Dissertation

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The impact of invasive plant species on plant diversity of riparian habitats

The ecological effect and options for actions, focusing on the restoration of running water and the mangament of protected areas

Auswirkung von invasiven Neophyten auf die Phytodiversität in Aulandschaften

Ökologische Zusammenhänge und Handlungsoptionen im besonderen Hinblick auf Renaturierungen von Fließgewässern und Managment von Schutzgebieten

Betreut von:
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Universität für Bodenkultur, Wien 2014

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„THE IMPACT OF INVASIVE PLANT SPECIES ON PLANT DIVERSITY OF RIPARIAN HABITATS

THE ECOLOGICAL EFFECT AND OPTIONS FOR ACTIONS, FOCUSING ON THE RESTORATION OF RUNNING WATER AND THE MANGAMENT OF PROTECTED AREAS

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ÖKOLOGISCHE ZUSAMMENHÄNGE UND HANDLUNGS-OPTIONEN
IM BESONDEREN HINBLICK AUF RENATURIERUNGEN VON
FLIEẞGEWÄSSERN UND MANAGMENT VON SCHUTZ-GEBIETEN“

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Wien, den 20.02.2014

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CONTENT

1. Introduction

1.1 Evolution of biological invasions

Invasive alien species (IAS) are the second most significant driver of biodiversity loss worldwide (ISSG 2014). Together with habitat destruction and the fragmentation of the spread of IAS is one of the most urgent issues for global and local nature conservation (EEA 2012; IUCN 2014). IAS have a negative impact on ecosystem services, natural ecosystem development, economic activities and human health (EEA 2012).

> *"Invasive alien species are animals, plants or other organisms introduced by man into places out of their natural range of distribution, where they become established and disperse, generating a negative impact on the local ecosystem and species.(IUCN, 10.09.2013)"*

The research that makes up the subject of this thesis focuses on the biological invasions in a human-caused context. Human activity causes dramatic environmental changes of global importance (Vitousek et al. 1996; Kowarik and Boye 2003; Walter et al. 2005; Thuiller, Albert et al. 2008). The spread of non-native species has been documented since the earliest history of humanity (Behre 1988). Plant species like the walnut (Juglans regia) were distributed in Central Europe with the troops of the Roman Empire (K Loacker, 2007, Bakels, 2003). Another plant species of landscape-forming character is the Mediterranean cypress (Cupressus sempervirens), which is originally not native to the Mediterranean landscape but became a part of the European cultural landscape (Bagnoli 2009). Europe´s history is rich in examples of species introductions. More than 10,000 alien species are today introduced and established in Europe (EEA 2012). This number of introduced species is increasing annually (Vitousek et al. 1997). Not all alien species automatically cause damage. Most non-native species do not manage to establish self-sustaining populations or do not compete with native species. About 15% of all introduced species have a negative ecological or economic impact (Hulme & Drake 2009, DAISIE 2013, EEA 2007).

> *"What causes a species to be labeled as invasive rather than simply non-native is its ability to harm native species through competition and predation, or by transferring pathogens and parasites, or through hybridization. (EEA 2012, p.10)".*

Current ecological impact of IAS is difficult to estimate, because of the lack of knowledge of new introduced species. Many IAS show a negative long-term effect, but a positive economic short-term effect (EEA 2012). For example, the introduction of the IAS *Robina pseudoacacia* from North-America to Europe, caused various positive economic effects in forestry. The fast growing wood shows excellent abilities in several scopes; meanwhile it invades natural habitats and causes irreversible damage on species composition (Richardson 1998). Once an alien species is established, it is hard and in many cases impossible to eradicate, like the *Solidago gigantea*. Beside human activity in a globalized world, global environmental changes, like the global change do support biological invasions. In economic terms the annual losses caused by IAS in European countries are estimated to be at in the rage of EUR 12 billion per year (EEA 2012, Kettunen et al. 2009). Agriculture, forestry and fisheries are the most affected. The impact of IAS can be direct, like the damage of Japanese knotweed (*Fallopia japonica*) to flood defense structures (EEA 2012). An example of the negative impact of IAS on human health is the giant hogweed (*Heracleum mantegazzianum*) of the common ragweed (*Ambrosia atemisiifolia*), which are one of the most pollen-allergic plants causing serious health risks for humans (Carlton 2003; Taramarcaz et al. 2005).
Terrestrial invertebrates and terrestrial plants are the biggest groups of IAS causing economic and ecological damage. The negative impact of terrestrial plants on ecological processes is estimated to be greater than on economic processes, compared to terrestrial invertebrates (EEA 2007). The global number of vascular plant species is estimated to be 300,000 species. Plant species have a key role in understanding disturbances on global diversity (Pearson 1995; Kreft & Jetz 2007). Europe´s flora counts around 12,000 native vascular plant species and around 700 established alien plant species (Vitousek et al. 1997). Almost 5.7% of the total number of European plants species are not native to Europe (Vitousek et al. 1997). Compared to Central European countries, Austria is a good example of the average situation of IAS´s. 4,060 vascular plant species grow in Austria. 1,110 alien plant species are non-native aliens (28%), 275 alien plant species (20%) are established in natural ecosystems. But only 17 alien plant species are considered as invasive, causing serious economic and ecological impact (Essl et al. 2002). This situation of biological invasion is comparable to Switzerland or Czech Republic (Vitousek et al. 1996; Pyšek et al. 2003; Walter, Essl et al. 2005).

1.2 Interspecific Competivity

The success of invasions relies not only on the abilities of the invasive alien species, but also on the biology of the resident native species. The characteristics of the native species decides whether a plant community will be invaded by IAS or not (Dukes 2002).

The aim of the research conducted and summarized herein is to compare the germination process of common native and invasive alien plant species in order to determine the possible advantages and disadvantages of individual species within the invasion of riparian areas and to thereby facilitate conservation efforts. The massive occurrence of invasive alien plant species can be correlated with the immediate suppression of native plants (Pyšek & Prach 1995; Richardson al. 2007). Besides gathering phenological data on riparian species' life history, the experiment provides information about their competitiveness in various controlled environmental conditions. A further goal is to investigate dominance in the early development stage under the exclusion of abundance of the species in the community, interspecific interaction, environmental conditions and the intensity of human disturbances.

These excluded factors will be discussed in subsequent studies. The major results of an experiment under controlled conditions using seeds of four invasive alien species – *Impatiens glandulifera, Impatiens parviflora, Solidago gigantea* and *Bunias orientalis* – and seeds of five native species: *Eupatorium cannabinum, Senecio sarracenicus, Tanacetum vulgare, Urtica dioicia* and *Salvia glutinosa* are discussed in this thesis in order to reveal management methods.

Figure1 IG Impatiens glandulifera seedings

1.3 Target: Biodiversity

The distribution of non-native species has consequences for evolutionary processes like hybridization, niche displacement or competitive exclusion of native species (Mack et al. 2000; Mooney &Hobbs 2000; Byers et al. 2002). Biodiversity is affected by IAS on all ecological scales. IAS cause damage to biological diversity at the genetic, species and ecosystem levels (Hejda et al. 2009, EEA 2012). The IUCN Red List of Threated Species, an officially recognized international instrument for estimating the biodiversity loss, marks 110 critically endangered native European species that are in danger because of IAS (DAISIE 2013; van Ham 2013; IUCN 2014).

> *"Biological diversity (biodiversity) means the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are a part; this includes diversity within species, between species and of ecosystems (IUCN, 2000)".*

In the last decade, research on the negative effect of IAS on biodiversity on the community level shows has increased and drawn greater attention (Parker et al. 1999; Dukes 2002) (Hulme & Bremner 2006; Hejda et al. 2009). To know the effect of IAS on the function of resident communities is important for the development of applied management (Gordon 1998; Pyšek & Richardson 2010). To measure the impact of IAS on plant communities, responding in plant cover, has the advantage that a wide variation within a large data set can be analyzed (Levine et al. 2003; Hejda et al. 2009). Differences in plant cover show the degree of dominance within plant communities. It is one of the most important ways to detect the impact of IAS on species richness and species evenness (Richardson et al. 2000; Hejda et al. 2009).

The negative impact on native plant communities depends on various factors and characteristics of the resident community (Dukes 2002). Therefore it is necessary to analyze the impact of IAS on a large scaled ecosystem, like the riparian area, regarding to differences habitat type and species abilities.

Further changes in development and structure of plant communities caused by IAS are to proof. Results of various studies show, that diverse systems of an high level of species richness are often less affected by biological invasion (Levine & D'Antonio 1999; Dukes 2002). A review of studies correlating species richness to the abundance of native and invasive plant species shows a diverse gap of results between manipulative and observational studies (Dukes 2002). While the majority of observational studies support the hypothesis that native species richness of the resident plant community reduces the success of invasions, manipulative studies show controversial results (Stohlgren et al. 1999; Dukes 2002; Levine et al. 2003). The research on the impact of IAS on riparian habitat will show results of observations in a riparian with a diverse structure of different habitat types correlated with manipulative experiments under controlled conditions.

To have detailed information on managing invasive alien species at the level of a vegetation community is a positive step towards protection of vulnerable natural or semi-natural ecosystem. Research on biological invasion is necessary to create successful programmes against biodiversity loss on local and regional levels. The protection of biodiversity is a highly calculated need for the sustainable development of future ecosystem services (Ehrlich & Ehrlich 1982; Ehrenfeld 1988). In the last two decades the aim to protect global biodiversity by reducing the impact of IAS has been recognized and discussed in several nature conservation instruments. The trend of legislative response to the danger of IAS is currently growing at various levels of government and legislation.

1.4 Legislative frameworks

1.4.1 International framework

Biological invasions are not constrained by borders. International programs and conventions are necessary to achieve a global efficiency. A broad number of conventions and intergovernmental organizations are considering the issue of biological invasions with varying different level of intensity (Clout et al 2009). Over 50 internationally agreed legal instruments deal with IAS in diverse sectoral context. The Plant Protection Convention (IPPC), the Convention on Biological Diversity (CBD), the Convention on Wetlands (Ramsar) and the Convention on International Trade in Endangered Species of Wild Flora and Fauna (CITES) are considered to be the primary conventions dealing with IAS focusing on the riparian area.

The Plant Protection Convention (IPPC)

The IPPC was signed 1951 with the aim of preventing the introduction of pests and plant products. This framework developed from the agricultural sector in response to the economic impacts of IAS. The IPPC was the initial reason for many international frameworks on IAS.

Convention on Biological Diversity (CBD)

At the Earth Summit in Rio de Janero, Brazil, in 1992, a strategy to ensure the conservation of biological diversity and sustainable use was agreed to by world leaders. The CBD requires the prevention, control or eradication of IAS which threaten ecosystems, habitats or species. At the 8th Conference of Parties in 2006 the CBS´s program has been taking the diversity of inland water ecosystems into account within the adoption of decision VII/4. The following approaches and guiding principles were important for the development of strategies of the European Union (EU) and the Council of Europe.

The Convention on Wetlands (Ramsar)

In 1971, in Ramsar, Iran, the Convention on Wetlands (Ramsar) was signed in order to provide international aims of conservancy programs on coastal and inland wetlands. Resolution VIII/18 considered the impact of IAS on wetlands. Parties to Ramsar have to undertake the risk assessments of alien species which influence the ecological character of wetlands.

The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)

CITES is a framework to subject the prohibition of trading endangered species. The impact of IAS has been discussed since the 13th meeting of the CITES Conference of Parties in 2004. Parties are asked to develop legislations and regulations controlling invasive alien species.

International Union for Conservation of Nature (IUCN)

On the international level non-binding intergovernmental forums have an important function to adopt action programs, resolutions or technical guidelines (IUCN 2000, Clout et al 2009). The IUCN, founded in 1948, plays a key role in developing international frameworks. It brings together over 80 states, 110 governmental agencies, around 800 non-governmental organizations and some 10,000 specialists from all over the world. The IUCN (International Union for Conservation of Nature) started to discuss the negative impact of IAS on biodiversity in 1996 at the 1th IUCN World Conservation Congress in Montreal, Canada. The work on guidelines for prevention of biodiversity loss caused by IAS was originally indicated by interests related to agriculture and human health. According to the 5th Meeting of the Conference of the Parties to the Convention on Biological Diversity guidelines with the aim to assist managers, decision-makers at all levels and to develop strategies, were published. In 1997 "the global Invasive species Program (GISP) was implemented in order to promote global cooperation in prevention and management of IAS. Global strategies and further guidelines were developed (Clout et al 2009). The Invasive Species Specialist Group (ISSG) of the IUCN provides the exchange of knowledge of IAS within a network of specialist from over 40 countries. The aim is to provide technical and policy advice (ISSG 2014).

1.4.2 Regional framework

Agreements on international standards are in the process of being implemented by different governmental sectors in national frameworks. Concerning the prevention of the introduction of IAS, governmental responses are useful to implement functional systems (Clout et al. 2009). Through a rage of legislative instruments of the European Union (EU) the prevention and control of IAS is considered and implemented in the national legislative instruments of the member states. Considering the research on the impact of IAS in riparian areas, two European community Directives show a greater relevance. The Habitat Directive plays a major role in the achievement of the targets of the EU's Biodiversity Strategy to 2020.
Therefore a network, Natura 2000 Network, of protected sites was initiated in order to protect areas of importance for European nature conservation aims. The study area is part of a Natura 2000 area, *"Tullnerfelder Donau-Auen"*. The aim of the Habitats Directive (1992) is to assure the long-term survival of Europe's most valuable and threatened species and habitats. Human activities are not excluded of the management of Natura 2000 sites, as long as the management is ecologically and economically sustainable (Sharpston 2010; Commision 2013). The aim of the research will be to assess the protection targets towards the biological invasion in the study area.

EU Biodiversity Strategy to 2020, Target 5:
"By 2020, Invasive Alien Species and their pathways are identified and prioritised, priority species are controlled or eradicated, and pathways are managed to prevent the introduction and establishment of new IAS (European Commission, 2011)."

The second legislative instrument of the European Union, implemented in national framework, is the EU Water Framework Directive. The Water Framework Directive was introduced in the year 2000 with the aim to reach “good water status” for all European waters by 2015 (Mostert 2003). Its aim is to achieve river basin management, including restoration programs, much like the one in place in the present research area. The Water Framework Directive plays a role for the ecological development of European riparian areas in the last decade (Seabloom et al. 2013).
Current legislative efforts are targeting the effective control of IAS in the EU. The European commission has prepared a proposal to the European Council and Parliament for an EU-wide strategy on invasive alien species (Hulme & Drake 2009). After the 1980s, over 30 eradication programs of invasive plants and animals successfully took place in the EU. Most of them were co-founded by the LIFE programs. LIFE is the EU´s largest financial instrument in place since 1992. National frameworks are criticized not to react rapidly on initial introduction of IAS (IUCN 2000; Essl et al. 2002; van Ham 2013).

1.5 Role of riparian areas

Riparian areas (riparian zones) are the fringe between aquatic and terrestrial ecosystems of rivers or streams. Riparian areas are important habitats for important ecological functions and European plant diversity (Gregory et al. 1991; Naiman et al. 1993; Naiman & Decamps 1997; Pfadenhauer 1997; Hood & Naiman 2000; Richardson et al. 2007). Riparian areas play an essential role in the complex of wild life ecosystems (Elmore & Beschta 2006). The high plant diversity of riparian areas is caused by a complex mosaic of diverse habitats, periodical return portions and the early succession stage of plant communities (Hood & Naiman 2000). This high amount of natural disturbances is on the one hand the reason for a high complexity of plant diversity, but on the other hand the reason for successfully establishing invasive alien plants (Pyšek & Prach 1994; Hood & Naiman 2000). Therefore the vulnerability of riparian ecosystems needs to be analyzed and discussed towards appropriate management methods in riparian areas.
The impact of biological invasions to island ecosystems has been recognized much earlier than the impact of IAS in many continental areas (Vitousek et al. 1997; Hood & Naiman 2000). After island ecosystems - freshwaters, which include riparian habitats, are the most effected ecosystems of biological invasion (EEA 2012). Over the last decades, European riparian zones are strongly affected by biological invasion (Pyšek et al. 1994; Essl et al. 2002; Schmitz & Lösch 2005).

Several studies have shown that riparian areas are among the natural habitats most affected by the invasion of alien plant species (Pyšek et al. 1994; Essl et al. 2002; Schmitz & Lösch 2005).

Rivers have a functional role for the dispersal of plants through the landscape (Nilsson et al. 1993; Johansson et al. 1996), and play a key role in the spread of diasporas of invasive alien plants (Sukopp 1969; Pyšek & Prach 1994; Johansson et al. 1996; Kolar & Lodge 2001; Kowarik & Boye 2003). But riparian areas are threatened by the invasion of alien plants in various ways: in general, invasion by alien plants is very successful within anthropogenic disturbed habitats. European riverine landscapes are often heavily altered by anthropogenic disturbances (Jungwirth et al. 1993; Poppe et al. 2003). Furthermore, the natural character of floodplain forests is based on a natural disturbance necessary for the life-cycle of riparian habitats (Jelem 1974). Within the present case study of influencing factors (Richardson et al. 2007), the effect of canopy, proximity to the river stream or flooding events will be analyzed and discussed.

The ecological and socio-economic impact of IAS has become the topic of increasing scientific research (EEA 2012). Although the topic of biological invasions has recently attracted growing interest (Lohmeyer & Sukopp 1992; Mooney & Cleland 2001; Sukopp 2001; Richardson & Pyšek 2006) only a few studies have discussed the consequences of biological invasion in the riparian area (Tickner et al. 2001; Imbert et al. 2012). Specifically the role of riparian seed banks and the phenogical abilities of invasive alien plants seem particularly underreported (Goodson et al. 2001). Many protected areas are related to riparian ecosystems (Manzano 2000). The management of invasive alien plant species in riparian areas is gaining attention for conversation of European biodiversity.

1.6 Challenges for ecological restoration programs

Ecological restorations are implemented to improve ecological functions of ecosystems that were affected by human-mediated changes to diversity (Richardson et al. 2007) (Noss 1990; D‘Antonio & Meyerson 2002). In the most cases restorations are following the target to return to previous conditions through reestablishment of species and structure of an ecosystem ((Richardson et al. 2007) (Decamps et al. 1988). Focusing on European riparian ecosystems, it is important to mention that riparian areas have been under anthropogenic influence for centuries. Since industrialization, the regulation of river streams and intense manipulation of Riparian vegetation took place. Today´s river restoration projects have not just to deal with issues of ecological engineering, but also with social and economic dimensions (Gobster & Hull 2000; Harris & van Diggelen 2009).

The present case study is focusing on the ecological restoration of the river Traisen. As part of the Life+ Traisen project, the downstream section and the outlet of the river Traisen will be brought into a natural structure. The uniqueness of this restoration project is that the creation of the new river course is not linked to historical data. A new river landscape will be created with the aim to improve hydrological and ecological conditions of the downstream section. The project Life+ Traisen is a positive practical example for implementing various dimensions and stakeholder requirements. The focus of the present study is to prove the handling of IAS within the project targets.

IAS do play a key role in the restoration process (D‘Antonio & Meyerson 2002). The presence of IAS is influencing the restoration method, the evaluation of target achievements and the monitoring process (D‘Antonio & Meyerson 2002; Merritt & Wohl 2002). Analyses of studies concerning plant invasions linking to restoration management show a need of clear principles in the restoration processes.

In riparian ecosystems restoration processes need to be considered carefully in order to achieve a positive effect in the long-term (Richardson et al. 2007). Due to the high variety and the spatial scale of riparian ecosystem functions, many studies recommend focusing on a multistep-goal scale (Palmer et al. 1997; Moerke & Lamberti 2004).

1.7 Objectives

This research is a contribution to the scientific discussion on invasive alien species (IAS). The study is correlated to practical advice to nature conservation and ecological restoration issues. Based on the knowledge of invasive species' germination and phenological ability, the impact on plant diversity and habitat preferences, as well as information on invasive species management in other cases, guidelines for the management of invasive species in riparian areas was developed in order to reduce the biodiversity loss caused by IAS. The following research questions were in the main focus of the study:

Species composition and vegetation development

Do invasive species cause significant changes in species composition of riparian vegetation?

The aim is to assess how the invasion does influence the vegetation development in the riparian area. To this aim, the abundance of native and invasive species was calculated.
Analyzes of the germination ability and the phonological development of native and invasive riparian species will show results for the competitive ability of IAS. The focus lies on the development of the invasive species *Impatiens glandulifera, Impatiens parviflora, Solidago gigantea* and *Bunias orientalis.*

Introduction mechanisms and habitat preferences

How do Invasive alien plant species invade riparian habitats?

In order to control the biological invasion, precise knowledge on introduction mechanisms are necessary. Prevention is just possible when pathways are detected.

Are factors like the proximity to open water bodies or roads significant for the spread of IAS?

The water body proximity, road proximity and cultivation intensity of each plot will be assessed for significant relevancies. Riparian areas are very rich in variety on habitats. Specifically, the study area shows a high diversity of riparian habitats.

Does the invasion of alien plants differ among habitat types?

Semi-dry grassland and floodplain forest in different ages may differ in their level of impact to biological invasion and require a different management scheme. The aim is to find out how does the abundance of IAS develop within different habitat types of the riparian area.

What are the consequences for restoration programs in riparian areas?

The focus lays on the objective, namely in which habitats an increase of occurrence of *Impatiens glandulifera, Impatiens parviflora, Solidago gigantea* and *Bunias orientalis* in the total plant cover is significant. Within this analysis the influence of canopy will be discussed. To detect habitat preferences are useful to argue the consequences for IAS management plans in riparian areas.

Plant diversity

Do IAS cause changes in riparian species richness?

Studies have shown that invasive plant species have a negative effect on native plant diversity. The effect of IAS on changes in species richness and evenness will be analyzed among habitat types focus on the development of diversity of invaded plots by *Impatiens glandulifera, Impatiens parviflora, Solidago gigantea* and *Bunias orientalis.*

Soil seed bank occurrence

Are seeds of IAS occurring in the riparian soil seed bank? Is the occurrence of seeds of IAS influencing the development of aboveground plant cover?

Guidelines on the management of IAS for restoration projects cannot neglect the seed bank occurrence of IAS. One aim is to detect how the structure of IAS`s seed in the seed bank structure (according the occurrence of seeds in 3 layers of depth and seed density) is. I want to analyze the relationship of seed occurrence to the development of aboveground plant cover development and discuss the consequences for restoration programs in riparian areas.

2. Method

2.1 Study area

The study area lies in Lower Austria, along the recent alluvium of the river Danube (Figure 2). The whole area was regulated in the 19th century, in order to control the floodings and increase the cultivative land use. Over the course of the construction of the Danube power plant at Altenwörth in the 1970's, the outfall of the river Traisen was dislocated. The new outfall of the river Traisen was placed 7.5km downriver into the tail water of the power plant. Ecological needs were not respected. The river Traisen was turned into a water canal without structures and connectivity to the riparian area. Due to the dislocation of the river bed which caused drying up of the water bodies, the species richness of the riparian area is endangered. Dry grasslands and semi-dry grasslands in the project area have an important role for the plant diversity of the riparian landscape (Lapin & Bernhardt 2013). Dry grasslands are areas of remarkably high species diversity that needs management to control the shrub encroachment. The ongoing shrub encroachment leads to the loss of species diversity and, in some cases, an increase of invasive alien plant species occurrences.

Figure 2 Location of the study area - Google earth

Although the dislocation influenced the ecological situation negatively, the species richness of the project area is calculated as very high. The occurrence of species of the fauna-flora-habitat directives is very high. Therefore the area was incorporated in the EU-wide network of Natura 2000 sites - the network of nature protection areas established under the Habitats Directive and Birds Directive (1979). As of November 30th, 2004 the area has been part of the Natua 2000 FFH site 16 *"Tullnerfelder Donau-Auen"* (NÖ 2013). The Natura 2000 site Tullnerfelder Donau-Auen is 19,483 ha in total size (Ellmauer et al. 1999). One of main focuses of this Natura 2000 site is the protection and support of riparian vegetation.

The ecological function of riparian forests, broadleaved alder forests and transitory phytocoenoses and dry grasslands of the riparian areas are important protected and supported objects of the Natua 2000 site *"Tullnerfelder Donau-Auen"* (Natura2000 2014).

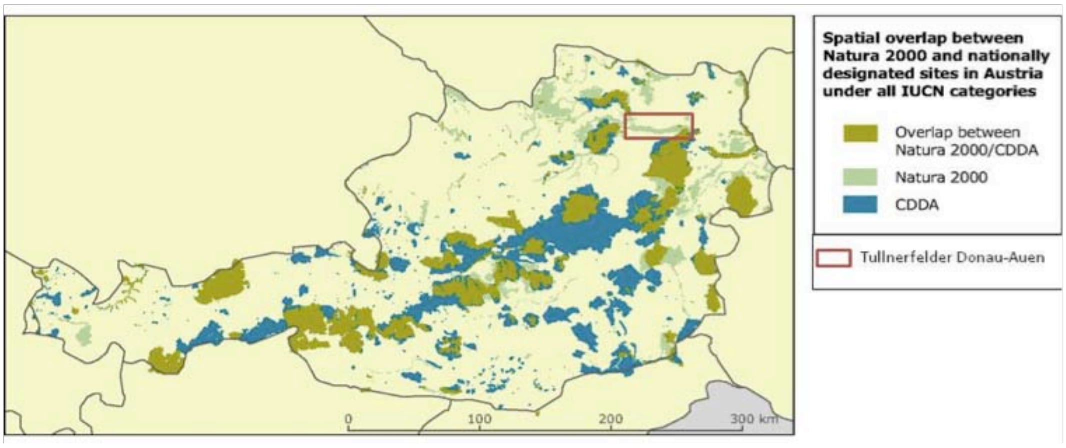


Figure 3 Map of all Austrian protected areas, focusing on the spatial distribution of Natura 2000 sites and nationally protected sites (CDDA - Common Database on Designated Areas); European Environment Agency (adopted map, EEA, http://www.eea.europa.eu/data-and-maps, 30.09.2013)

The Natura 2000 site *"Tullnerfelder Donau-Auen"* represents one of the largest connected riparian ecosystems of Austria (Figure 3). The high occurrence of endangered species is one of the major arguments for the national importance of the Natura 2000 site *"Tullnerfelder Donau-Auen"* for the Austrian and European nature conservation targets (NÖ 2013). The protection of gallery forests on the floodplains of the stream of the river Danube is the main focus of the management programs. Furthermore the management targets are to support the development of a mosaic of typical riparian ecosystems, consisting of wooded and not wooded ecosystems. The diminution of human disturbed areas, as well as the restoration of natural dynamics is the aim of the complex and interdisciplinary targets of management programs (NÖ 2013).

But the ecological structure of function of the riparian area is strongly endangered after the construction of the Danube power plant at Altenwörth and Greifenstein. The installation of power plants caused serious changes in hydrology to the area. Most abandoned meanders were disconnected from the main stream. Without this important cross-link to the main stream, the ecosystem is highly endangered. Furthermore the reduction of river flooding leads to massive disturbances within the gallery forests (NÖ 2013).

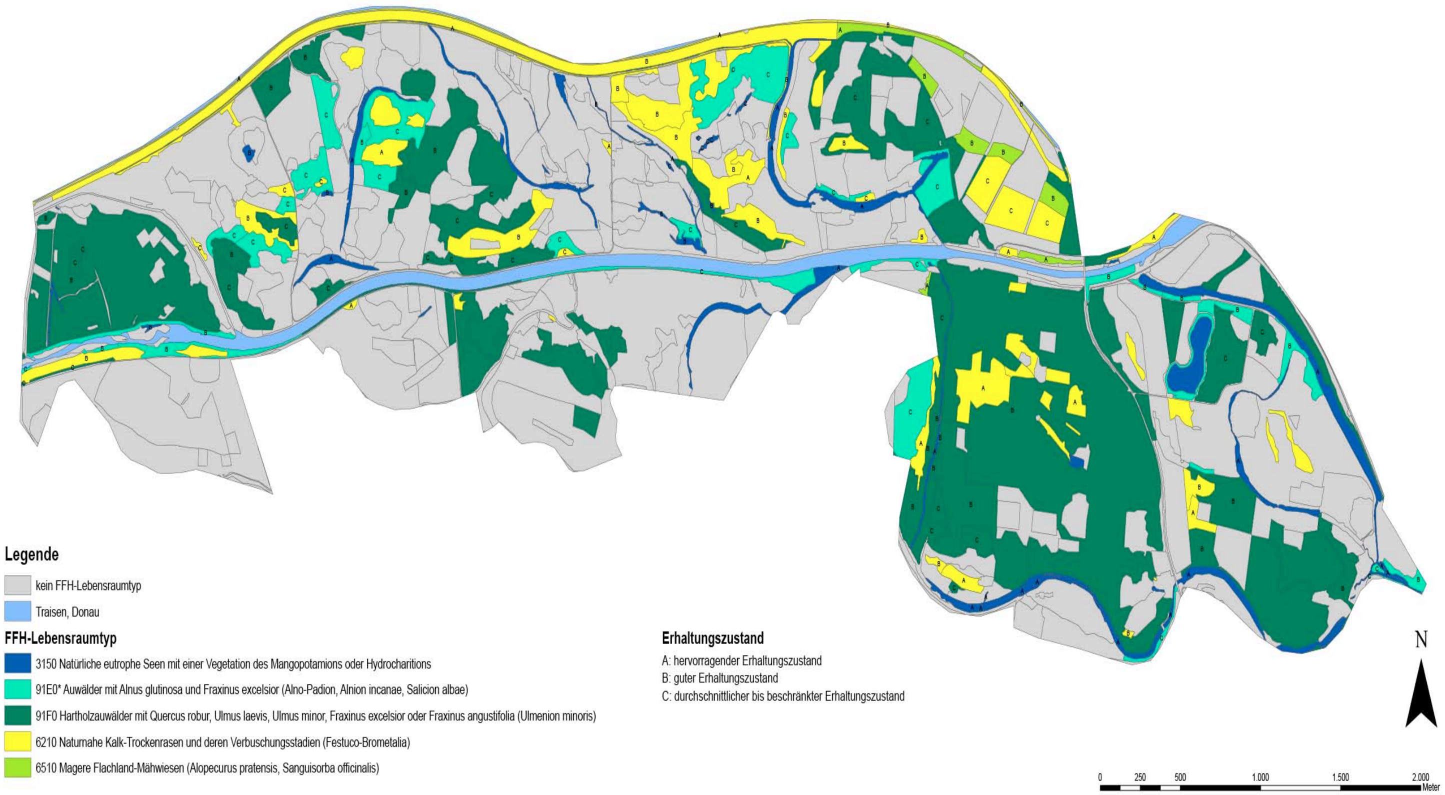


Figure 4 Map of the occurring FFH-habitat types in the study area. The FFH habitat type 3150 - Natural euthrophic lakes with Magnopotamion or Hydrocharition-type vegetation, 91E0 - alluvial forests with a linkage to alder (Alnus glutinosa) and ash (Fraxinus excelsior), 91F0 - hardwood alluvial forests with a linkage to Quercus robur, Ulmus laevis, Ulmus minor, Fraxinus excelsior oder Fraxinus angustifolia (Ulmenion minoris), 6210 - Semi-natural dry grasslands and shrubland facies on calcareous substrates (Festuco-Brometalia) and 6510 - 6510 Lowland hay meadows (Alopecurus pratensis, Sanguisorba officinalis) are recorded in the study area (Umweltverträglichkeitserklärung, LIFE+Lebensraum, 2010*

2.2 Restoration project Life + Traisen

The main aim of the project Life+ Traisen is the improvement of the hydro ecological condition of the downstream section and the outlet of the river Traisen. The planning of the project follows instructions of the European Water Framework Directive. Therefore the aim of the revitalization is to reach the “good ecological status”. The restoration of the artificial river bed is based on historical informations (Leopold & Wolman 1957; Eberstaller et al. 2000).

The plan is to build a totally new river bed of the river Traisen form kilometer no.7.5 and the outlet into the Danube. The building plan is separated into four phases of construction. The total length of the new, renaturalized downstream section of the river Traisen is 12.5km (Eberstaller 2005).

Besides hydrological and hydrobiological targets, it is one sub target to create natural riparian habitats with characteristic vegetation structures. The construction of a new cross riverbed profile is the main instrument to reach this sub target. In order to support autoecologycal processes of fauna and flora a flat lowering is planned, which is calculated to be 1.5m higher than the soil (0.3m). The cross riverbed profile of the future river bed will measures 200m in average (max. 275m). The new project aims to revitalise and restore the natural riverine habitats along vertical and longitudinal river structures. The actual cross riverbed profile measures 130m. The construction of falter and larger cross riverbed profiles supports mainly the FFH habitat types of softwood forests with populations of S*alix alba, Salix purpurea* and *Salix viminalis* and the habitat type grey alder forests with populations of *Alnus incana* and *Fraxinus excelsior* (Eberstaller 2005).
A part of the renaturation of the river bed it is planned to build rarely flooded ponds of the floodplain. The construction of these ponds leads to a higher diversity of aquatic habitats in the project areas.

According to the revitalization of the downstream section and the outlet of the river Traisen, clear improvements of the habitat situation are expected. Furthermore the project´s intention is to support the establishment of native riverine fauna and flora for the river (Eberstaller 2005). For the planned proposal a number of permits and authorizations were needed. The environmental impact assessment (UVP) that needs to be performed for certain relevant public and private projects to value the environmental impact on the proposal (Jahnel 2008) was evaluated positively.

Figure 5 Construction measures of the resoration project LIFE+ Traisen

2.3 Studied invasive alien species

The studied alien species were selected following the time of introduction and establishment. Non-native species that were introduced to Europe after 1492 are counted as alien species (Sukopp 1969; Adler & Fischer 1994, Essl et al. 2002). A main characteristic of the studied species is the ability to invade native habitats. The massive occurrence of this invasive alien species leads not just to an eradication of native species, but also to an endangerment of the stability of ecosystems and serious changes of habitat structures (Essl et al. 2002).

In the study area, eight IAS (*Acer negundo, Ailanthus altissima, Bunias orientalis, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera, Impatiens parviflora, Solidago gigantea*) are recorded in different occurrences. According to the target of the restoration project the main focus of the experiments and analysis was on the invasive species of the herb layer. Specifically *Bunias orientalis, Impatiens glandulifera, Impatiens parviflora* and *Solidago gigantea* are at the center of this research.

Impatiens parviflora DC.

I*mpatiens parviflora DC.* (Balsaminaceae) is one of the most invasive alien plants in European temperate forests (Trepl 1984; Pyšek et al. 1998; Chmura & Sierka 2007; Vervoort & Jacquemart 2012). *Impatiens parviflora* is an annual plant and native to central Asia, and was documented for the first time in Europe in 1831, when it was planted in the botanical garden in Geneva (Coombe 1956). Today *I. parviflora* occurs in shaded and humid forests and riparian woodlands (Obidzinski & Symonides 2000; Perglová et al. 2009). Within these habitat types, *I. parviflora DC* is able to adapt rapidly to various environmental conditions and invades native plant communities. The spread of *I. parviflora* DC is supported by anthropogenic disturbance such as forest management work (FaliNski 1998, Vervoort & Jacquemart, 2012).

Figure 6 Monotypic stand of Impatiens parviflora

Figure 7 Flower of Impatiens parviflora.

Figure 8 Seeds of Impatiens parviflora

Impatiens glandulifera Royle

Impatiens glandulifera Royle (Balsaminaceae) is considered to be one of the tallest (up to 3m) and most dangerous annual invasive alien plants in Europe (Willis and Hulme 2004; Hulme and Bremner 2005). *I. glandulifera* is native to the western Himalayas and became naturalized in Northern and Central Europe in the nineteenth century (Weber 2003). The global spread of *I. glandulifera* has a strong negative impact on natural vegetation - specifically in riparian areas, populations of *I. glandulifera* cause a reduction in species richness (Hulme and Bremner 2005; Hejda and Pyšek 2006). Apart from changes in vegetation coverage and seed bank composition, the nectar-rich flowers of *I. glandulifera* are also changing the preferences of pollinators (Chittka and Schürkens 2001).

Figure 9 Root and shoot axis of Impatiens glandulifera

Figure 10 Monotypic stand of Impatiens glandulifa along the rith river side of the river Traisen.

Figure 11 Seeds of Impatins glandulifera.

Figure 12 Flowering impatiens glandulifera.

Solidago gigantea Aiton

Solidago gigantea Aiton (Asteraceae) is a perennial invasive alien plant native to North America and was introduced in Europe as an ornamental garden plant around 1850 (Weber 2002; Kowarik & Gressel 2005). Today *S. gigantea* is one of the most common invasive alien plants in wetlands, meadows, forestations and river banks (Weber 2003). Due to the long blooming time of its numerous yellow flower heads in late summer and autumn, *S. gigantea* is still a very popular ornamental garden plant (Essl et al. 2002; Walter et al. 2005). Because of its fast clonally growth, *S. gigantea* competes very successfully with native species and once established its dense stands exclude almost all other plant species. The eradication of such dense stands is neither realistic nor affordable (Jakobs et al. 2004; Meyer et al. 2005).

Figure 13 Monotypic stand of Solidago gigantea in the east part of the study area.

Figure 14 Flowering Solidago gigantea

Figure 15 dry flower blossoms of Solidago gigantea in early spring

Figure 16 Seeds of Solidago gigantea

Bunias orientalis L.

*Bunias orientalis L. (Brassicaceae) i*s native to south-west Russia and was introduced to Central Europe in the eighteenth century. Today it is invasive in anthropogenically disturbed habitats in lowland limestone regions (Dietz et al. 1999). The life-cycle characteristics of this semi-rosette, polycarpic perennial plant have led to its rapid expansion and high colonization rates (Dietz and Ullmann 1997; Dietz et al. 1999). In various studies, *B. orientalis* shows a well-developed resistance to mowing and soil perturbation (Steinlein et al. 1996).

Figure 17 Monotypic stands of Bunias orientalis along the artificial embankment of the river Traisen in spring.

Figure 18 Flowering Bunias orientalis in spring

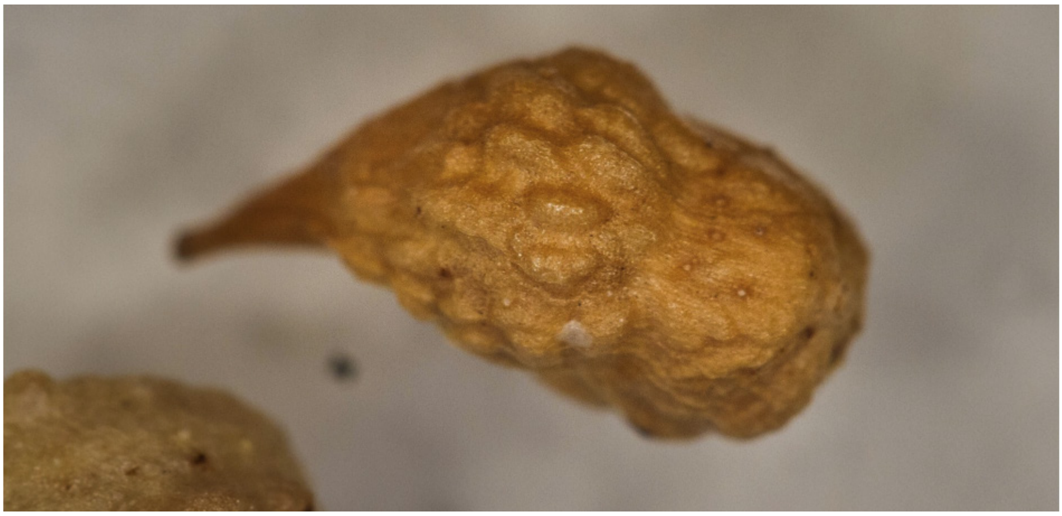


Figure 19 Seed of Bunias orientalis

Acer negundo L.

A. negundo L. (Aceraceae) is one of the most invasive plant species in riparian areas. This tree reaches the since of two to 25 meters tall, originally growing in North-America, where the distribution was limited to riparian zones. Due to human use and modification of riparian ecosystems, *A. negundo* invaded riparian ecosystems all over the world. In 1688, *A. negundo* was introduced to Europe for the first time (Oberdorfer 1994). The female tree produces approximately 20,000 seeds every year, which are distributed by the wind (Floraweb 2013). Trees become fertile very young (after 5 years) compared to native tree species.

Figure 20 Leafes of Acer negundo in autumn

Ailanthus altissima (Mill.) Swingle

The tree species *A. altissima* (Mill.) Swingle (Simaroubiaceae) is a very modest species and invasive to urban and riparian areas. *A. altissima* originates from Northern China and was planned for the first time in 1751 in England. Since then, *A. altissima* has successfully invaded riparian areas, ruderal ecosystems and urban habitats. Semi-natural dry grasslands are endangered by the invasion of *A. altissima* (Floraweb 2013). In urban areas *A. altissima* roots cause serious damage to buildings (Sheppard, Shaw et al. 2006). The abrasions can cause cardiac problems (Bisognano et al. 2005).

Figure 21 Young Ailanthus altissima on the edge of semi-dry grassland

Robinia pseudoacacia L.

R. pseudoacacia (Fabaceae) is an invasive tree species from North America. *R. pseudoacacia* is a pioneer species and have been planted in England since the seventeenth century. Fast growing and durable wood are interesting benefits for forestry utilization. Early maturity and distribution of seeds by the wind are also characteristic of *R. pseudoacacia*. Seeds show a long vitality, but for a successful germination the seeds need light. The invasion is often supported by the rapid vegetative reproduction. *A symbiosis* with nodule forming bacteria leads to a change of the soil nutrients availably and further to changes of whole ecosystems. Today the control of *R. pseudoacacia* is an important target for the management of protected areas across Europe (Hulme & Drake 2009).

Figure 22 Flowers of Robinia pseudoacacia

Rudbeckia laciniata L.

Rudbeckia lanciniata (Asteraceae) is an escaped ornamental plant and invasive to habitats in the proximity to running water. *Rudbeckia lanciniata* is native to North America. Since the seventheenth century, *R. lanciniata* has been planted in European botanic gardens. Since the 1980s the perennial invasive plant is reported as expanded to natural and semi-natural ecosystems. *Rudbeckia lanciniata* overwinters successfully in long creeping rhizomes. Its height, ranging from 50 to 250cm, supports the invasive ability of *R. lanciniata* (Francírková et al. 2001).

Figure 23 Flowers of Rudbeckia laciniata and Solidago gigantea

2.4 Vegetation sampling

The vegetation relevés includes all occurring vascular plant species. The plant species *Acer negundo, Ailanthus altissima, Bunias orientalis, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera, Impatiens parviflora, Solidago gigantea* are counted as invasive alien plant species (Essl et al. 2002). To assess the quality of plant diversity and vegetation coverage, a grid of coordinates was placed over the research area. A grid of 142 recording surfaces (plot) was chosen. All recording surfaces have a minimum distance of 300m between each other. Each coordinate of the grid is the north-western corner with a recording surface (10m x 20m) of 200m^2. The aboveground vegetation cover of each plot was sampled twice (early May and mid-August) in the years 2011, 2012 and 2013. The plots were marked and found again using a GPS readout (Geräts GPSMAP® 76CS, Garmin®). The coverage was collected within four different layers (Table 1). The scientific nomenclature of all plant species follows the classification Book "Exkursionsflora für Österreich, Lichtenstein und Südtirol" (Adler and Fischer 1994). The following parameters were recorded within the vegetation mapping in the years 2011, 2012 and 2013: habitat type, plant cover of occurring species, number of species and site conditions. The cover (%) of all vascular plant species were recorded and estimated following the adapted Braun-Blanquet scale (Table 2) (Braun-Blanquet 1964).

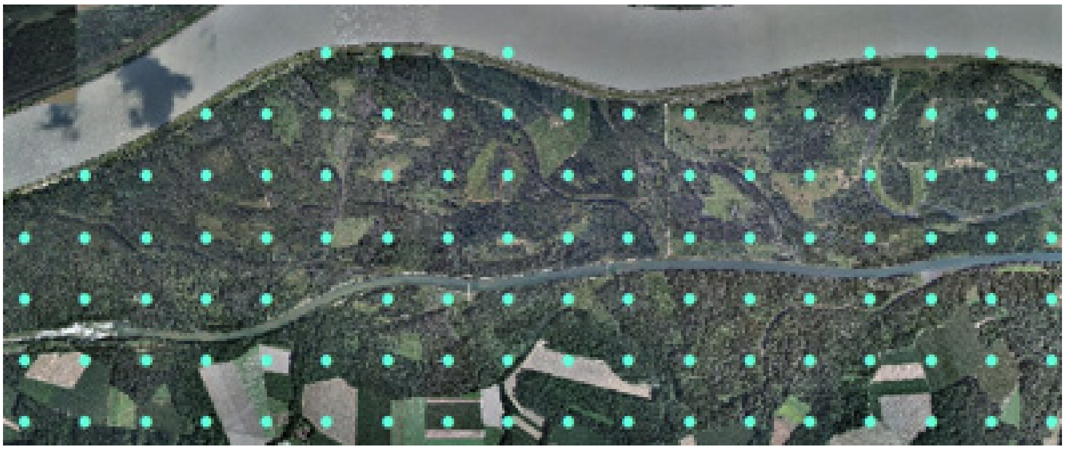


Figure 24 Schematic illustration of the grid of coordinates that was placed over the research area, used for the random selection of 142 recording surfaces

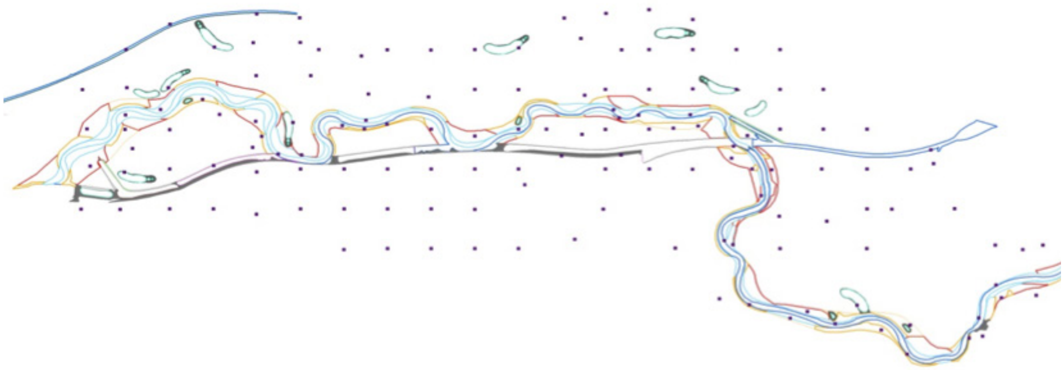


Figure 25: Distribution map of 142 plots regarding the contraction measures plan of the year 2010. The grid of coordinates was adapted due to technical modifications in the field..

Table 1: Layers of recorded vegetation coverage.

Level (in meters)	Appellation	Short form
0 to 1	Herb layer	KS
1 to 2	Shrub layer	StS
2 to 5	Tree layer I	BS I
5 to 20	Tree layer II	BS II

Table 2: Braun-Blanquet- scale: Five- part scale used for the mapping of invasive alien plants in the study area. Proportion of invasive species on the total vegetation coverage (1964).

Symbol	coverage
R	rare, 1-3 individuals
+	< 1% coverage
1	1-5% coverage
2	5-25% coverage
3	25-50% coverage
4	50-75% coverage
5	75-100% coverage

2.5 Invasive alien plant mapping

As a part of the observation and monitoring of the development of invasive alien plants in the study area, mapping was performed in the whole study area in June 2010 and June 2013. The mapping of the development of *Impatiens glandulifera*, *Impatiens parviflora, Solidago gigantea* and *Bunias orientalis* concentrates on the development of monotypic stands. Therefore an area was mapped as invaded, when the cover of the invasive species in the herb layer was higher than 50%. The exposition of the results was illustrated using the program ArcGis (Version 9.2, ESRI, 2008).

2.6 Soil seed bank

The seed bank analysis was performed on 89 recording surfaces. At the end of October 2011, the sediment was sampled five days prior to the natural stratification starting. At each site three plots were randomly taken using a soil cutter with a diameter of 5.2cm and a depth of 20cm. The plots were collected separated from the soil layer 0-5cm, 5-10cm and 10-20cm. All three plots of each soil layer were mixed to randomly taken soil layer samples (n=267) of the site and soil layer. The samples were stored in the dark (4°C) for no more than three weeks in plastic bags before analyzing. For assessing the size and composition of diasporas, a rinsing method was used (Bernhardt et al. 2008). The samples were washed with an automatic sieve system with consecutive mesh widths of 2mm, 1mm, 0,8mm, 0.5mm and 0.2mm. The washed and dried fractions were analyzed under the microscope or binocular. The reference collection at the Institute of Botany at the University of Applied Life Sciences and Natural Resources, Vienna was used to determine the species of the seeds.

In order to detect the number of seedlings emerging and their germination rate, a germination test was done after using the rising method. The detected seeds of each sample were put in Petri dishes, which were filled with absorbent papers, and kept for 40 days in the greenhouse without artificial day-night-control (temperature: 20–25°C; humidity: 60-90%). The seed was counted as germinated by the appearance of the radicula (Bernhardt et al. 2008).

In addition to the analysis of seed density in different soil layers, it was of practical interest to detect the contamination of soil with seeds of invasive alien plant species. Therefore, the seed abundance was analyzed for each constructional measure. All plots were classified into six constructional measures (road (n=10), standing water (n=7), buffer zone (n=14), HQ1 riverbed (n=14), and lowering (n=20), silting area (n=20) and not influenced area (n=15) that will take place in the order of the river's renaturation.

Figure 26 Soil cutter with a diameter of 5.6cm and a depth of 20cm.

2.7 Phenological observation

During the vegetation period of the year 2012 the vegetative and generative phenology was observed at eight sites in the study area. For a representative observation of the phenological characteristics of invasive plant species in the riparian area, the eight chosen sites were representative of the riparian area in terms of vegetation structure and biotoptype (tall herbaceous vegetation, embankment, European ash tree forest and poplar forest). A total number of four stands of *Solidago gigantea*, two populations of *Bunias orientalis,* two populations of *Impatiens glandulifera* and one population of *Impatiens parviflora* were observed thirteen times during the vegetation period of the summer of 2012. To detect the phenological development we used a vegetative (tab. 4) and a generative (tab. 5) scale (Dierschke 1989). Each scale consists of 12 stages, which describes a development stage. Besides the symphenological observation, the vegetation coverage and biotope types of the sites were documented. As such, the Braun-Blanquet-scale (Tab.4) was used. The weather parameter temperature (mean average temperature in "Grad Celsius") and precipitation (daily mean rainfall in "mm") were recorded in the time of the phenological observation. The climatic data were obtained from the nearest station "Tulln-Langenlebarn" (N 48.32°, EO16.12°). The data is provided from the "Central Institute for Meteorology and Geodynamics" (ZAMG, Hohe Warte 38, 1190 Vienna). For the analysis, the mean of the climate data type (temperature and precipitation) of each observation day was recorded. For the analysis the free software environment for statistical computing and graphics "R" (32-bit) was used (R Core Team, 2012).

Table 3: geographical location and description of the 8 observed sites.

ID	EO-coordinate	N-coordinate	river bank	road	Biotoptype
011/002	15 49,833	48 22,390	1	0	Tall herbaceous vegetation
06/012	15 50,491	48 22,196	1	1	Embankment
06/010	15 51,849	48 21,470	1	0	Tall herbaceous vegetation
76/248	15 49,304	48 22,146	1	1	European ash tree forest
77/249	15 49,555	48 22,216	1	1	Embankment
06/007	15 52,194	48 21,646	1	0	Poplar forest
004/004	15 50,274	48 22,312	1	1	Embankment
06/004	15 52,492	48 21,762	0	0	Tall herbaceous vegetation

Table 4: Vegetative scale.

Stage	Criterion of developmentstage
0	No shoot above ground
1	Emerging shoot without leaf development
2	First leaf developed (or: < 25%)
3	2-3 leaves developed (or: < 50%)
4	More than 75% of the leaves developed
5	Almost all leaves developed
6	(Vegetative) Fully developed plant
7	Start of yellowing, pedicel yellowed
8	Up to 50% turned yellow
9	More than 50% turned yellow
10	Above ground died back
11	Above ground disapeared

Table 5: generative scale.

Stage	Development stage
0	No flower bud
1	Flower bud visible
2	Flower bud clearly visible
3	Shortly before flowering
4	Ongoing flowering
5	Up to 25% bloomed
6	Up to 50% bloomed
7	Total blooming
8	Finish blooming
9	Totally finished blooming
10	Maturity of fruit
11	Distributing seeds

2.8 Germination Experiment

The aim of this experiment is to compare the germination process of native and invasive alien species in order to detect the advantages and disadvantages of species within the invasion of riparian areas. The results are valuable for the management of invasive species in protected areas or for the monitoring of resoration projects.

Seed collection

Seeds of the invasive alien species *Impatiens glandulifera, Impatiens parviflora, Solidago gigantea* as well as of the native species *Eupatorium cannabinum, Senecio sarracenicus, Tanacetum vulgare, Urtica dioica,* and *Salvia glutinosa* were collected in October 2012, when all seeds had matured. Only seeds of *Bunias orientalis* were already collected in August, because they mature earlier. All seeds of each individual species were collected in the floodplains of the river mouth of the river Traisen in Lower Austria. The collecting sites of each species represent typical habitats for those species in riparian areas.

Table 6: Overview of the collected native and invasive alien species.

Species	family	origin	Status	Number of Individuals	N-Coordinate	EO-Coordinate
Impatiens parviflora	Balsaminaceae	Western Asia	invasive	100	N48°21 550	EO 15°48 858
Impatiens glandulifera	Balsaminaceae	Himalaya	invasive	100	N48°22 216	EO 15°49 555
Bunias orientalis	Brassicaceae	Western Asia	invasive	177	N48°22 222	EO 15°48 881
Solidago gigantea	Asteraceae	North America	invasive	122	N48°21 550	EO 15°48 858
Eupatorium cannabinum	*Asteraceae*	Europe	native	110	N48°21 550	EO 15°48 854
Senecio sarracenicus	Asteraceae	Europe	native	150	N48°22 637	EO 15°50 514
Tanacetum vulgare	Asteraceae	Europe	native	88	N48°21 550	EO 15°48 858
Urtica dioica	Urticaceae	Europe	native	120	N48°21 550	EO 15°48 858
Salvia glutinosa	Lamiaceae	Europe	native	100	N48°21 550	EO 15°48 858

Eupatorium cannabinum L. (Asteraceae) is native to Central Europe and common in European riparian areas (Adler & Fischer 1994). It is a perennial herb, growing from 60cm to 1.5m, and prefers moist habitats, like floodplain forests or areas along stream banks with a high nutrient structure (Grieve 2005), where it can grow in dominant monotypic stands. Outside its native habitat in Europe, *E. cannabinum* is considered an invader into riparian zones (www.issg.org 17.05.2013; Clarkson 2003).

Senecio sarracenicus (Asteraceae) is a perennial plant native to riparian habitats in continental Europe and Asia (Oberdorfer 1962). It also grows along roads and other ruderal and disturbed habitats (Muller 2004). Although *S. sarracenicus* is quite competitive, it is an endangered species in many of its native habitats in Austria and Germany (Korneck and Sukopp 1988; Adler and Fischer 1994; Muller 2004).

Tanacetum vulgare (Asteraceae) is a robust perennial plant native to continental Europe and Asia. It grows in moist habitats, riparian and highly disturbed areas, along river banks, roads, and lowland pastures, and forms large colonies as a result of its spreading rhizomes. Due to its pharmacological characteristics, *T. vulgare* was introduced into North America, where it now must be managed as an invasive plant species (www.fs.fed.us 17.05.2013; www.issg.org 17.05.2013; Adler and Fischer 1994; Jacobs 2008).

Urtica dioica (Urticaceae) is a perennial plant native to the entire northern hemisphere. It occurs in ruderal and disturbed areas, gardens and riparian areas and is also used as a nitrogen indicator. It is very common and not endangered, and grows in large compact communities, with the size of individuals ranging from 30 to 150cm. The successful spread of *U. dioica* is facilitated by its strong vegetative speading via underground rhizomes (floraweb.de 05/15/2013; Korneck and Sukopp 1988; Adler and Fischer 1994).

Salvia glutinosa (Lamiaceae) is a native perennial plant native to Central Europe. It shows a strong occurrence in moist forests and riparian flood plain forests and is not endangered. *S. glutinosa* grows to between 30 and 120cm tall; its epizoochoric distribution is supported by sticky flowers and persistent seeds (floraweb.de 05/15/2013; Korneck and Sukopp 1988; Adler and Fischer 1994).

(I) Germination requirements

The seedlings were germinated in Petri dishes lined with filter paper. Each treatment group consisted of two replicates of 50 seeds (2x50 seeds) for each species. The seeds were kept moist using distilled water and the germination process was observed every other day. Two situations were simulated in the climate chamber:

Situation A (15/5): 12h/12h light, 15°C/5°C temperature, relative humidity 80%;
Situation B (25/10): 12h/12h light, 25°C/10°C temperature, relative humidity 80%;

(II) Seedling emergence

A seedling was counted as "positive" after the emergence of a radicle from the seed coat. Seedling emergence was measured as a percentage (%) and time to emergence of seedlings that emerged in the course of the experiment. The first day of the seeds being stored in the climate chambers was considered the starting day of the experiment.

(III) Seedling development

After the emergence of a radicle from the seed coat, the germinated seeds were taken from the Petri dishes and planted in pots filled with a mixture of sterile sand and soil. The germinated seedlings were then grown in the climate chambers under the same original simulated conditions (15/5 and 25/10) and their development defined in four different stages:

Table 7: Developmental stages of seedlings

No.	Stage of development
1	Yellow cotyledons, radicle visible
2	Cotyledons start emerging
3	Cotyledons are mostly green, above the ground and hanging on the curvature of the hypocotyl
4	Cotyledons are above ground and fully green; unifoliolates emerging

(IV) Effect of dry storage and cold-wet stratification

One part of the seeds was germinated after a period of dry storage. These seeds were stored in a dry and dark place at room temperature for three months (November, December and January). The other part of the seeds was germinated after a period of cold stratification. These seeds were stratified for three months in a dark and wet place at 5°C.

(V) Vitality of seeds

Furthermore, the vitality of ungerminated seeds was determined. Ungerminated seeds were checked for viability using the tetrazolium test: embryos were dissected and placed in a 0.1% solution of 2,3,5-triphenyl-2H-tetrazolium chloride (TTC), a quaternary ammonium salt and Redox-indicator displaying a change of color from colorless to red caused by a Redox reaction. In this process the tetrazolium cation of the dehydrogenase in the respiration chain is reduced to red TPF (1,3,5-triphenylformaza) (Rich et al. 2006). After a certain period of the germination test the seeds that did not germinate were cut lengthways into two pieces. The bigger half was put into a 1 % solution of 2,3,5-triphenyl-2H-tetrazolium chloride (TTC) in distilled water. After 6 hours in the dark and a temperature of 30°C the seeds were checked. By this time any viable embryos would have turned pink and could thus be declared dormant.

2.9 Data analysis

Species composition and vegetation development

The collected data were assembled with the program Hitab 5 (Wiedermann 1995). The analysis of the vegetation sampling data was analyzed using Microsoft Excel 2010. Statistical computing and graphics "R" (32-bit) was used (R Core Team, 2012).

For analyzing differences in susceptibility to invasion among habitat types, invaded (n=87) and not ivaded (univaded) plots (n=55) of the year 2013 were compared for each invasive plant species. "Invaded" plots with a percentage higher than 60% of total plant cover were selected for this analysis compared to "not invaded" plots with a percentage lower than 5% of total plant cover.

Introduction mechanisms and habitat preferences

The normality was tested on the model residuals using the Shapiro-Wilk test (Shapiro & Wilk 1965; Shapiro & Francia 1972). According to the Shapiron-Wilk test the data of vegetation sampling did not follow a normal distribution ($P < 0.05$). Therefore the Pearson correlation was used to analyze the relationship between species number, plant cover, habitat types and cover of IAS. The size of Pearson product-moment correlation coefficient is evaluated as follows: ($r < 0.50$) very low, ($r = 0.51$ to 0.89) low, ($r=0.80$ to 0.89) moderate and ($r >0.90$) high (Lee Rodgers and Nicewander 1988; Derrick et al. 1994). The Pearson correlation is recommended for landscape analysis and species composition correlations (Gustafson 1998; Deutschewitz et al. 2003; Raes et al. 2009).

Using the nonparametric Mann-Whithey-Wilcoxon Test (Mann–Whitney, Wilcoxon paired sample, or Spearman rank correlation), the significance of the difference among independent sample parameters was calculated, without assuming them to follow the normal distribution (Royston 1992; Cheung & Klotz 1997; Öztürk & Wolfe 2000). Differences among the following classification groups were tested: road proximity (n = 2 categoreis), open water body proximity (n = 2 categories), cultivation intensity (n = 3 categories) and tree canopy (n = 2 categories).

Table 8 Classification for the analysis of the influnce of road proximity.

No.	Road proximity	Description
1	On the roadside	Plot is directly infuenced by a road or in max. 2m distance
0	Not on the roadside	Plot is not directly infuenced by a road or at least max. 2m away

Table 9 Classification for the analysis of the influnce of open water body proximity.

No.	Open waterbody proximity	Description
1	On the edges of water bodies	Plot is near by a water body or in max. 2m distance
0	Not the edges of water bodies	Plot is not near by a water body or at least max. 2m away

Table 10 Classification for the analysis of the influnce of cultivation intensity.

No.	Cultivation intensity	Description
1	Near-nature	No human disturbance
2	Low	Occasional land-use, human disturbances do not lead to massive habitat changes
3	Intensive	Human disturbances; change of habitat characteristics

Table 11 Classification for the analysis of the influnce of tree canopy.

No.	Tree canopy	Description
1	Canopy	Develpped tree layer, min. 75% shading, or shade of surrounding trees.
0	No canopy	nN developed tree layer, max. 5% shading, no shade of surrounding trees.

Plant diversity

In order to calculate the influence of the development of the occurrence of invasive alien plants on the native diversity, the Shannon diversity index H‘ was chosen (Washington 1984; Magurran 1988). The Pielou`s pooled quadrate method was used for estimating diversity when a random sample is not expected. It was used to calculate the Pilou´s index of evenness J (Magurran 1988). The estimated species coverage was used for calculating the H‘ and J.

$$H' = -\sum_{i=1}^{S} p_i \log p_i$$

$$p_i = \frac{n_i}{N}$$

H‘ = Shannon‘s diversity index
S = Number of species
p_i = relative abundance of one species in N

$$H_{\max} = -\log(1/i)$$

$$J = H / H_{\max} = H / \ln S$$

H_{max} = ln(S) Maximum diversity possible
i = number of species
J = equitability (evenness)

The development of H` and J of the years 2011, 2012 and 2013 of all 142 plots was analyzed descriptive and using the the nonparametric Mann-Whithey-Wilcoxon Test (Mann–Whitney, Wilcoxon paired sample, or Spearman rank correlation) to detect significant changes after normality was tested on the model residuals using the Shapiro-Wilk test. This development was analyzed among plant coverage, species number and abundance of all 8 IAS recorded in the herb layer.

Soil seed bank occurrence and phenology

To prove the significance of the relationship between the change of the mean percent of the aboveground vegetation coverage from the study year 2011 to 2012 and density of seeds of invasive alien plants in the seed bank, a Fischer`s exact test was performed (Agresti 1992). We tested for a relationship between seed occurrence and the coverage from the year 2011 to 2012 in the herb layer with 2×2 contingency tables and used Fisher's exact tests when too many cell frequencies were < 5 because of low number of values. Furthermore the relationship between the density of seeds of invasive alien plants in the seed bank and the native species richness and native evenness was calculated using a regression analysis (Magurran 1988).

Data of the germination test were not transformed as needed to improve normality using the Shapiro–Wilk test (Royston 1992). An average was calculated over the total number of seeds from each species. Several variability values included a large number of zeros. The numbers of germinated seeds and developing plants were related to the two controlled temperature conditions and the two storage environments. The effects were tested using ANOVA. A Chi^2 test was used to analyze the significance of temperature, storage effect on the number of seedlings (Greenwood & Nikulin 1996). We tested for a relationship between germination and temperature as well as germination and could-wet stratification with 2×2 contingency tables.

For each observed population a diagram of the phenological spectrum was designed. Furthermore the vegetative and generative development characteristics are analyzed with a graph of each species. For this comparison the mean of the populations of a species is pictured. For the correlation with the climate data the mean of the populations of a species was used again. For the correlation if the development stage and the mean temperature, the Pearson's product-moment correlation was used.

3. Results

3.1 Impact on species composition and structure of plant communities

The vegetation sampling of 142 recording surfaces (plots) apprehended the occurrence of 8 invasive alien plant species *Acer negundo, Ailanthus altissima, Bunias orientalis, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera, Impatiens parviflora, Solidago gigantea* that were recorded within four different vegetation layers. In 2011, the plant cover mean is the lowest with 71.3% (n=142). The mean of the plant cover in 2011 is 71.3%, in 2012 90%, and 96% in 2013. The number of 14 species per plot on average did not change over the time period of investigation. The vegetation sampling of the herb layer in 2011 showed that 52 plots (n=142) were invaded by IAS. In 2012 in total 34 plots (=142) and in 2013 in total n=87plots (n=142) were invaded.

Table 12: Results of Pearson correlation coefficient for the species number x IAS abundance and plant cover x IAS abundance in the thre study years 2011, 2012 and 2013 (n=142).

	2011	2012	2013
Species number/ IAS abundance	-0.0758	-0.2766	-0.3819
Plant cover / IAS abundance	0.28574	0.40233	0.3803

The results of the Pearson product-moment correlation coefficient shows a very low negative relationship of species number and IAS cover ($r < -0.50$) within all three years. The correlation for the total plant cover and IAS cover is very low and positively correlated ($r < 0.50$) within all three years (Table 12).

The results of the nonparametric Mann–Whitney correlation test show significant differences among total plant cover in 2011 and 2013 ($P < 0.01$) (Figure 27). This is also true for the cover of IAS in the herb layer ($p < 0.01$). The species number did not differ significantly between the years 2011 and 2013 ($P = 0.692$).

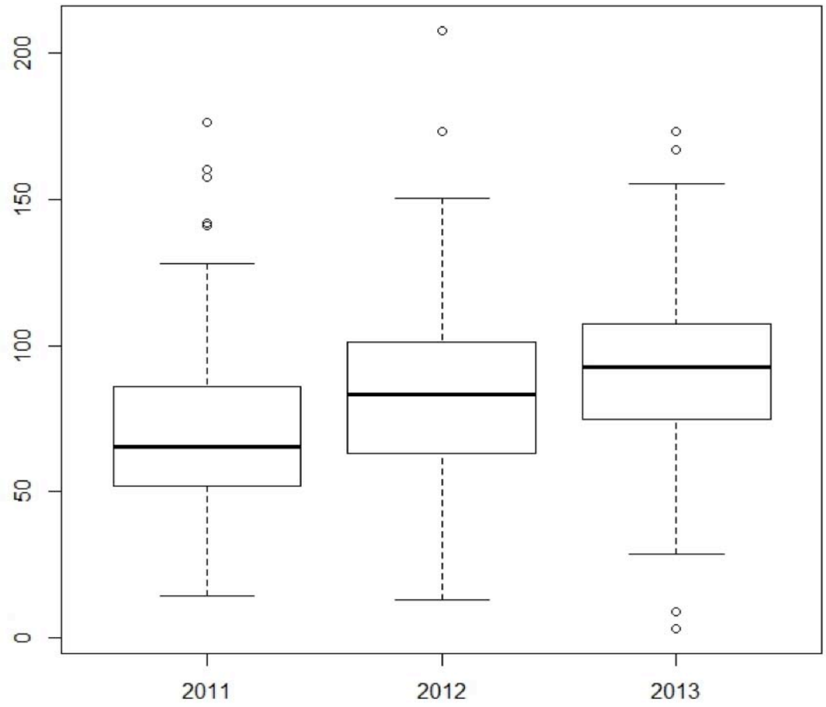


Figure 27 Total plant cover (all occurring species, invasive and not invasive plant species) of the year 2011, 2012 and 2013 of the herb layer (n=142).

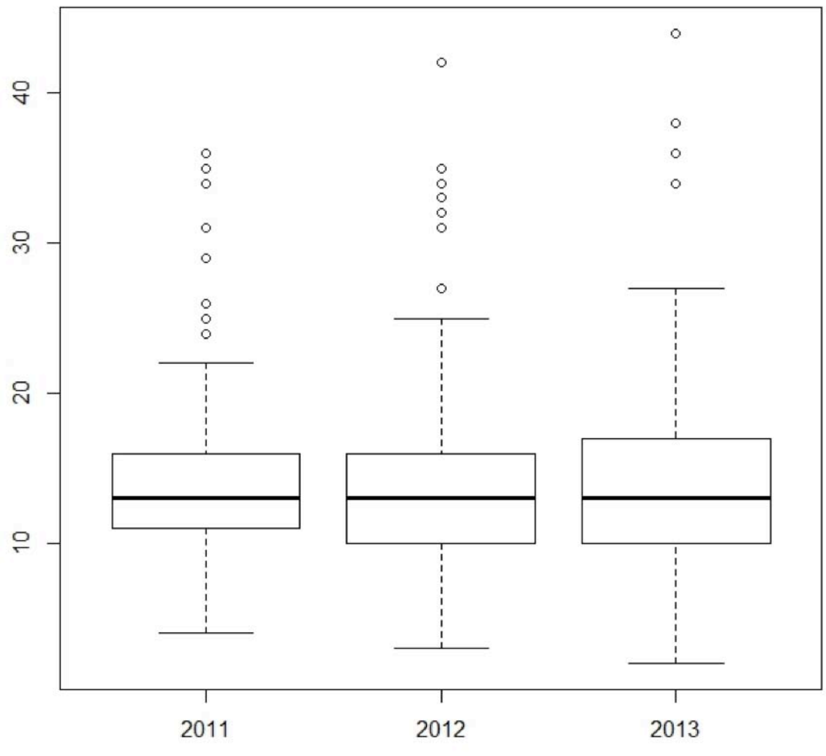


Figure 28 Total number of recorded species of the herb layer in the study years 2011, 2012 and2013 (n=142).

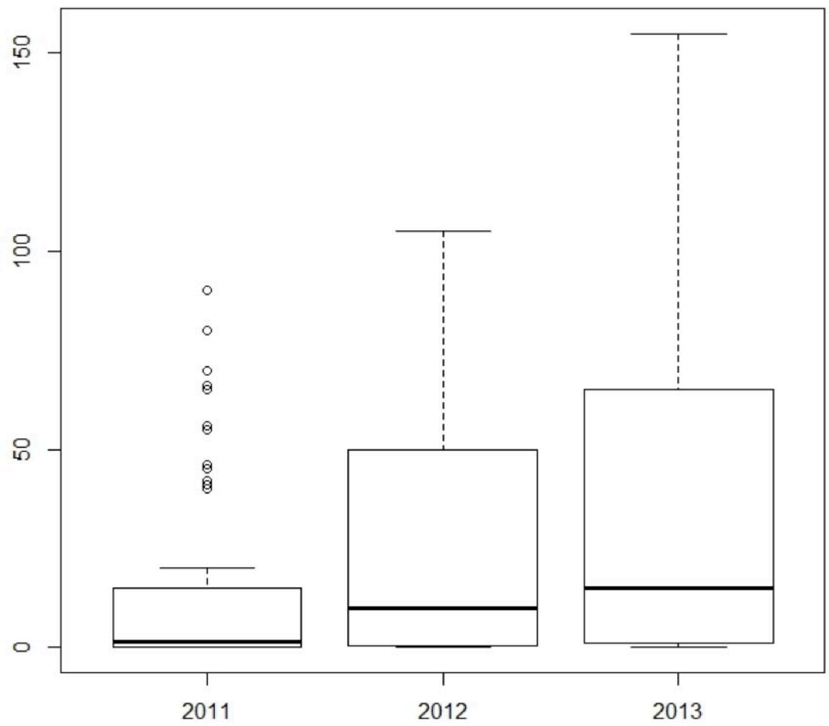


Figure 29 Sum of mean cover of IAS (Solidago gigantea, Impatiens parviflora, Impatiens glandulifera, Bunias orieintalis, Rudbekia lanciolata, Ailanthus altissima, Robinia pseudoacacia and Acer negundo) of the herb layer in 2011, 2012 and 2013 (n=142).

The invasive tree species *Acer negundo* and *Ailanthus altissima* never reached a plant over 5% (Table 13). The population did not increase within three study years. The invasive tree species *R. pseudoacacia* occurs with a rage from 0 to 15% in the years 2012 to 2011, and from 0 to 40% in the year 2013. In the year 2013, the mean abundance increased from 1.91% to 3.62%. The number of plot with occurrence of *R. pseudoacacia* increased from 9 in 2011 to 12 plots in 2013.

The most abundant invasive species is *Solidago gigantea* in all three years Table 13). The plant cover range reaches from 0 to 90%. The abundance in increasing from 8.14% in the year 2011 to 31.63% in 2013 (n=142 plots). The number of plots with occurrence of *S. gigantea* increased from 46 in 2011 to 58 in 2013 constantly. The results of the nonparametric Mann–Whitney correlation test showed low significant differences in the development of the populations of *Solidago gigantea* in 2011 and 2013 (P = 0.078).

The invasive species *Impatiens glandulifera* showed an abundance of 4.03% (n=142) in 2011 (Table 13). Its abundance increased in 2012 to 5.20% and decreased in 2013 again to the average abundance of 3.56%. According to the result of the Mann–Whitney correlation test increased from 2011 to 2013 were not significant (P = 0.887).

The abundance of *Impatiens parviflora* increased significantly from 5.43% abundance of the total plant cover (n=142) in 2011 to 17% in 2013 (P = 0.0002) (Table 13). The maximum cover rage increased from 65% in 2011 to 90% in 2012.

The mean abundance of *Bunias orientalis* increased from 0.19% in 2011 to 1.58% in 2013 (n=142) (Table 13). The maximum cover rage increased from 15% in 2011, 40% in 2012 and to 90% in 2013. The number of invaded plots increased from six plots in 2012 to eight plots in 2013. The perennial invasive species *Rudbeckia laciniata* occurred in 2011 on three plots with the maximum rage of 65% and mean abundance of 0.47%. In 2013, the maximum coverage increased to 90% and the mean abundance to 0.71% (n=142). According to the result of the Mann–Whitney correlation test increases from 2011 to 2013 were not significant (P = 0.538).

Table 13: The invasive alien plant species and in occurrence within the plant cover of the herb layer in all plots (n=142) in 2011, 2012 and 2013. The coverage shows the lowest (min.) and highest (max.) value of abundance in the total plant cover of each IAS and in total (plant cover total). The development of the abundance (S.total) is compared using the mean and standard deviation (Mean +-SD). The number of plots (n.plots**) show uneven distribution among species. The percentage of S.total* shows the decrease of increase of each species' abundance.*

2013						
	Coverage		S.total*		n.plots**	Percentage
	Min	Max	Mean	SD		of S.total*
Plant cover total (%)	3	174	90.96	28.47		
Species number	2	44	14.04	6.99		
Acer negundo	0	5	0.05	043	3	0.05
Ailanthus altissima	0	1	0.05	0.21	8	0.08
Bunias orientalis	0	90	1.58	9.10	8	2.52
Impatiens glandulifera	0	90	2.27	11.35	18	3.56
Impatiens parviflora	0	90	11.23	22.94	61	17.00
Robinia pseudacacia	0	40	0.51	3.62	12	0.71
Rudbeckia laciniata	0	90	0.71	7.57	4	1.10
Solidago gigantea	0	90	16.69	31.63	58	23.89

2012						
	Coverage		S.total*		n.plots**	Percentage
	Min	Max	Mean	SD		of S.total*
Plant cover total (%)	13.0	208.0	84.3	30.8		
Species number	3.0	42.0	13.9	6.6		
Acer negundo	0	5	0.05	0.43	4	0.06
Ailanthus altissima	0	1	0.05	0.20	8	0.07
Bunias orientalis	0	40	0.64	3.99	6	1.08
Impatiens glandulifera	0	90	3.66	13.92	21	5.20
Impatiens parviflora	0	90	7.99	19.61	58	13.47
Robinia pseudacacia	0	15	0.35	1.91	11	0.65
Rudbeckia laciniata	0	90	0.68	7.56	3	0.93
Solidago gigantea	0	90	13.34	26.37	55	20.41

2011						
	Coverage		S.total*		n.plots**	Percentage
	Min	Max	Mean	SD		of S.total*
Plant cover total (%)	14.5	176.5	71.3	29.5		
Species number	4.0	36.0	14.0	6.0		
Acer negundo	0	1	0.01	0.09	2.00	0.02
Ailanthus altissima	0	1	0.01	0.12	2.00	0.04
Bunias orientalis	0	15	0.19	1.39	6.00	0.29
Impatiens glandulifera	0	65	2.57	10.27	17.00	4.03
Impatiens parviflora	0	65	2.52	8.35	37.00	5.43
Robinia pseudacacia	0	15	0.19	1.34	9.00	0.49
Rudbeckia laciniata	0	65	0.47	5.45	3.00	0.88
Solidago gigantea	0	90	8,14	18,52	46,00	15,25

3.2 Spatial distribution

The comparison of the distribution from 2010 and 2013 show an ongoing encroachment of IAS. The stands of *Impatiens gladulifera* are related to habitats under direct influence of open water bodies. Between the years 2010 and 2013 the major spread of *Impatiens gladulifera* took place on the artificial embankments along the old river bed of the river Traisen. Semi-natural and natural habitats are not affected by the establishment of monotypic stands of *I. glandulifera.* Water body proximity and anthropogenic influence promote the expansion of *I. glandulifera.*

The increase of *Impatiens parviflora* between the years 2010 and 2013 is the highest of all invasive species in the study area. The distribution is extensively high within all forest habitats. Road proximity of anthropogenic influence does not significantly influence the distribution of *I. parviflora*. The spatial spread is extensive and rapid. Direct path ways for the invasion of *I. parviflora* are not detectable

Tabelle 14: Significance of Introduction mechanisms habitat parameters on the developnet of total cover.

	Effect	P- value
Road proximity	-	0.06232
Open water proximity	-	0.7333
Cultivation intensity		
1 vs 2	-	0.159
2 vs 3	-	0.3951
1 vs 3	-	0.5447
canopy	+	0.01026

Tabelle 15 Significance of Introduction mechanisms habitat parameters on the developnet of species number .

	Effect	P- value
Road proximity	+	0.00000001711
Open water proximity	-	0.8282
Cultivation intensity		
1 vs 2	-	0.277
2 vs 3	-	0.1522
1 vs 3	+	0.004169
canopy	+	0.004674

Tabelle 16 Significance of Introduction mechanisms habitat parameters on the developnet of cover of IAS.

	Effect	P- value
Road proximity	+	0.02939
Open water proximity	+	0.0000001085
Cultivation intensity		
1 vs 2	-	0.8027
2 vs 3	-	0.2141
1 vs 3	-	0.1454
canopy	-	0.07471

The distribution of *Bunias orientalis* affects the embankments of the river Traisen. The stands of *B. orientalis* are most successful along the area between the forest road and the left river bank of the river Traisen. In 2010, the populations were concentrating on punctual distribution. The mapping of 2013 shows an extensive spatial distribution along the left river bank. Water body proximity and anthropogenic influence are significant factors for the positive spread of *B. orientalis.*

The spread of *Solidago gigantea* in the study is related to disturbed habitats along roads, and grassland habitats, as is exemplified by tall oatgrass meadows and grassland fallows. Within these habitats the increase of population size from 2010 and 2013 was low compared to the increase of monotypic stands of *S. gigantea* within poplar tree forests. Proximity to forest roads is a significant factor for the spatial distribution of *S. gigantea.* The large monotypic of *S. gigantea* within meadow habitats, like in the east part of the study area, are mowed in the late summer period.

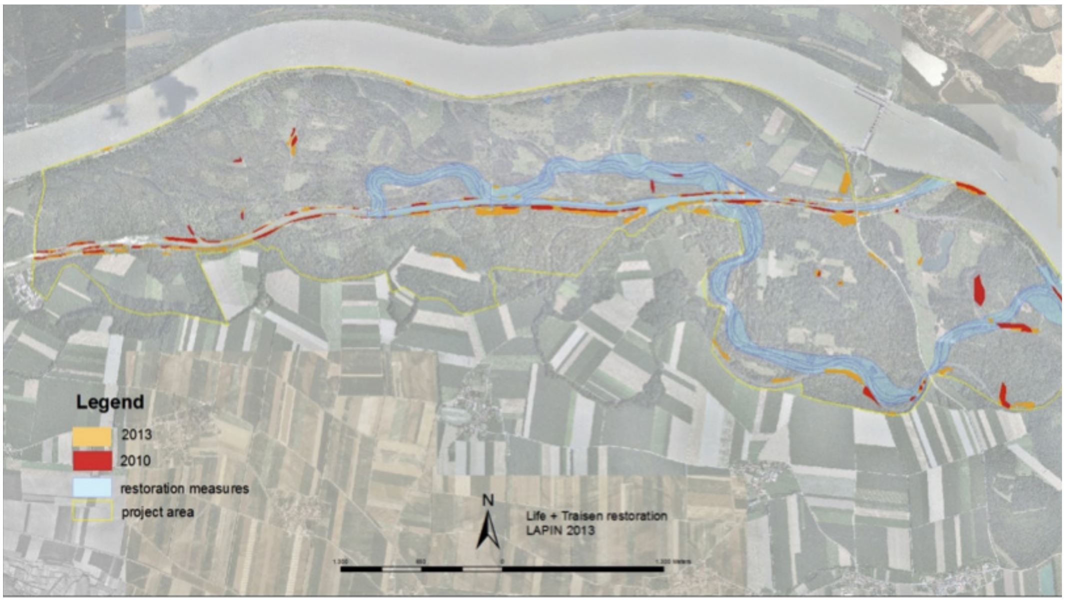


Figure 30 Developments of Impatiens gladulifera population from 2010 to 2013.

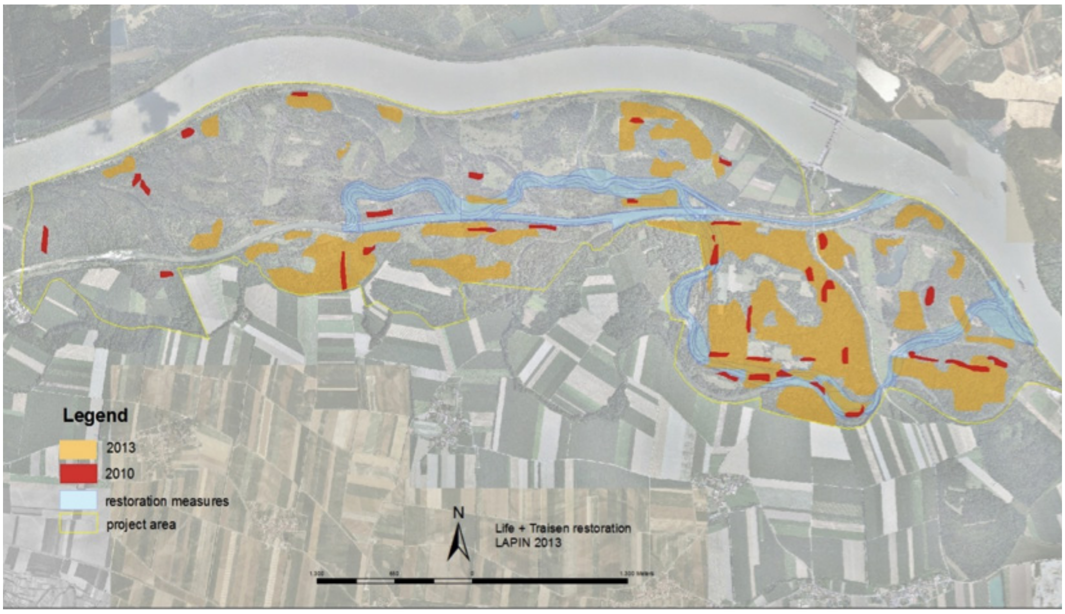


Figure 31 Developments of Impatiens parviflora population from 2010 to 2013.

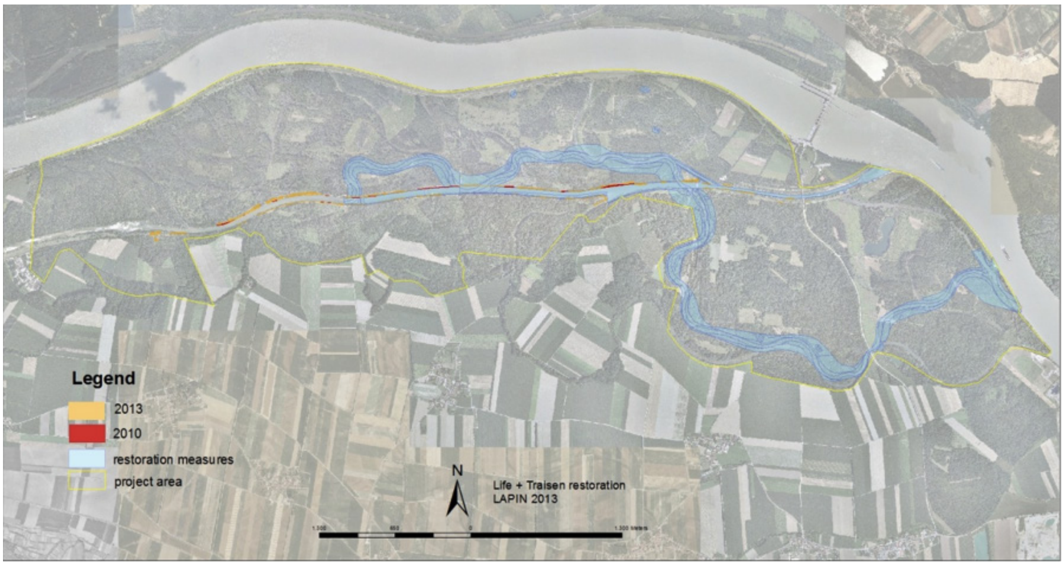


Figure 32 *Developments of Bunias orientalis population from 2010 to 2013.*

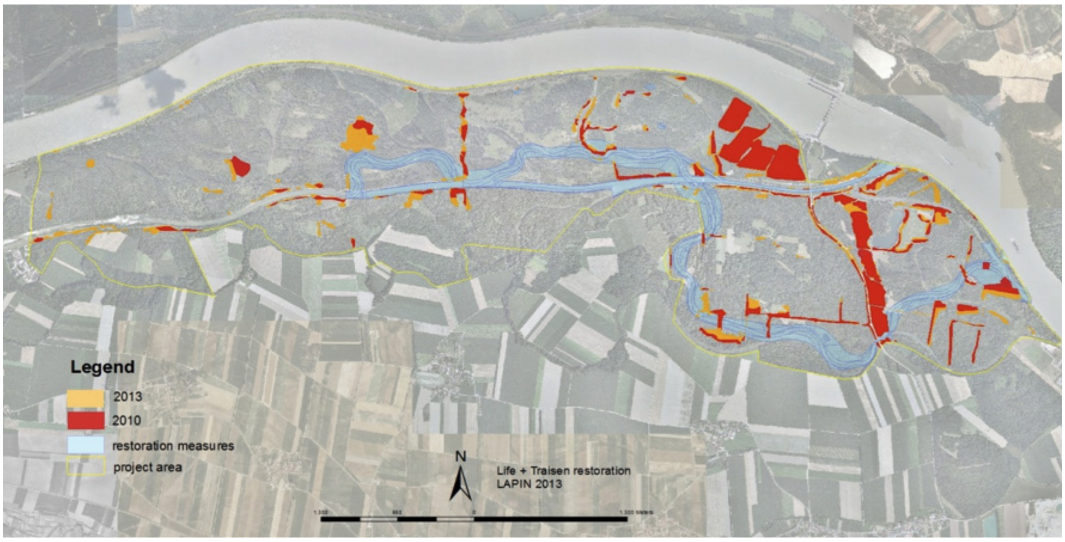


Figure 33 Developments of Solidago gigantea population from 2010 to 2013.

In total 41 plots (n=142) show a proximity to frequented forest roads. The analysis of the vegetation's development (Wilcoxon test) of plots with close proximity to direct roads showed significant differences of IAS abundance to plots not in proximity to roads between 2011 and 2013($P < 0.001$). But the species number of plots along roads decreased significantly during the three years ($P < 0.001$). The invasive species *Acer negundo* and *Ailanthus altissima* were not found along roads.

The invasive species *Solidago gigantea* (mean ± SD = 7.24 ± 14.43) had the highest abundance along roads in 2011. The results are not significant (P = 0.3563) comparing the year 2011 and 2013. The cover increased in 2013 (mean ± SD = 12.88 ± 24.76). The average plant cover of *S. gigantea* in plots with road proximity of 3.71% in 2011 and 5.12% in 2013 in the herb layer is lower compared to plots with no road proximity, where *S. gigantea* occurs with an average plant cover of 11.5% in 2011 and 18.78% in 2013.

Most significant differences among plots with and without road proximity exist for *Impatiens parviflora*. The results are significant (P = 0.001) when comparing the years 2011 and 2013. In 2011 invaded plots showed an IAS occurrence with a mean of 12.00% of plots in proximity to roads. In 2013, invaded plots with road proximity showed an abundance of average mean of 14.27%. Within the plots with road aproximity *Impatiens parviflora* shows an average total plant cover of 2.56% in 2011 and 6.04% in 2013 in the herb layer. Within plots with no road proximity *I. parviflora* occurs with an average plant cover of 3.18% in 2011 and 10.96% in 2013.

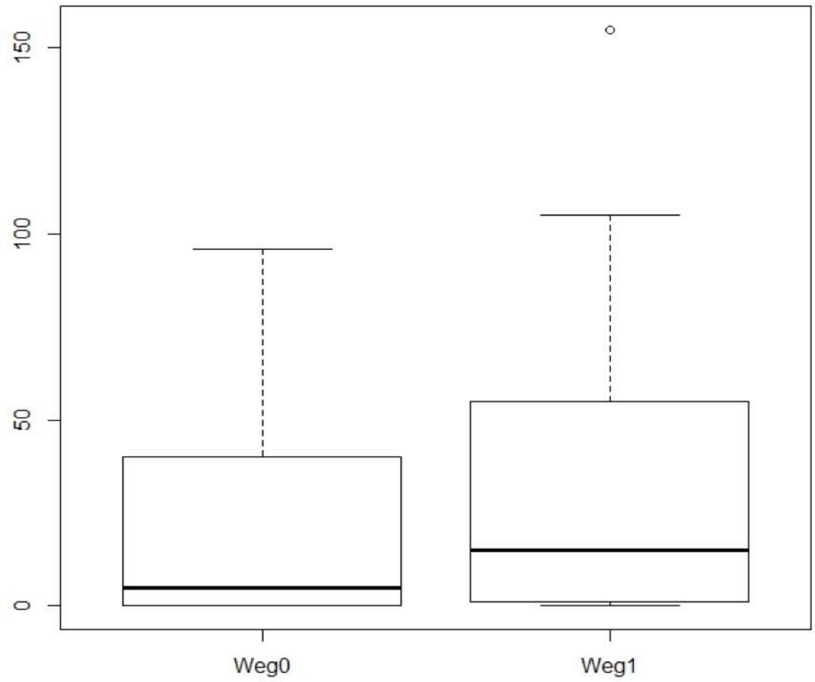


Figure 34 Total abundance of IAS of plots with road aproximity (Weg1) and without road aproximity (Weg0) in the herb layer in sum of 2011, 2012 and 2013 (n= 1263 plots).

In total 47 plots (n=142) show a proximity to open water bodies. The analysis of vegetation development of plots with a direct water proximity showed significant differences in IAS cover to plots without proximity to water (P < 0.001) among all three study years. All invasive species were found at least once in plots with direct proximity to water.

The invasive species *Solidago gigantea* (mean ± SD = 12.21 ± 22.41) had the highest cover along water in 2011. The abundance differed significantly among plots along water bodies (P < 0.001). The cover of S. *gigantea* increased in 2013 (mean ± SD = 20.98± 34.98).

Most significant differences among plots with and without water proximity are to detect for the invasive species *Bunias orientalis* (P < 0.001). In 2010, invaded plots showed an occurrence with a mean of 0.59% of plots with water proximity. In 2013, invaded plots with road proximity showed an abundance with an average mean of 4.79%. The average plant cover in the herb layer of *B. orientalis* is of 0.01% (n=142) in 2011 and 2.52% in 2013 (n=142). *B. oreintalis* was detected only along embarkments with direct water proximity.

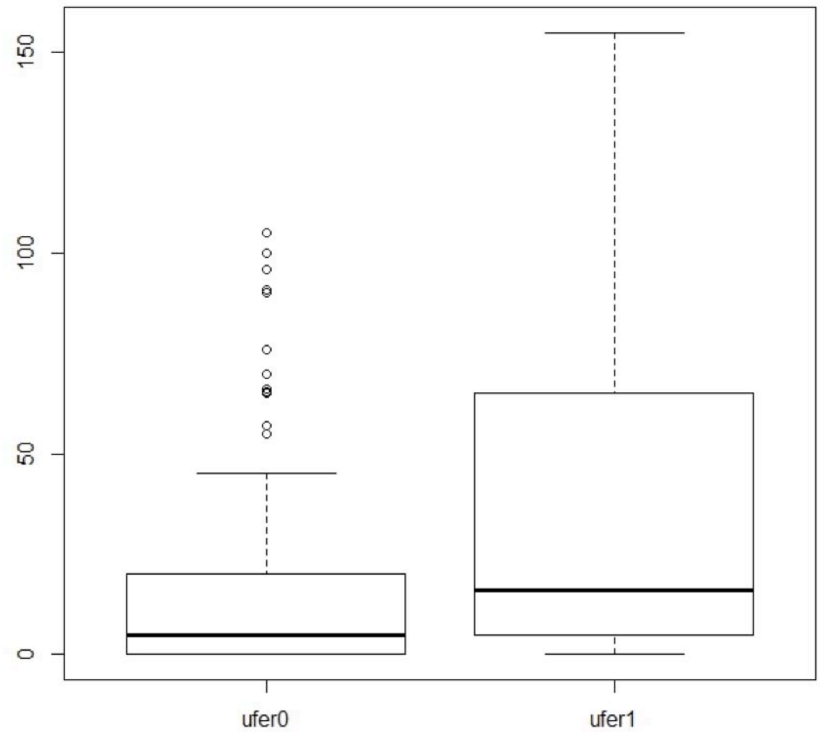


Figure 35 Total abundance of IAS of plots with water aproximity (Ufer1) and without road aproximity (Ufer0) in the herb layer in sum of 2011, 2012 and 2013 (n= 1263 plots).

Table 17: Water body proximity in 2011 and 2013. Mean and standard deviation (±SD) of abundance of IAS in 2011 and 2013 of plots with (S.water) and without (S.no.water*). direct water body proximity and average plant cover (%) and species number.*

	2011				**2013**			
	S.no.water*		S.water*		S.water*		S.water*	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Plant cover total (%)	70.14	26.44	73.61	34.99	91.04	29.00	90.81	27.67
species number	13.9	5.8	14.2	6.3	14.0	7.5	14.1	6.0
Acer negundo	0.02	0.11	0	0	0.07	0.52	0	0
Ailanthus altissima	0.02	0.14	0	0	0.05	0.21	0.04	0.20
Bunias orientalis	0.00	0.00	0.59	2.38	0.00	0.00	4.79	15.43
Impatiens glandulifera	0.92	6.84	5.91	14.51	1.07	9.25	4.70	14.53
Impatiens parviflora	2.21	6.48	3.15	11.29	11.68	23.61	10.32	21.73
Robinia pseudacacia	0.22	1.55	0.13	0.74	0.76	4.41	0.02	0.15
Rudbeckia laciniata	0.00	0.00	1.43	9.48	0.05	0.51	2.04	13.13
Solidago gigantea	6.13	16.01	12.21	22.41	14.57	29.97	20.98	34.69

Table 18: Road aproximity in 2011 and 2013. Mean and standard deviation (±SD) of abundance of IAS in 2011 and 2013 of plots with (S.road) and without (S.no. road*); Direct road proximity and average plant cover (%) and species number.*

	2011				**2013**			
	S.no.road*		S.road*		S.no.road*		S.road*	
	Mean	SD	Mean	SD	Mean	SD	Mean	SD
Plant cover total (%)	69.42	29.52	75.88	29.18	90.23	29.79	92.78	25.18
species number	12.7	4.4	17.1	7.9	12.7	5.8	17.22	8.64
Acer negundo	0.01	0.11	0.00	0.00	0.06	0.51	0.00	0.00
Ailanthus altissima	0.0	0.1	0.0	0.0	0.0	0.2	0.1	0.2
Bunias orientalis	0.00	0.05	0.66	2.55	0.05	0.50	5.37	16.45
Impatiens glandulifera	1.0	6.8	6.5	15.3	1.5	9.9	4.3	14.3
Impatiens parviflora	1.94	6.29	3.95	12.00	10.00	21.16	14.27	26.87
Robinia pseudacacia	0.1	0.5	0.4	2.3	0.3	1.6	1.1	6.3
Rudbeckia laciniata	0.66	6.47	0.00	0.00	1.00	8.97	0.00	0.00
Solidago gigantea	8.5	20.0	7.2	14.4	18.2	34.0	12.9	24.8

The visualization of the abundance of IAS in the plant cover of each plot shows differences among herb layer, shrub layer, tree layer I and tree layer II. The herb layer (0 to 1m) is most affected by the invasion of IAS. In total 109 plots were invaded by IAS in the herb layer (total n=142). The invasive species *Acer negundo, Ailanthus altissima, Bunias orientalis, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera, Impatiens parviflora* and *Solidago gigantea* were detected in the herb layers of the invaded plots. The cover of IAS within the herb layers differ.

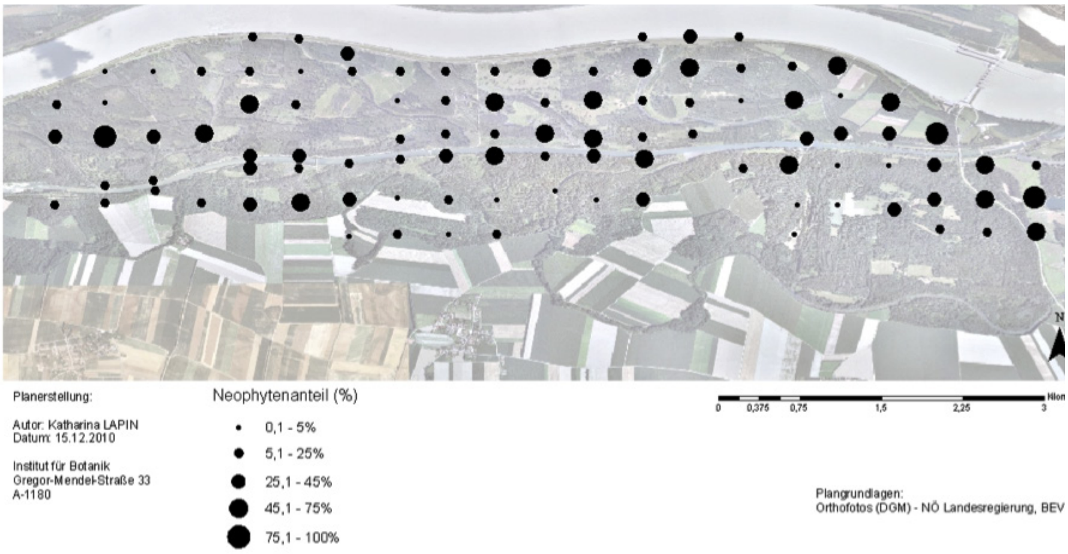


Figure 36 spatial distribution of IAS in the herb layer (2010). The abundance of IAS is shown in percentage of the total plant cover (%) separated in 5 classes (0.1 – 5%, 5.1 -25%, 25.1 – 45%, 45.1% - 75%, 75.1 – 100%). The invasive alien plant species Acer negundo, Ailanthus altissima, Bunias orientalis, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera, Impatiens parviflora, Solidago gigantea were recorded.

The shrub layer (1m to 2m) shows the lowest affection by IAS. The invasive species *Acer negundo, Ailanthus altissima, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera,* and *Solidago gigantea* were detected in the shrub layers of 18 invaded plots (total n=142). The invasive perennial and annual species, *Rudbeckia laciniata, Impatiens glandulifera* and *Solidago gigantea* are abundant in the shrub layer when the monotypic stands are developed on the invaded plots. In 2010, the invasion of the shrub layers is taking place mainly on embankments. The proximity to rivers are significant for the positive occurrence of IAS in the shrub layer.

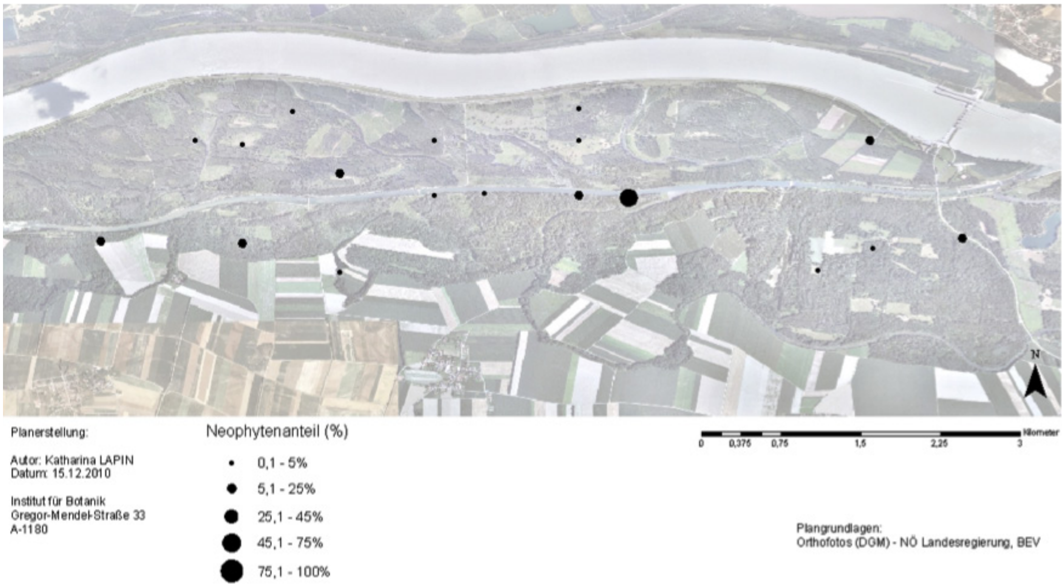


Figure 37 Spatial distribution of IAS in the shrub layer (2010). The abundance of IAS is shown in percentage of the total plant cover (%) separated in 5 classes (0.1 – 5%, 5.1 -25%, 25.1 – 45%, 45.1% - 75%, 75.1 – 100%). The invasive alien plant species Acer negundo, Ailanthus altissima, Bunias orientalis, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera, Impatiens parviflora, Solidago gigantea were recorded.

The tree layer I (2m to 5m) shows a low affection of IAS. The tree species *Acer negundo, Ailanthus altissima* and *Robinia pseudoacacia* were detected in the tree layer I within 23 plots (total n=142). The proximity to roads is significant for the positive occurrence of IAS in the tree layer I. The comparison of the development of IAS for the years 2010 and 2013 showed no significant increase of IAS in the tree layer I.

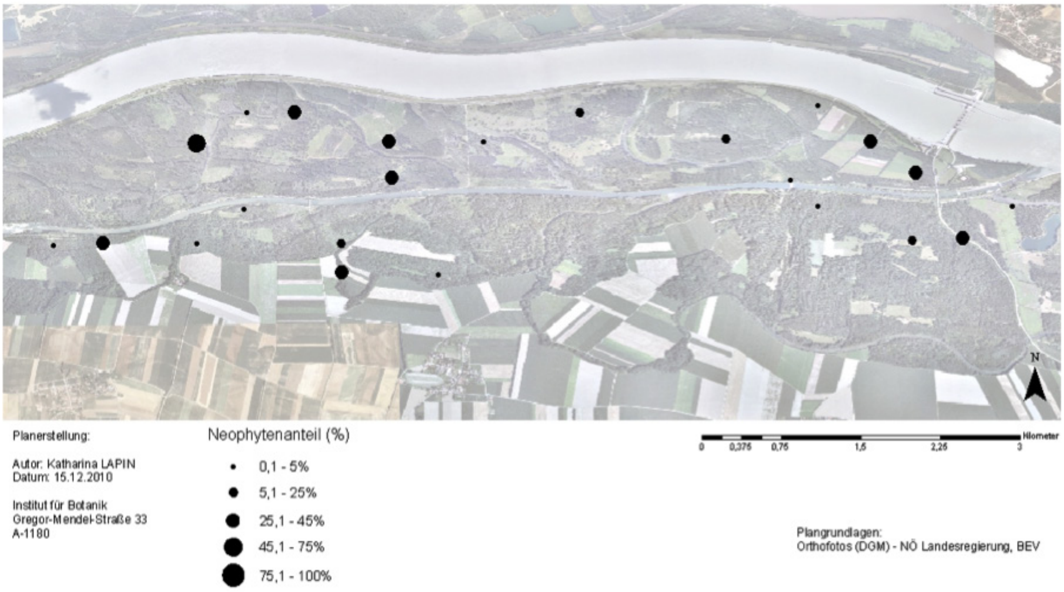


Figure 38 Spatial distribution of IAS in the tree layer I (2010). The abundance of IAS is shown in percentage of the total plant cover (%) separated in 5 classes (0.1 – 5%, 5.1 -25%, 25.1 – 45%, 45.1% - 75%, 75.1 – 100%). The invasive alien plant species Acer negundo, Ailanthus altissima, Bunias orientalis, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera, Impatiens parviflora, Solidago gigantea were recorded.

The tree layer II (5m to 20m) shows a higher affection of IAS than in the tree layer I. The tree species *Acer negundo, Ailanthus altissima* and *Robinia pseudoacacia* were detected in the the tree layer II within 45 plots (total n=142). The proximity to roads and the river stream are significant for the positive occurrence of IAS in the tree layer II. The main invasive tree species in the the tree layer II is *Robinia pseudoacacia*. The occurance of *Robinia pseudoacacia* is more frequent in habitats that are strongly anthropogenically influenced.

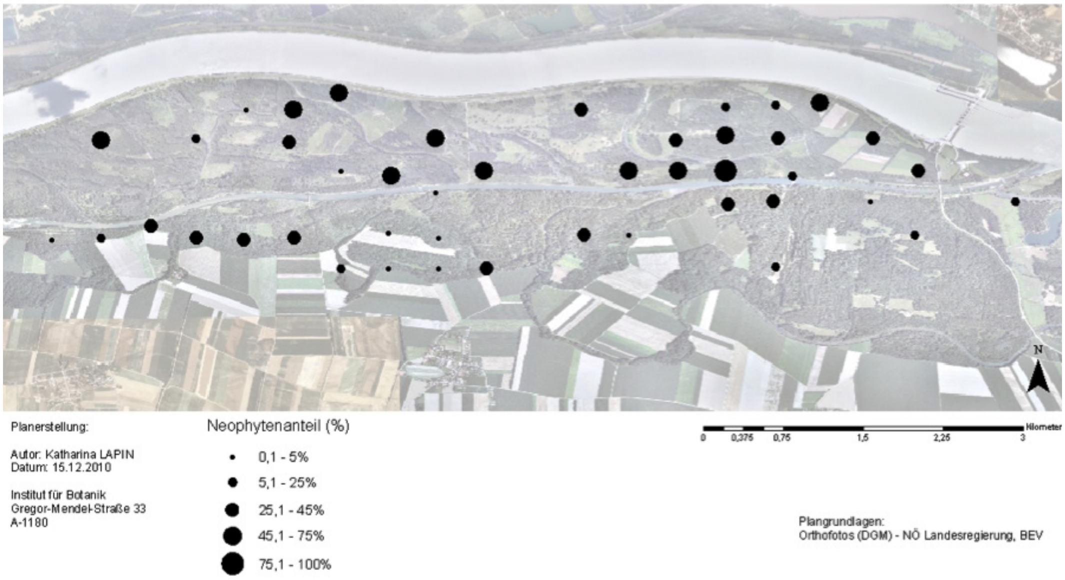


Figure 39 Spatial distribution of IAS in the herb layer II (2010). The abundance of IAS is shown in percentage of the total plant cover (%) separated in 5 classes (0.1 – 5%, 5.1 -25%, 25.1 – 45%, 45.1% - 75%, 75.1 – 100%). The invasive alien plant species Acer negundo, Ailanthus altissima, Bunias orientalis, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera, Impatiens parviflora, Solidago gigantea were recorded.

3.3 Impact on habitat types

The vegetation structure of the recorded 142 plots in the study years 2011, 2012 and 2013 are representing 11 habitat types: Ash tree forest, tall oatgrass meadows, grey alder tree forest, semi-dry grasslands, grassland fallow, tall forb community, cutover land, poplar tree forest, red pine tree forst, willow tree alluvial forest and embankment (Table 20). The result of the nonparametric Mann-Whithey-Wilcoxon test shows that the development of invasive species in the herb layer differs among habitat types ($P < 0.001$). The Pearson product-moment correlation shows a very low relationship among habitat type IAS cover as well as species number and IAS cover ($r < 0.05$). The habitat type tall forb community was invaded most by IAS during the study period. Habitat type grey alder tree forest shows a low abundance of IAS during the study period.

*Table 19 Results of Pearson correlation coeffient test for the species number (sp.n) x plant cover (cov) x IAS abundance in the three study years 2011, 2012 and 2013 (n=142 per year); * = n < 10 plots.*

	2011	2012	2013
ash tree forest **sp.n**	-0.288	-0.480	-0.608
ash tree forest **cov**	0.0282	0.211	0.152
tall oatgrass meadows **sp.n**	-0.636 *	-0.721 *	-0.8697 *
tall oatgrass meadows **cov**	0.555 *	0.812 *	0.984 *
grey alder tree forest **sp.n**	-0.321	-0.370	-0.417
grey alder tree forest **cov**	-0.105	-0.159	-0.042
semi-dry grasslands **sp.n**	-0.456 *	-0.705 *	-0.723 *
semi-dry grasslands **cov**	0.216*	0.235 *	0.586 *
grassland fallow **sp.n**	-0.547*	-0.915 *	-0.9611 *

	2011	2012	2013
grassland fallow **cov**	-0.594 *	-0.950 *	-0.129 *
tall forb community **sp.n**	-0.362	-0.621	-0.625
tall forb community **cov**	0.550	0.538	0.470
Cutover land **sp.n**	-0.109 *	-0.759 *	-0.112
Cutover land **cov**	-0.166 *	-0.528 *	0.514
poplar tree forest **sp.n**	-0.518	-0.403	-0.502
poplar tree forest **cov**	0.254	0.543	0.359
Embankment **sp.n**	0.355	0.094	-0.225
Embankment **cov**	0.375	0.664	0.668

Table 20 Development of IAS abundance amonge habitat types. The development over the years 2011, 2012 and 2013 are described using the plant cover (mean, ±SD) and total species number (mean, ±SD) of the average total plant cover. The abundance IAS shows the percentage of the invasive species Bunias orientalis, Solidago giganea, Impatiens granguifera, Impatiens parviflora, Alianthus altissima, Robinia pseudoacacia, Acer negundo and Rudbekia lanciotata of the average plant cover. The average abundance of the invasive species Bunias orientalis (Bu), Solidago giganea (Sol), Impatiens granguifera (Iglan) and Impatiens parviflora (Ipar) is shown separetly.

2011										
habitat type	**plots**	**plant cover**		**species number**		**abundace IAS ***	**average abundace ***			
		Mean	±SD	Mean	±SD	%	Sol	Iglan	Ipar	Bu
ash tree forest	40	73.15	29	12.525	4.7393	18.11	8.02	2.2	7.48	0
tall oatgrass meadows	4	84.62	32.73	18.50	7.41	38.40	38.40	0	0	0
grey alder tree forest	18	65.35	17.53	11.88	3.08	8.51	4.95	0	3.37	0
semi-dry grasslands	8	74.75	24.65	23.50	10.14	11.04	10.03	1.00	0	0
grassland fallow	3	73.17	12.36	18.67	8.96	18.68	18.22	0	0.46	0
tall forb community	13	73.5	30.87	15	4.88	29.51	22.08	3.77	0	2.09
cutover land	7	81.14	19.17	11.85	2.54	19.19	18.66	0	0.002	0
poplar tree forest	23	62.33	2.97	12.34	3.31	13.26	11.51	0	1.67	0
red pine tree forst	1	44	0	12	0	34.09	0	0	34.09	0
willow tree alluvial forest	1	63	0	11	0	11.11	7.93	1.59	0	1.59
embankment	23	78.74	42.04	15.61	7.20	29.68	8.23	14.08	3.31	0.36
tree hedge	1	30	0	12	0	0	0	0	0	0

2012										
habitat type	**plots**	**plant cover**		**species number**		**abundace IAS ***	**average abundace ***			
		Mean	±SD	mean	±SD	mean	Sol	Iglan	Ipar	Bu
ash tree forest	40	83.7	28.72	12.35	5.51	34.00	10.53	2.76	20.19	0
tall oatgrass meadows	4	95.63	40.98	17.75	5.06	33.99	33.99	0	0	0
grey alder tree forest	18	75.38	22.58	11.88	4.41	16.62	6.24	1.17	9.05	0
semi-dry grasslands	8	93.56	29.55	25.13	11.33	24.32	21.38	0.80	0.13	0
grassland fallow	3	115.83	7.14	22.66	11.06	26.61	25.91	0	0.29	0
tall forb community	13	90.27	32.78	15.30	5.92	44.23	32.51	5.11	0.51	4.69
cutover land	7	93.07	13.52	10.71	3.90	35.02	33.92	0	0.15	0
poplar tree forest	23	77.24	38.64	11.47	2.94	20.69	11.26	0.28	8.84	0
red pine tree forst	1	74.5	0	15	0	0	0	0	0	0
willow tree alluvial forest	1	81	0	11	0	30.86	6.17	6.17	0	18.52
embankment	23	88.82	31.47	15.43	6.58	43.95	13.46	16.20	8.57	1.03
tree hedge	1	32	0	14	0	0	0	0	0	0

2013										
habitat type	**plots**	**plant cover**		**speies number**		**abundace IAS ***	**average abundace ***			
		Mean	±SD	mean	±SD	mean	Sol	Iglan	Ipar	Bu
ash tree forest	39	93.63	27.81	13.46	5.98	39.41	15.31	2.52	20.29	0
tall oatgrass meadows	4	87.38	39.50	19.5	5.80	30.33	30.04	0	0	0
grey alder tree forest	18	95.26	18.09	12.35	3.82	23.43	6.85	0	16.49	0
semi-dry grasslands	8	108.44	25.96	27.63	10.17	24.56	21.33	0.58	0.12	0
grassland fallow	3	103.17	11.59	23.33	12.91	29.88	29.40	0	0	0
tall forb community	12	89.71	24.85	13.08	3.50	40.71	27.03	0.56	.93	12.08
cutover land	10	71.65	38.49	8.4	4.45	50.52	48.29	0.71	0.71	0
poplar tree forest	23	88.39	21.62	11.57	3.58	29.41	12.84	0.25	15.79	0
red pine tree forst	0	0	0	0	0	0	0	0	0	0
willow tree alluvial forest	1	111	0	10	0	31.53	13.51	4.50	0	13.51
embankment	23	91.52	34.81	15.22	6.97	47.89	19.33	9.03	11.16	3.80
tree hedge	1	36	0	14	0	0	0	0	0	0

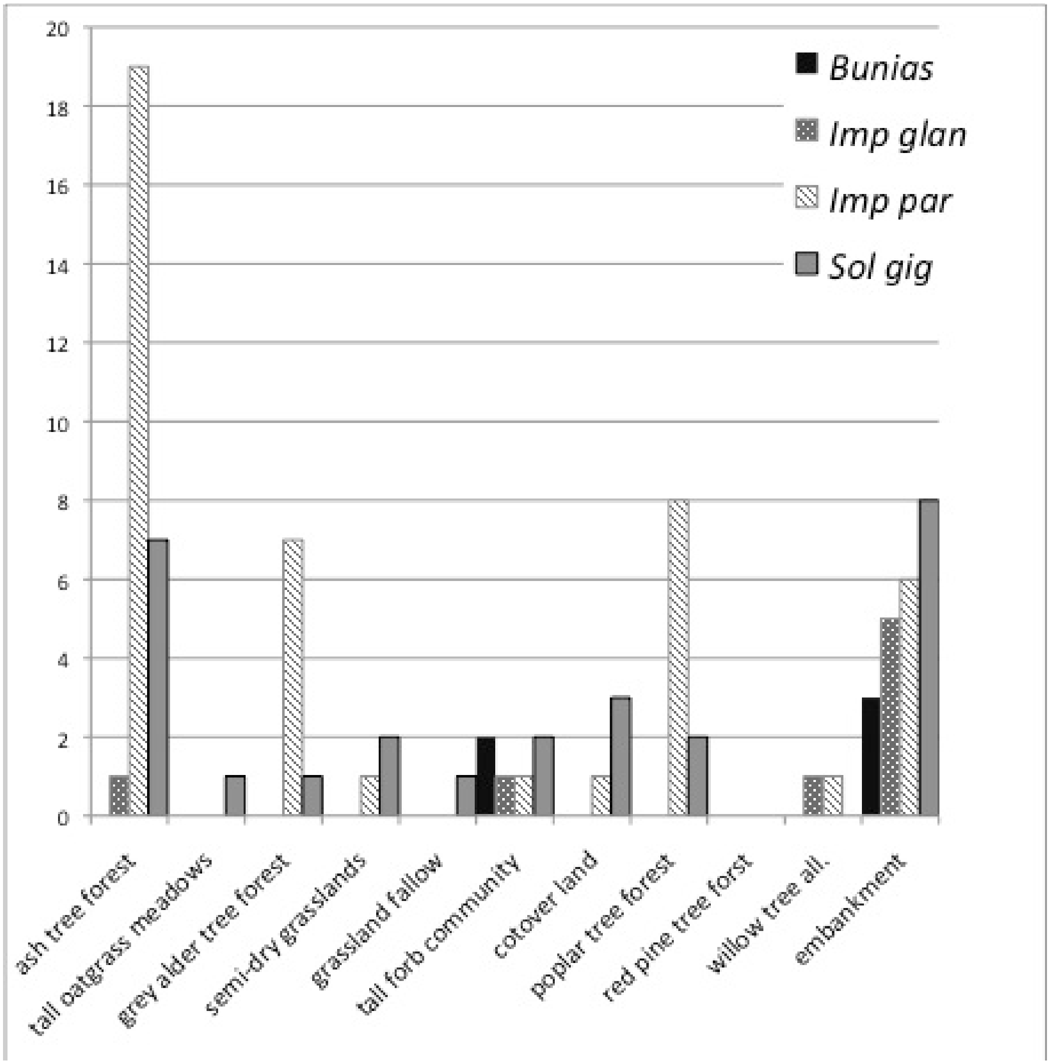


Figure 40 Rapid Invasion of IAS by habitat types. Number of plots (y-axis) with an increase of IAS Bunias orientalis (Bunias), Solidago giganea (Sol gig), Impatiens glanduifera (Imp glan) and Impatiens parviflora (Imp par) in the herb layer from 0% to over 60% from 2011 to 2013.

In total, 40 plots corresponded to the habitat type ash tree forest in 2012. In 2013, one plot was deforested (2013, n=39). The mean plant cover increased from 73.15% in 2011 to 93.63% in 2013. On average 12.5 species were recorded in the herb layer in the years 2011 and 2012 and 13.5 species in 2013. The invasive species *Solidago gigantea, Impatiens glanduifera, Impatiens parviflora, Alianthus altissima, Robinia pseudoacacia* and *Acer negundo* were detected. The abundance of IAS within the habitat type Ash tree forest was increasing. In 2011, invasive species represented in total 18.11% of the plant cover, in 2012 in total 34.0% and 39.41% in 2013 (Figure 40).

The changes of average abundance of *Impatiens parviflora* were strongest in 2012. The average abundancne of *I. parvifora* increased from 7.48% in 2011 to 20.19% in 2012. The populations of *Solidago gigantea* increased constantly from 8.02 % in 2011, to 10.53 in 2012, to 15.31% in 2013.

During the study period from 2011 to 2013 the abundance of IAS in the herb layer inceresed fom 0% to over 60% in 27 plots (n=40). Within this rapid invaded plots 19 plots were invaded by *Impatiens parviflora* and seven by *Solidago gigantea*.

Figure 41 Developed Ash tree forest

Tall oatgrass meadows

In total four plots corresponded to the habitat type tall oatgrass meadows. The land use of tall oatgrass meadows plots did not changed during the study period. The mean plant cover increased from 84.62% in 2011 to 96.3% in 2012 and decresed to 87.38% in 2013. In average 18.5 species were recorded in the herb layer in the year 2011 and 17.75% in 2012 and 19.5% species in 2013. One invasive species *Solidago gigantea*, was detected. The development of average abundance of *Solidago gigantea* was decreasing compared to the increasing total plant cover. The abundance of IAS within tall oatgrass meadows was constant with a low decreasing tendency. In 2011, invasive species represented in total 38.40% of the plant cover, in 2012 in total 33.99% and 30.33% in 2013.

During the study period from 2011 to 2013 the abundance of IAS in the herb layer increased fom 0 % to over 60% in one plot (n=4) by *Solidago gigantea*.

Figure 42 tall oatgrasss meadows in spring

Grey alder tree forest

In total 18 plots corresponded to the habitat type grey alder tree forest constantly during all three years. The mean plant cover in 2011 was 65.35% and increased to 95.26% in 2013. On average, 12 species were recorded in the herb layer. The average number of species did not change in grey alder tree forests. The invasive species *Solidago gigantea, Impatiens grandulifera, Impatiens parviflora, Ailanthus altissima* and *Acer negundo* were detected. In 2011 invasive species represented 8.51% of the plant cover, in 2012 16.62% and 23.43% in 2013.

The most abundant invasive plant species were *Solidago gigantea* and *Impatiens parviflora.* The percentage of the total plant cover of *I. parviflora* in grey alder tree forests increased from 3.37% in 2011 to 8.05% in 2012 and 16.39% in 2013.

During the study period from 2011 to 2013 the abundance of IAS in the herb layer increased from 0 % to over 60% in eight plots (n=18). Within this rapid invaded plot, seven plots were invaded by *Impatiens parviflora.*

Figure 43 Grey alder tree forest

Semi-dry grasslands

In total eight plots corresponded to the habitat type semi-dry grassland. The mean plant cover is increasing from 74.75% in 2011 to 108.44% in 2013. In average23.5 species in 2011 and 27,6 species in 2013 were recorded in the herb layer. The invasive species *Solidago gigantea, Impatiens glanulifera, Impatiens parviflora* and *Robinia pseudoacia* were detected. In 2011, invasive species represented 11.04% of the plant cover, in 2012 24.32% and 24.56% in 2013. The abundance of IAS did not increased. The most abundant invasive plant species was *Solidago gigantea* with a mean abundance of 10.03% in 2011, 21.38% in 2012 and 21.33% in 2013.

During the study period from 2011 to 2013, the abundance of IAS in the herb layer increased from 0 % to over 60% in three plots (n=8). Within these rapidly invaded plots, two plots were invaded by *Solidago gigantea* and one by *Impatiens parviflora*.

Figure 44 Semi-dry grasslands.

Grassland fallow

In total three plots corresponded the habitat type Grassland fallow. On average, twelve species were recorded in 2013 in the herb layer. The invasive species *Ailanthus altissima, Solidago gigantea* and *Impatiens parviflora* were detected. In 2011, invasive species represented 18.68% of the plant cover, in 2012 26.61% and 24.56% in 2013. The most abundant invasive plant species was *Solidago gigantea* with a mean abundance of 18.22% in 2011, 25.91% in 2012 and 29.40% in 2013. The increasing development of *S. gigantea* is remarkable.

During the study period from 2011 to 2013 the abundance of IAS in the herb layer inceresed fom 0 % to over 60% in one plot (n=3) by *Solidago gigantea.*

Figure 45 Grassland fallow.

Tall forb community

In total twelve plots are corresponded the habitat type tall forb community. The mean plant cover is 73.5% in 2011 and 89.71% in 2013. On average 15 species in 2011 and 13 species in 2013, were recorded in the herb layer. The invasive species *Ailanthus altissima, Bunias orientalis, Solidago gigantea, Impatiens glandulifera, Ropbinia pseudoacacia* and *Impatiens parviflora* were detected. In 2011, invasive species represented 29.51% of the plant cover, in 2012 44.23% and 40.71% in 2013.

The most abundant invasive plant species was *Solidago gigantea* with a mean abundance of 22.08% in 2011, 32.51% in 2012 and 27.03% in 2013. The development of *Bunias orientalis* is significant increasing within the tall forb communities. During 2011 and 2013 the abundance mean of 2.09% increased to 12.08%.

During the study period from 2011 to 2013 the abundance of IAS in the herb layer increased from 0% to over 60% in six plots (n=12). Within these rapidly invaded plots, two plots were invaded by *Bunias orientalis*, two plots were invaded by *Solidago gigantea.*

Figure 46 Tall forb community with dominance of the native Eupatorium cannabium.

Cutover land

In total, seven plots corresponded to the habitat type cutover land. The number of plots increased in 2013 to 10 plots, because of deforestations for the restoration constructions. The mean plant cover is decreasing from 93.07% to 71.65%. In average 13.5 species were recorded in 2011 in the herb layer and 8.4 species in 2013. The invasive species *Solidago gigantea* and *Robinia pseudoacacia* were detected. In 2011, invasive species represented 35.02% of the plant cover, in 2013 48.29%. The most abundant invasive plant species was *Solidago gigantea* with an increasing mean abundance.

During the study period from 2011 to 2013 the abundance of IAS in the herb layer increased from 0% to over 60% in four plots (n=7). Within these rapidly invaded plots, three plots were invaded by *Solidago gigantea.*

Figure 47 Cutover land 3 month after deforestation.

Poplar tree forest

In total 23 plots corresponded to the habitat type Poplar tree forest. The mean plant cover is 62.33% in 2011 and 88.39% in 2013. On average 11.47 species in 2011 and 21.62 species in 2013 were recorded in the herb layer. The invasive species *Impatiens glandulifera, Impatiens parviflora, Robinia pseudacacia, Rudbeckia laciniata* and *Solidago gigantea* were detected. In 2011 invasive species represented 20.69% of the plant cover, in 2012 20.69% and 29.41% in 2013. The most abundant invasive plant species were *Impatiens parviflora* and *Solidago gigantea.* The average abundance of *Impatiens parviflora* increased from 1.67% in 2011 to 15.79% in 2013.

During the study period from 2011 to 2013, the abundance of IAS in the herb layer increased from 0% to over 60% in ten plots (n=23). Within these rapidly invaded plots, eight plots were invaded by I*mpatiens parviflora*.

Figure 48 Young Poplar tree forest.

Red pine tree forst

In total, one plot corresponded to the habitat type Red pine tree forest. The mean plant cover is 74.5%. In 2012, the plot was deforested as part of the construction measures of the ecological restoration. On average 15 species were recorded in the herb layer. No invasive species were abundant in the herb layer.

Figure 49 Red pine tree forst.

Willow tree alluvial forest

In total, one plot coresponded to the habitat type Willow tree alluvial forest. The mean plant cover is 63% in 2011 and 111% in 2013. On average eleven species were recorded in the herb layer. The invasive species *Bunias orientalis, Impatiens glangulifera* and *Solidago gigantea* were detected. In 2011 invasive species represented 11.11% of the plant cover, in 2012 30.86% and 24.56% in 2013.

The most abundant invasive plant species was *Solidago gigantea.* The development of *Impatiens glandulifera* is remarkable. During 2011 and 2013, the abundance mean of *I. glandulifera* increased from 1.59% in average plant cover in 2011 to 4.50% in 2013.

Figure 50 Willow tree alluvial forest with old Salix alba occurence.

Embankment

In total, 23 plots corresponded to the habitat type embankment. The mean plant cover is 78.74% in 2011 and increased up to 91.52% in 2013. On average, fifteen species were recorded in the herb layer. The invasive species *Bunias orientalis, Impatiens glandulifera, Impatiens parviflora, Robinia pseudacacia Rudbeckia laciniata* and *Solidago gigantea* were detected. In 2011, invasive species represented 29.68% of the plant cover, in 2012 43.95% and 47.89% in 2013.

The most abundant invasive plant species were *Impatiens glandulifera* and *Solidago gigantea*. The development of *Impatiens glandulifera* during 2011 and 2013 showed an increasing tendency in 2011 and 2012. In 2013 the abundance decreased from 16.20% in 2012 to 9.03% in 2013.
During the study period from 2011 to 2013, the abundance of IAS in the herb layer increased fom 0% to over 60% in 22 plots (n=23). Within these rapidly invaded plots, eight plots were invaded by *Solidago gigantea* and six plots by *Impatiens parviflora,* five plots by *Impatiens glandulifera*, and three plots by *Bunias orientalis.*

Figure 51 Artificial embarkment of the river Traisen. right river side.

3.4 Diversification of environmental factors

Canopy

In total 81 plots (n=142) are affected by developed canopy of the tree layer I and II. The analysis of vegetation development of plots with influence of canopy showed significant differences of IAS abundance and number of species comparing plots without influence of canopy in the years 2011, 2012 an 2013 ($P < 0.001$).

The abundance of invasive species *Solidago gigantea, Impatiens glandulifera* and *Impatiens parviflora* show significant differences in the development within plots, with and without canopy. The mean cover of *I. parviflora* increased from 3.11% in 2011 to 13.82% in 2013 on plots with a canopy, and from 0% to 0.22% on plots without a canopy. The results of the Mann-Whithey-Wilcoxon test show significant differences among the cover of *I. parviflora* in the year 2011 and 2013 ($P < 0.001$). The mean cover of *S. gigantea* increases significantly ($P < 0.001$) from 7.43% in 2011 to 14.29% in 2013 on plots with a canopy and from 11.19% to 26.93% on plots without a canopy.

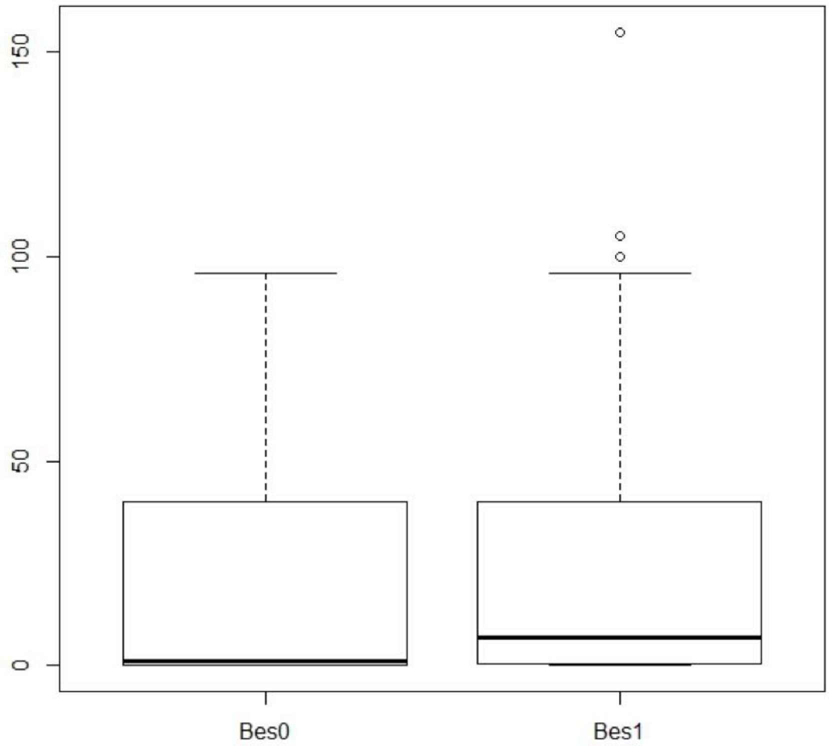


Figure 52 Abundance of IAS of plots among the two canopy classifications (Bes1= canopy, Bes0=no canopy) in the herb layer in sum of 2011, 2012 and 2013 (n= 1263 plots).

Table 21 Influence of tree canopy. The invasive alien plant species and in occurrence within the plant cove of the herb layer in all plots (n=142) in the years 2011, 2012 and 2013; Value of abundance in the total plant cover of each IAS and in total (plant cover total); The development of the abundance (S.no.canopy) is compared using the mean and standard deviation (Mean +-SD).The number of plots (S.canopy*) show uneven distribution among species.*

	2011					2012					2013			
	S.no.canopy*		S.canopy*			S.no canopy*		S.canopy*			S. no canopy*		S. canopy*	
	Mean	SD	Mean	SD		Mean	SD	Mean	SD		Mean	SD	Mean	SD
plant cover total (%)	80.00	32.40	69.24	28.50		97.06	28.63	81.30	30.70		90.20	36.71	91.14	26.36
species number	17.70	8.94	13.13	4.65		17.63	9.28	13.05	5.51		17.52	10.92	13.22	5.45
Acer negundo	0.00	0.00	0.01	0.10		0.00	0.00	0.06	0.48		0.19	0.96	0.01	0.10
Ailanthus altissima	0.00	0.00	0.02	0.13		0.06	0.21	0.04	0.19		0.06	0.21	0.05	0.21
Bunias orientalis	0.20	0.96	0.19	1.47		0.04	0.19	0.78	4.42		0.19	0.96	1.91	10.08
Impatiens glandulifera	1.85	7.74	2.74	10.80		0.22	0.97	4.47	15.36		0.19	0.96	2.77	12.57
Impatiens parviflora	0.00	0.00	3.11	9.19		0.07	0.27	9.84	21.39		0.22	0.97	13.82	24.80
Robinia pseudacacia	0.00	0.00	0.23	1.48		0.74	3.01	0.26	1.54		0.70	2.88	0.47	3.78
Rudbeckia laciniata	2.44	12.50	0.01	0.09		3.52	17.31	0.01	0.09		3.52	17.31	0.05	0.47
Solidago gigantea	11.19	25.21	7.43	16.63		22.07	34.62	11.29	23.75		26.93	40.02	14.29	29.01

Cultivation intensity

In total 80 plots show an intensive use (classification 3=intensive), 29 plots show a low intenstity of land use and 33 plots are near-natural habitats (n=142). The analysis of vegetation development showed no significant differences of IAS cover among plots of diffrent cultivation intesity ($P < 0.001$). Comparing the IAS cover of different cultivation intensity of the year 2011 and 2013, the Mann-Whithey-Wilcoxon test showed low significant differences ($P = 0.074$).

The invasive species *Bunias orientalis, Solidago gigantea* and *Impatiens glandulifera* occure with a higher mean on plots with higher cultivation intensity. *Impatiens parviflora* is more abundant in nature-near habitats.

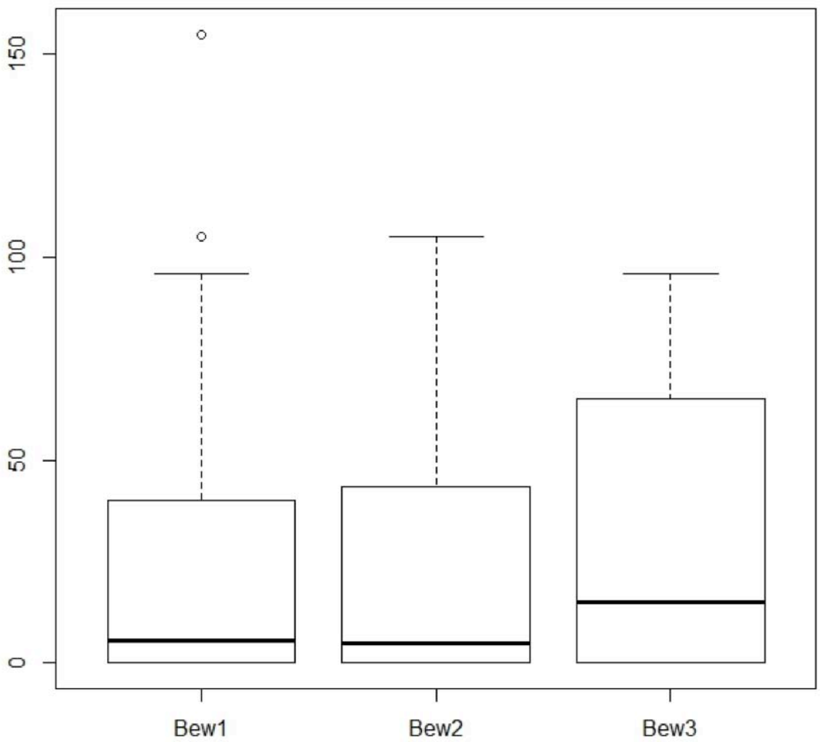


Figure 53 total abundance of IAS of plots among the three crassifications of cultivation intensitiy (Bew1=l near-nature, Bew2=low, Bew3=intensive) in the herb layer in sum of 2011, 2012 and 2013 (n= 1263 plots).

Table 22 Influnece of cultivation intensity. The invasive alien plant species and in occurrence within the plant covernumber of species of the herb layer in all plots (n=142) in the years 2011 and 2013; Value of abundance in the total plant cover of each IAS and in total (plant cover total); The development of the abundance is compared using the mean and standard deviation (Mean +-SD) in 3 classes: 1 (nature near), 2 (low intensity) and 3 (high intensity).

	2011							2013					
cultivation intensity	1		2		3			1		2		3	
	Mean	SD	Mean	SD	Mean	SD		Mean	SD	Mean	SD	Mean	SD
plant cover total (%)	68.73	29.82	76.53	32.31	72.36	24.88		92.51	27.71	92.40	27.86	85.53	31.18
species number	12.81	4.13	13.97	6.67	17.31	8.01		13.15	4.84	14.97	8.01	15.39	9.93
Acer negundo	0.02	0.12	0.00	0.00	0.00	0.00		0.02	0.12	0.00	0.00	0.16	0.90
Ailanthus altissima	0.03	0.16	0.00	0.00	0.00	0.00		0.03	0.17	0.13	0.34	0.02	0.09
Bunias orientalis	0.20	1.68	0.17	0.87	0.21	0.94		0.88	5.02	1.61	7.23	3.39	16.30
Impatiens glandulifera	1.71	8.75	3.36	10.12	4.03	13.89		2.15	11.16	0.84	2.91	4.03	16.25
Impatiens parviflora	3.73	10.03	1.61	7.00	0.21	0.94		16.16	26.63	6.81	17.57	2.94	11.88
Robinia pseudacacia	0.14	0.61	0.48	2.61	0.00	0.00		0.61	4.50	0.19	0.91	0.58	2.69
Rudbeckia laciniata	0.01	0.11	0.00	0.00	2.28	12.06		0.13	0.79	0.00	0.00	2.94	16.16
Solidago gigantea	4.75	11.69	13.27	26.64	11.66	21.29		12.32	26.95	20.71	34.84	23.97	38.14

3.5 Impact on plantdiverity

The mean Shannon index H`max (n=142) decreased from 2.6 in 2011 to 2.5 in 2013, during the mean plant cover was increasing form 71.3% in 2011 to 91% in 2013. The total number of species of 14 species per plot in average (n=142) is constant unchanged. Also mean Pilou´s index of evenness J (n=142) is decreasing from 0.7 in 2011 to 0.6 in 2013. The diversity of observed plots is decreasing linking the species richness and evenness. H` values of 2011 compared to 2013, using the Mann-Whithey-Wilcoxon test showed did not differ significantly (P = 0.690). The J values differ significantly among the years 2011 and 2013 (P = 0.01242).

Results of the Mann-Whithey-Wilcoxon test show significant differences among H` (P = 0.023) and J (P = 0.00243) on plots with IAS and without IAS in the herb layer (n=142 plots) in the year 2013 (Table 23)..

Comparing the development of H` of invaded plots among the years 2011 and 2013 no significant differences among H` of the years could be detected (P = 0.621). Compating the development of J of invaded plots among the years 2011 and 2013 significant diffrences among J among the years could be detected (P = 0.0063).

Table 23: Develompent of Biodiverstiy. The Shannon index (Hmax = -ln(1/i)) and Pilou´s index of eveness (HS/Hmax = Eveness J) was used to describe changes of diversity, comparing the total plant cover (n=142) and species number, showing the minimum value (Min) and maximum value (Max), mean ans standarddiviation (± SD) of the years 2011, 2012 an 2013.

2011					
	total plant cover	**i = species number**	**Hmax = -ln(1/i)**	**HS =sum -ln(p)*p**	**HS/Hmax = Eveness J**
Min	14.5	4.0	1.4	0.3	0.2
Max	176.5	36.0	3.6	3.3	0.9
Mean	71.3	14.0	2.6	1.7	0.7
± SD	29.5	6.0	0.4	0.5	0.1
2012					
	total plant cover	**i = species number**	**Hmax = -ln(1/i)**	**HS =sum -ln(p)*p**	**HS/Hmax = Eveness J**
Min	13.0	3.0	1.1	0.1	0.1
Max	208.0	42.0	3.7	3.4	0.9
Mean	84.3	13.9	2.5	1.7	0.6
± SD	30.8	6.6	0.5	0.6	0.2
2013					
	total plant cover	**i = species number**	**Hmax = -ln(1/i)**	**HS =sum -ln(p)*p**	**HS/Hmax = Eveness J**
Min	3.0	2.0	0.7	0.1	0.1
Max	173.5	44.0	3.8	3.5	1.0
Mean	91.0	14.0	2.5	1.6	0.6
± SD	28.5	7.0	0.5	0.7	0.2

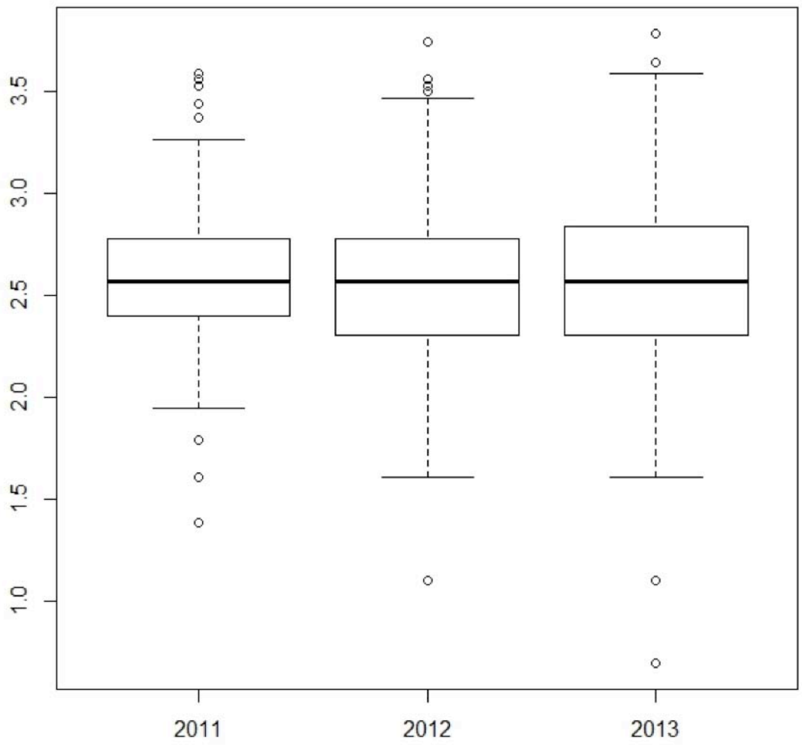


Figure 54 development of Shannon diversity index H´ of the species in the herb layer among the years2011, 2012 and 2013 of all plots (n=142).

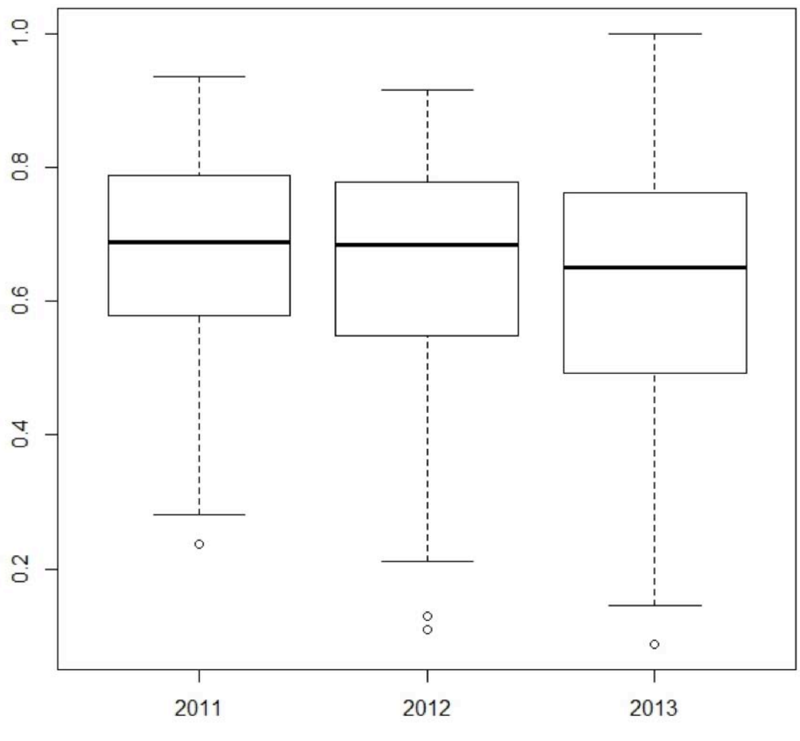


Figure 55 Development of Pilou´s index of evenness of the species in the herb layer among the years2011, 2012 and 2013 of all plots (n=142).

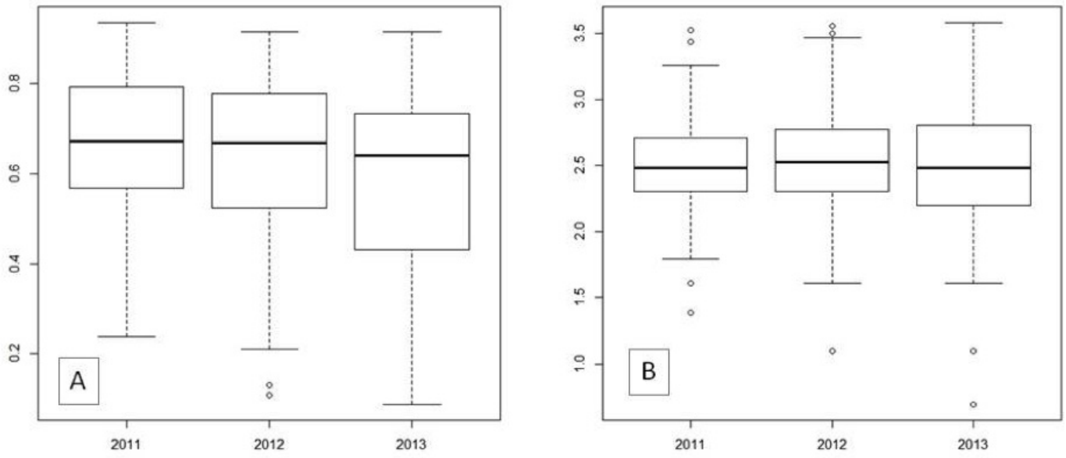


Figure 56 development of Shannon diversity index H´ of the species in the herb layer among the years2011, 2012 and 2013 of all plots (n=142), comparing plots without any occurance of IAS (A) and with ocurance (B).

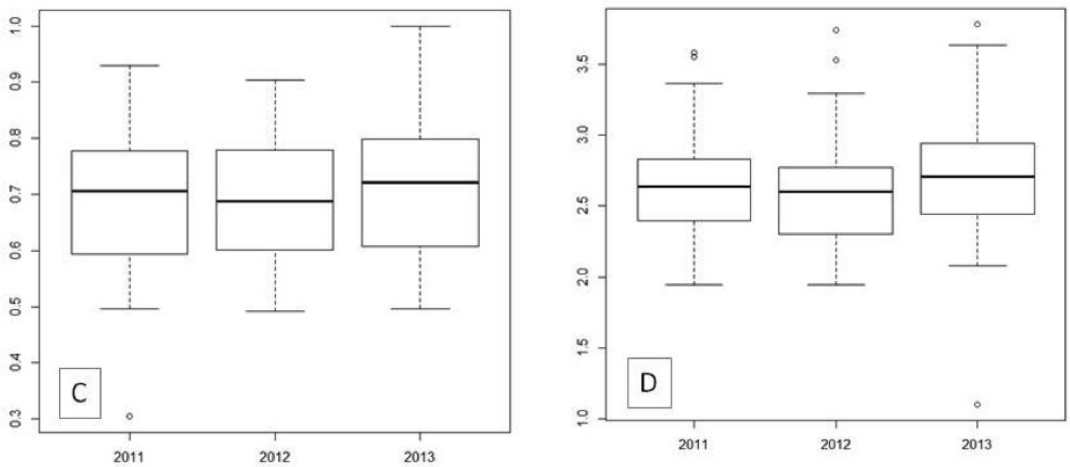


Figure 57 Development of Pilou´s index of evenness J of the species in the herb layer among the years2011, 2012 and 2013 of all plots (n=142), comparing plots without any occurance of IAS (C) and with ocurance (D).

3.6 Germination experiment

Germination requirements

In total, seeds of the native species *U. dioica* germinated very frequently and consistently (73%, n=400 seeds). *U. dioica*, 51.5% (n=200) of seeds germinated at 25/10°C and 94.5% of seeds germinated at 15/5°C. The highest total amount of germinated seeds was observed for the invasive species *S. gigantea.* 81.5% (n=200 seeds) of seeds of *S. gigantea* germinated at 25/10°C, and 54.5% (n=200 seeds) germinated at 15/5°C. and similar to the development of *U. dioica* (Fig.1). The germination rate of *T. vulgare* was around 40% with a particularly high rate in the first 10 days. Seeds of the native species *E. cannabinum* (18.8%, n=400) and *S. sarracenicus* (28.8 %, n=400) were similar in their number of germinated seeds and also showed similarities in their development of emerging seeds over time in that both species began to germinate late after day 20 but germinated after day 20. 10.5% (n=400) of the seeds of the native species *S. glutinosa* geminated, most of these within the first 20 days. Germination rates for *S. glutinosa* were 13.5% at 25/10°C and 7.5% at 15/5°C. Total germination capacity of the invasive species Seeds of the species *B. orientalis* and *I. parviflora* kept at 15/5°C and 25/10°C showed no germination activity. For the native species the lowest number of germinated seeds was recorded for the native species *S. glutinosa* and invasive species *I. glandulifera. I. glandulifera* were under 5% and therefore not representative for the further seedling development analysis (4%, n=400 sees) (Fig. 4). Seeds of the species *B. orientalis* and *I. parviflora* kept at 15/5°C and 25/10°C showed no germination activity (Table 3). The highest total amount of germinated seeds was observed for the invasive species *S. gigantea*. 81% (n=200 seeds) of seeds of *S. gigantea* germinated at 25/10°C, and 54.5% germinated at 15/5°C. For the native species *U. dioica*, 51.5% of seeds germinated at 25/10°C and 94.5% of seeds germinated at 15/5°C. The lowest number of germinated seeds was recorded for the native species *S. glutinosa* and invasive species *I. glandulifera.* Germination rates for *S. glutinosa* were 13.5% at 25/10°C and 7.5% at 15/5°C, and *I. glandulifera* showed rates of 2% at 25/10°C and 6 % at 15/5°C.

Observing the subplot of samples with cold stratification, the germination rates of seeds of the species *S. gigantea* and *S. glutinosa* were significantly higher at 25/10°C than at 15/5°C. The opposite was true for seeds of the species *I. glandulifera, S. sarracenicus* and *U. dioica*: these germinated more frequently in the lower temperature environment. The germination of the species *S. sarracenicus, T. vulgare* and *S. glutinosa* within the subplot of samples without cold stratification was significantly higher at 25/10°C than at 15/5°C, while germination of the species *I. glandulifera* and *E. cannabinum* was significantly lower at 25/10°C than at 15/5°C.

Table 24 Summary of the ANOVA: effect of could-wet stratification (storage) and temperature (15/5°C and 25/10°C) on total germination and time to germination of all species.

Factor	DF	SSQ	MS	F	P
Storage	1	4.60	4.63	0.0664	0.7971
Temperature	1	2469.7	2469.68	35.432	2.412e-08***
Storage x Temperature	1	0.4	0.36	0.0051	0.9429
Residuals	127	8852.1	69.5		

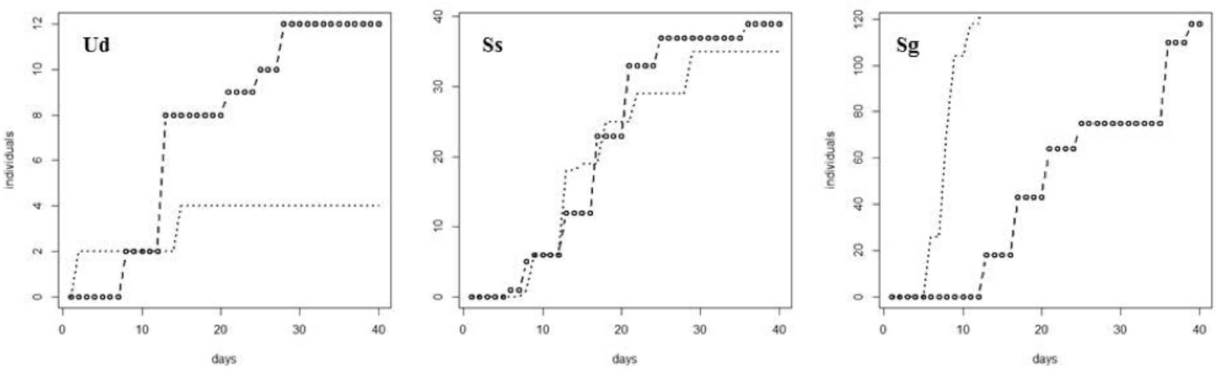


Figure 58 Number of germinated seeds of the species S. gigantea (Sg), S. sarracenicus (Ss) and U. dioica (Ud) at 15/5°C (--°°--) and 25/10°C (…..), (n=400 seeds per species), over the course of 40 days.

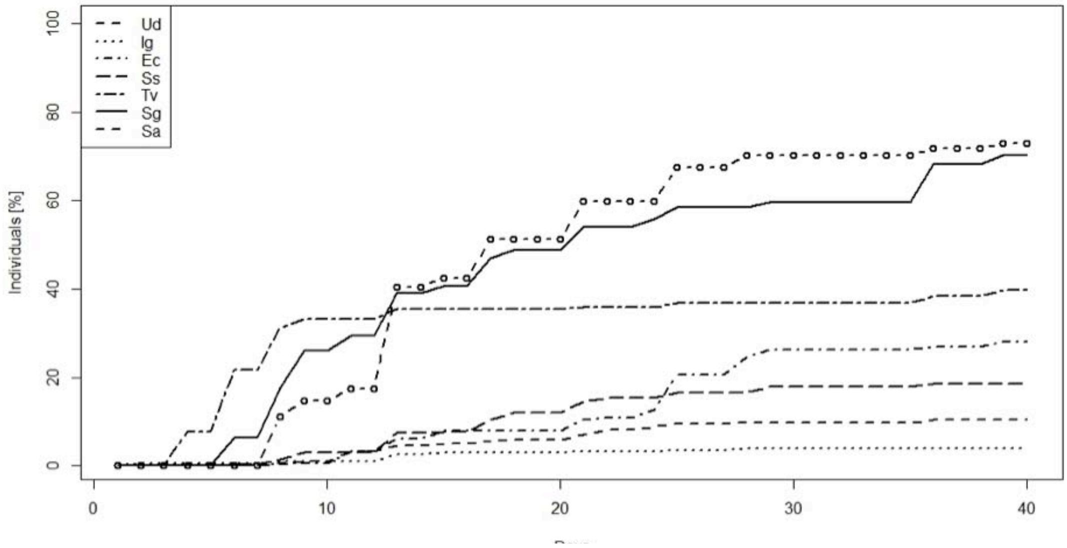


Figure 59 Percentage of germinated seeds of invasive alien species I. glandulifera (Ig), I. parviflora (Ip), S. gigantea (Sg) and native species E. cannabinum (Ec), S. sarracenicus (Ss), T. vulgare (Tv), U. dioica (Ud), and S. glutinosa (Sa) in total (n=400 seeds per species), over the course of 40 days.

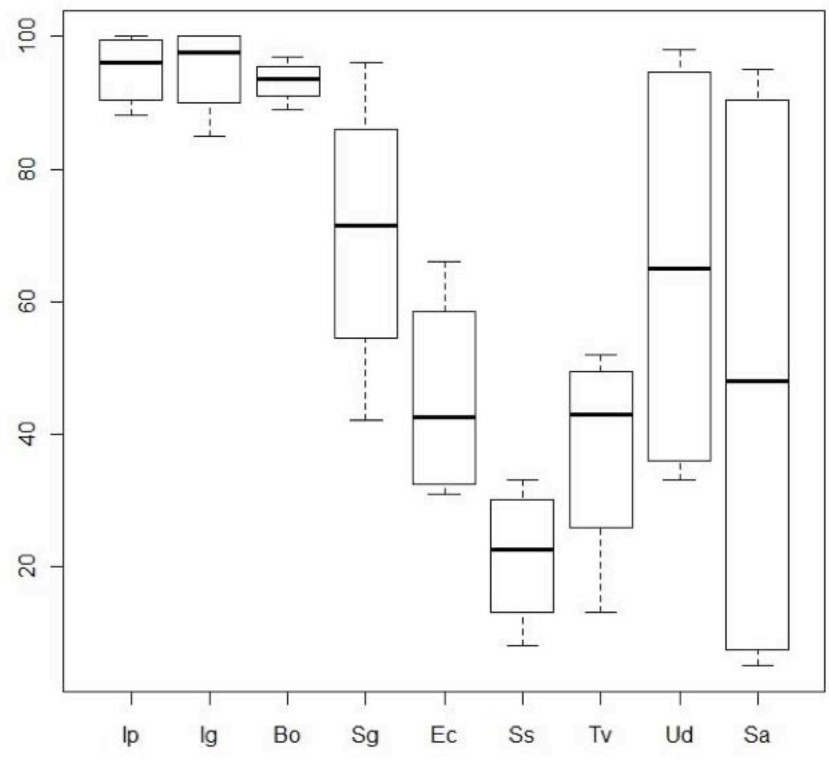


Figure 60 Boxplot of percentage of germinated seeds of invasive alien species I. glandulifera (Ig), I. parviflora (Ip), S. gigantean (Sg) and native species E. cannabinum (Ec), S. sarracenicus (Ss), T. vulgare (Tv), U. dioica (Ud), and S. glutinosa (Sa) in total (n=400 seeds per species), over the course of 40days.

Table 25 Percentage of seeds of all tested species that germinated after 40 days at 25/10°C and 15/5°C, split into the groups of seeds with cold-wet stratification and without. Values are averages across sites based on 2 replicates per species and site and 50 seeds per Petri dish.

Species	with cold stratification				without cold statification		
	germinated [%]		**difference**		**germinated [%]**		**difference**
	25/10 °C	15/5 °C	%		25/10 °C	15/5 °C	%
I. parviflora	0	0	0		0	0	0
I. glandulifera	3	9	-6		1	3	-2
B. orientalis	0	0	0		0	0	0
S. gigantea	96	42	54		67	67	0
E. cannabinum	36	31	31		11	34	-23
S. sarracenicus	8	20	-12		27	19	8
T. vulgare	47	47	0		52	13	39
U. dioica	51	98	-47		52	91	-39
S. glutinosa	10	5	5		17	10	7

The analysis of the subplot with cold-wet stratification at 25/10°C showed that on average, seeds of the invasive species *I. glandulifera* and *S. gigantea* as well as seeds of the native species *T. vulgare* and *U. dioica* emerged within the first 10 days. *T. vulgare* emerged fastest on average: by the 6th day. *S. gigantea. E. cannabinum, S. sarracenicus* and *S. glutinosa* emerged between the 10th and 20th day of the experiment on average.

The subplot without cold-wet stratification at 25/10°C showed no differences in this respect. Seeds germinated 2.5 days later on average than within the subplot with cold-wet stratification. Seeds of the species *I. glandulifera, S. gigantea, T. vulgare* and *U. dioica* emerged within the first 11 days, with *T. vulgare* once again the fastest with an average of germination time of 6 days. *S. gigantea. E. cannabinum, S. sarracenicus* and *S. glutinosa* emerged between the 10th and 21st day of the experiment on average.

Germination at 15/5°C with cold-wet stratification was very slow. It began after the 14th day for all species. *I. glandulifera, S. sarracenicus, T. vulgare* and *U. dioica* emerged before day 20 on average. *S. gigantea* and *E. cannabinum* generally emerged within 30 days and *S. glutinosa* within 32 days.

Germination at 15/5°C without cold-wet stratification was slower than at 25/10°C. *S. sarracenicus* emerged within 20 days (on average), followed by *S. glutinosa* and *U. dioica* (within 21 days). *S. gigantea* and *E. cannabinum* emerged within 30 days. *T. vulgare* emerged the quickest – and as the only species to emerge earlier without cold-wet stratification at 15/5°C: by day 13 on average.

Seedling development

After germination at 15/5°C, the invasive species *S. gigantea* developed its first primary leaf after 34 days in both subplots (with and without cold-wet stratification) (Figure 59). After germination at 25/10°C, seedlings of *S. gigantea* developed their first unifoliolates after 27 days with cold-wet stratification and after 32 days without cold-wet stratification. The native species *E. cannabinum* reached Stage 4 after 38 days at 15/10°C and after 28.5 days at 25/10°C, regardless of stratification. At 15/10°C, *S. sarracenicus* developed its first unifoliolates after 32 days with cold-wet stratification and after 29.5 days without cold-wet stratification. At 25/10°C, seedlings of *S. sarracenicus* reached Stage 4 after 27 days with cold-wet stratification and after 30 days without cold-wet stratification. After germination at 15/5°C, the invasive species *T. vulgare* developed its first unifoliolates after 38 days with cold-wet stratification and after 34 days without cold-wet stratification. At 25/10°C, seedlings of *T. vulgare* developed their first unifoliolates after 19 days regardless of stratification. At 15/10°C *U. dioica* reached Stage 4 after 35 days with cold-wet stratification and after 33 days without; at 25/10°C, Stage 4 was reached after 36 days with stratification and after 31 days without stratification. At 15/10°C, *S. glutinosa* developed its first unifoliolates after 35 days with or without cold-wet stratification, and at 25/10°C, its seedlings reached Stage 4 after 22 days with or without cold-wet stratification (Table 27).

Table 26 Mean of days required to reach the four phenological development stages (1 = yellow cotyledons, radicle visible, 2 = cotyledons start emerging, 3 = cotyledons are mostly green, above the ground and hanging on the curvature of the hypocotyl, 4 = cotyledons are above ground and fully green; unifoliolates emerging) for all germinated seedlings of native and invasive species at 15/5°C and 40 days of observation in total. Values are means across sites of each subplot based on two replicates per species and 50 seeds per Petri dish.

Species	with cold-wet stratification					without cold-wet stratification			
	Time to emergence [days]					Time to emergence [days]			
	Stage 1	Stage 2	Stage 3	Stage 4		Stage 1	Stage 2	Stage 3	Stage 4
	Mean (sd)	Mean (sd)	Mean (sd)	Mean (sd)		Mean (sd)	Mean (sd)	Mean (sd)	Mean (sd)
I. glandulifera	15.3 (7.5)	21.4 (1.3)	28.0 (1.7)	36.4 (1.5)		19.7 (6.1)	25.0 (4.2)	28.5 (3.5)	36.0 (2.8)
S. gigantea	21.2 (8.6)	27.2 (5.0)	31.5 (3.3)	34.5 (4.1)		26.3 (9.1)	22.7 (4.0)	29.6 (2.0)	34.1 (2.6)
E. cannabinum	28.2 (4.7)	29.8 (1.1)	31.8 (2.0)	37.8 (1.5)		25.0 (3.9)	28.0 (0.0)	31.7 (0.5)	38.0 (0.0)
S. sarracenicus	15.4 (4.3)	18.7 (2.5)	22.3 (2.1)	32.0 (0.0)		20.4 (7.9)	17.7 (9.8)	27.0 (0.0)	29.5 (3.5)
T. vulgare	14.1 (13.4)	16.9 (6.9)	28.2 (6.4)	38.0 (0.0)		13.4 (10.4)	16.7 (5.5)	30.0 (2.6)	34.0 (0.0)
U. dioica	17.6 (6.2)	22.7 (6.6)	25.7 (4.2)	35.2 (0.4)		20.2 (5.3)	24.4 (5.1)	29.4 (3.6)	33.4 (1.9)
S. glutinosa	31.4 (6.8)	3.8 (6.5)	31.0 (2.8)	35.0 (4.2)		20.2 (4.1)	27.5 (3.8)	30.6 (3.5)	34.8 (1.8)

Table 27 Mean of days required to reach the four phenological development stages (1 = yellow cotyledons, radicle visible, 2 = cotyledons start emerging, 3 = cotyledons are mostly green, above the ground and hanging on the curvature of the hypocotyl, 4 = cotyledons are above ground and fully green; unifoliolates emerging) for all germinated seedlings of native and invasive species at 25/10°C and 40 days of observation in total. Values are means across sites of each subplot based on two replicates per species and 50 seeds per Petri dish.

Species	with cold-wet stratification					without cold-wet stratification			
	Time to emergence [days]					Time to emergence [days]			
	Stage 1	Stage 2	Stage 3	Stage 4		Stage 1	Stage 2	Stage 3	Stage 4
	Mean (sd)	Mean (sd)	Mean (sd)	Mean (sd)		Mean (sd)	Mean (sd)	Mean (sd)	Mean (sd)
I. glandulifera	8 (8.1)	22 (0.0)	22 (0.0)	29 (0.0)		1.0 (0.0)	9.0 (0.0)	11 (0.0)	12 (0.0)
S. gigantea	9.9 (5.3)	9.3 (1.5)	13.0 (4.9)	27.3 (4.5)		9.1 (3.1)	10.6 (3.7)	16.0 (5.8)	31.5 (4.4)
E. cannabinum	16.1(6.5)	14.5 (4.9)	19.1 (7.7)	27.9 (5.2)		21.3 (9.6)	21 (3.31)	25.7 (9.3)	29.2 (5.2)
S. sarracenicus	16.3 (6.5)	10 (1.4)	18.3 (4.2)	27 (3.4)		17.1 (6.8)	10.3 (1.2)	17.1 (3.8)	30.4 (7.7)
T. vulgare	5.7 (1.4)	7.7 (3.2)	8.4(3.4)	19.1(3.4)		7.7(5.1)	6.8(0.4)	9.1(1.4)	19.3(3.6)
U. dioica	9.9 (2.6)	12.5 (4.6)	19.5 (3.9)	36 (0.0)		11.7(6.1)	11.6 (3.5)	16.0(5.7)	31(6.4)
S. glutinosa	12.9(7.3)	12.8(3.3)	19.3(4.8)	22.0(0.0)		14.6 (5.7)	13.3 (2.2)	18.1 (1.6)	22.3 (0.8)

Effect of dry storage and cold-wet stratification

The tested species *S. gigantea, S.sarracenicus, T. vulgare* and *U. dioica* differed within the subplots with and without cold-wet stratification and showed a high number of germinated seeds (>65%). The germination rates of seeds of the species *S. gigantea, S. sarracenicus,* and *U. dioica* without germination was not lower. The number of germinated seeds of *T. vulgare* without cold-wet stratification was significantly lower, and their germination occurred later (Figure 61) .

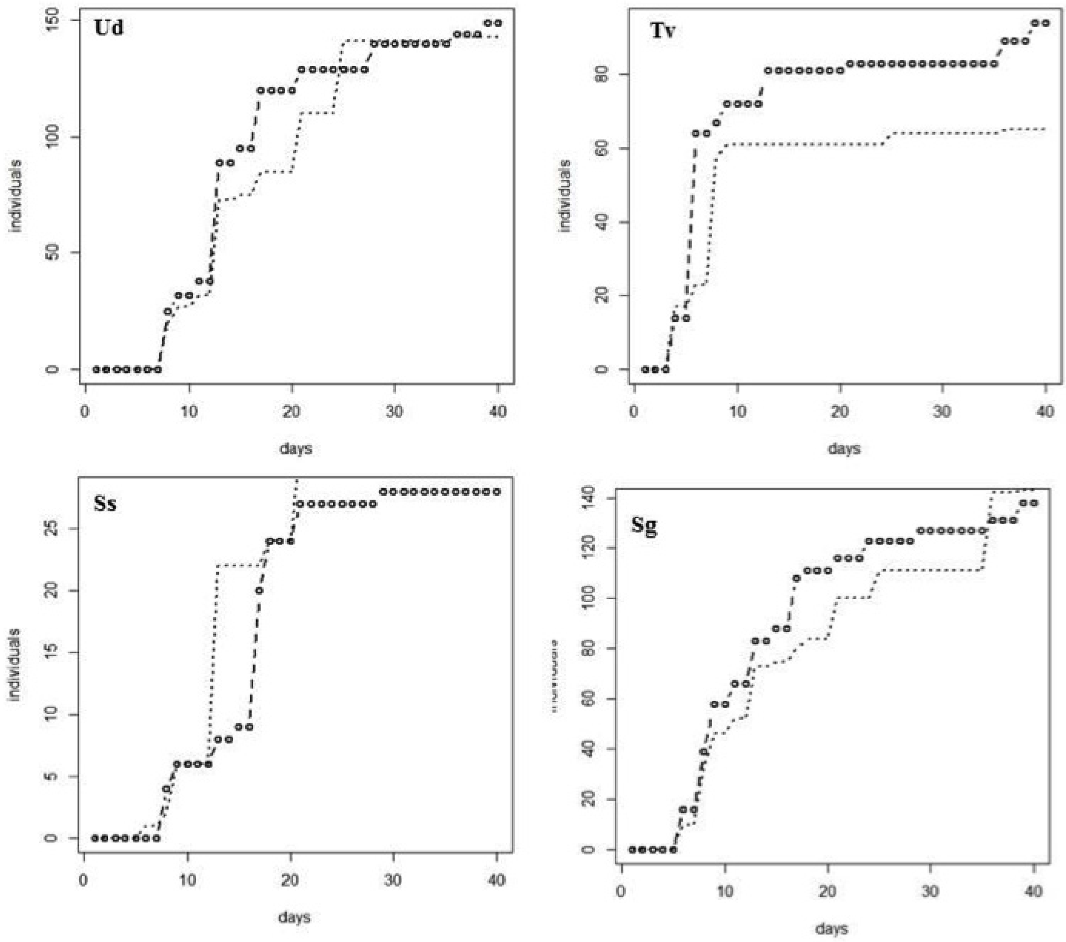
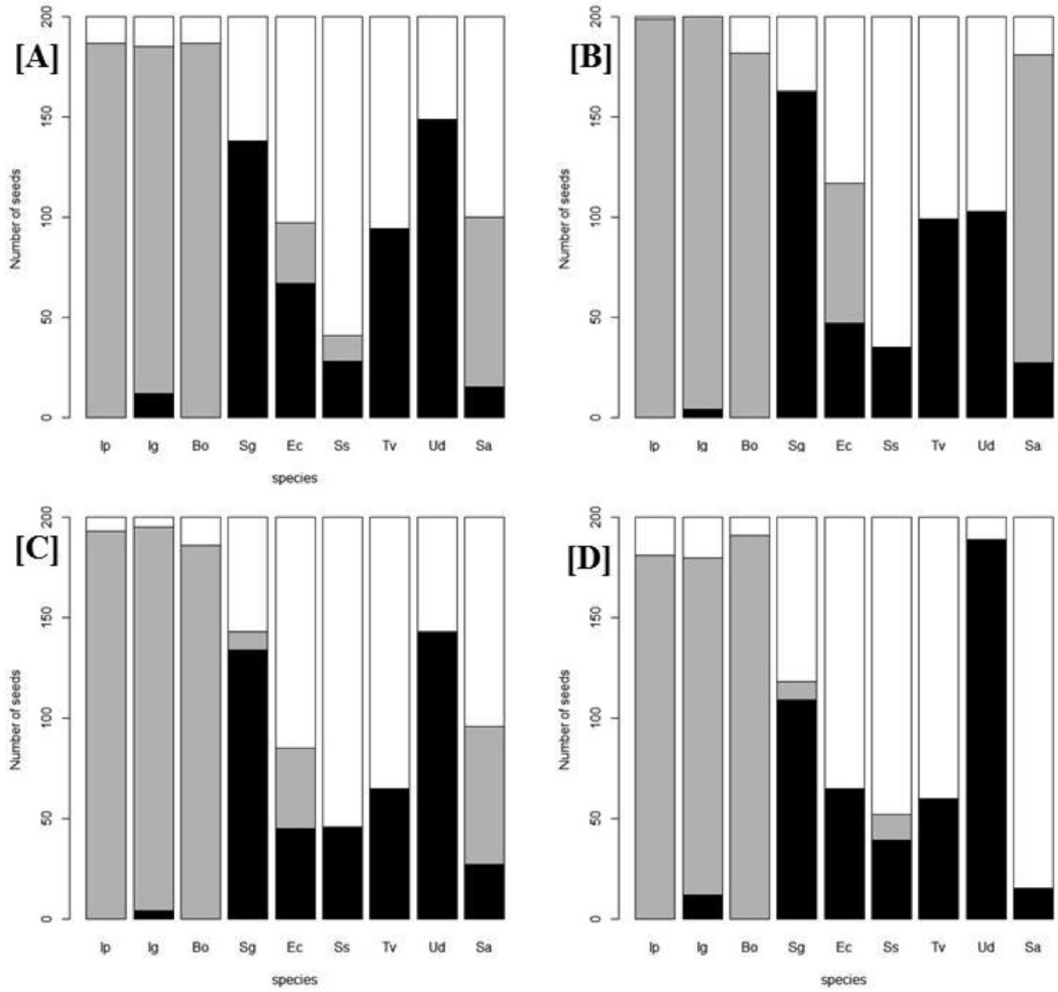


Figure 61 Number of germinated seeds of the species S. gigantea (Sg), S. sarracenicus (Ss), T. vulgare (Tv) and U. dioica (Ud) with cold-wet stratification (--°°--) and without cold-wet stratification (…..) (n=400 seeds per species), over the course of 40 days.

Vitality of seeds

The analysis of viability using the tetrazolium test showed a high number of dormant seeds among the ungerminated seeds of the invasive species *I. glandulifera*, *I. parviflora* and *B. orientalis*. Large differences in the percentage of dead seeds were detectable for *S. gigantea* with cold-wet stratification, where 4% of the seeds at 25/10°C were dead and 58% at 15/5°C were dead (Table 28). *U. dioica* showed a large disparity in the percentage of dead seeds relative to temperature. At 25/10°C, 49% of seeds with cold-wet stratification and 48% without were dead, while at 15/5°C these numbers dropped to 2% and 9% respectively. Seed vitality of the native species *S. glutinosa* also differed depending on germination temperature. At 15/5°C, 95% (with stratification) and 90% (without) of seeds were dead, while at 25/10°C, the numbers decreased to 5% and 14% respectively (Figure 62).



Figur 62 Absolute numbers of dead (white), dormant (gray) and germinated (black) seeds of invasive alien species I. glandulifera (Ig), I. parviflora (Ip), S. gigantea (Sg) and native species E. cannabinum (Ec), S. sarracenicus (Ss), T. vulgare (Tv), U. dioica (Ud), and S. glutinosa (Sa) comparing the subplots of seeds with cold-wet stratification [A], without cold-wet stratification [B], seeds at germination temperature at 15/5°C [C] and seeds at germination temperature at 25/10°C [D].

Table 28 Percentage of seeds of all tested species which had germinated or were dormant (not dead) after 40 days at 25/10°C and 15/5°C, and in groups of seeds with cold-wet stratification and without. Values are averages across sites based on 2 replicates per species and site and 50 seeds per Petri dish

Species	with cold stratification				without cold statification		
	not dead [%]		difference		not dead [%]		difference
	25/10 °C	15/5 °C	%		25/10 °C	15/5 °C	%
I. parviflora	99	88	11		100	93	7
I. glanduliferla	100	85	15		100	95	5
B. orientalis	93	94	-1		89	97	-8
S. gigantea	96	42	54		67	76	-9
E. cannabinum	66	31	35		51	34	17
S. sarracenicus	8	33	-25		27	18	9
T. vulgare	39	47	-8		52	13	39
U. dioica	39	98	-59		33	91	-58
S. glutinosa	95	5	0,9		86	10	76

Table 29 Percentage of seeds of all tested species which were dead after 40 days at 25/10°C and 15/5°C, and in groups of seeds with cold-wet stratification and without. Values are averages across sites based on 2 replicates per species and site and 50 seeds per Petri dish.

Species	with cold stratification				without cold statification		
	dead [%]		difference		dead [%]		differenc
	25/10 °C	15/5 °C	%		25/10 °C	15/5 °C	%
I. parviflora	1	12	-11		0	7	-7
I. glandulifera	0	15	-15		0	5	-5
B. orientalis	7	6	1		11	3	8
S. gigantea	4	58	-54		33	24	9
E. cannabinum	34	69	-35		49	66	-17
S. sarracenicus	92	67	25		73	81	-8
T. vulgare	53	53	0		48	87	-39
U. dioica	49	2	47		48	9	39
S. glutinosa	5	95	-90		14	90	-76

3.7 Soil Seed bank

A total number of 89 plots were recorded from the combined sampling of the aboveground vegetation coverage and the density of seeds of invasive alien plants. In total 163 seeds of the species *Impatiens glandulifera*, 83 seeds of *Impatiens parviflora*, 438 seeds of *Solidago gigantea* and 34 seeds of *Bunias orientalis* were detected. Results show the occurrence of detected seeds by soil layer 0-5cm, 5-10cm and 10-20cm. The most amount of seeds (n=499) were detected in the upper layer (0-5cm). In the mid-layer (5-10cm) in total 157 seeds were found and in the lowest layer (10-20cm) 61 seeds of invasive alien plants were detected.

All detected species have the highest amount of detected seeds in the highest layer (0-5cm). *Solidago gigantea* was the most abundant with a total of 318 seeds in all samples of layer 0-5cm, followed by 102 seeds of *Impatiens glandulifera*, 60 seeds of *Impatiens parviflora*, and 19 seeds of *Bunias orientalis.*

The total number of samples of the mid layer 5-10cm 157 seeds were detected. The highest amount of most abundant seeds was from the invasive alian plant species *Solidago gigantea* with a total number of 88 seeds.

In the lowest layer 10-20cm the fewest amount of seeds (n=61) was detected Solidago gigantea was the most abundant with a total of 32 seeds in all samples of layer 10-20cm, followed by 24 seeds of *Impatiens glandulifera* and 5 seeds of *Impatiens parviflora*. Of *Bunias orientalis* no seeds were detected in this lowest layer.

The analysis contamination of soil with seeds of invasive alien plant species for plots within different constructional measures shows a significant occurrence of seeds in all categories. The abundance of *Solidago gigantea* was most dominant. Especially in plots of the categories buffer zone and lowering a high abundance was recorded. The results show, that the abundance of seeds is getting lower with the deep. In the lowest layer 10-20cm the seeds of *Solidago gigantea* are most abundant. It is discuss, that although the number of seeds is low in the layer 10 to 20 cm, the contamination of soil with seeds of invasive alien plants is present.

The results of the germination of *Solidago gigatnea* test showed, that 717 seeds (n=396) germinated. The highest germination rate is from the species *Solidago gigantea,* due to the high abundance of seeds in all layers. In total 68.00% seeds of Solidago gigatea germinated. The total number of all germinated seeds decrease with the depth.

The analysis of the germination rate showed that 56.44% of the seeds of *Impatiens glandulifera* germinated. In total 24.39% of the seeds of *Impatiens parviflora* and 68.00% of the total amount of seeds of *Solidago gigantea* germinated positive. Further 44.12% of seeds of *Bunias orientalis* germinated during the germination test.

The analysis of the course of germination shows that 80% of germinable seeds germinated in the first 10 days. Seeds of *Impatiens glandulifera* and *Impatiens parviflora* reached the highest number of posive germinated seeds during the 10th and the 20th day. *Bunias orienatlis* was the last detected species that started to germinate. The first seed of *Bunias orientalis* germinated on the 16th day of the germination test.

Table 30 Total number of seeds of of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis, mean and standard diviation (+SD)recorded in the layers 0-5cm, 5-10cm and 10-20cm.

	0 to 5 cm			5 to 10cm			10-20 cm			
Species	**n**	**Mean**	**+ SD**	**n**	**Mean**	**+ SD**	**n**	**Mean**	**+ SD**	**Total**
Impatiens glandulifera	102	1.45	3.90	37	0.42	1.87	24	0.27	1.43	163
Impatiens parviflora	60	0.67	2.48	17	0.19	0.93	5	0.06	0.28	82
Solidago gigantea	318	3.57	9.00	88	0.99	4.69	32	0.36	1.40	438
Bunias orientalis	19	0.21	1.22	15	0.17	1.29	0	0.00	0.00	34

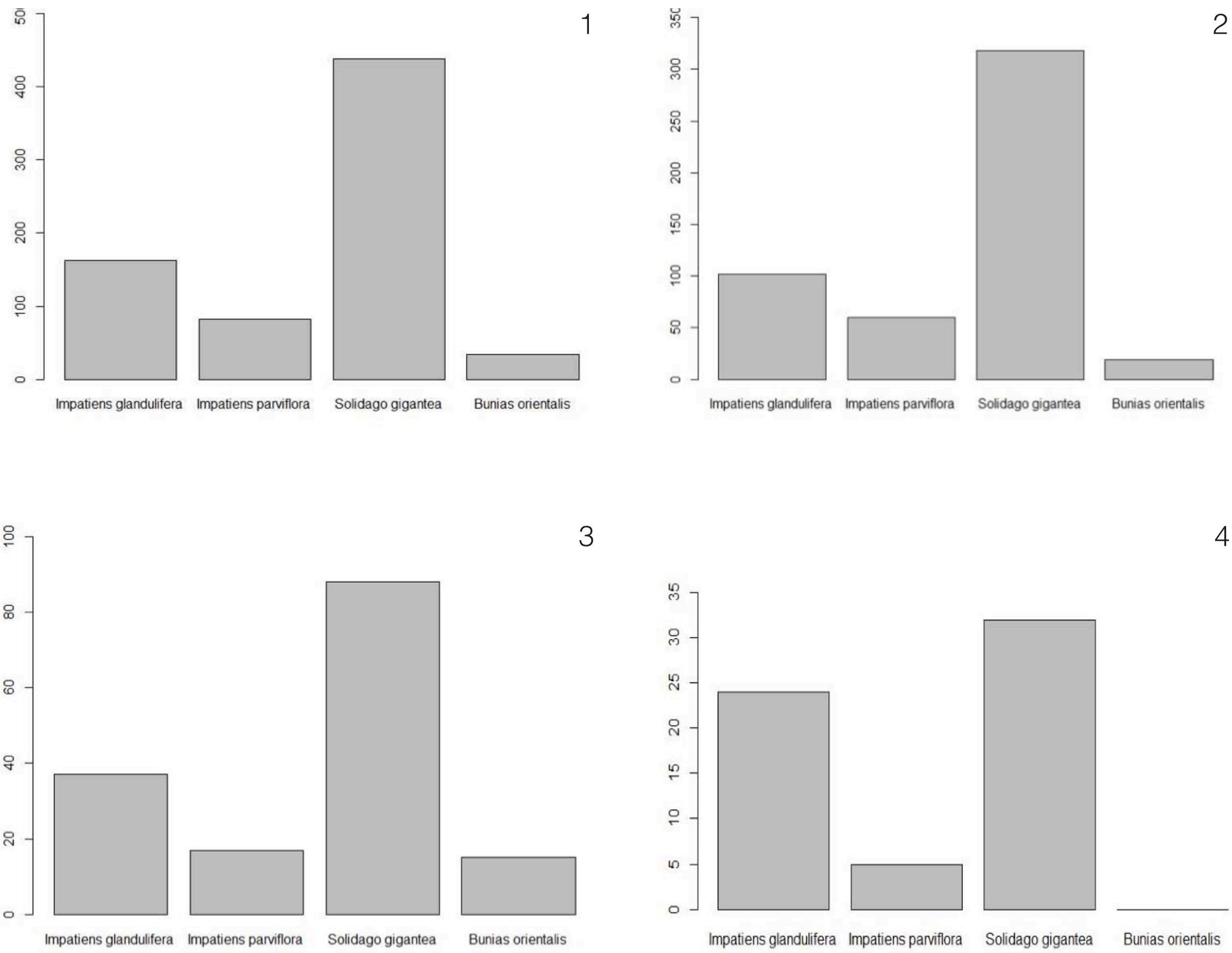


Figure 63 (1) Total number of detected seeds of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis.

(2) Total number of seeds of of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis in all samples of the layer 0-5cm.

(3) Total number of seeds of of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis in all samples of the layer 5-10cm.

(4) Total number of seeds of of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis in all samples of the layer 10-20cm.

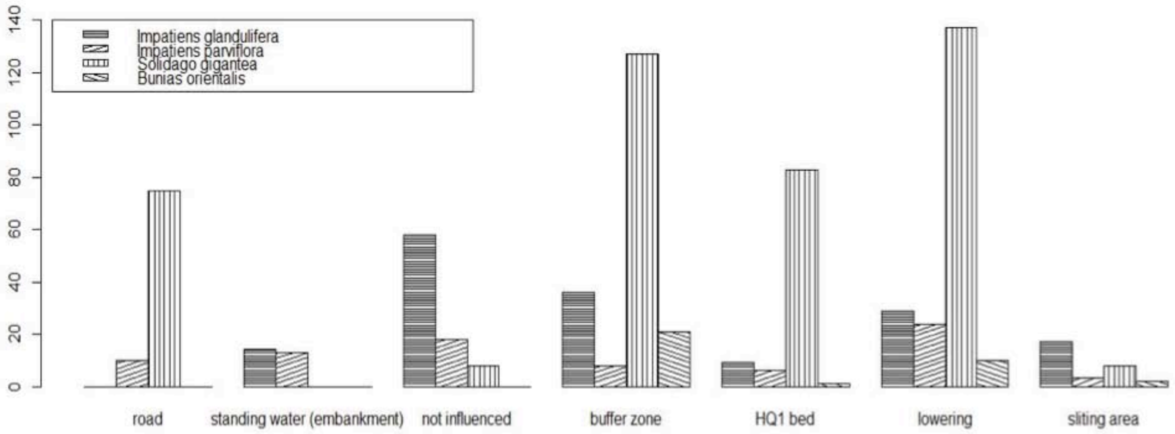


Figure 64 Total number of seeds of of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis for all samples of all layers within different constructional measures.

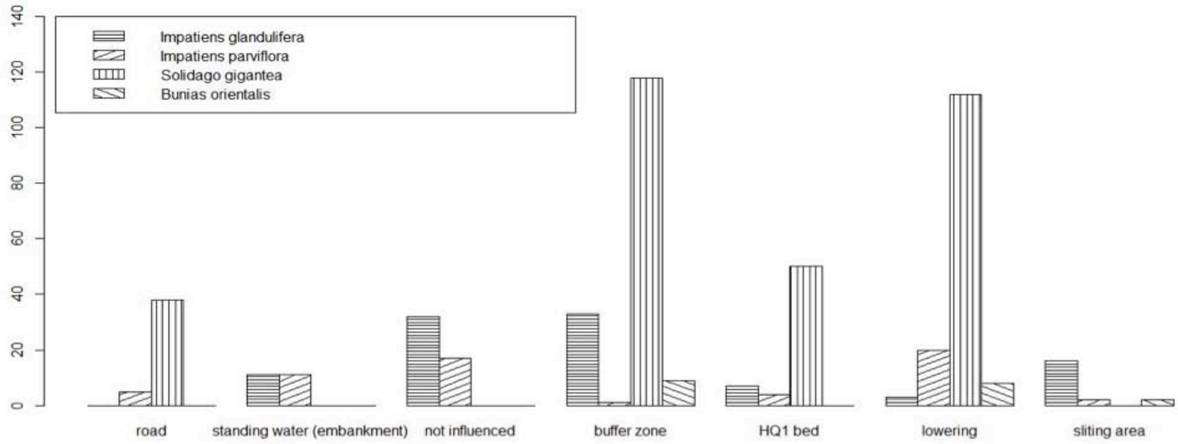


Figure 65 Total number of seeds of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis for all samples of the layer 0 to 5cm within different constructional measures.

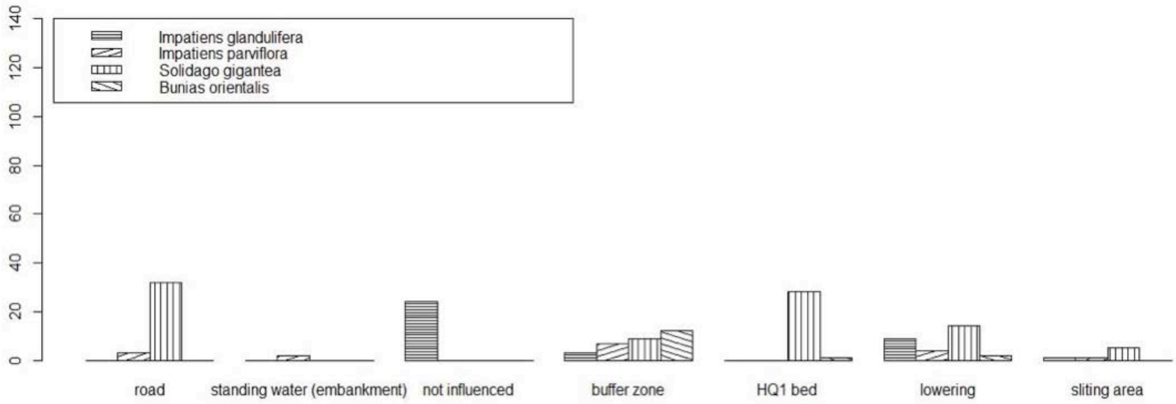


Figure 66 Total number of seeds of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis for all samples of the layer 5 to 10cm within different constructional measures.

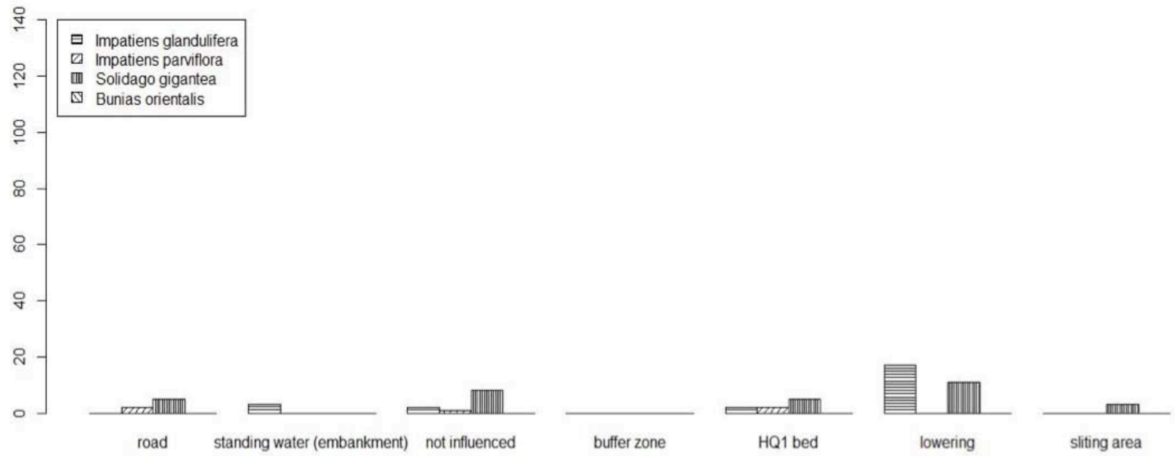


Figure 67 Total number of seeds of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis for all soil samples of the layer 10 to 20cm within different constructional measures.

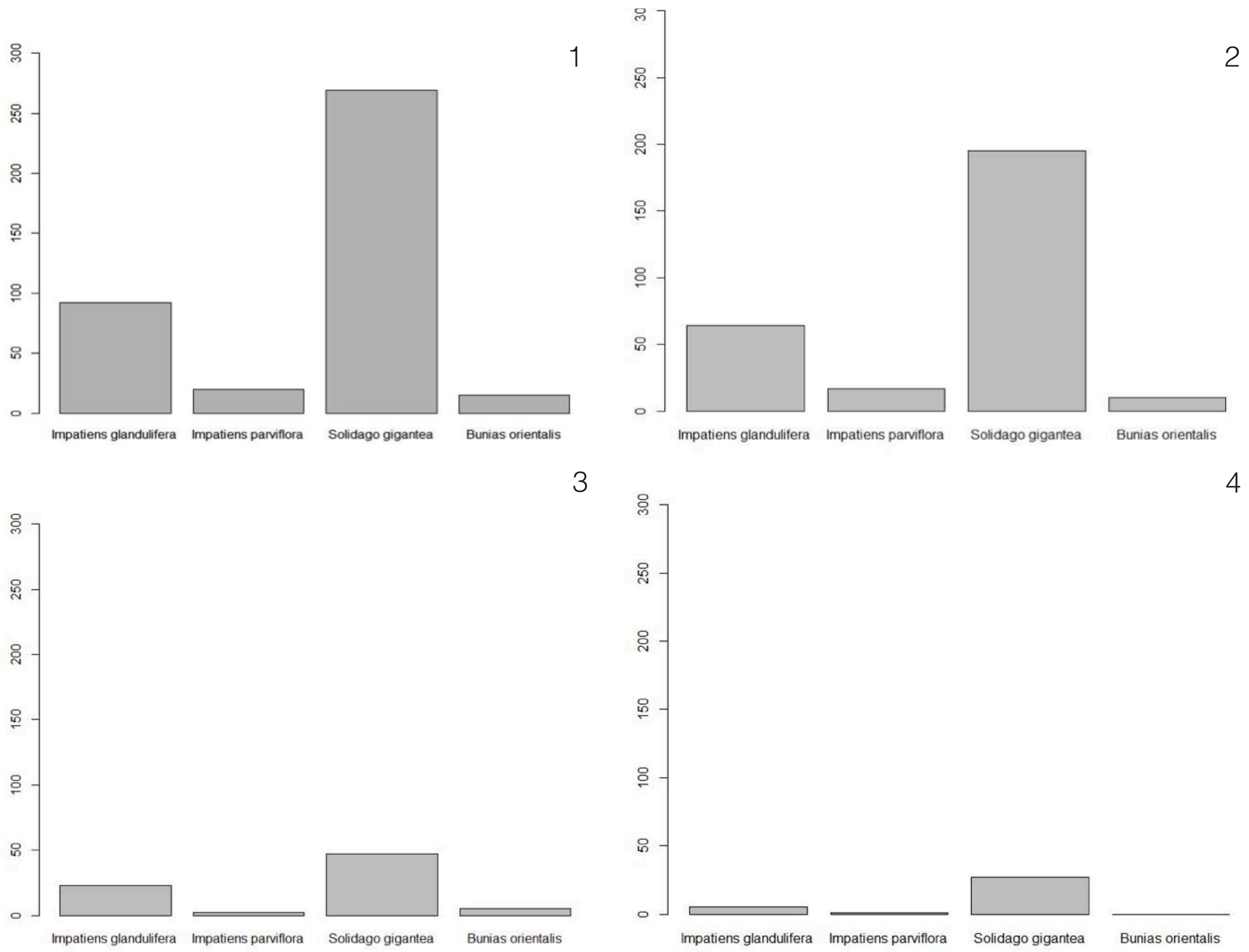


Figure 68 (1) Total number of germinated seeds of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis for all samples of all layers.

(2) Total number of germinated seeds of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis for all samples of the layer 0 to 5cm.

(3) Total number of germinated seeds of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis for all samples of the layer 5 to 10cm.

(4) Total number of germinated seeds of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis for all samples of the layer 10 to 20cm.

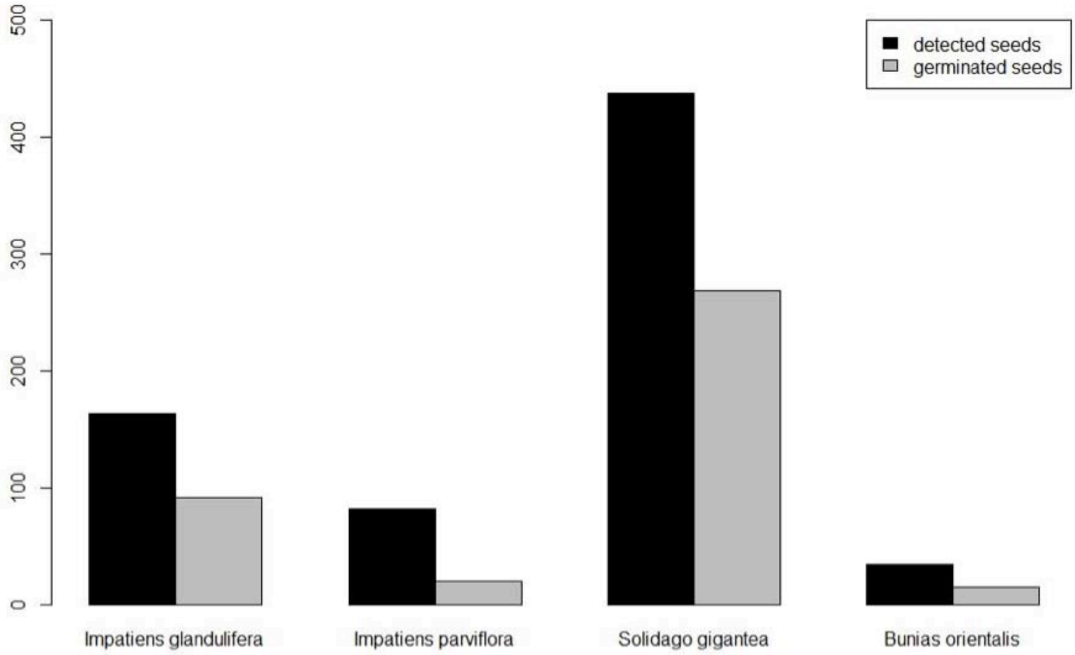


Figure 69 Comparison of the total number of detected seeds and the total number of germinated seeds of Impatiens glandulifera, Impatiens parviflora, Solidago gigantea and Bunias orientalis for all samples of all layers.

3.8 Aboveground coverage and seed bank occurrence

Invasive alien plant species *Bunias orientalis, Impatiens glandulifera, Impatiens parviflora* and *Solidago gigantea* dominated the plant cover of the the herb layer. Together *Bunias orientalis, Impatiens glandulifera, Impatiens parviflora* and *Solidago gigantea* comprised in average 71% of the total cover of the herb layer in the study year 2011. In the study year 2012 84% (n=89 plots) were covered by *Bunias orientalis, Impatiens glandulifera, Impatiens parviflora* and *Solidago gigantea*. In average (n=89 plots) 1.8 seeds of *Bunias orientalis*, 0.9 seeds of *Impatiens glandulifera*, 4.9 seeds of *Impatiens parviflora* and 0.4 seeds of *Solidago gigantea* were detected. *Solidago gigantea* was the most abundant invasive species. Generally species richness (Shannon index) and evenness (Pilou´s index) decreased as the occurrence of seeds of invasive alien species in the seed bank increase. Within each analyzed invasive alien species differences were detectable.

The total cover of the herb layer was tendential negative but not significantly related to occurrence of seeds of *Impatiens glandulifera* ($P=0.226$, $r^2=0.01679$) (Figure 70). The occurrence of seeds of *Impatiens parviflora* was not significantly related to the total cover of the herb layer ($P=0.851$, $r^2= 0.00041$). The occurrence of seeds of *Solidago gigantea* show a positive but not significantly relationship to the total aboveground cover of the herb layer ($P=0.6537$, $r^2= 0.00232$) (Figure 72). The occurrence of seeds of *Bunias orientalis* show a positive relationship to the total aboveground cover of the herb layer ($P=0{,}667$, $r^2= 0.002137$) (Figure 73).

The Shannon diversity index of aboveground plant species composition, representing the species diversity of each plot, is negative related to the seed diversity of *Impatiens parviflora* ($P=0.1273$, $r^2=0.02652$). The species diversity of each plot showed a significant negative relationship with seed density of Impatiens glandulifera ($P=0.03$, $r^2=0.05262$). The seed density of *Impatiens glandulifera* increased, while the species diversity decreased. There is no relationship between species diversity and seed density of *Solidago gigantea* ($P=0.7396$, $r^2=0.1112$). The seed density of *Bunias orientalis* show a positive not significant relationship to the species diversity index ($P=0.2002$, r^2=0.0.879).

The Pilou´s index of evenness (Magurran 1988), representing the evenness of the aboveground species composition in the herb layer of analyzed plots, is negative related to seed density of *Impatiens glandulifera* in the study year 2012 but not related in the study year 2011 ($P=0.546$, $r^2=0.0418$) (Figure 78). The evenness index and the seed density of *Impatiens parviflora* (P=0.2213, r^2=0.01714) showed a negative relationship (Figure 79). There was no relationship between the evenness index and the seed density of *Solidago gigantea* (P=0.6211, r^2=0.002821) (Figure 80).

The seed density of *Bunias orientalis* shows a positive not significant relationship to the evenness index (P=0.587, r^2=0.0034) (Figure 81).
The occurrence of seeds of invasive alien plants is not significant for the increase of invasive alien plants in the coverage of the herb layer of the year 2011 to the year 2012 (Increase in coverage - presence of seeds: E[x] = 0.26 < 5; increase in coverage – no seeds: Fisher exact test E[x] = 3.73 < 5; decrease in coverage - presence of seeds: Fisher exact test E[x] = 5.73 > 5; decrease in coverage - no seeds: Fisher exact test E[x] = 79.27 > 5). The performed Fischer´s exact test (Agresti 1992) showed results no significant relation between the detected seed occurrence and increase or decrease of coverage of invasive alien plant species in the herb layer from the year 2011 to 2012 (P =0.247).

The results show the presents of seeds of invasive species in the soil seed bank. But the spreading of invasive alien plats that were sampled aboveground, into not invaded areas is not related to the occurrence in the seed bank.

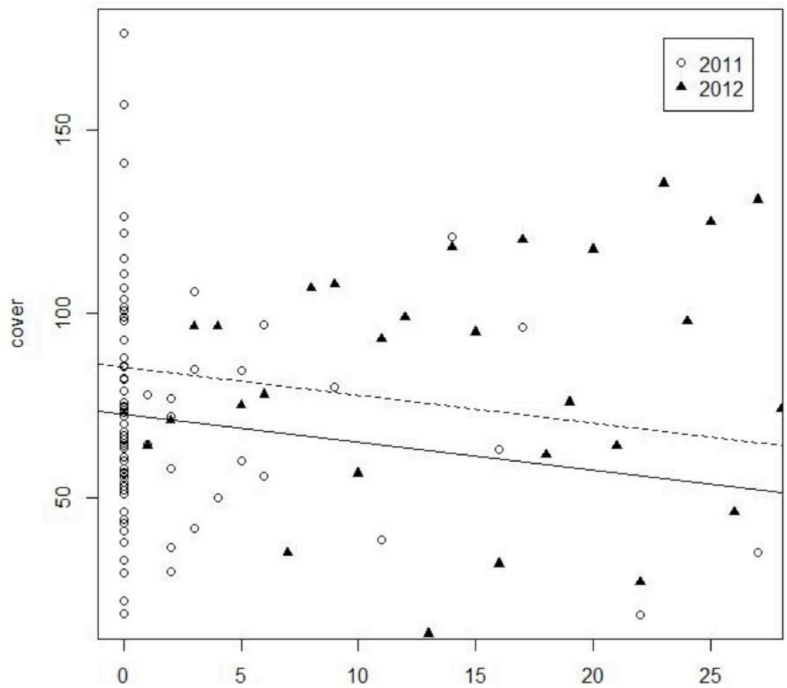


Figure 70 Regression analysis of the relationship between mean aboveground percent cover of the herb layer (2011 and 2012) and seeds of Impatiens glandulifera.

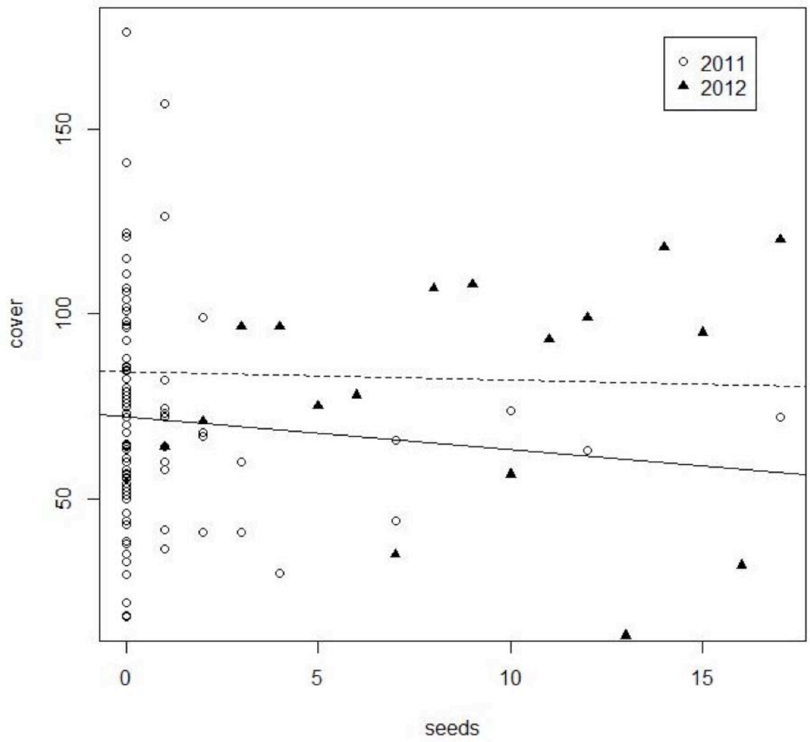


Figure 71 Regression analysis of the relationship between mean aboveground percent cover of the herb layer (2011 and 2012) and seeds of Impatiens parviflora.

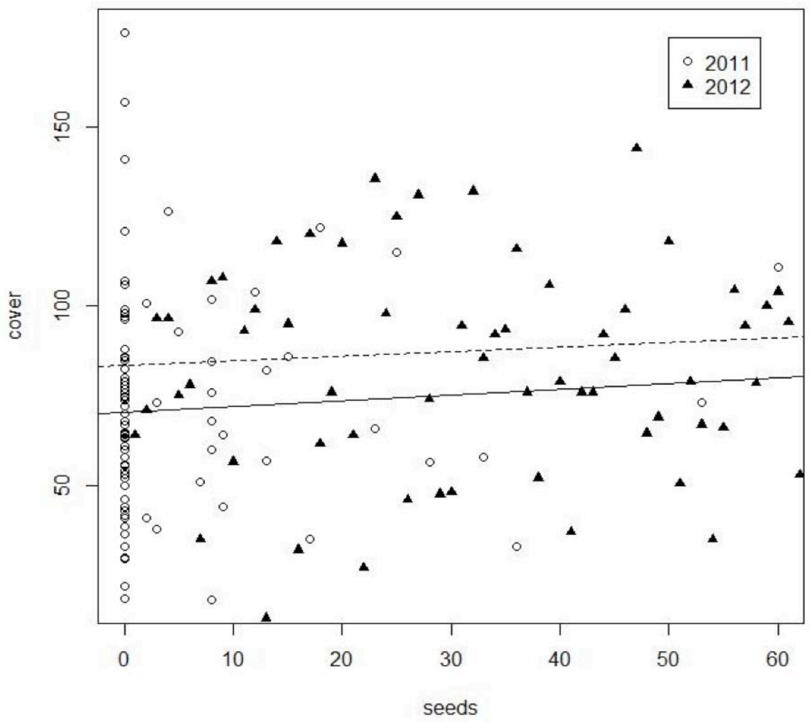


Figure 72 Regression analysis of the relationship between mean aboveground percent cover of the herb layer (2011 and 2012) and seeds of Solidago gigantea.

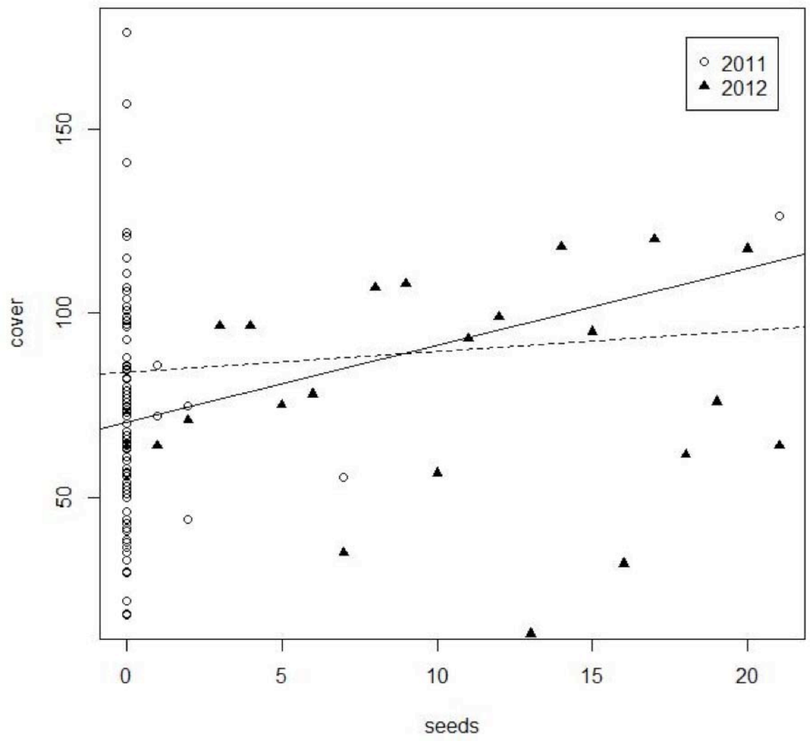


Figure 73 Regression analysis of the relationship between mean aboveground percent cover of the herb layer (2011 and 2012) and seeds of Bunias orientalis.

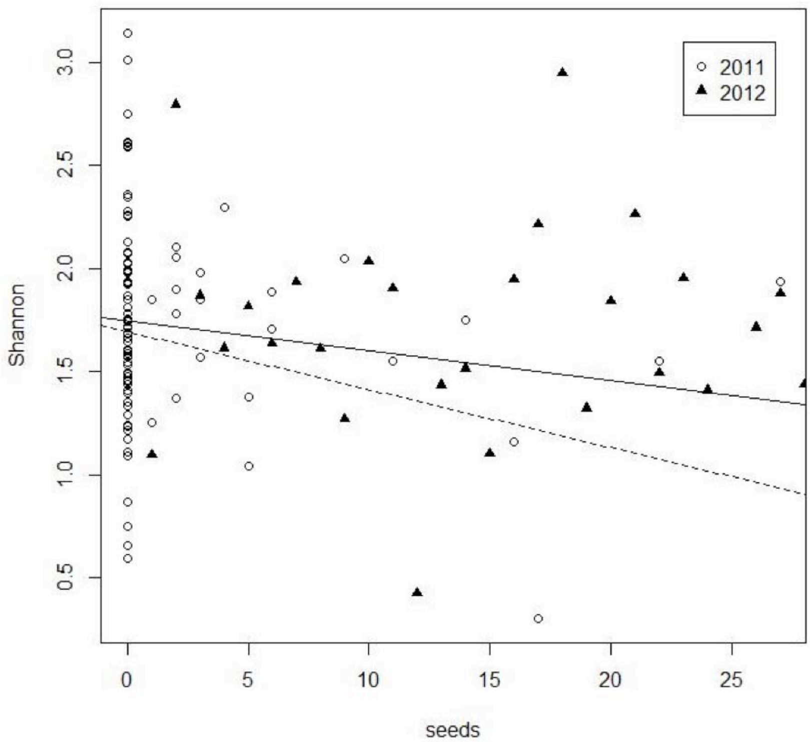


Figure 74 Regression analysis of the relationship between mean seed density of Impatiens glanduli-fera and mean Shannon diversity index of aboveground plant species composition in the study year 2011 and 2012.

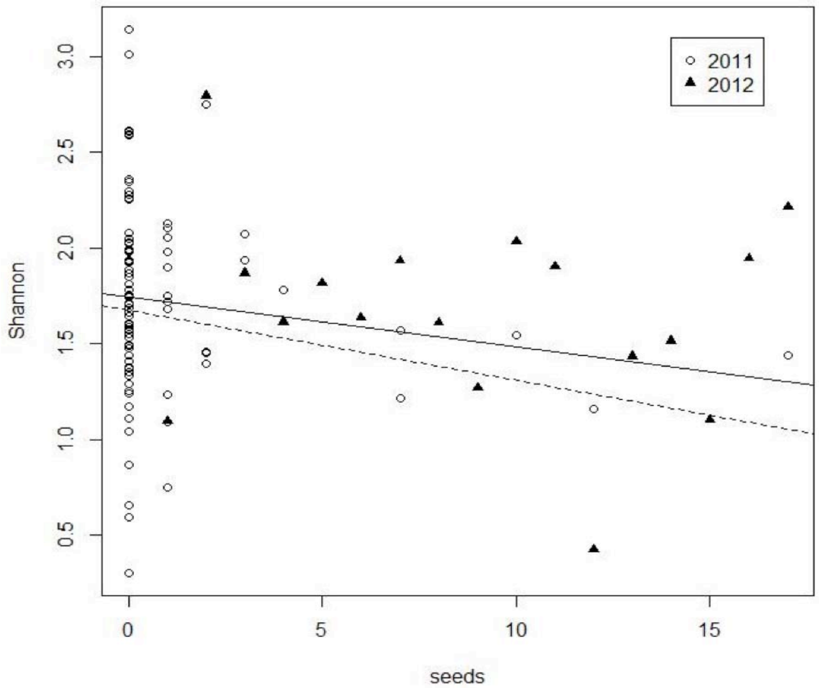


Figure 75 Regression analysis of the relationship between mean seed density of Impatiens parviflora and mean Shannon diversity index of aboveground plant species composition in the study year 2011 and 2012.

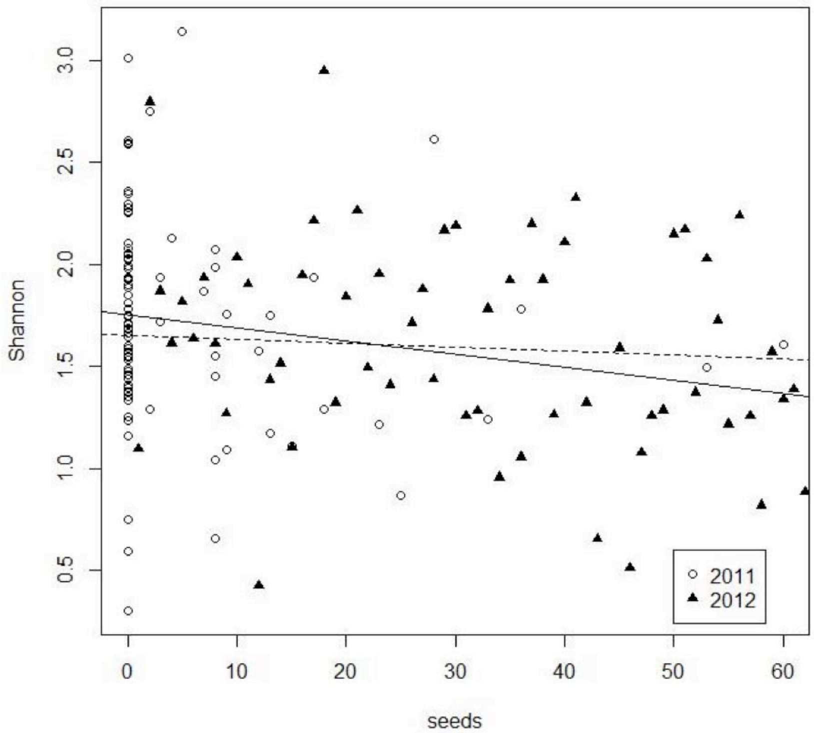


Figure 76 Regression analysis of the relationship between mean seed density of Solidago gigantea and mean Shannon diversity index of aboveground plant species composition in the study year 2011 and 2012.

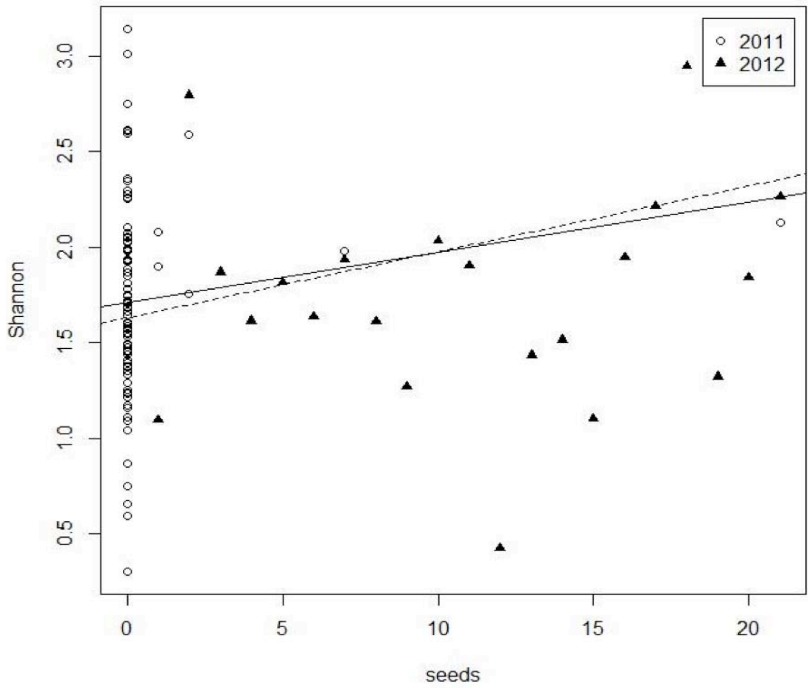


Figure 77 Regression analysis of the relationship between mean seed density of Bunias orientalis and mean Shannon diversity index of aboveground plant species composition in the study year 2011 and 2012.

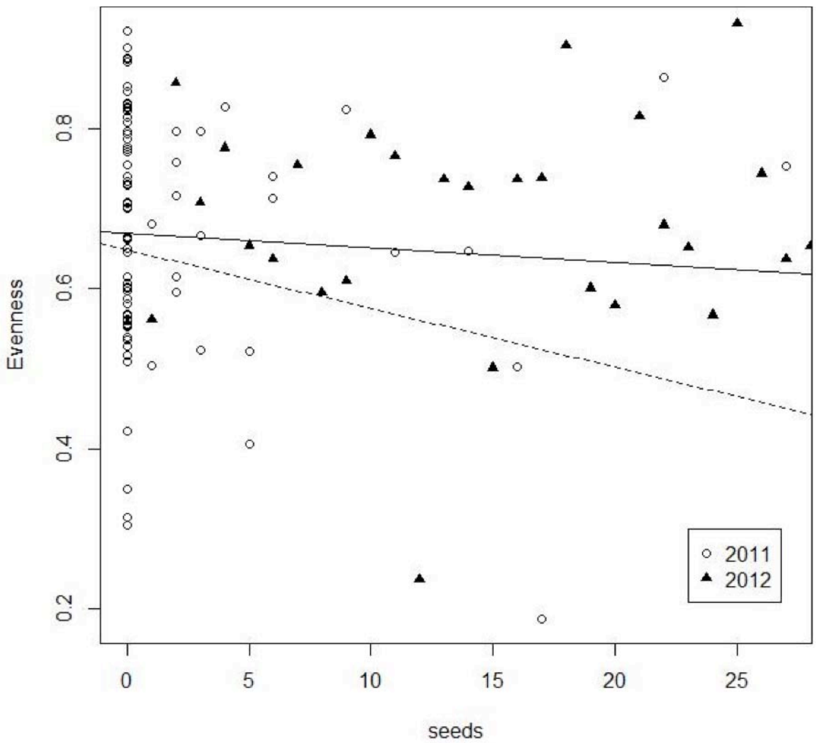


Figure 78 Regression analysis of the relationship between mean seed density of Impatiens glandulifera and mean Pielou`s evenness index of aboveground plant species composition in the study year 2011 and 2012.

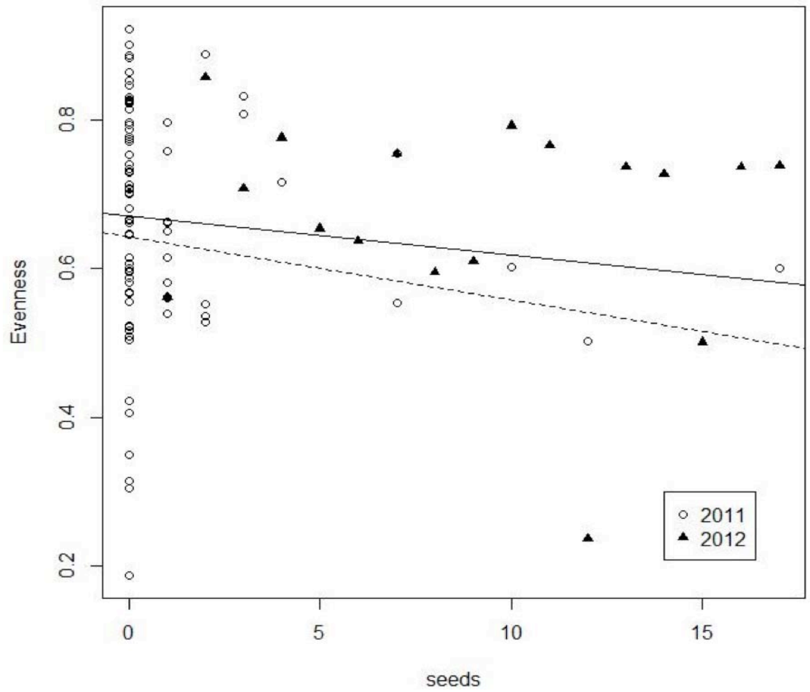


Figure 79 Regression analysis of the relationship between mean seed density of Impatiens parviflora and mean Pielou`s evenness index of aboveground plant species composition in the study year 2011 and 2012.

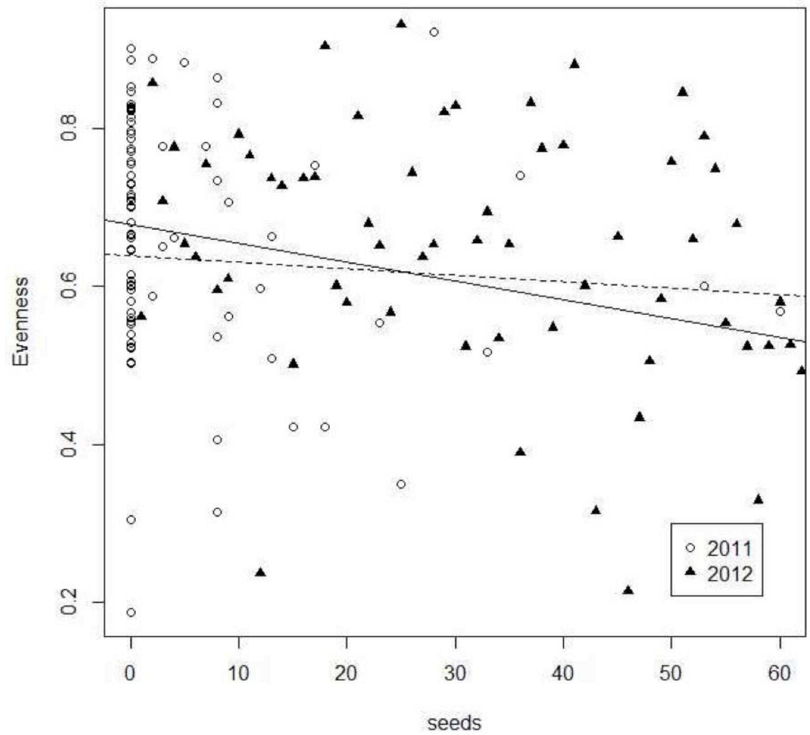


Figure 80 Regression analysis of the relationship between mean seed density of Solidago gigantea and mean Pielou`s evenness index of aboveground plant species composition in the study year 2011 and 2012.

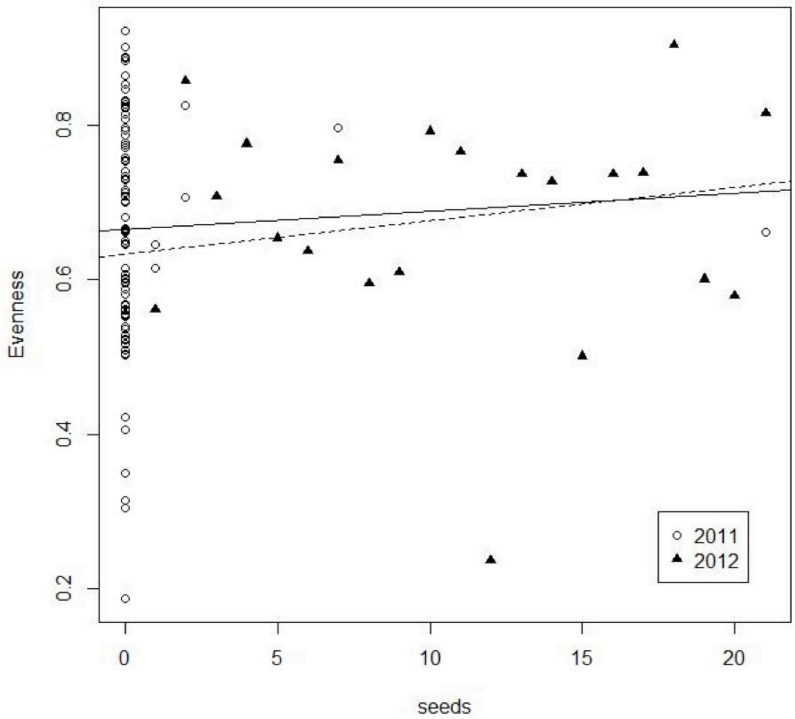


Figure 81 Regression analysis of the relationship between mean seed density of Bunias orientalis and mean Pielou`s evenness index of aboveground plant species composition in the study year 2011 and 2012.

3.9 Phenological observation

Vegetative development

The results show clearly the character of each observes invasive species. The first emerge of shoots were observed at the invasive species *Bunias orientalis*. The emerging shoots of *Bunias orientalis* were detected in April on both sites. In June the individuals of the *Bunias orientalis* –populations were fully developed (Figure 82). In the end of august the above ground plant components were died back, apart from the base leaf. The base leaf did not disappear until the end of the observation period.

The vegetative development of the species *Impatiens glandulifera* starts very fast in May (Figure 83). There are no differences between the two observed populations. Within two week the first shoot emerged and the leaves were developed up to 25%. In August the plants are vegetative fully developed. In October more than 50% of the vegetative plant components turned yellow.

The vegetative development of the invasive species *Impatiens parviflora* starts in May (Figure 84). Within one month the mean of the individuals of the population were vegetative fully developed. In June the plants were vegetative fully developed. The yellowing starts in the end of August. In the end of September more than 50% of the leaves are turned yellow. The vegetative components of the plants disappear in the beginning of October.

The vegetative development of the species *Solidago gigantea* starts in Mai very fast. There are no significant differences between the four observed populations. Within two week the first shoot emerged and the leaves were developed up to 25%. The populations of *Solidago gigantea* are vegetative fully developed in June. In the End of August the yellowing starts and lasts until October, when the plants start to die back (Figure 85).

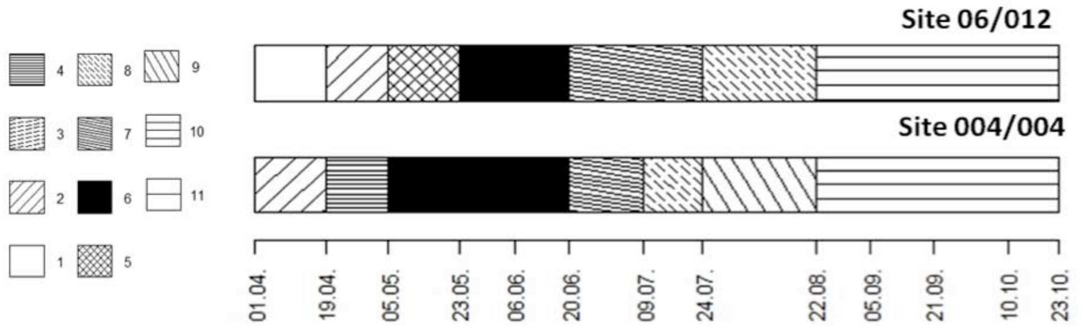


Figure 82 vegetative development of Bunias orientalis; from April to the end of October. Each site is pictured separately in a bar. The development is documented within 11 stages, following a defined scale of generative development process.

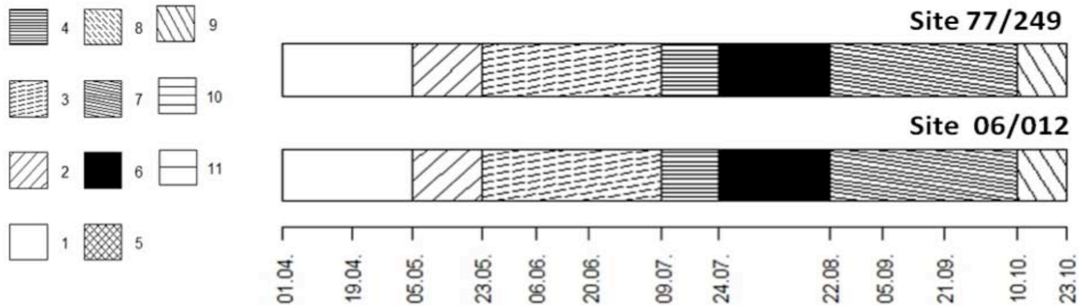


Figure 83 vegetative development of Impatiens glandulifera; from April to the end of October. Each site is pictured separately in a bar. The development is documented within 11 stages, following a defined scale of generative development process

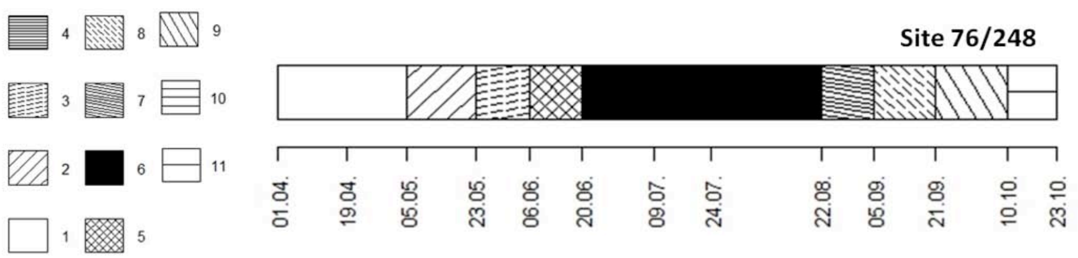


Figure 84 vegetative development of Impatiens parviflora; from April to the end of October. Each site is pictured separately in a bar. The development is documented within 11 stages, following a defined scale of generative development process.

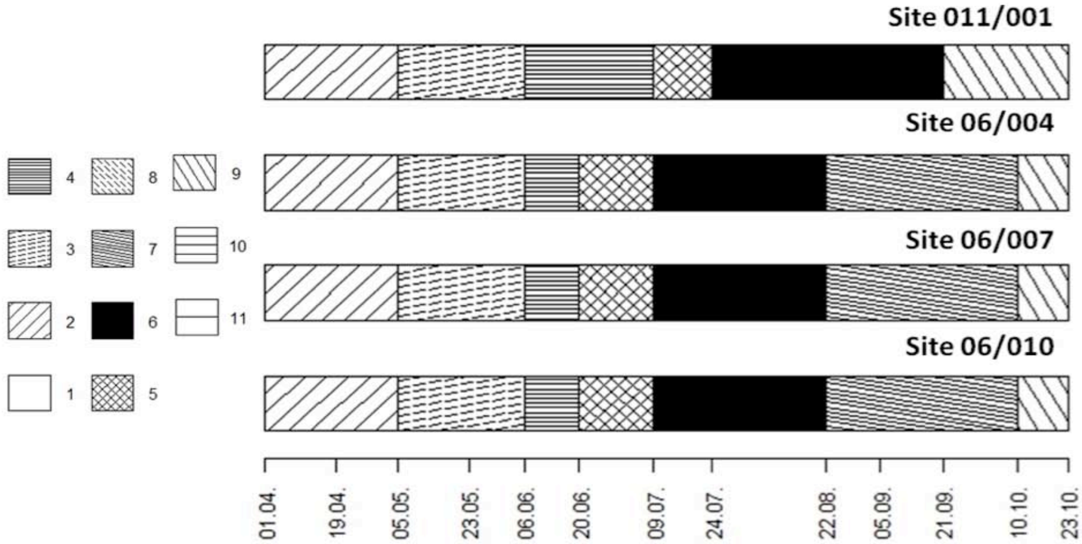


Figure 85 vegetative development of Solidago gigantea; from April to the end of October. Each site is pictured separately in a bar. The development is documented within 11 stages, following a defined scale of generative development process (Tab. 2).

Generative development

The generative development of *Bunias orientalis* stars in April. There are differences to detect between the two observed populations 06/012 and 004/004. The blooming of the population at the site 004/004 starts earlier (in April) then of the population at the site 06/012 (in May). The blooming finishes in both populations in the end of Mai. In June the fruits are mature. The distribution of seeds of the population 004/004 starts in July and of the population 06/012 in August (Figure 86).

The results of the observation of the generative development of *Impatiens glandulifera* show no relevant differences between the two observed sites. The emerge of visible flower buds starts in July. The flowering goes on in the end of July. The blooming finishes on both sites in October. The distribution of seeds of the mean of the population starts in October.

The generative development of *Impatiens parviflora* starts in the beginning of June (Figure.88). Very shortly after the emerging of the first visible flower bud in the beginning of the month June, the flowering starts in the end of the same month. The total blooming finishes in the End of July, when immediately the maturity of fruits begins. The distribution of seeds starts in the end of August.

The results of the observation of the generative development of *Solidago gigantea* show no significant differences between the populations of different sites (Figure 89). The first visible flower buds appear in June. In July the flower buds are clearly visible. The blooming starts in the beginning of June and lasts until October. The maturity of fruits and the distributing of seeds start in the end of October.

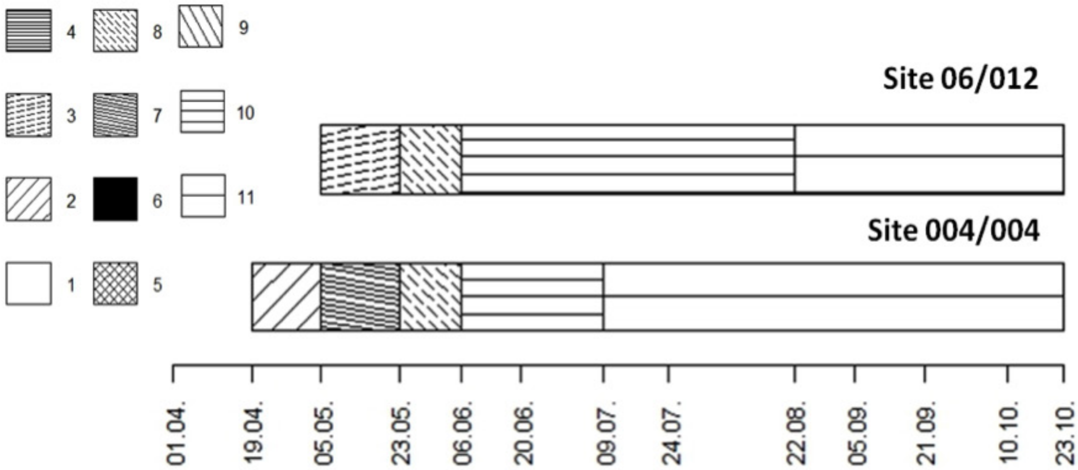


Figure 86 Generative development of Bunias orientalis; from April to the end of October. Each site is pictured separately in a bar. The development is documented within 11 stages, following a defined scale of generative development process .

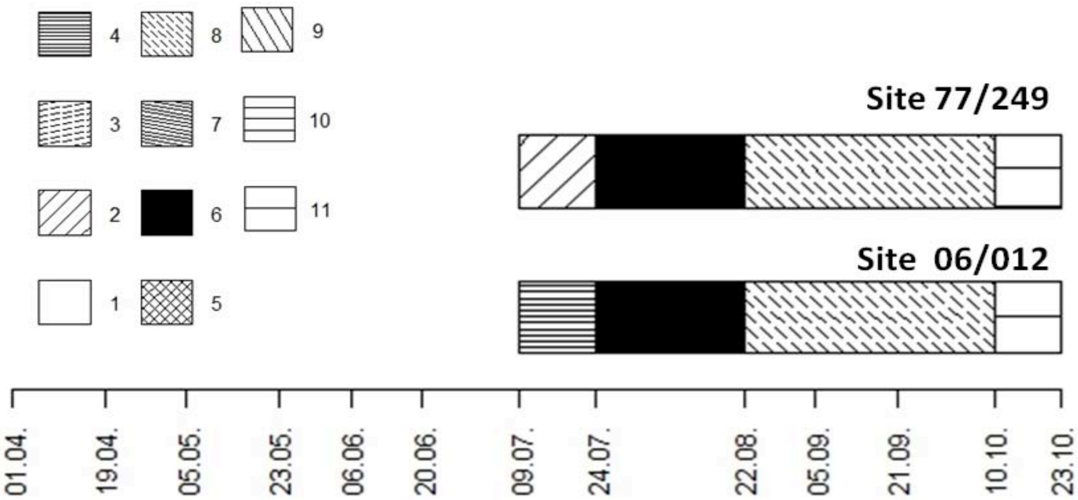


Figure 87 Generative development of Impatiens glandulifera; from April to the end of October. Each site is pictured separately in a bar. The development is documented within 11 stages, following a defined scale of generative development process.

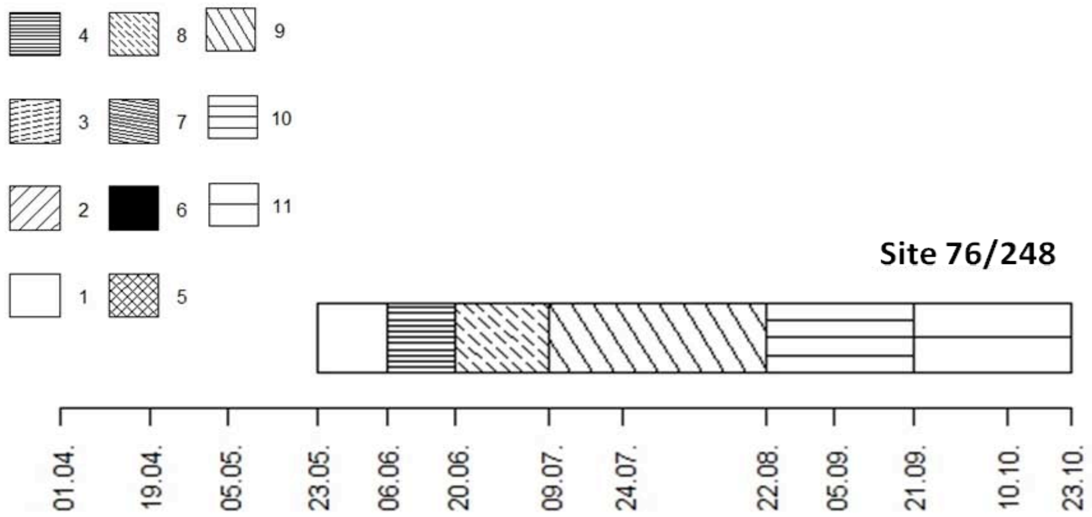


Figure 88 Generative development of Impatiens parviflora; from April to the end of October. Each site is pictured separately in a bar. The development is documented within 11 stages, following a defined scale of generative development process.

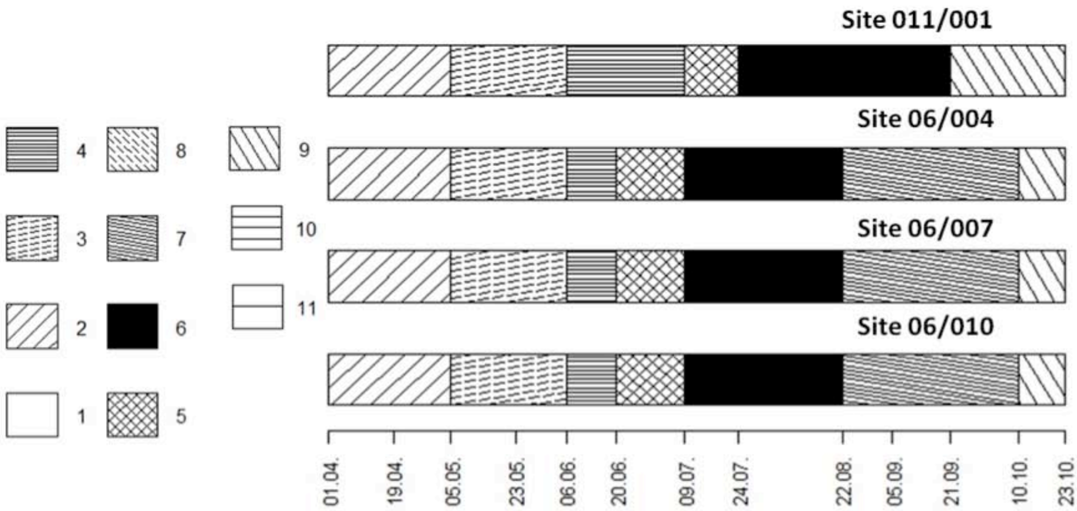


Figure 89 Gnerative development of Solidago gigantea ; from April to the end of October. Each site is pictured separately in a bar. The development is documented within 11 stages, following a defined scale of generative development process.

3.9.1 Growth characteristic of invasive species

Bunias orientalis

The comparison of the vegetative and the generative development of the invasive species *Bunias orientalis* shows, that the flower buds are visible 2 weeks after the emerging shoot, before the first leaf is developed. In the beginning of the month May *Bunias orientalis* is already vegetative fully developed and up to 50% of all flowers are blooming. The flowering lasts one month until of Mai. In June the fruits are mature. The distribution of seeds starts simultaneously with the dying back of the vegetative above ground plant components in July (Figure 90).

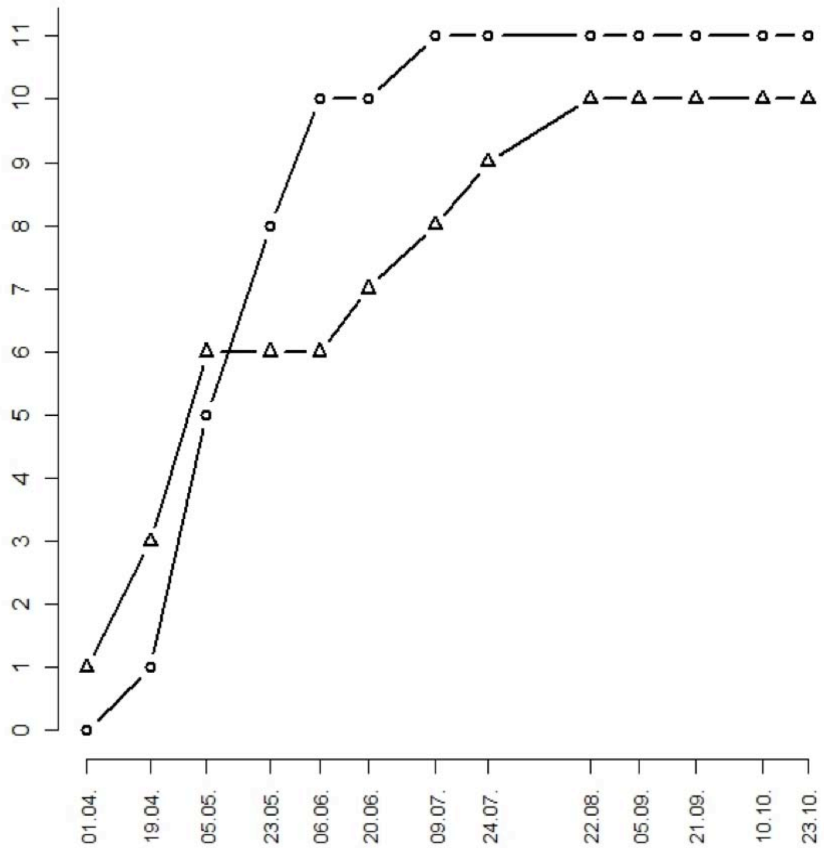


Figure 90 Vegetative and generative development of Bunias orientalis: the vegetative Line (Δ-symbol) and the generative Line (0) represents the phonological development measured in 12 stages (y-axis) during the observation period from the beginning of April to the end of October (x-axis).

Impatiens glandulifera

The comparison of the vegetative and the generative development of the invasive species *Impatiens glandulifera* shows, that the vegetative development starts two month earlier than the generative development, in the end of April. In June the first flower buds are clearly visible after more than 75% of the leaves developed. Once the vegetative plant components are fully developed, in August, the total blooming reaches its peak. The blooming lasts until the end of September when more than 50 of the leaves turned yellow. In the End of September the plant starts to die back above ground and to distribute seeds.

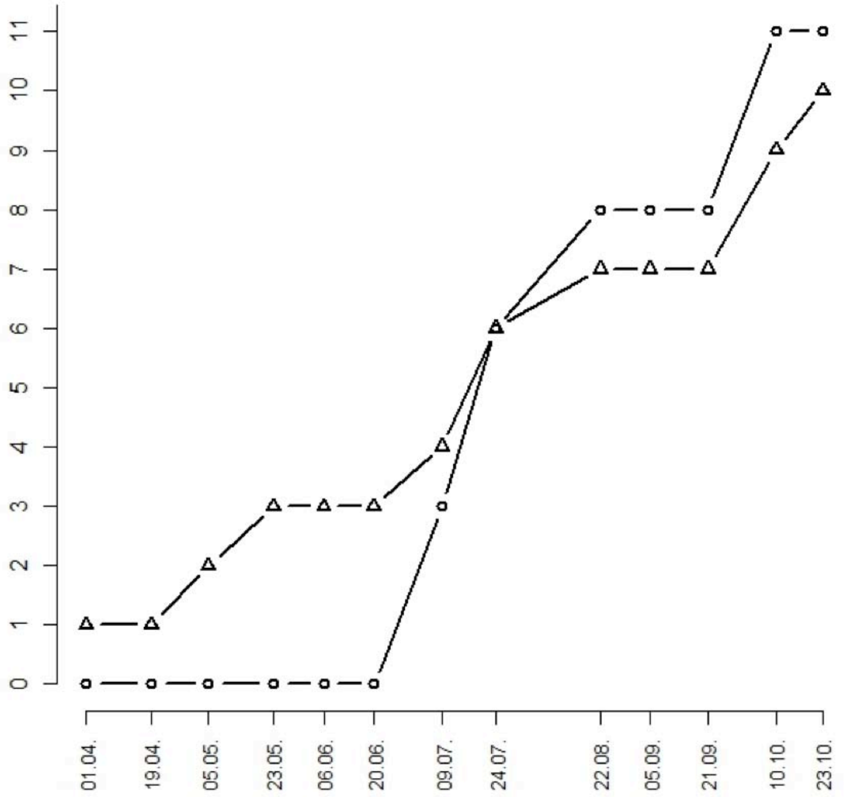


Figure 91 Vegetative and generative development of Impatiens glandulifera: the vegetative Line (Δ-symbol) and the generative Line (0) represents the phonological development measured in 12 stages (y-axis) during the observation period from the beginning of April to the end of October (x-axis).

Impatiens parviflora

The comparison of the vegetative and the generative development of the invasive species *Impatiens parviflora* shows, that vegetative growth starts in April, almost two month before the first bud is visible in the end of May. Afterwards the vegetative and generative development runs parallel. The flowering starts in June when almost all leaves are developed. It lasts until august when the vegetative plant components start to die back. The plant starts to turn yellow in September before the maturity of seeds is reached in the beginning of October.

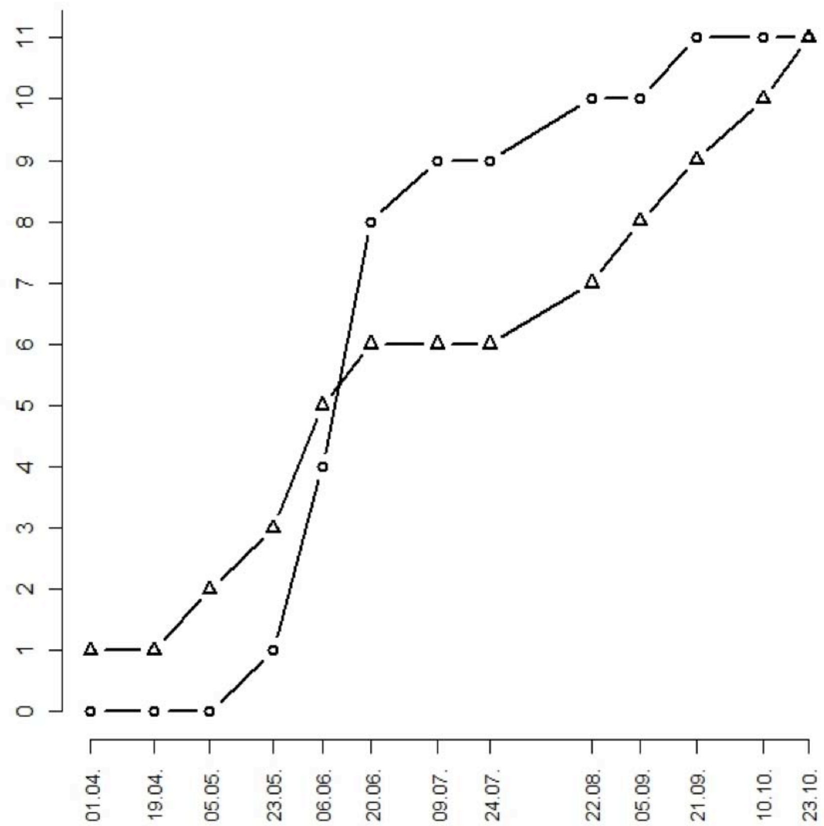


Figure 92 Vegetative and generative development of Impatiens parviflora: the vegetative Line (Δ-symbol) and the generative Line (0) represents the phonological development measured in 12 stages (y-axis) during the observation period from the beginning of April to the end of October (x-axis).

Solidago gigantea

The comparison of the vegetative and the generative development of the invasive species *Solidago gigantea* shows, that the generative development starts with the emerge of the first visible bud very late in June, when the plant is vegetative fully developed. After that the generative development goes on very quickly. Within one month, in July, the flowering starts and lasts for three months until the end of September. The distribution of seeds starts in the October, when up to 50% of the leaves are turned yellow.

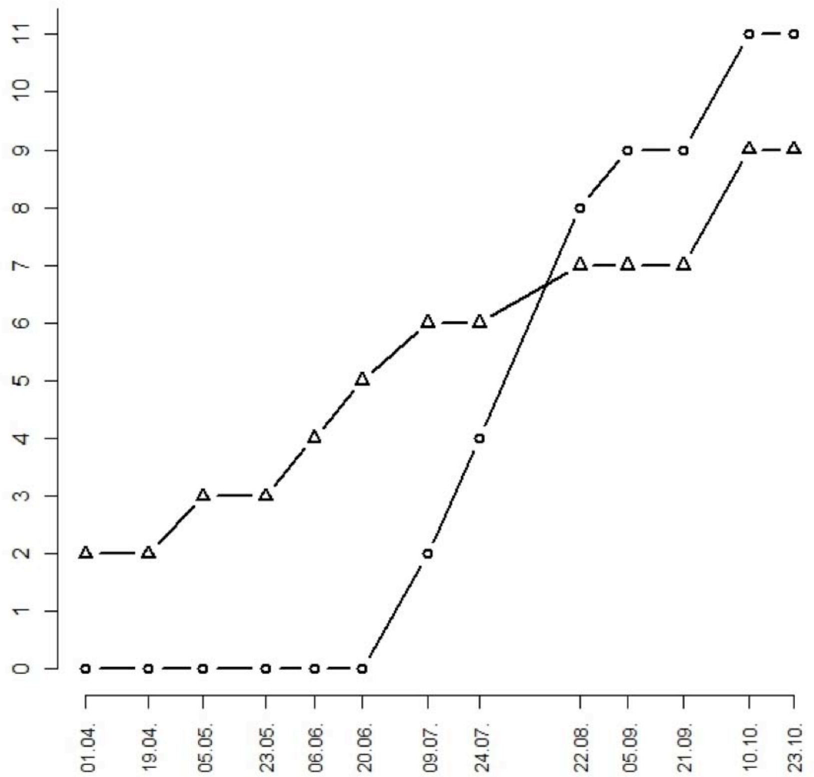


Figure 93 Vegetative and generative development of Solidago gigantea: the vegetative Line (Δ-symbol) and the generative Line (0) represents the phonological development measured in 12 stages (y-axis) during the observation period from the beginning of April to the end of October (x-axis).

3.9.2 Climate influence

The vegetative growth of *Bunias orientalis* starts at 5°C in the beginning of April. At the temperature mean of 14°C the mean of the populations of *Bunias orientalis* is vegetative fully developed and up to 50% of the flowerbuds are developed. In the end of may at the mean temperature of 21°C the blooming is finished and the 50% of the leaves are turned already yellow. The maturity of fruits is measured in June at the mean temperature of 15°C. At the mean temperature of 25°C in august the above ground plants components start to die back (Table 39).

The first shoot of *Impatiens glandulifera* emerge at the mean temperature of 5°C in April. The blooming starts in July at the mean temperature of 23°C. The plant is vegetative fully developed at mean temperature of 21°C in July. The blooming finishes at the mean temperature of 11°C in the end of September. At mean temperature of 9°C in October the vegetative components of *Impatiens glandulifera* start to die back (Figure 95).

The emerging shoots of the mean of the populations of *Impatiens parviflora* are measured at mean temperature of 5°C in April. The generative development is first recognized at the mean temperature of 11°C in the end of May. The flowering goes on in June at mean temperature of 14°C. In the middle of June up to the end of July the vegetative plant components are fully developed at mean temperature of 22°C. In the same time the blooming is finishing. At mean temperature of 21°C the fruits are mature the yellowing starts. The mean temperature of 10°C the vegetative plant components die back (Figure 96).

The vegetative growth of *Solidago gigantea* starts at 5°C in the beginning of April. The fist flower buds are visible at the mean temperature of 22°C in July, when the plant is vegetative fully developed. The flowering starts in the end of July at the mean temperature of 21°C. At the mean temperature of 11°C in September the blooming is finished and the yellowing starts. The maturity of seeds is reached in October at the mean temperature of 10°C (Figure 97).

The correlation test (Pearson‘s product-moment correlation) showed no correlation between mean temperature and growth stage ($-0{,}003 < r < 0{,}51$). The data of the present observation model is not suitable for correlation with climate data. Further analysis (germination experiments) is needed to detect the relationship of climate data.

Table 31Variations of climate data during the observation period by mean temperature (oC), precipitation (mm) and sunshine duration (h) compared to the generative (g) and vegetative (v) development stage of the species Bunias orientalis (Bu), Impatiens glandulifera (Igl), Impatiens parviflora (Ipa) and Solidago gigantea (Sog).

time	temperature (°C)	precipitation (mm)	sunshine duration (h)	Bu (v)	Igl (v)	Ipa (v)	Sog (v)	Bu (g)	Igl (g)	Ipa (g)	Sog (g)
01.04.2012	4.5 ± 2.6	0	3.6	1	1	1	2	0	0	0	0
19.04.2012	11.0 5± 3.8	3	7.9	3	1	1	2	1	0	0	0
05.05.2012	14.1 ± 4.7	0.4	5.6	6	2	2	3	5	0	0	0
23.05.2012	21.7 ± 5.0	0.6	10	6	3	3	3	8	0	1	0
06.06.2012	14.2 ± 5.0	0.2	7.1	6	3	5	4	10	0	4	0
20.06.2012	25.8 ± 4.9	27	10.2	7	3	6	5	10	0	8	0
09.07.2012	22.7 ± 2.7	5.7	8.1	8	4	6	6	11	3	9	2
24.07.2012	21.4 ± 5.6	0	12.5	9	6	6	6	11	6	9	4
22.08.2012	24.6 ± 6.0	0	9.5	10	7	7	7	11	8	10	8
05.09.2012	20.7 ± 4.2	0	11	10	7	8	7	11	8	10	9
21.09.2012	11.3 ± 5.6	0	10.8	10	7	9	7	11	8	11	9
10.10.2012	9.9 ± 2.3	0	7.7	10	9	10	9	11	11	11	11
23.10.2012	8.6 ± 1.3	0	0	10	10	11	9	11	11	11	11
average (time)	17±6.7	0.09 ± 0.6	0.31 ± 0.42								

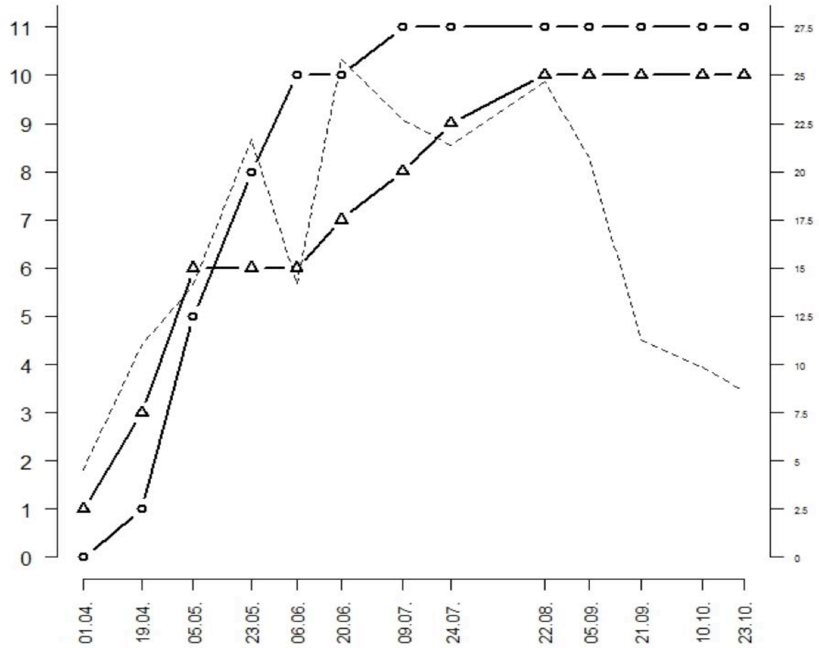


Figure 94 Comparison of the mean temperature (oC) (right y-axis) and the vegetative and generative development of Bunias orientalis: the vegetative Line (Δ-symbol) and the generative Line (0) represents the phonological development measured in 12 stages (y-axis) during the observation period from the beginning of April to the end of October (left x-axis).

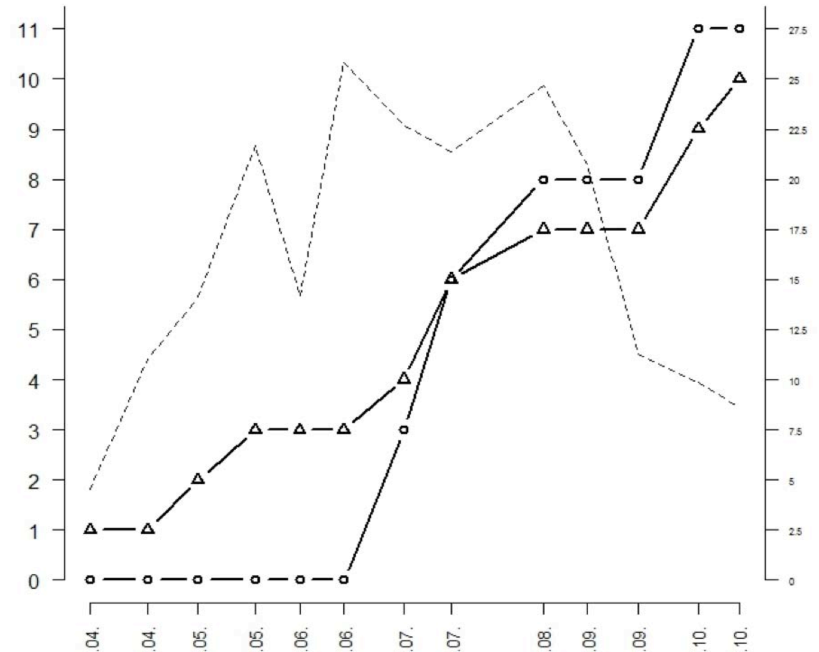


Figure 95 Comparison of the mean temperature (oC) (right y-axis) and the vegetative and generative development of Impatiens glandulifera: the vegetative Line (Δ-symbol) and the generative Line (0) represents the phonological development measured in 12 stages (y-axis) during the observation period from the beginning of April to the end of October (left x-axis).

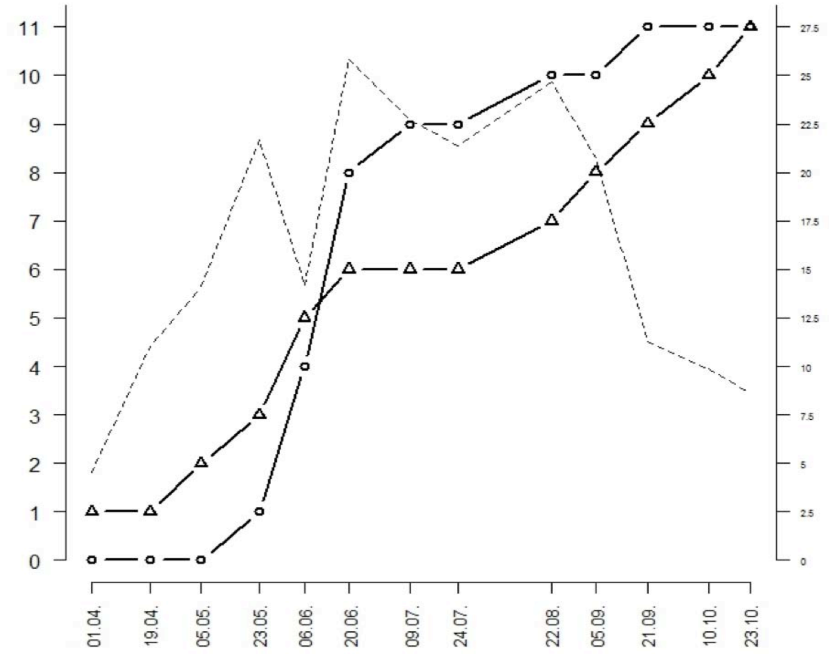


Figure 96 Comparison of the mean temperature (0C) (right y-axis) and the vegetative and generative development of Impatiens parviflora: the vegetative Line (Δ-symbol) and the generative Line (0) represents the phonological development measured in 12 stages (y-axis) during the observation period from the beginning of April to the end of October (left x-axis).

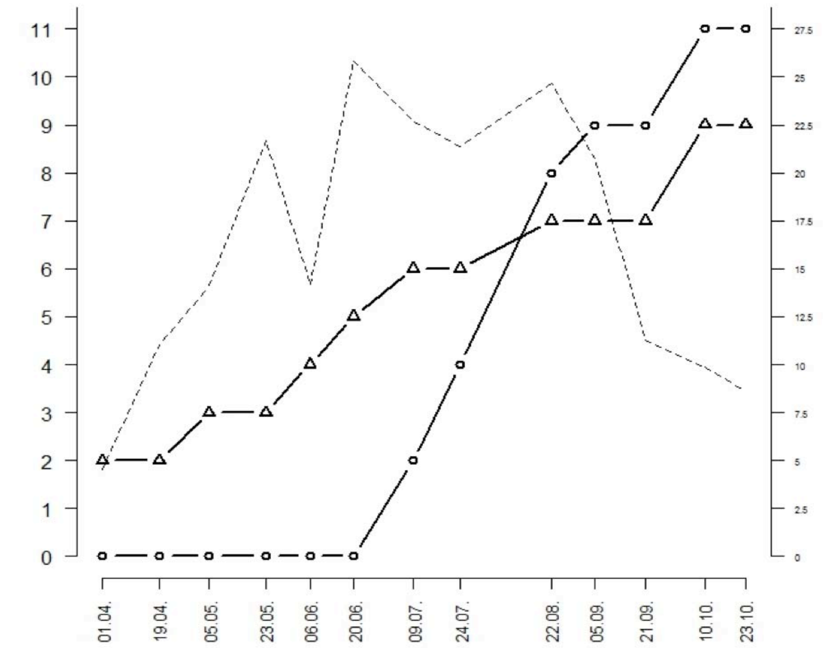


Figure 97 Comparison of the mean temperature (0C) (right y-axis) and the vegetative and generative development of Solidago gigantea: the vegetative Line (Δ-symbol) and the generative Line (0) represents the phonological development measured in 12 stages (y-axis) during the observation period from the beginning of April to the end of October (left x-axis).

3.9.3 Herbivory

None of the populations was infested by herbivores during the observation period. No damage by herbivores was detected. Herbivores had no influence on the vegetative or generative development.

Figure 98 Development of Solidago gigantea in 2013 in the study area (Battisti 2014).

4. Diskussion

4.1 Riparian vulnerability to biological invasion

The vulnerability of riparian habitats is comparable to island habitats, which are considered to be the most vulnerable ecosystems to invasion by IAS (Pyšek & Prach 1994; Vitousel et al. 1996; Wadsworth et al. 2000; Hood & Naiman 2000; Tickner et al. 2001; Walter et al. 2005;Richardson et al. 2007). In order to develop sustainable management and prevention methods, it is necessary to detect the reasons for the vulnerability of these riparian areas.

The high frequency of disturbances associated with flooding is one of the main characteristics of riparian vegetation (Franz & Bazzaz 1977; Hood & Naiman 2000). But at the same time disturbances, like floodings, are supporting the success of invasive plant invasions (Hood & Naiman 2000). Case studies on the invasion of riparian plant communities support the hypothesis, that a high species richness of the resistant native plant communities reduces the possibility of establishment of invasive alien plant species (Case 1990; Hood & Naiman 2000; Richardson et al. 2007).

In global terms, differences in classification of riparian areas need to be mentioned. The type of vegetation and plant communities varies with climate and geomorphology (Decamps et al. 1988; Naiman et al. 1993). River systems of the northern temperate zone are studied more intensively (Hood & Naiman 2000). The global comparison of the impact of invasive alien plants on riparian areas needs a critical understanding of local differences of plant diversity.

The state of riparian ecosystems in Central Europe in particular must be observed, for the ecological function of riparian ecosystems depends strongly on the respective river's flooding and elevated water tables (Naiman et al. 1993). Extensive abuse of water bodies in Central Europe over the past century has caused a serious degradation of many riparian ecosystems. In the past decades conservation and restoration of riparian landscapes has become a major issue for stakeholders on different levels (Briggs et al. 2002). Various river restoration projects have been established with the aim to revitalize the degraded riparian vegetation that has been harmed by river regulations and the loss of connectivity (Jungwirth et al. 1993; Poppe et al. 2003).

Differences in the invasibility of species rich communities are related to longitudinal variations in disturbance frequency and intensity (Hood & Naiman 2000). Further the results of the case study support the negative effect floods have on the highly invasive plant *Impatiens glandulifera*. Compared to the years 2011 and 2012, plant communities that were strongly invaded showed a significant increase of Impatiens glandulifera populations and a poor vegetative and generative development in the year 2013 - a year when a HQ100 flood plains took place (Figure 99). Therefore it is to assume that regular natural floods have a positive effect on the resistance of native plant communities against the invasion of *Impatiens glandulifera*. It is ecologically desirable that river restorations, like the LIFE+ Traisen project, support the recreation of natural riparian habitats and reduce the vulnerability of riparian habitats toward alien plant invasions.

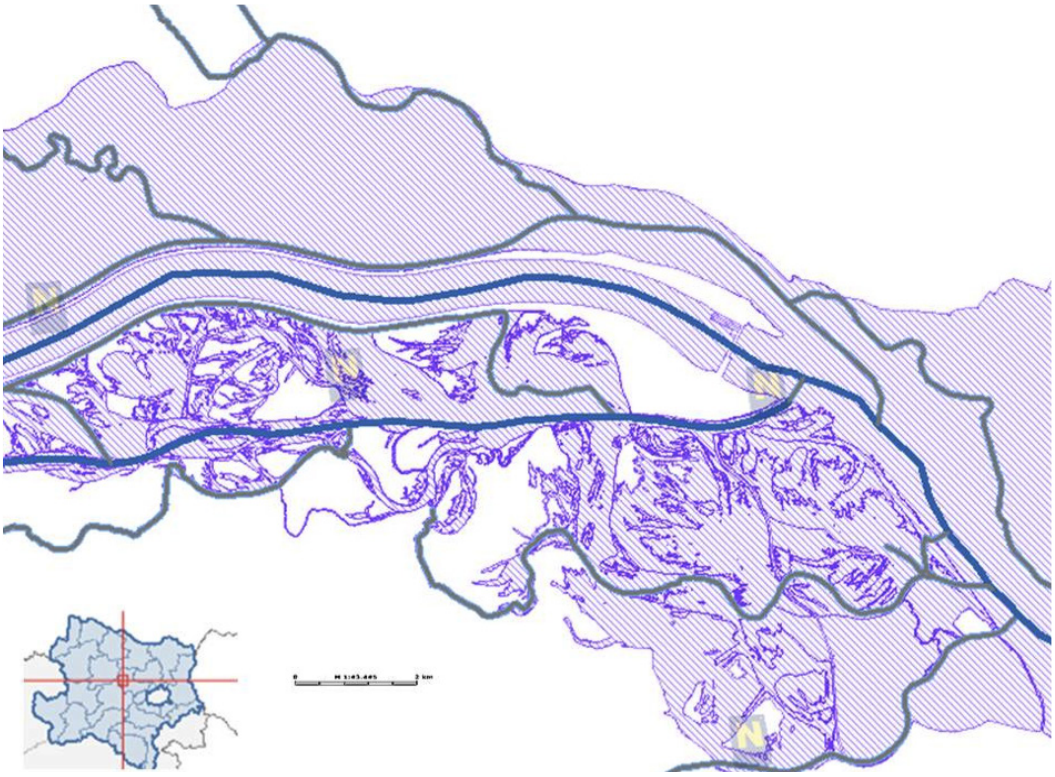


Figure 99 Area of the HQ100 flood plains (i.e. areas which statistically flood at least once every 100 years) , Land Niederösterreich, http://atlas.noe.gv.at 15.11.2013.

Shading of trees (canopy) is an important ecological function of riparian vegetation structure in relation to aquatic and semi-aquatic habitats (Naiman & Decamps 1997; Richardson et al. 2007). Results show differences in the effect of canopy on the success of invasive alien plants. Differences in canopy are characterizing riparian habitat structure.

One of the major aims of the ecological restoration Life+ Traisen is to create riparian floodplain forests in the immediate surroundings of the water bodies being restored. For the restoration of riparian floodplain forests it is recommended to use native plant material of the genus *Salix sp.* (Donat 1995; Holdenrieder et al. 1999; Landis et al. 2003; Florineth 2004). For the restoration of floodplain forests along the new course of the river Traisen, willow sticks will be used. Due to the results of the canopy tolerance of IAS and the phonological characteristics of the occurring IAS it is possible to draft recommendations. A high density of willow sticks and their early plantation supports the competitive ability of the vegetative growth of the native *Salix sp.* According to the vegetative sampling, the invasive species Bunias orientalis and Impatiens glandulifera, are endangering the restoration of species-rich riparian floodplain forests in the study area. But from the results of the canopy tolerance testing, we know that the most critical period will be over once the willow sticks will take root and achieve a plant coverage of 90%.

Figure 100 Analysis of willow population in the greater area of the riparian floodplain forests of the river Danube: Willow species occurance for the species composition of the riparian area of the river Traisen that could be used for the autocton restoration of the flppdplain forest of the new course of the river Traisen. The Willow species Salix alba, Salix rubens (fragilis x alba), Salix purpurea, Salix purpurea x viminalis, Salix viminalis and Salix x calodendron (viminalis x cinerea) were recorded in spring 2013.

4.2 Risk assessment

The occurance of invasive alien species increased significantly in the riparian area of the river Traisen. Annual and perennial species caused changes in the plant cover of the herb layer. But the total number of species did not change significantly. The results show that the diversity in the study area, measured in the Shannon index H` and Pilous´s index of evenness J, is decreasing. But the decrease of diversity could not have been related to the increase of IAS in the last three years. I assume that the time dimension is too short to discuss results for vegetation structure changes, but I could detect trends that need to be monitored over the coming years. Specifically the ongoing ecological restoration needs to be monitored and evaluated for its impact on biodiversity.

Invasion by alien plant species is constantly progressing (Hulme & Bremner 2005). Due to the rapid and ongoing distribution of invasive non-native plants, their impact on natural ecosystems is increasing (Lambdon et al. 2008).The negative impact of invasive alien plant species (IAS) on plant communities depends on habitat type and the structure of resident vegetation communities (Hejda et al. 2009). Results show that the invasion of the herb layer was most rapid in ash tree and poplar forests, embankments and grey alter tree forests. The main driver for this trend of invasion of tree habitats is the rapid distribution of *Impatiens parviflora* and the adaption of *Solidago gigantea* to the tree canopy. These aspects are recommended to be investigated in further studies. The consequence for nature conservation management is to evaluate the risk asset of each particular invasive plant species in correlation to the habitat qualities.

The major limiting resource of riparian plant species is the permanent availability of water. Fluctuation in river water levels and the groundwater table are influencing the establishment of invasive and native plant species (Richardson et al. 2007). During the study the year 2011 was characterized by medium hydrological conditions, followed by a dry year in 2012, and a year with wet hydrological conditions and HQ100 floodings in 2013. The development of IAS in the herb layer responded to this hydrological differences with changes in abundance. During the wet year in 2011, the number of *Impatiens glandulifera* increased compared to the dry year in 2012. The floods in the year 2013 caused a decreasing number of Impatiens glandulifera along the main channel of the regulated river Traisen. Compared to perennial invasive species, like *Solidago gigantea*, the changes of annual invasive species, like *Impatiens glandulifera* and *Impatiens parviflora* were more rapid. The seed bank occurrence has contributed to the establishment of

IAS (Décamps et al. 1995). Although the occurrence of seeds of IAS could not be related to the aboveground vegetation development within this study, the high number of IAS in the seed bank needs to be taken under further consideration within the restoration project.
The effect of IAS on species richness and biodiversity was discussed in several studies in the last decade (Funk et al. 2008). In global terms negative relationship between species richness and the abundance of invasive alien plant species results in biodiversity loss (Dukes 2002, Funk et al. 2008). At the local scale human disturbances are valued more significant for the loss of species diversity than the native competitive ability of species composition (Levine 2000). The results of the study support the hypothesis that plant invasion benefit strongly from human disturbances. For example, the invasive *Impatiens glandulifera* and *Solidago gigantea* establish within human disturbed areas, like forest roads or artificial embankments. The invasion into near-nature habitats is weak. For the establishment of IAS, human disturbances are playing a key role.

Results show clearly that invasion is following pathways. Plots along roads and embankments are stronger invaded than plots in semi-natural plots. The visualization of the abundance of IAS in the plant cover shows a higher abundance of IAS within the plant cover of plots with a closer proximity to the river Danube cycle path. The frequency analysis of the year 2012 counted 140,000 cyclists using the part of the Danube cycle path that crosses the riparian study area in the north and east (ECOplus2012).

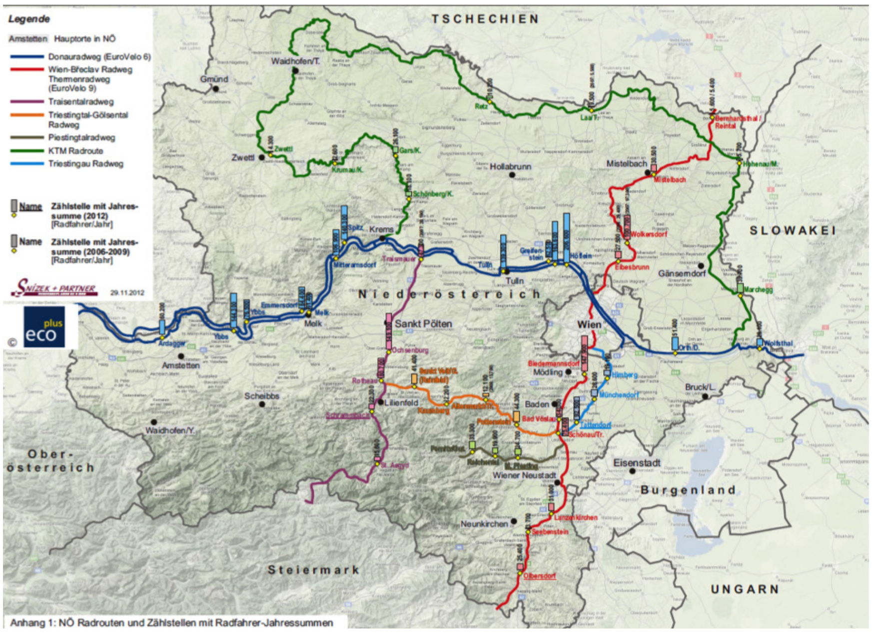


Figure 101 Frequency analysis of the main cycle pathways in Lower Austria, in 2012. (ECOplus, 2012)

The highest frequency of cyclists passes during the summer period in July and August. Over these two months 80,000 cyclists use this part of the Danube cycle route, crossing the study area. According to to the phenologcal observation completed in 2012, the months of July and August are critical periods for the distribution of invasive species' seeds, like those of the *Bunias orinetalis, Impatiens glandulifera* and *Impatiens parviflora*. The generative development is in the stage of mature seeds during this time. The rest of the seed distribution by cyclists along the Danube cycle route is high.

The importance of the regional economical income of this tourist infrastructure is increasing (Meschik 2012). The regional government of Lower Austria is planning the expansion of the Danube cycle route over the coming years in order to support the development of sustainable tourism and to strengthen the economic income for the region (ECOplus 2012).

Recent observations showed the increasing occurrence of IAS along the Danube cycle route (Reckendorfer et al. 1998; Donau-Auen 2009; ViaDonau 2013) The Danube cycle route needs to be considered as a potential pathway for IAS. The eradication of IAS populations along the Danube cycle route should be a top priority. Besides the control of present day IAS, we should actively monitor the area to determine the alien species that are not yet invasive, in order to prevent future invasions.

4.3 Evaluation of the life strategies of invasive species

Various studies discuss the connection between life strategies and habitat properties linking to the success of IAS (Binggeli 1996; Obidzinski & Symonides 2000). The evaluation of the life strategies of IAS within riparian habitats concentrates on invasive species with the highest impact on native species composition in the study area. The focus lies on the combination of riparian ecosystem characteristics and factors of invasive plant ecology with a significant importance for the biological invasion. The results are correlated with recommended and tested management and control mechanisms of previously published case studies.

Table 32: The ecology, invasiveness, habitat occurrence and ecological effects of IAS in the study area.

Species	*A. altissima*	*A.negundo*	*B. orientalis*	*I. glandulifera*	*I. parviflora*	*R. lanciniata*	*R. pseudoacacia*	*S. gigantea*
Family	Simaroubiaceae	Aceraceae	Brassicaceae	Balsaminaceae	Balsaminaceae	Asteraceae	Fabaceae	Asteraceae
Life form	phanerophyte	phanerophyte	geophyte	hemicryptophyte	hemicryptophyte	geophyte	phanerophyte	geophyte
Area of origin	China	N America	W Asia	India	Central Asia	N America	N America	N America
tolerant to canopy		+			+		+	+
EU climate distribution	Temp/Med	Temperate	Temp/Med	Temperate	Temperate	Temperate	Temperate	Temperate
AUT Conservation assessment *	Invasive	invasive	no impact	invasive	invasive	invasive	invasive	invasive
invasivness (riparian area) **	no impact	potentially inva-sive	invasive	temporary invasi-ve	invasive	invasive	potentially invasive	invasive
Introduction reasons	ornamental	ornamental	indirect	indirect	indirect	ornametal	ornamental, forestry	ornamental
Ecological effects								
changes in succession pattern		+				+	+	+
chages in species compositon	+	+	+	+	+	+	+	+
changes of nutrient cycles							+	
hybridisation*								
pathway - forest road			+		+		+	+
pathway - river stream		+	+	+		+		

Species	*A. altissima*	*A.negundo*	*B. orientalis*	*I. glandulifera*	*I. parviflora*	*R. lanciniata*	*R. pseudoacacia*	*S. gigantea*
Habitat ocurance								
ash tree forest		+			+		+	
tall oatgrass meadows	+					+		+
grey alder tree forest		+						
semi-dry grasslands	+	(+)						+
grassland fallow	+							+
tall forb community			(+)	+		+		+
cutover land		+			(+)		+	+
poplar tree forest		+			+			(+)
red pine tree forst								
willow tree alluvial forest		+		(+)	+			
embankment		(+)	+	+				
forest edge	+	+		+	+		+	+

* = (Water et al. 2005, Essl & Rabitsch 2002), invasiveness riparian area ** = based on reasech data evaluation; habitat occurance: + = main occurance, (+) accessory occurrence; other catagories: + = appropriate;

Impatiens parviflora

One characteristic of riparian plant communities is the structural diversification along vertical and horizontal gradients (Nilsson & Svedmark 2002). In most riparian plant communities the plant cover, abundance and species richness of the tree and shrub layer are more developed than the herb layer of riparian forest habitats (Nilsson 1992). Riparian plant communities in the herb layer are often very lacunose and with a high amount of patches which are free of vegetation. This is an enormous advantage for the distribution of *Impatiens parviflora*. The highly invasive plant species *I. parviflora* is very harmful to woodlands (Obidzinski & Symonides 2000; Chmura & Sierka 2007).

The short life cycle, high seed prediction and rapid growth of seedlings support the invasive success of *I. parviflora.* The combination of life strategies of I. parviflora with the naturally low plant cover of the herb layer in floodplain forests is the main reason for the rapid invasion of *I. parviflora* in riparian forest habitats (Obidzinski & Symonides 2000). The data of the present study support this hypothesis. The distribution of *I. parviflora* increased extensively from 2010 to 2013 within all forest habitat types. *I. parviflora* is competing successfully with native species of the herb layer. The decrease of species richness within the last three study years within the study area is related to the invasion of *I. parviflora.*

The introduction of *I. parviflora* is caused by indirect human activity. Fragmentation by roads, trampling and logging are supporting factors for the distribution of IAS (Pyšek & Pyšek 1991; Obidzinski & Symonides 2000; Kowarik 2008). The results of this case study confirm the connection of human disturbances as a factor for distribution of *I. parviflora*. In 2010 the population of I. parviflora was found mainly along forest roads. In 2013 *I. parviflora* invaded over 50% of all forest habitats.

The results of this and other European case studies show that species-rich forest habitats are less invaded by *I. parviflora* than forests with higher human impact (Case 1990; Chmura & Sierka 2007). Forest habitats with a higher native species density in the herb layer, or with a developed shrub layer, are less invaded by *I. parviflora* (Pyšek & Pyšek 1991; Chmura & Sierka 2007). The results of this case study confirm that natural and semi-natural forest habitats with a higher abundance of native species in the herb layer were invaded slower. The rapid development of monotypic stands of *I. parviflora* took place mainly in poplar wood forests.

Once this very aggressive colonizer *I. parviflora* is established along these "pathway-habitats" the protection of natural habitats from the invasion is unsuccessful (Obidzinski & Symonides 2000). Forest edges, canopy gabs and disturbances of soil are detectable factors that support the invasion of *I. parviflora*. Special monitoring efforts within the forest management on these pathways are necessary to diminish the spread (Chmura & Sierka 2007). Active eradication of individuals is necessary to protect natural forest habitats. Conventional conservation strategies for natural forests are not sufficient (Chmura & Sierka 2007), though recent data show a higher resistance of natural forest habitats. A successful control is only possible in the early stages of invasion.

Impatiens glandulifera

Riparian ecosystems are developed by natural disturbances. Natural floodings create a mosaic of plant communities at different stages of succession (Hood & Naiman 2000). Gaps in vegetation cover with a low density of plant species, or even no plant cover, are a common result of natural dynamics in ecosystems. Natural disturbances are important for the sustainable development of soft wood floodplain forests (Baptist et al. 2004). Unfortunately, the invasive species *Impatiens glandulifera* is taking advantage of these native disturbances in riparian ecosystems (Perrins et al. 1990; Pyšek & Prach 1995; Ammer et al. 2011). The rapid growth of *I. glandulifera* is endangering the reestablishment process of soft wood species, especially of the genus *Salix*, following a natural disturbance (Ammer et al. 2011).

Mechanical weed control techniques are commonly used for the management of *I. glandulifera* (Ammer et al. 2011, Hulme & Bremner 2006). The use of herbicides is inconsistent with conservation targets in protected riparian areas, such as for example the present study area, a Natura 2000 site. Various studies have tested the effect of mowing and hand weeding of monotypic stands. Hand weeding has the advantage that tree seedlings are not affected too much by the removal method. But the method is time and cost intensive and only suitable for individuals or little populations, where native plant species still thrive (Ammer, Schall et al. 2011). The results of the phonological observation show that the generative development starts in late June, when the vegetative development of the plant has almost reached the adult stage. This development stage, in the end of June, is recommended for timing control actions.

It is prudent to mention that a complete eradication of I. glandulifera is unrealistic (Wadsworth et al. 2000; Hejda et al. 2009). The management asks for a risk assessment compared to the native species on a local scale (Hejda et al. 2009). The results of the development of the *I. glandulifera* populations in 2013 compared to the on-going study years support the hypothesis that *I. glandulifera* is temporarily invasive in human disturbed areas and rarely invasive to undisturbed natural habitats (Ammer et al. 2011). Therefore the new role of ecological river restoration measures as an instrument for the control of plant invasion needs to be further discussed. Within this context the high seed productivity and tendency for long seed dormancy periods needs to be mentioned. The replacement and dislocation of soil that is contaminated with *Impatiens glandulifera* seeds is one of the most successful ways to facilitate the distribution of this invasive species (Drescher & Prots 1996, Hulme & Bremner 2006). The ecological restoration of the river Traisen contains a high number of dislocated cubic meters of soil. In the year 2010 and 2011 the soil seed bank was proofed within this study in order to detect the occurrence of Impatiens glandulifera seeds within the new river courses. The results show that seeds on I. glandulifera occur up to the depth of 20cm, but the most contaminated soil layer is the upper soil layer from 0 to 5cm. The density of IAS seeds was evaluated to be dangerous for the prospect of the successful reestablishment of the native vegetation. According to the result of this study the contaminated soil was not reused within the lowering areas of the river's new course. The analysis of the seed bank needs to be considered within the successful monitoring of IAS within restoration projects against *I. glandulifera.*

Bunias orientalis

Decades of river regulation and the intensive use of European water bodies led to an unnatural riverinne environment. The loss of connectivity of the river stream to the riparian landscape diminishes the habitat structure and biodiversity. Ruderal vegetation incraeses (Jungwirth et al. 1993; Chovanec et al. 2000; Muhar et al. 2007). The river Traisen is a good example for the negative impact of human activity on river systems. The embankments are unnaturally disconnected to the riparian complex of habitats. The requirements are appropriate for the invasion of IAS, which avoid habitats with canopy and prefer disturbed habitat structures.

Bunias orientalis is invasive to ruderal habitats with high anthropogenic impact (Dietz et al. 1999). In the study area, populations of *B. orientalis* develop along the artificial embankment of the river Traisen. Specifically, sections of the sunny embankment on the left river side, which are limited by a forest road to the north, are affected by the invasion of *B. orientalis.* The path ways for the introduction are clearly to detect along roads and embankment without canopy. This is an advantage for the monitoring. Areas with appropriate requirements need to be the focus of monitoring efforts in order to detect the invasion in the earliest stages. *B. orientalis* produces a high amount of seeds and start to germinate in the early spring. The phenological observation shows that the flowering starts early in April, and needs to be considered in management's actions.

Ongoing studies show that mowing before flowering is an successful control method, when it is repeated for a minimum of five years. The best control method is prevention. The eradication of individuals before the development of mature seeds is the most successful method to avoid invasion of *B. orientalis*. But *B. orientalis* has no chance to develop large monotypic stand in riparian areas, when the anthropogenic influence on the riparian landscape is diminished (Dietz et al. 1999). Besides many positive interactions, ecological restorations have a positive influence on the control of *B. orientalis* distribution.

Solidago gigantea

The mosaic of European riparian habitats includes often dry grasslands and permanent grassland, which are important for the species richness of a riparian area (Kurmann & Bernhardt 2013). The mowing method was used over centuries for local agricultural purposes. Today the abandonment of grasslands is regarded as a serious reason for the loss of biodiversity. Abandonment leads to shrub encroachment and the rapid intrusion of woody plants, following natural secondary succession. Changes in mowing timing often create gaps for the introduction of IAS. So the diversity of riparian grassland habitats is not just endangered by the abandonment but also by biological invasions of tall perennial rhizomatous invasive herbs like *Solidago gigantea.*

In the study area the highly invasive species *S. gigantea* successfully invades tall oat grass meadows, tall forb communities and forest edges of poplar tree forests. Two factors are responsible for the successful invasion of *S. gigantea.* One, tall oat grass meadows are underlying a management system that is supporting the distribution of seeds and that is not harmful to the clonal vegetative spreading. Massively invaded tall oat grass meadows are mowed in the late summer period; there the seed production has already started. The best timing for mowing is the in early June, when the vegetative growth is well-enough advanced, but the flowering has not yet started. Phenological measurements are recommended for the evaluation of the appropriate control timing in order to concern local phonological varieties. The mass of wind-dispersed seeds support the distribution of *S. gigantea* within a large number of different riparian habitat types. Smaller populations of *S. gigantea* are to be found all over the study area.

Forest roads are important pathways for *S. gigantea* in the study area. Once a population is established along disturbed areas like a forest road, the local spread into forest habitats is hard to control. The creation of dense clonal stands is a successful life strategy of *S. gigantea* for distribution (Sheppard 2005).

S. gigantea is a common garden plant (Kowarik & Gressel 2005). Any control management needs to include public perception and risk communication. (Sheppard et al. 2006). Individuals need to eradicate with all plant parts. The germination tests show a high germination rate under various conditions. The development of seeds needs to be controlled. Mowing these populations is useful when their generative development is considered. The control of *S. gigantea* is time intensive. All management actions of a successful monitoring initiative are recommended to be assessed over a period of at least 5 to 10 years.

Table 33 Management of the invasive species Bunias orinetalis, Impatiens glandulifera, Impatiens parviflora and Solidago gigantea based on phenologiclan observation under local envirometal conditions.

species	individuals	established populations	ongoing flowering	maturity of fruit	vegetative fully developed	distributing seeds	removal of the cuttings
Bunias orientalis	before flowering: pull out all plant components (including the root), planting native Pioneers; mowing twice a year, before flowering;		April	June	May	Juny	dispose all plant components
Impatiens glandulifera		mowing or clearing short of the ground	July	September	June	September	
Impatiens parviflora		annual mowing short of the ground	June	August	July	September	
Solidago gigantea		annual mowing before the flowering	July	October	July	October	compostable without seeds of flower buds

Rudbekia lancinata

Almost 90% of European invasive species are planted first for ornamental reasons and escaped later to natural habitats. The nutrient-rich environment of riparian areas is highly affected by biological invasions.

The invasive *Rudbekia lancinata* is establishing very successful in tall forb communities. The massive stands of *R. lancinata* dominated the native plant community in the herb and shrub layer. The vegetation sampling of 2011 to 2013 shows a negative impact of *R. lancinata* on the area. The spreading limited to an area of 300m^2 in the study area.

The negative impact of *R. lancinata* is almost not discussed and not mentioned in public information on IAS. The eradication of the perennial plant is cost and time intensive. Individuals have to be removed in all plant parts before seed development. The monotypic stands can be diminished over a period of 5 to 10 years by mowing in the early summer.

Ailanthus altissima
The mosaic of riparian habitats includes areas of different aridity. More arid areas, like forest edges or semi dry-grasslands, are common contact points for the introduction of *A. altissima*. Specifically, within the shrub encroachment of grasslands, *A. altissima* is a primary species. According to this study, the populations of *A. altissima* are not an invasive risk at the moment. Its occurance is limited to individuals in forest edges.

This invasive tree species has many advantages due to its life strategy. An adult tree of 30m height produces on average 300,000 seeds a year (Sheppard et al. 2006). Sprints from root cuttings support its vegetative growth.

Various efficient control methods against *A. altissima* exist. Individuals have to be eradicated with the total root stock. The ringbarking also shows great success when the tree has reached its adult size. Studies have reported several biological control methods (Sheppard et al. 2006). In the case of the study area we recommend to eradicate the existing individuals and to monitor the grasslands and preferred habitats, at least once a year. The control of *A. altisimma* at its present state is not yet cost or time intensive. Once the populations increase the control costs are expected to increase exponentially. Comparable riparian areas, like the National park "Donau-Auen" are an example for the negative impact of missing management on *A. atissima* in the early stages.

Robinia pseudoacacia

The invasion of *Robinia pseudoacacia* holds one of the most successful invasion histories in Europe´s alien species issue. A *R. pseudoacacia* was introduced as an ornamental plant and it is still a commonly traded garden plant. One of the first control management initiatives needs to be the prohibiton of any economic use of *R. pseudoacacia* (Kastler 2013). The highly invasive tree species spreads rapidly due to its successful competitive abilities with native flora. Due to rapid vegetative and sexual reproduction as well as nitrogen-fixing roots, monotypic stands of *R. pseudoacacia* developed fast and caused changes in native species compositi-on, displacing native species (Sheppard et al. 2006).

The spread needs to be controlled, especially in protected areas. The control is cost and time intensive, because of the species' durable root stock. A successful method is to ring bark the individuals. The monitoring needs to be assessed for at least 5 years (Essl et al. 2002). The spread of *R. pseudoacacia* is limited to planted individuals, which should be removed as soon as possible. More dangerous is the situation of cutover land in the east of the study area. Deforestation caused a massive invasion of *R. pseudoacacia* within the last years. The eradication of these monotypic stands seems rather unrealistic. But the spread to close by habitats needs to be prevented.

Acer negundo

The north American invasive species *Acer negundo* is well adapted to riparian environments (Kowarik & Boye 2003). *A. negundo* is tolerant to canopy and can invade natural forest habitats. In the study area the spread is limited to occurring seedlings across all different moist habitats. Older individuals are to be found on forest edges and along forest roads. These individuals need to be eradicated, including the root stock, in order to prevent the distribution of seeds.
The spread of *A. negundo* is correlated to human activity. The increase of *A. negundo* populations is caused by disturbances of the riparian environment (Kowarik & Boye 2003). Ecological restoration with the aim to support the development of natural soft wood habitats, focusing on the native genus *Salix*, is an important way to control *A. negundo.*

4.4 Competivity in germination and phenological development

Native species differ in their reaction on biological invasion (Hejda et al. 2009). Contemporaneously, the river's function as a corridor for invasive alien plant species shows in a massive invasion of nonnative plants to riparian habitats. The human disturbance of river ecosystems within the last century supports the invasion of alien plant species to riparian habitats and causes a change in the riparian vegetation composition (Hood & Naiman 2000; Weber & Jakobs 2005). The results of many studies show that native species in riparian habitats are not necessarily "weaker" than invasive alien plant species. The effects of environmental conditions and an intact riparian ecosystem for biological invasion must still be studied further. Seed characteristics alone do not explain the invasiveness of nonnative plants among native plant species (Perglová et al. 2009). It may therefore be assumed that the competitive character of plant species also depends on environmental conditions, intraspecific interactions and local habitat quality (Daehler 2003; Morrison & Mauck 2007; Bottollie-Curtet et al. 2013). Especially the competitive ability of invasive and native dominant plant species is correlated with species composition and interaction (Hillebrand et al. 2008; Bottollier-Curtet et al. 2013).

The most abundant native plant species within this study were the species *S. sarracenicus, T. vulgare* and *U. dioica*. Among other things, this confirms the successful ecological strategy and strong competitiveness of *U.dioica* (Grime et al. 1988; Taylor 2009). The best invasive competitor was *S. gigantea*, which shows a high germination rate within a short period of 3 weeks (Weber 2001; Jakobs et al. 2004).
Understanding the characteristics of invasive species' biology in comparison to native species or closely related species is important for progressing the research of biological invasion (Roy 1990; Richardson et al. 2007, Rejmánek 1996; Perglová et al. 2009). The focus of this study lay on the comparison of the germination and seedling characteristics of invasive and native species within the same habitat.

The cold-wet stratification at 15/5°C for 2 months had a pronounced effect on the number of germinated seeds of the invasive species *S. gigantea.* Native species *E. cannabinum* and *S. sarracenicus* were also noticeably affected by cold-wet stratification. In general, the percentage of germinated seeds decreased without cold-wet stratification, but the effect of different lengths of dry and cold-wet storage should be tested in more detail to discuss the results with comparable hypothesis (Perglová et al. 2009).

Results of the study also show that temperature has a clearly detectable effect on germination rates and seedling development. More seeds of the species *S. gigantea, T. vulgare* and *S. glutinosa* germinated at 25/10°C than at 15/5°C, while the number of germinated seeds of *S. sarracenicus* and *U. dioica* was higher at 15/5°C. Temperature also caused differences in seedling development for all tested species that showed germination activity. The comparison of the invasive species *S. gigantea* and the native species *S. sarracenicus* and *U. dioica* makes clear that the development of seedlings, both native and invasive, takes less time under warmer conditions. In the case of *S. gigantea, S. sarracenicus* and *U. dioica*, the competitiveness of invasive species is not higher in terms of seedling emergence and development. Competition in the germination stage apparently does not play a major role in the success of the invasive species *S. gigantea* compared to the native species *S. sarracenicus* and *U. dioica.* Rather, competition among young individuals may have a much stronger effect (Bottollier-Curtet et al. 2013). Studies show that the competitive advantage of invasive species is not dominant before several growing seasons (Andrews et al. 2009, Hejda et al. 2009). Specifically, in the long term, however, seed bank deposition and seed dormancy are indicative of successful invasion of disturbed riparian areas (Bottollier-Curtet et al. 2013).

The TTC-test showed a high number of ungerminated but dormant seeds of the species *B. orientalis* and *I. parviflora*. Many seeds have a long viability period and remain dormant until the environmental conditions are suitable (Baskin & Baskin 1998). In the case of the species *B. orientalis*, the invasion strategy is based on early emergence within a short gap before seedlings can be dominated by competitors. The long viability of seeds of *B. orientalis* allows them to wait for good germination conditions (Dietz et al. 1999). The high number of dormant seeds underlines the importance of understanding the seed bank characteristics of a particular seed bank composition. This can be useful for various restoration projects (Hopfensperger 2007). Nevertheless, the number of studies about riparian seed banks is unfortunately small (Goodson et al. 2001).

Uncontested riparian zones are highly vulnerable to biological invasion (Pyšek et al. 1998; Hood & Naiman 2000). The comparison of native riparian plant species and invasive plant species under controlled conditions is just one of many more possible ways to analyze requirements for understanding and managing the ongoing invasion of riparian habits.

4.5 Control, Eradication and Management

Results of a review of global and local strategies on biological invasions can be brought together in a system of seven steps. Within the following section the seven steps will be discussed.

4.5.1 Step one: Identification and Prevention

Preventing the introduction of alien plant species (IAS) is the cheapest and most effective option for the successful management of IAS (SpeciesSurvivalCommission 2000). Preventing invasions is also the most challenging step, where ecological functions and land use changes need to be considered in the long term (DÁntonio 2002). Successful prevention includes actions against the introduction of IAS with a scientifically documented negative impact, as well as the observation of IAS with scientific uncertain long-term impact on native species. It is useful to differ between the intentional and unintentional introduction of IAS. Concerning unintentionally introduced IAS, it is necessary to recognize that the impact of IAS is often strongly related to environmental conditions. The impact of established, non-native species varies between climate regions and ecosystems. Therefore the long-term impact is often unpredictable. The impact of some IAS, like Impateins gladulifera, is well documented on a global level. The intentional introduction of invasive alien species with well-detected negative effects can be legislatively permitted to raise preventive efficiency (EEA) (SpeciesSurvivalCommission 2000). But the impact of many IAS is unpredictable and difficult to generalize. The early identification on a local level depends on supporting the establishing of IAS and can save costs associated with IAS management (SpeciesSurvivalCommission 2000). An updated database of invasive alien species is a recommendable tool to identify them before they cause unnoticed damage (Lowe, Browne et al. 2000). To support the knowledge of IAS specifically, conservation managers on policy makers need to be informed about the supervision of risk management. International Programmes are helpful to identify risks of IAS early enough (Shine 2007).

4.5.2 Step two: Monitoring and early control

The eradication of individuals is highly necessary in early plant population development stages (SpeciesSurvivalCommission 2000). The eradication of a few individual invasive plans in concidered to be the best response the introduction of IAS (Genovesi 2005). Specifically, vulnerable ecosystems, like riparian areas, need to underlie consequent monitoring (SpeciesSurvivalCommission 2000). Since rivers are important pathways for the introduction of IAS (Johansson, Nilsson et al. 1996; Hood and Naiman 2000; Richardson, Holmes et al. 2007), frequent monitoring of riparian vegetation is useful and economically very efficient for early detection of biological invasions.
On an international level it is useful to identify the important pathways of IAS concerning the variation between countries according to trade and tourism routes. This supports the early detection of the unintentional introduction of IAS (SpeciesSurvivalCommission 2000). Furthermore, steakholders with any role contributing to the movement within these pathways need to be sufficient warned and trained (Hiebert 1997; SpeciesSurvivalCommission 2000; Pyšek and Richardson 2010)

4.5.3 Step three: defining the Outcomes of Control

Before starting the eradication of established IAS it is recommended to set the main goal. The relevant measures need to be set in order to quantifiy the results and to compare the outcomes with other methods in different regions (SpeciesSurvivalCommission 2000). Specifically, regarding the long-term effect of IAS, or the aspect of seed dormancy, it is necessary to set measures of success towards an active long-term monitoring.

4.5.4 Step four: Choosing control Method

Often various methods are used to control the same alien species. For many methods of eradication of alien species a lack of scientific results, due to the rapid spread, is documented. Many methods are published in gray literature (Genovesi 2005). The efficiency of control methods is in many cases highly correlated with the ecosystem structure.

In any case the chosen method has to be socially, culturally and ethically acceptable (SpeciesSurvivalCommission 2000). The most challenging criteria for choosing the best control method is its cost efficiency. The cost and benefits need to be evaluated with participating stakeholders in regard to the defined outcomes. Furthermore, the costs for the optional adaption of the monitoring should be clarified in advance. The control method often needs to be adapted to achieve the set outcomes.

4.5.5 Step five: eradication action

The eradication of established IAS is often associated with large costs. Many case studies show great success with the eradication of IAS when the methods are correlated to a precise preparation based on scientific knowledge (Genovesi 2005). Most management recommendations on the eradication of IAS are related to island ecosystems (Genovesi 2005). The development of successful eradication methods is often not easy to detect, because literature on the topic of IAS management is just available in grey literature (Vázquez & Simberloff 2003)(Genovesi 2005).
Within this step it is useful to integrate pubic stakeholders and communities into the eradication activities. In many cases the public awareness on IAS was increasing after taking part in an eradication activity.

4.5.6 Step six: Re-Introduction of native species

Successful eradication management is achieved with the sustainable recovery of the native plant diversity (Genovesi 2005). The re-introduction of native species helps the native diversity in the long-term. A sustainable native population should be the aim of each eradication action. With ecological restoration mechanisms it is required to plan and approve the re-introduction of native species. The main focus is on the reestablishment of the ecosystem`s condition´s and requirements (ISSG 2014).
Public participation has a positive effect on the re-introduction of native species. Socio-economic plans are recommended for the successful re-introduction regarding economic values of multiple uses for natural infrastructure. Participation with the community as well as with local politics is highly valued (Genovesi 2005).

4.5.7 Step seven: monitoring and evaluation

The control method needs to be evaluated constantly in order to achieve the set goal. Constant monitoring over a longer period is necessary in order to be able to avoid negative ecological or economical outcomes. The duration of these monitoring activities need to be adapted to the recreation of the target ecosystem.
The eradication of IAS is a topic of global importance. Therefore is essential to share experiences, knowledge on control methods, and efforts on public participation programs. An active network on a local, national and global level is necessary to reach positive outcomes on IAS research(SpeciesSurvivalCommission 2000).

4.6 Critical issues for Ecological restorations

In the last 20 years, ecological restorations became an important part of conservation efforts (Davis & Slobodkin 2004, Bradshaw 1993). Specifically, on a regional and local level restorations are important for nature conservation goals (Pfadenhauer 1999). The aim of ecological restoration is always to repair the ecosystem's functions (Richardson & Pyšek 2006). Strategies will differ according to the preferences of the stakeholders, and ecological restorations vary in their motivation, which can cause differences in the output of an ecological restoration project (Davis & Slobodkin 2004). To avoid any negative developments in ecological restoration projects, interdisciplinary cooperation is important (Suding et al. 2004).

> *„Ecological Restoration is the process of restoring one or more valued processes or attributes of a landscape (Davis and Slobodkin 2004)".*

At first, European river restoration projects orientated strongly hydrological and hydrobiological improvements. The EU Water Framework Directive (WFD), for example, focused mainly on hydrological measures, the connectivity of the riparian landscape with the river stream or the recreation of the native vegetation structure was only later taken into consideration (Rood et al. 2003; Muhar et al. 2007).

The motivations for ecological restorations vary (Pfadenhauer & Grootjans 1999) (Buijse et al. 2002). Often it is the aim to return to a more natural situation like it was prior to the invasion. The aim to recreate historic conditions is rather complicated to apply in the European cultural landscape, where human influence was unavoidable for centuries (Pfadenhauer 1999, Hohensinner et al. 2005; Richardson et al. 2007). Alien species were not a part of historic vegetation compositions and therefore need to be removed.

An alternative approach is to accept that humans are a part of the ecosystem. Following this approach, it is the aim of ecological restorations to improve ecosystem functions and structures, rather than to rebuild historic landscape (Richardson et al. 2007). Removal actions of alien species are not the main focus within this functional approach. The control of alien plant species was even criticized (Del Tredici 2004).

The manipulation of abiotic factors in an ecosystem's, life, for example flood frequency and intensity affect directly the recreation of native riparian vegetation (Rood et al. 2003). But the disturbances may allow for the establishment of alien species (Levine & Stromberg 2001; Richardson et al. 2007). The removal of IAS is considered as an essential component of ecological restoration projects.

The control of IAS needs to be considered in local circumstances and the intensity of IAS establishment (Funk et al. 2008, Richardson et al. 2007). The functional characteristics of the local species composition, like the phenology or seed bank occurance of native and local invasive species, are esseninal for restorative purposes (Funk et al. 2008). There is no "golden rule" that describes the control of IAS. It needs to be concidered that IAS can influence restoration projects in several ways. First the occurance of IAS is within a reason for the restoration. Second, IAS is colonizing after the restoration of an ecosystem or, third, IAS are already established in the area. The evaluation of the biological invasion is relevant for all ongoing mangement strategies (DÁntonio 2002).

Management actions need to orientate on the density and the species characteristics of each invasive species (DÁntonio 2002). The comparably low number of rather old individuals of *Robinia pseudoacacia* in the study area can be removed wthin the restauration constructions by root stock removal. In case of a larger and more developed population of *R. pseudoacacia,* a simple root stock removal would not be appropriate or could even be counterproductive, because the removal in long terms by ringbarking, for example, achieves greater success within monotypic stands.

The most important fact that needs to be mentioned is that the success of management and control actions against IAS are not successful in the short-term. Active monitoring over a long period of time is recommended (Richardson et al. 2007; Harris & van Diggelen 2009). The success of restoration measures need to be assured by a monitoring process lasting a period of 10 to 20 years (Pfadenhauer 1999).

Stabilization of the native plant species' composition is not just an aim of ecological restorations but also an important step towards IAS control (Tickner et al. 2001). To return to a natural course and connectivity to the riparian landscapes supports the competivity of native species to invasive species (Bottollier-Curtet et al. 2013). The success of the reestablishment of native species composition for restoration of human disturbed habitats can be supported if the site is surrounded by natural vegetation (Pfadenhauer 1999). Furthermore the renaturalisation of riparian landscapes and the eradication of unnatural elements decrease the number of human disturbed areas which are preferred areas for IAS establishments. For example the development of the invasive plant *Bunias orientalis* in the study area shows a concentration on unnatural embankments of the regulated river Traisen. Semi-natural habitats are not invaded by *B. orientalis*. This invasion could be avoided if the river is restored to a more natural course.

The natural reestablishment process of native plant species after restoration is seriously endangered when invasive alien species were established before the restorative measures. In most cases, the seed bank is decontaminated with IAS seeds (Prach et al. 2001). One life strategy of IAS is the high seed production and long survival of seeds in the seed bank (Gurnell et al. 2006). Results show the high density of IAS seeds in sites, where IAS have not occurred in the vegetation cover. The seed bank of the study area was evaluated as decontaminated. The use for the restoration of native vegetation can cause a high risk for the support of biological plant invasion.

Case studies of ecological river resoration projects are important for planning future restoration managment projects. Along the river Thur in Swizerland, the monitoring of IAS development after the river restoration showed an ongoing increase of alien plant species. It bears mentioning that no IAS established on gravel islands. The germination is interrupted by annual floodings (Krinke et al. 2005).

The research on IAS in the riparian area of the river Traisen show the negative impact of invasive alien plant species (IAS) on native vegetation composition, succession processes and species richness. To ignore the occurrence of IAS in a project area where an ecological restoration takes place is not to support. The aim of an ecological restoration is to improve ecosystem functions, and the creation of natural or semi natural habitats will fail, when IAS are not considered in the management and restoration plans.

The number of published examples of resoration projects that include a management process for invasive alien plant species is low, but increasing (DÁntonio 2002). In future it is nessearry to link reaseach on IAS with the applied objectives of ecological resoration management (Suding & Hobbs 2009).

4.7 Technical Guidelines for ecological restoration for the control and management of IAS in riparian areas

The timing of control actions is essential for a successful monitoring process. The success of control activities are correlated to the phenological development of the invasive species in the area. Technical guidelines were developed for implementing the main aspects of risk in the biological invasion into the processes of ecological restoration of running waters (Figure 104). The structure of the guidelines shows clearly the necessity to concider the danger of IAS within all stages of the restoration process. The successful management of IAS needs diverse management activities before, during and after the constructional measures.
Before the constructional measures, the situation of the project area as well as the greater area and any possible pathways need to be analysed ccording to the occurrence of IAS. A study of the vegetation will give information on the invasive species in the project area. Ecological restoration implementations show the change of topographic situation. Before the movement of soil layers the seed bank needs to be analysed, with the goal of detecting the seeds of IAS in the seed bank. In the case the seeds of invasive species are found, the soil needs to be considered as determinate. The deposition of soil material can lead to the establishment of IAS shortly after restoration. When IAS occurs in the project area, the invasive species character needs to be evaluated according to their risk assessment. The evaluation of the appropriate control method is based on the ecological life strategies of the invasive species, the competitivity of the native species and the spatial distribution of the occurring IAS. If no IAS are detectable in the project area, it needs to be assured that the construction measures do not support the establishment of IAS.

Construction measures of ecological restorations are massive human disturbances, although the intention in the long term is to have a very positive effect on the areas biodiversity and various ecosystem functions. Therefore an active monitoring process is wanted during the construction period. The focus lies on monitoring the pathways and the re-establishment of a native species' plant cover. The goal is to avoid the introduction of IAS into an uninvaded project area or to avoid the increase of existing populations. In order to decrease costs and the duration of human disturbances it is recommendable to use construction instruments, like forestry equipment and excavating machines, for the eradication activities of invasive plant populations. The reconstruction of a native plant cover is one of the main goals of ecological restorations. The transportation of plant material from outside into a restoration area is always a risk for the introduction of IAS. We recommend using local plant material. Willow sticks, which are often used for the greening of new river courses embankments, should be autochthone produced in the greater area. Furthermore determinated soil should be removed from critical restoration sections.

After the restoration constructions, the reestablishment of native species needs to be assessed to determine what the establishment of IAS should avoid. The evaluation of restoration successes should be established in order to evaluate the impact and development on diversity, vegetation structure, and ecological processes (Ruiz-Jaen & Mitchell Aide 2005). An active monitoring process has to be implemented. The monitoring needs to be planned for a duration of at least 10 years. Within this time the ecological processes along pathways like forest roads or embankments have to be proofed of invasions. The early eradication is necessary to avoid the development of monotypic stands within a recently restored ecosystem. The long-tern monitoring of IAS is a recommended investment following the completion of constructions in order to achieve the goal of restoration.

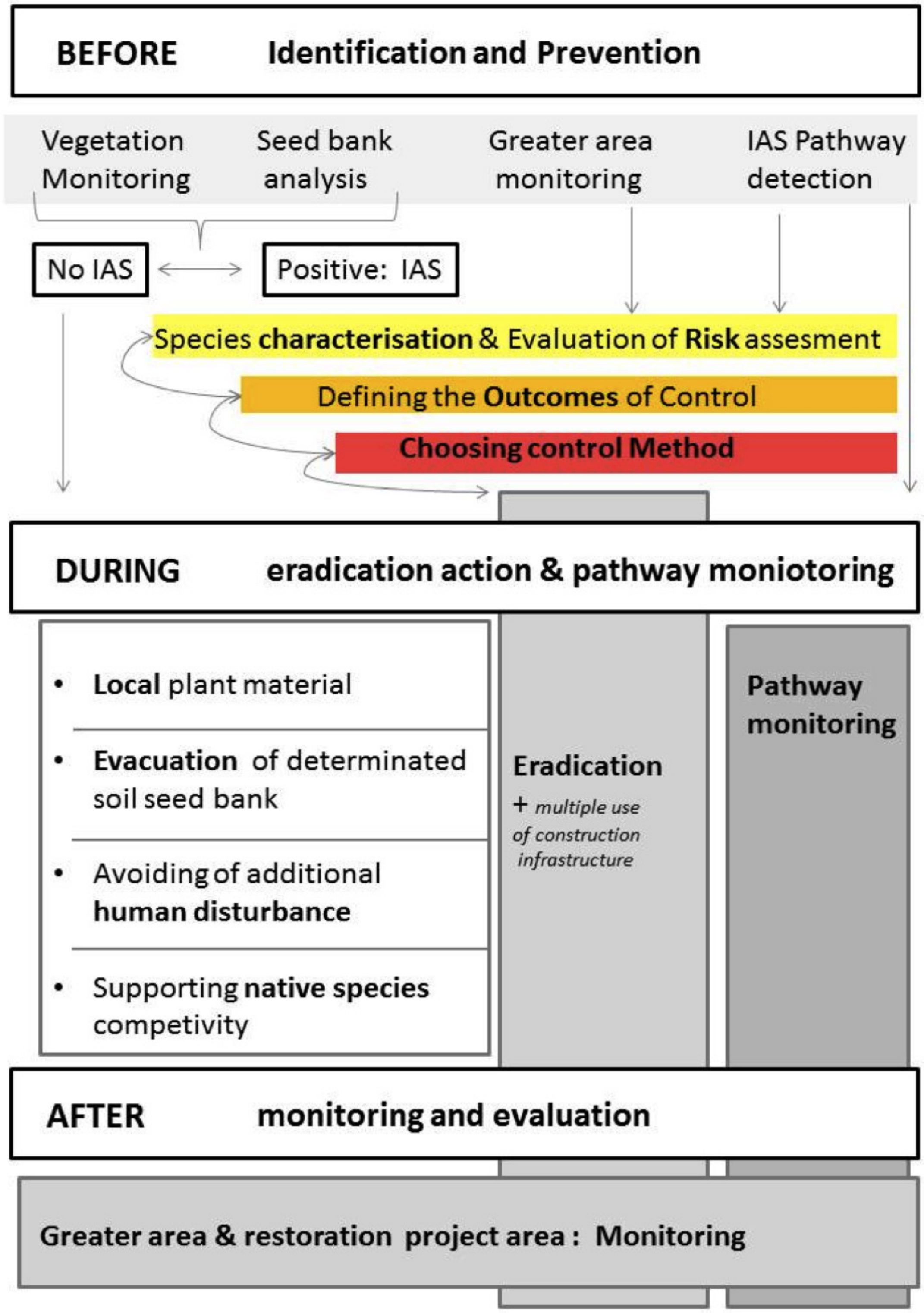


Figure 102 Guidelines for implementation the control and management of IAS in riparian areas for ecological restoration of running water

In the case of the Life + Traisen project biological plant invavions caanot be neglected in order to assure to fulfill the target to improve the environmental conditions fot the development of nature-near diversity. The Monitoring showed a massive spatial distribution of IAS. Especially the invasive species *Solidago gigantea, Imaptiens parviflora, Imatiens glandulifera* and *Bunias orientalis* are endangering the native vegetation stuctucture. Futher the project was heavily disturbed by human cultivation intensity in the forsts. The dislocation and regulation of the river Traisen changed the hydrological conditions and vegetation compositions as well as the habitat structure of the area (Figure 103).

The seed bank analysis supports results of the above ground vegetation monitoring. IAS occure massively in the area. Analyzing the greater area, it is obvious that the insavion is correlated to the large rage of the ecosystem of the river Danube and the urban rage of the greater area of Vienna. This is also detectable within the pathway monitoring. The development of the abundance of IAS is very strong along roads, tht are used for agricultural and silvicultural land use. The transportation of seeds with the riverine water bodies cannot be excluded.

The evaluation of the risk assessment shows the possibility to control IAS regarding the species characteristics. High priority is the prevention of distribution of IAS. Therefore Individuals habe hot be eradicated ed. Further the decontaminated soil is not to use for the restoration measures in the new river course. The eradication action is implemented in the restoration constructions. The main focus lies in the development of habitat quality for biodiversity. Areas with low structural diversity will be either converted to habitat types with a higher diversity value of the mangment will be chaged focusing the improvement of species richness, like for example changing the mowing regime against the high invasion of *Solidago gigantea* populations.

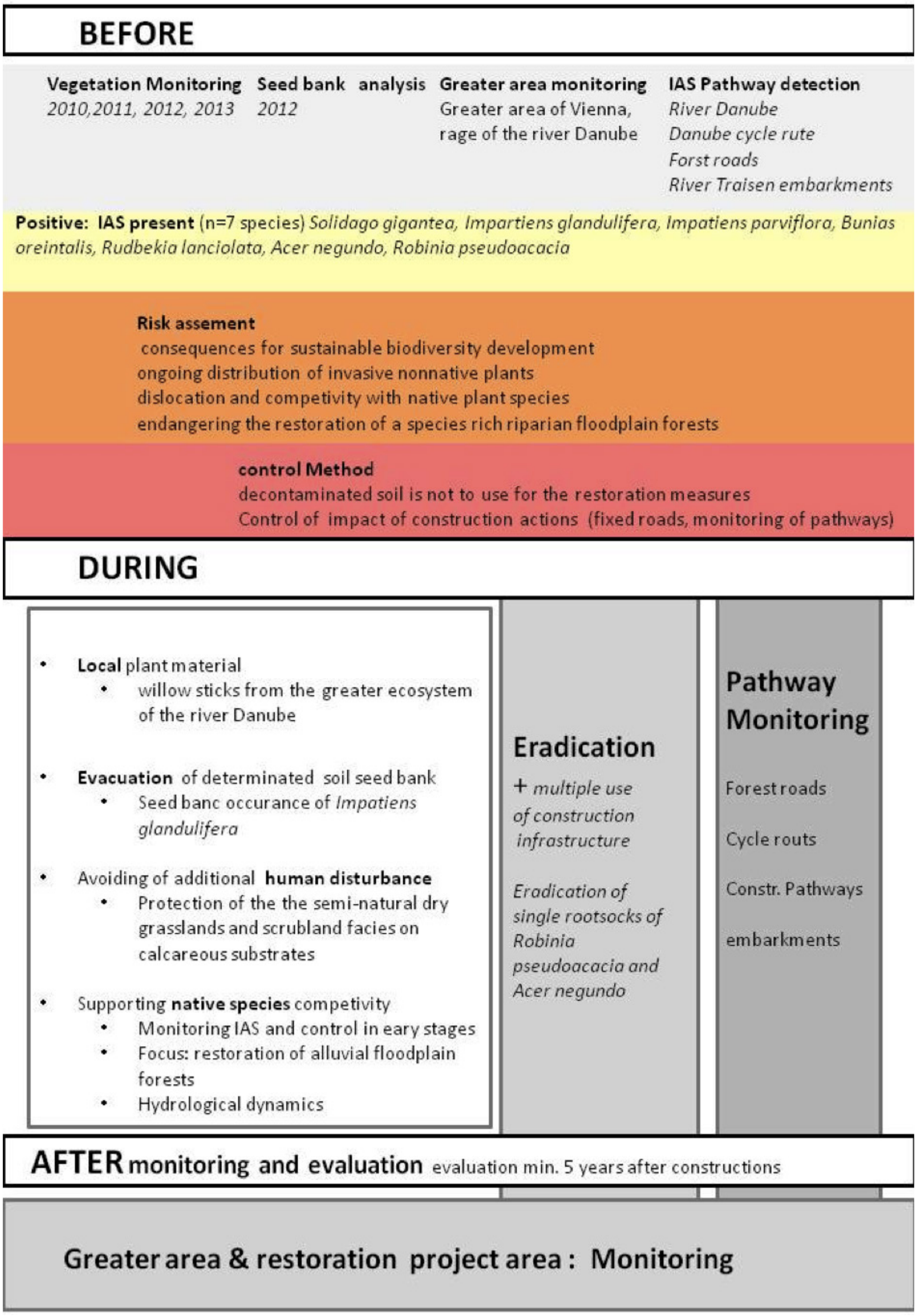


Figure 103 Adapting theoretical guidelines for the ecological restoration project LIFE+ Traisen.

During the construction period, the focus lies on the protection of FFH habitats from the disturbances of the construction measure, like for example the semi-natural dry grasslands and shrubland facies on calcareous substrates (Festuco-Brometalia). The eradication of the root stocks of developed *Robinia pseudoacacia* individuals is an example of the possibility for the recurring use of the construction infrastructure. The unnatural disturbances need to be concentrated to main roads, in order to control the influence on unchanged habitats. The re-creation of the fluvial habitats needs to access with native plant material, like willow sticks from the greater ecosystem of the river Danube. Or, if possible, even from the same project area.

After the conclusion of measures of the restoration, the monitoring process needs to be implemented for the duration of 10 vegetation periods, or at least 5 years. In the first 3 years the competitivity of young vegetation cover of the genus *Salix sp*. with the invasive *Impatiens glangulifera* seeds needs to be supported by the eradication of individuals in the early stage. The eradication of individuals is very important in order to avoid the development of monotypic strands of *Bunias orentalis* or Acer negundo along the recreated vegetation composition. Furthermore, the focus needs to observe the development of IAS populations in the greater area and along the pathways, like forest roads and bike ways in order to detect the danger of new invasive species that had not occured yet in the project area. The monitoring has to include the active eradication actions as well as the support of the native vegetation, like appropriate mowing regimes or the additional planting of willow sticks.

Additionally the Life + Traisen project is a unique chance for informative actions and the implementation of public and private stakeholders. The collaborations of multiple ecological disciplines raise the attractivity for public nature conservation interests. Active and passive programs for the increase of awareness to invasive species are important for the successful control of plant invasions.

4.8 Efficiency of nature conservation instruments

Invasive alien species have become one of the most serious challenges for global and local nature conservation efforts (Sukopp 2001; Pyšek et al. 2003; Cox 2004; Walter et al. 2005; McNeely 2006). The impact of IAS has become a fundamental issue for nature conservation strathegies (EEA 2012). Every year the amount of annunal budget spent on the control of IAS is increasing (DÁntonio 2002). The issue of invasive alien species is discussed in several international conventions and nature conservancy programs (Clout & Williams 2009). My review of the development of the international discussions on IAS shows a mosaic of binding and non-binding instruments. The applications of national legislative frameworks are not sufficient. Regarding the present research on the impact of IAS in riparian areas concerning restoration activities, several frameworks implemented in diverse conservation instruments are taking place.

The study area is a part of over 26,000 protected areas in the European Union, with the aim to guarantee a sustainable development of European species and natural ecosystems (NÖ 07.03.2013). The interdependence and perspectives of restoration programs within a Natura 2000 area was proved. The high level of species richness and diversity of habitats in the study area can be mainly seen as leftovers of the natural riparian ecosystem that was destroyed by several human interventions (Ellmauer et al. 1999). In rapid succession of the regulations of the river Danube and the artificial relocation of the river Traisen the species richness decreased. The restoration of the river Traisen within the project Life+ Traisen has the aim of constructing new typical riverine habitats and to improve the ecological conditions of the study area's diversity (Eberstaller et al. 2000).

Studies have shown a significant occurrence of invasive alien plant species in the study area. Specifically, embankments and sites close to open waterbodies within riparian habitats are invaded by *Impatiens glandulifera. Bunias orientalis* is rapidly developing a strong dominance along open structures with low shading. The proximity to roads supports occurrence of *Bunias orientalis.* Semi-natural habitats like the sites of ash tree forests are mostly invaded by *Impatiens parviflora*. Open structure sites and sites of the habitat types like tall oatgrass meadows and semi-dry grasslands are mostly invaded by *Solidago gigantea.*

The new renaturalized downstream section of the river Traisen is not just an opportunity to improve the ecological situation, but also a challenge to achieve the targets of diversity benefits without supporting the ongoing invasion of alien plant species. The characteristic vegetation structure of natural riparian habitats signifies among others forests with population of *Salix alba, Salix purpurea* and *Salix viminalis* (Pfadenhauer 1997). The construction of the new riverbed profile of the river Traisen supports mainly the FFH habitat types of softwood forests (Eberstaller 2005). The result of seed bank analysis and vegetation development sampling show that areas with high human disturbance are strongly affected. In order to reach the restoration´s aims to improve the diversity in the area a concert invasive species management needs to be developed. The vegetation samples showed a high percentage of the cover of invasive plant species of the herb layer in sites with a naturally, low species richness or sites with low total cover. The restoration will create a high number of uncovered and open grounds. The restoration needs to be considered carefully to achieve these improved targets (Suding et al. 2004; Richardson et al. 2007). The monitoring's aim is to observe this construction process from the earliest stage, and to implement measures as soon as possible. One option to diminish the expected growth of Impatiens glandulifera in the constitution area is to support the wanted growth of willow stick of *Salix alba, Salix purpurea* and *Salix viminalis*. This will be tested as a next step within the ecological restoration programme Life+ Traisen.

The plant diversity of the Natura 2000 *"Tullnerfelder Donau-Auen"* in the study area is highly endangered after the construction of the Danube power plants and the regulation of the river Danube. A significant high number of species of the Red List (Schratt 1990, Niklfeld & Schratt-Ehrendorfer 1999) were recorded within the vegetation samples pulled in 2011 and 2012 (Bernhardt et al. 2013). Using the established nature conservation instrument of the Red List, a level of diversity can be documented. The increase of the total number of invasive plant species is endangering the object of protection and the protection target. The topic of the biological invasion needs to be discussed and refected in the management and land-use programs in the area.

Due to the partly massive dominance of invasive species the removal of invasive plant species is economically not affordable and in most cases ineffective and therefore undesirable (Del Tredici 2004). With respect to ecosystem services the removal of alien plants from massively invaded river embankments is potentially counter-productive (Richardson et al. 2007). Studies have shown that the situation of conservation management and restoration in riparian ecosystems is different (Didham et al. 2005; Clout & Williams 2009). Open and dynamic hydrological structures create individual and riparian vegetation compositions. A successful conservation management process for invasive species and ecological restoration of riparian vegetation asks for participative solutions including human land-use claims and natural ecosystem services. Small-scaled and long-term restorations are recommended (Sweeney et al. 2002; Richardson et al. 2007). Analysis of the seed bank and the phenological development of invasive plant species help to specify monitoring measures. The aim to reach conservation targets includes active monitoring measures to protect uninvaded areas from biological invasion to maintain the integrity of the protected, natural areas for the protection of native flora.

4.9 Public Participation

IAS do not exclusively produce negative effects. Moreover IAS is related to various socio-economic benefits (EEA 2012). The ornamental value of the perennial garden plant *Solidago gigantea* is one example for a direct conflict of socio-economic benefit and negative ecological impact (Lapin & Bernhardt 2013). The social and economic dimension needs to be discussed in order to create comprehensive management guidelines. The promotion on public awareness and education is considered to play a key role in the control of IAS (SpeciesSurvivalCommission 2000).

Public participation is a mechanism to increase the public´s values on ecological projects (Clout et al. 2013). Regarding IAS that were introduced intentionally, like the garden plant *Solidago gigantea*, public behaviors were the main reason for the biological invasion. Raising awareness is considered to be one of the fundamental components for the useful control of the ongoing spread of IAS (Lapin & Bernhardt 2013).

The present case study shows so far a passive participation. Participation is limited to a passive receipt of information (Clout et al. 2013). This is also the case in similarly protected riparian areas. But riparian areas have the great advantage of being accepted as natural ecosystems with a high value for nature conservation. In Europe, riparian areas are mostly the last natural structure in a cultural landscape. Riparian areas are used as nature-based recreation areas. The public´s consideration is positively correlated to the riparian ecosystems. Furthermore, the public interest has been increasing over the last years (Elmore & Beschta 2006). Therefore the control of IAS in riparian areas is positively accepted and understood. Case studies show success in mobilizing volunteers to control invasive plants in riparian areas (Clout et al. 2013). The implementation of public participation has several effects on long-term monitoring and control activities in riparian areas.

In the case of the Life-Traisen project, engaging volunteers in a long-term monitoring would help to reduce costs and increase manpower on the project. One-time events could raise the public's awareness and the community´s sympathy for avoiding a cost-intensive restoration project. The invasive plant species in the study area are well-known to the public. Therefore it could be taken in consideration to train and certify independent volunteers to implement an annual monitoring activity. In the current case, it is important to involve land owners and decision makers. Any public participation needs to be planned carefully to ensure we reach the goal of ensuring the diversity of the riparian area.

4.10 Global Perspective

Biological invasions are a harmful element on the impact of global change on biodiversity (Hellmann et al. 2008). Various models let us expect an ongoing, increasing trend of the negative impact on ecological, economical and health security processes, if these drivers are neglected. The future development of the global distribution of plant species is related to land use changes and climate changes (Pressey et al. 2007; Thuiller et al. 2008). Studies show that the change of land use in combination with eutrofication support the distribution and successful establishment of IAS in local and global level (Burke & Grime 1996; Davis et al. 2000; Eisenhauer et al. 2013; Kollmann 2013).

The increasing global movement is another factor for the increasing distribution of plant species (EEA, 2012). International collaboration is wanted for the implementation of prevention measures. Studies show the need to take action on increasing international trade as one of the main drivers for the global distribution of alien species (Dukes & Mooney 1999, EEA 2012).

Climate change causes many environmental problems. Scenarios prove that increased global warming supports the distribution and establishment of IAS in the future (Mooney & Hobbs 2000; Hübner et al. 2008, EEA 2012). Various climate scenarios describe the consequences of climate change for IAS. The life-cycle of many IAS show a high adaptability to climate variations compared to native species (Lenoir et al. 2008). This is an advantage for the IAS`s competivity. Native plant species will be weakened by environmental changes and displaced by IAS. In addition to the direct competition of IAS and native species, IAS will take advantage of habitat changes, shifts of native plant species to a better climate optimum and disturbances (Dukes & Mooney 1999; Lenoir et al. 2008). Studies also show the massive tendency of land use changes in order to produce biofuels or other alternative energy resources. The ability to invasive spread of energy resource plants needs to be considered (EEA 2012). The importance of interdisciplinary conservation management is increasing, regarding the variety of changes in the future (Hellmann et al. 2008).

Modeling the future is always correlated to a simplification of the current reality. Variations are complex and need to be considered related to the function of the model. Higher complexity leads to an increasing specialization on a more specific habitat, for example. This may not be an advantage for broader decisions (Thuiller et al. 2008). Furthermore, models on IAS distribution have to deal with the unknown, namely whether a arriving alien species will actually become invasive and cause dramatic changes or not (EEA 2012). The invasive species Solidago gigantea for example, was introduced to Europe almost 100 years before it became one of the most invasive plants with a dramatic ecological impact (Lapin 2013). This needs to be taken into account for future conservation management planning (EEA 2012).

Nevertheless models are the basis for conservation planning (Thuiller et al. 2008). The increasing number of scientific contributions on models predicting global and local biodiversity changes shows an ongoing accuracy and simplicity for local use. Using accurate models helps to increase the efficiency of nature conservation management activities. The experience gained over recent decades improved awareness of the danger of IAS. According to ongoing experiments, management measures can become more effective. The combination with early prevention can help to stop the negative impact of IAS (Dukes & Mooney 1999, EEA, 2012).

Within the context of global diversity loss and biological invasions, ecological restoration are discussed very critical (Kollmann 2013). The impact during the construction period is a great impact on landscape ecology and often related to economic effects (Miller & Hobbs 2007; Bullock et al. 2011). To ensure the invasiveness of control methods of IAS within restorations, it is recommended to adapt IAS management concepts to the local level (Kollmann 2013).
Focusing on the situation of riparian areas the invasion of alien plant species is driven by a combination of factors. The future development is mainly dependent on management targets that support the native vegetation composition. Therefore the natural dynamic of running water is necessary, as is the aim of the LIFE+ Traisen project. Climate change asks for constant monitoring of non-native species in the riparian area. The focus of a riparian IAS monitoring is mainly on the moni-

toring of pathways. Riparian areas are on the one hand influenced by the trade route of rivers, like it is the case of the Danube in the the study area. On the other hand, riparian areas are popular attractions for nature-based tourism. In the case of the study area, in 2012 over 100,000 cyclists crossed the study area (ECOplus 2012). The Danube cycle path needs to be considered as a sensitive pathway for the distribution of IAS along a large European riparian area.

The present case study clarifies the complexity of IAS management in European riparian areas. In order to achieve future success in the protection of riparian diversity, an active monitoring demands for high awareness to the combination of driving factors and individual riparian requirements. In the end it is important to mention that the total eradication of IAS in the study area is not to be expected.

5. Conclusion

Biological invasions are not just an interesting scientific issue for vegetation ecologists, but rather a serious problem for humanity's sustainable development. The global natural environment is suffering from the establishment of IAS. Native species are dislocated and habitat structures are massively changed by IAS on the local scale. On the global scale IAS are endangering species richness, ecosystem services and are associated with increasing economic costs.

Europe´s situation is different to other global regions. Since 1500 AD.non-native species have been arriving in Europe and finding their use in various public locations. The natural environment is strongly influenced by human cultivation. Especially riparian areas were heavily affected by hydrological engineering measures. The history of the dislocation of the river Traisen in the study area is a good example of European environmental problems in riparian areas. The restoration project Life+ Traisen is an opportunity to implement IAS management to ecological restoration projects. The research completed in the preparation period prior to the construction of the restoration of the river Traisen is important data for the motoring of IAS during and after the construction periods. Successful management of IAS is based on local data on occurring IAS. Therefore it is recommended to analyse the IAS determination level before the commencement of restoration constructions.

Eight different IAS were found in the study area: *Acer negundo, Ailanthus altissima, Bunias orientalis, Robinia pseudoacacia, Rudbeckia laciniata, Impatiens glandulifera* and *Impatiens parviflora, Solidago gigantea*. The abundance of IAS is increasing. Specifically in artificial habitats, IAS establishment takes place. Tests under controlled conditions show a high competivity of native species to IAS. The influence of floods and other natural processes had a positive effect on the competivity of native plant species. The phenological development showed differences among invasive species. The results underlie that the timing of control methods is essential for the eradication of IAS. Furthermore, monitoring has to focus on the pathways of IAS. In the study area forest roads and water streams are the starting points of invasions. It is also important to consider differences among habitat types. The invasion of each habitat type has another identity and asks for individual controls and monitoring emphasis.

The impact on plant diversity needs to be observed for greater time periods. The establishment of non-native alien plant species often leads in the first moment to an increase in the number of species. The results show a low decrease of species richness in the riparian area. The negative impact of IAS is visible after the development of monotypic strands. At what moment this takes place during the invasion process is related to the IAS life strategy. But instead of retrospective controls, prevention of IAS population development is the most effective step towards the successful management of IAS. Therefore, an interdisciplinary collaboration of stakeholders is unavoidable. The ecological restoration of the river Traisen can be a great improvement for the riparian landscape when IAS monitoring is implemented at all stages of the restorations process.

The total eradication of IAS in European riparian areas are not to be expected, because of the high number of ecological and economical barriers. But the current environmental changes are even supporting biological invasions. Furthermore it should be the aim to support the native species' competitivity. Ecological restorations are an essential part of the redevelopment of sustainable plant diversity.

6. Zusammenfassung

Invasive Neophyten sind gebietsfremde Pflanzen, die sich in heimischen Lebensräumen massiv verbreiten und große Schäden an der heimischen Diversität verursachen. Weltweit schreitet die Verbreitung invasiver Arten rasch voran und wurde zu einer der größten Bedrohungen der globalen Diversität. Besonders Flusslandschaften sind in Mitteleuropa stark von der Verbreitung invasiver Arten betroffen. Um die Auswirkungen auf die Phytodiversität in Aulandschaften zu prüfen wurden im Rahmen des Life+ Traisen Projektes In der Vegetationsperiode 2010, 2011, 2012 und 2013 erfolgten umfassenden Untersuchungen über das Vorkommen und die Verbreitungsbiologie invasiver Arten durchgeführt. Die Untersuchungen zeigten deutlich, dass invasive Neophyten *Impatiens glandulifera, Impatiens parviflor, Solidago gigantea* und *Bunias orientalis* im Projektgebiet einen kritisch hohen Anteil an der Vegetationsstruktur haben. Es wurde beobachtet, dass Wege, Forststraßen und Uferbereiche als Korridor für die Verbreitung dienen. Phenologische Untersuchungen zeigten, dass die Etablierung von Massenbeständen durch Bekämpfung im frühen Stadium bei geringer Anzahl an Individuen eingeschränkt werden kann. Die erhobenen Daten und gewonnen Erkenntnisse dienen zur Entwicklung eines Leitfadens, der vorsieht das Vorkommen invasiver Neophyten in allen Ebenen des Planungs- und Bauablaufs von Renaturierungen von Fließgewässern und dem Management von Schutzgebieten mit einzubeziehen ist. Die Stärkung und Sicherung der Diversität heimischer Lebensräume sowie der sensible Umgang mit gebietsfremden Arten ist bei der Bekämpfung invasiver Arten von oberster Priorität.

7. Abstract

This research is a contribution to the scientific discussion on invasive alien species. Biological invasions are a harmful element on the impact of global change on biodiversity. Riparian areas are important habitats for important ecological functions and European plant diversity. The present case study is focusing on the ecological restoration of the river Traisen. As part of the Life+ Traisen project, the downstream section and the outlet of the river Traisen will be brought into a natural structure. Results show clearly that invasion is following pathways. Plots along roads and embankments are stronger invaded than plots in semi-natural plots. Preventing the introduction of alien plant species is the most effective option for the successful management. The study is correlated to practical advice to nature conservation and ecological restoration issues. Based on the data of invasive species' germination and phenological ability, the impact on plant diversity and habitat preferences, as well as information on invasive species management, guidelines for the management of invasive species in riparian areas was developed in order to reduce the biodiversity loss caused by IAS. The total eradication of invasive species in European riparian areas are not to be expected, because of the high number of ecological and economical barriers. Furthermore it should be the aim to support the native species' competitively.

8. List of tables

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10. Acknowlegment

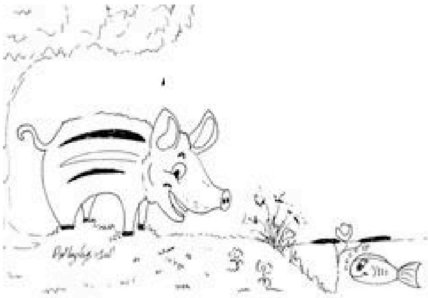
This work is part of the EU-project “Life+ Traisen”. I would like to thank the “Life+ Traisen” planning team for providing information on site management and restoration activities. I would also like to thank the Institute of Botany at the University of Natural Resources and Life Sciences, Vienna, for its technical assistance. I want to thank my advisor Prof. Dr. Karl-Georg Bernhardt for the opportunity to carry out my thesis and for patient guidance.

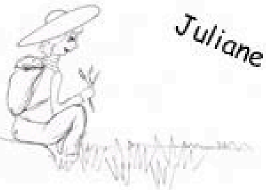
I extend my gratitude to the the forest andminstration of Grafenegg and Stift Herzogenburg for permission to work in the study area. Further I want to thank the project manager DI Helmut Wimmer, VERBUND, for support and the honest interest on the results. I thank Dr. Dipl.Ing. Thomas Kaufmann, office Freiwasser, and Dipl.Ing. Doris Eberstaller-Fleischanderl, ezb TB Eberstaller GmbH, for logistical support and constructive teamwork.

My thanks are due to Priv.-Doz. Mag. Dr. Karin Tremetsberger for critical reading and helpful comments on earlier drafts. Thanks also go to Dominik Aschauer for helping me trough many statistical questions, my dear cousin Alexandra Lapin for help with the difficulties of the English language. Further I want thank Mag. Dagmar Augustin for her never-ending support & motivation and Claudia Battisti for her professional assistance with graphical decisions. I also want to thank Mag. Kosa, the librarian of the Institue of Botany, for his help with literature search.

Thanks to my dear friends, my sports buddies and colleagues, who supported me in various ways. You joined me into the field, motivated me, celebrated with me the achievement of partial objectives, showed interested to my work and offered a lot of diversion that was so important to me! Danke, merci, mockrát děkuji, hvala lijepa, muchas gracias, amigos!

Finally, my highest expression of gratefulness goes to my dear parents Ulrike and Alexander Lapin, my grandparents Ilse and Wilhem Ruhdorfer and my brother Kyrill Lapin, my sisters Xenia Lapin and Teresa Mohn who went through ups and downs of this work together with me and always believed in my succeeding.





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Jörg & Ivana
Martin B. & Joachim
Patrick H.
Oma & Opa
Xenia
Tobi, Lätitia, Andrea & Gerald
Dominik
Martin D.
Margaret
Agnes

Thank you!

Katharina

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