if the development of area loss was significantly related to time, regression analyses were performed. To test for significant differences between main and side channels or between sectors, paired one-sided Wilcoxon tests were performed.

Main parameters for the spatial analysis were:

% area of 1938 - To illustrate the development of areas over time the absolute areas of the composite pictures were recalculated into percentages per reference years. As percentages, areas with different absolute values could be compared.

Woodland progression rate per year – Another measure for afforestation was the progression of woodland relative to wetlands in meters per year. To document the ranges for each sector, the changes in the boundaries of woodlands around wetlands were measured at transects representing the maximal, minimal and the general progression. They were then recalculated by dividing by the number of years of the whole time of investigation, as well as over the representative time intervals.

Median area loss per time interval - To give a measure of general loss over time in the spatial analysis, the area loss was expressed in % change per time interval. The median had to be taken since in most cases normal distribution was not present.

Apart from these parameters of spatial analysis another parameter was defined to describe the velocity of area loss. It was the **rate of area loss per year**: The standardization for the factor time minimized the influence of different lengths in time intervals. Additionally, the percentile measure allowed the comparison of changes in areas with different initial sizes. It was computed in percent per year as follows:

$$\text{Rate of area loss} = (- 1 + (F_2 / F_1)^{(1/t)}) * 100$$

F₁: area at the beginning of the time interval [m²]

F₂: area at the end of the time interval [m²]

t: duration of the time interval [y]

A disclaimer here is the fact, that the parameter time interval was the denominator in the exponent. Comparisons of areas over small time periods therefore overestimated rates of wetland change. For the sake of better understandability the rate was expressed as “rate of wetland loss”. Furthermore, losses of area were expressed as negative values, and gains as positive rates. “High losses” are therefore numerically low or negative. To make it more understandable for the reader the word “loss” already expresses the negative algebraic sign. This variable was used in the spatial analysis, to illustrate the differences in velocity of area loss for wetlands. Here the rates were calculated from the summarized subareas standing for the main and side channels in each sector. These subdivision rates of area loss and their corresponding median were also calculated for water bodies in the subareas. In the multi linear regression analysis this variable was used as the dependent variable for statistical analysis, the response variable (see below).

Statistical regression analyses

To look for statistical connections between the predictors and the rate of area loss per year and to see if there were different relationships within the main and side channels in each sector, multiple linear ordinary least square regression analyses were performed for these subdivisions. This was only done for wetlands and not for water bodies, for reasons described in the results. To ensure comparability, in every subdivision the same predictors were used. The response variable as mentioned was the annual rate of area loss and predictor variables were as listed below.

To maximize linear relations between each predictor and the response, the response was transformed by calculating the cube root of its values. If this did not help, the respective predictor was also transformed or a polynomial term of this predictor was integrated into the model. The selection of predictors to be used was based on a stepwise exclusion of the most insignificant predictors from the maximal model, which included all predictors and reasonable interactions (Crawley 2007). After the “principle of parsimony” the most simple, but most explaining model was taken as final. Diagnostics after Faraway (2002) to check the assumptions for OLS regressions were checked for the maximal and the final model. Autocorrelations within the data were tested with the Durbin Watson Test and by autocorrelation plots, the so-called correlograms. All the analyses were done with the software R 2.12.1.

In order to minimize the effect of errors in digitalized rates of wetland loss, longer time intervals were used to make statistical comparisons. Theoretically one would have taken proximate time samples for comparison. A problem with the dataset was, however, that sampling times were sometimes only one year apart, resulting in overestimations of area changes. In order to correct for this bias, samples were compared from alternate time points; one sample with the second sample thereafter, and not the proximate sample. By doing this, the autocorrelation between the time intervals was reduced to less than 42 % and was significant for lag 1 and maximum at lag 2. Besides, unavoidable inaccuracies by confining the boundaries of the areas carried less weight since changes in area were supposed to be bigger. Table 2 shows the time intervals used in the analysis.

In contrast to the spatial analysis, the actual dates of photography were used in the regression analyses. That means areas lacking photographs were not combined with photographs of other near dates to represent the main date of the composite picture. The employed time points and intervals are shown in Table 2. The only years that completely cover the area of investigation on a single date are colored.

Predictors or explanatory variables of the regression model

In order to statistically, investigate possible causal relationships between the rate of wetland loss and known factors for terrestrialization processes, dewatering, and in the long run afforestation, predictors were defined and calculated on the basis of logical arguments and data availability for each subarea and time interval. The list of predictors was created on the basis of available material and classical factors known to be associated with the process of wetland loss. Of the original 22 predictors 8 were kept in the end, since they were the most integrative and explanatory for the main factors and at the same time available for the biggest part of the subareas. Additionally, a spearman's correlation matrix was used as indication for strongly collinear variables, which were excluded as well.

Time periods	Duration in years
01.11.1938-17.08.1960	21.81
17.08.1960-06.08.1973	12.98
17.08.1960-07.10.1973	13.15
14.06.1968-01.08.1980	12.14
06.08.1973-06.05.1986	12.76
07.10.1973-06.05.1986	12.59
01.08.1980-25.05.1992	11.82
01.08.1980-16.06.1992	11.88
06.05.1986-15.05.1997	11.03
06.05.1986-04.10.1996	10.42
25.05.1992-20.07.1999	7.16
25.05.1992-10.06.2000	8.05
16.06.1992-20.07.1999	7.10
04.10.1996-28.04.2004	7.57
04.10.1996-24.06.2005	8.73
15.05.1997-28.04.2004	6.96
15.05.1997-24.06.2005	8.12

Table 2: Time intervals used for regression statistics. Colored time periods cover the whole area of investigation.

Geomorphological predictors:

Area: The average of measured areas defined at the starting and endpoints of a time interval [m^2]. In smaller areas, terrestrialization processes react stronger to nutrient input and internal processes and are often in a more progressed state (Lüderitz 2009). Area is therefore, supposed to have a positive relationship with the rate of area loss.

Mean patch fractal dimension (MPFD): The fractal dimension is a measure of the complexity of a shape defined as a value between 1 and 2. For most simple polygons like

circles or rectangles the fractal dimension is equal to 1, the more complex its borders are the more it tends towards 2. MPFD's were determined with the extension Patch Analyst for ArcView 9.2 by averaging the calculated fractal dimensions of all patches of a subarea. The two measurements used in time interval comparisons were then averaged (Farina, 2006). The longer the shoreline relative to the area, the more a subarea was influenced by litter fall and terrestrialization processes were supposed to be stronger, even though narrow water bodies could show less primary production and lower temperatures due to shade caused by trees.

Spatial distribution or location predictors:

Distance from the inlet (Dtl): The distance reflects the path of superficial water in meters running from the inlet through the channel network of the Lobau to each subarea. It predicted the effect of backflow connection to the river. The nearer to the inlet a subarea was located, the more enhanced its terrestrialization by sedimentation or nutrient input respectively was assumed to be at higher loads of river water (Hein 2000; Reckendorfer & Hein 2000; Amoros & Bornette 2002; Reckendorfer et al. sub.). The data were generated with ArcGIS Linear Referencing Tools based on the DTM from 2006.

Vertical distance from the river Danube: This is a descriptive parameter for the proximity of each subarea to the Danube riverbed. The meters of separation reflected potential connections between subareas and the alluvial aquifer of the Danube. The closer a subarea was located to the river, the more enhanced autogenic processes in a water body by nutrient rich seepage water could be (Amoros & Bornette 2002, Hein 2000). They were calculated with the ArcGIS Near-function from the mid point of each subarea to the right shoreline of the Danube.

Hydrological predictors:

Frequency of connection (days*y⁻¹) The surface connectivity to the river Danube in each subarea was determined on the basis of the threshold connectivity based on the ground-surface-water model and the geodataset (Figure 2). The frequency of the respective discharges was computed from the data of the water gauge P Fischamend for each subarea over each time interval. Admittedly, this predictor only indicated the frequency of backflow influence of the river water for a subarea– depending on its distance to the inlet. Since thresholds for backflow at distinct discharges of the Danube were only available for main channels, this predictor was used for the sake of comparability.

Duration of water coverage in $\text{days} \cdot \text{y}^{-1}$. The discharges at which each subarea was assumed to be covered with water, were also products of the ground-surface water model. (Figure 2 below) The frequency was measured on the basis of the flood discharges measured at the water gauge P Fischamend for each time interval. Astatic water bodies can show higher terrestrialization processes, since they are often in the ultimate stage. The development of the surrounding forests is less disturbed by high water levels.

High flooding frequency of HQ2: This was determined as the average days per year within a particular time interval, when the Danube's discharge of $5850 \text{ m}^3 \text{ s}^{-1}$ (HQ2) had been exceeded. Again basis for the data assessment were the discharges, measured at the water gauge P Fischamend. Floods of the Danube exceeding the annual maximum discharge transport disproportionately more particulate matter (Nachtnebel 1998) and nutrients were therefore assumed to have a higher importance on sediment and nutrient intake (Schiemer et al. 2006) and hence terrestrialization and afforestation. According to the distribution of the frequencies over the course of time, this predictor was an indication for when certain rates of area loss occurred.

Deepening of the Danube's riverbed represents an important factor for water loss in the area and could be inferred from data collected over the time period by the regulated low water level at water gauge P Fischamend. This characteristic water level of the river expresses the duration of time exceeding 94% determined over 30 years and was published by the Danube Commission for certain years. Subsequently, the corresponding water levels with regard to deepening of the channel were interpolated for every year. The differences over each time interval were then computed (Figure 5). Since the discharges of the dataset of water gauge Fischamend were based on the riverbed of 1956, the predictors "frequency of connection", "duration of water coverage" and "flood frequency of HQ2" did not include the effects of the deepening of the Danube. Although one might question the omission of this correction, it would have been difficult to incorporate the correction into the previous predictors, since no data on the riverbed over time were available and corresponding discharges for the water levels could not be calculated.

When a change in one predictor caused a change in the relationship between another predictor and the response or vice versa, it was defined as an interaction between predictors (Allison, 1977). In the predictors chosen for this analysis, interactions were found between the "frequency of connection" and the "distance from the inlet", and between the "high flooding frequency at HQ2" and the "distance from the inlet". Since the sediment and nutrient load of flooded water was higher nearer the inlet, the frequency of connection and flood frequency at HQ2 had an increasing effect as distance from the inlet

decreased. Another interaction was found between the “duration of water coverage” and the “vertical distance to the river Danube”. Because most of temporary water bodies were filled mainly by groundwater, one might have assumed that the proximity to the Danube would have produced higher nutrient loads in these water levels and thus affected terrestrialization. A consequence here would be that water cover in astatic water bodies would have had different effects on terrestrialization, from positive to negative, depending on the proximity to the Danube.

Results

Spatial patterns in the development of wetlands – an overview

As an overview of the changes that occurred during the period studied, there is a spatial comparison of the wetlands between 1938 and 2004 in Figure 8. That was made by overlaying the digitized areas of both years to demonstrate the change. In effect it illustrates the distribution of changes and loss of wetlands that occurred over 65.5 years. The table in the figure shows the areas in million m² over time, as well as the percentages in area compared to 1938 lost in particular time intervals. A disclaimer in the table, as well as in the following analyses, is the absence of data in the area comparison with the composite picture of 1960. Subareas 3,4 and 40 were missing in this picture and hence affected calculations for the upper Lobau. When one examines the end points of this study, 1938 and 2004, it is evident that 30.8 % of the initial wetlands in the Lobau had been replaced by forest. This translates into an area of approximate 1 mio m² of the former 3.2 mio m² wetlands. Taken another way, the endpoints could represent a continuous annual decrease in wetlands of around 0.6 % over the course of the study.

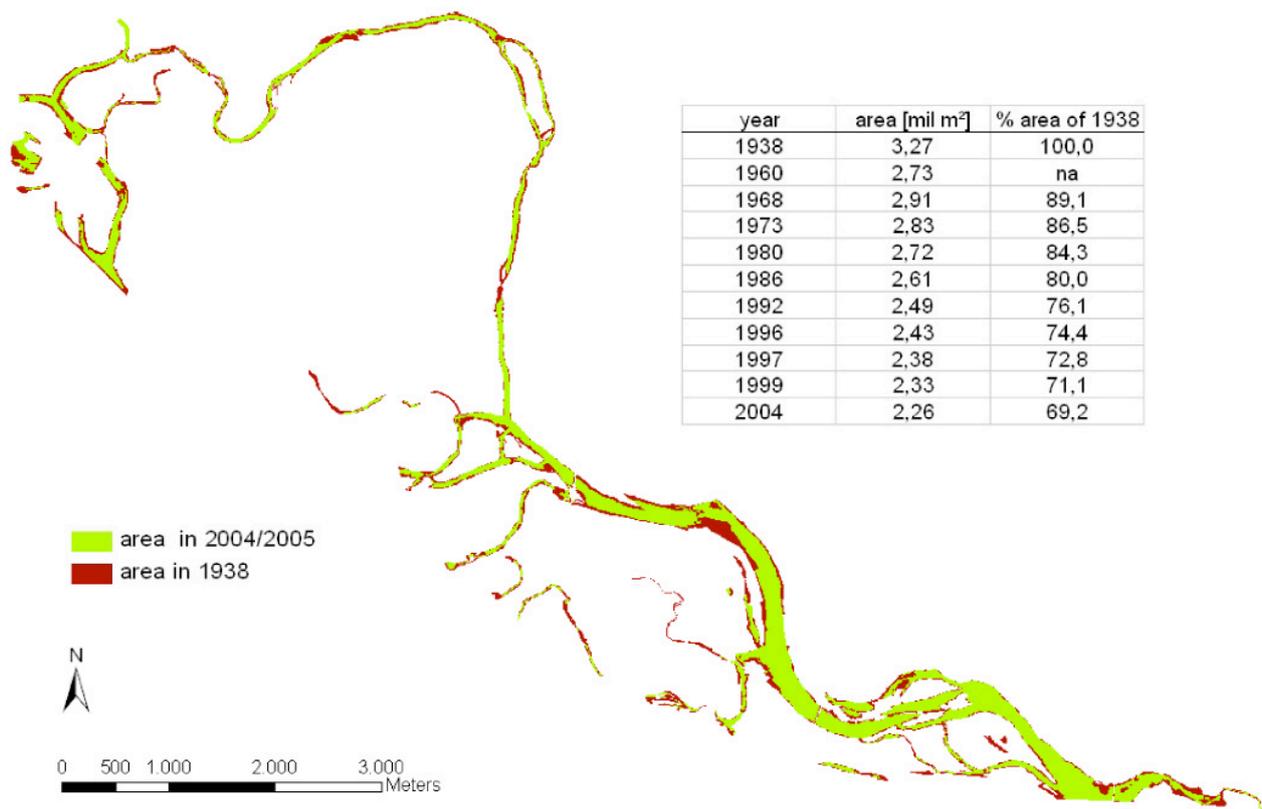


Figure 8: Wetland loss over the course of the Study: An overlay of the digitalized wetland areas in 2004 (green) and 1938 (red). A table of the areas measured and loss of area over the study has been included in the figure.

Despite these complications, one can recognize some prominent aspects of the change from wetlands to woodlands. It appears that the side channels were more exposed to afforestation than the main channels of the Lobau's water system. Some narrow side channels were even found to disappear completely.

In Sector A, the upper Lobau, there appeared to be a pronounced but balanced loss of wetlands in both main and side channels. This subdivision among channels here is not simply a reflection of size, as appears to be the case in B and C. Panozzalacke (5) and Dechantlacke (3) have been classified as side channels of A, even though they are large water bodies. The main channels of A are indeed narrow compared to main channels in B and C.

In the main channels of Sector B, close to the Danube, there appeared to be a large loss of wetlands in the centrally located subareas 16 (Mittelwasser) and 2 (Eberschüttwasser) that were separated by a ford, the Mühlleitner Furt. In general, this area had a high proportion of helophytic vegetation, consisting primarily of reed. The former slip-off slope on the right side of the shoreline appeared as wide wetlands covered with sediments in

1938. Here the highest loss of area took place. According to the photographic material, it had become a pioneer forest already in 1960. It was classified as a *Salix purpurea* forest in 1997 (Burger & Doganbacher, 1999), and has remained so until today.

The main channels of Sector C, the subareas 34 and 1 are adjacent to the Danube inlet, before the first weir. These channels showed the highest loss of wetlands for this sector. In this case, the boundary of the forest narrowed successively over time. To this there were large losses to areas with sediment or supposed reed development between 1938 and 1960 and, 1973 and 1980.

The measure of area loss in percentages that was used during the whole study was affected or even perhaps dependent on the absolute sizes of the wetlands. A small absolute loss of wetlands in a small area could have been overestimated compared to the relatively higher losses of bigger areas. This can be exemplified in a comparison of main channels in the three sectors over the study. In Sector A these channels are narrow. In order to assign numerical values to the changes, transects of main channels in each sector for the highest and lowest area loss in Figure 8 were made. From them, the progression of woodland from the side was measured and is listed in Table 3.

For Sector A main channels, the measured progression of woodland between 1938 and 2004 was between 3 cm per year in subarea 20, and 72 cm per year in subarea 6. Conspicuous time intervals with extreme woodland progression were not found in the study. For the main channels in Sector B, the most extreme progression was observed in subarea 2 between 1938 and 1960, where it amounted to 5.16 m per year. Afterwards, a similar progression of 70 cm per year as in Sector A was found. Other transects in this sector showed no progression of woodlands. As in B, Sector C also showed widely distributed progression rates of woodland over time and space. The highest rate of 3.68 m per year was found between 1938 and 1960 in subarea 34. To illustrate the heterogeneity in this subarea, another adjacent transect had a maximum progression of 2.25 m per year between 1960 and 1980, while between 1938 and 1960 its progression was only 97 cm per year (not shown). After peaks in Sector C's main channels, the progressions were reduced to levels lower than those in Sector A, about 0.39 and even 0.00 for the adjacent transect. The lowest progression of woodlands over the study period was found in subarea 10, 2 cm per year. All in all, when one examines woodland progressions, one can state that Sectors B and C showed definitively higher maximum rates. However, except for specific high periods in the hot spots of B and C, in Sector A woodland showed homogeneously similar progression.

	Subarea	width [m]			woodland progression rate [m y ⁻¹]		
		1938	1960	2004	1938 - 1960	1960 - 2004	1938 - 2004
A min	20	49.7		46.7			0.05
A max	6	97.4		34.23			0.96
B min	50	199.2		199.4			0.00
B max	2	241.8	128.7	82.4	5.19	1.06	2.43
B mid	16	163.6	127.2	120.0	1.67	0.23	0.71
C min	10	124.5		123.0			0.02
C max	34	124.6	44.7	27.68	3.68	0.39	1.48

Table 3: Changes in the width of wetlands for main channels over the study – The woodland progression rate. For transects in the main channels of each sector the widths of wetlands are shown for the years 1938, 1960 and 2004. For the whole time of investigation as well as the major time of change, rates were calculated [m y⁻¹]. To indicate the range of each Sector, the transect represent the minimum (min) and maximum (max) and if necessary a "common" transect (mid). The subareas containing these transects are also shown.

Observations of water bodies

Due to the high variability in the water expanse data explained in the methods, an illustrative overview of the change in water bodies over the time of investigation could not be made. But there were some relevant observations that were seen in the data comparisons during classification. One was that parallel to the wetland loss in the subareas 34 and 1 of Sector C, next to the Danube inlet. Here, there was a high loss in water expanse. Aggradation or dewatering of the wide channel after 1938 was paralleled by afforestation. By 2004 there was only a narrow channel remaining.

Another conspicuous point in the comparisons of water bodies was that in Sector A, landscape management occurred between 1938 and 1960. The former channels in the subareas of Groß-Enzersdorfer Arm and Eberschüttwasser (41, 39, 27 and 38) had been meadows, loose woods with only small remnants of water in 1938. On the other hand, when following the development from 1960 to present, these areas changed back into water-covered channels, probably as a result of human intervention. Since wetlands also showed an increase in area, at that time one could assume that woodlands were cut. As a consequence, data from these subareas between 1938 and 1960 was omitted from the analyses. The Dechantlacke (subarea 3) as water body did not even exist in 1938 and was dug out before 1960. Similarly in Sector C the lake of subarea 52 was built before 1980. Dredging measures in these areas were documented by members of the Nationalpark

Donau Auen. Accordingly, in the subareas of Groß-Enzersdorfer Arm and Eberschüttwasser, the time of dredging was estimated to have been in the 60s and 70s. Another aspect was, that in 1938 water expanses of the main channels were visually smaller than later photographs perhaps due to a higher proportion of helophytic vegetation. In later years mainly hydrophytic vegetation was detected at these locations. Since misinterpretation of the images was probable, the data from 1938 were omitted in the analyses of water expanse.

Insufficient correction of water expanses by water level

In spite of the problems mentioned above, an attempt to examine the development of water bodies over time correcting for the water tables was deemed necessary to examine terrestrialization processes, including dewatering of the area. Measured water expanses were therefore recalculated to a value corresponding to an equal water table. This was supposed to reduce variation and to investigate possible relationships between the rate of water loss and the assumed explanatory predictors in further statistical analysis.

The data in Figure 9 are exemplary containing the results from the test zone for main channels around P17. In the upper graph, the theoretical relationship between main channel water expanse and water table levels has been plotted. In addition, the digitized water areas of the images have been plotted against the corresponding water level. With this data one can compare the expected water expanse with the measured expanse. A linear relationship in the modeled changes of expanse versus water table ($n = 12$, $p < 0.001$) was calculated. The measured water expanses, however, were highly variable, although they did tend to increase at higher water tables. At low water tables the measured values fit the model better. The second part of the analysis was to determine whether the deviation from the expected water expanse was reduced and a trend in decreasing water expanses over time could be measured. This would have been a strong argument for terrestrialization and/or water loss in the area and would have allowed better use of the water body data in the statistical analyses. In the lower graph the differences between measured and expected expanses have been plotted according to calendar time. Unfortunately there were no clear effects of sampling date on these differences. Hence the water expanse data could not be improved by simply correcting for water tables with a model.

The main reasons for these problems may have been the slip-off slopes of the main channel shorelines that were often overgrown with helophytic vegetation. For undercut slopes where visible vegetation was absent the ground surface may have provided

accurate estimations of water expanse. The fraction of bare to vegetated shoreline should probably have been considered.

The results from the side channel analyses around P16 produced similar results and have therefore not been shown. The estimated changes in water expanse were, however, not linear for the side channels. The models produced an exponential pattern. The reason for this was probably the difference in channel structure between the two groups, since the relatively shallow embankment of side channels changed the linear interaction.

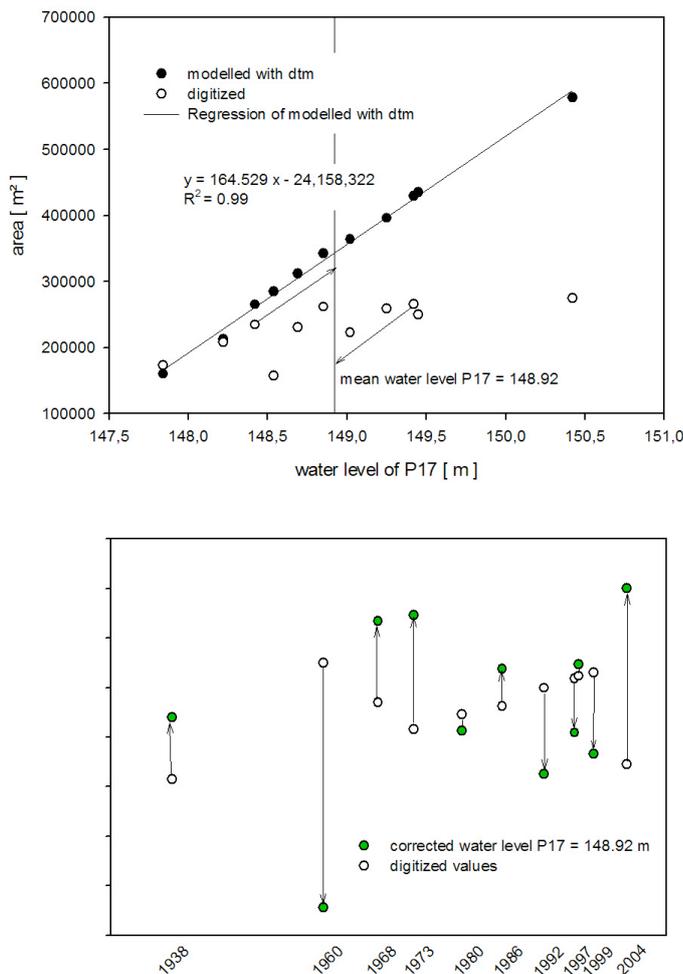


Figure 9: A comparison of theoretical and measured water area in the test zone for main channels at different water levels (above) and the corrected and measured water expanses over the course of the study (below). Theoretical water expanses (full circles) were calculated for each water level of the representing water gauge P17 based on the DTM. The measured water expanses (empty circles) were then corrected by recalculating to the mean water level by means of the regression formula of the modeled values. The corrected water expanses (green circles) were plotted with the measured ones over time. Neither a decreasing pattern indicating terrestrialization, nor an improvement in variability was found.

Development of wetlands and water bodies in the three sectors

A first step in the analysis was the quantitative documentation of wetland and water body changes in the three sectors over sequential time points. To do this, the values in a given dataset were converted to a percentage of area present in 1938, and alternatively a percentage of the 1968 area for Sector A. The initial areas were defined as 100 %. The data for wetlands and water bodies are shown in Figure 10a and b. For the wetlands, one can see that there was a continuous decrease in size in all sectors. (Figure 10a) Sector A

had the most pronounced loss with only 67.3 % of the area remaining in 2004. Overall, a median loss of wetland of -3.60 % per corresponding time interval for that sector was found.

The area of Sectors B and C decreased in a similar, but less pronounced manner than A. There was nonetheless an obvious difference between the two sectors. In contrast to C, the middle Lobau (Sector B) exhibited a higher loss of wetland area of 15.3 % between 1938 and 1960. With the ensuing loss, the wetland here was reduced to 67.6 % by losing a median area of 2.21 % per time interval after 1960. For Sector C the loss was less, being reduced to 76.7 % of its area in 1938, but with a similar median area loss of 1.98 % per time interval. Between 1938 and 1960 Sector C had a loss of 5.8 %.

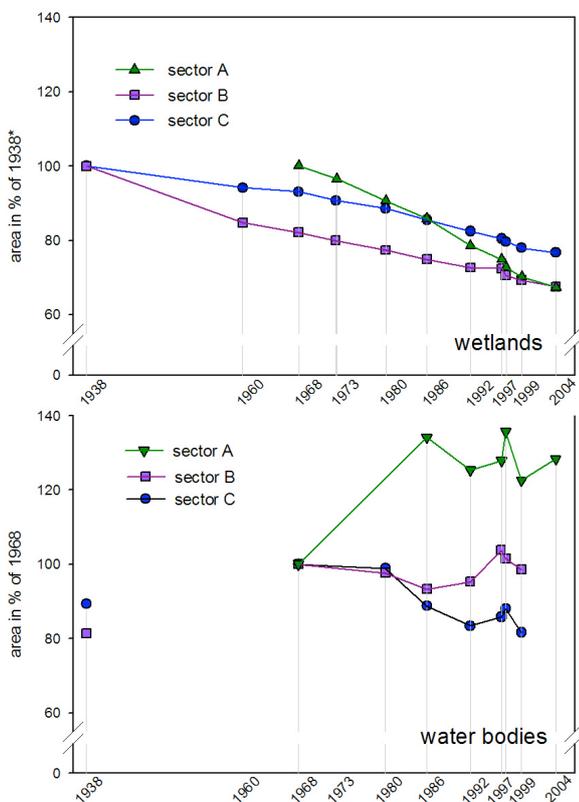


Figure 10: The patterns of changes in wetland and water body areas over the course of the study. The areas are expressed as percentages relative to the earliest representative time point. For water bodies, doubtful data due to extreme water levels or low picture quality were omitted. The dominance of wetland loss versus loss of water bodies increases from Sector C to A with the degree of disconnection.

With respect to changes in water body sizes, the patterns appeared to be different, see Figure 10b. As might have been expected on the basis of water table differences and problems explained above, more variable patterns were documented for this parameter. Still general trends were recognizable. In contrast to the loss of wetlands in Sector A, an increase of around 28.3 % water expanse in 2004 relative to 1968 was found. This was primarily due to a steep increase between 1968 and 1986 of 34.2 %. The water expanses in Sector B remained relatively constant after 1968. Finally, in Sector C there was a decrease in water body area of about 15 %. Here the loss was most pronounced between

1980 and 1992, when it amounted to a 15.5 % loss of wetlands. Outside of that time period, the pattern in C looked very similar to B. In all sectors there was a conspicuous increase in water expanse between 1992 and 1997 and a subsequent decrease by 1999. The pattern could not be directly related to changes or differences in the water tables available from the 4 water gauges. For sake of illustration, the water body data from 1938 have been included in the figure for Sector B and C and expressed as the percentage of the 1968 data. It is interesting to note that the relative area sizes in 1938 (C>B) were the reverse of those after 1960 (B>C).

Further analyses of wetlands with subdivisions of main and side channels

In order to get a more detailed picture of how afforestation occurred in the sectors, the data from Figure 10 have been reanalyzed using the hydrological and geomorphological characteristics of the water bodies as grouping parameters. As described in the methods, a differentiation was made between main and side channels in each sector. The changes in wetland area within these two groups of each sector are shown in Figure 11. As in the previous figure, percentages were calculated relative to areas found in 1938 and alternatively 1968 for sector A.

As seen in the previous figure, there was a steady decrease in area in each of the three sectors for both main and for side channels. In all cases these changes were significant over time ($n = 11$ (9 for sector A), $p < 0.001$). It is striking here, that the main and side channels of Sector A showed a similar pattern of change. In main channels 32.2 % of wetlands were replaced by forest between 1968 and 2004, in side channels 33.6 %. Their median loss in wetlands per corresponding time interval amounted to 3.3 % in main, and 3.8 % in side channels. In contrast, the side channels of sectors B and C had more pronounced afforestation than the main channels. In accordance to Figure 10 the wetland's loss in Sector B was greater than in C, a 12.6 % loss difference for the side channels and 8.4 % for the main channels compared to C.

In Sector B the wetland loss from 1938 to 1960 was also present but not specific for either channel type. Both channel types lost around 15 % of area over that time interval. In the following intervals the main channels decreased much slower with a median area loss of 1.23 % per interval until 2004, when 25.1 % had been lost compared to 1938. Conversely, the side channel area decreased after 1960 with a median loss of 3.95 % per interval until 2004, when 49.3 % of area had been replaced by forest. As mention earlier, there was an abrupt loss of wetlands between 1996 and 1999. This was found in each sector. In A it was

found in both main and side channels. In Sectors B and C this phenomenon was only observed in the side channels. In Sector C about 16.7 % of the wetlands in main channels and 36.6 % in side were lost to forest by 2004. The main channel's area decreased with a median of 1.55 %, the side channels with 2.95 % per corresponding time interval.

Statistical comparisons of the differences in area losses between main and side channels in each sector showed, that the losses in main channels of B and C were significantly lower than that in the side channels (Wilcox signed rank test, $n=10$, $p=0.001$ each).

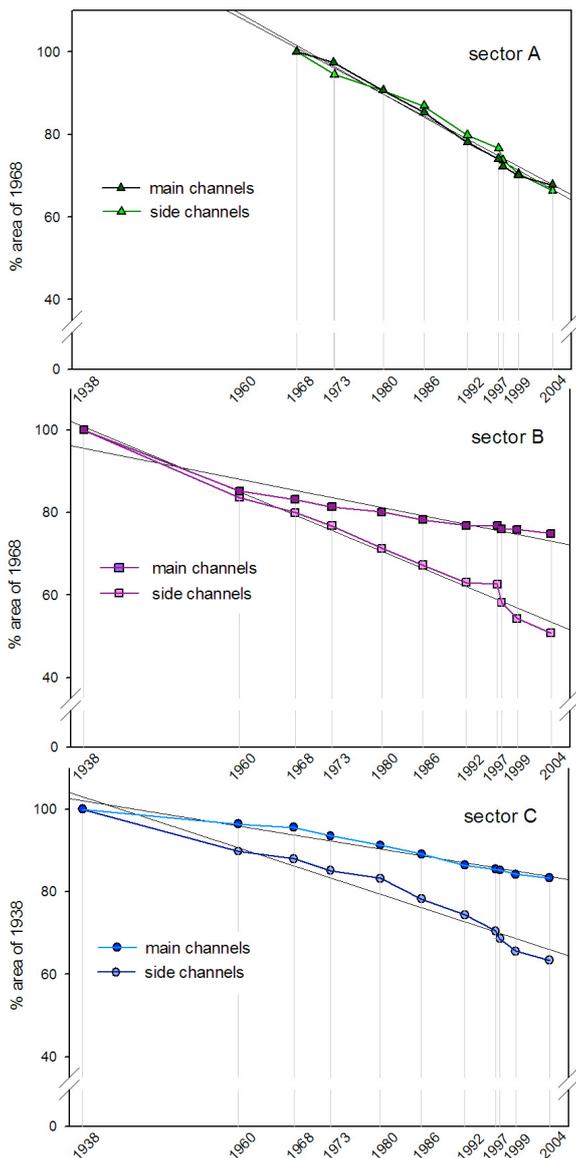


Figure 11: The patterns of wetland loss over the time of the study in main and side channels of each sector. Areas are expressed as percentage relative to 1938 or 1968 for sector A. All channels types show a significant loss over time ($p < 0.05$). In B and C side channels show significant higher losses than main channels. ($p < 0.05$, $n=11$, one sided signed Wilcoxon rank sum test)

A problem with the analysis to this point was that the time intervals between composite pictures for analysis were quite variable, ranging from 0.6 (1996 – 1997) to 21.8 (1938 – 1960) years. This had two effects. One was that methodological errors from digitizing were exaggerated in short time intervals, when no large changes in woodland could have been assumed. The other was that the rate of change per year could not be accurately estimated from the analysis, since short time periods in the denominator of the exponent

often produced numeric overestimation. (See methods) To rectify this situation and to quantify the loss of wetlands as percentage in change per year, the proximate plus one sample was chosen as the time interval for rate change analyses. An exception was the longest interval between 1938 and 1960, which stayed unchanged. This reduced the range to 7.0 - 21.8 years. The resulting rates of wetland loss per year between these sampling points are shown in Figure 12.

In Sector A the principal trends in both main and side channels were that the rate of wetland loss per year increased over time i.e. there was an increase in velocity of afforestation. The median losses were $-1.14\%y^{-1}$ for main and $1.13\%y^{-1}$ for side channels. The maximal rate of $-1.72\%y^{-1}$ was found between 1992 and 1999. Differences between the two subdivisions of channels were small, with a median of $0.07\%y^{-1}$.

In contrast to A, Sectors B and C showed increases in the rate of area loss for side channels, while the rates of wetland loss in the main channels remained relatively constant. In Sector B, it appeared that the rate in area loss even lessened slightly with time, i.e. the velocity of afforestation decreased. The median rate change for main channels was $-0.31\%y^{-1}$ and decreased between 1938 and 2004 from $-0.73\%y^{-1}$ to $-0.20\%y^{-1}$. For side channels there was a trend over the study with a median rate decrease of $-0.99\%y^{-1}$. The decrease was mainly caused by the more extreme rates between 1992 and 2004 of -1.9 and $-2.1\%y^{-1}$, which translated into an absolute area of more than 4500 and 5300 $m^2 y^{-1}$ respectively. Comparing the side channel data in Figure 12 with the previous figure, it appears that the loss of wetlands documented in Figure 11, between 1996 and 1999 caused the unusually low rates of change between 1986 and 1996 that are plotted in Figure 12.

In Sector C, side channels showed a similar change in the rates of area loss per year as in B, but were consistently smaller. Nonetheless, the values showed a significant decrease over the study ($p = 0.025$, $n = 8$, $R^2 = 0.6$). The median rate change here was $-0.80\%y^{-1}$. In contrast to the other sectors, there was almost no change or variability in the rates found for the main channels of Sector C. However, after 1968 the rates were about 0.2% higher, than for main channels of Sector B. As in the other sectors the fastest area loss was observed between 1992 and 1999, where the rate was $-1.75\%y^{-1}$ area loss per year.

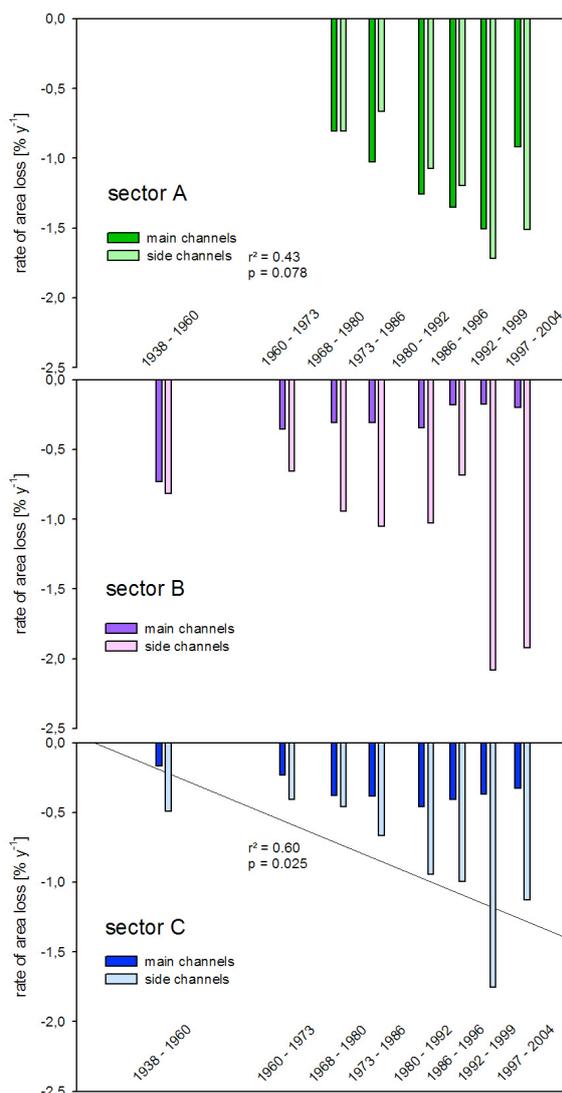


Figure 12: The rate of wetland loss (% change per year) in each sector for main and side channels and time interval of the study. Sector A and main and side channels of Sector B and C show an accelerating afforestation over time.

For the sake of completeness, it is appropriate to examine the afforestation patterns of individual subareas within each sector of the Lobau. These data are shown in Figure 13. Again, the parameter for change was the percentile wetland loss relative to 1938 (1968 in Sector A) for each subarea and time interval. Noting that the solid lines are main channels and the dashed side channels, it is conspicuous that the side channel decreases were larger and more variable. In accordance with Figures 9-11, one may also note that main channels showed smaller amounts of wetland loss. The conformity in the pattern of loss between main and side channels of Sector A, that was discussed earlier, can also be seen here. There were still “outliers” in A (subareas 26, 4, 25) that had higher area losses. In Sector A these were small and narrow side channels, similar to many of the side channels of B and C (compare Figure 8). All other subareas of A had similar gradients in decline, albeit these slopes were generally steeper than main channels of Sectors B and C. Exceptional for main channels in Sector A was the pronounced loss of wetlands in four

subareas of the Groß-Enzersdorfer Arm (27, 38, 39 and 41) between 1980 and 1997. These areas were almost completely terrestrialized in 1938. During the ensuing decades measures were undertaken to re-establish their structure. (See description of Figure 1)

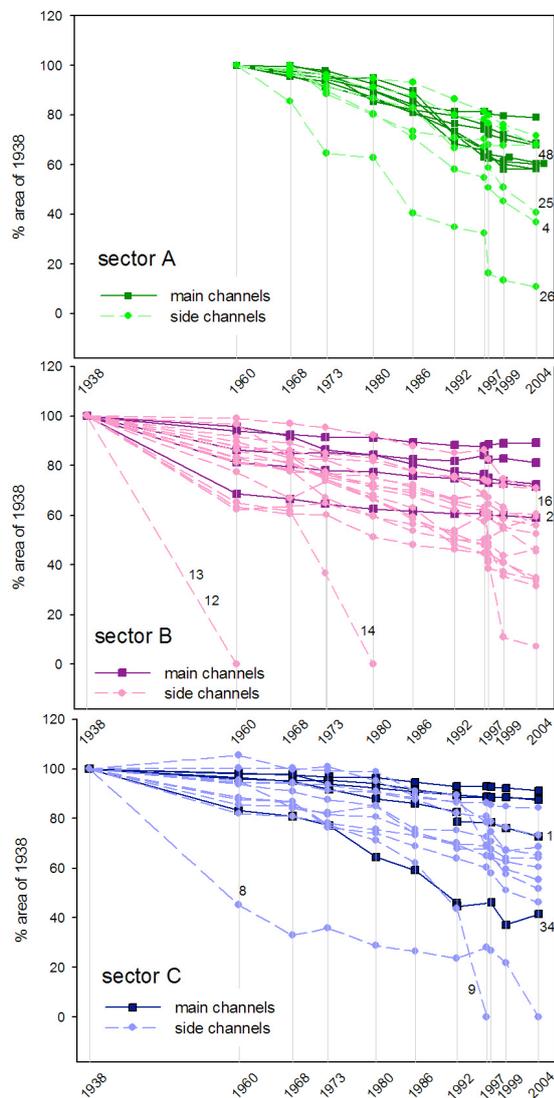


Figure 13: The patterns of wetland loss in individual subareas (% change relative to 1938) over the course of the study. Main channels (solid lines, dark) show lower variability compared to side channels (dashed lines, light).

In Sector B side channels principally showed the largest ranges of wetland loss. Here, four water bodies were completely eliminated by forest, whereas subarea 12 – 14 were all part of a former side arm south of Mittelwasser, that was extremely narrow in 1938. With regard to main channels of Sector B the differences found in the patterns resulted mainly from specific area losses between 1938 and 1960. Actually subarea 2 and to a certain extent subarea 16 showed the highest overall losses, but almost no losses after the initial decrease found in the first time interval. (compare Figure 11) In contrast, subarea 44 showed the most continuous decrease of wetland loss found in the main channels of this sector.

Similar differences between side and main channels were found in Sector C with some exceptions. For instance, the subareas 8 and 9, seemed to have been completely replaced by forest at the end of the study period. These were the smallest subareas in this sector. Among the main channels of Sector C, the greatest loss of wetlands occurred in subareas 34 and 1. They were the subareas most frequently connected to the Danube River because they are located next to the inlet. The loss in area occurred over the whole study period. Subarea 34 was somewhat unusual because of its narrowness. It actually had more characteristics of a side than a main channel, but since it showed even higher area losses than other side channels it would be outstanding even as such.

Analyses of Water bodies

After the analyses of wetlands it was important to examine whether the same patterns were recognizable in the changes of water expanse within the Lobau. As previously seen in Figure 10 and the accompanying exposé the overall patterns of water expanse appeared to differ among the sectors. For this reason the water expanses have been plotted as the percentile change in measured water expanse relative to 1968 in the three sectors and the two groups of subareas. The data are shown in Figure 14. As was earlier the case in this report, the water expanse areas for 1938 have been included but expressed this time as a percentage of the 1968 area. Again, the differences in water expanse could have been caused by differences in water levels at the time of the photography. The measured water levels for each sector on the dates of sampling were therefore compared with the expanses. In the figure, these are shown as bars on the bottom of each graph. The right ordinate defines their measures in meter above Adria. For Sector B, the gauges at P17 and P16 were considered to collectively estimate the water levels of Sector B. Incomparable composite pictures because of either extreme high or low water levels, or extremely low quality of the image were excluded. Finally, the median of the rate of annual area loss as a measure of the velocity of change, has been written in bold letters for main and side channels of each sector. As for wetlands they were calculated for the sake of robustness from one sample year to the one after the next.

For all sectors it appears that the changes in the area of expanse did not reflect the variation found in the measured water table. There was no obvious pattern. Taking a look at the side channel data, striking examples can be found between 1980 and 1986 in Sectors B and C. Here there were abrupt losses of more than 15 % in water expanse without corresponding changes in the water levels. With regard to the main channels in

Sectors B and C, the constant developments in water expanses did not follow the fluctuating water levels of the water gauges P17 and P Fischamend.

There were also other problems in associating water levels to water expanse. During and after floods, there were more extreme water levels in individual basins. Hence the high water levels of the Danube in 1960, were reflected in the water expanse data of the Lobau. Connectivity of the main channels with the inlet reached the subarea 44. In 1980 another extreme water level was measured in P11. For the composite pictures of 2004 the water gauges 16, 17 and Fischamend showed extremely low values. Although interesting in their own sense, these extreme data sets did not contribute much to the analyses undertaken here. The years were therefore omitted from the water expanse analysis.

Overall it does appear that the negative change in water expanse was more pronounced in the side channels than in the main channels. However, in Sectors B and C this phenomenon seems to have been based on the high losses of 37.0 and 36.4 % area between 1980 and 1986. The hydrograph of P17 in Sector B and P Fischamend for Sector C showed that there was little variation in the water levels. Interestingly, when the patterns of change were examined in detail, it was found that the water levels for P17 (in B but next to C) remained relatively constant between 1982 and 1988 at around 148 m. a. A., with little fluctuation compared to time intervals before or afterwards. Water gauge P16 showed a period of lower fluctuations between 1978 and 1990. However, for P Fischamend no conspicuous low mean water levels or rare fluctuations were found. (Appendix 1)

In Sector A, the data for main channels demonstrated an overall water expanse increase after 1968. Over the study period there was a positive rate in area increase of 1.40 % per annum. The pattern of area change over time was very similar between total water bodies in the sector shown in Figure 10 and the main channels depicted in Figure 14. It may have been a simple product of mass law in that the main channels represented 70.6 % of the total of water bodies here. As mentioned earlier, the major increase in water expanse occurred between 1968 and 1986. Chronicles of the Nationalpark Donau Auen have also documented dredging activity in the Groß-Enzersdorfer Arm (subareas 27, 38, 39, 41) that took place in the 1960's and 70's. In contrast, the side channels that included the two large subareas Dechant- (3) and Panozzalacke (5), showed a more or less constant decrease in area with a median water expanse loss rate of $-1.53 \% y^{-1}$, even though repeated dredging in Dechantlacke have also been confirmed from members of the Alluvial Zone National Park.

In Sector B it was evident that absence of change in water bodies described in Figure 10 was a product of the patterns in the main channels. Here the dominance of main channels in the water expanse (70.4 % of the total) as in A was the probable cause. Outside of a marginal decrease in expanse between 1968 and 1980, there was a marginal but steady increase in main channel area over the rest of the study. The median rate of change was $+0.57 \text{ \%y}^{-1}$. The pattern of change in the side channels differed to a certain extent. Overall there was a negative trend with a median rate of -1.10 \%y^{-1} and the abrupt loss between 1980 and 1986 as mentioned above. An exception to this was the positive development between 1992 and 1996. Here, the water tables data did not provide a basis for understanding changes in water expanse, since P16 did not show any pronounced differences between the dates and P17 levels varied independently from water expanse changes.

The data in Sector B from 1938 are of interest. At that time side channels were almost 56.2 % larger in water expanse than in 1968, 30 years later. Conversely, the expanses of main channels were 31.4 % less than 1968. As mentioned above in the 1938 composite picture there were difficulties in defining vegetation types especially in main channels and consequently uncertainties in the boundaries between water bodies and wetlands. Underestimations of the main channel's open water expanse may then have been expected. Moreover there were a few side arms in subareas 15, 17, 18 and the upper end of 19 and 28 present in 1938 that had disappeared by 1968.

Sector C exhibited patterns in main and the side channels similar to those in B. One pronounced difference was the fact that the pattern of change in main channel expanses was negative, compared to Sector B and especially to A. In the images of 1999 just 84.8 % of the water expanse existed. The median rate of loss was -0.96 \%y^{-1} . Similar to the other sectors, main channels composed 68.4 % of the total area. The pattern of change in the side channels was similar to that found in Sector B although less pronounced. Except for the abrupt decrease between 1980 and 1986, no trend was found. With regard to 1938, differences were not as high as reported for Sector B. The data are nonetheless of interest. The water level data showed that the Danube was extremely low at this time. The fact that the side channels were 3.6 % smaller than in 1968, may therefore have been expected. However main channels showed marginally higher values of 85.2 % compared to 1968, in contradiction to the low water level.

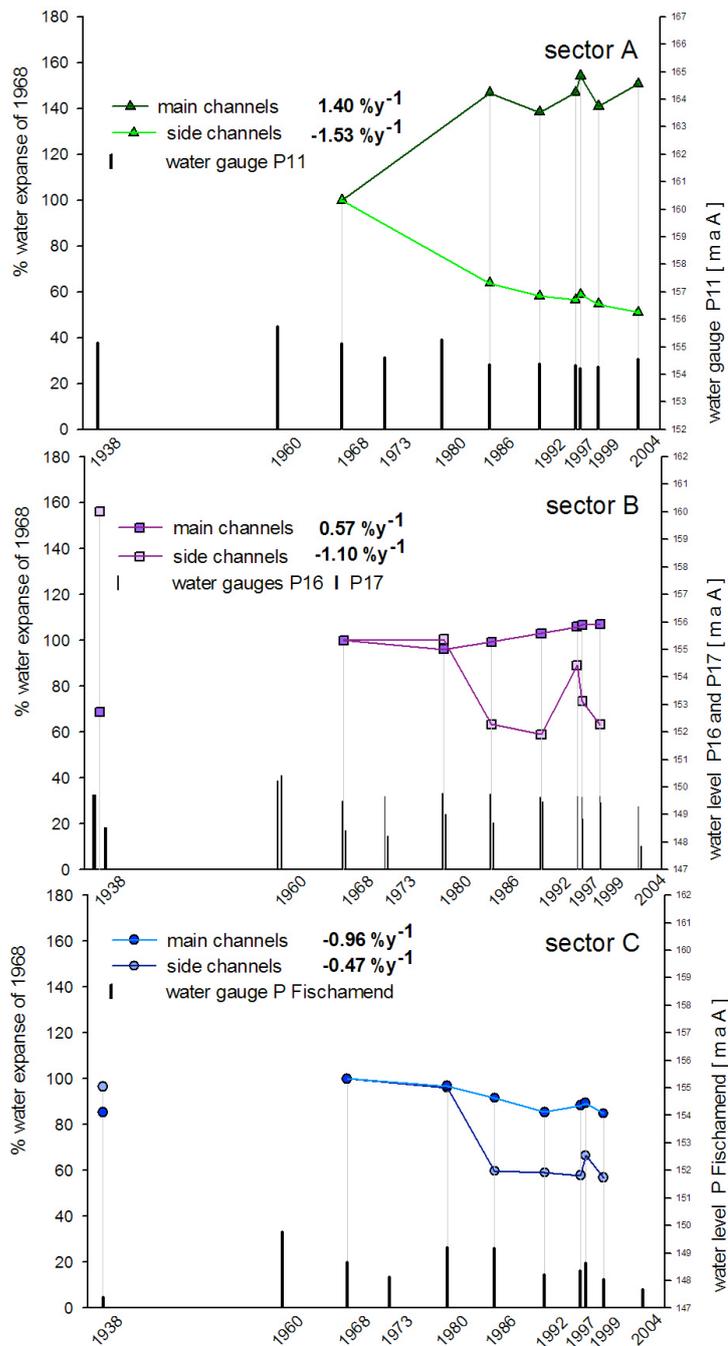


Figure 14: The patterns of water body areas and water levels (at the corresponding gauges) in sectors and channel types over the course of the study. The areas are expressed as percentage of 1968. The values for 1938 were kept for comparison. To express the degree of loss the median rates of annual loss [$\% y^{-1}$] was calculated from one data point to the next but one. Doubtful data caused by extreme water levels or low picture quality were omitted.

Multiple Linear Regression Analyses of Wetland Loss

Multiple regression analyses for main and side channels in each sector were performed with the rate of area loss per year as the response variable for each subarea. The goal was to find out, whether the developments in area loss of wetlands reported were related to factors, commonly assumed to be associated with terrestrialization and water loss and in the long run with afforestation of the floodplains. Table 2 contains the resulting estimations of contributions of those predictors that were found to have significant effects in the model. Relevant parameters in the analysis were the regression coefficient b , mainly expressing the direction of partial correlation with the response and the partial coefficient of determination, which demonstrated the percentage of explained variance by each predictor, when others were held constant.

To compare the fit of the models in each subdivision visually, the percentages of explained variance have been depicted as the adjusted coefficient of determination in Table 4. In this way, the percentages have been adjusted or corrected for differences in degrees of freedom between analyses. As an overview for accuracy of the models, in Figure 15 the observed response values have been plotted against the modeled ones. A 100% correlation with these values, expressed as line serves as a basis for comparison. Each model shown in the figure was highly significant ($p < 0.001$) and contains an intercept. These have not been drawn for the sake of clearness.

In general, it is hard to draw conclusions about differences in relationships among the predictors and the rates of area loss per year, since no unified pattern was found across the subdivisions. The direction of the partial correlation was also often not in accordance with the theoretical expectation of the predictor. For this reason each subdivision is described separately with two possible aspects of the results. The first is the specific interaction with afforestation in that area. The second is possible confounding factors or simple methodological error. The descriptions here have also been limited to the spatial and temporal distribution of the highest and lowest rates of area loss per year.

Nonetheless, when one examines the figure, it is clear, as had been seen in Figures 11, 12 and 13, that side channels showed higher rates of area loss, and lower negative values. This was true for all sectors. The size of the area and the deepening of the Danube, as well as to a certain extent water-body-shape were stable in direction and important regressors for the side channels. Accordingly, one could conclude, that side channels showed the most negative rates of area loss in small and complex shaped subareas as a response to the riverbed incision of the Danube. In main channels only the predictor "high flooding frequency" could be extracted as a similarly important and stable

factor in Sector A and C, showing that afforestation occurred primarily during time intervals with fewer high floods in more recent times. Besides these interactions, no other pronounced patterns were found in the main channels. In spite of this, in Sector B and C the individual predictors chosen were found to explain a higher degree of variation. In side channels their models explained less variation and more predictors were significant and therefore included in the final models. Additionally, the models did not reliably estimate the most negative rates of area loss, as can be seen in Figure 15. In comparison Sector A showed a contradictory pattern between the model fit of main and side channels. Here predictors for main channels explained less variability than side channels, and the model did not estimate the highest area losses reliably. Finally none of the predictors showed the expected direction in relationship with the response. One might conclude, that the afforestation processes of the main channels of Sector A were strongly affected by other factors that were not considered in this analysis.

Sector A main channels

For Sector A, it was possible to construct a model that explained 48% of the variance in terrestrialization ($p < 0.001$, $n = 62$, adjusted $R^2 = 0.48$). This was actually the worst model result among the analyses of main channels, which also produced underestimations for the highest rates of area loss (the most negative). The relationships among predictors and responses always went in the unexpected direction. This is an indication that the spatial distribution may have had more of an effect than the chosen predictors. Thus, other, explanatory variables would have to be brought into the model to improve its predictive value. The flood frequency at HQ2 explained 39 % of the variance, when other parameters were held constant. Finally the vertical distance from the Danube and lateral to the inlet had significant effects on the model, respectively accounting for 14 and 10% of the variance. Since all the subareas had been assumed to be water-covered by the same level of discharge from the Danube, the interaction of water cover with the distance from the inlet was dropped at the onset of the analysis. The result of this logical step reduced variance explanation by 20% and was perhaps one reason for the underestimations in the model.

The exclusion of the other remaining predictors had little effect on the model in that it caused a loss in fit of less than 1%. Nonetheless, there were some interesting trends. For instance, the model expressed more negative rates of area loss in years with less flooding, and in subareas farer away from the Danube and the inlet. In particular, the subareas in Groß-Enzersdorfer Arm (39, 27, 38, 41), located far away from the Danube, showed the

most negative rates between 1980 and 1999. These time intervals were characterized by very low flood frequencies (see Figure 4) and, the changes here were actually underestimated by the model. Oddly enough, these subareas showed an increase in area between 1938 and 1960. Here one might assume either a methodological cause in preprocessing of the pictures of 1938 or compounding anthropogenic effects associated with forest management, particularly since the water bodies of these subareas were excavated before 1960. This data was therefore excluded from the statistical analysis for that time range. In contrast, subarea 20 of Sector A was closest to the Danube and to the inlet. Between 1938 and 1960 it had less negative rates of area loss than in the more recent intervals.

Sector A side channels

The rate of area loss for the side channels in Sector A was best explained by the predictors, deepening of the Danube riverbed, water coverage and area that respectively accounted for 28, 22 and 15% of the variance. Besides these, shape also contributed a 7% effect, a trend without the defined statistical significance ($p = 0.053$). In contrast to the main channels, all predictors showed the expected relationships with the rate of area loss. In total 61% of the variation of the data could be explained ($p < 0.001$, $n = 54$, adjusted $R^2 = 0.61$). Excluding the other predictors produced a loss in fit of 3%.

In general all subareas of Sector A were supposed to have been superficially connected to the Danube at HQ 100 (Figure 2 above). In fact, this water level did not occur during the time period studied here. Instead, a groundwater connection can be assumed. Due to the location close to the river, smaller subareas with more complex shapes and more astatic water bodies were probably more susceptible to area loss resulting from riverbed deepening. As already mentioned, the deepening of the riverbed was more pronounced in later time intervals of the study. (Figure 3) These kinds of interactions were clearest in subareas 26, 25, 4 and to a certain extent 48. They were the smallest wetlands that correspondingly showed the most negative rates of area loss, between 1986 and 2004. With regard to water coverage, all water bodies of this sector were estimated to be covered with water at a discharge of the Danube of $910 \text{ m}^3\text{s}^{-1}$, except for subarea 26 that had coverage at $2200 \text{ m}^3\text{s}^{-1}$. Particularly this subarea showed the most negative rates. (Figure 3 below) Less negative rates were found in the subareas 3 and 5 over the whole time period and subarea 48 between 1938 and 1973.

			Area	MPFD	DistInlet	DistDb	fConnect	wCover	fFlood	DeepDb	fConnect* DistInlet	fFlood* DistInlet	wCover* DistD
	n	adjR ²	b*10 ⁻⁵ p R ²	b*10 ⁻¹ p R ²	b*10 ⁻⁵ p R ²	b*10 ⁻⁴ p R ²	b*10 ⁻² p R ²	b*10 ⁻³ p R ²	b*10 ⁻¹ p R ²	b p R ²	b*10 ⁻⁵ p R ²	b*10 ⁻⁴ p R ²	B p R ²
A main	62	0.48			-2.56	-1,07			4.16				
					0.10	0.14			0.39				
A side	54	0.61	0.256	-9.54				7.50		30.92			
			0.15	0.07				0.22		0.28			
B main	47	0.56			-32.1		-8.64				1.05		
					0.26		0.14				0.10		
B side	144	0.39	1.11	-6.71	6.94	2.05	9.73		-2.06	26.68	-1.04		
			0.14	0.03	0.08	0.04	0.05		0.03	0.10	0.05		
C main*	50	0.84	3.01*10 ⁻¹		5.09				1.35				
			0.65		0.19				0.16				
C side	98	0.44	1.11					-2.91	-4.67	19.29		1.22	
			0.29					0.06	0.06	0.06		0.07	

Table 4: Overview of Model Outcomes: For each model the number of cases **n** and the adjusted coefficient of determination **adjR²** was plotted. Furthermore for each significant predictor as well as the interaction terms in the last three columns the regression coefficient **b** and the partial coefficient of determination **pR²** are also shown. All models were highly significant ($p < 0.001$), the predictors significant ($p < 0.05$). The predictors for each subarea in each time interval: **Area**: mean area, **MPFD**: mean patch fractal dimension expressing the shape of each subarea, **DistInlet**: Distance to the inlet, **fConnect**: Frequency of Connection, **wcover**: water coverage, **fFlood**: High flooding frequency, **DeepDb**: Deepening of the Danube. * In the main channels of Sector C the predictor values for “Area” had to be logarithmized.

Sector B main channels

Main channels of Sector B had the lowest rates of area losses per year. It was mentioned above that this may have been related to the fact that these subareas were large in comparison to those of Sectors A and C. The rates were best explained by the predictor's distance from inlet and the frequency of connection. Their interaction was also important. They contributed respectively 26, 14 and 10% to the explanation of variance. The whole model explained 56% of the variance ($p < 0.001$, $n = 47$, adjusted $R^2 = 0.56$). It predicted that the distance from the inlet and the frequency of connection negatively affected the rate of wetland loss. Exclusion of the remaining predictors produced a loss in fit of only 2% underlining the robustness of the model. Mittelwasser subareas 2 and 16 showed the most negative rates in area loss between 1938 and 1960. But subarea 44 had strong negative values over the whole time of investigation as already observed in Figure 13 that led to the surprise mentioned. However, it was by far the most variable subarea in this subdivision and included parts that were classified as side channels. Subareas 50 and 49 showed the least negative rates of afforestation.

Sector B side channels

Side channels of Sector B were generally more variable with regard to the model prediction than the side channels of other sectors. The best model that could be constructed produced a relatively low explanation of variance, 39% ($p < 0.001$, $n = 144$, adjusted $R^2 = 0.39$). There were problems with the model as seen in Figure 15. Maximal negative rates were highly underestimated by the model. Nonetheless, according to the model, the strongest predictors for area loss were area, deepening of the Danube riverbed and the distance from the inlet, which respectively explained 14, 10 and 8% of the variance. Besides the three aforementioned predictors, the frequency of connection, its interaction with distance from the inlet, the vertical distance to the Danube, the flood frequency and finally the shape also influenced the slope of the regression significantly. However each of these predictors explained 5% or less of the variance on their own. With the exception of the frequency of connection, all predictors showed the relationship with the rate of area loss that one would have expected from classical models. Exclusion of the remaining predictors had little effect on the model with a loss of less than 1%. The model mainly predicted, that the rate of area loss per year followed the smaller size of subareas and the increase in riverbed depth. Besides these predictions, the complexity of water body shape, the proximities to the inlet and Danube, low values in connectivity and higher

frequencies of flooding, followed increases in area loss. There was a significant negative interaction between connectivity and distance from the inlet.

Subareas 18, 15, 17, 22 and 19 had the highest rates of area loss between 1986 and 2004, parallel to the period of the intense deepening. In subarea 14 rates were high until 1973. Consequently, one might assume that the diversity of the subareas in this sector was a cause for the lower fit of the model. The mentioned side channels except for 22 were small subareas located near the Danube and the inlet. Diversity among the subareas also produced methodological problems. For instance, subareas 12, 13 and 14 for later time intervals had to be excluded from analysis. In the process of area loss they could no longer be recognized at one time point (see Figure 13) and so values for their shape could not be determined.

Sector C main channels

Main channels in Sector C showed the clearest fit to the model in comparison to the other sectors. The model constructed explained 84% of the variance with the predictors: area, distance from the inlet and flood frequency ($p < 0.001$, $n = 50$, adjusted $R^2 = 0.84$). Here the logarithm of area had to be included in the model to fit the model assumptions.

This was also an important predictor explaining 65% of the variance on its own. Distance from the inlet and the flood frequency accounted for 19 and 16%. Exclusion of the remaining variables even improved the adjusted R square. According to the model, smaller sizes of subareas, proximity to the inlet and low levels of flood frequency increased the rate of area loss. When one examines the data from the model it is clear, that subareas 34 and 1, located next to the inlet showed the highest rates of area loss after 1973, parallel to the decrease in the flood frequency. Conversely, as seen in Table 2, subarea 34 showed the highest rate of woodland progression between 1938 and 1960. In the subareas 51, 10 and 47 there was little evidence of wetland loss in recent time intervals. Accordingly, there were no obvious temporal patterns.

Sector C side channels

As in the side channel model for Sector B, the side channels in C were variable as well albeit with 44% explanation of the variance. ($p < 0.001$, $n = 98$, adjusted $R^2 = 0.44$). As can be seen in figure 15, this model also underestimated the most negative rates of area loss to a high degree. Nonetheless, there were other significant predictors in the model. Area, explained 29% of the variance on its own. Water coverage, the deepening of the Danube riverbed, the flood frequency each explained 6% of the variance. The interaction

between riverbed deepening and the distance from the inlet explained 7%. Exclusion of the remaining predictors caused a decrease in fit of < 2%. Principally the model predicts that the rate of area loss per year followed decreasing size of subareas, water coverage frequency, flood frequency and stronger deepening of the Danube riverbed. The subareas 52, 46, 9 and 8 had the highest rates of area loss per year. With the exception of subarea 8, afforestation occurred between 1980 and 2004. In subarea 8 this occurred between 1938 and 1986. Subareas 23 and 33 showed the smallest amounts of area losses. For 23 this was particularly true between 1938 and 1986. Subareas 23 and 33 were the largest subareas in this Sector. In addition, these subareas had the highest frequency of water coverage.

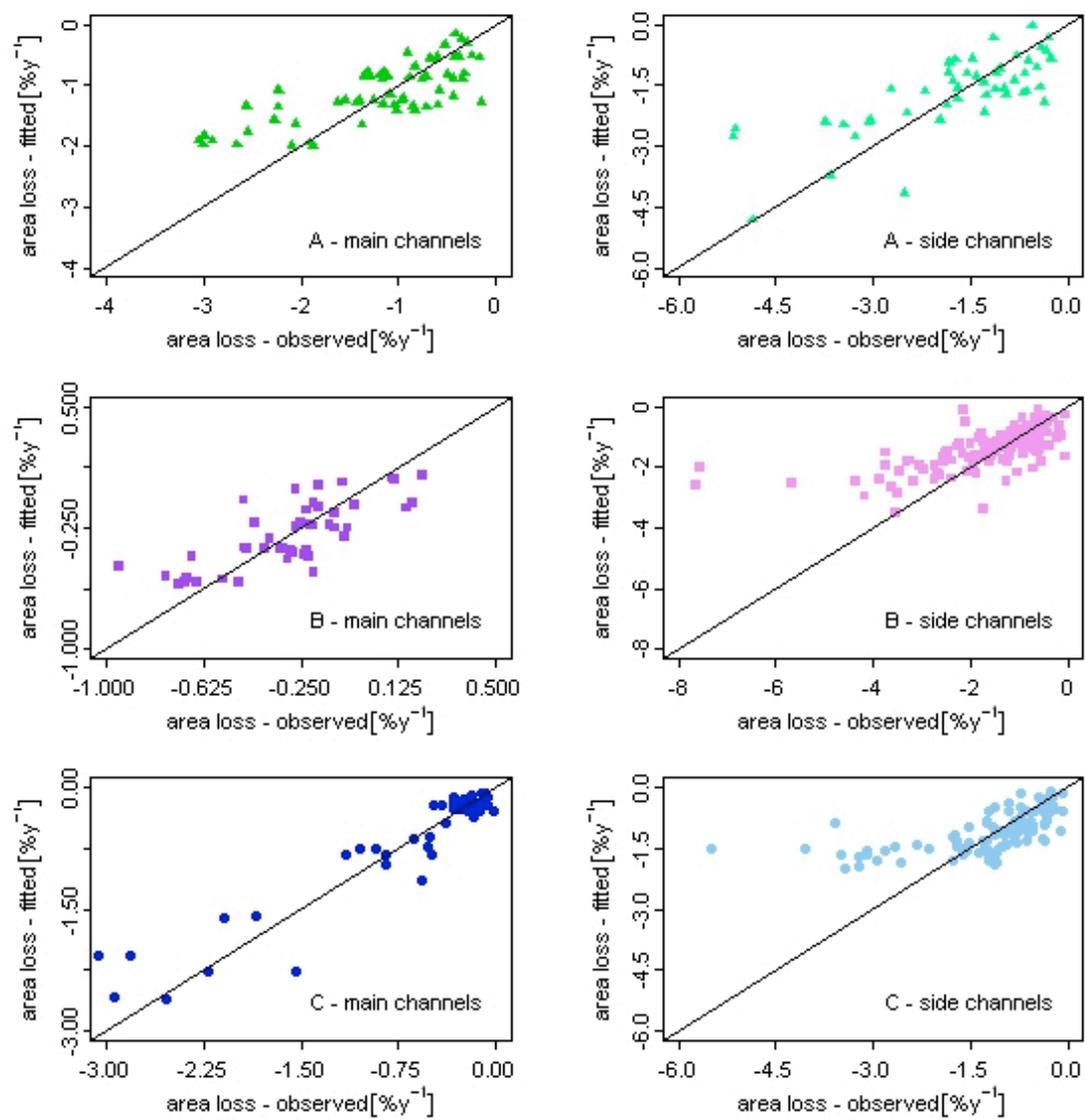


Figure 15: Comparison of observed with modeled wetland Loss in main and side channels of each sector. The lines indicate the ideal models with a 100 % correlation.

Discussion

During this study the overall loss of wetlands and water bodies in the Lobau was documented with a multi temporal analysis based on aerial images. Afforestation and terrestrialization coupled with a sinking groundwater table took place in the whole area over the whole time of investigation. The results underline the fact that aquatic and semi aquatic habitats in the area are prone to extinction. The spatial and temporal patterns of the changes found were, however, complex. Due to the broad nature of information integrated into the measured units and the limits in detectability, compromises had to be made to allow interpretation and conclusions about the processes involved. The methodological possibilities and limits of aerial pictures in the analysis of changes in dynamic systems, like former floodplains, are therefore first issues to be addressed.

The obvious unit to detect in aerial pictures is the change of area types. Detectable areas in historical pictures with different types and qualities can only produce coarse estimations, of dynamic processes and their consequences over long periods of time. The loss of water bodies is an expression of terrestrialization, the aggradation of fine sediment, coupled with groundwater loss. Water body loss further integrates allogenic and autogenic terrestrialization, potentially enhanced by sediment and nutrient intake from the river. The amount of effect depends on the location in the floodplain and the degree of connectivity. By using the integrated unit "loss of water expanse" differentiation of the individual contributions to water loss cannot be made.

Also the fact, that the direct consequences of terrestrialization is only partially detectable, makes interpretation more complicated and can cause more processes to be integrated at the same time. Fine sediment aggradations are only detectable on the shorelines. The aggradation in the water is not visible and does not immediately result in a loss of water expanse. To point out the importance of the waterbed aggradation, Reckendorfer et al. (2011) predicted complete terrestrialization of the water bodies in sector B and C within this century. The forecast was based on the thickening of fine sediment layers since river regulation. Another aspect was that when the shoreline was covered with helophytic vegetation, this boundary had to be used to define the open "water expanse". Loss in water expanse therefore also integrated processes coupled with the water-sided expansion and contraction of helophytes. Klötzli (1971) and Schröder (1979, 1987) have assumed eutrophication as a possible reason for the "die-back" of *Phragmites* in many lakes. The contraction and vegetation loss is a self-propagating system for eutrophication. In water bodies with large amounts of reed, the influence of nutrient rich river water could

even increase the measurable water expanse size. Similar losses of reeds with water expanse increases have been described by Dienst et al. (2004) after floods in Lake Constance. Other studies have examined the interactions between reeds and terrestrialization in more detail. In the Lobau, Kirschner et al. (2001) examined the effects of helophytes (*Phragmites australis*, *Tupha latifolia* and *Schoenoplectus lacustris*) on terrestrialization in Sector B subareas 2, 49 and 50. Helophytes were estimated to contribute 60 – 73 % of the mean annual fine sediment aggradation (3mm). Thus, a contraction of helophytic vegetation can occur, even though an aggradation of fine sediments and therefore terrestrialization has taken actually place. Another important aspect is the shape of the riverbed. This interacts with the predisposition and velocity of terrestrialization in each subarea individually. Flat banks favor the population of submergent and emergent macrophytes, whereas steep banks and deep water hinders them and reduces the potential of autogenic terrestrialization.

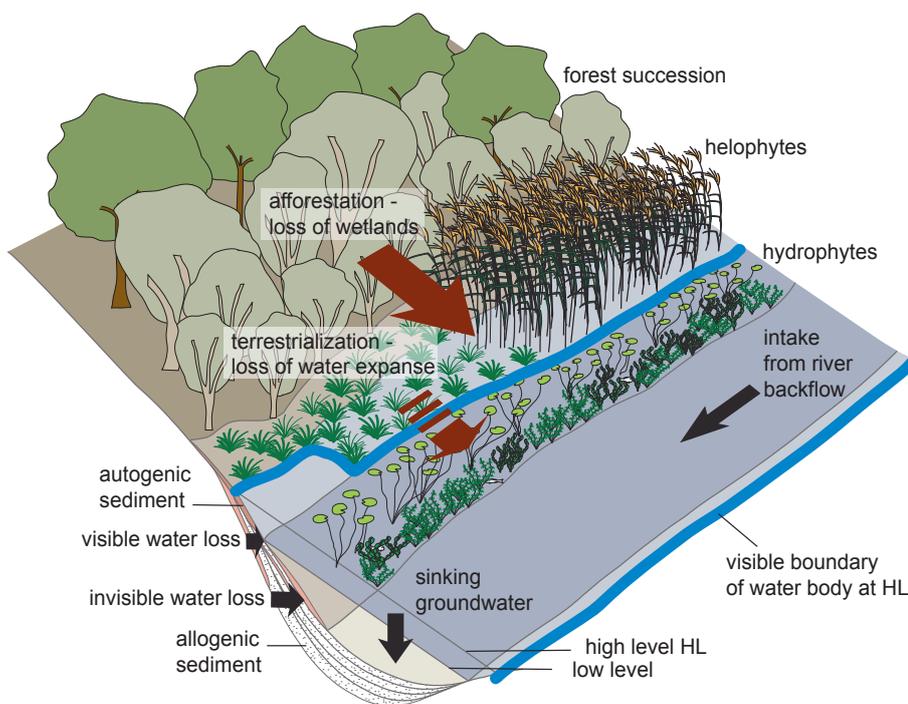


Figure 16: A Schematic Representation of Detectable Wetland and Water Body Loss in “Former Floodplains” with Aerial Images. The processes of afforestation and terrestrialization are shown (red arrows) with related causes (red arrows). Visible measures of water expanse (blue lines) do not reliably represent water boundaries. Afforestation progresses faster than terrestrialization.

Recalculation of water body loss to a mean water level did not suffice to produce a regular pattern over time. The reason for the high variability in water expanse data independent of water levels was probably due to the dependency of area change on myriads of processes that may have interfered with one another. The problems in classifying water bodies associated with differentiation of helo- and in-season hydrophytes in different types and

qualities of images, have already been described in the methods and results. Wetland loss and afforestation were easier to detect in all types of pictures and was therefore a more precise measuring unit for the analyses. Nonetheless, this parameter may even integrate more ecological and hydrological processes than expected. On one hand, terrestrialization and deepening of the groundwater table establish prerequisites for afforestation, namely a soil basis. On the other hand, plant ecological factors can interact to modulate afforestation.

In summary, even with the difficulties and interpretation limits mentioned above, overall patterns and indications of afforestation and terrestrialization associated with questions addressed were found. As an addition to Figure 6, where exemplifying photographic material visualized problems in detectability, a pictorial representation of the processes, limits of detectability and difficulties in interpretation has been shown in Figure 16. In this study the term “terrestrialization” was used to describe the visible loss in water expanse via allogenic and autogenic processes coupled to the sinking groundwater tables. The problems in water body detection associated with vegetation are evident in the drawing. The presence of different types of vegetation in the images outline furthermore the difficulties encountered during interpretation. The drawing also illustrated the dominant role of afforestation versus loss of water expanses.

In the area investigated there was a loss of 100.5 ha of semi aquatic and aquatic areas between 1938 and 2004 (65.5 y). That represented 30.8% of the areas present in 1938. In aquatic habitats losses were less pronounced but still present. Increases in water expanses were found in Sector A, and probably caused by human intervention, dredging measures. Former studies of Doppler (1991) and Eberstaller-Fleischanderl & Hohensinner (2004) confirmed the loss of aquatic and semi-aquatic areas in the Lobau, documenting both terrestrialization and increases in afforested areas. In these studies, other boundaries for the area of investigation were used and the definition of area types were based on different criteria. This has made quantitative comparisons with the present study difficult. One question addressed in the study was the comparison of wetland losses to afforestation and losses of water bodies to terrestrialization. There was a stronger trend towards afforestation than terrestrialization in main channels. The dominance of afforestation increased with the degree of disconnection. In Sector C, terrestrialization showed the same percentile trend as afforestation. In B, afforestation dominated increased relative to C and no terrestrialization was found. These effects were most striking in Sector

A, where the highest afforestation of main channels was observed. The amount of wetland loss in A was more than 30% in 2004 compared to 1968, in spite of the 50 % increase in water body expanse. The afforestation in Sector A occurred therefore independently from dredging measures. Comparing photographic data of 1960 and 2004 in Figure 6 shows an example of dominating afforestation versus terrestrialization. The progressing woodland even fragmented former arms, whereas changes of water expanse were mainly caused by expanding or contracting boundaries of helophytic vegetation.

The argument, that percentages of afforestation in the main channels of A were overestimated on the basis of their small size, was relativized by the main-channel woodland progression rates in all sectors. The homogeneous and continuous wetland losses in A were coupled to progression rates that were similar to those of other sectors, albeit in B and C the progression in space and time were characterized by more extremes. An alternative explanation for the high afforestation in A could be that more potential wetland was present from the fragmentation of its main channels before 1938. Additionally, water fluctuations in A were the lowest of the whole Lobau (Weigelhofer et al. sub.). The terrestrialized areas are comparable to meadows with low groundwater fluctuations. On the contrary, in Sectors B and C, more amphibic, terrestrializing area with helophytes was found. Groundwater fluctuations were more pronounced and terrestrialized areas were composed of gley soil (Schratt- Ehrendorfer 2011). This may have set limits to afforestation in these sectors.

Generally, nutrient intake from river water can stimulate high productivity in floodplain forests (Schnitzler, 1997). Nonetheless, comparably or even higher afforestation rates in more disconnected areas with lower river nutrient input could also be possible. Due to the supply of water and air for the mature soil of floodplain, with an organic layer, degradation, e.g. that of litter fall, is fast. Floodplain soils are therefore nutrient rich and allow fast growth without riverine nutrient input (Margl, 1972). Furthermore Schnitzler (1997) also reported, that in non-flooded sites, mycorrhiza and other fungal communities can be a substitute for this input. Hence, in areas with higher water level fluctuations, afforestation could be more limited by inundation than enhanced by nutrient intake from the river. The lower afforestation in the main channels of B and C could therefore have been a product of the higher frequency of inundations. Subarea comparisons of afforestation, flood frequencies and the degree of disconnection can be used to strengthen these arguments. The subarea 34 in Sector C and the dredged subareas in A are close to the river, but showed high rates of afforestation. However, these increases were limited to recent times, and probably more closely related to groundwater loss and low flooding frequencies.

These interactions were documented in the regression analysis. Jelem (1972) reported that the original forest community composition in the Lobau had changed so much, that it is hard to recognize characteristic floodplain communities anymore. Many flood-sensitive species, e.g. *Acer pseudoplatanus*, *Carpinus betulus* or *Tilia* had replaced flood-resistant soft wood species. The phenomenon of precocious ecological transition in forest development is defined as truncated succession, in the sense, that climax species followed species of pioneer succession and intermediate stages were skipped (Schratt-Ehrendorfer 2011).

Another plant ecological approach to flood plain succession is to consider the specific adaptations of predominant species link them to afforestation. In the absence of scouring floods and concomitant habitat renewal, the floodplain forest in the Lobau went through a continuous succession. By the 1970's primary softwood forest were reduced to shoreline willows in patches of the lower Lobau. The subsequent succession was characterized by *Populus alba* that required mature soil and is known to be competitive even in gley soils of terrestrialized areas. This species then populated a large segment of shoreline area in former arms (Jelem 1972). When present, *P. alba* reproduce vegetatively with basal shoots that can extend as far as the tree's height (Margl 1972). In this way, acceleration of afforestation in the upper Lobau and the side channels of B and C may have occurred. They supposedly had lacked softwood forests early on, due to the degree of isolation that was present before regulation. This enabled *Populus alba* to become dominant.

The terrestrialization in main channels associated with lowering degrees of disconnection suggests that allogenic sediment intake caused more water body loss, than in main channels of sectors where fine sediment aggradation was related to autogenic processes. In Sector C, ongoing loss in water expanse was even observed in main channels. Here, the subareas 34 and 1 are located between the inlet and the first weir. Water body loss was higher here than other subareas of this sector. This underlines the potential importance of allogenic intake in terrestrialization. In contrast, water expanses of Sector B were found to increase by about 7% between 1968 and 2004. Reckendorfer & Hein (2000, 2004) and Reckendorfer et al. (sub.) found a clear relationship between the rate of sedimentation and the distance from the inlet in main channels of the lower Lobau. With the measurements of the sediment thickness and composition they documented a pronounced impact of allogenic processes in main channels 34 and 1. They found that these effects were present up to the Gänshaufentraverse, the weir separating Sector C from B.

In areas where the effects of allogenic terrestrialization supposedly exceeded autogenic loss of water bodies, the difference was probably a consequence of sedimentation. This conclusion is reinforced by comparisons of estimated mean sedimentation rates associated with the types of terrestrialization. Reckendorfer et al. (2011) found that Sector C main channels 34 and 1, were dominated by allogenic sediment intake with sedimentation rates of 9 and 14 mm y⁻¹ and for the following three subareas after the first weir had rates of 8, 5 and 2 mm y⁻¹. These rates were two to five times higher than those found in the main channels of Sector B after the Gänshaufentraverse where aggradation was dominated by autogenic processes. The mean rates in the main channels of Sector B were only 3 mm y⁻¹. This was also confirmed by Kirschner et al. (2001).

Other non-autogenic processes like the expansion and contraction of helophytic vegetation certainly play an important role as mentioned above but their importance relative to autogenic processes in area losses is not clear. In figure 6 two areas were shown where helophytic vegetation defined the “growth” and “decline” of water expanses. Together with differences in riverbed morphology, fluctuations of the water level and the degree of eutrophication, these processes can produce a complex array of patterns in terrestrialization and afforestation as has been seen in the Lobau. For example, in the main channels of Sectors A and B there was no evidence of terrestrialization on the basis of the parameters measured in this study. The complexity here is also related to inconsistencies between this study and earlier analyses of the floodplains where larger areas, less detail and smaller databases were used. Jelem (1972) compared a field survey with data from 1963, when open waters were documented, and concluded that terrestrialization had occurred. Also Eberstaller-Fleischanderl (2004) found a principal loss in water bodies in the same time period of this study on the basis of map comparisons. In comparison, this study has underlined the fact that such generalizations of area loss have to be treated with care. The study has not negated the presence and the severeness of terrestrialization but it has demonstrated the need for detailed analyses in order to understand the complexity of spatial and temporal patterns and the factors undermining the loss of habitat.

There have been some earlier discussions of habitat shift patterns after river regulation. Schiemer (1999) did this for the Danube floodplains east of Vienna. According to his schematic representation, the Lobau should presently be in the last phase of total habitat shift. When one compares the patterns described in this study with his projections, it appears that this phase was present in 1938, around 50 years after river regulation.

Schematically there should have been an increase in wetlands before this phase, however, it is questionable whether that ever happened. In 1938, there was pronounced terrestrialization in the main channels of Sector A. It can be assumed that right after regulation, there was an increase in wetlands and large aquatic areas dried out. Areas of former through-flow side arms characterized with coarse sediment transformed into so-called Heißländer (dry habitats) due to the low water retention capacity of the sediment. The shortening of the Danube by channelization during river regulation caused and increased current velocity and with it a strong drainage of groundwater from the whole area (Schratt-Ehrendorfer, 2011). The pattern described above has, nonetheless, not been supported by other studies. Eberstaller-Fleischanderl & Hohensinner (2004) reported that all types of aquatic habitats and wetlands have continuously declined since regulation, with the exception of aquatic side channels. The reverse continuous pattern was reported for afforestation. As mentioned earlier the discrepancies here may be related to the amounts and detailed characteristics of the data used to draw conclusions.

In the main channels of Sectors B and C areas dominated by allogenic sediment showed more terrestrialization than areas dominated by autogenic fine sediment. On the other hand, afforestation, of the wetlands produced a different picture. These, wetlands were less afforested in Sector C than in B and A. Sediment differences in allogenic or autogenic dominated terrestrialized areas are known to be related to the organic content of the soil and its potential as a source of microorganisms growth (Reckendorfer & Hein 2004). The sediment grain size may also play a role. If so, soil aspects would have to be considered here to decide whether differences in organic content could have been critical for the different patterns of afforestation. In general, flood plain soil is highly productive (Schnitzler 1997), thus, differences in afforestation between sectors may be closely related to the sediments, the geomorphology of the channel beds and to differences in inundation frequency, as already mentioned above and indicated by the data.

The third question addressed was to compare areas dominated by autogenic terrestrialization to see whether nutrient input of superficial river connection had an effect on area loss compared to areas less influenced. Here, the comparison of main channels in Sector A and B would be useful. A is more disconnected from the river laterally and vertically. However, several water bodies of A were exposed to dredging measures and/or recreational activity and cannot serve as a comparison here. Nonetheless, the water expanses in Sector B's main channels did not show losses either. A continuous increase was documented with no terrestrialization. If so one would have to assume that nutrient

intake from the river produced an increase and not a loss of water expanse. If so the expanse may have been checked by the large amounts of helophytic vegetation bordering the water bodies of B. The Eutrophication Hypothesis developed by Schröder (1979, 1987) and Klötzli (1971) argues that a large biomass production of Phragmites due to high nitrogen and phosphate loads is accompanied by instability in their culms. Mechanical destruction causes a high amount organic matter to accumulate resulting in anoxic conditions and the production of phytotoxins. Higher eutrophication takes its course and with it the “die-back” of macrophytes as mentioned above. According to the fact that river water today contains more nitrogen and phosphate, than it did in former times (Schrott-Ehredorfer 2011) an increase in the trophic level could be a possible reason for the increases found in open water. The main channels of Sector C may also have been affected by these processes, because their shorelines were covered with a large amount of helophytes. More data would be necessary to clearly address this question.

Two final points should be mentioned with regard to afforestation. To begin with, the pioneer forest of *Salix purpurea* appeared in the main channels of Sector B between 1938 and 1960, in subareas 16 and 2. The result was a high amount of afforestation in this sector. *Salix purpurea* settles on raw gravel of former waterways and is actually a specific type of afforestation for floodplains. There is no organic layer or water retention capacity in its soil basis so this species requires water and nutrients from floods (Jelem 1972). After regulation these areas were probably the last stand for this type of forest. Now afforestation of wetlands begins on mature soil. A second point is that subarea 44 of the main channel in Sector B is the only one that showed a continuous wetland loss. This supports the statement that afforestation seems to have been hindered by floods and water fluctuations. This subarea was most disconnected being located after the 5th weir, away from the inlet and almost completely covered with helophytes. Comparing overall fluctuations of the mean day water levels of P16 with the ones of P17, it becomes obvious that subarea 44 was exposed to smaller water level fluctuations. This is especially true because subarea 44 is separated from the remaining main channels of this sector (see Appendix). As shown in the regression analyses, afforestation was more pronounced here on the basis of its disconnection and the branched shape of its shoreline.

As expected, side channels showed higher losses in water expanses and wetlands than main channels in all sectors. The losses of wetlands were also consistently more pronounced. Terrestrialization of side channels in Sector B and C occurred between 1980 and 1986. No clear trends were observed outside of this time interval. Side channels of

Sector A, however, showed a continuous loss in water expanse. For wetlands, in this sector, the amount of total loss of side channels was comparable to the losses in waterbodies. Eberstaller-Fleischanderl & Hohensinner (2004) have also reported higher losses in side channels than in main channels. In this report, the aerial pictures of 1938 and 2001 show an area loss of around 20% in side channels (personal observation). For one-sided connected arms (main channels) he reported losses of 8%. In spite of differences in classification criteria and the boundaries of the area investigated, these results are comparable with data in the present study.

The individuality of water expanse fluctuations is principally higher in side channels than in main channels, causing more incomparability and variability between the time points. Their shorelines can be expected to be more even, so that small differences in water levels produce larger and more variable area measurements. An indication of these difficulties can be found in this study where water expanses were recalculated to a mean water level. For side channels there was an exponential relationship between the modeled relationships and the measured areas whereas for main channels it was linear. The thickness of fine sediment layers may enhance individuality, because thick layers have a clogging effect on groundwater connectivity (Schiemer 1995, Reckendorfer & Hein 2000, 2004). It is therefore difficult to estimate the water level of side channels based on 4 water gauges that are not in their immediate area. A final problem here is the pronounced effect of shade and distortion of the shoreline image by trees in side channels, due to their narrowness. This has a strong potential for misinterpretation and impreciseness and therefore was another cause for variability among time points.

In theory, side channels are in a more advanced stage of terrestrialization. Their trophic state is eutrophic or even hypertrophic (Lüderitz 2009). Due to their small size and isolation, they can be compared to shallow lakes. As such they are strongly influenced by nutrient intake either by local terrestrial input, floods or seepage water (Schiemer 2006). Reckendorfer & Hein (2000, 2004) and Reckendorfer et al. (sub.) have found higher amounts of particulate organic matter (POM) and phosphorus in the side channels of the upper and mid Lobau. In these areas, thick layers of organic fine sediment can enhance the chance for anoxic conditions and re-suspension of phosphorus can occur, enhancing autogenic processes (Schiemer 1995). Significantly lower redox potentials were measured in side- than in main channels (Reckendorfer & Hein 2004) indicating to the high potential for re-suspension. In a eutrophic isolated pool of the upper Lobau, Bondar et al. (2007) measured frequent anoxic layers in the highest zone of the sediment. This also documented the high potential for re-suspension. Terrestrialization processes therefore

are assumed to be much stronger and faster, than in bigger and deeper basins. A stronger terrestrialization in side channels, than in main channels could therefore be assumed, even though the data in the present study did not show clearly document the effect. Here the effect may have been overshadowed by other ecological, hydrological and anthropological or methodological factors.

Concerning afforestation of side channels, more wetland loss was found than in main channels. In Sectors B and C the wetland loss of side channels was significantly lower. The losses also accelerated over time. The results of the statistical analysis also demonstrated that afforestation in smaller, complex areas of side channels paralleled the deepening of the Danube, and thus occurred in more recent times. Using the same data set, Reckendorfer et al. (2011) came to the conclusion that these factors are among the most important ones for the loss of aquatic and semiaquatic habitats in both main and side channels of the Lobau. Since quantitatively more subareas of side than of main channels were classified in the analysis, a similar result was possible.

As was the case for the small and fragmented main channels of Sector A, one reason for afforestation of side channels could have been that side channels had a higher amount of wetland area than water expanse. Some small or narrow subareas were already completely terrestrialized in 1938. Thus, more potential wetland was available for afforestation. Compared to the main channels in A, the side channels were smaller and even more fragmented. Another reason for the enhanced afforestation of the side channels could, again, have been the high gap dynamic of floodplain forests reported by Schnitzler (1997) due to the vegetative reproductive strategies in the second succession of species like *Populus alba*, or *Ulmus*, *Alnus*, or *Crateagus*. The deepening of the Danube could also have been an accelerating factor for afforestation due to the lower flood frequency and the lower groundwater table fluctuations. Many side channels became shallow lakes, where the connection to the groundwater level was decoupled earlier than in deeper channels and as a result enhanced terrestrialization took its course (Lüderitz 2009). On the positive side, the riverbed of the Danube in this section has recently been stabilized with coarser sediment that should increase the vertical connectivity and hinder further incision. One can only hope that this will lessen future dewatering in the Lobau.

The results of the statistical analysis also showed, that the loss of wetlands depends on diverse, more complex parameters that might not have been available or considered during analyses. Moreover, due to the high individuality of each subarea, regression

models based on a small number of cases, as in this study, might have been too “precise” to get interpretable results. The models were therefore mainly used to detect spatial and temporal patterns of afforestation. Many of the results have already been incorporated into discussion points. For side channels the predictors “area”, the “deepening of the Danube” during the time intervals and to a certain degree “shape” were found to be stable predictors, showing constant, explainable relationships with the rate of wetland loss and explained a larger amount of variability. Their importance did not depend on a few extreme subareas. The predictors were therefore considered as an indication of direct and perhaps causal relationships. For main channels, this was only true to a certain degree for the predictor “high flooding frequency”. This predictor was important in Sector A and C, showing that afforestation occurred primarily during time intervals with the fewer high floods. No other pronounced patterns were found in the main channels.

The afforestation of side channels is therefore statistically better connected to predictors describing autogenic terrestrialization processes and dewatering of the area than those associated with allogenic. Exceptions to this were found in the side channels of Sector B, where hydrological predictors were significant, but explained a minor amount of variability. Hydrological predictors like the distance to the Danube and the frequency of connection did not seem to have had a dominant explanatory role in afforestation of most sectors, since they did not follow the relationship with the rate of wetland loss predicted by classical studies.

One reason for this could have been, that the statistical consideration of each sector individually, with their inherent degrees of connectivity might have decreased the differences between the predictor values too much compared to the variability of the response values. Additionally with this grouping, the number of cases decreased too much for highly individual subareas. The number of “replicates” necessary for a confident interpretation was therefore often not attained. Thus, it might have been better to have used one model for all data points. In that case, hydrological parameters could have been significant and interpretable in a causal way. An example for an extreme subarea was 26 in the side channels of Sector A. Its predictor value for water coverage differed from the other subareas. Since it was also the area with the most severe rates of wetland loss it caused this predictor to explain much of the variability. Unfortunately, no certain conclusions could be drawn, because comparable subareas as replicates were not present. The same situation occurred for subarea 34 in the main channels of Sector C. Here the strongest rates of wetland loss in most time intervals were found. Since this area was comparably small, and data were relativized for size, it may have lead to an

overestimation of this predictor's importance in the model. The missing importance of hydrological factors in wetland loss may indicate, that afforestation depends more on geomorphological and plant ecological factors than being the direct result of terrestrialization. However, there was an inaccuracy in the predictor "frequency of connection". It is its homogeneous for all main channels except for subarea 34, the one right next to the inlet. Hence, classes of backflow conditions based on figure 3 (above) might have been better, since the input of the river would have more realistically estimated for each subarea. Unfortunately, comparable data were not available for the side channels.

Conclusions

The terrestrialization coupled with a sinking water table caused a severe loss of aquatic habitats in the Lobau between 1938 and 2004, especially in the side channels. In main channels areas with allogenic sediment intake showed obvious terrestrialization. However in the whole area, the loss of wetlands caused by afforestation seemed to have been even more pronounced. Afforestation increased with the degree of disconnection from the Lower to the Upper Lobau. The isolated backwaters of side channels showed the highest degree of wetland loss. The frequency of inundation was found to slow down the development of forest. The aspect of ongoing afforestation, the loss of biodiversity in floodplain forests and the negative effect of flow conditions on afforestation could therefore be another pro-argument for decision makers to reconnect the Lobau. Schnitzler, 1997 pointed out the loss of forest diversity in former floodplains. With this, its fauna has also become endangered. Examples are the European Pendulin Tit *Remus pendulinus* or the spittlebugs of the family Aphrophoridae, which are bound to soft wood forests (Schratt-Ehrendorfer 2011). In Europe, the only remaining primeval alluvial forest of this "aesthetic and biodiverse" of this kind is located in the Lanzhot Reserve, Slovakia. However, to ensure the maintenance of former floodplains as important refuge habitats, saving water bodies is an important "Ansetzen an den Ursachen", since inundation, water level fluctuations, the grain size of the ground and scouring erosive floods are prerequisites for floodplain forest maintenance, its growth and biodiverse, species composition.

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Hydrological improvement of former floodplains in urban areas: potential and limits.

Zusammenfassung

Die Lobau ist ein ehemals angebundenes Auegebiet der Donau im Osten Wiens. Seit der Donauregulierung im Jahre 1875 hat sich ihr Landschaftsbild drastisch verändert, was mit der Verlandung der Altarme, einem sinkenden Grundwasserspiegel und dem Voranschreiten des Auwaldes einhergeht. Viele aquatische und semiaquatische Habitate sind für gefährdete Arten schon verschwunden. Eine multitemporale Landschaftsanalyse wurde mittels Luftbildern von 1938 bis zum Jahre 2004 durchgeführt, um die räumliche Verteilung der amphibischen Flächen- und Wasserkörperveränderungen zu dokumentieren und zu quantifizieren. Mittels hydrologischer Daten wurde weiters untersucht, in wie weit die Veränderungen mit allgemein bekannten Faktoren für die Flächenveränderungen zusammenhängen.

Die Untersuchung hat gezeigt, dass sich vor allem die amphibischen Flächen gegenüber den Wasserkörpern verringerten. Der Wald scheint sich daher stärker auszubreiten, als die Wasserkörper verlanden. In den Hauptarmen der Lobau nahm die Bedeutung des Verlusts amphibischer Flächen gegenüber dem von Wasserkörpern mit dem Grad der Entkoppelung vom Hauptgerinne zu. In den isolierten Seitenarmen hingegen waren die Verluste beider Flächentypen am Größten. Dementsprechend haben sich die stärksten Verluste amphibischer Flächen in den Seitenarmen, aber auch im Hauptarm des obersten Sektors gezeigt. Dort wurde sogar eine Beschleunigung des Voranschreitens des Waldes festgestellt. Weiters hat die Untersuchung gezeigt, dass Waldwachstum in Zeiten häufiger Überflutungen eingeschränkt gewesen zu sein scheint. Der Verlust von Wasserflächen konnte im Hauptarm allerdings nur im untersten Sektor der Lobau festgestellt werden. Allogener Sedimenteintrag scheint daher zu einem stärkeren Flächenverlust zu führen als die Auflandung durch autogen produzierte Feinsedimente. Die Seitenarme zeigten zwar die stärksten Verluste von Wasserfläche, allerdings nur in bestimmten Zeitintervallen. Neben Verlandungserscheinungen stellen somit auch das Voranschreiten des Waldes, der Verlust von Biodiversität in Auwaldarten und die negativen Auswirkungen stark eingeschränkter Strömung ein wichtiges Argument für zukünftige Anbindungsmaßnahmen der Lobau dar.

Curriculum vitae

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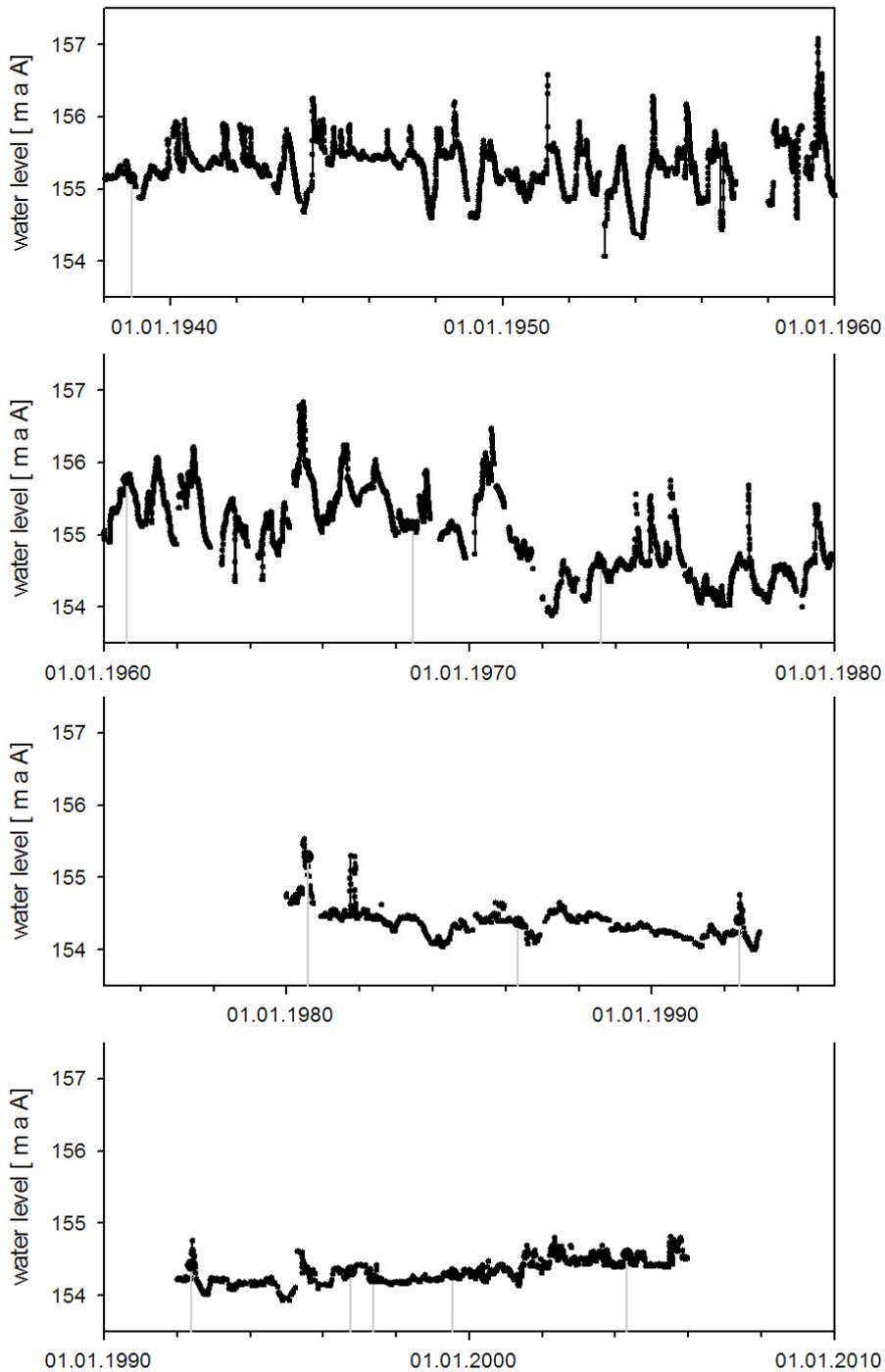
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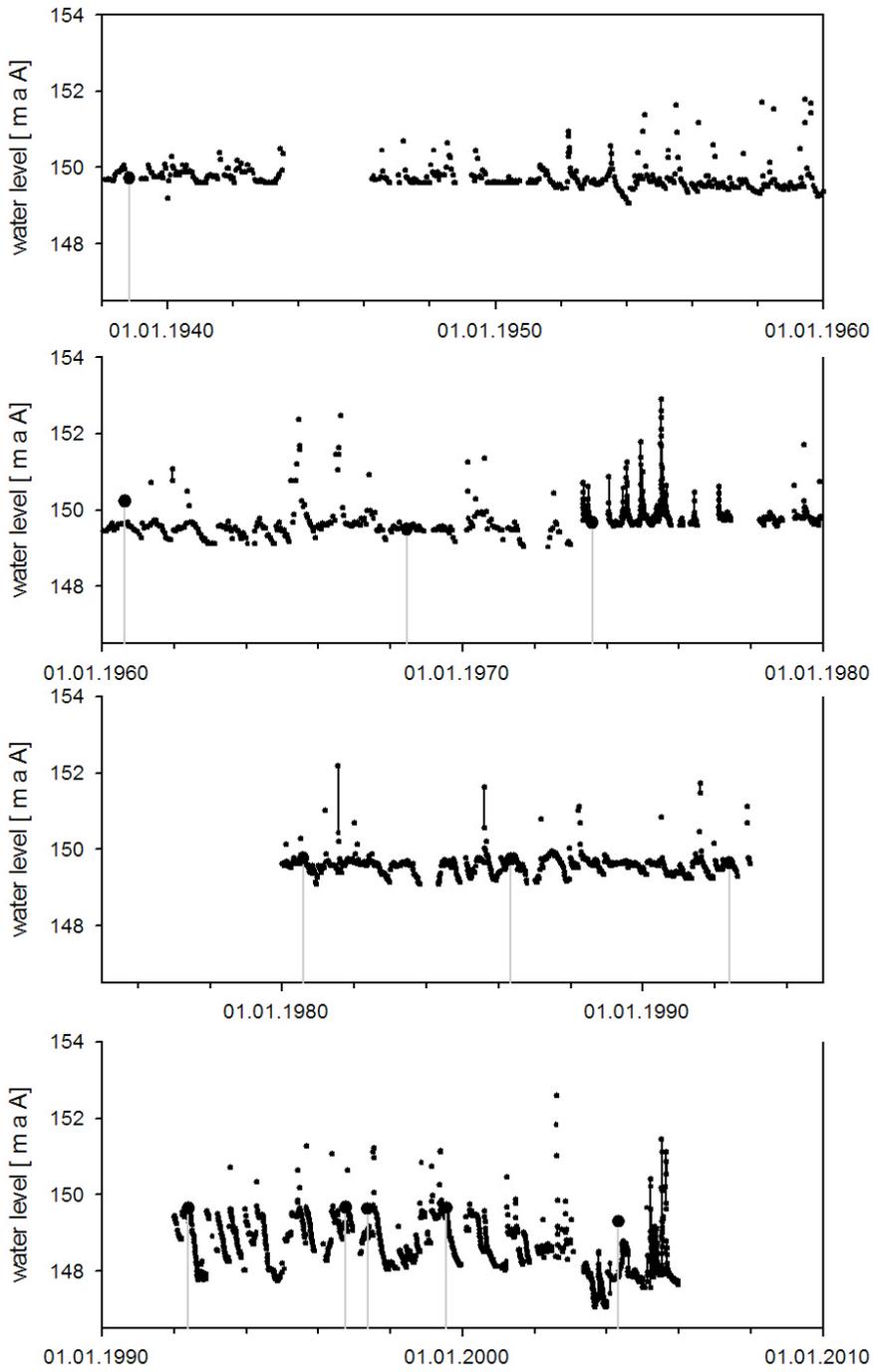
Appendix:

Daily water gauge measurements of the four water gauges over the time of investigation. The days of shooting are indicated in grey. The measurements served as an indication for the development of water levels in corresponding subdivisions. P11: Sector A; P16: upper part of Sector B; P17: lower part of Sector B; P Fischamend: Sector C

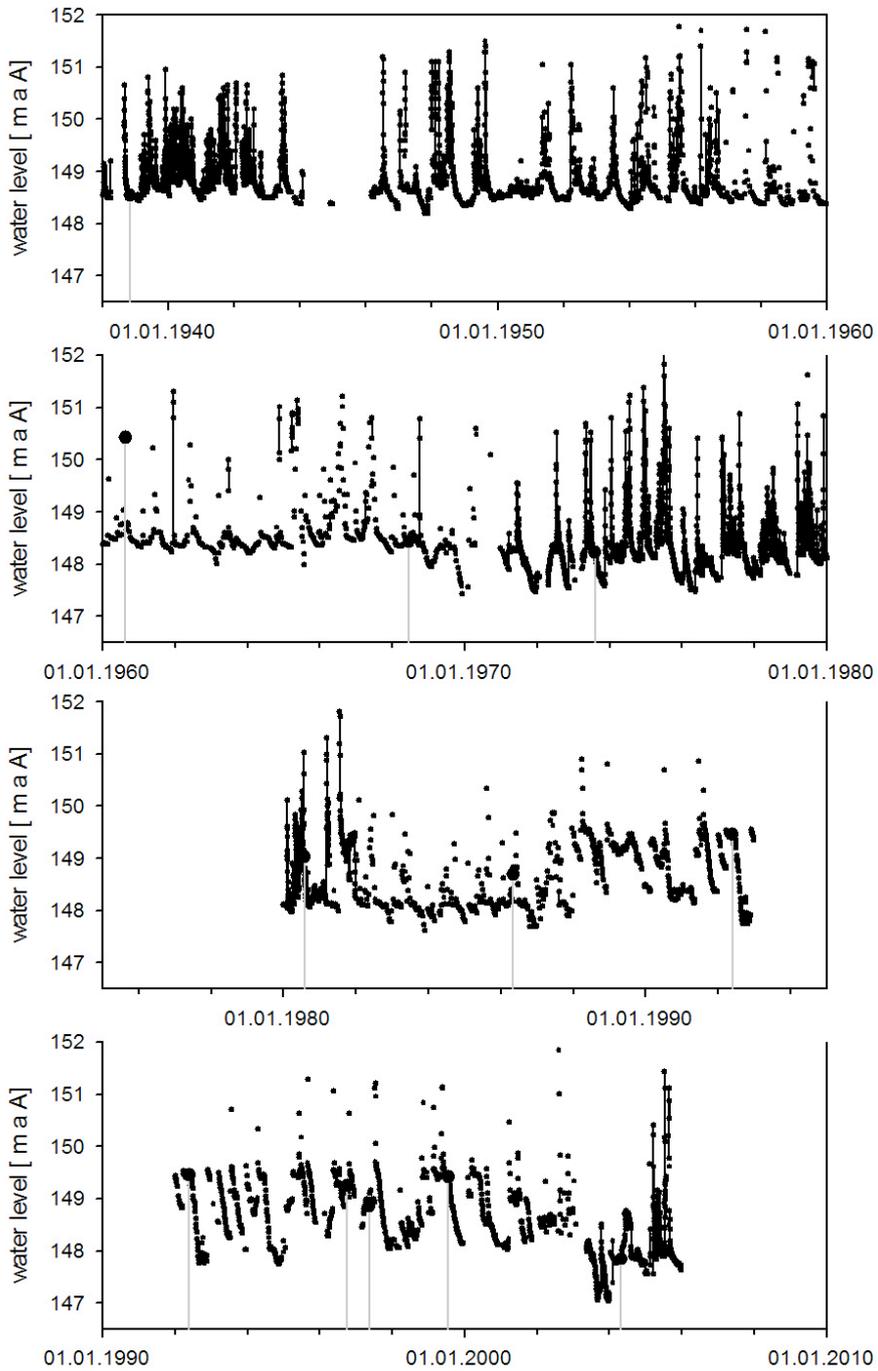
P 11



P 16



P 17



P Fischamed

