Plant succession research on disturbed sites as a tool for bioindication

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Kurzfassung

Die vorliegende Dissertation befasst sich mit den Möglichkeiten vegetationsökologischer Methoden für die Ingenieursarbeit an gestörten Sonderstandorten. Diese Standorte bedürfen einer besonderen Planung, da im Allgemeinen entweder potentielle Gefahren von ihnen ausgehen, oder die Störungen eine normale Nutzung dieser Flächen unmöglich oder zumindest sehr schwierig machen. Beim Beispiel Deponieflächen wird gezeigt, dass Substrateigenschaften der Deponieabdeckung und Emissionssituation mit Hilfe der Spontanvegetation beschrieben werden können. Dadurch kann der Nachsorgezeitraum von alten Deponien effizient bestimmt bzw. überprüft werden. Gerade bei den Emissionen sind die Ergebnisse der Vegetationsökologie den klassisch angewandten, chemisch-physikalischen Analysemethoden an Aussagekraft überlegen. An einem Fallbeispiel einer Deponie im Mürztal (Steiermark) wird die Anwendbarkeit der Methoden zur Emissionsbeurteilung bewiesen. Das zweite Störungsbeispiel stellen Hochwasserbegleitflächen dar. Durch eine 10 Jahre lange Datenreihe wird die Sukzession an ausgewählten Flächen nach dem Jahrhundertereignis 2002 im Kamptal (Niederösterreich) dokumentiert. Veränderungen der Körngröße des Substrats, Änderungen im Nährstoffangebot der Flächen und der wachsende Konkurrenzdruck innerhalb der Vegetation können überwacht werden. Dadurch können Nachnutzungs- bzw. Pflegemaßnahmen standortsangepasst geplant werden. Das dritte diskutierte Beispiel ist eine geologische Massenbewegung des Jahres 1999 im vorderen Bregenzerwald (Vorarlberg). Die Vegetation eines Teiles des betroffenen Hanges wurde fünf bzw. 13 Jahre später erhoben. Pflanzen zeigen nicht nur den allgemeinen Zustand des Substrates, insbesondere das geologische Material des Untergrundes, sondern auch relativ kleine Unterschiede, die sich z.B. im Basengehalt des Bodens niederschlagen, können exakt aufgezeigt werden. Auch hier könnten Maßnahmenpläne treffsicher auf das jeweilige Substrat angepasst werden und die Effizienz der Maßnahmen dadurch erheblich erhöhen. Die Arbeit beweist die große Bedeutung der Vegetationsökologie in Ingenieursfragen und eröffnet konkrete Anwendungsmöglichkeiten, die kostensparend und effizient eingesetzt werden können. Weitere Forschungsfragen werden in der Arbeit aufgezeigt, deren Ergebnisse sogar noch bessere Aussagen ermöglichen könnten.

Abstract

This thesis focuses on the possibilities of vegetation ecological methods for engineering at disturbed sites. These sites need special planning as they either bear potential risks or the disturbances disable or at least complicate any conventional land use on the sites. The first example deals with landfill sites. Substrate conditions of the landfill cover material and the emission situation can be described via spontaneous vegetation. Based on this information the duration of the aftercare maintenance of old landfills can be estimated in an efficient way. The monitoring of the aftercare phase can be realized by vegetation ecological methods as well. Especially emission control can be done more easily and efficiently by means of spontaneous vegetation compared to classical chemical-physical analyses. The feasibility of the methods to assess the landfill emissions was exemplified at a landfill in the Mürz valley (Styria). The second example comprises flood plain areas. Data from 10 years after a flood in the valley of the river Kamp (Lower Austria) were evaluated. Vegetation ecological methods displayed changes in the particle size composition of the substrates, changes in nutrient supply and the increasing concurrence within the vegetation. Again maintenance and after use plans could be adapted more properly and site specific using such information. The third example is a case study from the Bregenzerwald (Vorarlberg). A geological mass movement had taken place in 1999. The vegetation of a part of the affected area was assessed 5 and 13 years after the movement. The plants displayed the general substrate consistence – especially the geological underground material. Even small differences e.g. of the soil reaction were shown in detail. These results lead to exact, site specific and thereby efficient management plans. The thesis proves in all the cases the high potential of vegetation ecology as a helpful tool for engineers. Specific practical questions are presented, where these methods could help to improve engineering in an efficient and comparably cheap way. Further questions are stated to improve these applications in future.

1 Plant succession research on disturbed sites as a tool for bioindication

Introduction

Bioindication is a useful tool to describe the state of ecosystems. Plants are widely applied for bioindication. The appearance of specific plants, the frequency and the combination of plant species depend on many environmental factors and therefore reflect them in a comprehensive way (Ellenberg et al., 1992). A plant community responds to changing conditions. Rooting plants are connected life-long to the place where they germinated. Bioindication is based on the fact that species or species combinations and whole plant sociological communities require certain site conditions. Site characterizations based on this fact provide more detailed information than chemical or physical variables (e.g. pH or lime content) as they integrate the impact of all factors with all factor interactions over the whole year or even over many years. Due to this fact they serve as indicators of changes or damages in the environment. Vegetation ecology does not only reveal the current stage, but also tells us something about the past as it integrates previous events and long-lasting processes.

Engineers in practice take advantages of the method as it enables them to identify and describe disturbed sites in the landscape. Such sites occur in many facets and they demand different levels of technical and scientific maintenance. The goal is to use the same vegetation ecological methods in different cases of disturbed sites. The methods must be applicable and provide reliable and relevant information.

Bioindication in the nemoral zonobiome

The main drivers of vegetation development in the long term are usually the climatic conditions of the site. Walter and Breckle determined zonal areas of homogenous vegetation due to climatic factors (1986). Main driving variables of the distribution of these areas are the degree of latitude and the altitude. Most areas of Europe are part of the nemoral zonobiome VI. It is characterized by four seasons with snow and frost down to –10 °C in winter and warm, rainy summers and two, relatively long transition seasons (spring, autumn). Mean annual precipitation is about 500 to 1500 mm. The vegetation period lasts about six months, and the minima in temperature during the vegetation period range above + 5 °C. Typical zonal vegetation is characterized by deciduous forests with relatively few tree species. Herbs are dominated by hemicryptophytes and geophytes. During winter time, buds and seeds are situated at or under the soil surface. In higher altitudes the proportion of coniferous trees increases, in subalpine regions coniferous trees dominate.

Within a zonobiome, climatic differences are due to gradients in continentality and altitude. The vegetation pattern is diversified because of the geological ground material and relief, but also small scaled climatic disturbances. Finally biotic interactions with plants, animals and most important human activities lead to different habitat conditions as well. From Western to Central Europe there are only marginal changes. Especially continentality increases from the Atlantic Sea to the Asian central steppe, evidently in Central and Eastern Europe (beginning in Eastern Austria), where colder winters with less snow protection damage the buds of most of the deciduous tree species. Late frost destroys young leafs and blossoms. The stronger drought in summer time also favors the xeromorphic needles of coniferous species (Ellenberg, 1996).

Ecological factors influencing plant communities

Vegetation is widely used as an indicator of ecological variables. Species indicator values play a crucial role in applied plant ecology, even though certain reservation exists to any mathematical processing such as averaging, etc. (Diekmann, 2003).

Water

Generally all parameters are connected with others. In case of water, grazing and slope (inclination) play an important role as they affect vegetation and soil and consequently the water holding capacity (Popp et al., 2009). Changes in the water level in turn have a reflexive interaction with vegetation, which displays the dynamics and connectivity of the soil-plant-atmosphere-continuum.
In peatland the water table represents the driving factor for the vegetation composition. A fluctuating water table can lead to a shift from graminoid to ericoid plant covers and a change in the dominating Sphagnum species (Breeuwer et al., 2009).
Restoration of more arid ecosystems, like e.g. in the zonobiome IV of the Mediterranean Basin, strongly depends on the morphological and physiological traits of the species. Survival rate was found to be highest for legumes, followed by C4 species and leafless shrubs. Typical late-successional woody species, like *Pinus halepensis*, *Olea europea* or *Pistacia lentiscus* – highly recommended for restoration – showed a significantly decreased survival rate (Padilla et al., 2009). Besides water supply, poor soil conditions – typical for the highly degraded Mediterranean vegetation – had important influence on the results.

CO_2 concentration

In the course of the discussion about climate change the CO_2 concentration in the air became a parameter of special interest for plant growth. Erbs et al. (2009) found out that water use efficiency is increased at an elevated CO_2 level. This result corresponds to findings of other authors, who also recognized the number of leaves per tree to be positively correlated to elevated CO_2 levels (Overdieck and Strassemeyer, 2005).

Light

The availability of light is strongly correlated with vegetation succession and therefore also with mechanical disturbances. Successional stages by spontaneous succession of 40 years on former agricultural land strongly correlate with Ellenberg's indicator value of light concerning the herbaceous layer (Dölle et al., 2008). Nitrogen conversion and leaf mass respectively correlate as well with light availability (Hallik et al., 2009).

Nutrients

Nutrient supply and soil quality properties are main factors influencing plant growth. Compost application can improve these factors, which leads to higher productivity (Courtney and Mullen, 2008; Gomiscek, 1999; Nishanth and Biswas, 2008; Tambone et al., 2007). However, the improvement of nutrient supply does not necessarily increase plant growth. Species strategy and life trait play the crucial role in the conversion of an increased nutrient level into biomass (Gerdol, 2005). Vice versa plant growth also changes soil properties like the pH or the soil organic matter (Thiry and Van Hees, 2008).

Grazing

In arid regions the tree regeneration is fostered when seedlings are growing amongst tussocks. The positive effect can be explained by improvement of soil properties and/or by the effect of the "nurse plants". Grazing by large herbivores was found to have a strong negative effect. Among tussocks this effect was less distinct than with other or without vegetation. This is interesting, especially for the restoration of arid ecosystems (Anthelme and Michalet, 2009).

In alpine regions soil fertility and grazing significantly influence specific leaf area and even more leaf dry matter content, leaf size and plant height (Rusch et al., 2009). Plant species composition changes considerably as well. Grazing as a single factor is not significant.

Besides the negative impacts, also positive effects of livestock on vegetation can be found. Dispersion of seeds via epizoochory is an effective method. Couvreur et al. (2005) investigated the dispersion time and distance of the seeds of 12 plant species in the furs of Galloway cattle and Haflinger horses.

Overgrazing by geese led to an increase of summer annuals and total species richness in a Hungarian dry sandy pasture. This can be explained by an increase of small scale disturbances and an increase of seed supply of zoochorous species (Matus et al., 2005). The distribution potential of the ornithochoric species *Taxus baccata* was investigated by Iszkulo and Boratynski (2005). They found out that the main parameter of colonization is the distance to the next mother trees. Precipitation and temperatures in winter and spring were not significant.

A study in the Central Alps examined the effect of grazing exclusion on subalpine and alpine vegetation. The results showed that grazing promotes stress tolerant species with hardly any nutritive value. Competitive species were forced by the exclusion of grazing (Mayer et al., 2009).

Thorny shrubs can facilitate tree regeneration significantly. Especially hardly resistant coniferous saplings (*Abies alba*, *Picea abies*) were supported, whereas the effect was less distinct for the deciduous species *Acer pseudoplatanus* and *Fagus sylvatica* (Vandenberghe et al., 2009).

Shrub population control by grazing is often desired by conservation management. The question arises at which stage population growth is mainly influenced, and thus to gain the best result. Magda et al. (2009) found out for Scotch broom (*Cytisus scoparius*) that safe early life stages had the greatest importance for survival. Even consecutice years of less seed production are not problematic, when the scarce mast years are successful in reproduction. These results can be used to optimize encroachment control through ruminants.

Grazing was also detected to be a main factor of persistence over centuries of sub-halophytic flora in grassland reclaimed from the sea. When species competition was not suppressed by grazing, competitive species came to dominance. Halophytic vegetation was more influence by the salt concentration in the soil than by grazing (Bonis et al., 2005).

Mechanical disturbances

The seed persistence in comparably rarely disturbed plant communities (woodland, pasture) is low, whereas in frequently disturbed habitats such as arable fields it is high. Seed longevity increases with tillage frequency. Additionally, shape and mass

of seeds are adapted to the specific disturbing effects (Albrecht and Auerswald, 2009). The adaption of weed species to the crop type is considerable. In cereal fields archaeophytes are more present, whereas in root crop fields neophytes are dominating (Lososová and Cimalová, 2009). It can be hypothesized that most of the neophytic weed species are typical companions of the root crop species, whereas the archaeophytes are better adapted to the habitat conditions of the cereal species.
When mechanical disturbances stop, arable fields turn into pioneer forests again. The time frame for this succession can differ from 20 to 40 years (Dölle et al., 2008). A natural example of succession after mechanical stress is the flora development in glacier forelands. Plant strategy and successional status are the driving factors for the longevity of species (Erschbaumer and Retter, 2004).
A parameter of interest in vegetation dynamics, which comes to the fore by human transformation of land use, is the fragmentation of landscape. Biodiversity and extinction risk are related to habitat size and singularity of habitats. However, the validity of this general statement differs strongly among species depending on the reproduction trait. Corresponding results were found with respect of mobility of animal species (Öckinger et al., 2009).

Competition

Besides soil and climatic impacts on plant growth and species development also biotic factors can be relevant. An interesting aspect is the description of plant success. On the one hand there are plant growth and biomass production on the other hand there is the goal of a maximized reproduction rate. In many cases there is a trade-off between sexual and vegetative reproduction. However, there are also species that do not show such effects, for instance *Lolium perenne* (Thiele et al., 2009). The ability to colonize new sites is an important property of species viability. Due to climate change in alpine to nival regions, new areas become available for colonization. Most good colonizers in these areas are characterized by diaspores with pappi or narrow wings (Vittoz et al., 2009).

Allelopathy is a wide-spread strategy in species interactions (Fitter, 2003). Plants use allelochemicals – substances to suppress other plant species, but also microorganisms - in order to dominate a habitat. The efficiency of allelochemicals was proven for several species, for instance by Khanh et al. (2009), Kremer and Ben-Hammouda (2009), Qasem (2004) and Sampietro et al. (2007).
Its importance became significant in forests (Blanco, 2007; Wang et al., 2008), but also in agricultural crop production (John et al., 2010). New promising strategies focus on the use of allelopathic effects in weed control (Fan and Marston, 2009; Farooq et al., 2011; Flamini, 2012).

Methods of bioindication

Depending on the specific question there is a big amount of methods applied in vegetation ecology. Generally the indicative properties of every plant species in a community have to be weighted according to its importance for the community. Such properties can be indicator values. For their weighting, different possibilities are in use, as e.g. orthogonal coverage, number of individuals, or the mass of all individuals of a certain species (of the shoots or the whole plant). Orthogonal coverage is common in use as it can be estimated comparably easy. Simple species lists, however, attribute the same weight to every species.

Indicator values

Very significant variables are the indicator values according to Ellenberg et al. (1992). These indicator values give an idea regarding the habitat demands of plant species. Different factors are indicated – climatic ones (light, temperature and continentality) and edaphic ones (moisture, soil reaction and nutrient supply). Furthermore there exist indicators for the tolerance of salt and/or heavy metals in the soil. The indicator values range from 1 (little of the respective factor) to 9 (a lot of the respective factor). Only in case of moisture the further values 10, 11 and 12 are introduced for aquatic plants.

Depending on the question, a set of the most purposive indicator values is used. In case of site investigations often the edaphic indicator values are included. Disturbances usually change or at least influence especially the substrate, and in doing so they change or at least influence especially the moisture and nutrient supply. Even the soil reaction can be influenced. On landfill sites leachate can furthermore increase the salt concentration in the soil (Konold and Zeltner, 1981). Leachate emission as such is comparably rare, and it was not taken into account in detail in the attached papers.

The indicator values of plants in a community were assessed in combination with their coverage. The species coverage was estimated with a combination of different scales. The most widespread one created by Braun-Blanquet (1964) was combined with the finer one established by Londo (1976) as described in Smidt et al. (2007):

r	-	1 – 3 individuals (rare species)
+	-	more individuals but with a negligible coverage
0.1	-	< 1 % coverage
0.2	-	1 – 3 % coverage
0.4	-	3 – 5 % coverage
1	-	5 – 15 % coverage or even less, but high number of individuals
2	-	15 – 25 % coverage or even less, but high number of individuals
3	-	25 – 35 % coverage
4	-	35 - 45 % coverage
5	-	45 - 55 % coverage
6	-	55 - 65 % coverage
7	-	65 - 75 % coverage
8	-	75 - 85 % coverage
9	-	85 - 95 % coverage
10	-	95 - 100 % coverage

This combination provides a maximum in resolution. Estimation according to a fine scale implies a risk of lacking reproducibility. The estimation error can be minimized if all relevés are performed by the same person. The advantage is high resolution and thus information about changes of small scale habitats that are very common at disturbed sites. Often species survive at a low level as the seed supply is provided by surrounding habitats. The fine scale prevents these species to be overrepresented in the statistical data evaluation. Species with no distinct indicator value for a certain variable (species with an indication "x") were excluded from the statistical evaluation as well as neophytic species for which up to now no indicator values at all have been elaborated.

Principal Component Analysis (PCA) was carried out using Unscrambler 9.2 (Camo). For the PCA, matrices were created by summing up the coverage of all species with a certain indicator value. Thus the matrix consisted of R- and N-value with nine

columns each and the F-value with 12 columns. Ubiquitous species reflecting moderate or average conditions (defined by humidity indicator F, soil reaction indicator R or nutrient indicator N between 5 and 7) were removed from the data matrix, as they do not reflect extreme site conditions. The resulting parameter set for the PCA comprised the F-values “1” to “4” and “8” to “12”, the R-values “1” to “4” and “8” to “9”, the N-values “1” to “4” and “8” to “9”.
Beside the indicator values according to Ellenberg et al. (1992), also species indicating waterlogged sites or changing moisture were used as variables. Both site types with low or changing oxygen supply are facilitated by clay minerals. At landfill sites, clay and loam are relevant components in cover layers, in flood plain areas they differentiate the small scale habitats and at a geological mass movement they are relevant for the stability of the slope. Therefore these variables were hypothesized to give good insight in the sites’ history.

Life traits

Further attention focused on the life trait classified according to the following scale:

1	-	annual species
2	-	biennial species
3	-	perennial herbs and grasses
4	-	woody species

For many species this classification was difficult, as they can reproduce annually but can survive even for several years. Following the demands of further mathematical procedures, these species were assigned to only one trait. The resulting error was minimized by the high number of species in all relevés. These life traits can – in some cases - indicate the time span since the latest severe disturbance in the past. Especially for landfill gas detection this information is useful as this kind of disturbance does not cause a mechanical damage of the substrate surface. In flood plain areas, the life traits can clearly demonstrate the successional speed at different relevé sites.

High dimensional statistical methods were performed with the original data to overcome the reservation against mathematical processing of indicator values, even without taking their averages into account. Principal Component Analysis (PCA) was used in all papers with the coverage of species having a certain indicator value as input data. The results of the PCA are visualized in a Scores Plot with the relation of the relevés (data points) and a Loadings Plot with the influence of the original variables on the Principal Components (Brereton, 2003). Instead of one weighted average-value nine variables were taken, one for each step of the Ellenberg scale. The first variable contained the coverage of all species with an indicator value of 1, the second one all species with value 2, and so on. In case of F-value this lead to twelve variables, in case of R- and N-value to nine. As species with indicator values from 5 to 7 usually do not indicate specific site conditions, these variables (F-, R- and N-value from 5 to 7) were excluded from the data set. The remaining data set comprised nine variables for the F-value (F 1 to F 4 and F 8 to F 12), and 6 for the R- and N-value. The exclusion of the variables without high indication ability increased the percentage of data explanation by the first and second Principal Component. The relation of the different relevés in the Scores Plot was almost not affected by the "noise" variables. With this data set the distribution of the data was described without any further mathematical processing. The life traits were also included in the variable set for PCA.

Further variables

Also evenness and the Sørensen index (Sørensen, 1948) were taken as variables. The index was calculated as follows:
Index = 2*a/(2*a+b+c); a = number of species in both lists, b and c are the number of species in just one of the two lists. The index ranges from 0 to 1 (the latter when the species lists are equal).
Evenness was calculated as follows:
$E = H'/H_{max}$; $H_{max} = \ln s$; s = total number of species

H' is the Shannon-Wiener-function (Tremp, 2005):
$H' = -\sum_{i=1}^{s} p_i \ln p_i$; $p_i = \frac{n_i}{N}$
s = total number of species
N = Sum of coverage percentages of all species
n_i = coverage percentage of species i
p_i = relative proportion of species i ranging from 0 to 1

Biodiversity has come to the fore in ecosystem management for several years. Evenness as a diversity variable became relevant especially in the discussion regarding this goal for flood plain areas. As discussed in the respective paper, biodiversity is strongly connected to plant strategies (Grime, 2001).
Sørensen index was chosen as a useful variable to quantify similarities and the changes over time or as consequences of different substrate patterns.

Disturbed sites

Disturbed sites display different effects and degrees of the disturbance. The examples presented in this thesis focus on heavily disturbed sites. Usually such sites need maintenance due to various kinds of present or past catastrophic changes of the habitat conditions. The disturbances can be man-made or natural disasters. Established ways of land use usually are not possible or recommended any more. Neither building activities nor agriculture or forestry are favored. Spontaneous vegetation plays a crucial role in the succession of these areas even when plants and seed mixtures, respectively, are applied. The assessment of the current state and the potential risks is an essential basis for the further development and maintenance plan.
Three relevant types of disturbed sites are shown in this thesis: landfill sites, flood plains and landslides. Landfills are entirely anthropogenic sites. Flood plains also may be heavily influenced by human activities; geological landslide areas, however, is discussed as an example for mainly natural disasters that lead to major disturbances.

Landfill sites

At the investigated landfills, present and potential emissions can be distinguished. Ongoing emissions are relevant for the short and middle-term management plan; potential emissions determine the long-term measures. The monitoring of landfills by means of gas measurements usually comprises the assessment of currently occuring emissions. Potential emissions can be estimated only by investigations of the landfilled material itself. In both cases (present and potential emissions) there are two different approaches: either via chemical and physical properties or by their cumulative effect on organisms. To characterize current emissions, chemical or physical variables are measured at the different emission paths. In the case of leachate, such variables comprise pH value, electrical conductivity, load with organic compounds, metals, salts,... Eco-toxicological tests or the biological oxygen demand (BOD_5) consider cumulative effects. Whenever leachate escapes to the surface vegetation can serve as an indicator. Current gaseous emissions like methane concentration are measured by means of FID (Flame Ionization Detector)-devices. Unfortunately, these emissions strongly intermit in time and space. Therefore satisfying results demand a dense grid of measurements several times a year. The evaluation of vegetation provides the opportunity to improve the results in a cheap, fast and unerring way. Vegetation reacts on landfill gas emissions in different ways. It compiles and reflects chemical and physical conditions over a long period of time. Different species are affected in a specific way (Marchiol et al., 2000). Leachate as well affects vegetation and causes different reactions (Marchiol et al., 1999). Vegetation damages on landfills are recorded by different authors (Blume et al., 1979; Kreh, 1935)

Flood plain areas

Vegetation monitoring in flood plain areas is a useful tool to show the dynamics of a river bank. Height and changes of the ground water table can be assessed as well. Habitat conditions often change very fast in time and space. The composition of soil material, nutrients and water availability can shift from maximum to minimum within a distance of some meters. From the perspective of nature conservation vegetation naturally established after a flood deserves attention and is therefore the pivot of maintenance and protection plans. Vegetation ecological methods serve as monitoring tools also in these cases.

Landslides

Geologically driven landslides have to be monitored to estimate further risks. As these landslides usually are caused by water and the rock and soil composition, the indicator values of plants can provide relevant information. Especially the geological base material often cannot be assessed by means of aerial photography. Usually outcrops and bores are the only possibilities to obtain direct data. Vegetation ecology, of course, can provide information about the rooted layers, but generally the rooted zone of the soil is strongly connected to the geological underground. Especially water infiltration into soils is highly relevant for movements. Drainage ditches to reduce infiltration are rather questionable in clayey material and do not guarantee a satisfactory solution.

Discussion and Outlook

The results presented in this thesis lead to several further questions, but also to possible applications. In case of landfills the opportunity of emission monitoring by means of vegetation can promote a system of permanent landfill control during the after-care phase. Also the end of the after-care phase can be determined in a cheap and comprehensive way. Usually vegetation itself very often is part of the maintenance plan (Prach and Pyšek, 2001; Simmons, 1999; Stalljann and Wendt, 2002). Including the knowledge about the emission situation and the susceptibility of different species, the restoration of the landfill could be arranged in line with the site specific conditions. Further research could improve the prediction of gaseous emissions. Especially plant physiological investigations could be included, when taking specific images of plant damages into account.

Considering management plans for flood plains, vegetation ecology can visualize different substrate conditions and the distance to the water tables. By this information proper goals for the ecosystem can be stated: which parts should be left to natural succession; where and when measures should be realized. Especially grazing activities or extensive agricultural use have to be planned and can be adapted according to the habitat properties indicated by the spontaneous vegetation.

The survey of geological mass movements could profit from a detailed substrate mapping. Measures like a herringbone pattern drainage system become ineffective, when clay and loam dominate in the soil. Under these conditions probably the increase of transpiration by means of shrub or tree plantation seems more favorable. Willow, alder and poplar species could be used according to the different site conditions (Schiechtl, 1995; Schiechtl, 1998). The species and clone choice can also be adapted to the indicative ability of the spontaneous vegetation. Especially among willows there are species for different ranges of water supply and soil reaction (Hörandl et al., 2002).

Altogether this thesis proves that vegetation ecological methods supply engineers with relevant information about the conditions at disturbed sites. The applicability is proven. The main advantages are comparably fast and thereby cheap results and spatial information, valid for a certain time frame (at least the life time of the species found). As vegetation is influenced by the cumulative site conditions, the results provide a comprehensive and reliable site description. Further information could be gained by plant physiological measurements to get deeper insight into the stress phenomena of plants as indicators for changes in the site properties.

Bibliography

Albrecht, H., and K. Auerswald. 2009. Seed traits in arable weed seed banks and their relationship to land-use changes. Basic and Applied Ecology 10:516-524.

Anthelme, F., and R. Michalet. 2009. Grass-to-tree facilitation in an arid grazed environment (Aïr Mountains, Sahara). Basic and Applied Ecology 10:437-446.

Blanco, J.A. 2007. The representation of allelopathy in ecosystem-level forest models. Ecological Modelling 209:65-77.

Blume, H.-P., R. Bornkamm, and H. Sukopp. 1979. Vegetationsschäden und Bodenveränderungen in der Umgebung einer Mülldeponie (Damages to Vegetation and Alterations of Soil near a Sanitary Landfill). Zeitschrift für Kulturtechnik und Flurbereinigung 20:65-79.

Bonis, A., J.B. Bouzillé, B. Amiaud, and G. Loucougaray. 2005. Plant community patterns in old embanked grasslands and the survival of halophytic flora. Flora 200:74-87.

Braun-Blanquet, J. 1964. Pflanzensoziologie. 3rd ed. Springer, Vienna, New York.

Breeuwer, A., B.J.M. Robroek, J. Limpens, M.M.P.D. Heijmans, M.G.C. Schouten, and F. Berendse. 2009. Decreased summer water table depth affects peatland vegetation. Basic and Applied Ecology 10:330-339.

Brereton, R.G. 2003. Chemometrics - Data Analysis for the Laboratory and Chemical Plant John Wiley & Sons Ltd., Chichester.

Courtney, R.G., and G.J. Mullen. 2008. Soil quality and barley growth as influenced by the land application of two compost types. Bioresource Technology 99:2913-2918.

Couvreur, M., K. Verheyen, and M. Hermy. 2005. Experimental assessment of plant seed retention times in fur of cattle and horse. Flora 200:136-147.

Diekmann, M. 2003. Species indicator values as an important tool in applied plant ecology - a review. Basic and Applied Ecology 4:493-506.

Dölle, M., M. Bernhardt-Römermann, A. Parth, and W. Schmidt. 2008. Changes in life history trait composition during undisturbed old-field succession. Flora 203:508-522.

Ellenberg, H. 1996. Vegetation Mitteleuropas mit den Alpen in ökologischer, dynamischer und historischer Sicht. 5th ed. Ulmer.
Ellenberg, H., H.E. Weber, R. Düll, V. Wirth, W. Werner, and D. Paulißen. 1992. Zeigerwerte von Pflanzen in Mitteleuropa. 2nded ed. Goeltze, Göttingen.
Erbs, M., J. Franzaring, P. Högy, and A. Fangmeier. 2009. Free-air CO_2 enrichment in a wheat-weed assembly - effects on water relations. Basic and Applied Ecology 10:358-367.
Erschbaumer, B., and V. Retter. 2004. How long can glacier foreland species live? Flora 199:500-504.
Fan, P., and A. Marston. 2009. How can phytochemists benefit from invasive plants? Natural Product Communications 4:1407-1416.
Farooq, M., K. Jabran, Z.A. Cheema, A. Wahid, and K.H. Siddique. 2011. The role of allelopathy in agricultural pest management. Pest Management Science 67:493-506.
Fitter, A. 2003. Making allelopathy respectable. Science 301:1337-1338.
Flamini, G. 2012. Natural herbicides as a safer and more environmentally friendly approach to weed control: A review of the literature since 2000, pp. 353-396 Studies in Natural Products Chemistry, Vol. 38.
Gerdol, R. 2005. Growth performance of two deciduous *Vaccinium* species in relation to nutrient status in a subalpine heath. Flora 200:168-174.
Gomiscek, T. 1999. Rekultivierung von Deponien mit abfallbürtigen Substraten und Energiepflanzen in Hinblick auf Wasserhaushalt und Biomasseertrag Institute of Waste Management, University of Natural Resources and Applied Life Sciences, Vienna.
Grime, J.P. 2001. Plant strategies, vegetation processes, and ecosystem properties. 2nd ed ed. Wiley, Chichester.
Hallik, L., O. Kull, U. Niinemets, and A. Aan. 2009. Contrasting correlation networks between leaf structure, nitrogen and chlorophyll in herbaceous and woody canopies. Basic and Applied Ecology 10:309-318.
Hörandl, E., F. Florineth, and F. Hadacek. 2002. Weiden in Österreich und angrenzenden Gebieten Institute of landscape planning and ecological engineering, University of Natural Resources and Life Sciences, Vienna, Vienna.
Iszkulo, G., and A. Boratynski. 2005. Different age and spatial structure of two spontaneous subpopulations of *Taxus baccata* as a result of various intensity of colonization process. Flora 200:195-206.
John, J., J. Shirmila, S. Sarada, and S. Anu. 2010. Role of Allelopathy in vegetables crops production. Allelopathy Journal 25:275-312.
Khanh, T.D., L.C. Cong, T.D. Xuan, Y. Uezato, F. Deba, T. Toyama, and S. Tawata. 2009. Allelopathic plants: 20. Hairy beggarticks (Bidens pilosa L.). Allelopathy Journal 24:243-254.
Konold, W., and G.H. Zeltner. 1981. Untersuchungen zur Vegetation abgedeckter Mülldeponien Landesanst. für Umweltschutz Baden-Württemberg, Inst. für Ökologie u. Naturschutz.
Kreh, W. 1935. Pflanzensoziologische Untersuchungen auf Stuttgarter Auffüllplätzen. Jahreshefte des Vereins für vaterländische Naturkunde in Württemberg 91:59-120.
Kremer, R.J., and M. Ben-Hammouda. 2009. Allelopathic plants. 19. Barley (hordeum vulgare L). Allelopathy Journal 24:225-242.
Londo, G. 1976. The decimal scale for relevés of permanent quadrats. Vegetatio 33:61-64.
Lososová, Z., and S. Cimalová. 2009. Effects of different cultivation types on native and alien weed species richness and diversity in Moravia (Czech Republic). Basic and Applied Ecology 10:456-465.
Magda, D., E. Chambon-Dubreuil, C. Agreil, B. Gleizes, and M. Jarry. 2009. Demographic analysis of a dominant shrub (*Cytisus scoparius*): Prospects for encroachment control. Basic and Applied Ecology 10:631-639.
Marchiol, L., C. Mondini, L. Leita, and G. Zerbi. 1999. Effects of Municipal Waste Leachate on Seed Germination in Soil-Compost Mixtures. Restoration Ecology 7:155-161.
Marchiol, L., S. Cesco, R. Pinton, and G. Zerbi. 2000. Germination and Initial Root Growth of Four Legumes as Affected by Landfill Biogas Atmosphere. Restoration Ecology 8:93-102.
Matus, G., M. Papp, and B. Tóthmérész. 2005. Impact of management on vegetation dynamics and seed bank formation of inland dune grassland in Hungary. Flora 200:296-306.
Mayer, R., R. Kaufmann, K. Vorhauser, and B. Erschbamer. 2009. Effects of grazing exclusion on species composition in high-altitude grasslands of the Central Alps. Basic and Applied Ecology 10:447-455.
Nishanth, D., and D.R. Biswas. 2008. Kinetics of phosphorus and potassium release from rock phosphate and waste mica enriched compost and their effect on yield and nutrient uptake by wheat (*Triticum aestivum*). Bioresource Technology 99:3342-3353.

Öckinger, E., M. Franzén, M. Rundlöf, and H.G. Smith. 2009. Mobility-dependent effects on species richness in fragmented landscapes. Basic and Applied Ecology 10:573-578.

Overdieck, D., and J. Strassemeyer. 2005. Gas exchange of *Ginkgo biloba* leaves at different CO_2 concentration levels. Flora 200:159-167.

Padilla, F.M., R. Ortega, J. Sánchez, and F.I. Pugnaire. 2009. Rethinking species selection for restoration of arid shrublands. Basic and Applied Ecology 10:640-647.

Popp, A., N. Blaum, and F. Jeltsch. 2009. Ecohydrological feedback mechanisms in arid rangelands: Simulating the impacts of topography and land use. Basic and Applied Ecology 10:319-329.

Prach, K., and P. Pyšek. 2001. Using spontaneous succession for restoration of human-disturbed habitats: Experience from Central Europe. Ecological Engineering 17:55-62.

Qasem, J.R. 2004. Allelopathic plants: 11. Cardaria draba (L.) Desv. Allelopathy Journal 13:165-172.

Rusch, G.M., C. Skarpe, and D.J. Halley. 2009. Plant traits link hypothesis about resource-use and response to herbivory. Basic and Applied Ecology 10:466-474.

Sampietro, D.A., J.R. Soberón, M.A. Sgariglia, E.N. Quiroga, and M.A. Vattuone. 2007. Allelopathic plants. 17. Sugarcane (Saccharum officinarum L.). Allelopathy Journal 20:243-250.

Schiechtl, H.M. 1995. Pflanzen als Mittel zur Bodenstabilisierung, p. 50-62 Ingenieurbiologie - Wurzelwerk und Standsicherheit von Böschungen und Hängen. Jahrbuch 2 der Gesellschaft für Ingenieurbiologie e.V., Vol. 2. Sepia, Aachen.

Schiechtl, H.M. 1998. Die Bedeutung der Erlen in der alpinen Ingenieurbiologie, p. 201-212 Ingenieurbiologie - Wurzelwerk und Standsicherheit von Böschungen und Hängen. Jahrbuch 7 der Gesellschaft für Ingenieurbiologie e.V., Vol. 7. Sepia, Aachen.

Simmons, E. 1999. Restoration of landfill sites for ecological diversity. Waste Management and Research 17:511-519.

Smidt, E., J. Tintner, and K. Meissl. 2007. New Approaches of Landfill Assessment and Monitoring, p. 191-225, *In* A. A. Velinni, ed. Landfill Research Trends. NOVAPublisher.

Sørensen, T. 1948. A method of establishing groups of equal amplitude in plant sociology based on similarity of species and its application to analyses of the vegetation on Danish commons. Biologiske Skrifter / Kongelige Danske Videnskabernes Selskab 5:1-34.

Stalljann, E., and P. Wendt. 2002. Deponiebegrünungen als Erosionsschutz und Möglichkeit zur Reduzierung von Sickerwasser. Müll und Abfall 4:187-192.

Tambone, F., P. Genevini, and F. Adani. 2007. The effects of short-term compost application on soil chemical properties and on nutritional status of maize plant. Compost Science and Utilization 15:176-183.

Thiele, J., R.B. Jorgensen, and T.P. Hauser. 2009. Flowering does not decrease vegetative competitiveness of *Lolium perenne*. Basic and Applied Ecology 10:340-348.

Thiry, Y., and M. Van Hees. 2008. Evolution of pH, organic matter and 226radium/calcium partitioning in U-mining debris following revegetation with pine trees. Science of the Total Environment 393:111-117.

Tremp, H. 2005. Aufnahme und Analyse vegetationsökologischer Daten Ulmer, Stuttgart.

Vandenberghe, C., C. Smit, M. Pohl, A. Buttler, and F. Freléchoux. 2009. Does the strength of facilitation by nurse shrubs depend on grazing resistance of tree saplings? Basic and Applied Ecology 10:427-436.

Vittoz, P., N. Dussex, J. Wassef, and A. Guisan. 2009. Diaspore traits discriminate good from weak colonisers on high-elevation summits. Basic and Applied Ecology 10:508-515.

Walter, H., and S.-W. Breckle. 1986. Ökologie der Erde UTB.

Wang, H.X., G.Z. Li, D.M. Yu, and Y.M. Chen. 2008. Barrier effect of litter layer on natural regeneration of forets: A review. Chinese Journal of Ecology 27:83-88.

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Can vegetation indicate landfill cover features?

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ABSTRACT

Generally, great efforts are made in measuring features of landfill covers. However, conventional physical or chemical parameters reach their limits in indicating the small scale changes of the habitats. Bio-indication is a proven tool to assess habitat conditions. The advantages of vegetation monitoring are obvious: cheap, easy, and integrating over time and space. Our study displays, how vegetation can indicate landfill cover features by adapting some common evaluation methods. Ellenberg's ecological indicator values were used, but ubiquitous species were excluded from multivariate data analysis of the Ellenberg values. Four groups of habitats were distinguished according to their cover material: (i) loamy substrates; (ii) wet hollows and areas with mature compost; (iii) fresh compost and mechanically biologically treated waste; (iv) slag from municipal solid waste incineration and leachate-influenced areas with fresh untreated waste or sewage sludge. The differences were assessed by ecological indices. The results give a promising impression of the potential vegetation monitoring has in the indication of landfill cover features.

Introduction

Assessment of landfill top cover features is equally important for landfill managers and authorities, also with respect of contaminated soil or abandoned waste dumps. Measurement of chemical or physical parameters is expensive, and gas emissions, compaction or contamination can vary strongly both in time and space. The vegetation on top covers seems to be a more proper indicator: It reveals integrative information about the spatial variation of the conditions and may allow extrapolations about the landfill development, at least over the life time of the plant species involved. Furthermore, the method of phyto-sociological relevés is fast and cheap. Up to now, the combination of phyto-sociological relevés with Ellenberg's (1992) ecological indicator values has not been used frequently for the assessment of landfill vegetation. The heterogeneity of top covers and the big amount of influencing factors seemed to hamper a reliable result. Konold and Zeltner (1981) combined the vegetation data of 22 different German landfill sites, but did not take into account the occurrence of different habitat types. As a result, an evaluation of substrate conditions seemed impossible. Only the influence of leachate and gaseous emissions was reflected.

We assume that up to now classical plant sociology has not produced sufficient landfill-relevant results yet. The published associations do not fully mirror the dynamics on a landfill. Many of the plant communities growing on landfills in Austria even might not have been classified yet (Forstner, 1982, 1984; Hübl and Holzner, 1974; Mucina et al., 1993).

While an experienced European botanist would not need substantial statistical support for the characterization of different landfill habitats based on the existing vegetation, landfill owners and even botanists from outside Europe might be grateful for a statistical reassurance that landfill top covers can be divided into different habitat types by principal component analysis. We hypothesized that multivariate data analysis provides suitable methods to display the data structure by combining various parameters. In this paper the ecological indicator values after Ellenberg et al. (1992), adapted by Karrer and Kilian (1990) for Austrian conditions, are combined with the life span of the respective species. The indicator values are in common use to describe substrate conditions (Diekmann, 2003) even though the mathematical treatment of Ellenberg's ordinal scale reveals some problems.

Diversity is seen as a target parameter for environmental protection and eco-engineering. β-Diversity and evenness have therefore become important parameters to describe the biodiversity and by this the ecological value of vegetation units. On landfills several species-poor plant communities occur. But they coexist with highly diverse ones of ruderal origin, where a great number of species stands for a mosaic of small-scale habitat arrangements or different successional stages.

The aim of this paper was to prove to which extent history, construction methods and substrate quality of landfill top covers

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Table 1
Climatic conditions of the 13 landfill sites.

Site no.	Federal state	Climatic region	Mean annual precipitation (mm)	Mean annual temperature (°C)	Altitude above sea level (m)
1	Lower Austria	Subatlantic	600	9.5	200
2	Lower Austria	Subatlantic	660	9.2	270
3	Styria	Alpine	900	6.1	650
4	Vienna	Pannonian	520	9.8	170
5	Tyrol	Alpine	1180	8.4	550
6	Lower Austria	Subatlantic	600	9.2	280
7	Lower Austria	Pannonian	500	8.8	230
8	Lower Austria	Pannonian	600	9.5	370
9	Upper Austria	Subatlantic	840	8.2	350
10	Upper Austria	Subatlantic	830	9.8	290
11	Lower Austria	Pannonian	510	8.9	240
12	Lower Austria	Pannonian	550	9.9	210
13	Styria	Alpine	800	7.2	550

result in different floristically circumscribed vegetation types that can be assessed by the above mentioned, quick and cheap methods of vegetation science.

Materials and methods

Sites and phytosociological assessments

For the presented study 13 landfill sites in Austria were investigated. They are distributed all over the country, with a predominance of Eastern Austria where landfill sites are more frequent, and belong to climatically different regions (Table 1).

Firstly, these landfill sites were divided into homogeneous areas. These areas were characterized by the landfill managers with regard to substrate type and peculiarities as well as to dimension of top cover and eventual occurrence of leachate and gas emissions (Table 2). Similar homogeneous areas were found on different landfill sites. On these structurally well described areas phyto-sociological relevés were conducted, e.g. relevés of the homogeneous area type “A” were performed on six different landfill sites. The time of application of the substrate and thereby the start of succession was not exactly known in most cases. This time span ranged between 3 and 15 years.

Altogether, on the homogeneous areas with sufficient site information 165 phyto-sociological relevés were conducted. Species nomenclature follows Fischer et al. (2005). The coverage of the species was estimated using the scale presented in Smidt et al. (2007). It is a modified Londo scale (Londo, 1976) with an additional “r” for single individuals in a relevé.

The relevés were analysed by the program VegePro (©Irene Ahamer). VegePro calculates weighted mean indicator values for the *F* (moisture situation)-, *R* (soil reaction)- and *N* (nutrient supply)-indicator values and the life span (LF). The program combines the cover percentages in the relevés and the modified Ellenberg indicator values for the Pannonian region by Karrer and Kilian (1990). Furthermore, life span data scale is used as following (Smidt et al., 2007): “1” – annual species, “2” – biennial species, “3” – perennial forbs, and “4” – perennial woody species.

Defining homogeneous areas

First of all, weighted mean indicator values were calculated for all relevés comprising all scores of the Ellenberg scale from 1 to 9 (respectively 12 for the *F*-value). Mean life span was calculated for all relevés as well. The impact of the factor “homogeneous area” was tested by ANOVA ($\alpha = 0.05$) and Bonferroni’s *t*-tests as post hoc test (Abdi, 2007).

Second, a principal component analysis (PCA) was carried out using the Unscrambler 9.2 software (©Camo) to distinguish different groups of homogeneous areas, each of those representing a separate habitat type. The species of every relevé were arranged according to their respective indicator values (for *F*, *R*, *N*, as well as for LF) so that the cover percentage of all species with a certain *F*, *R*, *N* or LF-value could be summed up separately. This means that the entire species number of one relevé was split according to their indicator values.

As ubiquitous species with *F*-values from 5 to 7, *R*-values from 5 to 7, and *N*-values from 5 to 7 would not indicate extraordinary site conditions, species with these Ellenberg values were then excluded from the procedure. So only the cover percentages of the remaining species with a meaningful indicator value were summed up. The resulting parameter set for the PCA comprised the *F*-values “1” to “4” and “8” to “12”, the *R*-values “1” to “4” and “8” to “9”, the *N*-values “1” to “4” and “8” to “9”. LF-values ranged from “1” to “4” as indicated above. In Tables 3 and 4 of the combined presence the most frequent species included in the calculation are marked by a “√”as good indicator species.

The stepwise calculation of the indicator values improved interpretation and evaluation of the data. The resulting principal component analyses displayed the homogeneity of various habitat types, each of these types comprising 2 or more homogeneous areas.

Constancy (relative number of relevés containing a species), cumulative species cover and “combined presence” of all species of the resulting groups were calculated. “Combined presence” was calculated by multiplying constancy and cumulative coverage. We preferred the calculation of “combined presence” to a simple assessment of constancy. This approach is justified because most of the frequent species on landfill covers develop their indicative qualities not only by their simple presence but also especially by massive growth and cover under most suitable conditions.

For a detailed differentiation, the two homogeneous areas A (slopes with loamy substrate) and B (methane oxidation layers) were compared by parameters for diversity, evenness and species number. H', calculated by the Shannon function, was used as the parameter of diversity (Tremp, 2005), evenness was calculated as follows (Häupler, 1982):

$$E = \frac{H'}{H_{\max}}; \quad \text{where} \quad H_{\max} = \ln s; \quad s = \text{total number of species}$$

A Welch-test was used instead of the common *t*-test to differentiate the two groups ($\alpha = 0.05$, $\beta = 0.10$) (Rasch et al., 2009).

To compare β-diversity between the homogeneous areas A and B, the coefficients of Sørensen (1948) and Jaccard (1912) were used. They are assumed to be the most proper ones to describe similarities of two different communities (Chambers, 1983). Area A comprised 29 relevés, among them 8 relevés from landfill site 1 (Table 1), and area B 22 relevés, 10 of which also from site 1.

Table 2
Site information of the homogeneous areas (A–J: sufficient information regarding the substrate; K–O: insufficient information not shown).

	A	B	C	D	E	F	G	H	J
Relevés of site no.	1, 2, 8, 10, 11, 12	1, 11	1, 2, 3, 5, 6, 8, 10, 12, 13	2, 13	2, 7, 10, 13	2, 6, 8, 9, 10, 11, 12, 13	4	12, 13	7, 10, 11
Inclination	0–30°	Flat	0–20°	Flat	0–10°	Flat	0–30°	Flat	0–10°
Exposition	South to east	No	Mixed	No	Mixed	No	East	No	Mixed
Substrate type and peculiarities	Sandy to clayey loam	Compost	More or less loamy material	MBT[a] material	Untreated waste, sewage sludge, fresh soil excavation heaps	Old, mature compost	MSWI slag[b]	Loam, wet hollows	Sandy to loamy, dense liner (HDPE[c] plus loam layer)
Dimensions of top cover (m)	0.5–1.5	0.8–1.0	0.2–2.0	0	0	0.8–1.5	0	0.2–2.0	0.2–2.0
Leachate	No	Unknown	No	Unknown	Occurring	Unknown	Unknown	Unknown	Unknown
Gas	No	Methane oxidation layer	Unknown	Unknown	Unknown	Unknown	Unknown	Unknown	Unknown
Seed mixture applied by landfill manager	Unknown	Unknown	Seed mixture with unknown species composition	None	None	Unknown	Seeded species not found onsite	Unknown	Seeded species: e.g. *Festuca rubra*, *Lotus corniculatus*

[a] Mechanically biologically treated municipal solid waste.
[b] Municipal solid waste incineration slag.
[c] High density polyethylene.

Table 3
The 15 species with highest combined presence in the habitat types "ACJ" and "FH", respectively, and their share of total combined presence, in brackets the indicator values for *F*, *N*, LF; good indicator species are marked by a "√".

Habitat type ACJ	%	Habitat type FH	%
Elymus repens (*x*; 7; 3)	27.5	√*Urtica dioica* (6; 9; 3)	60.9
Festuca rubra ssp. *rubra* (6; *x*; 3)	24.6	√*Galium aparine* (*x*; 8; 1)	5.9
Dactylis glomerata (5; 6; 3)	12.7	√*Phalaris arundinacea* (8; 7; 3)	5.7
Plantago lanceolata (*x*; *x*; 3)	3.8	√*Artemisia vulgaris* (6; 8; 3)	3.8
Achillea millefolium agg. (*x*; *x*; 3)	3.2	*Elymus repens* (*x*; 7; 3)	3.8
√*Lotus corniculatus* (4; 3; 3)	3.1	*Arrhenatherum elatius* (*x*; 7; 3)	3.6
Arrhenatherum elatius (*x*; 7; 3)	2.4	*Calamagrostis epigejos* (*x*; 6; 3)	3.0
√*Artemisia vulgaris* (6; 8; 3)	2.3	*Dactylis glomerata* (5; 6; 3)	1.4
Cirsium arvense (*x*; 7; 3)	1.6	√*Calystegia sepium* (6; 9; 3)	1.2
Poa pratensis (5; 6;3)	1.5	√*Chelidonium majus* (5; 8; 3)	1.2
Calamagrostis epigejos (*x*; 6; 3)	1.4	√*Sambucus nigra* (5; 9; 4)	1.1
√*Securigera varia* (4; 3; 3)	1.3	√*Impatiens glandulifera* (8; 7; 1)	0.9
Phleum pratense (5; 7; 3)	1.2	√*Chenopodium album* (4; 7; 1)	0.9
Lolium perenne (5; 7; 3)	1.2	√*Phragmites australis* (10; 7; 3)	0.6
Tripleurospermum perforatum (*x*; 6; 2)	1.1	√*Anthriscus sylvestris* (5; 8; 3)	0.5
Sum over all 278 species of the habitat type	100.0	Sum over all 135 spec. of the habitat type	100.0

First we calculated the indices for the comparisons of all relevés within a homogeneous area (A × A resp. B × B), then the relevés were compared with the relevés of the other homogeneous area (A × B). This was done for all relevés of the homogeneous areas (results in Fig. 4(a)). Furthermore only the relevés of landfill site 1 (8 out of 29 for A and 10 out of 22 for B) were taken for the comparisons (results in Fig. 4(b)). Last the relevés of landfill site 1 were taken for the comparisons A × B (8 × 10) and of different landfill sites for the comparisons A × A (21 × 21) resp. B × B (12 × 12) (results in Fig. 4(c)). Evenness, diversity and species number per relevé were also compared by ANOVA ($\alpha = 0.05$) and Bonferroni's *t*-tests as post hoc tests (Abdi, 2007).

Results

Plant communities

A reliable assignment of the relevés to published phyto-sociological associations cited by Oberdorfer (1983), Schubert et al. (2001) or Mucina et al. (1993) turned out to be impossible in most cases. Annual ruderals like *Chenopodium album* for instance intermingled homogeneously with *Elymus repens* or *Artemisia vulgaris*, character species for different phyto-sociological classes. So we tried to assign the species combinations found at least to relevant alliances. A short description of the communities follows below:

A: Species of the alliances Convolvulo-Agropyrion repentis (Görs, 1966) and Sisymbrion officinalis (Tüxen, 1950), partly Erigeron annuus-sociation (Görs, 1968) (Arction lappae)
B: Species of Chenopodium album-sociation (Atriplicion nitentis, Passarge, 1978) and alliance Convolvulo-Agropyrion repentis (Görs, 1966) (Artemisietea vulgaris – Agropyretalia repentis), plus *Dactylis glomerata* and *Festuca rubra* (maybe seeded)
C: Mixture of seeded grass species and Tanaceto vulgaris-Arrhenateretum elatoris (Fischer, 1985), Cynosurion cristati (Tüxen, 1947) and Convolvulo repentis-Agropyretum repentis (Felföldy, 1942)

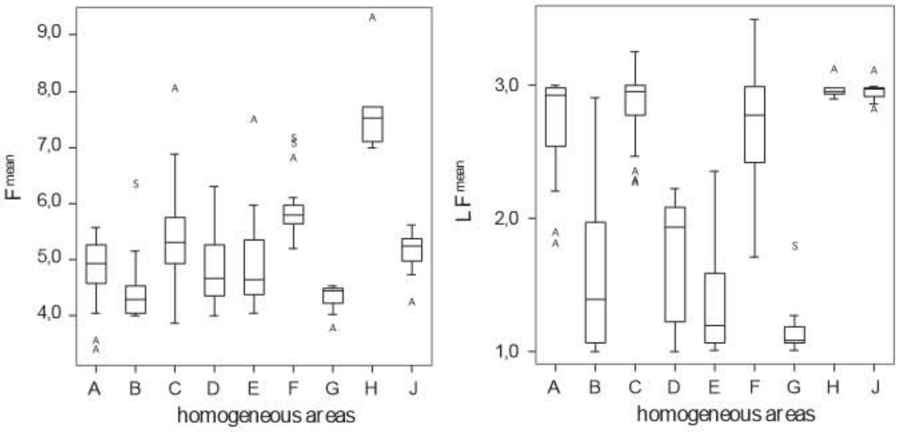

Fig. 1. Median and distribution of ecological indicator value F_{mean} (moisture) – [Ellenberg et al., 1992 resp. Karrer and Kilian, 1990] – and the life span (LF_{mean}).

D: Atriplicion nitentis (Passarge, 1978) plus Elymo repentis-Sisymbrietum loeselii (Mucina, 1993)
E: Atriplicion nitentis (Passarge, 1978) but *Sisymbrium* spp. absent
F: Urtico dioicae-Aegopodietum podagrariae (Tüxen, 1947) ex Görs (Görs, 1968) with additional accompanying grass species (*Elymus repens*, *Arrhenaterum elatius*, *Calamagrostis epigejos*, *Festuca rubra*)
G: Brometum sterilis (Görs, 1966) (Sisymbrietea officinalis – Sisymbrietalia officinalis – Sisymbrion officinalis, Tüxen, 1950) and other species characterizing the same alliance, additionally *Bassia scoparia*, *Senecio vernalis*, and *Anthriscus caucalis*

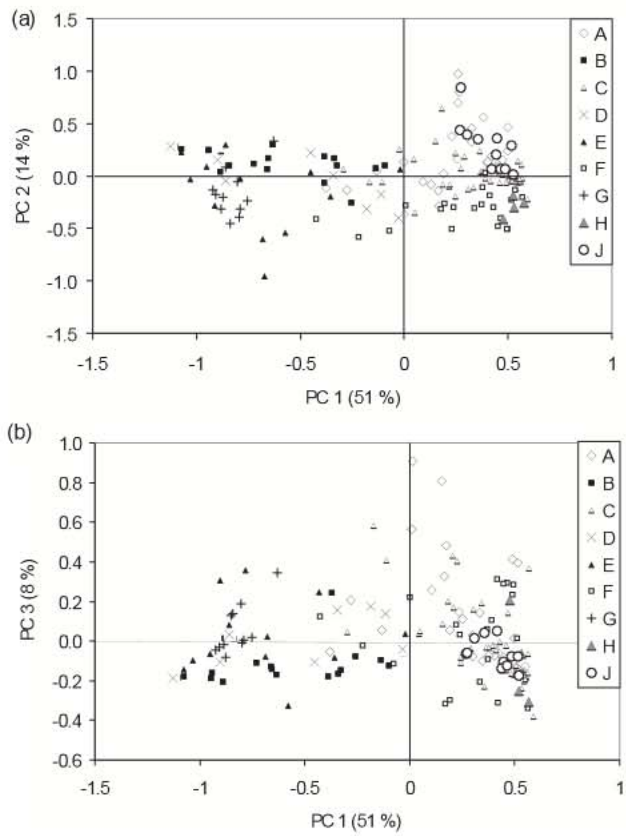

Fig. 2. Scores plots of the first and second (a) and the first and third (b) principal component (data comprise 165 relevés).

H: Mixture of Senecion fluviatilis (Tüxen, 1950), Phragmition australis (Koch, 1926) and Caricion elatae (Koch, 1926) species
I: Seeded grassland, *Festuca rubra*, *Lotus corniculatus*, *Plantago lanceolata*, *Dactylis glomerata* and *Achillea millefolium* agg. dominating

Weighted mean indicator values

In Fig. 1 means of Ellenberg's indicator values for moisture (*F*) and the life span scores (LF) for the nine floristically different homogeneous areas are given. Only the life span differentiates roughly. The homogeneous areas A, C, F, H, and J are dominated by perennial species, whereas the others are dominated by annual and biennial species, respectively. Except for the areas dominated by wetland plants (*H*) the Ellenberg values do not display obvious differences

Table 4
The 15 species with highest combined presence in the habitat types "BD" and "EG", respectively, and their share of total combined presence, in brackets the indicator values for *F*, *N*, LF; good indicator species are marked by a "√".

Habitat type BD	%	Habitat type EG	%
√*Chenopodium album* (4; 7; 1)	38.4	√*Bassia scoparia* (?;?; 1)	31.3
√*Amaranthus retroflexus* (4; 7; 1)	35.6	√*Stellaria media* (x; 8; 1)	8.9
Dactylis glomerata (5; 6; 3)	8.0	√*Anthriscus caucalis* (5; 6; 1)	8.3
√*Sisymbrium loeselii* (4; 5; 1)	4.9	√*Chenopodium album* (4; 7; 1)	7.4
√*Artemisia vulgaris* (6; 8; 3)	3.4	√*Sisymbrium loeselii* (4; 5; 1)	7.1
Elymus repens (x; 7; 3)	2.3	√*Senecio vernalis* (4; 6; 1)	3.9
√*Bassia scoparia* (?;?; 1)	1.9	√*Atriplex prostrata* (6; 9; 1)	3.4
√*Atriplex sagittata* (x; 7; 1)	1.5	√*Bromus sterilis* (4; 5; 1)	3.4
√*Chenopodium ficifolium* (4; 7; 1)	0.5	√*Echinochloa crus-galli* (5; 8; 1)	3.2
√*Urtica dioica* (6; 9; 3)	0.3	√*Lactuca serriola* (4; 4; 2)	2.8
√*Populus nigra* (8; 7; 4)	0.3	*Elymus repens* (x; 7; 3)	2.7
√*Ballota nigra* (5; 8; 3)	0.3	√*Tripleurospermum perforatum* (x; 6; 2)	2.7
√*Carduus acanthoides* (4; 7; 2)	0.3	√*Solanum lycopersicum* (0; 0; 1)	2.3
Arrhenatherum elatius (x; 7; 3)	0.3	*Taraxacum officinale* agg. (5; 8; 3)	1.9
Calamagrostis epigejos (x; 6; 3)	0.2	√*Persicaria lapathifolia* ssp. *lapathifolia* (8; 8; 1)	1.9
Sum over all 76 species of the habitat type	100.0	Sum over all 122 species of the habitat type	100.0

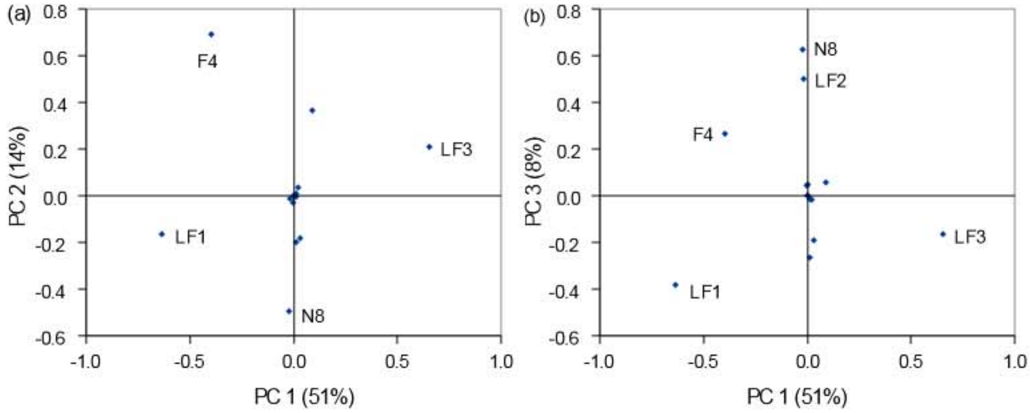

Fig. 3. Loading plots of the first and second (a) and the first and third (b) prinicipal component (data from 165 relevés).

among the groups with respect to substrate moisture. The results for soil reaction (*R*), nutrient supply (*N*) show similar proximity of the medians to each other As it is the case with the *F*-value (not shown in Fig. 1).

Principal component analysis

The first three principal components (PCs) explain 73% of the data variance. Fig. 2(a) explains the first and the second principal component of the PCA scores plots; Fig. 2(b) the first and the third principal component. The meaning of the 3 axes is displayed by the loading plots in Fig. 3(a) and (b). Relevés from the areas A, C and J are grouped together; F and H form another habitat type, neighbouring B and D, E and G.

The first PC separates the relevés mainly according to the prevalence of annual or perennial species. The second and the third PC are separating according to the indicator values *F4* (slightly dry) and *N8* (nutrient rich). Additionally, the third PC displays differences in the presence of annual and biennial species. The indicator values for *R* (soil reaction) are not significantly separating the relevés, indicating that soil reaction does not differ substantially among the different substrates.

Tables 3 and 4 list the most present species of the four different habitat types on landfills. The habitat type ACJ is dominated by ubiquitous grassland species with low indicative potential. It is developing into a ruderal meadow. Ellenberg indicator values of just three species among the 15 most important species could be used for the PCA. The other habitat types mostly are dominated by species with a high indicative potential (indicator values of 11 respectively 13 out of 15 species each were used for the PCA). Habitat type FH is dominated by nitrophilous perennials of moist to wet habitats. The most present species in the habitat types BD and EG are moderately nitrophilous annual species of only moderately moist to dry habitats.

Comparison between the homogeneous areas A and B

Jaccard/Sörensen indices for areas A and B

The tests for both Sörensen- and Jaccard-index gave the same results. Therefore only boxplots for Sörensen-indices are shown in Fig. 4. The differences were tested by ANOVA and Bonferroni's *t*-test as a post hoc-test.

The hypothesis of equality is rejected in the comparison between A × B and A × A respectively B × B, but accepted for the comparison A × A and B × B ($\alpha = 0.05$, $\beta = 0.2$), even when the relevés of the comparisons A × A resp. B × B are taken from different landfill sites (but the same homogeneous area) and the relevés of the comparisons of A × B were recorded on the same landfill site. This indicates that the Sörensen indices within A and within B are comparable.

This result makes clear that the influence of the homogeneous area (with its substrate) on vegetation is bigger than the influence of the landfill site (with its climatic conditions). Significant differences were proven for the diversity (Shannon-index) and the species number per relevé (Fig. 5(a) and (c)), but equality for the evenness (Fig. 5(b)).

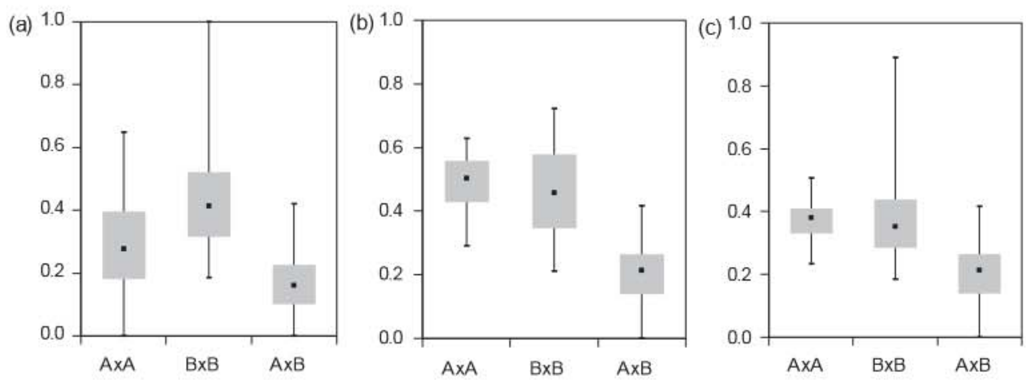

Fig. 4. Boxplots of Sörensen indices within and between the homogeneous areas A and B. (a) For all relevés, (b) for all relevés of landfill site 1 (Table 1) and (c) with the comparisons A × A and B × B from different sites and A × B with relevés from site 1 (Table 1).

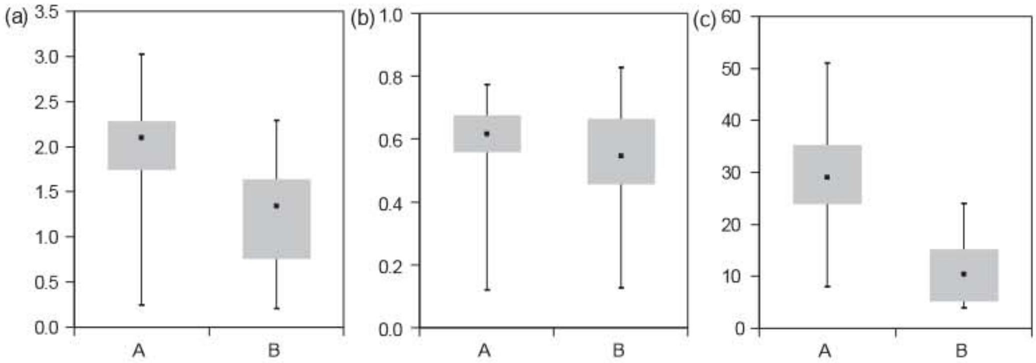

Fig. 5. (a) Shannon-index, (b) evenness, and (c) species richness of the homogeneous areas A and B.

Discussion and conclusion

The results confirm a diagnostic potential of plants for landfill cover properties. On every landfill the vegetation was grouped by habitats with discrete substrate and emission situations. In the presented study this could be based only on the pre-information by the landfill managers. The vegetation on the landfills on the whole reflected a variety of strongly differing habitats. On sandy to loamy material with thick cover layers (Table 2) vegetation develops into ruderal meadows with high successional stability. On old compost material and wet hollows a nitrophyte community of mid-term stability established. Compost on methane oxidation layers and mechanically biologically treated waste both form a habitat type suitable for nitrophytic annual species. Fresh waste and municipal solid waste slag result in a ruderal pioneer stage dominated by weedy species of open soil. Ways of further succession from this stage cannot yet be predicted with certainty.

Most of the species are part of the spontaneous vegetation. This result is of special interest as recultivation very often relies on seed mixtures with obviously little success. Although almost half of the landfills had received a seed mixture of unknown composition (Table 2), the species usually present in commercially available mixtures are hardly found in the relevés. This corresponds to results obtained in various human-disturbed habitats in the Czech Republik. Prach and Pyšek (2001) pointed out that there spontaneous succession often is the cheapest and most successful method for restoration, reflecting a high degree of naturalness.

As these artificial habitats are influenced by many factors and probably by a variety of small-scale disturbances, a definition of plant associations in the sense of phytosociology is not useful. A classification was possible just on the level of plant classes. Not very surprisingly, *Chenopodietea*, *Agropyretea* and *Artemisietea* dominated. Pyšek et al. (2004) found the same classes dominating urban synanthropic vegetation in the city of Plžen. As this approach did not yield highly informative results, we looked for a more distinctive method of grouping relevés. This was realized by using PCA.

The small scale changes in landfill site conditions lead to an unusual relation between constancy and coverage of species. On the one hand some species are constant in habitats different from their usual ones. As an example we found in some relevés *Chenopodium album*, *Elymus repens*, and *Artemisia vulgaris* with the same high coverage mixed together. Forstner (1984) called a similar combination of species "*Chenopodium album*-pioneer stage". On the other hand, some – especially annual – nitrophilous species (for instance *Bassia scoparia*) become dominant in a few favourable microsites. By multiplication of the two parameters constancy and cover, a new parameter could be introduced. We call it "combined presence", a tool to show which species were really pre-destined to function as indicators and to differentiate between substrate conditions. This means that only indicator species with a relatively high combined presence seem to be reliable tools for our purpose to distinguish between different top cover qualities. These species can be divided in grasses common in disturbed grass land, annual nitrophilous species and nitrophilous perennials.

Elymus repens is a common grass with different habitat preferences reproducing both by seeds and rhizomes (Szczepaniak, 2009). It is recorded as one of the dominant species in roadside vegetation in north England (Akbar et al., 2009). In less disturbed habitats its coverage decreases for the benefit of *Festuca* species (Centeri et al., 2009). *Dactylis glomerata* is a very adaptive species with lots of clones. This fact explains the great variability of habitats colonized by the species. It demonstrated remarkable suppressive effects on competing annual species in a field experiment in a southern Australian pasture (Tozer et al., 2009).

The most dominant taxa of the annual nitrophilous species are *Chenopodium album*, *Amaranthus retroflexus* and *Bassia scoparia*. *Chenopodium album* and *Amaranthus retroflexus* are both known as typical species of tuber crop fields and especially *Amaranthus retroflexus* is forced by high amounts of available nitrogen (Ellenberg, 1996). *Amaranthus retroflexus* shows a high affinity to compost amended habitats (Table 4). Amisi and Doohan (2010) recorded that growth and especially seed production of this species was significantly increased, when soil was enriched with compost, whereas seedling emergence was reduced. The reduced emergence leads to less competitiveness of amaranth and fosters ruderal strategists. A further effect of interest could be the capability of *Amaranthus retroflexus* regarding the accumulation of metals. This property was tested by Chehregani et al. (2009) in an Iranian lead and zinc mine. Heavy metals probably do not play an important role in landfill habitats as the contents in municipal solid waste and its biodegraded landfill products are low compared with mining sites (Bilitewski et al., 1997) and below the toxicity limits for the species (Geiger et al., 1993) – but definitely much higher than in uncontaminated soils. They possibly influence species composition. As in this study no chemical data were obtained, we cannot prove this for our sites. At least *Amaranthus retroflexus* exerts allelopathic effects on other species (Liu and Ma, 2009). All these factors could explain the high dominance of this species.

Also *Chenopodium album* is a typical ruderal species (Grime, 2001) and a dominant weed (Gupta et al., 2008) with enormous high seed availability that takes advantage of the plenty of nutrients. It seems to be especially fostered by the strongly disturbed landfill habitats. Furthermore *Chenopodium album* avoids shade, but has high variation between local populations (Haraguchi et al., 2009; Nishimura et al., 2010). An important trait of the species on leachate-affected habitats is its adaptability regarding salt concentration, as reported by Yao et al. (2010).

Bassia scoparia is a species of ruderal habitats with a certain tolerance of salt (Ajmal Khan et al., 2009; Salehi et al., 2009), also proven to be tolerant when irrigated with leachate of municipal solid waste landfills (Erdogan et al., 2008). The optimum temperature for *Bassia scoparia* germination is relatively high (Jami Al-Ahmadi and Kafi, 2007; Khan et al., 2001).

Among the nitrophilous perennials *Urtica dioica* is the dominant species. It is nutrient demanding, and common on compost amended ruderal sites (Taylor, 2009). Some authors also refer on its tolerance of moderate heavy metal contents in the substrate (Shams et al., 2010; Spongberg et al., 2008).

Usually, the calculation of Ellenberg's mean indicator values comprises all species of every relevé. The ubiquitous species, however, with their indicator values from 5 to 7, are useless for indicating substrate and habitat properties. They overlay the evaluation of the data with a variance which cannot be interpreted. In this case, the first principal component would explain only 36% of variance instead of 51% if these intermediate Ellenberg values are omitted.

It became obvious that for areas which the landfill managers had covered with loamy substrate and for which different additional information was available on similar morphology, history, etc., also the vegetation type was comparable. Wet hollows and areas with mature compost as substrate grouped one near to the other in the PCA. Methane oxidation layers with fresh compost result in similar vegetation types to fresh mechanically biologically treated waste. Slag from municipal solid waste incineration leads to an extreme substrate similar to leachate-influenced areas with fresh waste or sewage sludge. Even long time after construction these substrate conditions are mirrored by vegetation.

The most present species in the group of the loamy substrate are ubiquitous species with low indicative potential, whereas the cover features of the other groups can be described by vegetation very well.

We showed that relevés from different landfills, but belonging to the same habitat type, correspond better to each other with respect of α- and β-diversity and species number than relevés from one landfill, but belonging to different habitat types. Mean species richness of the homogeneous area A (median of 29 species per relevé) and also life traits of the species there correspond to results obtained from comparable habitats on a landfill in Berlin (Rebele and Lehmann, 2002). The homogeneous area B with highly disturbed site conditions contains significantly less species (median of 11 species per relevé). Three of the four landfill habitat types distinguished in this paper could not be revegetated by seed mixtures typical for gardens or street shoulders. As far as seed mixtures had been applied on the sites (see Table 2) the seeded species either did not occur at all or did not grow well during our research. Habitat conditions of these typical landfill habitats are indicated well by the most present species.

It is clear that this more general aptitude test has to be complemented by further studies especially on the exact response of the vegetation to methane emissions, salt and heavy metal concentrations, allelopathic effects, and cover layer thickness. Nevertheless, a good indication of substrate quality and the emission situation seems to be possible by studies of flora and vegetation cover there. Thus, the time and cost demanding assessment by physical or chemical parameters could be simplified by the vegetation ecology approach. Specific site conditions regarding climate and surrounding vegetation influence the specific species composition but not the indication quality. Certainly, the specific situation of each site has to be taken into account, when restoration focuses on specific floral elements and Red List species (Kirmer et al., 2008). However, the question of the paper, whether vegetation can indicate landfill cover features distinctly can be answered with "yes"!

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References

Abdi, H., 2007. The Bonferroni and Šidák corrections for multiple comparisons. In: Salkind, N. (Ed.), Encyclopedia of Measurement and Statistics. University of Texas, Richardson.

Ajmal Khan, M., Gul, B., Weber, D.J., 2009. Seed germination of *Kochia scoparia* under saline conditions: Responses with germination regulating chemicals. Pakist. J. Bot. 41, 2933–2941.

Akbar, K.F., Hale, W.H.G., Headley, A.D.D., 2009. Floristic composition and environmental determinants of roadside vegetation in north England. Polish J. Ecol. 57, 73–88.

Amisi, K.J., Doohan, D., 2010. Redroot pigweed (*Amaranthus retroflexus*) seedling emergence and growth in soils amended with composted dairy cattle manure and fresh dairy cattle manure under greenhouse conditions. Weed Technol. 24, 71–75.

Bilitewski, B., Härdtle, G., Marek, K., 1997. Waste Management. Springer, Berlin.

Centeri, C., Herczeg, E., Vona, M., Balázs, K., Penksza, K., 2009. The effects of land-use change on plant-soil-erosion relations, Nyereg Hill, Hungary. J. Plant Nutr. Soil Sci. 172, 586–592.

Chambers, J.C., 1983. Measuring species diversity on revegetated surface mines: an evaluation of techniques. USDA, Forest Service, Research Paper.

Chehregani, A., Noori, M., Yazdi, H.L., 2009. Phytoremediation of heavy-metal-polluted soils: screening for new accumulator plants in Angouran mine (Iran) and evaluation of removal ability. Ecotoxicol. Environ. Safety 72, 1349–1353.

Diekmann, M., 2003. Species indicator values as an important tool in applied plant ecology – a review. Basic Appl. Ecol. 4, 493–506.

Ellenberg, H., 1996. Vegetation Mitteleuropas mit den Alpen. Ulmer, Stuttgart.

Ellenberg, H., Weber, H.E., Düll, R., Wirth, V., Werner, W., Paulißen, D., 1992. Zeigerwerte von Pflanzen in Mitteleuropa. Goeltze, Göttingen.

Erdogan, R., Zaimoglu, Z., Sucu, M.Y., Budak, F., Kekec, S., 2008. Applicability of leachates originating from solid-waste landfills for irrigation in landfill restoration projects. J. Environ. Biol. 29, 779–784.

Felföldy, L., 1942. Szociológiai vizsgálatok a pannóniai flóraterület gyomvegetációján. Acta Geobot. Hung. 5, 87–140.

Fischer, A., 1985. "Ruderale Wiesen" – ein Beitrag zur Kenntnis des Arrhenaterion-Verbandes. Tuexenia 5, 237–248.

Fischer, M.A., Adler, W., Oswald, K., 2005. Exkursionsflora für Österreich, Liechtenstein und Südtirol. Biologiezentrum der OÖ Landesmuseen, Linz.

Forstner, W., 1982. Ruderale Vegetation in Ost-Österreich. Teil 1. Wiss. Mitt. Niederösterr. Landesmus. 2, 19–133.

Forstner, W., 1984. Ruderale Vegetation in Ost-Österreich. Teil 2. Wiss. Mitt. Niederösterr. Landesmus. 3, 11–91.

Geiger, G., Federer, P., Sticher, H., 1993. Reclamation of heavy metal-contaminated soils: field studies and germination experiments. J. Environ. Qual. 22, 201–207.

Görs, S., 1966. Die Pflanzengesellschaften der Rebhänge am Spitzberg. Nat. Landschaftsschutzgeb. Baden-Württ. 3, 476–533.

Görs, S., 1968. Der Wandel der Vegetation im Naturschutzgebiet Schwenninger Moos unter dem Einfluß des Menschen in zwei Jahrhunderten. Nat. Landschaftsschutzgeb. Baden-Württ. 5, 191–284.

Grime, J.P., 2001. Plant Strategies, Vegetation Processes, and Ecosystem Properties. Wiley, Chichester.

Gupta, A., Joshi, S.P., Manhas, R.K., 2008. Multivariate analysis of diversity and composition of weed communities of wheat fields in Doon Valley, India. Trop. Ecol. 49, 103–112.

Haraguchi, A., Li, B., Matsuki, S., Nagata, O., Suzuki, J.-I., Hara, T., 2009. Variation and plasticity of photosynthesis and respiration in local populations of fat-hen *Chenopodium album* in northern Japan. Plant Species Biol. 24, 189–201.

Häupler, H., 1982. Evenness als Ausdruck der Vielfalt in der Vegetation. Cramer, Vaduz.

Hübl, E., Holzner, W., 1974. Vorläufiger Überblick über die Ruderalvegetation von Wien. Acta Inst. Bot. Acad. Sci. Slov., Ser. A 1, 233–238.

Jaccard, P., 1912. The distribution of the flora in the alpine zone. New Phytol. 11, 37–50.

Jami Al-Ahmadi, M., Kafi, M., 2007. Cardinal temperatures for germination of *Kochia scoparia* (L.). J. Arid Environ. 68, 308–314.

Karrer, G., Kilian, W., 1990. Standorte und Waldgesellschaften im Leithagebirge Revier Sommerein. Mitt. Forstl. Bundesversuchsanstalt Wien 165, 1–244.

Khan, M.A., Gul, B., Weber, D.J., 2001. Influence of salinity and temperature on the germination of *Kochia scoparia*. Wetlands Ecol. Manage. 9, 483–489.

Kirmer, A., Tischew, S., Ozinga, W.A., Lampe, M.v., Baasch, A., Groenendael, J.M.v., 2008. Importance of regional species pools and functional traits in colonization processes: predicting re-colonization after large-scale destruction of ecosystems. J. Appl. Ecol. 45, 1523–1530.

Koch, W., 1926. Die Vegetationseinheiten der Linthebene. Jahrb. St. Gallisch. Naturwiss. Ges. 61, 1–146.

Konold, W., Zeltner, G.H., 1981. Untersuchungen zur Vegetation abgedeckter Mülldeponien. Landesanst. für Umweltschutz Baden-Württemberg. Inst. für Ökologie u. Naturschutz.

Liu, S., Ma, D.W., 2009. The allelopathy of different development stages of *Amaranthus retroflexus* L. on root border cells of cucumber. Shengtai Xuebao/Acta Ecol. Sinica 29, 4392–4396.
Londo, G., 1976. The decimal scale for relevés of permanent quadrats. Vegetatio 33, 61–64.
Mucina, L., 1993. Die Ruderalvegetation von Westslowakei. Inst. of Plant Physiology, Vienna.
Mucina, L., Grabherr, G., Ellmauer, T., 1993. Die Pflanzengesellschaften Österreichs, Band 1 – Anthropogene Vegetation. G. Fischer, Jena.
Nishimura, E., Suzaki, E., Irie, M., Nagashima, H., Hirose, T., 2010. Architecture and growth of an annual plant *Chenopodium album* in different light climates. Ecol. Res. 25, 383–393.
Oberdorfer, E., 1983. Süddeutsche Pflanzengesellschaften. VEB Fischer, Jena.
Passarge, H., 1978. Übersicht über mitteleuropäische Gefässpflanzengesellschaften. Feddes Repert. 89, 133–195.
Prach, K., Pyšek, P., 2001. Using spontaneous succession for restoration of human-disturbed habitats: experience from Central Europe. Ecol. Eng. 17, 55–62.
Pyšek, P., Chocholoušková, Z., Pyšek, A., Jarošík, V., Chytrý, M., Tichý, L., 2004. Trends in species diversity and composition of urban vegetation over three decades. J. Veget. Sci. 15, 781–788.
Rasch, D., Kubinger, K.D., Moder, K., 2009. The two-sample *t* test: pre-testing its assumptions does not pay off. Stat. Pap., 1–13.
Rebele, F., Lehmann, C., 2002. Restoration of a landfill in Berlin, Germany, by spontaneous and directed succession. Restor. Ecol. 10, 340–347.
Salehi, M., Kafi, M., Kiani, A., 2009. Growth analysis of *Kochia* (*Kochia scoparia* (L.) Schrad.) irrigated with saline water in summer cropping. Pakist. J. Bot. 41, 1861–1870.
Schubert, R., Hilbig, W., Klotz, S., 2001. Bestimmungsbuch der Pflanzengesellschaften Deutschlands. Spektrum, Heidelberg.
Shams, K.M., Tichy, G., Fischer, A., Sager, M., Peer, T., Bashar, A., Filip, K., 2010. Aspects of phytoremediation for chromium contaminated sites using common plants *Urtica dioica*, *Brassica napus* and *Zea mays*. Plant Soil 328, 175–189.
Smidt, E., Tintner, J., Meissl, K., 2007. New approaches of landfill assessment and monitoring. In: Velinni, A.A. (Ed.), Landfill Research Trends. Nova Science Publications, Inc., New York, pp. 197–226.
Sørensen, T., 1948. A method of establishing groups of equal amplitude in plant sociology based on similarity of species and its application to analyses of the vegetation on Danish commons. Biologiske Skrifter/Kongelige Danske Videnskabernes Selskab 5, 1–34.
Spongberg, A.L., Hartley, L., Neher, D.A., Witter, J., 2008. Fate of heavy metal contaminants in a former sewage treatment lagoon, Hancock County, Ohio. Soil Sediment Contam. 17, 619–629.
Szczepaniak, M., 2009. Biosystematic studies of *Elymus repens* (L.) gould (Poaceae): patterns of phenotypic variation. Acta Soc. Botan. Polon. 78, 51–61.
Taylor, K., 2009. Biological flora of the British Isles: *Urtica dioica* L. J. Ecol. 97, 1436–1458.
Tozer, K.N., Chapman, D.F., Cousens, R.D., Quigley, P.E., Dowling, P.M., Kearney, G.A., Cameron, C.A., 2009. Effects of perennial species on the demography of annual grass weeds in pastures subject to seasonal drought and grazing. Crop Pasture Sci. 60, 1088–1096.
Tremp, H., 2005. Aufnahme und Analyse vegetationsökologischer Daten. Ulmer, Stuttgart.
Tüxen, R., 1947. Der Pflanzensoziologische Garten in Hannover und seine bisherige Entwicklung. Jahresber. Naturhist. Ges. Hannover, 1942/43–1946/47, 113–287.
Tüxen, R., 1950. Grundriß einer Systematik der nitrophilen Unkrautgesellschaften in der Eurosibirischen Region Europas. Mitt. Florist. Soziol. Arbeitsgem. N. F. 2, 94–175.
Yao, S., Chen, S., Xu, D., Lan, H., 2010. Plant growth and responses of antioxidants of *Chenopodium album* to long-term NaCl and KCl stress. Plant Growth Regul. 60, 115–125.

3 Risk assessment of an old landfill regarding the potential of gaseous emissions – A case study based on bioindication, FT-IR spectroscopy and thermal analysis

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Risk assessment of an old landfill regarding the potential of gaseous emissions – A case study based on bioindication, FT-IR spectroscopy and thermal analysis

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ABSTRACT

Risk assessment of two sections (I and II) of an old landfill (ALH) in Styria (Austria) in terms of reactivity of waste organic matter and the related potential of gaseous emissions was performed using conventional parameters and innovative tools to verify their effectiveness in practice. The ecological survey of the established vegetation at the landfill surface (plant sociological relevés) indicated no relevant emissions over a longer period of time. Statistical evaluation of conventional parameters reveals that dissolved organic carbon (DOC), respiration activity (RA_4), loss of ignition (LOI) and total inorganic carbon (TIC) mostly influence the variability of the gas generation sum (GS_{21}). According to Fourier Transform Infrared (FT-IR) spectral data and the results of the classification model the reactivity potential of the investigated sections is very low which is in accordance with the results of plant sociological relevés and biological tests. The interpretation of specific regions in the FT-IR spectra was changed and adapted to material characteristics. Contrary to mechanically–biologically treated (MBT) materials, where strong aliphatic methylene bands indicate reactivity, they are rather assigned to the C—H vibrations of plastics in old landfill materials. This assumption was confirmed by thermal analysis and the characteristic heat flow profile of plastics containing landfill samples. Therefore organic carbon contents are relatively high compared to other stable landfills as shown by a prediction model for TOC contents based on heat flow profiles and partial least squares regression (PLS-R). The stability of the landfill samples, expressed by the relation of CO_2 release and enthalpies, was compared to unreactive landfills, archeological samples, earthlike materials and hardly degradable organic matter. Due to the material composition and the aging process the landfill samples are located between hardly degradable, but easily combustible materials and thermally resistant materials with acquired stability.

1. Introduction

Due to the increased awareness of hazards originating from old landfills and the implementation of stricter regulations including the treatment before final disposal, investigations ex post are often necessary to confirm the harmlessness regarding the environmental risk. Over the last decade several systematic investigations were carried out in some Austrian regions (Holubar et al., 2004) to evaluate old disposal sites where municipal solid waste was landfilled in the sixties and seventies. Two main properties of landfilled waste are in the focus of interest: its toxicity and the reactivity. The latter has attracted more attention due to the contribution of landfill gas emissions to relevant climate effects (Bogner et al., 1995; Tesar et al., 2007; Manfredi et al., 2009; Tintner et al., 2012). Toxicity of municipal solid waste plays a minor role in old landfills without baseliners due to leaching effects (Holubar et al., 2004). Comprehensive investigations are necessary to identify and quantify the impact on the environment. Contaminated leachate of reactor landfills with a collection system requires a long lasting monitoring (Scharff et al., 2011). The release of hazardous substances is mainly influenced by physical and chemical properties of the landfill matrix and dominated by sorption and desorption processes.

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Due to the high costs of a comprehensive investigation program in practice only a small number of samples is in general collected. The remaining reactivity of old landfills is estimated on the basis of the emitted gaseous compounds at the landfill surface. Due to highly intermittent emissions over time and space gas measurements cannot properly assess the actual emissions. Desiccated sections and preferential flows do not reflect the potential situation and can lead to misinterpretation. More information is obtained by plant ecological investigations (plant sociological relevés). The procedure is described below. Vegetation integrates the emission variability over time and provides spatial information with high resolution. The number and composition of plant species and the related indicator values allow conclusions to be drawn about the type and the intensity of emissions (Tintner and Klug, 2006, 2011).

The emission potential and the related risk for the future can only be derived from the solid material and the quality of organic matter. Detailed information on waste organic matter, short time of analyses, easy handling, reliability and low costs in favor of a sufficient number of samples are objectives of waste analytics. Therefore innovative analytical methods that comply with this purpose have been developed during the last decade. Fourier Transform Infrared (FT-IR) spectroscopy and thermal analysis characterize the material by the unique pattern (Provenzano et al., 1998; Dell'Abate et al., 2000; Provenzano et al., 2000; Smidt et al., 2002; Smidt and Lechner, 2005; Smidt and Schwanninger, 2005; Smidt and Meissl, 2007). Their potential for the evaluation of mechanical–biological treatment (MBT) has been proven by several studies (van Praagh et al., 2009; Böhm et al., 2010; Smidt et al., 2010). The combination with multivariate statistical methods allows a substantial data analysis under diverse perspectives.

The objective of this study was to determine the gas emission potential within the scope of the required risk assessment of an old landfill (abbreviated: ALH) in Styria. The decision on additional measures, e.g. *in situ* aeration should be based on the results obtained. Plant sociological relevés (vegetation ecology) before sampling represent a new approach to assess the actual situation by bioindication. FT-IR spectroscopy and thermal analysis were used besides conventional parameters for the characterization of solid waste matter. Whereas the sociological relevé reveals the status quo as a result of landfill conditions over a longer period of time, FT-IR spectral and thermal analyses provide information on the reactivity potential. The proof of the effectiveness in practice and applicability for landfill risk assessment was an important step to promote the development of the applied tool set.

2. Materials and methods

2.1. Sample origin

Landfill ALH samples originated from the two oldest sections (I and II) of a landfill in Styria (Austria), with an area of 4.5 hectares and a depth of 3–5 m. According to the information by the landfill operator municipal solid waste, bulky waste and residues from biologically treated waste were deposited in the sixties and seventies in these sections. Sampling was carried out according to statistical principles. The number of required samples was determined to be 75 ($\alpha = 0.05$ and $\beta = 0.2$). The calculation was based on the variability of the parameter respiration activity with the standard deviation of 1.68 mg O_2 g^{-1} DM (dry matter). Sampling was carried out by digging. The necessary material amount for any sampling point in the grid was calculated according to the following equation: sample (kg) = 0.06 × particle size (mm), resulting in 20 kg (CEN/TR 15310-1, 2006). The particle size was determined by visual inspection of the material. From the heap a mixed sample was collected according to the splitting procedure. Apart from big objects (e.g. chassis of old phones) the whole sample was collected and shredded to 20 mm.

The classification models (Esbensen, 2002) comprised 110 landfill samples (class "Landfill" with low and very low gas forming potential) and 190 MBT samples (class "MBT" comprising all stages of degradation and therefore different reactivity). The sample set originated from previous investigations. Six samples of high reactivity originating from a reactor landfill (reactive samples in Fig. 5b) were classified besides 75 ALH samples. For the TOC prediction model 125 landfill samples with low, middle and high gas forming potential were used (sample set of 110 samples mentioned above +15 reactive landfill samples from previous investigations) in addition to 75 ALH samples. The sample pool for the stability assessment comprised the 110 landfill samples with low and very low gas forming potential, 36 stable organic materials (lignite), samples (e.g. charcoal, organic waste with mineral components) from archeological sites and 75 ALH samples.

2.2. Sample preparation

For respiration activity (RA_4), gas sum (GS_{21}) and the production of eluates the fresh material was used. The determination of several parameters such as biochemical oxygen demand (BOD), NH_4—N, pH, sulfate and chloride was conducted on the eluate. For other analyses the samples were dried at 105 °C, ground and screened <0.5 mm. For spectral and thermal analyses the material was additionally homogenized by pestle and mortar.

2.3. Plant sociological relevé

Twenty-two plant sociological relevés were carried out prior to digging according to Tintner and Klug (2006) based on indicator values created by Ellenberg et al. (1992). These investigations were extended to the whole landfill area (sections I–IV) to reveal the differences between previously landfilled materials and an obviously disturbed section (III), where MBT material is currently landfilled and thereby validate the method. Relevés are carried out at homogenous areas with a representative vegetation cover. Their size depends on the vegetation type. More than 25 m^2 are necessary for segetal and ruderal associations (Westhoff and van der Maarel, 1973). For data evaluation a combination of the scales published by Braun-Blanquet (1964) and Londo (1976) were used. The plant species were listed and the indicator values of each species were weighted by its coverage within the relevé area. Indifferent species or species with unknown indicator values were excluded from the calculation.

The processing of the indicator values was described by Tintner and Klug (2011). Data were integrated in a scheme of vegetation ecological investigations (plant sociological relevé) at different landfill sites, established by Tintner and Klug (2011), based on a principal component analysis (PCA). Data of previous investigations that were the basis for the classification (Fig. 1) are indicated by gray symbols, each representing a specific group of plants and substrate conditions. They are assigned to the four corners in the PCA, but overlapping occurs.

Chemical and physical properties of the substrate are revealed by the development of a characteristic composition and distribution of plant species that are indicators for the existing environmental conditions over a longer period of time. Contrary to chemical and physical parameters that show the current stage of the landfill at the moment of sampling, the relevé integrates the emission status at least over the life time of the species. Table 1 compiles characteristics of the substrates, the corresponding indicator plants and the conditions they indicate.

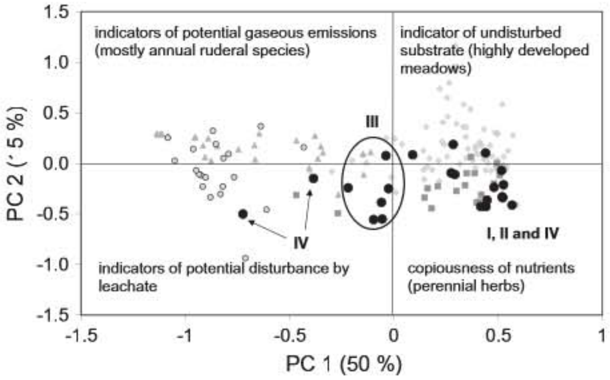

Fig. 1. Grouping of 22 relevés (black dots) in four landfill sections (I–IV) based on a PCA according to the indicator scheme established by Tintner and Klug (2011); four groups of indicators represented by gray symbols.

2.4. Chemical, physical and biological parameters

Loss of ignition (LOI), total nitrogen (TN), total organic carbon (TOC), pH value and NH_4—N were determined according to the Austrian Compost Ordinance (BMLFUW, 2001). TN, total (TC) and inorganic (TIC) carbon were determined by combustion in the CNS (carbon–nitrogen–sulfur) analyzer (VarioMax) according to Austrian Standard Methods OENORM S 2023 (Austrian Standards Institute, 1986). The TOC was calculated (TC – TIC = TOC). Dissolved organic carbon (DOC) was measured according to DIN EN 1484 (European Committee for Standardization, 1997), the biochemical oxygen demand during a 5-day incubation period (BOD_5) according to DIN EN 1899-1 (European Committee for Standardization, 1998). Respiration activity (RA_4) was determined during a 4-day incubation period in a Sapromat (Voith Sulzer) by recording the oxygen uptake in milligram per gram dry matter (mg O_2 g^{-1} DM), the gas generation sum (GS_{21}) during a 21-day incubation period under anaerobic conditions by recording the generated gas volume per kilogram dry matter (NL kg^{-1} DM) according to OENORM S 2027 1-2 (Austrian Standards Institute, 2004a, 2004b). Sulfate and chloride were quantified by ion-chromatographic measurements (Dionex DX-120 with automated sampler Dionex AS 40).

2.5. FT-IR spectroscopy and thermal analyses

FT-IR spectra were recorded in the attenuated total reflection (ATR) mode by a Bruker alpha® instrument in the wavenumber range from 4000 to 400 cm^{-1}. The homogenized sample was directly applied on the ATR reflection module containing a diamond crystal with a measuring area of approximately 4 mm^2 and a pressure applicator. Twenty-four scans per spectrum were collected at a resolution of 4 cm^{-1} and a zero-filling factor of 2 and corrected against ambient air as background. The average of four spectra (maximum deviation of the four spectra from the average spectrum <5%) was vector normalized for multivariate data analysis. Simultaneous thermal analyses were carried out with a STA 409 CD Skimmer (Netzsch GmbH) in an Al_2O_3 pan with the following combustion parameters: temperature range 30–950 °C, heating rate 10 K min^{-1}, gas flow 150 ml min^{-1} (80% He and 20% O_2), sample amount 16.00 mg. Simultaneous thermal analysis comprises thermogravimetry (TG) and differential scanning calorimetry (DSC) and the analysis of combustion gases with the coupled mass spectrometer. TG and DSC were applied to reveal the mass loss and the heat flow profile respectively of the whole sample. Combustion gases were recorded in the mass spectrometer. Data evaluation focused on mass 44 (ion current of CO_2). The enthalpy was calculated by integration of the area below the DSC profile and a horizontal base line between 30 °C and 600 °C, starting at 30 °C. The CO_2 release was calculated by integration of the area below the CO_2 profile and a horizontal base line within the same temperature range. The calculation results in a measure of the relative intensity.

2.6. Sampling design and data evaluation

Sampling design and data evaluation of conventional parameters (ANOVA) were performed using the software SPSS 15.0. For multivariate data analysis the Unscrambler (CAMO 9.2®) was applied. Multivariate data evaluation focused on principal component analysis (PCA), classification by means of soft independent modeling of class analogy (SIMCA) and partial least squares regression (PLS-R) as described by Esbensen (2002) and Brereton (2002). SIMCA is a procedure for classification based on PCA class modeling. "Soft modeling" that is often used in chemical pattern recognition means that two classes can overlap. Thus it is possible that samples have characteristics of both defined classes, or of neither of the defined classes. By means of PLSR the variations of *y*-variables are related to the variations of *x*-variables with the purpose to predict the *y*-variable (e.g. TOC) by *x*-variables (e.g. heat flow data).

3. Results and discussion

Plant sociological relevés of the landfill ALH (black dots – Fig. 1) indicated that no relevant emissions had taken place in sections I, II and IV during several years apart from two exceptions (relevés in section IV marked by arrows). Relevant in this context means that hazardous emissions can reduce plant growth. In section I and II perennial grasses like *Festuca rubra* and *Lolium multiflorum* were dominant. These species are capable of producing high amounts of biomass but are rather not adapted for disturbances (negative

Table 1
Vegetation information on landfills.

Group of vegetation associations	Typical plants and their properties	Substrate characteristics	Indication
Highly developed meadows	Perennial grass species, no loss of vegetation	Moderate humidity to weak dryness, moderate nutrient content, undisturbed	Neither leachate nor gas emissions are expected
Perennial herbs	Perennial nutrient, especially nitrophilous herbs	Old compost and mature MBT deposits with thin cover layers	No gaseous emissions, leachate with low salt concentration is possible
Indicators of potential gaseous emissions (mostly ruderals)	Distinct change in the composition of species, dominance of annual and dryness preferring species; physiological damages, reduced life span	Reactive organic matter	Moderate to strong gaseous emissions, mechanical disturbances and recent deposits
Indicators of potential leachate emissions	Dominance of salt preferring species, subtly differentiated spectrum of species, not competitive without saliferous substrates	Relatively high salt concentration	Salt concentration related to leachate emissions

Table 2
Data of conventional and biological analyses including the mean value, the upper and the lower limit of the confidence interval (α = 5%, n = 75).

Parameter	Dimension	Mean	Confidence interval of the mean values	
			Lower limit	Upper limit
pH-value		8.1	8.0	8.1
Electrical conductivity	mS cm^{-1}	1.2	1.1	1.3
NH_4–N	mg kg^{-1} DM	456	417	494
SO_4–S	mg kg^{-1} DM	133	91	174
Cl^-	mg kg^{-1} DM	837	718	954
Total nitrogen (TN)	% DM	0.94	0.89	0.99
Loss of ignition (LOI)	% DM	31.4	30.1	32.7
Total organic carbon (TOC)	% DM	17.0	16.2	17.7
Total inorganic carbon (TIC)	% DM	1.0	0.9	1.1
C/LOI		0.54	0.53	0.55
Respiration activity 4 days (RA_4)	mg O_2 g^{-1} DM	5.3	4.9	5.7
Gas sum 21 days (GS_{21})	NL kg^{-1} DM	5.0	4.1	5.9
Biological oxygen demand (BSB_5)	mg O_2 kg^{-1} DM	69	54	85
Dissolved organic carbon (DOC)	mg C kg^{-1} DM	942	846	1038

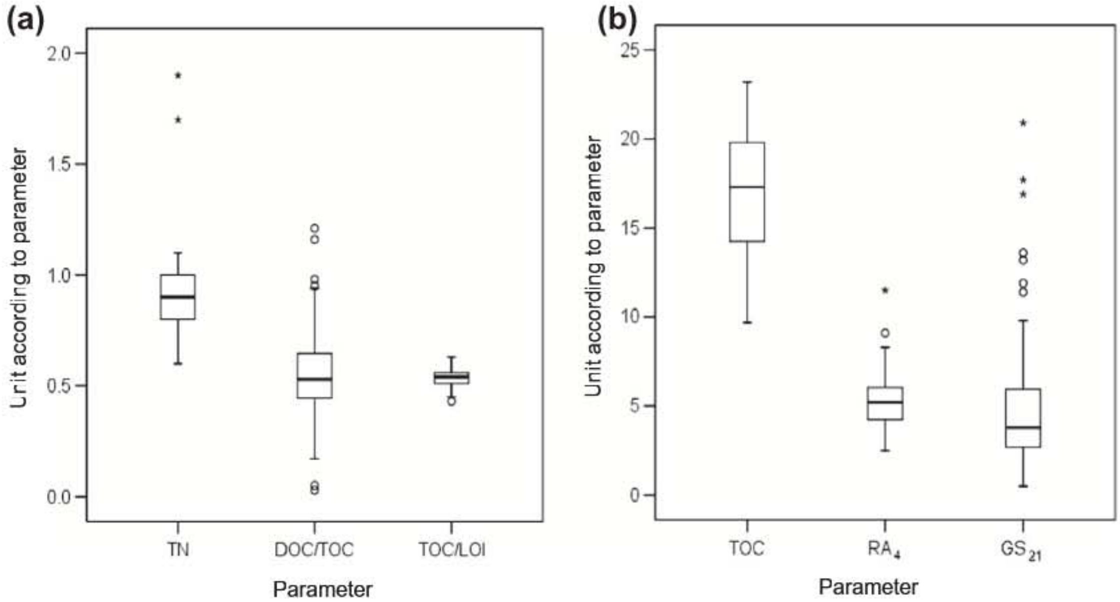

Fig. 2. Boxplots indicating median, minimum, maximum, 25% and 75% quartile of parameters (unit according to parameter): (a) TN (% DM), DOC (% of TOC), TOC/LOI ratio, (b) TOC (% DM), RA_4 (mg O_2 g^{-1}DM), GS_{21} (NL kg^{-1} DM); circles = data within the range of 1.5 quartiles, asterisks = data beyond the distance.

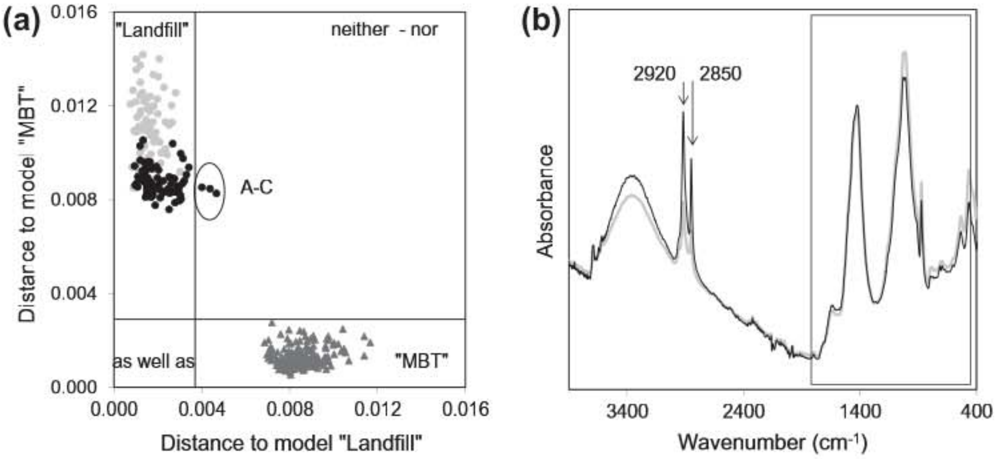

Fig. 3. (a) Classification of landfill ALH samples (black dots, A–C) in the SIMCA model based on FT-IR spectra (wavenumber 4000–400 cm^{-1}) visualized by the Coomans plot and (b) average spectrum of the three outlying landfill samples (black line) and average spectrum of the landfill class (gray line).

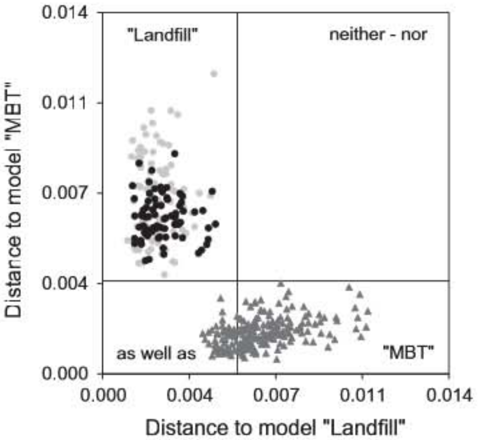

Fig. 4. Classification of landfill ALH samples (black dots) in the SIMCA model based on FT-IR spectra (wavenumber 1850–400 cm^{-1}) visualized by the Coomans plot.

environmental impact). They need optimal conditions and are typical for intensive grassland. Section IV is dominated by nitrophilous herbs and grasses like *Urtica dioica*, *Phalaris arundinacea*, *Elymus repens*, *Calamagrostis epigejos* and *Solidago gigantea*. The only shrub species is *Salix* x *rubens*, indicating the typical habitats of all these species originating from rarely disturbed wetlands. Such conditions arise from the close-grained cover layer and the cold climate resulting in relatively low transpiration rates. The two outlying relevés in section IV were carried out at the border of the landfill where contamination by leachate is possible. The dominant species are highly ruderal, annual genera *Chenopodium* and *Atriplex*. Section III that is still in operation reflects the expected disturbances. As shown in Fig. 1 data of sections I, II and IV are found in the right corners of the PCA, most of them in the class that indicates a nutrient-rich substrate.

3.1. Conventional parameters and biological tests

The mean values of all samples and the limits of the confidence interval that depend on the variability of data are indicated in Table 2 (required probability $\alpha = 5\%$, number of samples $n = 75$).

Fig. 2 displays the relevant parameters related to waste organic matter as boxplots with median, minimum, maximum, 25% and 75% quartile. Circles characterize data within the range of 1.5 quartiles, asterisks indicate data beyond this distance.

It is evident that the TOC does not comply with the required limit value of 5% DM of the Austrian Landfill Ordinance (BMLFUW, 2008). Based on the special regulation for MBT materials the assessment of reactivity was therefore mainly supported by biological parameters RA_4 and GS_{21}. Results of RA_4 and GS_{21} feature very low reactivity.

Additionally data were evaluated by a multiple linear regression. The influence of all parameters in Table 2 on the GS_{21} as the relevant parameter to assess future emissions was calculated. For the variable selection a stepwise method was used. Based on this statistical evaluation the DOC, RA_4, LOI and TIC were identified as influencing variables. The coefficient of determination ($R^2 = 0.60$) of the resulting model indicates that almost 60% of the variability of GS_{21} can be explained by these four variables. The fact that the DOC is higher ranked than the TOC is plausible and reflects the limited significance of the TOC with regard to reactivity.

3.2. Determination of reactivity by assignment of landfill spectra to classification models

FT-IR spectroscopic and thermal analyses were included to clarify the discrepancy between the relatively high TOC content and the low reactivity and to unambiguously reveal the characteristics of reactive and non-reactive landfill materials. Due to the correlation between reactivity and the band intensity of FT-IR spectra, prediction models were developed for MBT materials based on a Partial Least Squares Regression (PLS-R) (Böhm et al., 2010).

For landfill materials with a divergent composition the relation between the biological behavior and the chemical information of the spectrum becomes weaker in the area of lower reactivity and is not satisfying. Therefore a different approach of data evaluation focused on classification of the 75 landfill ALH spectra by means of a SIMCA-model that distinguishes two groups of waste materials: unreactive landfills and MBT materials of different degradation stages after the mechanical treatment. "Unreactive" in this context means compliance with the limit values of the Austrian Landfill Ordinance (BMLFUW, 2008) regarding the reactivity parameters RA_4 and GS_{21}. The model was based on the whole wavenumber range from 4000 to 400 cm^{-1}. Fig. 3a demonstrates the model and the classified landfill ALH spectra (black dots), visualized by the Coomans plot. Apart from three samples (A–C) the spectra

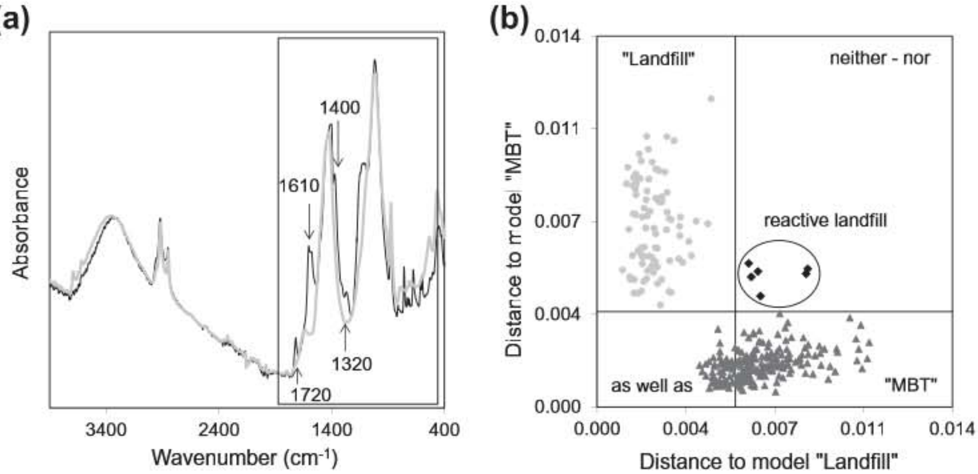

Fig. 5. (a) Average spectrum of reactive landfill samples (black line) and average spectrum of the landfill class (gray line) and (b) classified reactive landfill samples in the corner "neither-nor".

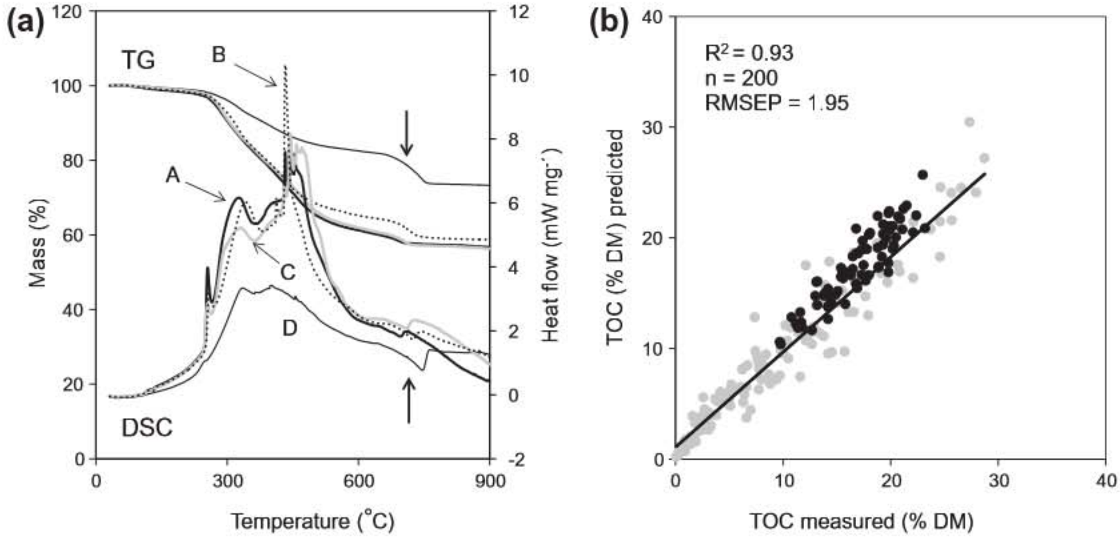

Fig. 6. (a) TG- and DSC profiles of four selected ALH samples (A–D), including the three outlying samples (A–C) in the classification model (Fig. 3a) and (b) correlation of the measured and the predicted TOC of 75 ALH (black dots), stable and reactive landfill samples (125), based on DSC profiles, TOC analysis and PLS-R (RMSEP = root mean square error of prediction).

were assigned to the corner of old unreactive landfills. In order to identify the characteristics of these three outlying samples, their average spectrum was compared to the average spectrum of the landfill samples in the classification model (Fig. 3b). The comparison reveals that the investigated landfill samples feature very strong aliphatic methylene bands at 2929 and 2850 cm^{-1}. These bands are very useful to assess the reactivity of MBT materials or biogenic waste and play a crucial role in reactivity prediction models as shown in previous studies (Böhm et al., 2010; Meissl et al., 2007). Depending on the stage of degradation they reflect adequately the breakdown of biomolecules by decreasing band intensities (Smidt and Meissl, 2007; Smidt and Schwanninger, 2005). However, strong aliphatic bands of landfill samples can indicate the C—H vibration of plastics which leads to misinterpretation of reactivity. Apart from the difference due to the aliphatic methylene bands spectra of the "Landfill" class and the spectra A–C are very similar.

Thus a second classification model was based on the wavenumber range 1850–400 cm^{-1} (framed wavenumber region in Fig. 3b). If the aliphatic methylene bands are excluded, all landfill ALH samples are classified in the "Landfill" corner (Fig. 4). The high portion of plastics in the three samples was verified by registration of the ingredients. Due to the missing methylene bands and their discrimination power the classes "MBT" and "Landfill" move together in the second classification model and the landfill ALH samples become more similar to the model samples that were collected without particles >20 mm (especially plastics). Stable MBT materials and samples from MBT landfills feature properties of both classes (corner "as well as"). The distance from class "MBT to "Landfill" is 42 for the first classification model (Fig. 3a) and 11 for the second one (Fig. 4). The model (class) distance is determined by fitting members from two defined classes to their own model as well as to the other model. It is calculated on the basis of pooled residual standard deviations. The distance from a model to itself is 1. Distances of more than 3 indicate a significant segregation between the defined classes (Esbensen, 2002).

As shown in Fig. 5a spectra of reactive landfill samples feature more bands (framed wavenumber region 1850–400 cm^{-1}) that can be assigned to functional groups of metabolites, e.g. C=O vibration of carboxylic acids at 1720 cm^{-1}, CO_2 vibration of carboxylates at 1610 cm^{-1}, NH_2 and C—N vibrations of amines at 1400 cm^{-1} and 1320 cm^{-1} respectively. Due to the high reactivity that is reflected by the spectral pattern, samples do not belong to the "Landfill" class as it only represents landfilled waste materials of low and very low reactivity. Waste samples from a reactor landfill differ from currently produced MBT material in composition. Due to this fact they are not assigned to the "MBT" class either despite the presence of reactive samples in this class (Fig. 5b).

3.3. Thermal behavior

Thermal analysis was performed to confirm the assumption that the strong aliphatic methylene bands were caused by the C—H vibration of plastics. Fig. 6a displays the TG- and the DSC-profiles of four selected ALH samples (A–D), three of them (A–C) with a high plastic content (outlying samples in Fig. 3a). The sharp peaks of exothermic reactions in the heat flow profiles are clearly distinguished from the mineral–organic matrix and indicate intact materials in the landfill with a defined combustion temperature, e.g. objects, bags, textiles of plastics. Contrary to resistant organic materials in the landfill, decaying waste organic matter features a different thermal pattern with broad, blurred and missing distinct exothermic peaks. At most two mixed organic fractions can be distinguished in the temperature range 250–350 °C and 350–550 °C.

Based on the heat flow profile a prediction model for the TOC by means of a PLS-R was developed including 75 ALH and 125 landfill samples of low, middle and high (>limit value of the Landfill Ordinance) gas forming potential (Fig. 6b). Samples of landfill ALH are located in the region of higher TOC values which confirms again that TOC and reactivity are not necessarily related.

3.4. Thermal stability

Previous investigations of MBT- and landfill materials have shown that the enthalpy referring to organic dry matter (oDM) is a reliable indicator of acquired stability during the degradation process (Smidt et al., 2010, 2011). The acquired stability against microbial attack is mainly caused by the interaction of organic matter with mineral components. Apart from this mechanism stability in terms of biodegradability is also an implication of the chemistry of the material itself (e.g. plastics, wood, charcoal). These effects can be distinguished by means of the thermal behavior. For interpretation ALH samples were inserted in a diagram of

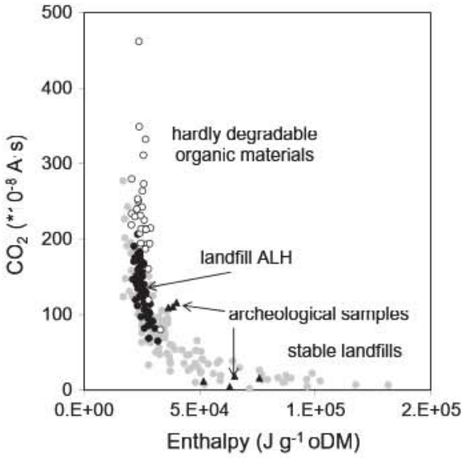

Fig. 7. CO_2 release and enthalpy of stable landfills (gray dots), archeological samples (black triangles), hardly degradable organic materials (black circles) and landfill ALH (black dots).

CO_2 release versus enthalpy referring to oDM (Fig. 7). In order to represent both kinds of stability, the sample set comprised hardly degradable organic materials such as lignite, charcoal and low-quality coals (stability by material properties) and unreactive landfill samples, earthlike cover layers and materials from an archeological excavation (acquired stability). Hardly degradable organic materials feature high CO_2 release and relatively low enthalpy (oDM). They are located at the left side along the *y*-axis. The remaining organic fraction of old landfills is characterized by low organic matter content and high enthalpy (oDM) that reveals a strong interaction of organic and mineral compounds. The corresponding samples are found close to and along the *x*-axis. Samples of landfill ALH are located between the two extreme positions. It can be concluded that the low reactivity is caused as well by hardly or not degradable ingredients (plastics) as by interactions of organic and mineral components. Archeological samples that are dominated by charcoal are found next to ALH samples. Four samples mainly contain mineral components and organic residues (ancient waste dump). They are therefore located close to stable landfill materials with high enthalpies and low CO_2 release.

4. Conclusion

The study has shown that more detailed information about the current and the future status of the landfill regarding its reactivity is available by the applied tool set. Plant sociological relevés before sampling can identify inconspicuous areas and hot spots of the landfill. Such information is useful if the number of samples is limited due to any reason. The different composition of old landfills compared to MBT materials affects the significance of the aliphatic methylene bands in infrared spectra. Therefore the interpretation of these bands requires a relevant modification. Contrary to MBT materials, where the intensity of the C—H vibration (aliphatic methylene bands) mainly depends on the reactivity, it is primarily influenced by plastics in landfill materials. Moreover, low or high plastic contents are a result of the sampling procedure (whole sample or on-site screening <20 mm). The interpretation of the methylene bands in landfill materials was supported by comparison of single spectra and thermal data. Besides the divergent interpretation of the aliphatic methylene bands the correlation between the biological behavior and chemical (spectral) characteristics is insufficient with the increase of mineral components. Therefore classification of landfill samples according to their spectral characteristics was more reliable than prediction of reactivity parameters. The comprehensive investigation of many landfill samples upgrades the library of spectral and thermal data and can serve as data base for other purposes.

The combination of an efficient tool set that provides comprehensive information for lower costs in favor of a sufficient number of samples is a main concern to advance waste analytics for the landfill assessment. Inclusion of eco-toxicity tests that could answer the second relevant question about toxicity of unknown old deposits might complete the powerful tool set in the future.

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References

Austrian Standards Institute, 1986. Analytical Methods and Quality Control of Compost. OENORM S 2023.
Austrian Standards Institute, 2004a. Stability Parameters Describing the Biological Reactivity of Mechanically Biologically Pretreated Residual Wastes – Part 1: Respiration activity (AT4). OENORM S 2027-1.
Austrian Standards Institute, 2004b. Stability Parameters Describing the Biological Reactivity of Mechanically–Biologically Pretreated Residual Wastes – Part 2: Gas Generation by Incubation Test (GS21). OENORM S 2027-2.
Bundesminister für Land- und Forstwirtschaft Umwelt und Wasserwirtschaft (BMLFUW), 2001. Verordnung des Bundesministers für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft über Qualitätsanforderungen an Komposte aus Abfällen (Kompostverordnung). BGBl. Nr. 292/2001.
Bundesminister für Land- und Forstwirtschaft Umwelt und Wasserwirtschaft (BMLFUW), 2008. Verordnung des Bundesministers für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft über Deponien (Deponieverordnung 2008). BGBl. Nr. II 39/2008.
Bogner, J., Spokas, K., Burton, E., Sweeney, R., Corona, V., 1995. Landfills as atmospheric methane sources and sinks. Chemosphere 31 (9), 4119–4130.
Böhm, K., Smidt, E., Binner, E., Schwanninger, M., Tintner, J., Lechner, P., 2010. Determination of MBT-waste reactivity – an infrared spectroscopic and multivariate statistical approach to identify and avoid failures of biological tests. Waste Manage 30 (4), 583–590.
Braun- Blanquet, J., 1964. Pflanzensoziologie, 3rd ed. Springer, Wien, New York.
Brereton, R.G., 2002. Chemometrics: Data Analysis for the Laboratory and Chemical Plant. John Wiley & Sons Ltd., Chichester, England.
CEN/TR 15310-1, 2006. Characterization of Waste – Sampling of Waste Materials – Part 1: Guidance on Selection and Application of Criteria for Sampling Under Various Conditions.
Dell'Abate, M.T., Benedetti, A., Sequi, P., 2000. Thermal methods of organic matter maturation monitoring during a composting process. J. Therm. Anal. Calorim. 61 (2), 389–396.
Ellenberg, H., Weber, H.E., Düll, R., Wirth, V., Werner, W., Paulissen, D., 1992. Zeigerwerte von Pflanzen in Mitteleuropa.
Esbensen, K., 2002. Multivariate Data Analysis – in Practice. Alborg University, Esbjerg.
European Committee for Standardization, 1997. Guidelines for the Determination of Total Organic Carbon (TOC) and Dissolved Organic Carbon (DOC). DIN EN 1484.
European Committee for Standardization, 1998. Water Quality – Determination of Biochemical Oxygen Demand After *n* Days (BODn) – Part 1: Dilution and Seeding Method with Allylthiourea Acid Addition. DIN EN 1899-1 (H51).
Holubar, P., Katinger, H., Braun, R., Donat, C., Fritz, J.I., 2004. EVAPASSOLD – Evaluierung und Erstabschätzung von Altablagerungen. Project Report.
Londo, G., 1976. The decimal scale for relevés of permanent quadrats. Vegetatio 33 (1), 61–64.
Manfredi, S., Tonini, D., Christensen, T.H., Scharff, H., 2009. Landfilling of waste: accounting of greenhouse gases and global warming contributions. Waste Manage. Res. 27 (8), 825–836.
Meissl, K., Smidt, E., Schwanninger, M., 2007. Prediction of humic acid content and respiration activity of biogenic waste by means of Fourier transform infrared (FTIR) spectra and partial least squares regression (PLS-R) models. Talanta 72, 791–799.
Provenzano, M.R., Senesi, N., Piccone, G., 1998. Thermal and spectroscopic characterization of composts from municipal solid wastes. Compos. Sci. Util. 6 (3), 67–73.
Provenzano, M.R., Ouatmane, A., Hafidi, M., Senesi, N., 2000. Differential scanning calorimetric analysis of composted materials from different sources. J. Therm. Anal. Calorim. 61 (2), 607–614.
Scharff, H., van Zomeren, A., van der Sloot, H.A., 2011. Landfill sustainability and aftercare completion criteria. Waste Manage. Res. 29 (1), 30–40.

Smidt, E., Lechner, P., 2005. Study on the degradation and stabilization of organic matter in waste by means of thermal analyses. Thermochim. Acta 438 (1–2), 22–28.

Smidt, E., Meissl, K., 2007. The applicability of Fourier Transform Infrared (FT-IR) spectroscopy in waste management. Waste Manage. 27 (2), 268–276.

Smidt, E., Schwanninger, M., 2005. Characterization of waste materials using FT-IR spectroscopy – process monitoring and quality assessment. Spectrosc. Lett. 38 (3), 247–270.

Smidt, E., Lechner, P., Schwanninger, M., Haberhauer, G., Gerzabek, M.H., 2002. Characterization of waste organic matter by FT-IR spectroscopy: application in waste science. Appl. Spectrosc. 56 (9), 1170–1175.

Smidt, E., Böhm, K., Tintner, J., 2010. Application of various statistical methods to evaluate thermo-analytical data of mechanically–biologically treated municipal solid waste. Thermochim. Acta 501 (1–2), 91–97.

Smidt, E., Böhm, K., Tintner, J., 2011. Evaluation of old landfills – a thermoanalytical and spectroscopic approach. J. Environ. Monit. 13 (2), 362–369.

Tesar, M., Prantl, R., Lechner, P., 2007. Application of FT-IR for assessment of the biological stability of landfilled municipal solid waste (MSW) during in situ aeration. J. Environ. Monit. 9 (1), 110–118.

Tintner, J., Klug, B., 2006. Standortsbeurteilung auf Deponien mithilfe der Vegetationsökologie. Abfallforschungstage 2006, 297–309.

Tintner, J., Klug, B., 2011. Can vegetation indicate landfill cover features? Flora 206 (6), 559–566.

Tintner, J., Kühleitner, M., Binner, E., Brunner, N., Smidt, E., 2012. Modeling the final phase of landfill gas generation from long-term observations. Biodegradation 23 (3), 407–414.

van Praagh, M., Heerenklage, J., Smidt, E., Modin, H., Stegmann, R., Persson, K.M., 2009. Potential emissions from two mechanically–biologically pretreated (MBT) wastes. Waste Manage. 29 (2), 859–868.

Westhoff, V., van der Maarel, E., 1973. The Braun-Blanquet-approach. Ordination and classification of communities. In: Whittaker, R.H. (Ed.), Handbook of Vegetation Science Part 5, first ed. Springer, pp. 617–726.

4 Early Successions in the Floodplain of the Kamp River (Austria)
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Early Successions in the Floodplain of the Kamp River (Austria)

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Abstract: In 2002, large riparian parts of the river Kamp in Lower Austria were affected by a flood which eradicated almost all vegetation at the riverside. From 2003 to 2012 the authors documented early stages of succession at five specific sites with characteristic vertical and horizontal distances to the river. Phytosociological relevés were performed. Ecological indicator values, life traits, diversity indices (species number, evenness) and Sørensen index were used to describe the development within this period. The results demonstrate that, after five years, succession had usually reached a stage where highly competitive species like *Phalaris arundinacea* or *Alnus glutinosa* dominate. The first year after the flood annuals, lots of them common ruderals, dominated. The second year a change to biennial species took place. This transition was paralleled by a numeral maximum of species in the second year. Differences were only evident at a site where vegetation had not been completely eradicated by the flood. At this site a rudimentary stage of annuals was directly followed by an increase of *Phalaris arundinacea* that had been formerly present as well. Succession dynamics were stronger where conditions did not favor competitive species, which led to higher evenness values. Evenness values decreased, whenever *Phalaris* or *Alnus* achieved the dominance. As a matter of fact, these vegetation types showed a minimum in biodiversity. Maxima of evenness were found in vegetation with a high percentage of ruderals and short-living species.

Keywords: Early successional stages, biodiversity, evenness, ecological indicator values.

INTRODUCTION

Vegetation development is steered by site conditions on the one hand and diaspore supply on the other hand. A typical succession after a disastrous flood would commence with a pioneer phase of short-lived, annual species. Usually these species are succeeded by biennials dominating for some years. The later phases are characterized either by perennial grasses and forbs, which can be very invasive, or by shrubs and fast growing pioneer trees. The latest successional stages comprise slowly growing tree species, which are usually rather competitive and shade tolerant (Grime 2001). Along rivers the typical pattern is periodically or episodically disturbed by floods and the resulting changes of the river bed structure and material. Typical floodplain plant species are fostered by their tolerance against low oxygen supply in the root zones and fast regeneration after mechanical stress.

In August 2002 large parts of the Kamp valley in Lower Austria were affected by an extraordinary flood. Although such events are disastrous for people living and working in the region, for ecologists they usually provide a natural possibility to study ecosystem dynamics in a large-scale experiment.

This work took the chance to document the succession at sites with different preconditions in the Kamp valley. Based on phytosociological relevés both the distribution of plant species and their interactions (dominance, facilitation, and competition) were assessed. Various concepts are available and commonly in use to describe vegetation changes (Bråkenhielm and Liu 1998, Ellenberg 1996, Grime 2001, Tilman 1988). However, it is still difficult to find significant parameters to describe favorable plant succession (Critchley 2000). Undisturbed successions most often lead to rather species poor communities, although it has become a common goal to keep biodiversity as high as possible. Therefore six different sites along the river were chosen, where site specific developments of the plant layers were expected. Thus we hoped to find out whether biodiversity issues can be applied under these preconditions.

SITES AND METHODS

Site Description

In order to document different possible successions, five specific sites were selected that represented different preconditions for vegetation succession (Muhar *et al.* 2004). Table **1** gives an overview over the situation at the sites.

The valley width near site A is smaller than 250 m. The potential natural vegetation near the river would be dominated by black alder (*Alnus glutinosa*) or by silver willow (*Salix alba*). The mean water runoff of the river at this place was between 5 and 25 m³/s at a gradient of about 3 %. The site was situated near a dam of a power station at an elevation of about 250 mas. l. The relevé area was surrounded by black alder trees.

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Table 1. General data at the Study Sites

	Site A	Site B	Site C	Site D	Site E
Area (m²)	**40**	**40**	**35**	**40**	**30**
Vertical distance to water level (m)	1.0	0.8	0	0	0.5
Horizontal distance to the main river (m)	30	80	0	0	50
Direct influence of the flood 2002	high	high	high	medium	high
Direct influence of the small flood 2006	none	none	low	low	low

Sites B, C and D were located some hundred meters downstream. Site B was situated on a sand bank of about 50 cm in height, which had been created by the flood in 2002. Site C was situated directly at the river bank. Therefore the site was flooded twice in subsequent years, but without complete destruction of the vegetation. Site D is protected against floods by a stone hillock. The last site E is situated about 11 km downstream near a new arm of the river created by the flood. In 2003 this new arm contained no water; thus an assessment of the vegetation was possible. In the following years the arm was always filled with water, therefore scarce vegetation was observed. Only in 2007 the arm was dry and vegetated again.

Methods

In September 2003 all sites were monitored for the first time and phytosociological relevés were performed. Ecological indicator values for the vegetation as described by Ellenberg (Karrer and Kilian 1990, Ellenberg et al. 1992) were used, and the life traits and life spans of the species were assessed to display the change of vegetation over a period of ten years. The development of the indices for species diversity and evenness were also calculated in order to understand the changes in site conditions. The nomenclature of the species according to Fischer et al. (2008) was applied. The cover of every species was estimated using a combination of the Braun-Blanquet (1964) and Londo (1976) scales as described in Smidt et al. (2007) or Tintner and Klug (2006). Weighted ecological indicator values for moisture, soil reaction and nutrient supply (Ellenberg et al. 1992) were calculated for every relevé.

$$F_{mean} = \frac{\sum(w_i \times F_i)}{\sum w_i}$$

where F means moisture value and w means weight.

As the Kamp valley is situated at the western border of the Pannonian climate region, the indicator values from Karrer and Kilian (1990), adapted to Eastern Austrian conditions, were used instead of Ellenberg´s values. For each species of a relevé, the respective indicator was weighted by the species coverage, and by this a weighted average value for the entire relevé was calculated. This calculation was supported by the software VegePro (© Irene Ahamer).

The herbal layer species were grouped according to their life spans into annuals, biennials, perennial grasses and forbs and perennial woody species (shrubs and trees). For many species this classification is difficult, as they can reproduce annually but can survive even for several years. Following the demands for further mathematical procedures, these species were assigned to only one trait. The resulting error was minimized by the big amount of data points in one relevé.

Furthermore the total number of species and the total cover at every relevé were evaluated which allows the calculation of the relative and absolute portion of cover represented by the specific life traits. The indicator values and the life traits were calculated for both the total vegetation and the herbal layer. Species numbers were taken from the herbal layers of the relevés.

As parameter of diversity we used H' - calculated by the Shannon-Wiener function – (Tremp 2005).

Evenness was calculated as follows:

$E = H'/H_{max}$; where $H_{max} = \ln s$; s = total number of species.

To compare ß-diversity between years and sites, the coefficient of Sørensen (1948) was used. This method compares two vegetation lists. The index is calculated as follows:

Index = 2*a / (2*a + b + c); where a = number of species in both lists, b and c are the number of species in just one of the two lists

Principal Component Analysis (PCA) was carried out using Unscrambler 9.2 (Camo). PCA matrices were created where the coverage of all species with a certain indicator value was summed up. Therefore the matrix consisted of the R- and the N-value with nine columns (one column for each step of the indicator scale) and the F-value with 12 columns respectively. The ubiquitous species reflecting moderate or average conditions (defined by humidity indicator F, soil reaction indicator R or nutrient indicator N between 5 and 7) were removed from the data matrix, as they do not reflect extreme site conditions. The resulting parameter set for the PCA comprised the F-values "1" to "4" and "8" to "12", the R-values "1" to "4" and "8" to "9", the N-values "1" to "4" and "8" to "9". Life traits were indicated in ranges from "1" to "4" as reported by Tintner and Klug (2011).

RESULTS AND DISCUSSION

Species Dynamics

Site A had been completely deprived of vegetation by the flood of 2002. In the first year after the flood a typical annual stage was established. Mainly species of the Chenopodietea dominated. In the second year vegetation diversified (Fig. (**1**). Perennial grasses (especially *Phalaris arundinacea* and *Agrostis stolonifera*) grew up, annual

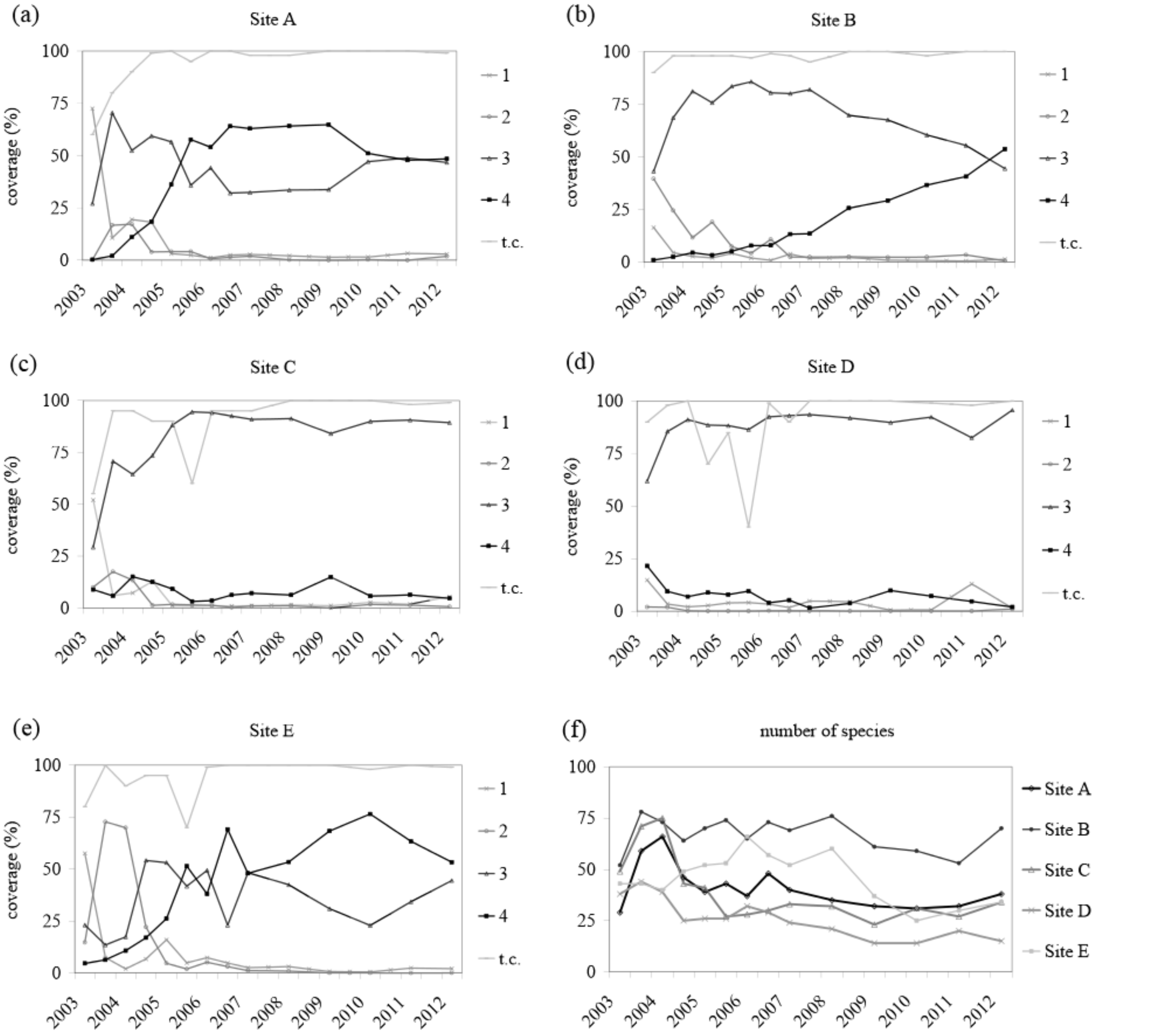

Fig. (1a-e). Development of the portions of ground cover by different life traits for each relevé in percentage of the total vegetation coverage: 1 - annual, 2 – biennial species, 3 - perennial grasses and forbs, 4 - woody species; t. c. - total vegetation coverage in percentage of the total relevé area; (**f**): total species number of each relevé over the years.

species were still present and biennial species (*Verbascum phlomoides* and *Erigeron canadensis*) performed best compared to the other years. First sprouts of *Alnus glutinosa* appeared. In the third year, *Phalaris arundinacea* and *Alnus glutinosa* increased their coverage. Only in summer an amount of non-invasive annual species were added (*Vicia hirsuta*, *Oxalis stricta*). In the following years, under the competition of the shrub-shaped *Alnus glutinosa,* the coverage of *Phalaris* decreased permanently. However, it remained the dominant species among perennials, even when more and more shade-tolerant species grew up (e. g. *Poa trivialis*, *Lamium maculatum*). During the last years these species increased their coverage significantly together with nitrophilous species of floodplain forests (e. g. *Urtica dioica*, *Carduus crispus*).

Site B mainly is made up by sandy substrate. In the first year *Erigeron canadensis* dominated the site. Also *Artemisia vulgaris* and some perennial grasses (*Elymus repens* and *Poa trivialis*) were frequent. Annual species were also present (*Persicaria dubia*, *Apera spica-venti*). In the following years *Artemisia* decreased more and more. For some years, a few *Trifolium* species became frequent. The annual *Apera spica-venti* as well reached high cover values for some time. For the whole period under observation medium size grasses were frequent (*Poa trivialis*, *Poa palustris*, *Agrostis stolonifera*). After three years, perennials with a higher moisture demand established themselves (*Phalaris arundinacea*, *Juncus effusus*). However, only after four years a clear tendency was observed towards site specific *Salix fragilis*, *Alnus glutinosa*, additionally *Betula pendula*, *Carpinus betulus*, *Pinus sylvestris* and *Tilia cordata* that immigrated from the surrounding forested slopes. The moisture (F)-value increased continuously from 5.6 in 2003 to 7.6 in 2012.

Site C is situated directly at the river bank. The substrate is coarse gravel. The vegetation had been completely eradicated by the flood. In the first year mainly annual species emerged (*Persicaria* sp.). However, the total coverage of the area was comparably low (55 %). From the second year onward *Phalaris arundinacea* had performed best among all species. Obviously its rhizomes had survived the flood of 2002 in the substrate. Various other species typical for wetlands (*Poa trivialis*, *Stellaria aquatica*, *Solanum dulcamara*) became frequent but not dominant. The last year potentially invasive species established more and more (*Urtica dioica*, *Mentha longifolia*). In 2012 tree species did not cover much, even when some individuals grew up to more than two meters (*Alnus glutinosa*, *Salix fragilis*). In the spring of 2006 a small flood superposed the substrate with a new layer of gravel and sand. The vegetation was partly buried, and the total coverage was low (60 %). However, the plants survived, and therefore the succession continued despite the disturbance.

At site D the vegetation was not totally eradicated in 2002. This can be explained by the mentioned stone hillock and some trees, which protected the site. However, in the first year the coverage by annual species was only about 15 %. In the beginning two *Veronica* species covered about 25 % together, but from the third year on they were absent. The coverage of *Juncus effusus* was also quite frequent in the second year, but declined to less than 1 %. An interesting fact is the decrease of *Salix fragilis* and all other woody species. Their coverage amounted to more than 20 % in the first year. From the second to the fourth year the coverage remained at a level between 7 and 10 % and declined over the years. In 2007 all woody species together covered less than 2 %. The species which has dominated the site since 2004 is *Phalaris arundinacea*. Even the small flood in the spring of 2006) did not change the situation as it was the case at site C and E. On the contrary, the coverage of *Phalaris* increased even from about 60 % in 2005 to about 80 % in summer 2006. A remarkable detail is the presence of *Scirpus radicans* in the relevé. The species is rare throughout Austria.

Site E started with a typical annual stage (*Setaria pumila*, *Oxalis stricta*, *Bidens tripartita*, and others). In the second year biennial species dominated (mainly *Erigeron annuus* and *Erigeron canadensis*). The third year the vegetation became more and more heterogeneous. *Trifolium* species but also perennial grasses like *Poa palustris* became more frequent, but did not came to dominance. At the same time tree species grew up. The second flood affected the herbal layer of site E as well. As a consequence, the total cover of the site decreased. Two months later, however, a great number of new species was identified, mainly perennial grasses and forbs (Fig. (**1**). After five years the aspect of this site had changed very much. In large parts the trees reached a height of more than four meters. Between these trees meadow species, invasive ruderal species (*Cirsium arvense*, *Mentha longifolia*, *Calamagrostis epigejos*, *Phalaris arundinacea*, and others) and less-invasive ruderals intermingled. Until 2012 mainly shade-tolerant wetland species performed well (*Urtica dioica*, *Poa palustris*, *Stellaria aquatica*). The site developed similarily to site A (Fig. (**3**). A remarkable result is the performance of *Robinia pseudacacia*. This tree is well known in the Kamp valley as an invasive neophyte. It is a typical pioneer on dry sites in the Pannonian area (Protopopova et al. 2006). In the Pannonian climate of the Kamp River *Robinia pseudacacia* had become such a problem that the county government had started projects to repel this species. Therefore the performance of *Robinia* is of special interest. In the first year after the flood *Robinia pseudacacia* covered about 1 %, like *Alnus glutinosa*, *Populus nigra* and *Salix fragilis*. As the site is situated about 0.5 m above the water table, *Robinia pseudacacia* could cope with the competition of floodplain pioneers until the third year. Then the three other species grew more efficiently, not only in height but also in coverage. Until 2012 especially the coverage of *Populus nigra* collapsed from about 50 % to less than 1 %, whereas the coverage of *Robinia* increased continuously up to 30 %. The second species with a continuous increase of its coverage up to about 10 % is *Populus alba*.

The new arm below site E was investigated only in September 2003 and in August 2007. In 2003 short living ruderals and sprouts of black poplar were present. In 2007 the vegetation had a different character. Only few annual species (4.8 %) and sprouts of trees (0.2 %) were found. Yet the perennial grasses *Phalaris arundinacea* and *Agrostis stolonifera* conquered the area together with *Artemisia vulgaris*. The succession had not started from an early point, but the surrounding perennial vegetation penetrated into the gaps. Especially the two grasses succeeded in immigrating with their rhizomes and stolones over a distance of only a few meters. From 2008 to 2012 the site was filled with water all the time.

Results of Life Traits, Indicator Values and Diversity

Fig. (**1**) displays some effects typical for early successional stages. Annual species in the first year are followed by biennial species in the second year and perennials later. After some years, woody species come to dominance. Site A and site E demonstrate this best. At site B perennial species dominate even in the second year, whereas the coverage of shrubs and trees is still increasing. At site C and especially site D the perennial *Phalaris arundinacea* dominated the vegetation from the second year. Woody species seem to be suppressed there by the high water table.

Fig. (**1**) also shows the changes of total species numbers. Successions often lead to a maximum of short-lived species in the second year (sites A, C and D). On the one hand the annual species are still present and the biennial species reach their maximum in growth and diversity, on the other hand perennials and especially woody species have already started to establish themselves. Later on, annual and biennial species decrease in number and vitality. Site B shows a small scale vegetation pattern. No species gained dominance until 2012. Due to local disturbances by animals, even annual and biennial species have survived on a low coverage level. Site E became more heterogeneous by the increase of tree species. After the first two years, the trees grew up in some parts of the relevé area. In the sunlit gaps in between, various micro-habitats have established. However, since 2008 the number of species has been reduced drastically.

At some sites the average Ellenberg moisture value shows a marked increase (Fig. (**2**) as typical wetland species (*Alnus glutinosa*, *Phalaris arundinacea*) increased their

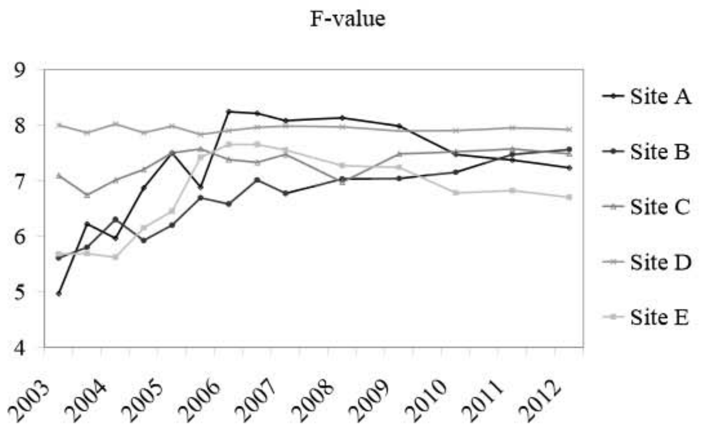

Fig. (2). F- (moisture) indicator value in the course of the years.

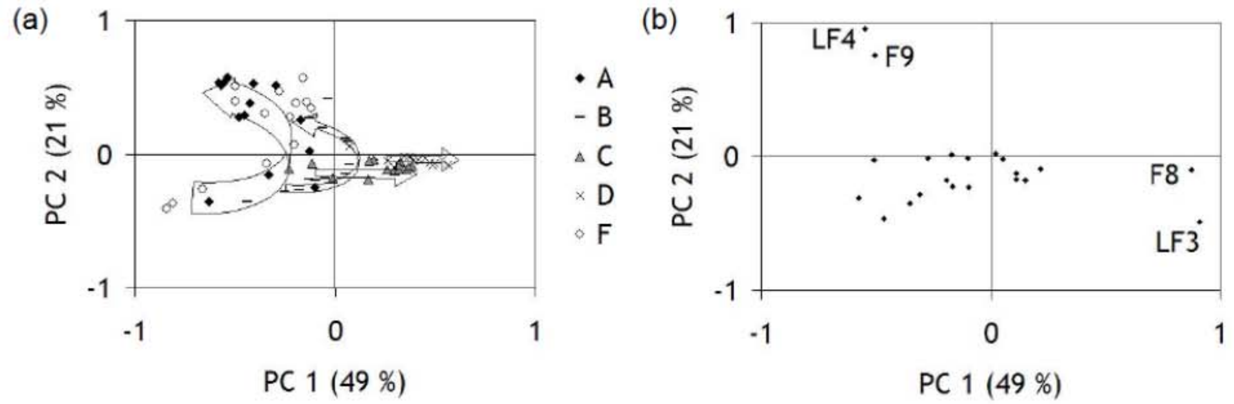

Fig. (3a). scores plot, the arrows indicating the development over the years (explanation of PCs: PC 1: 49 %, PC 2: 21 %); (**b**) loadings plot. F means moisture indicator value, LF means life trait.

coverage. The annual and biennial species are less moisture demanding. Therefore the F-value started at quite low levels in the first year. Probably even the substrate increased its portion of fine blown-in particles and additionally by the deposit of organic matter. Sites C and D had never lost their connection to the water table; therefore their F-values remained more or less stable.

The indicator values for nutrients and soil reaction did not change significantly.

Castillo-Campos *et al.* (2008) found out, that in tropical forests in Veracruz, Mexico, species richness was higher in secondary vegetation than in primary forests. Shrubs and herbs contributed more to the vegetation of the former. Similar results are reported by Kreyling et al. (2008) from a long-term succession in British Columbia, Canada.

The following results of PCA illustrate to which successional stages the sites develop (Fig. (**3**). The first two principal components (PC) of the parameter set described in the Sites and Method chapter of all relevés are shown in figure 3. The loadings plot explains how the parameter set is combined in the PC. Input parameters are the ecological indicator values and the life trait-values. Site A and E tend towards alder/ willow forests whereas sites C and D tend more towards a vegetation dominated by *Phalaris arundinacea*. All sites except D started at an early succession stage (Fig. 3a, lower left corner). Site B started quite similar to site C, but in the last years the tendency towards an alder/ willow forest became obvious.

The calculations of diversity support the observations described above. Evenness was quite high in all relevés of 2003 (Fig. (**4**). This is remarkable especially for site D. At all sites with a development towards the dominance of one wetland species, evenness decreased. At sites B and E the succession did not yet reach such a stage, therefore also evenness is still quite high. During the biennial stage in site E, where *Erigeron* spp. dominated, evenness was comparatively low.

In Fig. (**5**) the Sørensen indices of the relevés are displayed with respect to the latest relevé date (2012). Changes in species composition were substantial between 2006 and 2007 and between 2010 and 2012 compared to the other years. Nevertheless, the strongest changes were recorded between the first and the second year. This corresponds to the maximum of species number in these years. It can be concluded that the development over these five years was remarkable. Site A and site E show the most constant increase of the index. In total the sites A, B and E showed the greatest changes as they started from a very early stage of succession. Site D showed the smallest changes, probably due to the small flood of 2006.

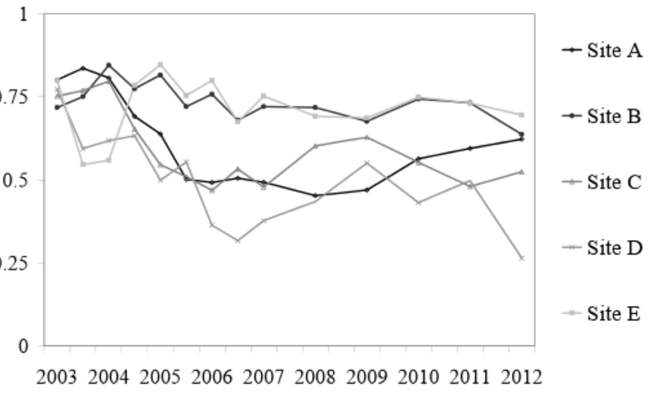

Fig. (4). The development of evenness in all relevés.

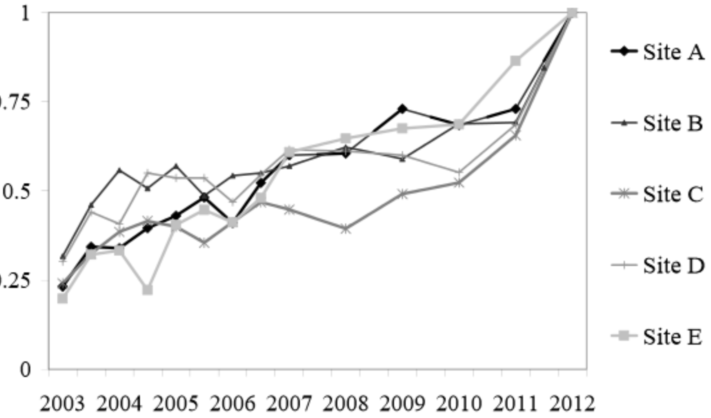

Fig. (5). Sørensen-indices relative to relevés of 2012.

Table 2. Sørensen-Indices Related to Relevés of 2012 and all Relevés of the Years 2003, 2007 and 2012 (First Letter: Site, Number: Year)

	A2003	B2003	C2003	D2003	E2003	A2007	B2007	C2007	D2007	E2007	A2012	B2012	C2012	D2012	F2012
A2012	0.23	0.22	0.22	0.15	0.30	0.60	0.27	0.30	0.22	0.38	1.00				
B2012	0.16	0.32	0.31	0.25	0.22	0.47	0.57	0.30	0.31	0.52	0.42	1.00			
C2012	0.13	0.21	0.24	0.17	0.21	0.43	0.27	0.45	0.28	0.32	0.41	0.43	1.00		
D2012	0.23	0.21	0.19	0.30	0.07	0.29	0.12	0.42	0.62	0.35	0.18	0.25	0.29	1.00	
E2012	0.21	0.18	0.14	0.18	0.20	0.46	0.28	0.23	0.35	0.61	0.51	0.45	0.31	0.38	1.00

The relationship of the sites A to E and their development is demonstrated in Table **2**. In the last year sites A and E matched best. Interestingly the sites C and D (both dominated by *Phalaris arundinacea*) differ considerably in their species lists.

Early successional stages as described in this paper seem somehow exotic with regard of maximizing biodiversity, as species numbers or evenness are highest in early stages with a high frequency of annual and biennial ruderal (often neophytic) species. Chabrerie *et al.* (2001) found the richest plant communities in the Seine estuary between regimes with natural *and* anthropogenic disturbance.

CONCLUSIONS

The five sites showed different, but typical early successions after a flood. Some developed from an annual over a biennial towards a perennial stage dominated by trees (in this case *Alnus glutinosa* or *Salix fragilis*) or herbaceous plants - mainly grasses (*Phalaris arundinacea*). Where none of these species comes to early dominance, the succession is decelerated.

The development of the sites could be documented by various parameters. Average ecological moisture values increased at the sites A, B, C and E. This can be explained by the increase of the typical vegetation of river sides and wet plains. As the plants grow and root into deeper zones, they reach the water table. This is a clear difference between the adapted species and the ruderals of early stages of succession. Therefore the degree of ruderalization can be displayed by the F-indicator value. PCA was a good tool to get more information about the weighted ecological indicator values of every single relevé. This information is lost by calculating averages.

Evenness is high immediately after the flood, after other disturbances, and where ruderal species can survive alongside with the typical wetland species. As soon as competitive species like black alder or well-adapted clonal grasses like *Phalaris arundinacea* increase their coverage, evenness decreases. This must be taken into account, whenever evenness is taken as a measure for successful ecological engineering (Pedersen et al. 2007). Corresponding results are reported by Zhang et al. (2007). The parameter evenness depends to a high degree of plant strategies (Grime 2001). Vegetation with dominant competitive species tends to low evenness, whereas ruderals (and especially stress-tolerant species, which are less frequent in early succession stages,) form a type of vegetation with higher evenness. Also α-diversity, measured as number of species, is higher in early succession stages – especially the biennial stage – with a high percentage of ruderals.

CONFLICT OF INTEREST

The authors confirm that this article content has no conflicts of interest.

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REFERENCES

Bråkenhielm, S & Liu, Q (1998) Long-term effects of clear-felling on vegetation dynamics and species diversity in a boreal pine forest. *Biodiversity and Conservation* 7, 207-20.

Braun-Blanquet, J (1964) *Pflanzensoziologie - Grundzüge der Vegetationskunde*, Springer, Vienna.

Castillo-Campos, G, Halffter, G & Moreno, CE (2008) Primary and secondary vegetation patches as contributors to floristic diversity in a tropical deciduous forest landscape. *Biodiversity and Conservation* 17, 1701-14.

Chabrerie, O, Poudevigne, I, Bureau, F, Vinceslas-Akpa, M, Nebbache, S, Aubert, M, Bourcier, A & Alard, D (2001) Biodiversity and ecosystem functions in wetlands: A case study in the estuary of the Seine River, France. *Estuaries* 24, 1088-96.

Critchley, CNR (2000) Ecological assessment of plant communities by reference to species traits and habitat preferences. *Biodiversity and Conservation* 9, 87-105.

Ellenberg, H (1996) *Vegetation Mitteleuropas mit den Alpen in ökologischer, dynamischer und historischer Sicht*. 5th ed., Ulmer, Stuttgart.

Ellenberg, H, Weber, HE, Düll, R, Wirth, V, Werner, W & Paulißen, D (1992) *Zeigerwerte von Pflanzen in Mitteleuropa*. 2nd ed. Goeltze, Göttingen.

Fischer, MA, Oswald, K & Adler, W (2008) *Exkursionsflora für Österreich, Liechtenstein und Südtirol*. 3rd ed. Biologiezentrum der Oberösterreichischen Landesmuseen.

Grime, JP (2001) *Plant strategies, vegetation processes, and ecosystem properties*. 2nd ed. Wiley, Chichester.

Karrer, G & Kilian, W (1990) Standorte und Waldgesellschaften im Leithagebirge Revier Sommerein. *Mitteilung Forstlicher Bundesversuchsanstalt* Wien 165, 1-244.

Kreyling, J, Schmiedinger, A, Macdonald, E & Beierkuhnlein, C (2008) Slow understory redevelopment after clearcutting in high mountain forests. *Biodiversity and Conservation* 17, 2339-55.

Londo, G (1976) The decimal scale for relevés of permanent quadrats. *Vegetatio* 33, 61-4.

Muhar, S, Poppe, M, Egger, G, Schmutz, S & Melcher, A (2004) *Flusslandschaften Österreichs Bundesministerium für Bildung*, Wissenschaft und Kultur, Vienna.

Pedersen, ML, Friberg, N, Skriver, J, Baattrup-Pedersen, A & Larsen, SE (2007) Restoration of Skjern River and its valley-Short-term effects on river habitats, macrophytes and macroinvertebrates. *Ecological Engineering* 30, 145-56.

Protopopova, VV, Shevera, MV & Mosyakin, SL (2006) Deliberate and unintentional introduction of invasive weeds: A case study of the alien flora of Ukraine. *Euphytica* 148, 17-33.

Smidt, E, Tintner, J & Meissl, K (2007) *New Approaches of Landfill Assessment and Monitoring*, In: Velinni, AA (Ed.), Landfill Research Trends, NOVA Publishing NY., pp. 191-225.

Sørensen, T (1948) A method of establishing groups of equal amplitude in plant sociology based on similarity of species and its application to analyses of the vegetation on Danish commons. *Biologiske Skrifter/ Kongelige Danske Videnskabernes Selskab* 5, 1-34.

Tilman, D (1988) *Plant strategies and the dynamics and structure of plant communities* Princeton Univ. Pr, Princeton.

Tintner, J & Klug, B (2006) *Standortsbeurteilung auf Deponien mithilfe der Vegetationsökologie*, In: Kühle-Weidemeier, M (Ed.), Abfallforschungstage, Cuvillier, Hannover.

Tintner, J & Klug, B (2011) Can vegetation indicate landfill cover features? *Flora: Morphology, Distribution, Functional Ecology of Plants* 206, 559-66.

Tremp, H (2005) *Aufnahme und Analyse vegetationsökologischer Daten* Ulmer, Stuttgart.

Zhang, GS, Wang, RQ & Song, BM (2007) Plant community succession in modern Yellow River Delta, China. *Journal of Zhejiang University. Science. B.* 8, 540-8.

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5 Vegetation ecology as a tool for description of a geological land slide in Vorarlberg, Austria

Abstract

Landslides often are reasoned by reactions of the geological subsoil material. Usually a high water content in clayey substrate material leads to movements on the surface. The monitoring of such areas has to take these characteristics into account. Conventional monitoring methods focus on substrate sampling by core drilling or by means of geophysics. Changes in time and small scale spatial deviations can be assessed only by expensive monitoring series. Vegetation ecology provides information with a high resolution in space and - up to certain degree - also in time. This paper proves the applicability of ecological parameters like diversity index and indicator values as description tools for the geological subsurface.

Introduction

The biggest landslides with affected areas larger than more than 0.5 km² in Austria in historical time are situated at the Pechgraben (near Großraming), at the Gschliefgraben (near Gmunden) in Upper Austria and at the Rindberg (near Sibratsgfäll) in Vorarlberg. All these areas are situated in the geological formation of Ultrahelvetikum (Alexander et al., 1964; Faupl, 1975; Prey, 1983; Widder, 1988). Mainly marl, combined with argillite, sand- and siltstones are dominating, often strongly intermingling each other. On the long term, such a geological underground leads to strong erosion. On the short term, the movement procedure depends on small scale morphology, water infiltration and clay content. The detailed assessment of the geological situation requires a huge amount of drilling holes, which is very expensive.

The geological underground mainly determines soil composition. The soil components again – combined with climatic variables - determine the preconditions for vegetation, above all for species composition. Vegetation ecological indication has a tradition of already about 50 years. Generally the methods are used in the context of nature conservation, land use change and other anthropogenic influence on habitats. Very often vegetation would even have the potential to release information for technical planning, management and especially maintenance plans. Movement rates in the recent time, the potential movement risk and susceptibility of the habitats are relevant for such planning. They depend on the geological underground material, which is influencing the soil formation and vegetation composition.

The goal of the current investigation was to prove the indicative ability of vegetation on relevant variables such as moisture distribution and the geological underground.

Material and Methods

Site description

The landslide area of Rindberg is situated near the village Sibratsgfäll in Vorarlberg, Austria. In 1999 a slope area of about 1.5 km² shifted downwards with movement rates up to a meter per day and total distances of about 480 metres within half a year. Reasons for the movement were the geological situation on the one hand, and extraordinary rainfall of more than 500 mm in May 1999 on the other hand (Bauer, 2000). The geology of the slope can be divided into four different units containing Flysch rocks (sandstones), but also different rocks of the Ultrahelvetikum (marl, argillite). The thickness of the sliding substrate increases wedge-shaped from the highest point to the bottom of the slope to more than 65 m.

The assessment area covered the top part of that slope. In this selected area five divisions can be distinguished (figure 1). The westernmost division, called Sommerstadel (A), had been a pasture before 1999. The strongest motions had occurred under the mountain top of the Feuerstätter Kopf (1645 m) called Wilds Ries (B) as well as at Feuerstätter west (C) and

Feuerstätter east (D). Before 1999 this area was covered with forest. The easternmost division called Lustenauer Ries (E) is a hollow-shaped old pasture.

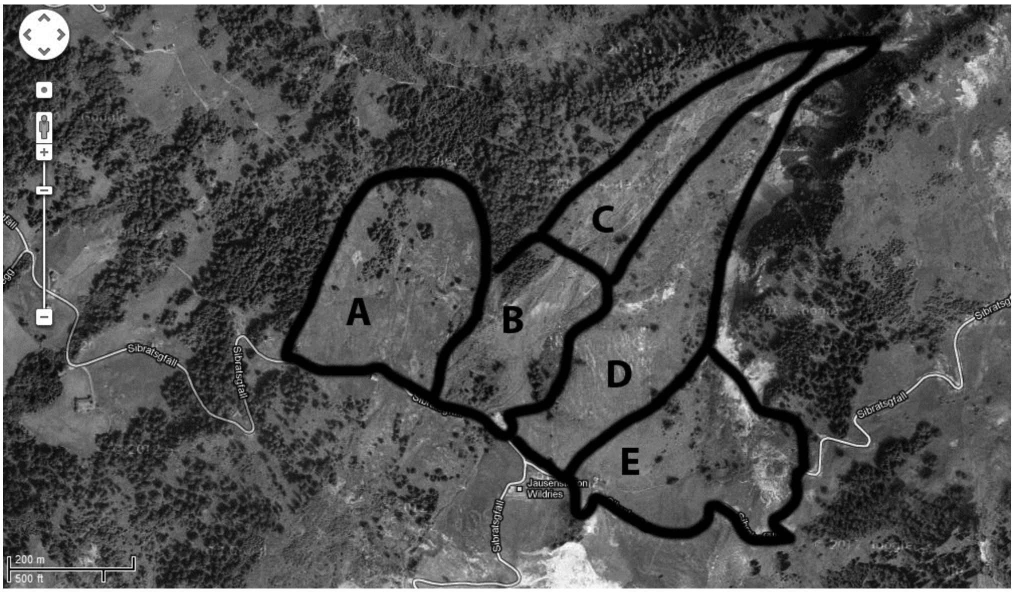


Figure 1: Study area of about 27 ha with the divisions Sommerstadel (A), Wilds Ries (B), Feuerstätter west (C), Feuerstätter east (D) and Lustenauer Ries (E); aerial photo © Google Maps, 2013.

Methods

In the years 2003 and 2012 84 phytosociological relevés altogether were performed. Species nomenclature followed Fischer et al. (2008). For the estimation of plant cover a combination of the Braun-Blanquet (1964) and Londo (1976) scales was used as described in Smidt et al. (2007) and Tintner and Klug (2006). For every relevé, weighted ecological indicator values for moisture (F), soil reaction(R) and nutrient supply (N) (Ellenberg et al., 1992) were calculated. The respective indicator for each species of a relevé was weighted by the species coverage, and by addition a weighted average value for the entire relevé was calculated. This calculation was supported by the program VegePro (© Irene Ahamer). According to Ellenberg et al. (1992) the coverage of species indicating changing moisture, and the coverage of species indicating waterlogging were calculated. Principal Component Analyses was performed with the Ellenberg values F, R, N and life traits as described in the first chapter of this thesis.

The relevés were grouped into the following three vegetation types: 1) meadows and pastures, 2) marshes, 3) tall herbaceous and clear-cutting vegetation. Meadows and pastures in the following are referred to as "meadows". Species lists were composed and weighted by a combined variable called "presence". This variable was calculated by multiplying the abundance of a certain species and its cumulative coverage over all relevés of a vegetation type.

Species of the ground layer were grouped into annuals, biennials, perennial herbaceous and perennial woody species (shrubs and trees) as indicated by Tintner and Klug (2011).

Additionally the total species number was evaluated. Also the relative and absolute portion of cover represented by the specific life traits was calculated. The indicator values and the life traits were calculated over the total vegetation and for the herb layer separately; species numbers were taken from the ground layers of the relevés.

Further variables were used as described in the first chapter of this thesis - the Shannon-Wiener function (Tremp, 2005), evenness and the Sørensen index according to Sørensen (1948)

The variables (weighted average indicator values, life traits, evenness, species number) were tested by cross-classification (with the factors “part” and “vegetation type”, $\alpha = 0.05$). As homogeneity of variances was not given, Tamhane-T2 Test (Tamhane, 1977) was used as a post-hoc test ($\alpha = 0.05$). Sørensen indices between the groups were tested by Oneway-ANOVA, Tukey HSD (Rasch et al., 1999) was used as a post-hoc test ($\alpha = 0.05$).

In addition to the relevés, eight soil samples were collected: two from Sommerstadel, four from Lustenauer Ries, one from Wilds Ries and one from Feuerstätter west. Measurements of grain size distribution, pH and calcite content (Scheibler method) were performed (Austrian_Standards_Institute, 1999). Grain size distribution was measured by means of wash-sieving up to a size of 20 µm. The smaller grain-size fractions were screened by Sedigraph (SediGraph III micromeritics ®).

Results and Discussion

Vegetation data

The relevés were split up according to the different vegetation types and the five parts of the investigated area. The factor (part and vegetation type) combinations with at least four relevés were evaluated statistically. 73 out of the 84 relevés could be used to proceed.

Table 1 and 2 assemble the lists of the15 most present species in meadows, marshes, as well as in tall herbaceous and clear-cutting vegetation independent of the division of their origin.

Table 1: The 15 species with highest combined presence in meadows and marsh, respectively, and their share of total combined presence; in brackets the ecological indicator values for F, R, N; good indicator species are marked by √; ~ indicates changing moisture conditions, = indicates water logging

meadow	%	marsh	%
*Cynosurus cristatus (*5; x; 4)	18.4	*Juncus effusus* (7; 3; 4)	17.8
√ *Festuca nigrescens* (x; 3; 2)	8.6	√ *Carex paniculata* (9; 6; 4)	9.6
√ *Nardus stricta* (x~; 2; 2)	8.5	√ *Scirpus sylvaticus* (8; 4; 4)	9.1
Agrostis capillaris (x; 4; 4)	7.1	√ *Molinia caerulea* (7; x; 1)	8.8
Trifolium pratense (x; x; x)	6.6	*Equisetum sylvaticum* (7; 5; 4)	8.1
Dactylis glomerata (5; x; 6)	6.3	√ *Equisetum palustre* (8; x; 3)	6.0
Anthoxanthum odoratum (x; 5; x)	4.5	*Agrostis capillaris* (x; 4; 4)	4.1
Plantago media (4; 7; 3)	3.0	*Anthoxanthum odoratum* (x; 5; x)	3.9
Trifolium repens (5; 6; 6)	2.6	√ *Caltha palustris* (9=; x; x)	3.2
Juncus effusus (7; 3; 4)	2.5	√ *Potentilla erecta* (x; x; 2)	3.1
Lotus corniculatus (4; 7; 3)	2.5	√ *Tussilago farfara* (6~; 8; x)	2.2
Alchemilla vulgaris agg. (x; x; x)	2.5	√ *Agrostis stolonifera* (x=; x; 5)	2.1
Centaurea jacea (x; x; x)	2.1	*Lysimachia nemorum* (7; 7; 7)	1.7
Plantago lanceolata (x; x; x)	1.9	*Dactylis glomerata* (5; x; 6)	1.7
Lolium perenne (5; 7; 7)	1.4	√ *Equisetum fluviatile* (10; x; 5)	1.4
Sum over 15 most present species	78,5	Sum over 15 most present species	82,8
Sum over 62 further species	21,5	Sum over further 64 species	17,2
Sum over all 177 species of the habitat type	100.0	sum over all 179 spec. of the habitat type	100.0

Table 2: The 15 species with highest combined presence in tall herbaceous and clear-cuting vegetation, and their share of total combined presence; in brackets the ecological indicator values for F, R, N; good indicator species are marked by √; ~ indicates changing moisture conditions

tall herbaceous and clear-cutting vegetation	%
Rubus idaeus (x; x; 6)	22.7
Dactylis glomerata (5; x; 6)	12.4
√ *Tussilago farfara* (6~; 8; x)	9.0

Picea abies (x; x; x)	6.7
Rubus fruticosus agg. (x; x; x)	3.5
Dryopteris filix-mas (5; 5; 6)	3.3
Agrostis capillaris (x; 4; 4)	3.1
Petasites albus (6; x; 5)	2.7
Equisetum sylvaticum (7; 5; 4)	2.6
√ *Urtica dioica* (6; 7;8)	2.0
√ *Carex flacca* (6~; 8; 4)	1.8
Carex sylvatica (5; 6; 5)	1.8
Geranium sylvaticum (6; 6; 7)	1.3
Fragaria vesca (5; x; 6)	1.3
√ *Chaerophyllum hirsutum* (8; x; 7)	1.3
Sum over 15 most present species	75,5
Sum over further229 species	24,5
Sum over all 244 species of the habitat type	100.0

The species lists reflect the use of vegetation. In meadows/ pastures the dominant species are medium-size grasses with relatively low nutrient demand. Livestock obviously does not introduce very high amounts of nutrients. In marshes most of the species have a high water demand but a relatively low nutrient demand. Many of them are good indicator species. The tall herbaceous and clear-cutting vegetation is dominated by comparably ubiquitous species appearing over a big span of the scales (water supply, soil pH, nutrient content). In comparison with the other habitat types the nutrient demand is higher. This can be explained by the higher mineralization rate of this vegetation type which is very often exposed to full sunlight.

The first evaluation dealt especially with mean indicator values, life traits and diversity parameters. The Tamhane-T2 test revealed significant differences for the indicator values F, R and N, the percentage of biennial and perennial species, the diversity parameter H’ and evenness. These results are presented in the following figures and tables.

Significant interactions were found for the percentage of biennial species and the percentage of indicators for changing moisture. All other interactions were not significantly different from zero, though due to the big standard deviation of the error term the relevant power of the interaction tests are quite poor for all variables except the F-value. So for all other variables the acceptance of the hypothesis “no interaction exists” is doubtful.

Figure 2 and table 3 contain the results for the indicator value F (soil moisture). They demonstrate the moist to wet conditions of the marshes and the higher moisture conditions at tall herbaceous and clear-cutting vegetation. The meadows are characterized by a medium water supply.

The divisions were mainly differentiated by the occurrence of only in Feuerstätter west, Wilds Ries and Sommerstadel. A reason could be found in geology, as in Feuerstätter east and Lustenauer Ries the underground is formed by calcareous rock mass, whereas the geology of the rest leads to other soil types with a higher content of clay minerals. These clay minerals are obviously necessary to compact the soil to an extent that marsh vegetation can establish itself.

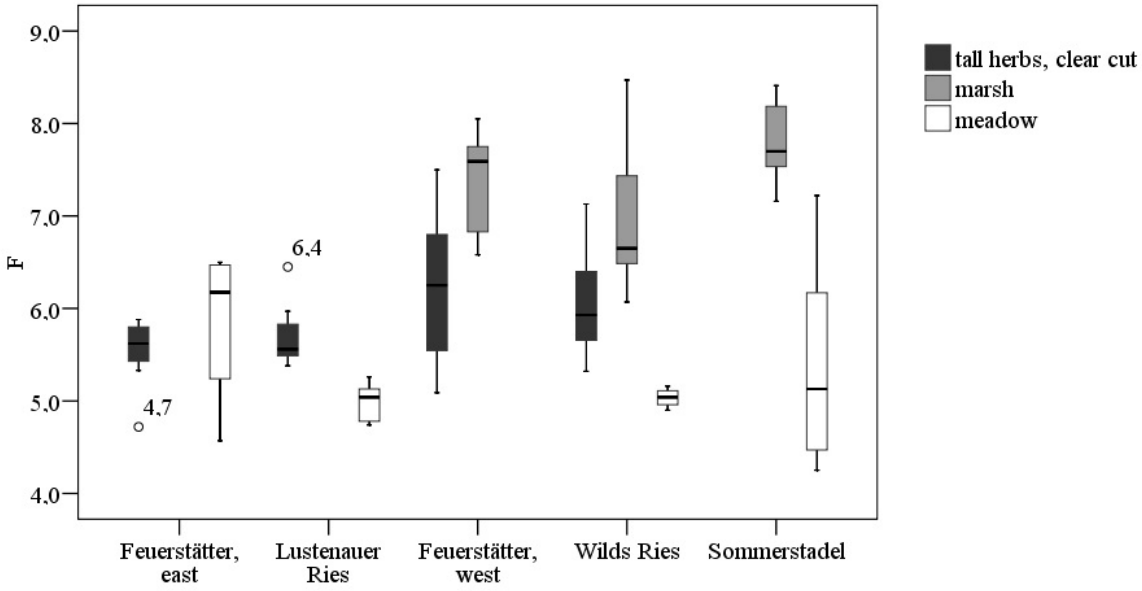


Figure 2: Boxplots of the F-indicator values of the different divisions and vegetation types

Table 3: Results of the Tamhane-T2 test with respect of the F-value

part	group 1	group 2	group 3
Lustenauer Ries $_{n=12}$	5.4 ± 0.5		
Feuerstätter east $_{n=14}$	5.6 ± 0.5	5.6 ± 0.5	
Wilds Ries $_{n=18}$	6.2 ±1.0	6.2 ±1.0	6.2 ± 1.0
Feuerstätter west $_{n=15}$			6.7 ± 0.9
Sommerstadel $_{n=14}$		6.8 ± 1.5	6.8 ± 1.5
vegetation type	**group 1**	**group 2**	**group 3**
meadow $_{n=19}$	5.3 ± 0.8		
tall herbs, clear-cutting vegetation $_{n=32}$		5.9 ± 0.6	
marsh $_{n=22}$			7.4 ± 0.7

In figure 3 and table 4, results for the indicator value R (soil reaction) are presented. They reflect mainly the geological underground, stressing the assumption that in Lustenauer Ries and Feuerstätter east more calcareous material can be found in the underground. Interestingly, this is independent of the vegetation type. There are no interactions between the factors part and vegetation type. The higher lime content is visible even if the histories of the vegetation and its land use, respectively, are the same. Among the vegetation types tall herbaceous and clear-cutting vegetation has the highest R-values; the lower growing meadows and marshes depend rather on the geological underground as they produce less biomass and thus less humic substances to keep the soil pH in a neutral range.

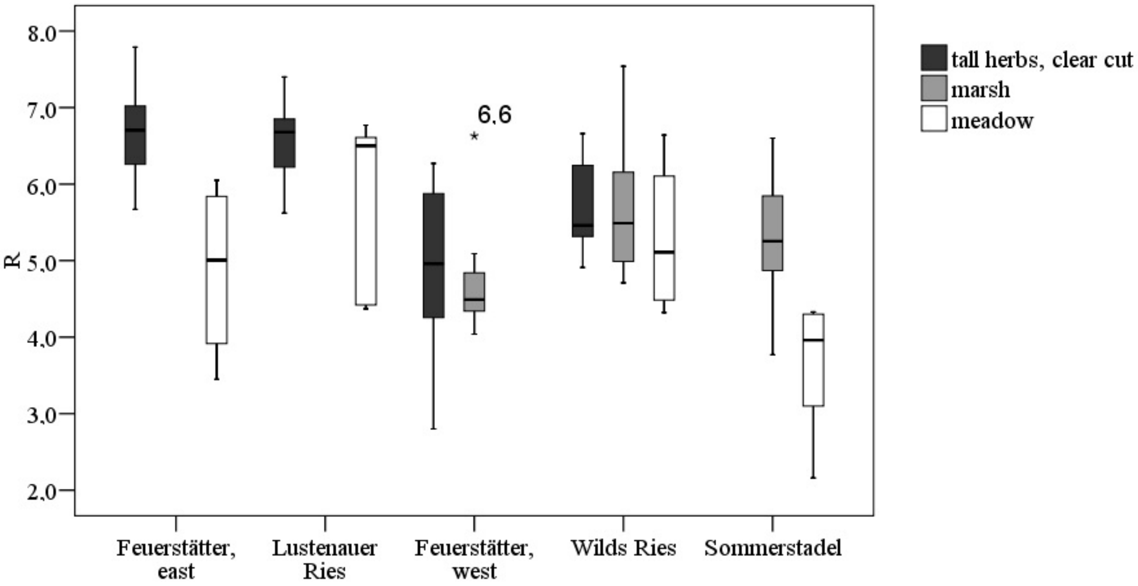


Figure 3: Boxplots of R-indicator values of the different divisions and vegetation types

Table 4: Results of the Tamhane-T2 test with respect of the R-value

Part	group 1	group 2	vegetation type	group 1	group 2
Sommerstadel $_{n=14}$	4.6 ± 1.2		meadow $_{n=19}$	4.8 ± 1.3	
Feuerstätter west $_{n=14}$	4.9 ± 1.0		marsh $_{n=22}$	5.3 ± 0.9	
Wilds Ries $_{n=11}$	5.6 ± 0.9	5.6 ± 0.9	tall herbs, clear-cutting vegetation $_{n=32}$		6.0 ± 1.1
Lustenauer Ries $_{n=5}$		5.7 ± 1.0			
Feuerstätter east $_{n=12}$		6.2 ± 1.2			

In figure 4 and table 5 the results of the indicator value N (nutrients) are displayed. Regarding the vegetation type, they could be expected. Tall herbaceous and clear-cutting vegetation store most biomass, thus leading to the highest amount of plant available nutrients. Meadows and marshes range on a significantly lower level. This corresponds to the relatively low livestock in that region. Sommerstadel has significantly lower values than the other divisions. On the one hand, this again is based on the clayey and relatively lime-reduced soil; on the other hand Sommerstadel is the division with the smallest movements. Even now, 13 years after the land slide, the Feuerstätter area, Wilds Ries and even Lustenauer Ries show clear open areas, cracks and crevices. Due to these cracks, but also due to the destroyed former vegetation cover, nutrients were released which now results in higher N-values.

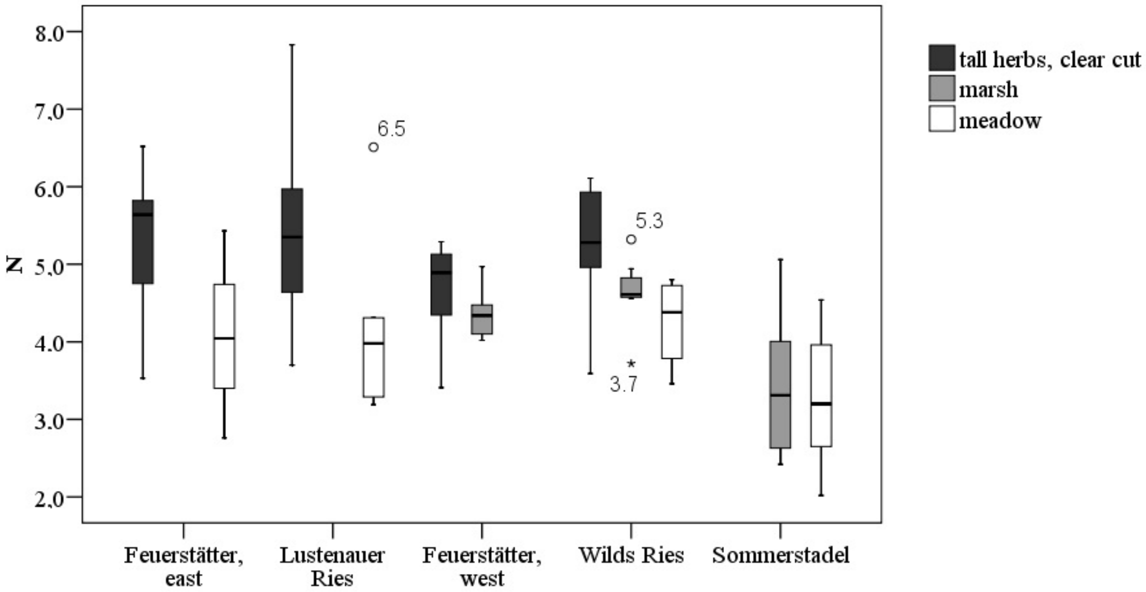


Figure 4: Boxplots of the N-indicator values of the different divisions and vegetation types

Table 5: Results of the Tamhane-T2 test with respect of the N-value

part	group 1	group 2	vegetation type	group 1	group 2
Sommerstadel $_{n=14}$	3.4 ± 0.9		meadow $_{n=19}$	3.9 ± 1.1	
Feuerstätter west $_{n=15}$		4.5 ± 0.5	marsh $_{n=22}$	4.1 ± 0.8	
Wilds Ries $_{n=18}$		4.8 ± 0.8	tall herbs, clear-cutting vegetation $_{n=32}$		5.2 ± 1.0
Feuerstätter east $_{n=14}$		4.9 ± 1.1			
Lustenauer Ries $_{n=12}$		4.9 ± 1.4			

Figure 5 and the table 6display the results for the species numbers. Tall herbaceous and clear-cutting vegetation have a higher number of species than marshes. Meadows are in-between. This can be explained by a higher heterogeneity (corresponds to the results of the diversity parameter H´).

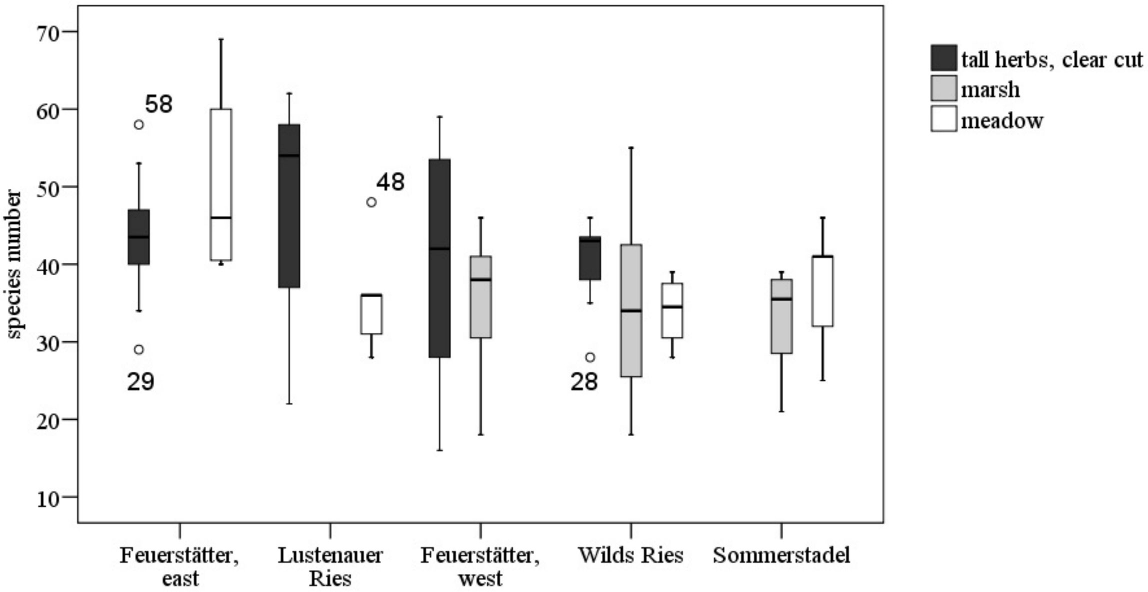


Figure 5: Boxplots of species numbers in the relevés of the different divisions and vegetation types

Table 6: Results of the Tamhane-T2 test with respect of the species number

part	group 1	group 2	vegetation type	group 1	group 2
Sommerstadel $_{n=14}$	35.2 ± 10.0		marsh $_{n=22}$	34.2 ± 10.0	
Wilds Ries $_{n=18}$	36.6 ± 9.6	36.6 ± 9.6	meadow $_{n=19}$	39.1 ± 10.0	39.1 ± 10.0
Feuerstätter west $_{n=15}$	37.8 ± 13.0	37.8 ± 13.0	tall herbs, clear-cutting vegetation $_{n=32}$		42.7 ± 11.6
Lustenauer Ries $_{n=12}$	42.3 ± 13.3	42.3 ± 13.3			
Feuerstätter east $_{n=14}$		45.5 ± 10.1			

The distributions of life traits differ significantly with respect of vegetation type. Even though the data vary strongly, tall herbaceous and clear-cutting vegetation contain more woody species ready to turn the areas into forests in the long term. Meadows have a significantly higher portion of biennial species, which could be explained by small-scale disturbances by grazing, but also by more moderate effects of the former landslide.

Table 7: Results of the Tamhane-T2 test with respect of the coverage of biennial and perennial herbaceous and woody species

biennial species:			perennial woody species:		
vegetation type	group 1	group 2	vegetation type	group 1	group 2
marsh $_{n=22}$	0.7 ± 1.0		marsh $_{n=22}$	2.4 ± 4.9	
tall herbs, clear cutting vegetation $_{n=32}$	1.6 ± 3.8		meadow $_{n=19}$	4.5 ± 7.2	
meadow $_{n=19}$		5.4 ± 5.7	tall herbs, clear-cutting vegetation $_{n=32}$		21.3 ± 21.1

perennial herbaceous species:

vegetation type	group 1	group 2	group 3
tall herbs, clear cutting vegetation $_{n=32}$	74.2 ± 20.7		
meadow $_{n=19}$		89.4 ± 9.3	
marsh $_{n=22}$			96.3 ± 5.5

Table 8: Results of the Tamhane-T2 test with respect of H' and the evenness

H': vegetation type	group 1	group 2	Evenness: vegetation type	group 1	group 2
marsh $_{n=22}$	2.2 ± 0.4		marsh $_{n=22}$	0.6 ± 0.1	
meadow $_{n=19}$		2.5 ± 0.3	tall herbs, clear cutting vegetation $_{n=32}$	0.7 ± 0.1	0.7 ± 0.1
tall herbs, clear cutting vegetation $_{n=32}$		2.5 ± 0.4	meadow $_{n=19}$		0.7 ± 0.1

Figure 6 displays the results of PCA. The relevés are displayed in the Scores Plot. Especially the vegetation type could be found in the arrangement in the Scores Plot (figure 6a). The Loadings Plot displays the influence of the Ellenberg-steps on the Principal Components. The relevant steps of the Ellenberg values for that separation are the F-value of 4, the R and N-value of 2 (figure 6b). They mainly separate the meadow relevés. Woody species (LF 4) separate mainly tall herbaceous and clear cut vegetation. However, the types are not distinguished very clearly.

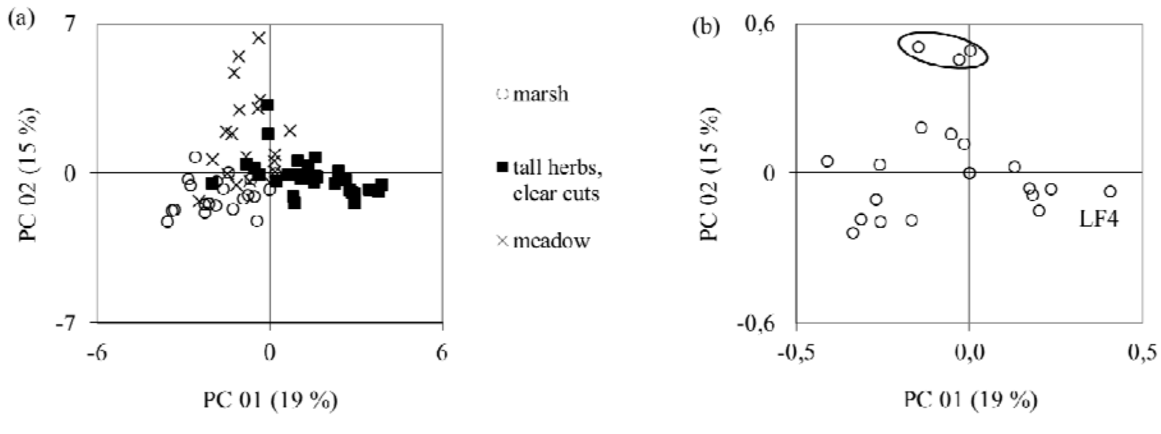


Figure 6: (a) Scores Plot of the first and the second PC of 73 relevés (b) corresponding Loadings Plot. The ellipse on the top contains the Ellenberg steps F4, R2 and N2, the utmost circle at the right (LF4) marks the woody species portion

Further evaluation dealt with Sørensen indices. The purpose of this evaluation was to determine whether differences in species lists are bigger within a certain group or among groups. The different vegetation types and the different divisions served as groups.

Figure 7 displays the heterogeneity of marshes in different parts. Interestingly, the marshes of Feuerstätter west and Sommerstadel were quite homogenous, whereas within Wilds Ries the marshes were heterogeneous, similarly as between the different divisions. All three divisions are located on soils with high clay content and a comparably low pH-value. Tall herbaceous vegetation was found in Feuerstätter west and east. The eastern division is well-endowed with carbonates. In this case the two divisions were comparably homogenous within the divisions, but the homogeneity was significantly lower between the two divisions (figure 8). Meadows were found in Sommerstadel, Feuerstätter east, Wilds Ries and Lustenauer Ries (figure 9). Within its group the relevés of Wilds Ries were homogenous. The other divisions were in tendency more homogenous than the intercrossing of different divisions. However, a distinct separation is not possible. Compared to marshes or tall herbaceous vegetation, the meadows were less different among divisions. This could be explained by the influence of a comparable amount of livestock throughout the study area, whereas the other vegetation types are more related to soil and geological underground.

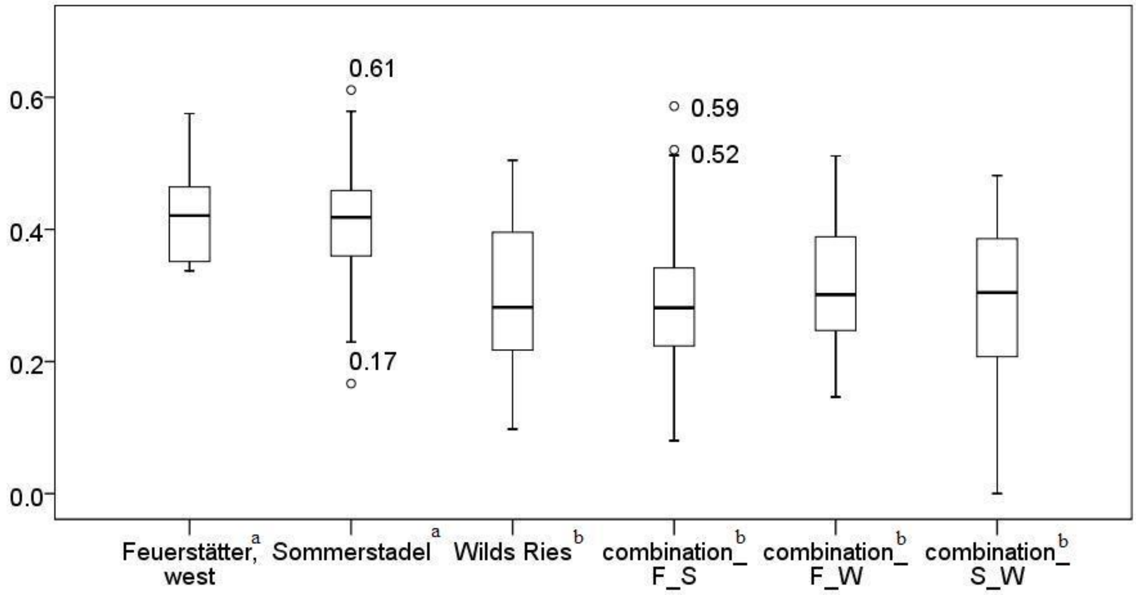


Figure 7: Sørensen indices of marshes at Feuerstätter west (F), Sommerstadel (S) and Wilds Ries (W) and between these divisions, [a, b] indicate homogenous groups according to Tukeys HSD test (α = 0.05)

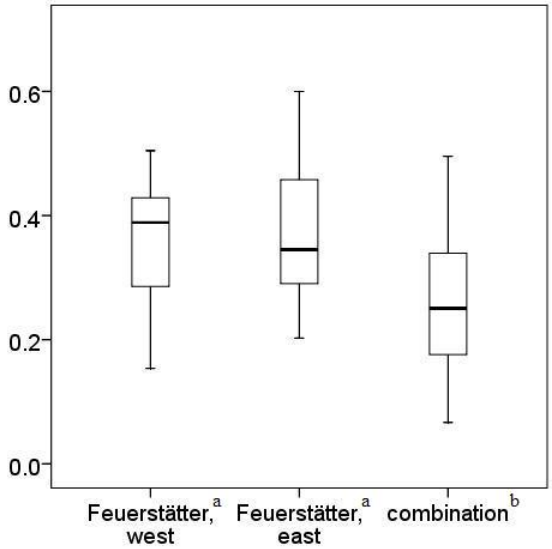


Figure 8: Sørensen indices of tall herbaceous vegetation at Feuerstätter west and Feuerstätter east and between these divisions, [a, b] indicate homogenous groups according to Tukeys HSD test (α = 0.05)

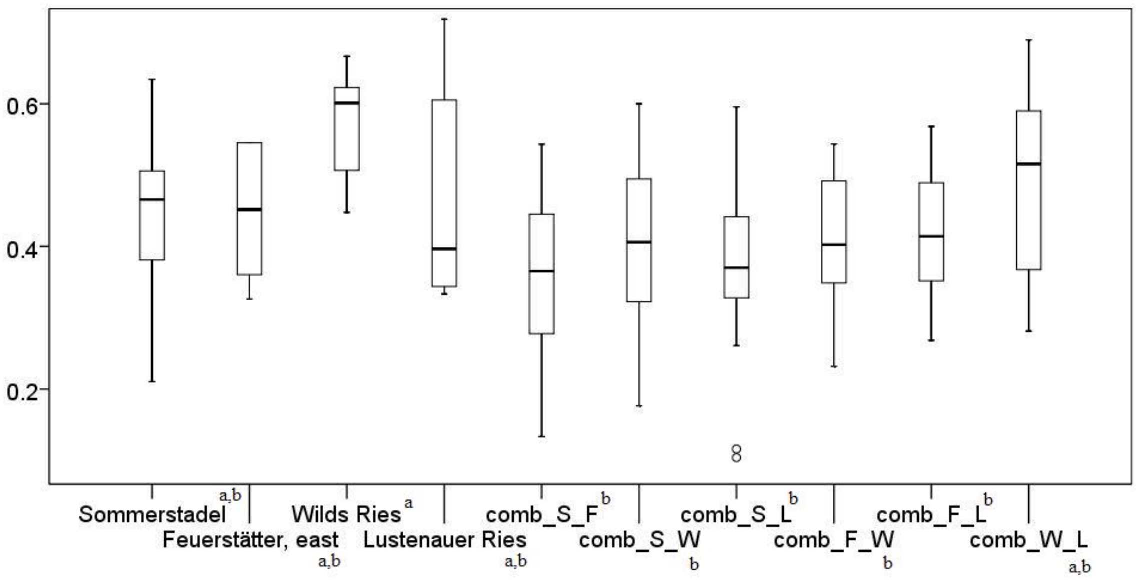


Figure 9: Sørensen indices of meadows at Sommerstadel (S), Feuerstätter east (F), Wilds Ries (W) and Lustenauer Ries (L), and between these divisions, [a, b] indicate homogenous groups according to Tukeys HSD test ($\alpha = 0.05$)

In figure 10 the indexes of the vegetation types at Feuerstätter east are shown. Especially meadows present themselves as homogenous. The crossed vegetation lists of tall herbaceous vegetation and meadows are significantly less homogeneous. At Feuerstätter west, marshes and tall herbaceous vegetation were assessed. The latter could not be significantly distinguished from the combination of the types (figure 11). At Sommerstadel, marshes and meadows occurred. Whereas the two types were homogeneous, the differences between the types were significantly bigger (figure 12). Altogether the species lists of tall herbaceous vegetation and marshes are closer to each other than to meadows.

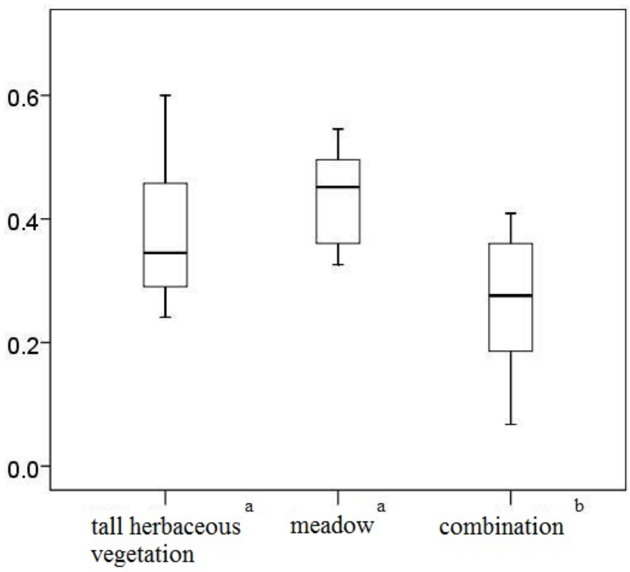


Figure 10: Sørensen indices of the relevés of the part Feuerstätter east derived from tall herbaceous vegetation, meadows, and the combination of both. [a, b] indicate homogenous groups according to Tukeys HSD test ($\alpha = 0.05$)

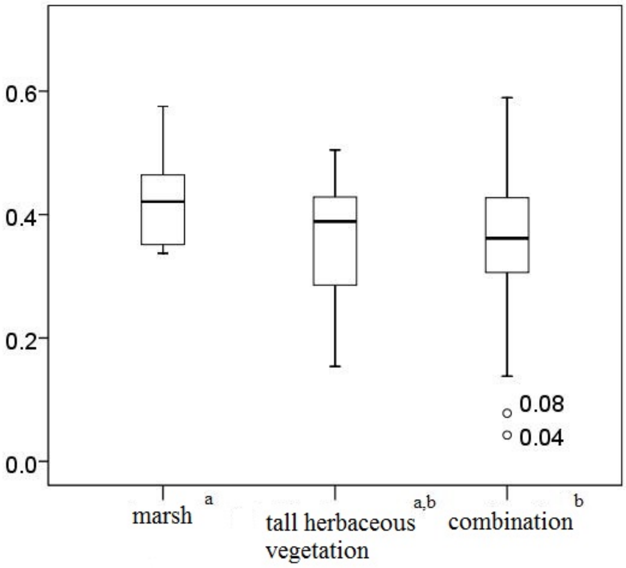


Figure 11: Sørensen indices of the relevés of Feuerstätter west derived from marshes, tall herbaceous vegetation, and the combination of both. $^{a, b}$ indicate homogenous groups according to Tukeys HSD test (α = 0.05)

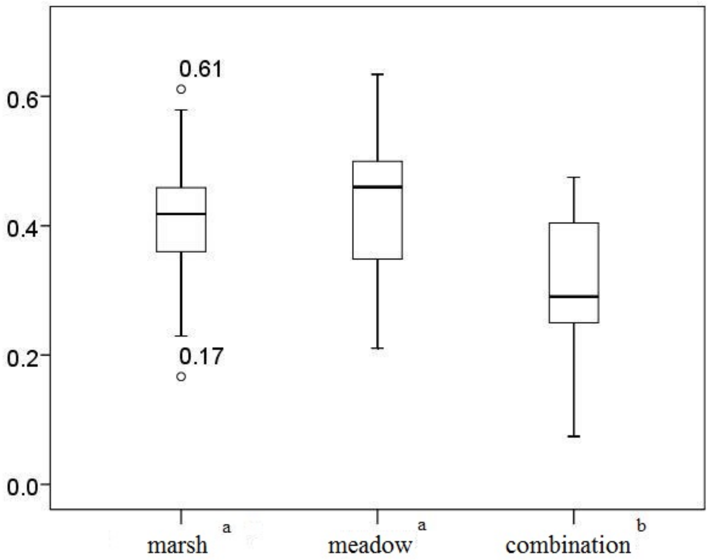


Figure 12: Sørensen indices of the relevés of Sommerstadel derived from marshes, meadows, and the combination of both, $^{a, b}$ indicate homogenous groups according to Tukeys HSD test (α = 0.05)

Soil and stone samples

The results of the soil analyses are arranged in the following tables and figures.

Figure 13 displays the grain size distributions of the collected soil samples. According to the Austrian Standards (ÖNORM L 1050), the samples from Sommerstadel are both classified as clay; at both Wilds Ries and Feuerstätter west, one sample is classified as clay and one as loamy clay. At Lustenauer Ries one sample is loamy clay and one is loam.

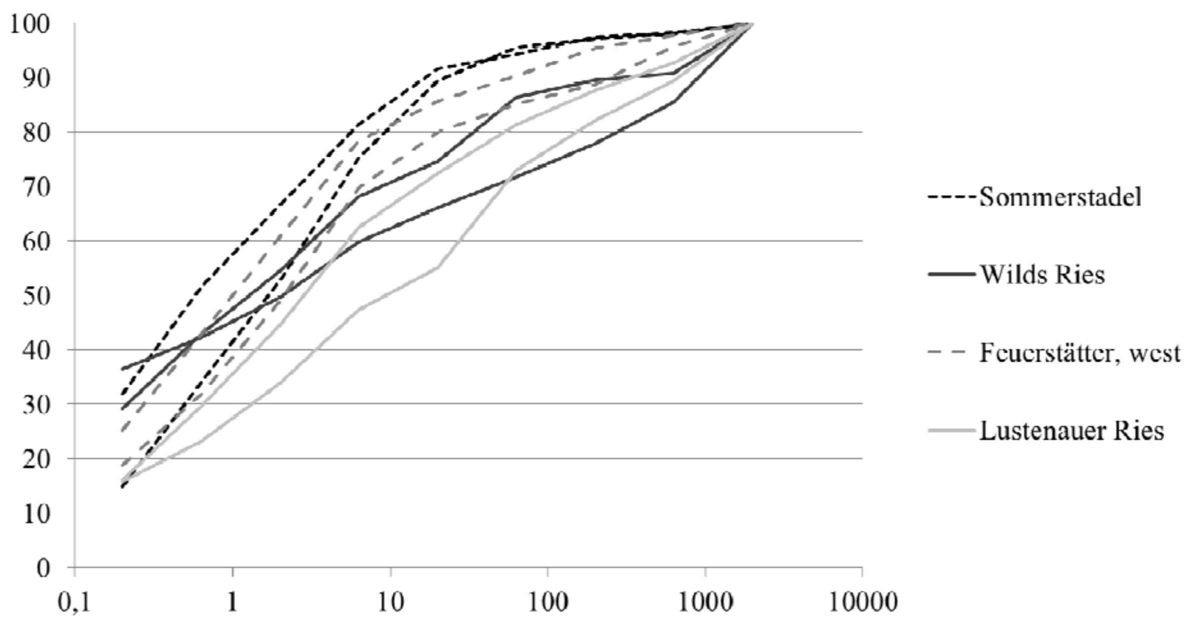
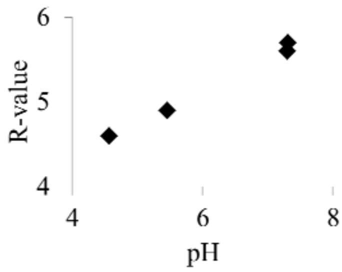


Figure 13: Grain size distributions of the eight soil samples

The following table assembles pH and calcite content of the samples.

Table 9: pH eluted 1+5 (sample + H_2O resp. $CaCl_2$), $CaCO_3$ (%) measured with the Scheibler method.

	pH $_{H_2O}$	pH $_{CaCl_2}$	$CaCO_3$
Sommerstadel	4.9	4.2	0.3
	4.9	4.2	0.2
Wilds Ries	8.1	7.2	0.5
	8.3	7.4	0.4
Feuerstätter west	5.2	4.5	0.1
	6.7	6.4	0.1
Lustenauer Ries	7.3	7.1	7.8
	8.3	7.6	7.6



The R indicator values and pH (especially from the $CaCl_2$-eluate) are highly correlated (mean values are displayed in the adjacent figure). This demonstrates the indication quality of vegetation regarding the soil pH. Interestingly, even a relatively low amount of carbonate is enough to keep the carbonate buffer working.

Conclusions

The main goal of this paper was to prove the indicative power of vegetation about the geological underground material. Meadows, marshes and tall herbaceous plus clear-cutting vegetation were separated as vegetation types, among which meadows are influenced rather by livestock than by soil and/or geology.

Small amounts of carbonate content in the soil lead to a working carbonate buffer, which is clearly indicated by vegetation. This information can be useful, if e.g. different clones of willows should be used as re-vegetation elements, which differ in their demand on soil pH.

Among all variables derived from vegetation, the Ellenberg indicator values seem to be the most powerful ones. R-value is a good indicator of soil pH and by that of the geological background of the soil material. F-value indicates indirectly the clay content, which usually is most relevant for mass movements. N-value mainly distinguishes vegetation types and land use. The differences between the groups can be quantified easily by diversity indices like e.g.

Sørensen index. Species number and also the percentage of life traits did not lead to such conclusive results.

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References

Alexander, K., P. Bloch, W. Sigl, and W. Zacher. 1964. Helvetikum und "Ultrahelvetikum" zwischen Bregenzer Ache und Subersach (Vorarlberg). Verhandlungen der Geologischen Bundesanstalt Sonderheft G:134-146.

Austrian_Standards_Institute. 1999. ÖNORM L 1084 Chemical analyses of soils - determination of carbonate, Vienna.

Bauer, W. 2000. Rutschung Sibratsgfäll in der Parzelle Rindberg. Bregenzerwald-Heft 19:26-35.

Braun-Blanquet, J. 1964. Pflanzensoziologie - Grundzüge der Vegetationskunde Springer, Vienna.

Ellenberg, H., H.E. Weber, R. Düll, V. Wirth, W. Werner, and D. Paulißen. 1992. Zeigerwerte von Pflanzen in Mitteleuropa. 2nd ed ed. Goeltze, Göttingen.

Faupl, P. 1975. Kristallinvorkommen und terrigene Sedimentgesteine in der Grestener Klippenzone (Lias - Neokom) von Ober- und Niederösterreich. Jahresberichte der Geologischen Bundesanstalt 118:1-74.

Fischer, M.A., K. Oswald, and W. Adler. 2008. Exkursionsflora für Österreich, Liechtenstein und Südtirol. 3rd ed. Biologiezentrum der Oberösterreichischen Landesmuseen.

Londo, G. 1976. The decimal scale for relevés of permanent quadrats. Vegetatio 33:61-64.

Prey, S. 1983. Das Ultrahelvetikumfenster des Gschliefgrabens südsüdöstlich von Gmunden (Oberösterreich). Jahresberichte der Geologischen Bundesanstalt 126:95-127.

Rasch, D., L.R. Verdooren, and J.I. Gowers. 1999. Fundamentals in the Design and Analysis of Experiments and Surveys R. Oldenbourg, München, Wien.

Smidt, E., J. Tintner, and K. Meissl. 2007. New Approaches of Landfill Assessment and Monitoring, p. 191-225, *In* A. A. Velinni, ed. Landfill Research Trends. NOVAPublisher.

Sørensen, T. 1948. A method of establishing groups of equal amplitude in plant sociology based on similarity of species and its application to analyses of the vegetation on Danish commons. Biologiske Skrifter / Kongelige Danske Videnskabernes Selskab 5:1-34.

Tamhane, A.C. 1977. Multiple comparisons in model I one-way ANOVA with unequal variances. Communications in Statistics - Theory and Methods 6:15-32.

M. Kühle-Weidemeier (ed.) 2006. Abfallforschungstage, Hannover. Cuvillier.

Tintner, J., and B. Klug. 2011. Can vegetation indicate landfill cover features? Flora: Morphology, Distribution, Functional Ecology of Plants 206:559-566.

Tremp, H. 2005. Aufnahme und Analyse vegetationsökologischer Daten Ulmer, Stuttgart.

Widder, R.W. 1988. Zur Stratigraphie, Fazies und Tektonik der Grestener Klippenzone zwischen Maria Neustift und Pechgraben/ OÖ. OÖ Geonachrichten 126:95-127.

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Publikationsliste

Tintner J., Smidt E., Böhm K., Matiasch L. (2012) Risk assessment of an old landfill regarding the potential of gaseous emissions - A case study based on bioindication, FT-IR spectroscopy and thermal analysis, Waste Management 32(12), 2418-2425.

Tintner J., Kühleitner M., Binner E., Brunner N., Smidt E. (2012) Modeling the final phase of landfill gas generation from long-term observations. Biodegradation 23, 407-414.

Tintner J., Klug B. (2011) Can vegetation indicate landfill cover features? Flora: Morphology, Distribution, Functional Ecology of Plants 206, 559-566.

Angermeier R., Tintner J., Smidt E., Ottner R., Matiasch L., Binner E., Böhm K. (2011) Development of mechanically-biologically treated municipal solid waste under different vegetation types. Journal of Environmental Engineering 137(5), 340-347.

Huber-Humer M., Tintner J., Böhm K., Lechner P. (2011) Scrutinizing compost properties and their impact on methane oxidation efficiency. Waste Management 31(5), 871-883.

Smidt E., Böhm K., Tintner J. (2010) Application of various statistical methods to evaluate thermo-analytical data of mechanically-biologically treated municipal solid waste. Thermochimica Acta 501(1-2), 91-97.

Tintner J., Smidt E., Böhm K., Binner E. (2010) Investigations of biological processes in Austrian MBT plants. Waste Management 30, 1903-1907.

Böhm K., Smidt E., Binner E., Schwanninger M., Tintner J., Lechner, P. (2009) Determination of MBT-waste reactivity - An infrared spectroscopic and multivariate statistical approach to identify and avoid failures of biological tests. Waste Management 30, 583–590.

Smidt E., Meissl K., Tintner J., Ottner F. (2010) Interferences of carbonate quantification in municipal solid waste incinerator (MSWI) bottom ash – Evaluation of different methods. Environmental Chemistry Letters, 8(3), 217-222.

Smidt E., Meissl K., Tintner J., Binner E. (2009) Influence of input materials and composting operation on humification of organic matter. GSB Dynamic Soil, Dynamic Plant, Special Issue 1 "Compost 1", 50-59.

Meissl K., Smidt E., Schwanninger M., Tintner J. (2008) Determination of humic acids contents in composts by means of near and mid infrared spectroscopy and partial least squares regression (PLS-R) models. Applied Spectroscopy 62/8, 873-880.

Smidt E., Meissl K., Tintner J. (2008) The influence of waste sample preparation on reproducibility of thermal data. Thermochimica Acta 468, 55-60.

Meissl K., Smidt E., Tintner J. (2008) Reproducibility of FTIR spectra of compost, municipal solid waste and landfill material. Applied Spectroscopy 62/2, 190-196.

Tintner J., Klug B. (2008) Monitoring and future use planning of a MSWI bottom ash landfill by means of vegetation ecology. Österreichische Wasser- und Abfallwirtschaft 60(7-8), 123-128.

Smidt E., Meissl K., Tintner J. (2007) Investigation of 15-year-old municipal solid waste deposit profiles by means of FTIR spectroscopy and thermal analysis. Journal of Environmental Monitoring 9, 1387-1393.

Smidt E., Tintner J. (2007) Application of differential scanning calorimetry (DSC) to evaluate the quality of compost organic matter. Thermochimica Acta 459/1-2, 87-93.

Book chapter

Böhm K., Tintner J., Smidt E. (2011) Modeled on Nature – biological processes in waste management. In Kumar S (ed.): Integrated Waste Management vol I. Intech, 153-178.

Smidt E., Tintner J., Meissl K., Binner E. (2009) Resource recovery by composting – materials, techniques, quality assessment. In: Composting Processes, Materials and Approaches (Ed.), Nova Science Publisher, New York, pp. 316.

Meissl K., Smidt E., Tintner J., Binner E. (2007) Humus - A quality criterion for composts. Infrared spectroscopy (FTIR) - A new evaluation tool and its application in practice. Lechner P. (Ed.), ISBN 978-3-900962-69-2.

Smidt E., Tintner J., Meissl K. (2007) New approaches of landfill assessment and monitoring. In: Landfill Research Trends, A.A. Velinni (Ed.), Nova Publisher, New York, pp. 191-225.