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initial ecosystem development 2005-2010

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Content

1.   Introduction

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Numerous ecological studies have been carried out over the past decades. Many of them were focused on subunits of landscapes others dealt holistically with composite landscapes of various landscape elements and land-use practices (e.g., Ellenberg et al., 1986; Fränzle et al., 2008). Based on these comprehensive approaches, networks such as the International Long-Term Ecological Research Network have been established, including a large number of monitoring sites (Waide et al., 1998). The need for such long term monitoring studies has been postulated again by Müller et al. (2010), as they provide opportunities for investigating the relationships between different processes and patterns within the system, and to better predict their future development (Lovett et al., 2007). Recently, several additional networks have emerged in this context such as the Critical Zone Exploration Network (Anderson et al., 2008), the Biodiversity Exploratories (Fischer et al., 2010) or the Terrestrial Environmental Observatoria (www.tereno.net).

Most of these ecosystem research approaches, however, are dealing with mature ecosystems. The analysis of young ecosystems in their initial stages of development might be a way to disentangle the complex process web (Schaaf et al., 2011). In contrast to the high complexity of mature natural systems, ecosystems are initially less complex and less interlinked because only a few initial controls are interacting at this stage (Jørgensen et al., 2000). Several studies have been carried out on initial ecosystems, e.g., investigations at Surtsey Island near Iceland or the research conducted following the eruption of Mount St. Helens (USA). But generally, only limited knowledge exists on the very initial phase of ecosystem development (Bardgett & Wardle, 2010) although early events, processes and structures cause imprints on the system, which influence the development at later stages (Walker & del Moral, 2003).

Fath et al. (2004) suggested a model of ecosystem development based on the assumption that ecosystem succession can be interpreted as a series of four distinct growth and development phases: (i) 'boundary growth', (ii) 'structural growth', (iii) 'network growth', and (iv) 'information growth'. Thus, a continued input of resources is necessary to maintain the system in phase (i). During early succession, biomass will increase in the second phase

followed by a growth in connectivity of the system in phase (iii), e.g., as a result of developing food webs. The fourth phase is suggested as the change of system behavior from exploitative patterns to more conservative, energetically more efficient patterns. This becomes evident by an increasing complexity during ecosystem development as additional patterns and processes emerge.

Water and element cycling play a key role in the functioning and stability of ecosystems. Their analysis requires clearly defined system boundaries, which is typically approached by using natural catchments (Likens, 1989). Theoretically, a catchment offers the opportunity to quantify input and output parameters of a system, as well as the possibility to quantify the water and element budgets (Kirchner, 2009). In reality, catchments often cannot be regarded as clearly defined sub-units of a landscape, but have to be considered at least in part as 'black boxes". Also research on catchments has been mainly conducted in mature or at least non-initial ecosystems. Important internal interdependencies between the different underlying physical, chemical and biological structures with the surface properties or with the hydrological behavior are still not well understood (Kendall et al., 2001).

To overcome these limitations of naturally developed ecosystems, artificially created systems might be an alternative (Eckhardt et al., 2003). However, due to the challenges of constructing an artificial catchment most of these approaches are restricted to smaller dimensions. For example 'Hydrohill' in China has a size of 490 m² (Kendall et al., 2001). Moreover, interactions of different abiotic and biotic ecosystem compartments have often been excluded in such studies. Recently, the Biosphere 2 Hillslope Experiment has been initiated near Tucson, Arizona (USA). It aims at exploring how climate, soil and vegetation interact and drive the evolution of the hydrologic hillslope behavior (Hopp et al., 2009). Artificial catchments can also be a great opportunity for holistic ecosystem approaches, especially if above- and below-ground interactions are studied. One of the most important advantages of using artificial catchments for scientific purposes is the fact that the boundaries and inner structures of the system can be planned and defined in advance. Hence, an artificial catchment can be an important tool for validating, optimizing and developing model approaches.

Within the post-mining landscape Welzow-Süd (Brandenburg, Germany), the artificially created catchment 'Chicken Creek' was constructed in 2004/2005 in close cooperation of the Brandenburg University of Technology at Cottbus (BTU) and the Vattenfall Europe Mining AG (VEM), to analyze initial structures and processes in a well-defined system. 'Chicken Creek' catchment enables the study of interactions and feedbacks between the developing structures and related processes under clearly defined boundary conditions. The research approach places special emphasis on the spatial and temporal dynamics of both structures and processes – above- and below-ground. Furthermore, the study of the gradual development of this ecosystem is aimed to better understand the behavior of more mature systems.

Volume 1 of this series (Gerwin et al., 2010) gives detailed insight into the construction process and initial properties of the 'Chicken Creek' catchment. Generally, the site can be divided into three major sections: (i) the backslope area, (ii) the footslope, and (iii) the pond basin. The site is completely fenced to avoid disturbances by human visitors or game animals. In November 2009, the surface boundary of the catchment was adjusted to the subsurface boundary in the course of construction works of the adjacent hill 'Wolkenberg' (Fig. 1.1).
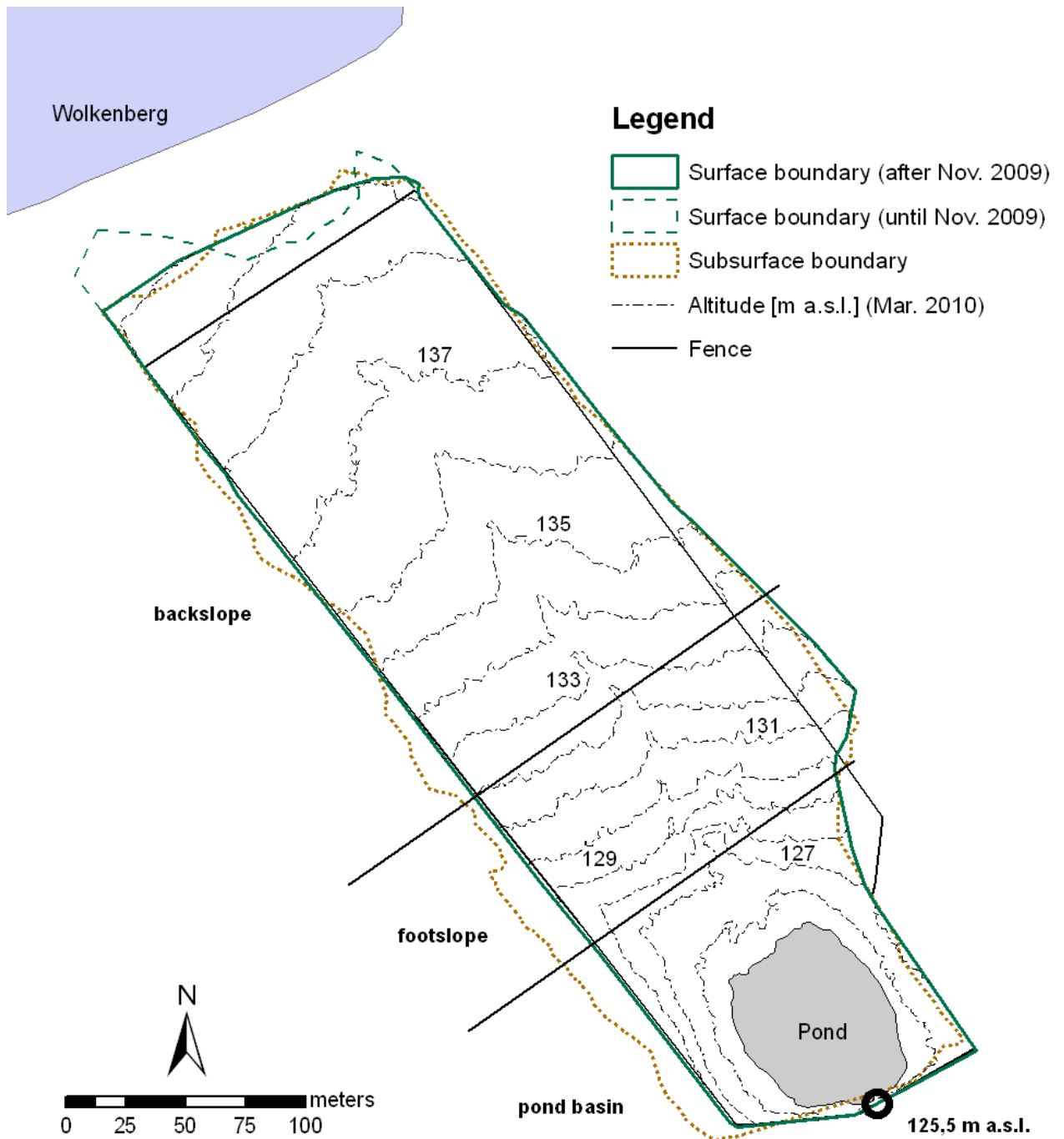

Fig. 1.1:   General structure of the 'Chicken Creek' catchment.

Basic monitoring of the 'Chicken Creek' catchment started immediately after the end of construction works in 2005. Monitoring of environmental parameters is carried out to measure meteorological, hydrologic and soil conditions as well as the surface development. In addition, specific monitoring programs are carried out including vegetation and soil fauna as well as limnological parameters within the pond (Fig. 1.2). A first comprehensive report on monitoring activities and results gathered from 2005 to 2008 is given in the second volume of this series (Schaaf et al., 2010).
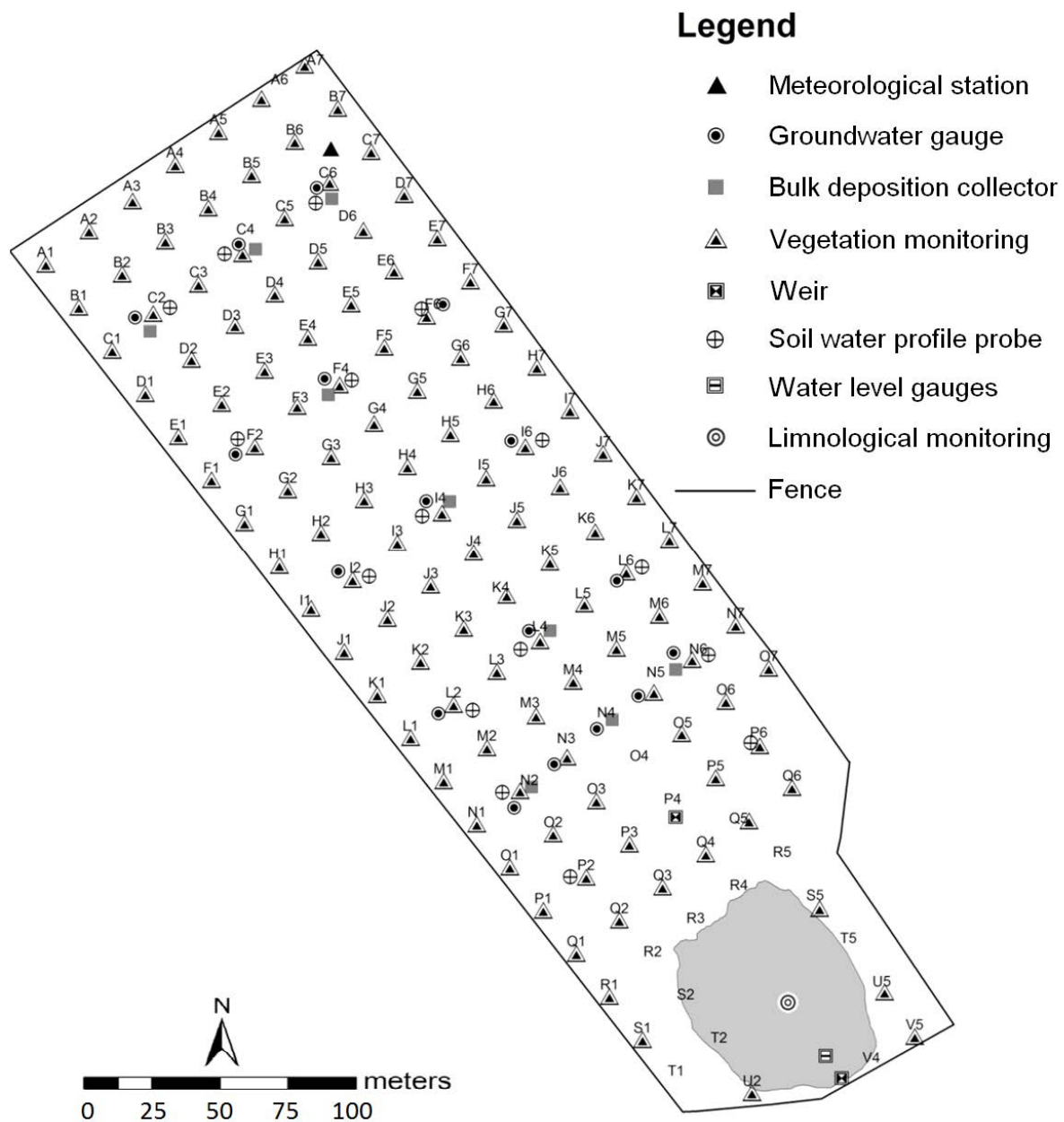

Fig. 1.2a: Grid based monitoring installations at the 'Chicken Creek' catchment.

As soil sampling in 2005 proved the Quaternary substrate to be relatively homogenous both spatially and with depth (Gerwin et al., 2010), all further installations were oriented along a basic 20 m x 20 m grid (Fig. 1.2a). Due to the evolution of patterns within the catchment in the following years, these installations were successively complemented to better represent the differentiating site conditions (Fig. 1.2b).
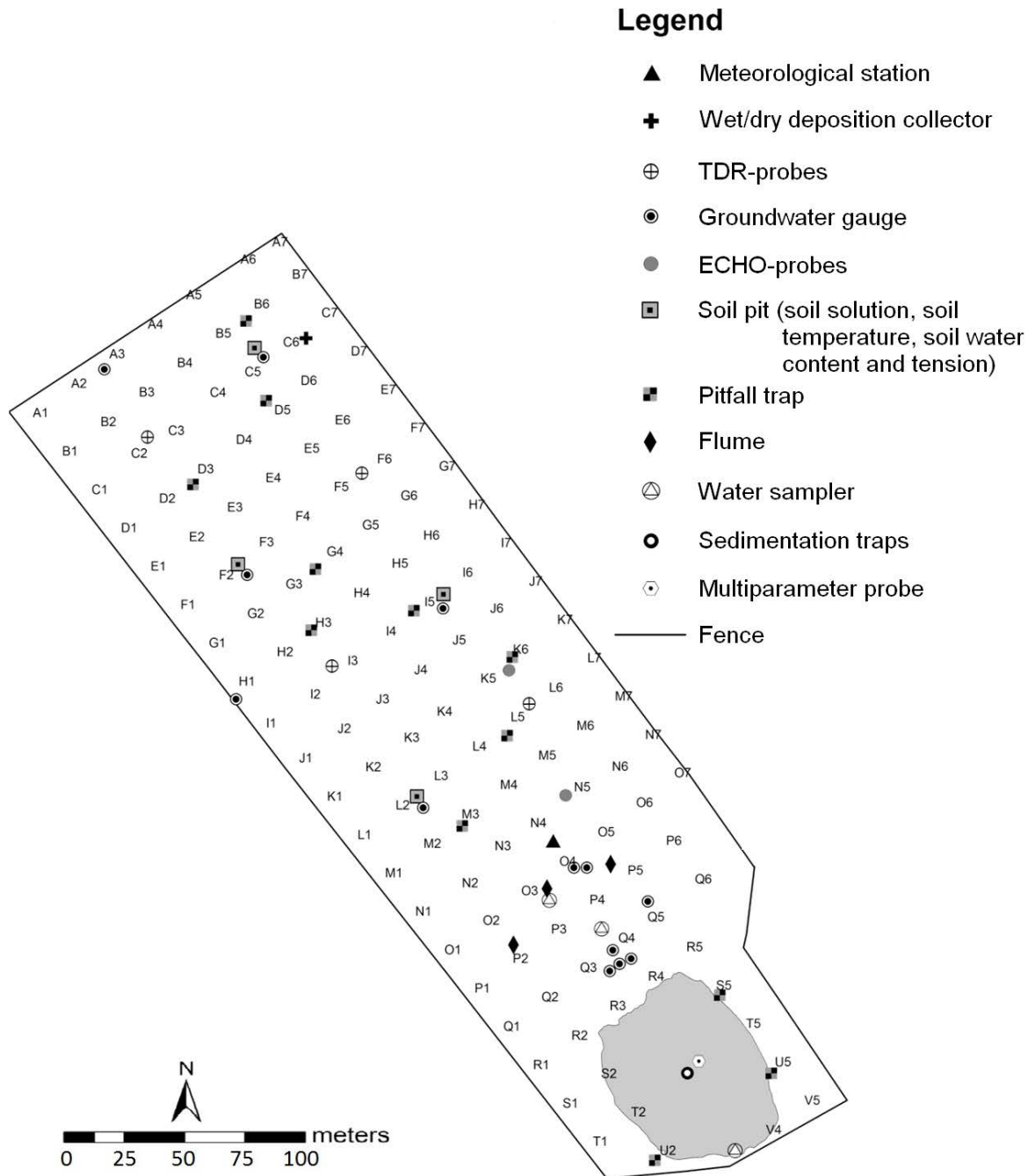

Fig. 1.2b: Complementary monitoring installations at the 'Chicken Creek' catchment.

E.g., initial streams appeared as the consequence of heavy rainfall events in 2005 and 2006. Hence, flumes for measuring flow rates were installed in the main erosion rills. Further, due to unexpected heterogeneities in the inner structure of the catchment, the initial network of groundwater gauges was completed on the basis of modeling results. Other installations were added to answer questions arising from the observed structural development of the site: ECHO probes were installed to investigate possible differences in soil water content between vegetated and still bare patches in the eastern part of the catchment (Fig. 1.2b). Several individuals of Black locust had to be removed adjacent to the northern weather station and to paths in 2010 as they hampered measurements and field work.

For the monitoring of surface structure development, aerial photos were taken from a helicopter in 2006 and 2007. Since 2008 a micro-drone equipped with a digital camera is used. Additional aerial photographs taken regularly since the start of construction works in 2004 are provided by VEM (Fig. 1.3). Furthermore, aerial images were processed by automated digital photogrammetry to obtain Digital Elevation Models during routine surveys of VEM. Finally, high-resolution data were recorded in September 2010 resulting in a Digital Surface Model and a Digital Terrain Model (cf. chapter 2).

Altogether, two weather stations are operated and groundwater levels are recorded at 30 locations within the site. Two main weirs were installed to measure groundwater runoff and total runoff: At the outlet of the sub-surface clay walls above the pond, a weir was installed during the construction of the site. Another weir was placed at the outlet of the pond representing the total discharge point for the whole catchment. In three main erosion gullies flumes are installed to register both the continuous base flow from groundwater discharge and episodic run-off events (cf. chapter 3).

Both weirs and one of the flumes are equipped with automated water sampling units taking daily samples for water quality analyses. Soil solution is sampled and soil moisture conditions are determined at four soil funnels that were excavated down to the saturated layer depth. Soil moisture is additionally monitored by means of several permanently installed pipes. As atmospheric deposition might be an important source for nutrient input, bulk deposition is collected at 18 locations within the catchment (cf. chapter 4).

The amount of viable seeds in the substrate used for the catchment construction was analysed in 108 soil samples taken in autumn 2005. Moreover, the transport of plant seeds by wind was assessed using 55 seed rain traps. Since autumn 2005, the initial vegetation development of the study site has been recorded annually at 119 grid points (minimum). In 2005, the vegetation was assessed directly after finishing the catchment area construction works. In the following years, vegetation records have been repeated regularly in summer (cf. chapter 5).
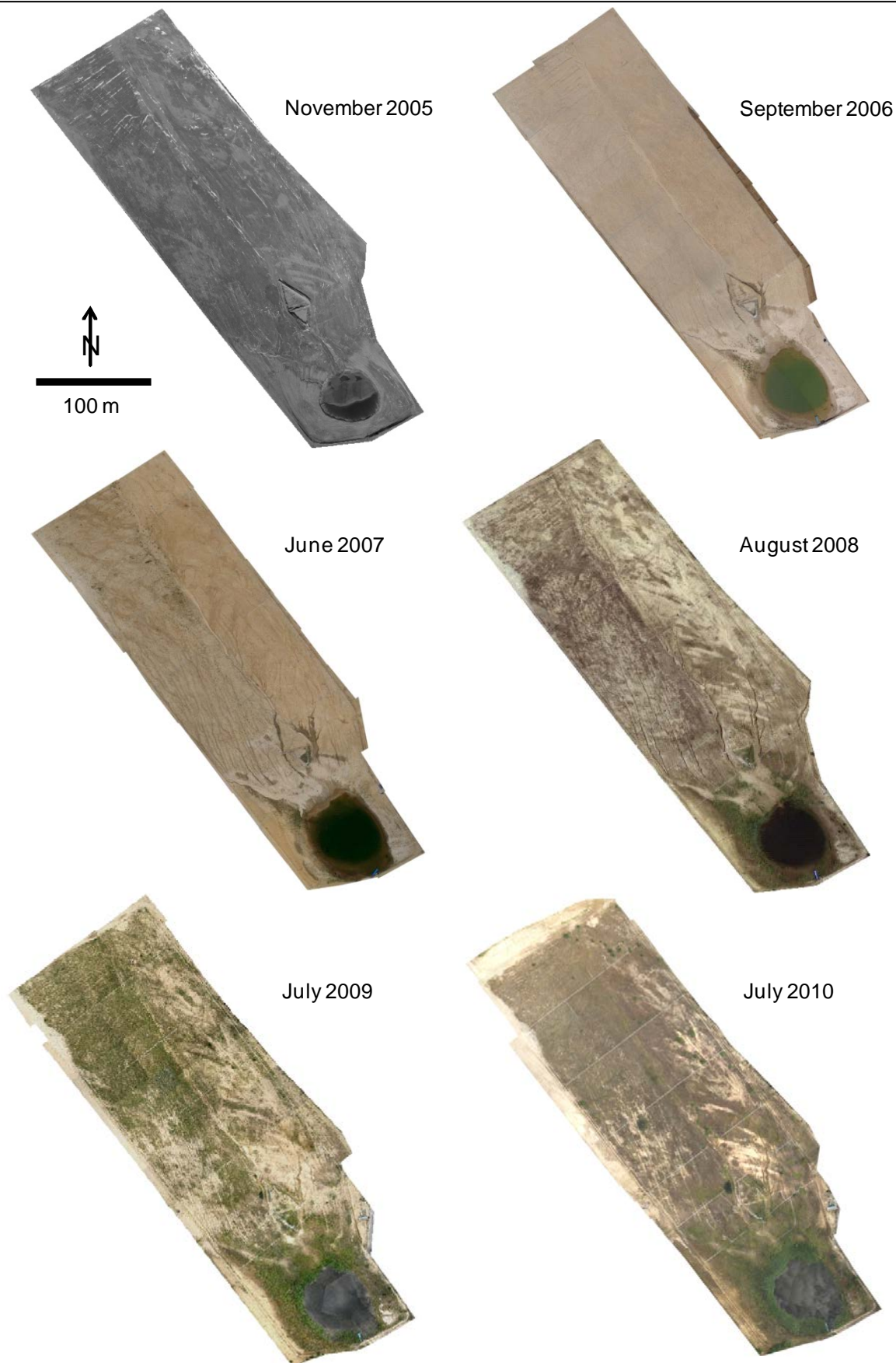
Fig. 1.3: Aerial photos of the 'Chicken Creek' catchment provided by Vattenfall Europe Mining AG (2005, 2008) as well as based on helicopter (2006, 2007) and microdrone flights (2009, 2010).

The development of the soil food web during the initial stage of primary succession was investigated as well. Before regular zoological sampling was begun, two preliminary sampling campaigns were undertaken directly after the end of construction works in October 2005 and in May 2006. Regular sampling began in 2007 at typical times of peak abundances, namely in April and October of each year (54 samples each) (cf. chapter 6).

The limnological studies on 'Chicken Creek' pond started in 2005 as occasional sampling and have been intensified since February 2008. Since then, various physical, chemical and biological parameters have been recorded (cf. chapter 7). Moreover, the extent of the accumulated sediment layer, its chemical characteristics, and the effect of decomposing plants on phosphorus mobilization in the pond have been analysed (cf. chapter 8).

This volume summarizes the monitoring activities and results at the 'Chicken Creek' catchment for the period 2005 to 2010. The development in all ecosystem compartments mentioned above is assessed, classified, and compared to more mature systems with regard to functional relevance and succession stage. In a final synopsis, the whole catchment is evaluated with regard to ecosystem development and an outlook is given on the expected trends both in the short and the medium term (cf. chapter 9).

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2. Aspects of initial surface development

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2.1 Introduction

The morphology of a land surface as well as the spatial distribution of sediment properties are critical initial conditions in ecosystem development. Newly exposed or deposited sediments are initially relatively instable and can undergo rapid alteration regarding their surface geometry and internal structures due to erosion, deposition and other physical and biochemical transformation processes.

The first processes of sediment erosion by water on recently exposed material usually are rain splash and sheet wash (Govers & Poesen, 1988), causing spatially varying sediment redistribution over comparably short distances and most parts of a hillslope. A major threshold in erosion-affected surface development is the beginning of the incision of erosion rills by concentrated surface runoff, which initiates further rill network growth, stream channel development and increased erosion rates (Horton, 1945). Depending on climatic conditions, and on the soil texture and structure at the surface, wind erosion can contribute to the overall erosion (Breshears et al., 2003), sometimes even exceeding the mass movement rates of water erosion (Visser et al., 2005). The impact of aeolian processes is especially significant during the very first phase of surface development. While sediments transported by water mostly remain in the catchment, the aeolian transport involves sediment exchange with the surrounding areas. The interaction between wind and water may as well play a role during the initial ecosystem development; however, erosion modeling can currently hardly differentiate between these two forms of erosion (e.g., Flanagan & Visser, 2004).

A variety of ecosystem processes are affected by developing surface structures. Surface geometry can influence the spatial distribution of soil moisture and groundwater levels, and the evolution of flow paths, the velocity of surface runoff as well as the movement of solid particles by gravity are highly dependent on morphology (Moore et al., 1991). Developing erosion rills and stream channels can serve as landscape corridors and thus accelerate the transport of water, solutes and sediment through the system (Bracken & Croke, 2007).

Furthermore, the comprehensive effects of denudation, deflation and aeolian deposition can alter mineralogical compositions (Chappell et al., 2005) of the initial surface through selective in- and output of particles. The modified particle and mineral composition can change chemical compositions and element budgets (Sterk et al., 1996) and the physical conditions (Herrmann et al., 1994) of the uppermost soil layer. Such initial relocation processes, together with the primary heterogeneity of the original material again affect soil formation processes (Birkeland, 1984). Pedodiversity is promoted when erosion and deposition hinder the proceeding of soil development through exposure or application of unweathered material, so that a mosaic of areas at different development stages is created. Erosion rills and ephemeral stream channels are preferred colonization spots for biota (Molina et al., 2009), while structures like sand patches emerge from aeolian translocation, which may affect the development of vegetation patterns (Rietkerk et al., 2002). The differentiation of morphology and sediment particle composition is thus critical for the development of diversity in an ecosystem.

Consistent with the above studies, both surface morphology and surface sediment composition were found to be rapidly altered in the 'Chicken Creek' catchment by erosion and deposition. The most obvious effect was the development of a widespread network of erosion rills and deeply incised gullies. Furthermore, field surveys indicated that wind erosion and aeolian deposition had a considerable effect on surface structures and sediment redistribution (Maurer & Gerke, 2011). This chapter aims at the reconstruction and description of surface development and the underlying processes for the first five years of catchment evolution. Considering the complexity of initial surface structural development, the focus is on the impact of physical processes on surface development at the catchment scale. An integration of results from soil surface evolution analyses using multi-temporal analysis of aerial imagery and digital elevation data (Schneider et al., 2011) and multi-scenario modeling of wind erosion (Maurer & Gerke, 2011) is presented.

2.2 Materials and methods

2.2.1 Characterization of surface development from DEMs and aerial photographs

Several states of evolution of the catchment surface have been recorded in aerial photographs and digital elevation data:

- High resolution aerial photographs were recorded in survey flights using a helicopter or a microdrone and, after referencing and rectification, composed to aerial image mosaics (Veste et al., 2010). Aerial photographs were taken on September 22nd, 2006, June 14th, 2007, July 10th, 2008, and July 1st, 2009.

- Aerial photographs of the catchment surface were recorded and processed by automated digital photogrammetry to obtain digital elevation models (DEMs) during routine photogrammetric surveys in the mine surveying department of VEM AG. Based on given camera parameters and flying height, Dominik (2007) computed a vertical accuracy σ_Z of 0.147 m for these DEMs. Elevation datasets for ten states of surface evolution between November 26th, 2005 and March 4th, 2010 were provided, however two of these elevation datasets covered only parts of the catchment area. Quality of the elevation datasets was evaluated by critical comparison with other datasets and errors were diminished by referencing to d-GPS data (Schneider et al., 2012).

- On September 8th, 2010, a high-resolution DEM was recorded using airborne LIDAR by MILAN Geoservice GmbH, Spremberg. As multiple-return LIDAR data were recorded, two datasets resulting from this measurement were provided: a Digital Surface Model (DSM), depicting the surface of vegetation cover in the catchment, and a Digital Terrain Model (DTM), depicting, with a few exceptions in areas with particularly dense vegetation cover, the ground surface. A vertical accuracy σ_Z of 0.05 m to 0.07 m was given for these elevation models.

Based on these datasets, the structural and morphological evolution of the surface caused by processes of erosion and deposition was reconstructed following a combination of approaches:

- For the description of the development of surface geometry, combined analyses of DEM and high-resolution aerial photograph time series were carried out in the 3D-environment of the software GOCAD Suite 2.5.2 (Paradigm Ltd., George Town, Cayman Islands).

- For a reconstruction of change in sediment volume between two states of development, gridded 3D models of volume change were constructed from pairs of photogrammetry-based DEMs. Models were constructed using GOCAD as follows: The vertical distance, dz, between the surface elevations of two DEMs (Z^t and Z^{t+1}):

  i) $\quad dz_i = Z_i^t - Z_i^{t+1}$, $i = 1, \ldots n$

  was computed for all n nodes of the surface models. Based on elevations of the older DEM and the distances dz_i, 3D volume bodies with cells of 1 m by 1 m size were constructed and the spatially distributed difference volume [$m^3\ m^{-2}$] was calculated. Direction of change was obtained by vertical projection of dz_i from the surface models. To improve comparability of the models, rates of volume change per year [$m^3\ m^{-2}\ a^{-1}$] for each model cell were calculated from the volume differences and length of the time intervals defined by the pair of DEMs.

- In order to obtain a high-resolution model of elevation change during the period of development from November 2005 until September 2010, the photogrammetry-based DEM of the surface in November 2005 and the high-resolution ALS-based DEM for September 2010 were used to construct a differential model, which depicts the change in surface elevation in this time interval for each node of the September 2010 DEM.

- In order to assess uncertainties in the models of elevation and volume change, the standard deviation in elevation change σ_c was obtained by (Lane et al., 2003):

  ii) $\sigma_c = \sqrt{\sigma_1^2 + \sigma_2^2}$,

  where σ_1 and σ_2 are the general accuracy of the two elevation models defining the time interval of development depicted in the model of change. Areas with elevation or volume change higher than a minimum amount of change that can be detected with a confidence of 68%, following Lane et al. (2003), were then defined in the models (cf. Schneider et al., 2011).

- Based on the DEM of November 2010, morphometric parameters for the developed surface were computed in SAGA GIS (SAGA USER GROUP Association, Göttingen). Local slopes (β) were derived using the method of Zevenbergen and Thorne (1987). The specific catchment areas (*SCA*) of DEM cells, i.e., the number of model cells from that surface runoff is routed to one specific cell, was derived using the Multiple Flow Direction Algorithm *MFD* (Freeman, 1991) for a representation of flow accumulation on the catchment surface. Based on local slopes and the *SCA*, the Stream Power Index *SPI* (Moore et al., 1991) and the Topographic Wetness Index *TWI* (Beven & Kirkby, 1979) were calculated:

  iii) $SPI = SCA \cdot \tan(\beta)$,

  iv) $TWI = \ln \dfrac{SCA}{\tan \beta}$.

  High values of the *SPI* reflect areas where the potential erosive power of surface flow, as determined by morphology, is high. High values of the *TWI* indicate areas of morphologically determined flow accumulation and have been correlated with, e.g., potentially high soil moisture contents, water tables, evapotranspiration or plant cover (Florinsky, 1998).

2.2.2 Multi-scenario modeling of wind erosion

The effects of wind erosion are often quantified using modeling approaches. If no direct measurements were carried out, as is the case with 'Chicken Creek', simulation results can be validated by remote sensing data (e.g., Quincey et al., 2007; de Vente et al., 2009). For modeling wind erosion on the 'Chicken Creek' catchment, we used the PCRaster version

(Visser et al., 2005) of the standalone erosion submodel of the Wind Erosion Prediction System (WEPS), version 1.0 (Wagner, 2002). Required WEPS input parameters are soil particle distribution, the strength and distribution of surface crusts, the amount and distribution of available loose material on the crusts, the amount of clods/rocks on the surface, the quantification and spatial distribution of surface roughness and aggregate density. While initial soil particle distributions are known from the 20 m · 20 m raster soil sampling, the well-defined constructed initial surface conditions facilitated the estimation of remaining surface parameters. The area of loose sediment cover, e.g., was estimated from surface photos. The parameter surface roughness is based on wind tunnel measurements on similar surfaces, and crust strengths and initial amount of erodible material were estimated by assuming analogies with surfaces in semi-arid regions. The boundary conditions, namely wind speed data from the onsite weather station, were implemented as WEPS time series. The effect of soil moisture, which can inhibit particle entrainment by wind, was included by cutting out the predominantly moist periods from the time series. To shorten computation time, periods with wind speeds below erosion threshold were also cut out from the time line. In WEPS, topographic elements like slopes and gullies are not considered.

A number of uncertain input parameters and boundary conditions had to be considered in the simulations. To cover the range of uncertainties, the following sets of scenarios were selected:

- Scenarios accounting for uncertainties in surface properties. Input parameters related to surface properties were derived from surface photographs and field observations. Since these estimations included uncertainties, scenarios were selected for low, average, and high wind erosion susceptibilities.

- Scenarios assuming sediment influx of erodible environments. External inputs of aeolian sediment must be considered as a possibility, since the surrounding sand surfaces were initially only scarcely vegetated and high wind speeds were common. In these scenarios, we assumed that the isolated position of the catchment had no effect on sediment transit and deposition.

- Scenarios evaluating sensitivity for texture distribution patterns with increasing heterogeneity (Fig. 2.1). Based on the initial soil raster measurements, four texture distribution scenarios were used to evaluate the effect of increasingly heterogeneous texture distribution on simulated aeolian mass transport. Texture distribution was assumed to be (i) homogeneous over the whole catchment area, (ii) partitioned in two homogeneous parts, (iii) partitioned in four parts, and (iv) heterogeneously distributed according to ordinary kriging interpolation (linear variogram model).

- The four texture distribution patterns were applied for each scenario, which added up to twelve erosion susceptibility scenarios (3 susceptibilities · 4 texture patterns) plus eight scenarios considering external influence (2 erodible surroundings · 4 texture

patterns), resulting in a total of twenty scenarios. This number was deemed to be sufficient for the determination of upper and lower limits of potential aeolian sediment translocation.
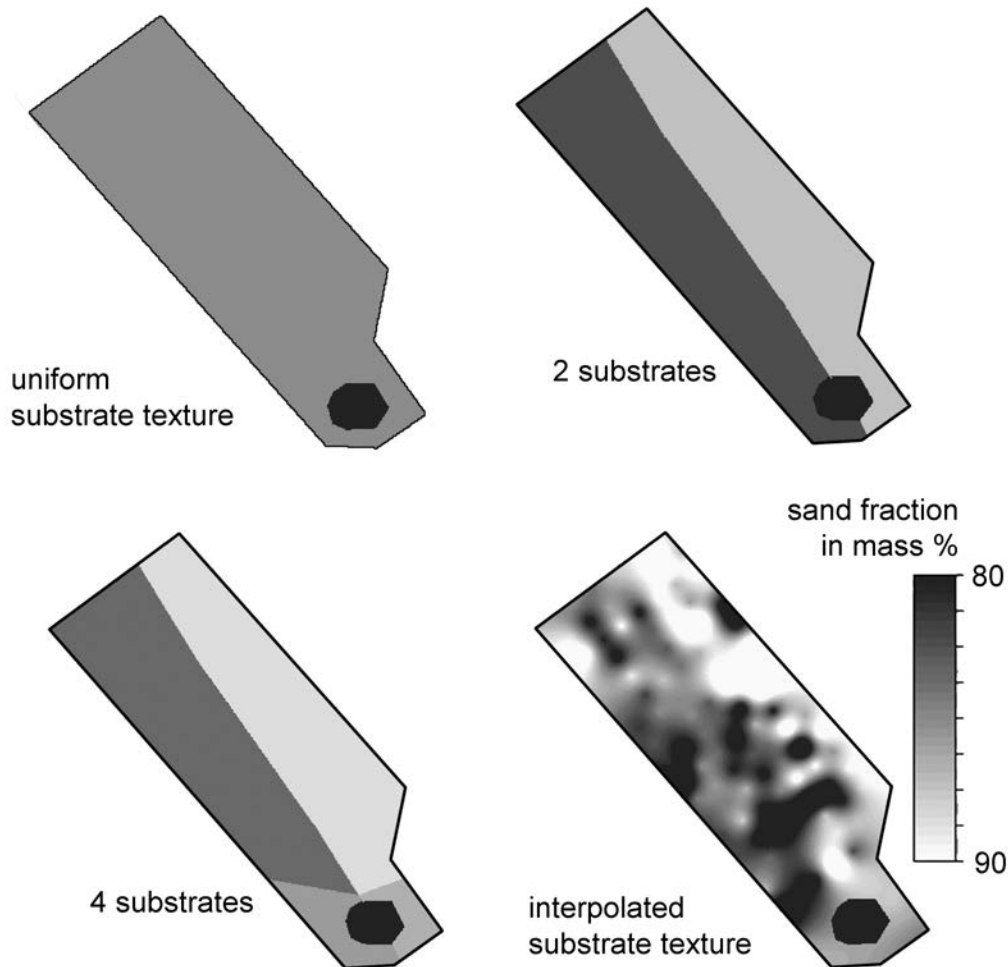

Fig. 2.1:   Some input parameters for wind erosion modeling were implemented in the form of raster maps. In this example, the spatial distribution of the sand fraction is given for the four texture distribution sub-scenarios (Maurer & Gerke, 2011).

The supervised classification of an aerial photograph (taken on a helicopter flight in June 2007 using a Canon EOS 400D digital camera; cf. Veste et al., 2010) was evaluated using ground truth information from surface photographs. For example, bright patches on the aerial image were identified as sand accumulations on surface photographs, e.g., surfaces showing wind ripples. We assumed that such sand accumulations outside of the fluvial drainage network, are exclusively of aeolian origin. Based on signature definition, each pixel of the aerial image was assigned to a surface class using the maximum likelihood algorithm. Information of class area coverage was combined with estimated aeolian sediment thicknesses derived from surface photographs.

2.3 Results and discussion

2.3.1 Characterization of surface development from DEMs and aerial photographs

Digital elevation models of the catchment surface and aerial photographs reflect the increasing differentiation of surface morphology (Fig. 2.2).
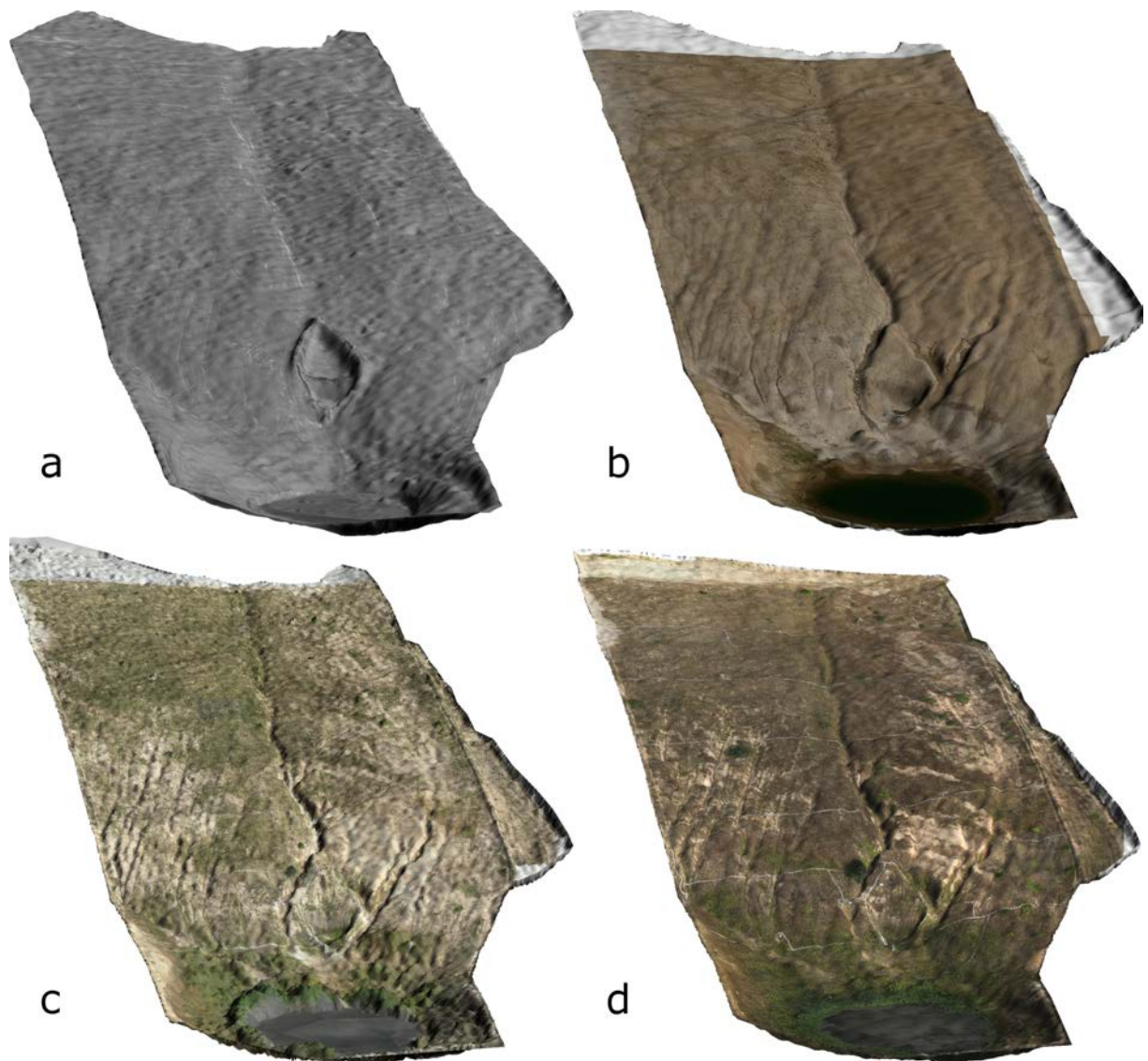
Fig. 2.2: Photogrammetry-based DEMs of the catchment surface, overlaid with aerial photographs for visualization of surface structures; a) DEM for November 26th, 2005, black and white aerial photograph recorded on November 26th, 2005 (© VEM AG); b) DEM of November 21st, 2007, helicopter-based aerial photograph mosaic recorded on June 14th, 2007; c) DEM of June 13th, 2009, drone-based aerial photograph mosaic recorded on July 1st, 2009; d) DEM of March 4th, 2010, drone-based aerial photograph mosaic recorded on July 27th, 2010.

In the DEM depicting the almost initial state as of November 2005, the surface is represented as relatively smooth with spatially distributed variations which are most probably due to the general inaccuracy of the elevation data. In the DEM for November 2007, the effects of erosion rill incision in the central and lower slope areas are apparent. In the southwestern (SW) part of the slope, several small and parallel erosion rills are observable, while in the northeastern (NE) part one major rill running towards the trapezoidal spring area is visible. The aerial photograph mosaic shows that vegetation cover was beginning to develop in the alluvial fan areas. The DEMs and aerial photograph mosaics of 2009 and 2010 reflect that patterns of the erosion rill network remained relatively stable during further surface development. The major erosion rills appear to be more clearly incised; however no further headward growth of the rills from 2007 on can be observed.

The rapid growth of vegetation cover is clearly observable from the aerial photographs. Figure 2.3 gives the spatial distribution of rates of change in sediment volume [m^3 m^{-2} $year^{-1}$] for six time intervals, as derived from models of volume change. A standard deviation in volume change of $\sigma_v = 0.208$ m was derived from the general accuracy of the input DEM, and additional systematic errors or artefacts might further limit the quality of the differential model. Thus, the small amounts of change in sediment volume occurring during the comparably short time intervals could not be sufficiently quantified. However, the models give an approximation of the development of spatially distributed sediment erosion and deposition.

The model of volume differences for the first months of development (November 2005 – May 2006, Fig. 2.3a) indicates relatively high rates of elevation increase and decrease which are distributed irregularly across the hillslope. In the model for the next interval of surface development (May 2006 – November 2006, Fig. 2.3b), the incision of the first erosion rills in the central and in lower parts of the slope is observable, and spatially concentrated sedimentation is indicated in the central part of the downslope area. The model of change for November 2006 – November 2007 (Fig. 2.3c) shows a higher spatial concentration of erosion to the central rill and major erosion rills, especially in the NE part of the catchment. High rates of sedimentation are indicated in the western part of the footslope area. Erosion rates were highest around the trapezoidal spring area in the center of the slope, while a transition from erosion to sedimentation is observable in the westernmost erosion rills in the downslope area (i in Fig. 2.3b, Fig. 2.3c).

Rates of erosion and sedimentation derived from the models for later time intervals are generally lower. In the model for November 2007 – August 2008 (Fig. 2.3d), the continuing incision of rills is observable, and a higher concentration of erosion to narrower rills, compared to the time interval before, is indicated. Intensity of rill incision around the trapezoidal spring area appears to be lower compared to models for the previous intervals, while in the areas further upslope no clear difference in erosion intensity can be observed.

In the models for August 2008 – June 2009 (Fig. 2.3e) and June 2009 – March 2010 (Fig. 2.3f), the interpretation of spatial distribution of erosion is strongly restricted by areas of spatially concentrated volume change that may result from vegetation cover or processing of the DEMs. Relatively high rates of erosion are indicated in single areas of the central rill rather than equally distributed along the rill (ii in Fig. 2.3e,f). The model for June 2009 – March 2010 (Fig. 2.3f) indicates further backslope displacement of erosion intensity as well as sedimentation in previously incised areas around the trapezoidal spring area.
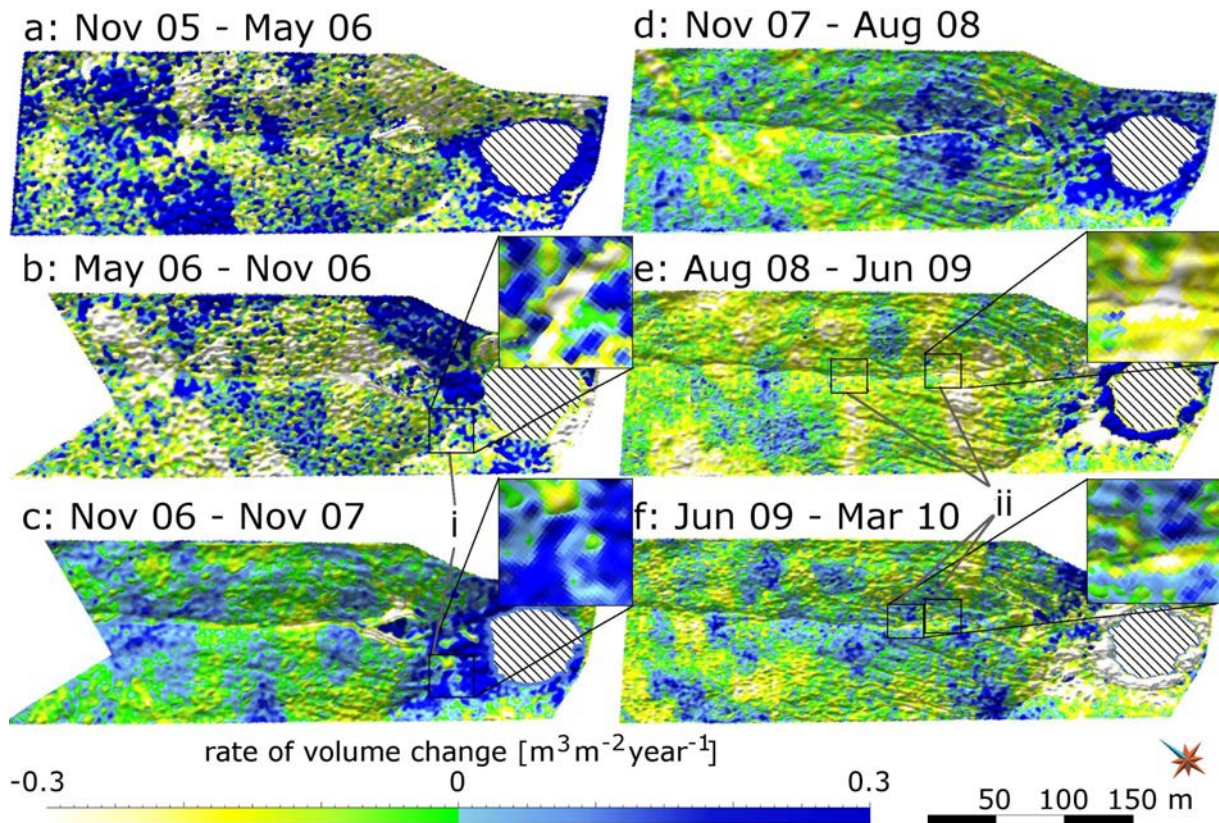

Fig. 2.3:   Maps of change in sediment volume derived from 3D models of difference between available DEMs for a) November 2005 to May 2006, b) May 2006 to November 2006, c) November 2006 to November 2007, d) November 2007 to August 2008, e) August 2008 to June 2009, and f) June 2009 to March 2010; negative numbers (white to green colors) show erosion, positive numbers (blue colors) show deposition; roman numbers mark (i) areas of transition from erosion to deposition and (ii) erosion and confluences of tributaries to the central erosion rill; the pond surface indicated by the hatched areas was excluded from the analysis; models b) and c) do not cover the whole monitoring area.

The net change in elevation for the total period of surface development from November 2005 to September 2010 is reflected in the map of elevation differences based on the

photogrammetric DEM for the initial surface and the high-resolution ALS DEM for the further developed surface (Fig. 2.4).

change in surface elevation [m]

-0.5   0   +0.5
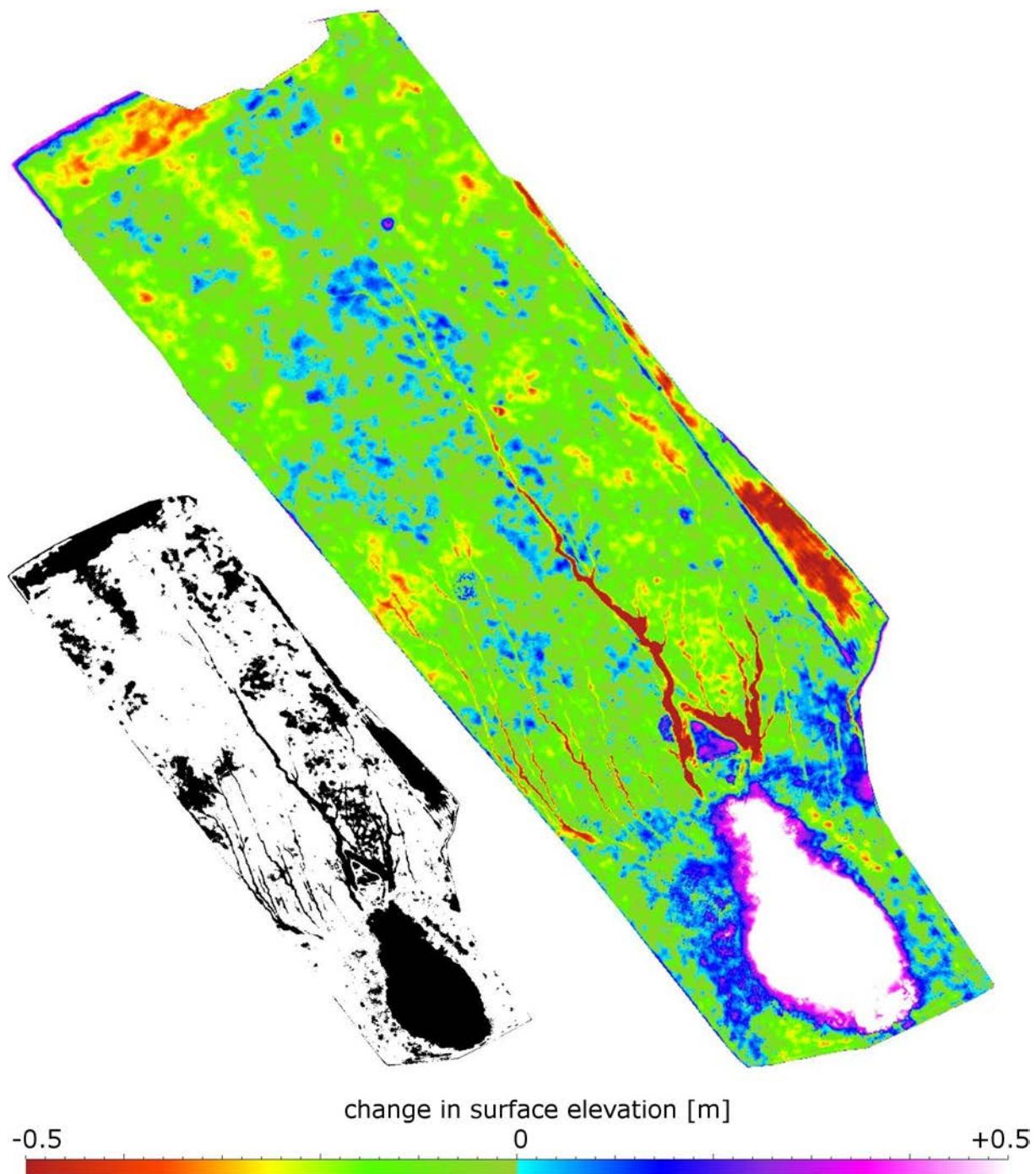
Fig. 2.4:   Differences in surface elevation between the ALS-based DEM of September 8th, 2010 and the photogrammetry-based DEM of November 26th, 2005; negative values (green to red) indicate erosion, positive values (blue to white) show deposition; the small map shows areas of elevation change above the minimum level of detection defined from the general DEM accuracy with a confidence limit of 68%; the pond area was excluded from the evaluation.

High intensity erosion is observable in the central erosion rill and its main tributaries on the NE part of the catchment. Lateral widening of the central rill as a result of high-intensity erosion at confluences of tributaries is reflected. For the SW part, the model clearly shows that erosion intensity was less concentrated to major erosion rills, but high-intensity erosion occurred in more, narrow-spaced rills. A transition line between areas with high-intensity rill erosion and the sedimentation-dominated footslope areas, running in perpendicular direction to the slope around the outlet of the upper weir, is observable. The highest rates of alluvial sedimentation were clearly concentrated in the central footslope below the trapezoidal spring area. High decrease in elevation in the area outside of the monitoring area in the eastern part of the catchment, resulting from further construction works, is also observable in the model. Results of the delineation of areas with elevation change above a minimum level of detection defined with a confidence limit of 68% show that the inaccuracy of the input DEMs limits the significant detection of change to the major erosion rills and the alluvial fan below the trapezoidal spring area. High rates of change occurring in interrill areas of the hillslope might have resulted from sheetwash erosion or settling of the sediment. However, it has to be considered that definition of a minimum level of detection only takes into account the general inaccuracy of the elevation data. Therefore, it cannot be excluded that these patterns result from systematic errors in the elevation models or are influenced by referencing to d-GPS data.

2.3.2 Estimation of aeolian sediment relocation

For average erodibility scenarios, simulated sediment losses from the catchment area are about two magnitudes higher than for minimal erodibility scenarios, ranging from 14 Mg to 54 Mg in two years, while sediment loss rates amount to 1.2-4.5 Mg ha^{-1} a^{-1}. For average and high susceptibilities, the spatial texture distribution has an influence on the net sediment balance: More heterogeneous distributions mostly result in higher shares of easily entrainable sand fractions in the downwind northeastern quarter, which is reflected by higher losses from the catchment area. Interpolated texture distribution causes sediment losses to increase dramatically compared to uniform texture. For high erodibility scenarios, total losses of 2.7 Mg and 6.4 Mg are calculated for scenarios with two and four different soil textures, and high losses of 43 Mg are simulated for interpolated texture (cf. calculated soil loss rates in Tab. 2.1). When assuming equally erodible surroundings, total soil losses in all four texture distribution scenarios are reduced by an amount of about 11-14 Mg compared to their counterparts. WEPS model results for surroundings with 50% lower emission potential show reduced net sediment losses, while still being several times higher than losses in scenarios assuming equally erodible surroundings.

For WEPS simulations, the 'Chicken Creek' pond was defined as a static, non-erodible clay surface in order to act as a "trap" for aeolian sediment. Assuming average erodibility resulted in sediment inputs into the pond of about 1.7-6.2 Mg; when assuming high erodibilities, inputs rose to between 10 Mg and 39 Mg. Spatially distributed texture patterns seemed to have significant impact when assuming heterogeneous texture distributions. Here, the southwestern sector with high contents of fine sand (0.05-0.1 mm) acted as a major source of wind-blown material transported in upwind direction into the pond.

Tab. 2.1: WEPS modeling results for wind erosion scenarios on 'Chicken Creek'; given are soil loss rates in Mg ha^{-1} a^{-1}; net aeolian deposition did not occur; the simulation period lasted 27 months from October 2005 to December 2007 (modified from Maurer & Gerke, 2011).

	Uniform texture	2 substrates (E and W)	4 substrates (NE, NW, SE, SW)	Interpolated texture values
Low erodibility	-0.01	-0.01	-0.01	-0.01
Average erodibility	-1.2	-1.4	-1.5	-4.5
High erodibility	-7.0	-7.9	-9.1	-8.1
Surroundings 50% erodible	-2.5	-1.3	-1.4	-0.9
Surroundings equally erodible	-0.0	-0.2	-0.5	-3.6

The distribution patterns identified via supervised image classification indicate that wind-blown sediments tended to accumulate mainly in the eastern half of the catchment (Fig. 2.5). This coincides with the main downwind direction. Likewise, deflation zones are much more common on the western half of the catchment. These patterns roughly coincide with sediment distribution maps of scenarios with more complex texture distributions. In these scenarios (4 soils, kriging interpolation), sediment accumulates especially on the eastern half before being transported out of the simulation area. The quantitative analysis of the supervised classification yielded an estimated mass of aeolian sediments on the simulation area of about 116 Mg (Maurer & Gerke, 2011). The calculated magnitude of aeolian translocation of about 4-8 Mg ha^{-1} a^{-1} fits to observations of wind erosion on bare sand surfaces (e.g., Funk et al., 2004).
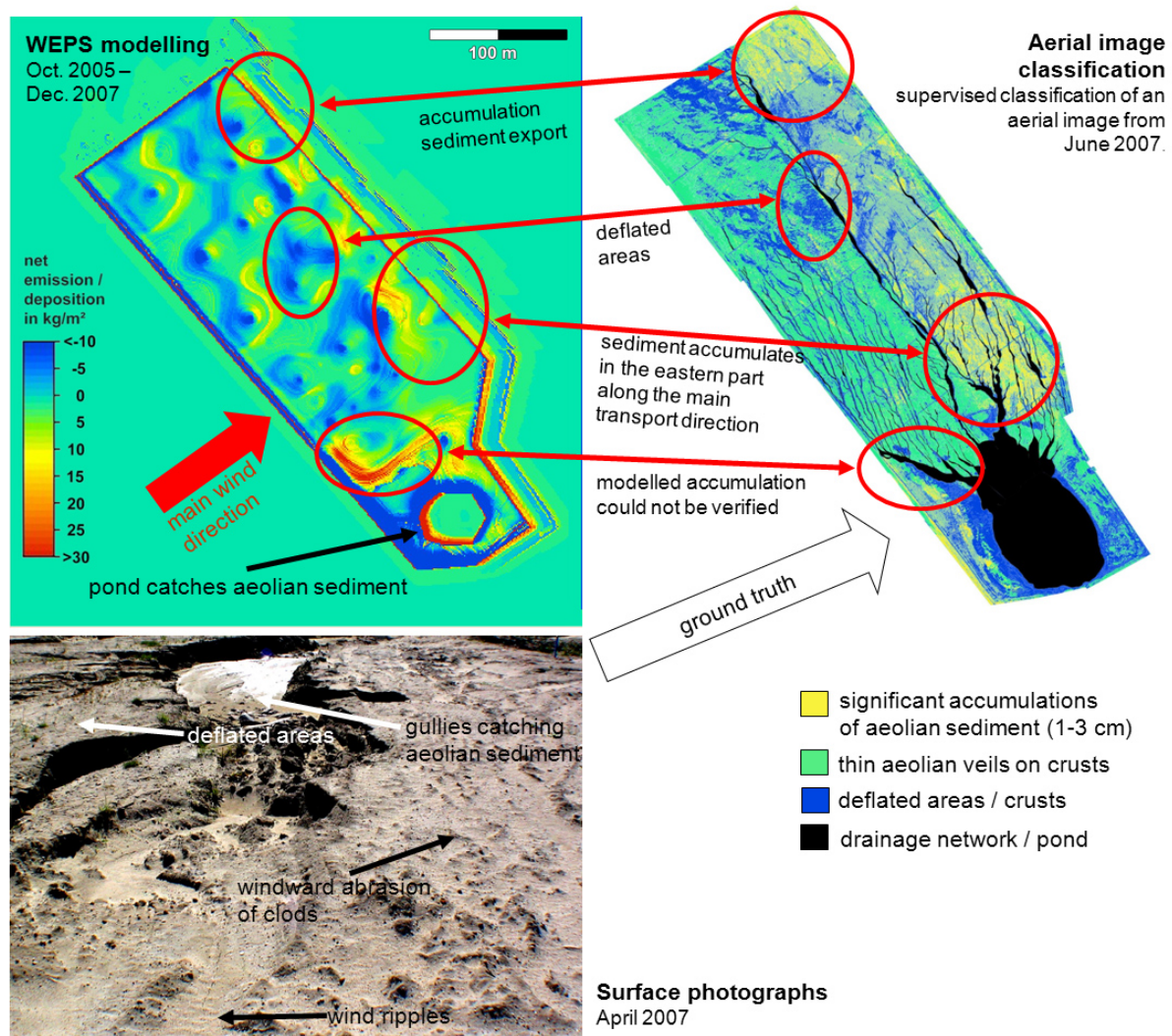
Fig. 2.5:   The magnitude of wind erosion on 'Chicken Creek' was estimated via a multi-scenario approach using the Wind Erosion Prediction System (WEPS); simulation results (upper left side, here best fitting scenario with interpolated soil texture and assuming high erodibility) were verified with data from the literature and aerial image analysis (upper right side) (modified from Maurer & Gerke, 2011).

Besides the uncertainty of input parameters describing surface conditions, other factors that add to the uncertainty of results must be considered: (i) The role of the surrounding environment: assuming additional external aeolian input, overall net sediment balances are even likely to change to a regime where net-inputs are higher than net-outputs. However, due to the steadily growing protective plant cover on most of the surrounding areas, it is unlikely that this influence was as massive as in the scenarios for an equally erodible environment. (ii) The influx of vegetation was considered to be negligible for the simulation period (October 2005 – December 2007), based on studies on the effect of biomass cover on wind erosion processes (e.g., Michels et al., 1995; Sterk & Spaan, 1997). This assumption, however, was

not confirmed directly by measurements on-site. (iii) Crust formation was not included as a dynamic process. Instead, we assumed an initial state where the surface was covered either by relatively easily erodible crusts, representing compacted material as well as fragile initial physical and biological crusts. (iv) Surface conditions in WEPS are not updated with respect to water erosion processes, which means that changes in surface topography (i.e., surface roughness) are not taken into account.

However, from the verification of simulation results with estimations from aerial image analysis and literature, and despite the considerable uncertainties, we can conclude that scenarios assuming high erodibilities and spatially complex texture distributions seem to reproduce the magnitude of wind erosion at least in a realistic manner. From that, we can conclude that the assumptions for the wind-erosion specific surface parameters made in these scenarios are the best approximation of the real initial conditions.

2.3.3 Surface morphology in September 2010

The morphology of the catchment surface in September 2010 is reflected in the ALS based DEM (Fig. 2.6a). Diversification of surface structures resulting from processes of erosion and deposition is clearly observable. The map of slopes derived from the DEM (Fig. 2.6b) further highlights morphological structures of the surface, as steep rill walls are captured in the high-resolution DEM, resulting in particularly high slopes. The effect of morphology on surface flow paths along the hillslope is reflected in the map of the model cells contributing areas (Fig. 2.6c). On the SW part of the hillslope, flow paths are about equally spaced, running parallel and are relatively straight, while on the NE part high amounts of surface flow are collected in one main erosion rill which is draining to the eastern branch of the central rill. In the central rill, 'meandering' flow is observable in areas where the gully was considerably widened by lateral erosion at confluences with tributaries.

The map of *SPI* values (Fig. 2.6d) reflects areas that are prone to erosion as a result of morphology. High *SPI* values occur in the already developed erosion rills. Outside of the rills, *SPI* values area higher at the sides of the central rill and in the footslope area immediately above the alluvial fans. High values of the index in the alluvial fan areas are a result of the overestimation of flow concentration. The map of *TWI* values (Fig. 2.6e) reflects areas that are subject to differing wetness conditions as a result of morphology. High values occur at rillbeds, while at the steep walls of major rills lower values indicate potentially dryer conditions. Generally, *TWI* values are higher at the backslope area and around the pond because of the lower inclination of the slope, and areas of high *TWI* values occur at confluences of surface flow paths.
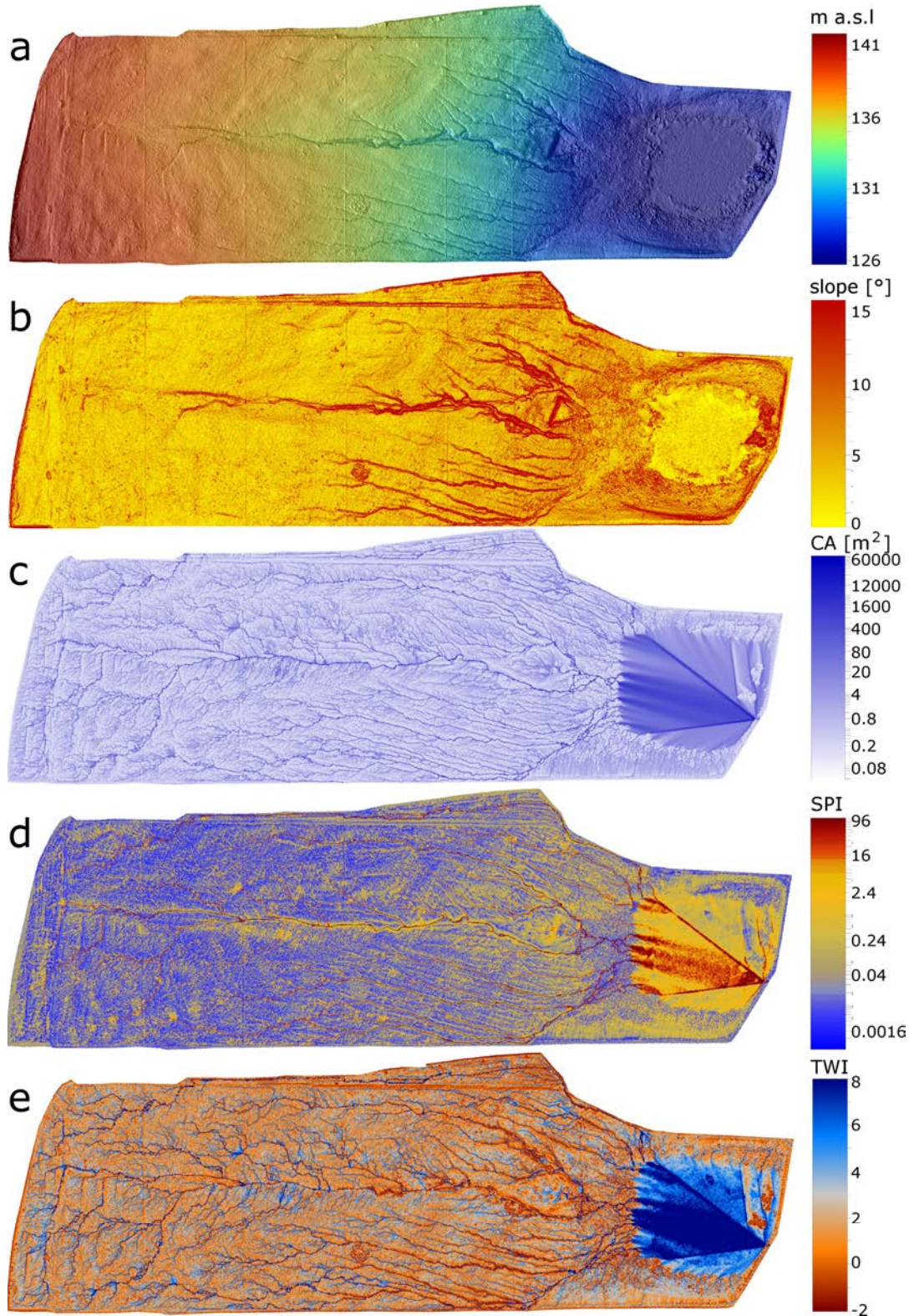
Fig. 2.6: Digital terrain model based on airborne laser scanning (a) and morphometric parameters derived from the model (b-e); a: Point spacing of the DTM is 0.25 m, for visualization of surface structures the DEM is overlaid with a map of analytical hill shading; b: map of local slopes for the DEM cells; c: map of Specific Catchment Area values (given in number of cells); d and e: maps of values of the Stream Power Index and the Topographic Wetness Index.

2.3.4 Hydro-geomorphic surface development in relation to initial and boundary conditions

Although the temporal resolution and accuracy of elevation models were not sufficient to exactly reconstruct or identify thresholds in erosion-affected surface structural development, the combination of approaches allowed a characterization of surface evolution and its interdependences with other structures and processes of ecosystem evolution.

Results of the analyses of surface morphology development from 2005 to 2010 affirmed that different types of initial surface structures, described in Gerwin et al. (2010), have affected surface development:

- The main patterns in surface flow paths and thus the main patterns of the developed rill network are following the overall topographic structure of the artificially-constructed surface. The location of the central main gully was predetermined by the construction of the two parallel slope parts facing together. The trapezoidal spring area in the center of the slope was delineated with walls which caused the branching of the central gully and thus the formation of two deeply incised rills on both sides of the spring area. Directions of surface flow and thus erosion rills are following the main slope directions throughout the catchment, so that erosion rills in the NE part are generally running toward the central gully (in southern direction), while in the SW part rills are orientated in southeastern direction and running towards the central gully in the backslope area, but are running in southern direction in the lower slope areas (cf. Fig. 2.2).

- Although the surface of the catchment was flattened before it was left to undirected development, it is probable that small-scale structures remained from the construction works, e.g., caterpillar tracks only covered by thin layers of loose sediment (Gerwin et al., 2010) or wheel tracks of tractors used for smoothing the surface. A comparison of aerial photographs recorded shortly before and after the end of construction works with the digitized rill network of 2007 (Fig. 2.7) indicates that rill network patterns were orientated along these surface structures. In the uppermost slope area, erosion rills were running perpendicular to the central gully, obviously in the same direction as bulldozer tracks from the construction works. In the SW part of the slope, aerial photographs show densely spaced and very pronounced bulldozer tracks, which might have influenced the development of the densely spaced and narrow erosion rills in this area. Furthermore, the comparison shows that the course of the gully that the rills on the SW part drain to is orientated along the transition zone between material dumped in the western part of the catchment and material that was probably dumped in a separate delivery in the footslope area.

- Differing sediment properties in the SW and the NE part of the catchment (Gerwin et al., 2010) can be assumed to cause different infiltration conditions and erodibility of the sediment. These factors may have caused the evolution of more and densely spaced erosion rills in the SW part in contrast to the spatial concentration of erosion to few rills in the NE part.
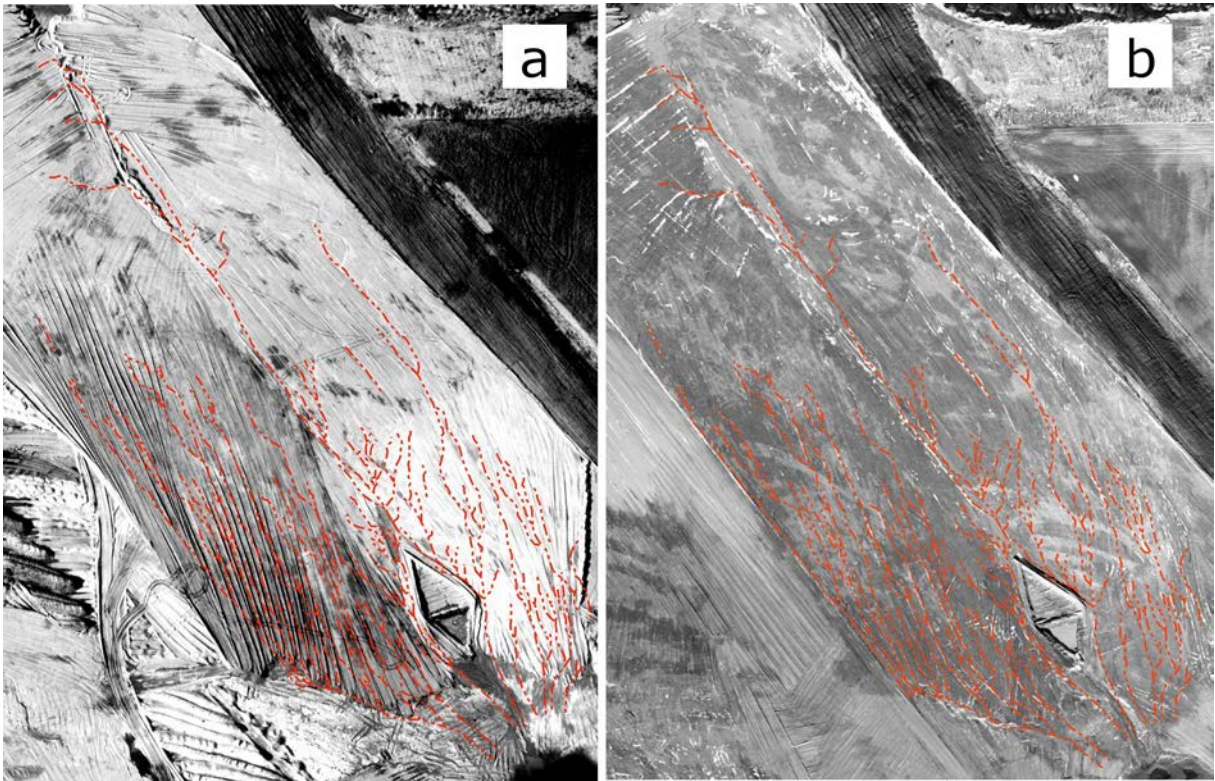

Fig. 2.7:   The rill network as mapped from aerial photographs of June 14th, 2007 (orange) in comparison to aerial photographs of the construction phase; a) aerial photograph taken on August 12th, 2005, before final flattening shows bulldozer tracks on the surface b) aerial photograph taken on November 26th, 2005 after final leveling of surface shows remaining small-scale structures (e.g., tracks from tractor wheels).

Influences of these three types of initial conditions cannot be exactly distinguished, most probably combined influences of initial conditions occur in different parts of the hillslope: A comparison of aerial photographs of the construction phase (Fig. 2.7) and overall topography (Fig. 2.6a) shows that the tracks of construction machinery follow the main slope directions in most parts of the catchment, so that these structures probably did not counteract the effects of overall topography on surface runoff routing. However, they might have considerably influenced flow accumulation patterns, e.g., the closely spaced construction structures in the SW part of the catchment might have prevented flow accumulation over a large drainage area and instead routed surface flow along many parallel drainage lines, causing the incision of

closely spaced but relatively small erosion rills. Differences in drainage density between the SW and NE part might also be a result of differing infiltration properties or erodibility of the differing sediments in the two parts of the catchment.

The reconstruction of change in sediment volume in time intervals (Fig. 2.3) does not show clearly defined rill erosion for the first six months of surface development. This is, however, most probably due to the inaccuracy of the DEMs. Field observations and aerial photographs show that the incision of first erosion rills started already with the first precipitation events after the termination of the construction works (Biemelt et al., 2010; Gerwin et al., 2010). An early formative surface runoff event occurred in consequence of snowmelt on frozen soil in January 2006 (Biemelt et al., 2010). Rill network growth was further accelerated by a number of high rainfall events concentrated in June and August 2006 (Gerwin et al., 2011). Most favorable conditions for rill incision were in May 2007, when considerably high amounts of precipitation followed a period of very dry conditions (Biemelt & Nenov, 2010). This was reflected in the rapid growth of the rill network until 2007. The rapid formation of the rill network during the first two years of catchment evolution has most probably also been influenced by the formation of mineral and biological soil crusts (Badorreck et al., 2009; Fischer et al., 2010), which on one hand can reduce the erodibility of surface areas, but on the other hand reduce infiltration and cause an increase and spatial concentration of surface runoff. The fact that no further headward growth of erosion rills was observed after 2007 (cf. Fig. 2.2) might be a consequence of the lower precipitation intensities in the following years (cf. chapter 3). Field observations and aerial photographs furthermore show that, along with the overall increase in plant cover (cf. chapter 5), especially peripheral inactive erosion rills became colonized by vegetation. This colonization can be assumed to cause a stabilization of the erosion rills, e.g., by increasing infiltration rates, reducing the velocity and erosive power of surface runoff or retaining eroded material (Molina et al., 2009). While peripheral rills became increasingly affected by biotic factors with proceeding ecosystem development, central areas of the rill network became affected by groundwater discharge when the major rills had incised down to the clay layer around the spring area or to the groundwater body along the central erosion rill where baseflow was observed (cf. chapter 3). This effect can be assumed to cause slowly continuing incision of these rills and thus a decreased dependence on the temporal dynamics of precipitation.

2.4   Conclusions

Despite the limitations due to scarce data and data uncertainty, the combined analyses of aerial images, elevation data, and model scenarios, supplemented by terrestrial validation, allowed a characterization of hydro-geomorphic surface evolution during the first five years. During this initial period, the soil-geo system was considerably altered by sediment

relocation. The most formative processes were erosion and deposition of sediments by water erosion. Analysis of elevation data and aerial photographs showed that hydro-geomorphic development in the first phases of development was characterized by vertical rill incision, growth of the rill network, and a high dependence on abiotic initial conditions. Preliminary visual observations also indicated a noticeable contribution of wind erosion to the total sediment balance. Results from wind erosion modeling and aerial image analysis suggest that aeolian sediment relocation was about two magnitudes lower than that caused by water erosion. Sediment relocation and the emergence of erosional and depositional structures resulted in rapid surface differentiation. High erosion rates in the first years of development also caused major morphological changes in the 'Chicken Creek' pond (cf. chapter 8). After the first phase of rapid growth and vertical incision, evolution of the erosion rill network was later found to be predominantly characterized by lateral erosion. Furthermore, the areal extent of actively eroding rills decreased, which probably corresponded with the establishment of vegetation in formerly more instable regions. Reduced erosion rates in the catchment are reflected in the decreased rates of sediment accumulation in the pond between 2008 and 2010 (cf. chapter 8). The proceeding consolidation of surfaces, e.g., by crusts and vegetation, will probably completely inhibit wind erosion processes on the catchment, while aeolian sediment import from the surrounding mining landscape could remain relevant well beyond this very first phase.

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3. Water budget components

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3.1 Introduction

After the technical construction of the 'Chicken Creek' catchment a rapid development especially of surface and vegetation took place (cf. chapters 2, 5). In relation to this, changes in the hydrological behaviour have to be expected. In addition, external driving forces affect the input and output of water and energy. This chapter presents the measured time series of hydrologic parameters as a basis for further detailed investigations of structure-process interactions.

3.2 Materials and methods

The initial sensor installations described in Biemelt et al. (2010) were continuously maintained and during the last two years supplemented by additional measurements. The following parameters were recorded:

- precipitation, solar radiation, air temperature, humidity, wind speed at two weather stations,

- soil water content at 0.1 m, 0.3 m, 0.5 m 0.8 m depth and soil water tension at 0.3 m, 0.5 m, 0.8 m 1.5 m depth at four profiles (at the soil pits),

- soil water content at 16 locations at 0.1 m, 0.2 m, 0.3 m, 0.4 m, 0.6 m and 1.0 m depth (PR2/6-profile probes, Delta-T-Devices),

- soil water content using TDR-probes at 0.15 m, 0.3 m, 0.5 m, and 0.8 m depth at four locations,

- soil water content at 0.03 m and 0.1 m depth along two transects close to the grid points K5/K6 and N5/N6, respectively (ECH2O-EC5 probes, Decagon Devices),

- groundwater levels at 30 locations,

- discharge at the outlet point (spring area) in the middle of the clay wall (weir 1/ upper weir),

- discharge from the central gully (flume 2) and from the main gully in the eastern part of the catchment respectively,

- water level in the pond, and

- total catchment runoff (at the catchment/pond outlet, weir 2/lower weir).

In the lower part of the catchment, between weir 1 and the pond, sediments accumulated by erosion from the hillslope forming an alluvial fan of several decimeter thickness. More sediment was deposited into the pond (cf. chapter 8). The initial construction of weir 1 enabled a free drainage towards the pond (Fig. 3.1a).

Fig. 3.1:   Weir 1 a) after installation, November 2005, b) drainage pipe to the pond, June 2007, c) sediment spillage, August 2010, d) plastic shielding, September 2010.

Due to the formation of the alluvial fan the surface elevation around the weir increased by about 0.5 m compared to the level at the time of installation. As a result, free drainage from

the weir was hampered. Therefore, in June 2007 a PE pipe was installed below the surface to ensure direct drainage from the weir to the pond (Fig. 3.1b). After a heavy rainstorm in August 2010, the weir was partially covered by sediment (Fig. 3.1c). The sediment was removed and the weir was shielded by plastic plates (Fig. 3.1d).

Because of the sedimentation in the pond and the rapid plant growth (Fig. 3.2) the storage volume for water in the pond was reduced. Therefore, the shape of the pond was surveyed in summer 2010 again. Using Trimble®-R8 d-GPS receiver the coordinates of about 400 points were recorded.

Fig. 3.2:   'Chicken Creek' pond in May 2011 (photo: Elmer).

In May 2010, additional soil moisture sensors (ECH2O-EC5, Decagon Devices) were installed in two transects, where consistent surface colour differences were observed from aerial images (Fig. 3.3, right). Four sensors each were horizontally installed in depths of 3 cm and 10 cm along transects of 1.5 m (Fig. 3.3, left). Soil samples were taken from the places where the sensors were installed and analyzed for texture (by sieving and sedimentation after Köhn, 1940), total carbon (CT) and carbonate content (Scheiblermethod, DIN 18129).

Additional TDR probes were installed in August 2010 at four locations. Each unit was equipped with four TDR-probes installed at 0.15 m, 0.3 m, 0.5 m and 0.8 m depth. The locations of the units mirror the locations of the soil pits along the longitudinal axis of the catchment (cf. chapter 1, Fig. 1.2b).

Fig. 3.3: Position of the two soil water sensor transects within the catchment (aerial image from July 2010), pictures of the two soil water transects at grid points K5/K6 (top) and N5/N6 (bottom) during sensor installation in May 2010.

3.3 Results and discussion

3.3.1 Precipitation

Meteorological parameters were recorded at 'Chicken Creek' since September 2005. In the months January, February and April, the five-year means (2006-2010) of monthly precipitation (original amounts) at the 'Chicken Creek' catchment were considerably lower than the long-term means (Fig. 3.4). In contrast to this, in the months July, August and September the five-year means of monthly precipitation in the catchment were considerably higher than the long-term mean values. April was the month with the lowest and August the month with the highest precipitation amount in terms of the five-year mean. Compared to the long-term data, precipitation at the 'Chicken Creek' catchment showed a more seasonal differentiation with minima in spring and maxima in autumn in the years from 2006 to 2010. In addition, the precipitation was characterised by high variability within years as well as

between the same months of different years. The highest monthly precipitation sum in 'Chicken Creek' was 205 mm, recorded in August 2010. However, only 16 mm rain fell in August 2007. Thus, August was the month with the highest variability of precipitation. Very low precipitation amounts were registered for January 2010, April 2007, April 2009 and September 2006. Annual precipitation varied between 349 mm in 2006 and 726 mm in 2010 (mean 569 mm; original amounts).
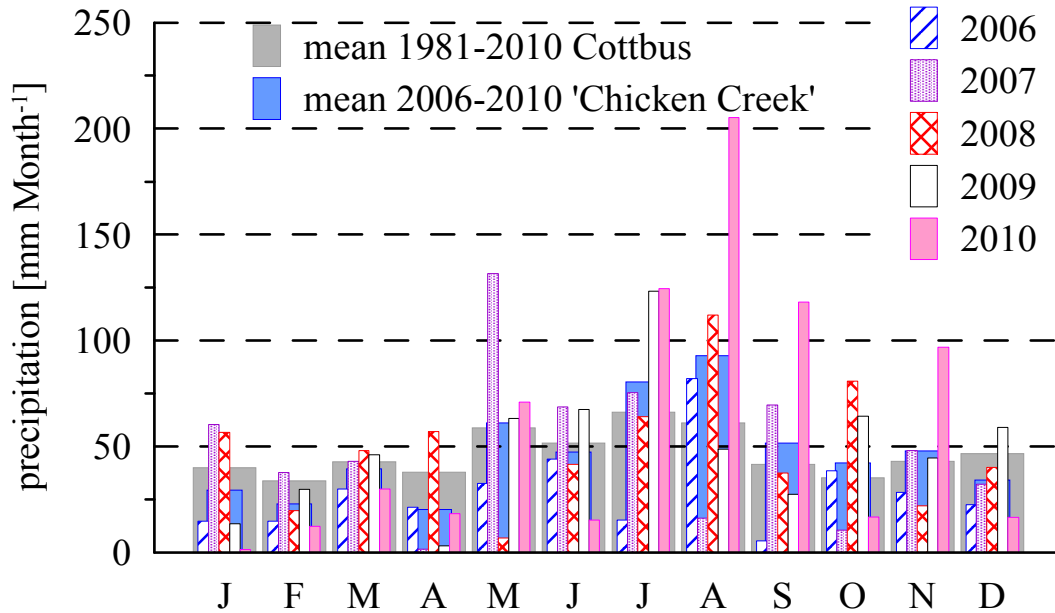

Fig. 3.4:   Monthly amount and five-year monthly mean (2006-2010) of precipitation from the 'Chicken Creek' catchment (original amounts) and long-term monthly mean of precipitation from Cottbus (German Meteorological Service, DWD).

3.3.2   Soil water

Consistent increase of soil water contents with depth could not be observed at any of the four locations (Fig. 3.5). For example, at soil pits I and C the water content at 10 cm depth exceeded that at 30 cm depth for most of the time. This is a clear indicator of the heterogeneity of dumped substrates. The amplitudes of change decreased with increasing depth. Very low soil water contents were observed down to 30 cm depth during periods with soil frost at the beginning of 2009 and 2010 because the FDR probes can register only liquid water.

At soil pits C and I the soil water content at 80 cm depth increased to about 50 Vol.-% and 40 Vol.-%, respectively in autumn 2008. This was also observed temporarily at 50 cm and at 30 cm depth at the same soil pits. During times with such high values of soil water content the probes were below the groundwater table (saturated conditions).
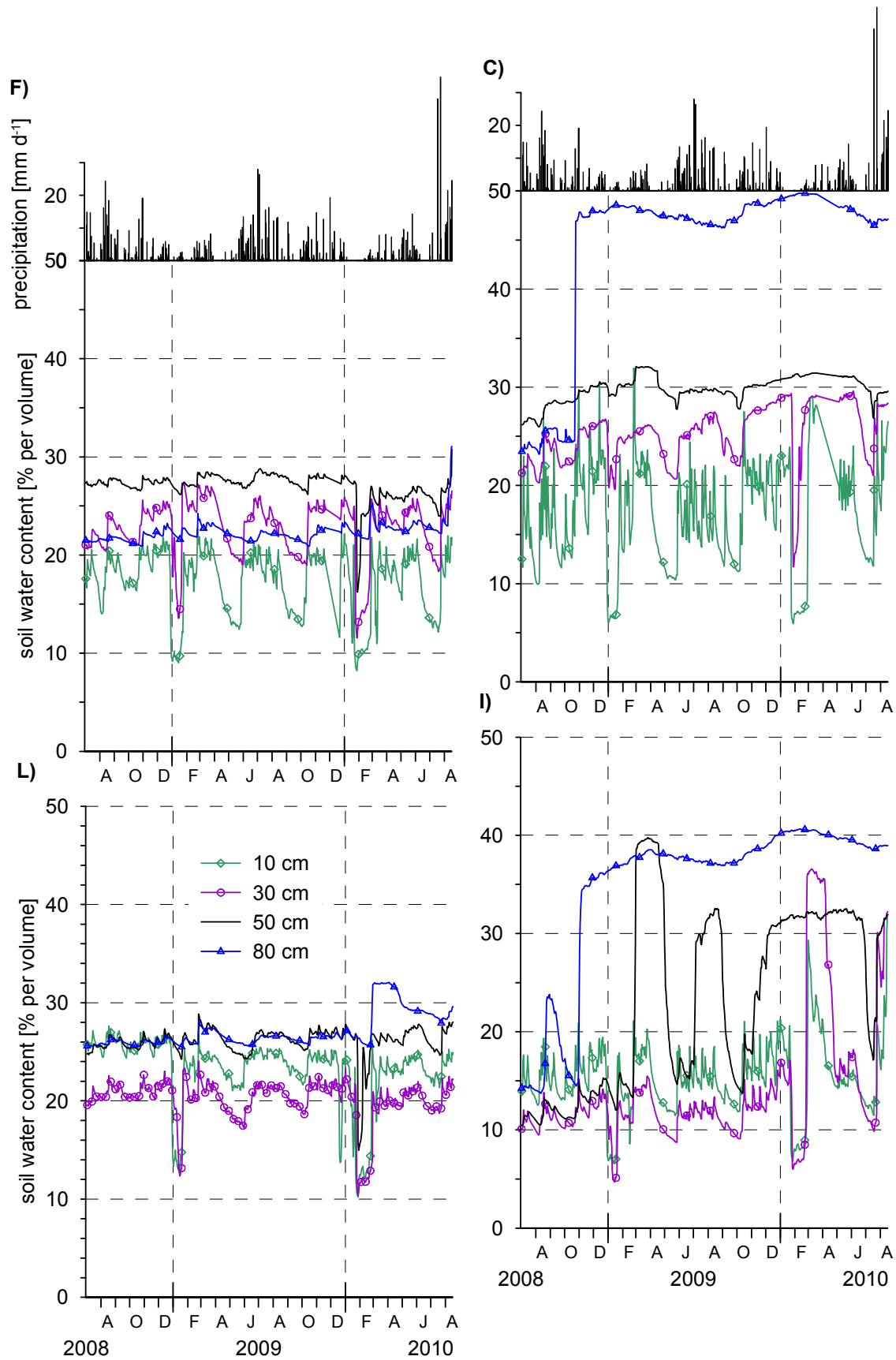
Fig. 3.5: Daily soil water content at 10 cm, 30 cm, 50 cm and 80 cm depth measured with ML2x Theta probes at the soil pits at grid rows C, F, I and L.

The maxima of soil water content in the case of reoccurrence of saturation after drying were not constant. For example, at soil pit I at 50 cm depth the soil water content reached values of about 40 Vol.-% in spring 2009. During the next saturation period in summer 2009 and also later the soil water content at this depth did not exceed 35 Vol.-%. As a possible reason for this could be structural changes in the soil pore system of the dumped substrates. This phenomenon was not observed at the other depths. The measured soil water tension indicated comparatively wet conditions for the whole measurement period (Fig. 3.6). Short episodes of higher soil water tensions were observed at 30 cm and 50 cm depth in summer 2008, 2009 and 2010 at soil pit L and at soil pit C in summer 2010. Field retention curves calculated from measured tensions and soil water contents revealed strong hysteresis between drying and wetting periods (Biemelt & Gerke, 2011). Seasonal data indicated temporal changes in the water retention, which seem to correspond with episodic water saturation during the establishment and rising of the groundwater table. A capillary fringe of about 25 cm can be identified from the field retention curves for soil pit I. Because of the high groundwater levels (Fig. 3.9) the pit installations had to remain deactivated from August 2010 to May 2011.

For the calculation of actual evapotranspiration as remaining value of the water balance the storage change in the unsaturated zone is needed. In principle the measured time series can be used for this. But, substantial uncertainties result for example from the short period of measurements, from the spatial heterogeneity of the substrates and from temporal changing of the vertical extent of the saturated and unsaturated zone. At present a complete time series of soil moisture for a whole year only exists for the year 2009 (Fig. 3.6). Hence, an appropriate determination of soil water storage changes only from measured time series is not possible.

Soil crusts were found very soon within the first years at the soil surface. Abiotic mineral crusts (mainly from gypsum precipitation, cf. chapter 4), physical crusts as a result of rain splash (puddle erosion) as well as different types of biological soil crusts (BSC) were described (Fischer et al., 2010b). Depending on type and composition both physical crusts (e.g., Bresson & Boiffin, 1990) and BSC are known to positively or negatively influence infiltration rates and surface runoff processes (Littmann et al., 2000; Yair et al., 2008).

Soil water content in 3 cm depth was consistently higher below biological soil crusts and vegetated patches (points 3 and 4) compared to adjacent sites with bare soil (Fig. 3.7, 3.8). Similar differences were found for both 3 cm and 10 cm soil depth. As expected the reaction to precipitation events was generally higher in 3 cm depth compared to 10 cm. At transect K5/K6 the differences in soil water content between the positions were larger in 10 cm depth compared to 3 cm depth (Fig. 3.7). All sensors reacted quickly to precipitation events with generally higher amplitudes below the vegetated patches. Only during winter, the sensors in 10 cm depth showed higher amplitudes below the bare soil surface. This could be caused by freezing and thawing events, but it should be even more pronounced in 3 cm depth.
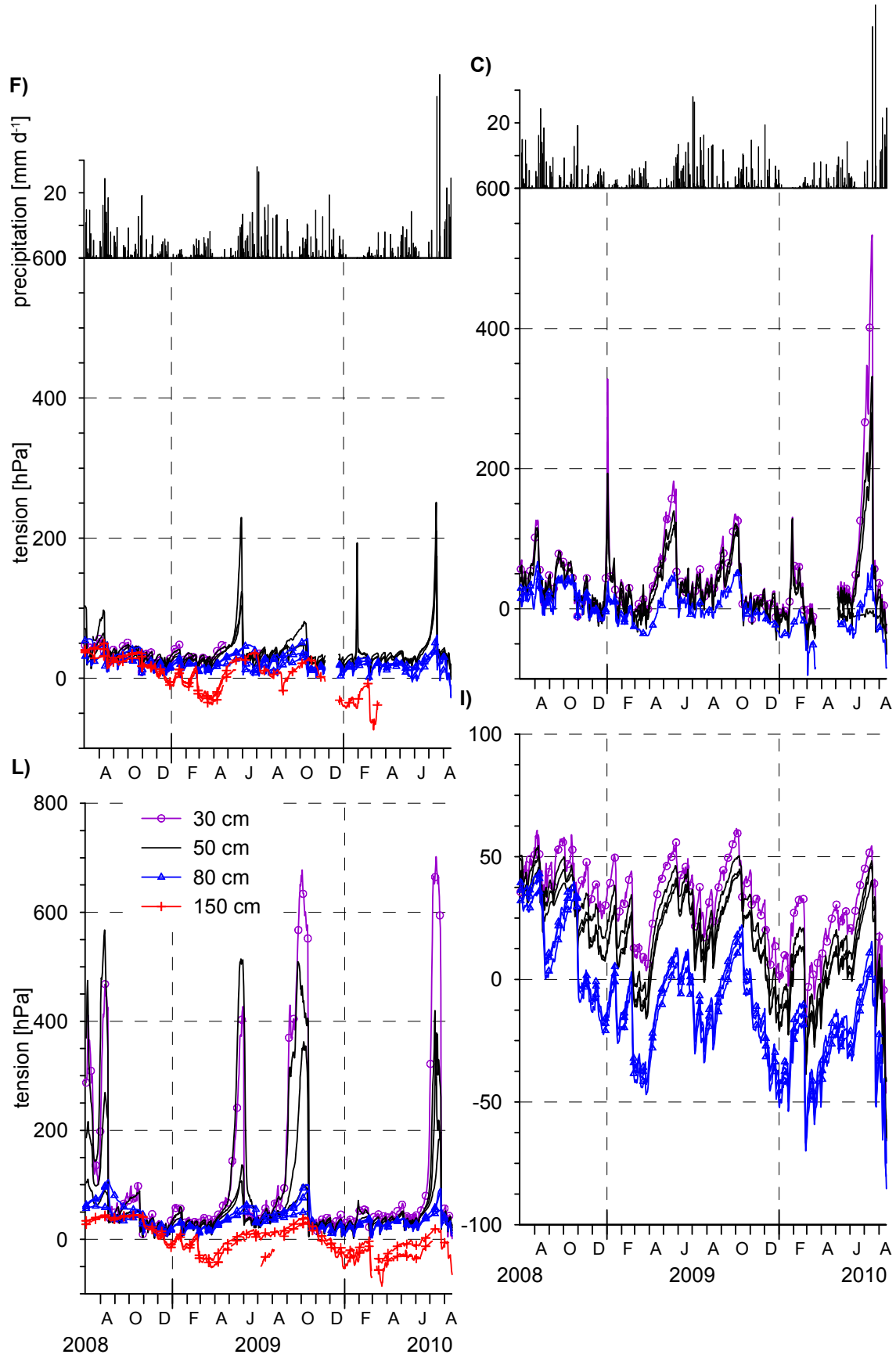
Fig. 3.6: Daily soil water tensions at 30 cm, 50 cm, 80 cm and 150 cm depth at the soil pits at grid rows C, F, I and L (note the different scales on the y-axes).
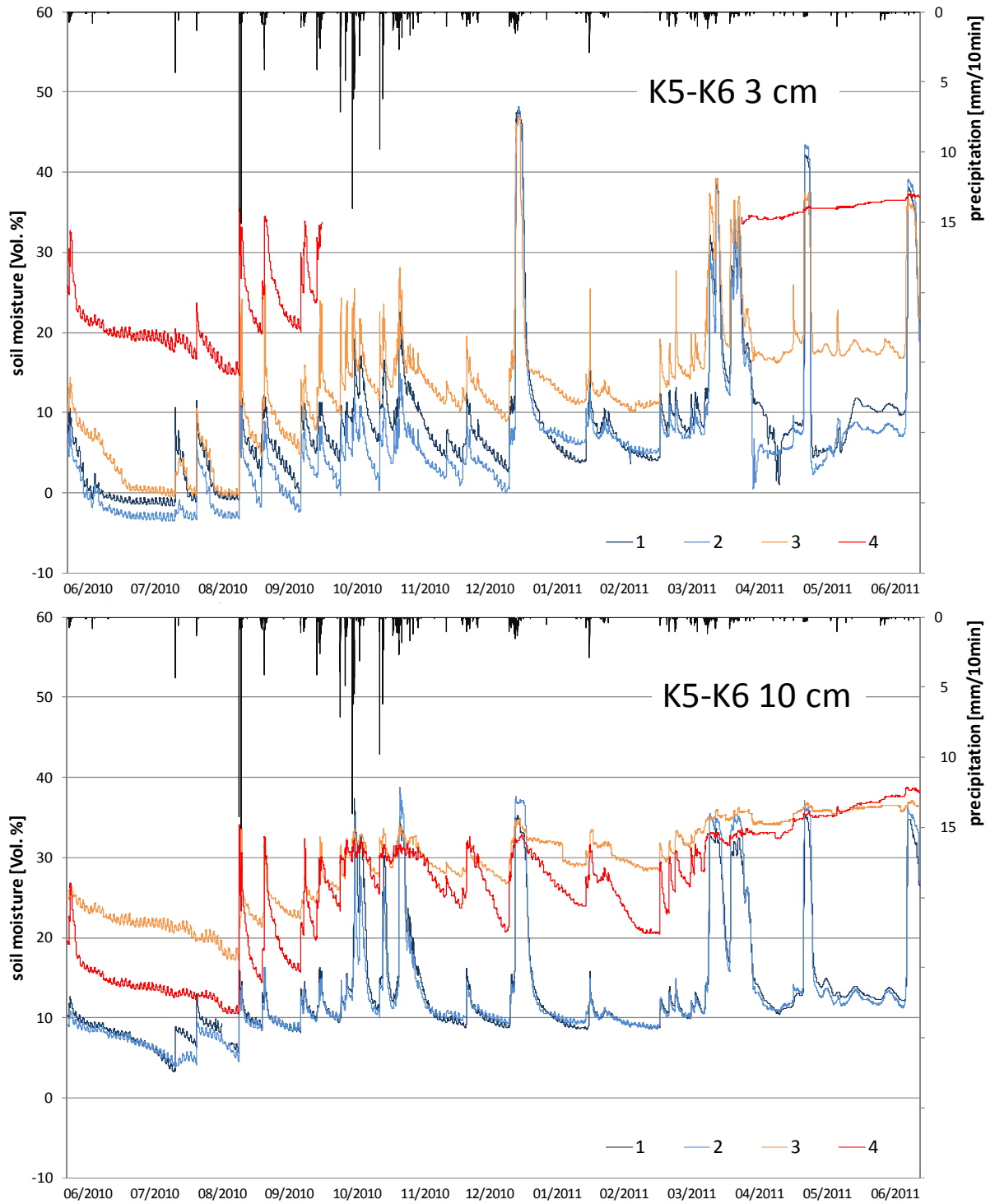
Fig. 3.7:   Soil water content in 3 cm (top) and 10 cm (bottom) depth along the transect at K5/K6.

At transect N5/N6, differences between the positions are slightly smaller. But a clear differentiation can be seen between positions 1 and 2 (bare soil) and positions 3 and 4 (soil crust), again with higher amplitudes below the crust. In 10 cm depth the soil water content at positions 2, 3 and 4 was very similar and only at position 1 the water content was consistently

lower. Soil texture data (Tab. 3.1) revealed higher silt (U) and clay (C) contents (mean) below the crust (11.6% U, 5.3% C) compared to the bare soil (5.8% U, 1.5% C). Whereas carbonate contents showed no clear tendencies, C_T contents were elevated at positions below the BSC indicating higher organic carbon contents (Tab. 3.1).
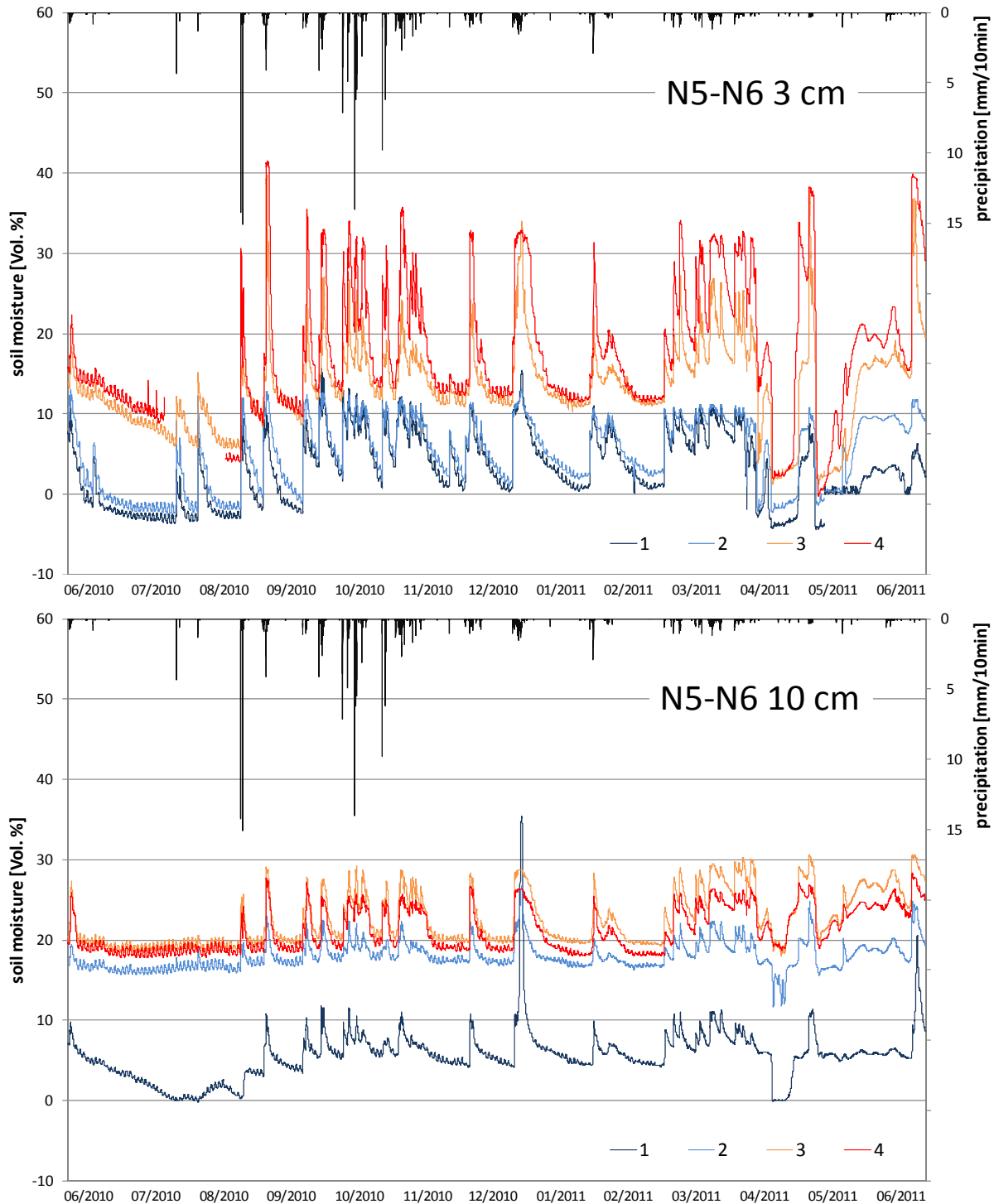

Fig. 3.8:   Soil water content in 3 cm (top) and 10 cm (bottom) depth along the transect at N5/N6.

Tab. 3.1: Soil characteristics of samples taken at the position of the soil moisture sensors.

site	depth [cm]	position	sand [%]	silt [%]	clay [%]	C_T [%]	$CaCO_3$ [%]
K5/K6	**3**	**1**	95,5	4,0	0,5	0,23	2,70
		2	83,3	11,3	5,4	0,31	2,14
		3	92,1	6,2	1,7	0,24	1,45
		4	83,5	11,7	4,8	0,52	1,75
	10	**1**	95,3	4,2	0,5	0,19	2,35
		2	93,8	5,5	0,7	0,19	1,93
		3	73,8	15,3	10,9	0,67	3,67
		4	91,5	6,4	2,1	0,45	2,54
N5/N6	**3**	**1**	91,3	6,7	1,9	0,30	1,82
		2	94,1	5,1	0,8	0,24	1,41
		3	82,0	13,0	5,0	0,58	3,17
		4	81,6	13,2	5,2	0,35	2,38
	10	**1**	93,6	5,1	1,3	0,42	4,84
		2	95,0	4,2	0,8	0,35	3,59
		3	79,7	13,9	6,4	0,47	3,06
		4	80,4	13,0	6,6	0,47	4,31

These results indicate higher infiltration and water holding capacity below the crust and/or faster surface runoff on bare soil. The lower silt and clay contents under the bare soil surface may be a result of preferential depletion of fine material by lateral water erosion. Water repellency, which is often reported for BSC, could not be detected for the soil crusts at the catchment (Fischer et al., 2010b). It remains a chicken and egg question, whether the observed differences are a result of different initial soil surface properties, e.g., by erosion intensity, or if they are caused by the establishment of soil crusts. Fischer et al. (2010a) concluded that surface stability is one of the main prerequisites for BSC establishment.

3.3.3 Groundwater levels

Except for grid points N4 and N2, the time series of groundwater levels at all other grid points were quite similar (Fig. 3.9). During the first four years (until 2009) the groundwater levels continuously increased, showing seasonal fluctuations with local minima in autumn and local maxima in spring.
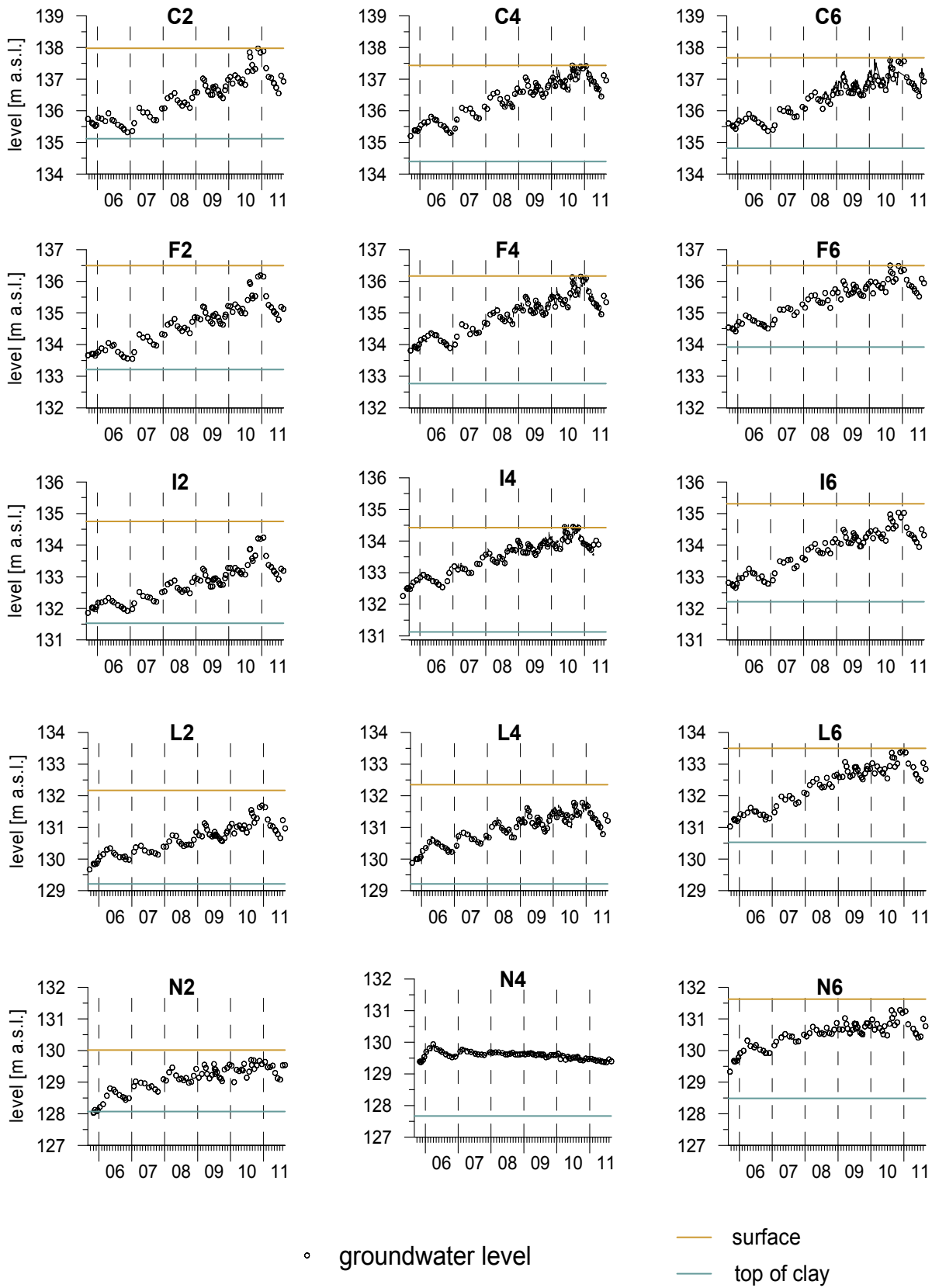
Fig. 3.9: Time series of groundwater levels (the positions of the graphs indicate the location of the observation wells in the catchment).

Due to intense rainfall in late summer and autumn of 2010 (Fig. 3.4), the groundwater levels increased rapidly and reached the surface in the upper, in the middle and in the eastern part of the catchment (grid points C2, C4, C6, F2, F4, F6, I4, I6, L6) (Fig. 3.10a). The erosion rills and gullies drained the aquifer and functioned as receiving water course during this extraordinarily wet period (Fig. 3.10b). Afterwards the groundwater levels decreased from January 2011 to July 2011 by about 1 m, due to very low precipitation in this period. The deviating groundwater regime at the observation well N4 can be attributed to the spatial proximity to the central erosion gully. The aquifer is drained by the erosion gullies since their fast incision into the ground in spring and summer 2007. The groundwater level at N4 decreased slowly from 2007 to 2011, indicating the gradual deepening of the erosion gully.

Fig. 3.10:   Photos of August 28th, 2010: a) local saturation of the soil between F4 and F6, b) groundwater discharge in the central gully near I4.

3.3.4   Discharge at specific locations within the catchment

After completion of the construction work at the 'Chicken Creek' catchment, the impact of external influences started. Precipitation events rapidly produced runoff on the bare soil surface and erosion gullies emerged. An H-flume (flume 2) was installed in the central gully for measuring the fast, short-time occurring runoff with a high temporal resolution. An example for a fast runoff event on August 11th/12th, 2008 is shown in Figure 3.11. The precipitation event lasted 150 minutes. 23.4 mm rain fell in total in three consecutive intervals, with maximum intensities of 5.2 mm, 4.9 mm and 3.6 mm per 10 minutes. The

water level in the pond increased by about 18.5 cm and 580 m³ water flowed into the pond. 170 m³ of this amount passed through the flume. The discharge level at weir 2 was not reached. Thus, the whole water amount resulting from this precipitation event was completely retained in the catchment. Further event-based investigations will help to analyse the runoff behaviour concerning changing conditions.
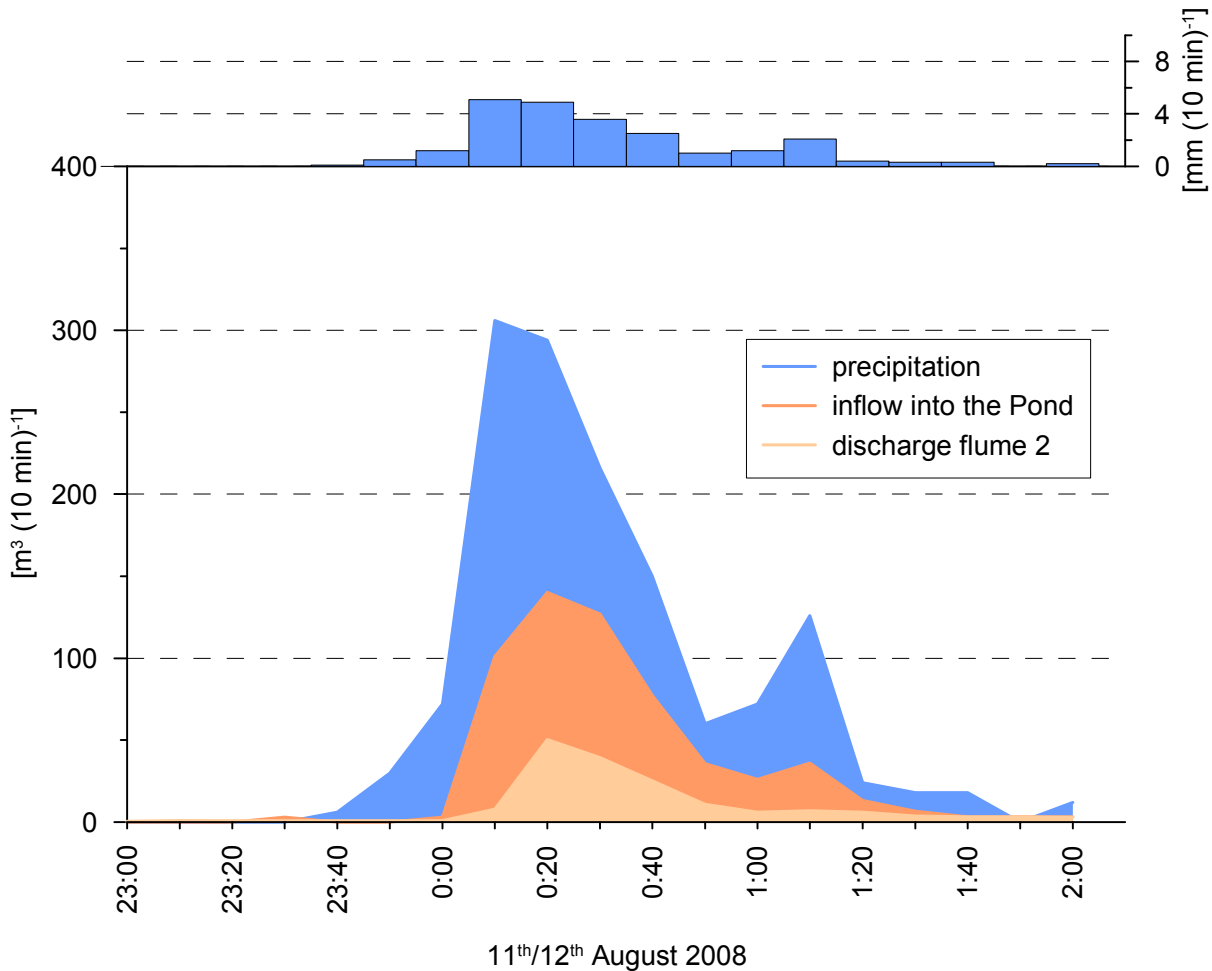

Fig. 3.11: Precipitation, runoff observed at flume 2 (2-ft-H-flume) and inflow into the pond during the rainstorm event on August 12th, 2008.

Figure 3.12 shows the discharge records at different locations within the catchment. Weir 1 was installed immediately after completion of the construction work and has provided discharge measurements since November 2005. As a result of the incision of erosion gullies down to the subsurface clay wall in the footslope area, baseflow was observed not only at weir 1, but also in these gullies. Since baseflow amounts are comparatively low, the H-flume at the central gully was complemented with a tipping bucket device in May 2008. The discharge measurement in the main gully in the eastern part of the catchment started in June 2010 (cf. Gerwin et al., 2010; Mazur et al., 2011).
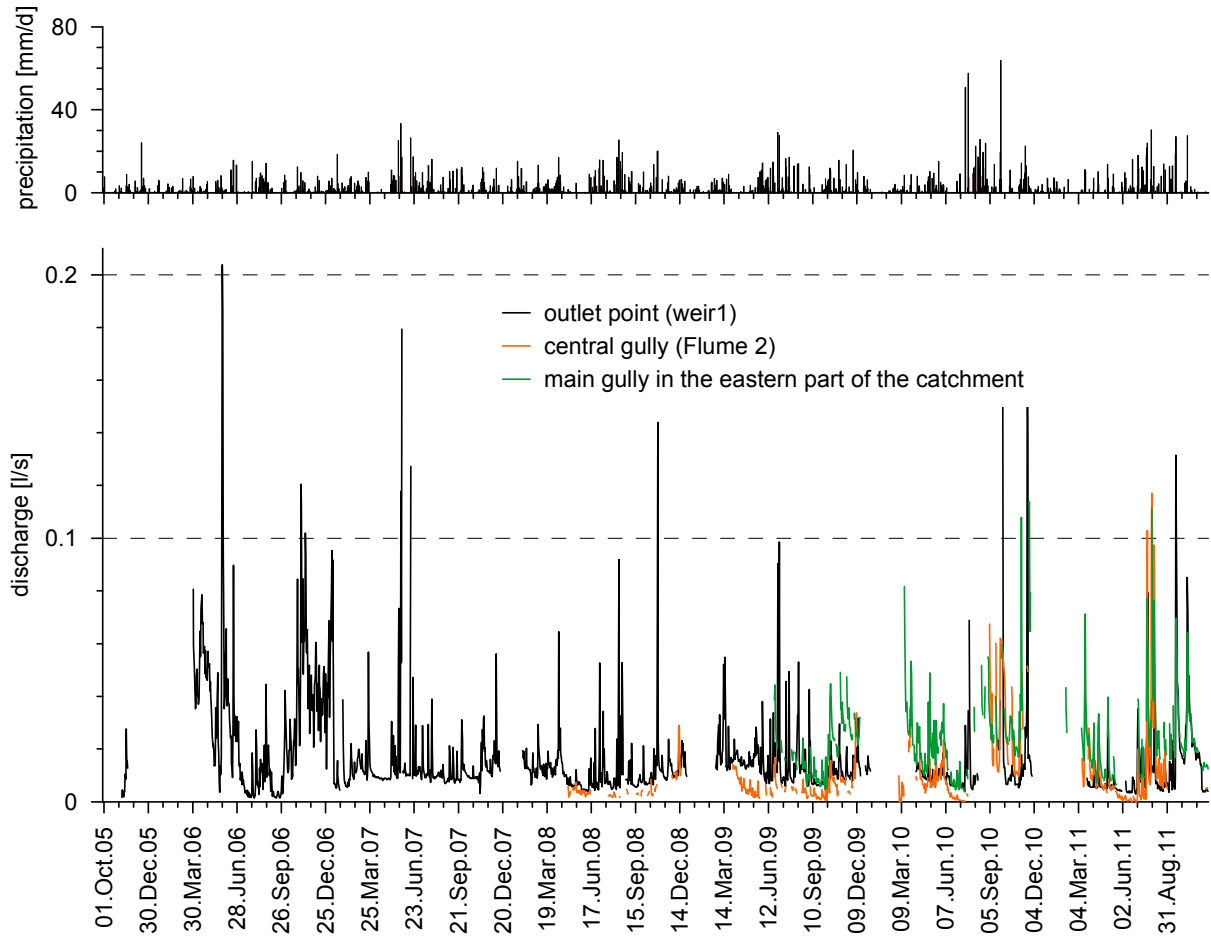
Fig. 3.12:   Daily values of precipitation and discharge at specific locations within the catchment.

The time series for the whole observation period showed discharge values with comparable magnitudes at all three locations (Fig. 3.13). The discharge measurements at these locations were interrupted at several times due to frost or disturbances associated with extensive deposition of sediment after heavy rainfall events. Additionally, the measurement of peak discharge values in the erosion gullies was limited by the capacity of the tipping buckets.

The discharge time series from the terrestrial part of the catchment allow the analysis of selected episodes in detail. For example, Figure 3.13 shows the course of discharge, groundwater level, and precipitation from April 2010 to July 2010. For this comparison, the groundwater observation well at grid point L4 and not at N4 was chosen. The latter was not representative for the groundwater level dynamics in the whole terrestrial part of the catchment, due to its close proximity to the central gully (Fig. 3.9). The discharge at the three locations was determined by both precipitation and baseflow (Fig. 3.13).

The most intense reactions to precipitation were registered at the main gully of the eastern part of the catchment. The discharge amplitudes in the central gully and in the spring area

(weir 1) were more or less similar. After the rainy May 2010 a dry period between June 12th and June 30th, 2010 followed. The discharge at the three locations decreased relatively fast to values below 0.01 l s[-1]. At weir 1 and at the main gully in the eastern part the discharge decreased to almost similar values. In the central gully the discharge decreased more rapidly. The groundwater level at grid point L4 reached a local maximum on June 7th, 2010 and fell by 50 cm during the dry period afterwards. A time lag of several days between rainfall and the onset of groundwater level change became obvious. The discharge at all three locations decreased with falling groundwater levels. This decreasing trend was expectedly weaker compared to the decrease in discharge after a precipitation event.
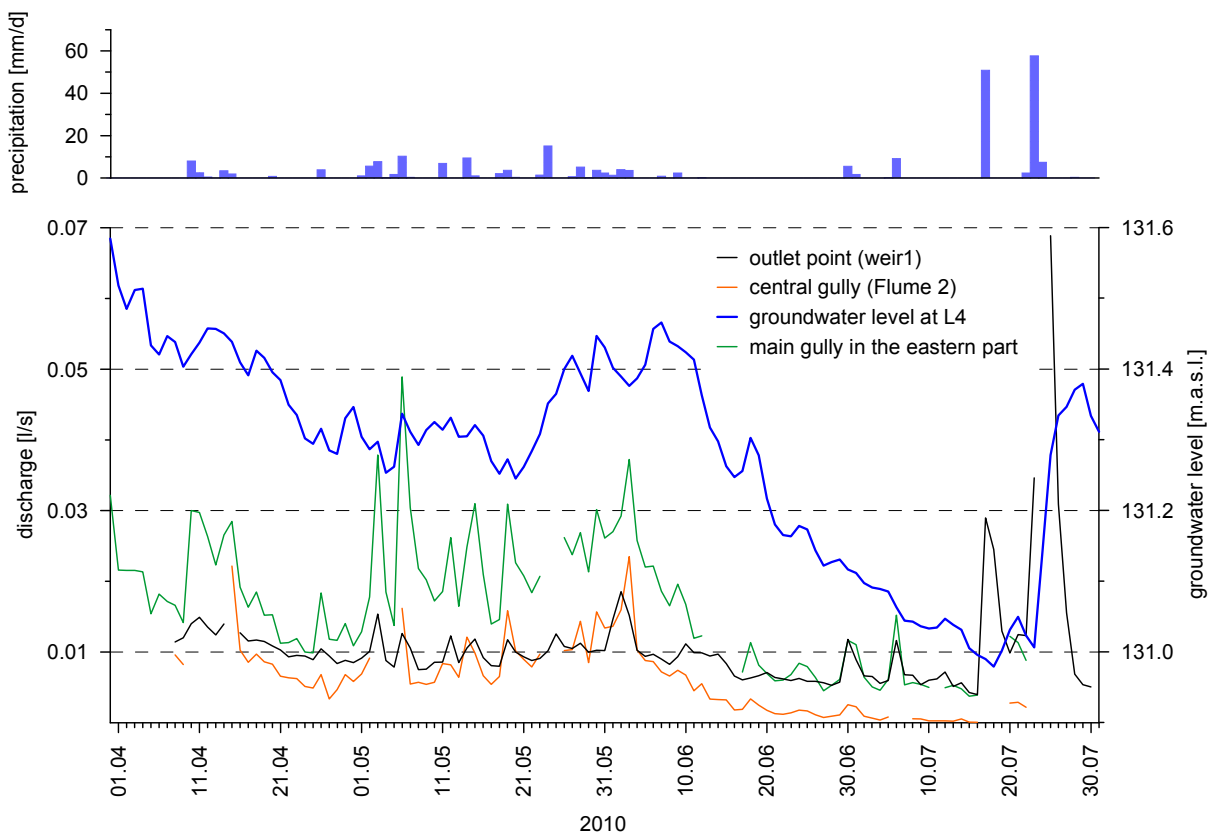

Fig. 3.13: Daily precipitation, groundwater level and discharge at different locations within the catchment from April 1st to July 31st, 2010.

3.3.5 Total Discharge

The catchment generated discharge for the first time in January 2007 (Fig. 3.14). In spite of the significant variations of monthly discharge, an increasing trend was observed between 2005 and 2010. Obviously, this was connected to the gradual filling of the pore volume in the aquifer. However, in the second half of 2010 the increasing discharge amounts were caused by extremely high precipitation amounts. Until November 2009, monthly discharge was lower

compared to the precipitation amounts. In January, February, October and December 2010 the discharge was considerably higher than precipitation. In October 2010 high discharge values occurred as a result of saturation of the area after heavy rains in the previous month (Fig. 3.14). This is also true for December 2010 although precision of measurements of snowfall events is reduced due to wind drifting and due the fact that only liquid precipitation was registered with tipping buckets. This limitation became obvious in January and February 2010. The measured precipitation amounted to only 1.4 mm and 12.3 mm, respectively. In these two months, precipitation fell in the form of snow except for a melting period from January 17th to January 19th, 2010 and on February 21st, 2011. A continuous snow cover was observed between December 28th, 2009 and February 25th, 2010. The water equivalent stored in the snow cover was determined after snowfall events on three days in January and February 2010. We measured 14 mm on January 7th, 2010, 38 mm on January 14th, 2010 and 55 mm on February 3rd, 2010 (3 locations and 3 repetitions for each location). Therefore, the real precipitation amounts exceeded the values registered by tipping buckets significantly.
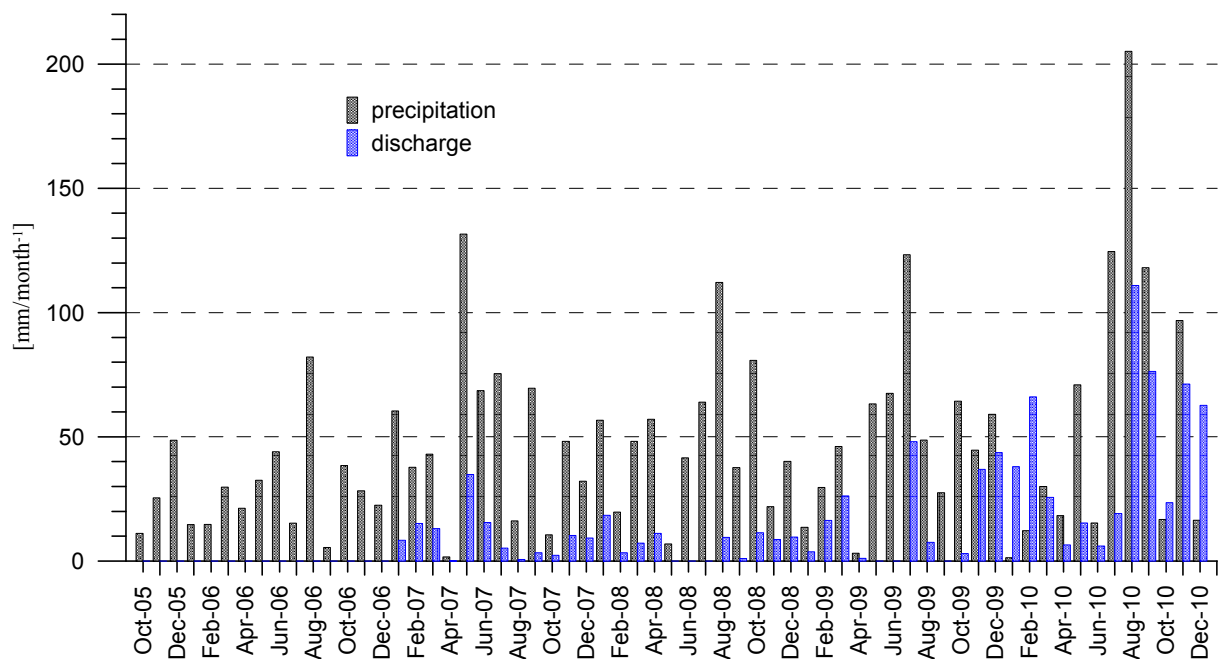

Fig. 3.14: Comparison of monthly precipitation and monthly discharge from the 'Chicken Creek' catchment.

During dry periods in the warm season the pond level dropped below the discharge level and at this time no discharge occurred indicating that evaporation from the pond surface exceeded the inflow to the pond (Fig. 3.15). After precipitation events, the water level in the pond increased rapidly. Since the beginning of 2007 permanent discharge has occurred during the cold seasons.
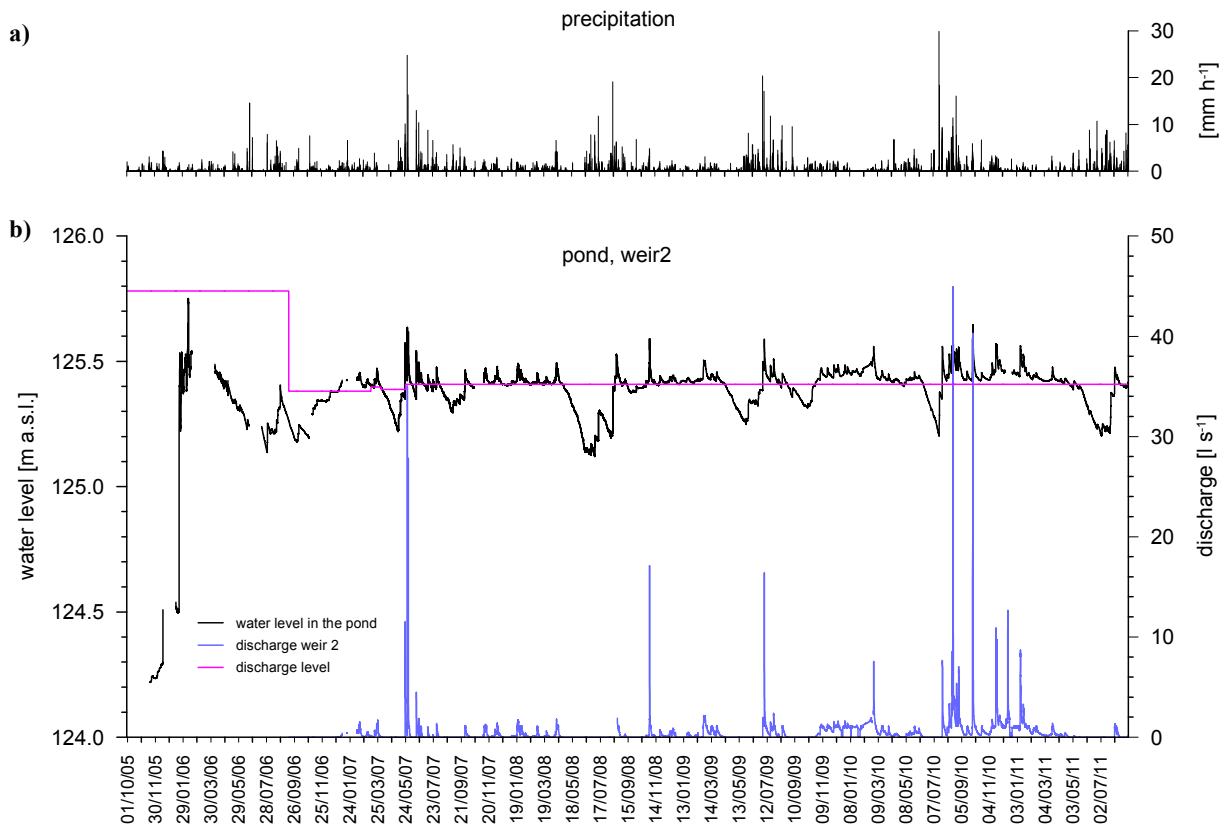
Fig. 3.15: Hourly values of a) precipitation, b) water level in the pond and discharge from the 'Chicken Creek' catchment (the purple line marks the discharge level, i.e., the lowest water level in the pond at which discharge can occur).

The pond displayed its storage function not only during times with water levels lower than the discharge level, but also during times when fast runoff occurred, for example as a result of intense rainfall or snowmelt events. The geometry of the weir enables the calculation of discharge from water level, but it also reduces the outflow from the pond. This is of major relevance for studies of fast runoff components. Therefore, the fast inflow into the lake, which was directly linked to precipitation events, was calculated taking the pond storage change, the precipitation onto the pond surface and the pond outflow into consideration. Figure 3.16 illustrates the storage effect of the pond for several rainfall events in August 2010. Because of the high water levels due to extreme precipitation, permanent discharge from the pond was observed. The inflow into the pond reached significantly higher values and showed higher dynamics. The pond storage attenuated these high dynamics, which produced a smoother curve of pond discharge.
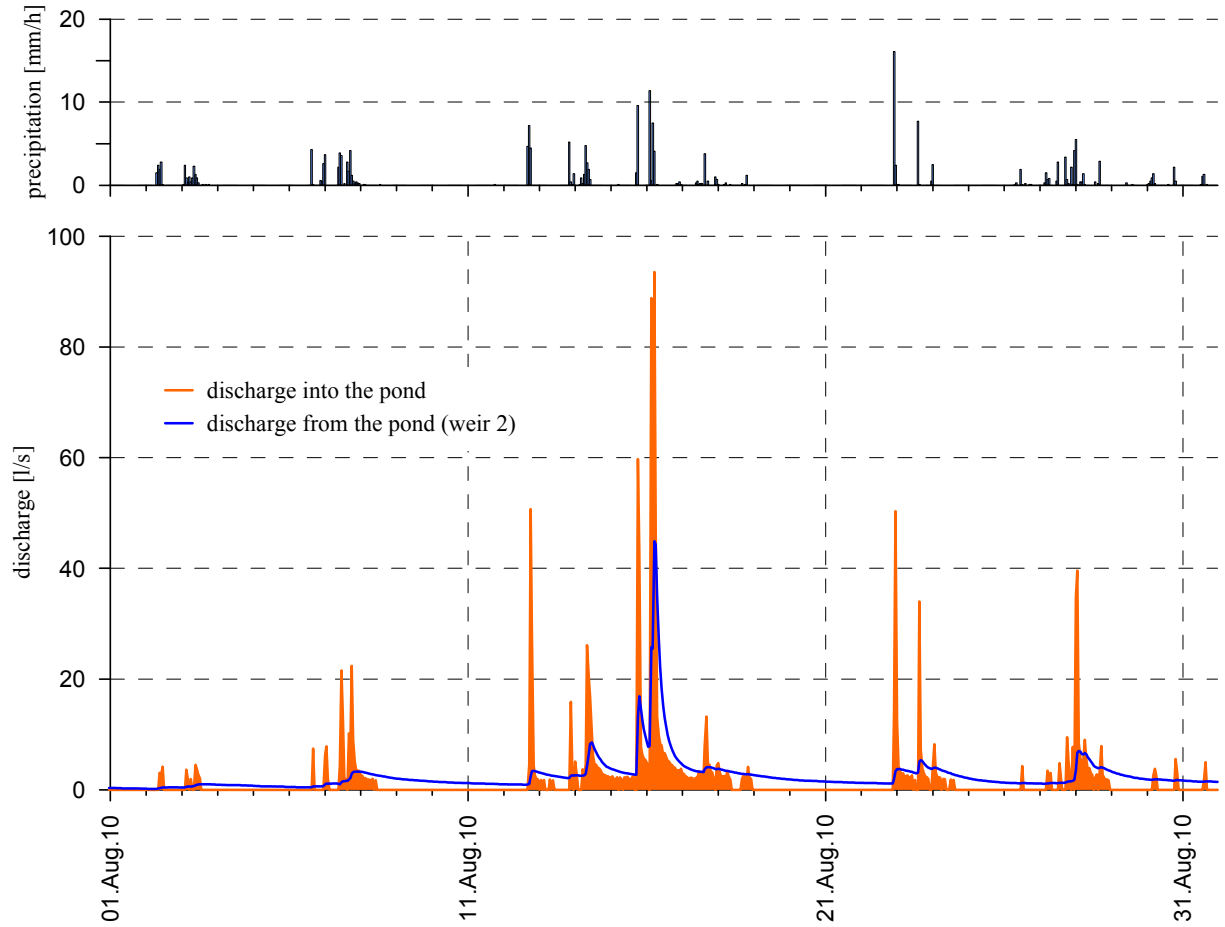
Fig. 3.16:   Hourly values of precipitation, discharge into the pond and discharge from the pond for August 2010.

3.3.6   Annual water balance

The annual water balance of the 'Chicken Creek' catchment (including the pond) was derived by subtracting discharge and ground water storage change from precipitation (corrected amounts, cf. Biemelt & Nenov, 2010) (Tab. 3.2), which are the directly measurable components of the water balance. The remaining values are the sum of actual evapotranspiration and change in unsaturated soil water storage. The potential evapotranspiration ($ET_{pot.}$) was calculated as grass reference evapotranspiration (Biemelt & Nenov, 2010) (Tab 3.2, column A). The annual sums varied between 644 mm in 2010 and 771 mm in 2007. The change in ground water storage was conservatively estimated using the annual changes in mean groundwater levels throughout the catchment multiplied by the mean difference in water content between field capacity and total pore volume. This difference was estimated at 25 Vol.-% (Tab. 3.2, column D). According to the measured soil moisture data and the retention curves annual changes of the water storage in the unsaturated zone are relatively small. Assuming mean changes of 5 Vol.-% and the decreasing extent of the

unsaturated zone (= soil surface - groundwater level), we estimated possible changes in water storage in the unsaturated zone (Tab. 3.2, column E).

Tab. 3.2: Water balance components of the 'Chicken Creek' catchment [mm year^{-1}].

year	A ET$_{pot.}$	B precipitation (corrected)	C discharge	D groundwater storage	E unsaturated zone storage	F Balance F=B-C-D-E
2006	794	403	0	-15	96	322
2007	771	667	118	138	84	327
2008	737	660	80	89	62	429
2009	697	665	187	80	45	353
2010	644	794	517	100	27	150

The remaining values of the water balance (Tab. 3.2, column F) varied between 150 mm in 2010 and 429 mm in 2008. These values represent actual evapotranspiration from the terrestrial parts of the catchment plus direct evaporation from the pond surface. Bormann (2011) reported actual evapotranspiration values between 157 mm and 268 mm in his model simulations of the 'Chicken Creek' catchment that did not include the pond. Simulation runs for the catchment with ten different models calculated actual ET values between 80 mm and 560 mm (Holländer et al., 2009) indicating large uncertainties in these model simulations. Therefore, the values given in column F in Table 3.2 lie well within this range of uncertainty.

3.4 Conclusions

Similar to systems after glacial periods, volcanic eruptions or fire, external and internal factors are very important in the initial phase. In the course of ecosystem development, certain equilibrium is achieved and the hydrological functions of the landscape unit emerge. Different processes run on different spatial and temporal scales. At any time during a precipitation event and at any spot within the catchment, the infiltration capacity of the substrate controls the infiltration rate. In the first years, surface runoff processes like runoff generation after precipitation events or snow melt dominated catchment hydrology inducing severe surface and gully erosion (cf. chapter 2). Hence, the increasing water level of the pond was dominated by surface runoff. The filling of the substrate pore volume took several years defined by the time required for the groundwater levels to reach the surface. In 2010, exceptional high amounts of rainfall led to an almost complete saturation of the sediment body. Surface and gully runoff that also drained the groundwater resulted in high discharge rates.

Only weak interactions between groundwater and the hydrological conditions at the surface were observed in the first years of ecosystem development. However, with rising groundwater levels, the main erosion gullies developed to (at least locally) perennial streams and drained groundwater. Overall, the external impacts described above exceeded trends caused by the development of the catchment itself. For example, a trend of decrease in total discharge due to increased transpiration by the increase of vegetation biomass (cf. chapter 5) was not observed during the study period.

The well determined water balance components are precipitation, total discharge and the storage of water in the saturated zone. This is not true for the actual evapotranspiration and the storage in the unsaturated zone, for which an exact quantification is difficult. For the future, additional studies to determine the actual evapotranspiration are needed to decrease the uncertainty in the water balance of the catchment.

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4.   Soil solution and water chemistry

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4.1   Introduction

Soil solution is a major medium for transport processes in soils and at the same time a reactive phase that interacts with other soil components via physical, chemical and biological processes (Schaaf et al., 1995; Schaaf, 2004). Soil solution composition can be used as an indicator for soil processes (Schaaf & Hüttl, 2005). Combined with water fluxes in the soil, the analysis of solutes enables the calculation of element fluxes in the soil compartment (Gast et al., 2001; Weisdorfer, 1999; Schaaf, 2001). Within the monitoring programme of the SFB/TR 38, soil solution is sampled at four grid points at the catchment site. The data should provide information on the spatio-temporal development of soil solution composition.

Also at the catchment scale, water is the main transport medium. The transport and cycling of water and elements integrates all the many abiotic and biotic compartments of an ecosystem over scales and relates processes and patterns to the overall ecosystem functioning (e.g., Jørgensen, 2009; Schaaf et al., 2011). The analysis of water and element cycling plays a key role in understanding the functioning, stability, elasticity and resilience of ecosystems. Clearly outlined budget areas are critical for this kind of studies. Catchments and watersheds are therefore frequently used as natural, fundamental spatial landscape units (Likens, 1999) offering the opportunity to quantify input parameters as well as the output from the system, and to calculate water and element budgets (Neal et al., 2003; Schleppi et al., 1998). Catchments integrate complex processes over several scales and can be treated as dynamic systems (Kirchner, 2009) functioning as 'a mirror of the landscape' or as an organism (Likens & Borman, 1995).

Atmospheric deposition is an important input pathway for elements into ecosystems (Littmann, 1994; Ihle, 2001; Schaaf, 2004). In initial ecosystems, nutrient availability is a limiting factor for primary production. In this context, atmospheric input might be an important source for nutrients for the vegetation at 'Chicken Creek'. In Central and Northern Europe, the high atmospheric nitrogen deposition can influence development of ecosystems. In nutrient-limited ecosystems, like dry grassland ecosystems, atmospheric nitrogen inputs have a large impact on primary productivity and can modify species composition, vegetation dynamics and the direction of succession (Bakker & Berendse, 1999).

The point of pond discharge at the lower end of the catchment is the only path for flowing water and solutes to leave the catchment and can therefore be used to calculate the total output from the catchment. Water and element budgets were calculated from bulk precipitation/deposition and pond discharge/output.

4.2   Materials and methods

Bulk deposition was sampled initially at 18 grid points (cf. Schaaf et al., 2010). Since the data of bulk precipitation and rainwater composition showed no consistent spatial trend over the catchment area, sampling points were reduced to 9 in November 2009 (Fig. 4.1).
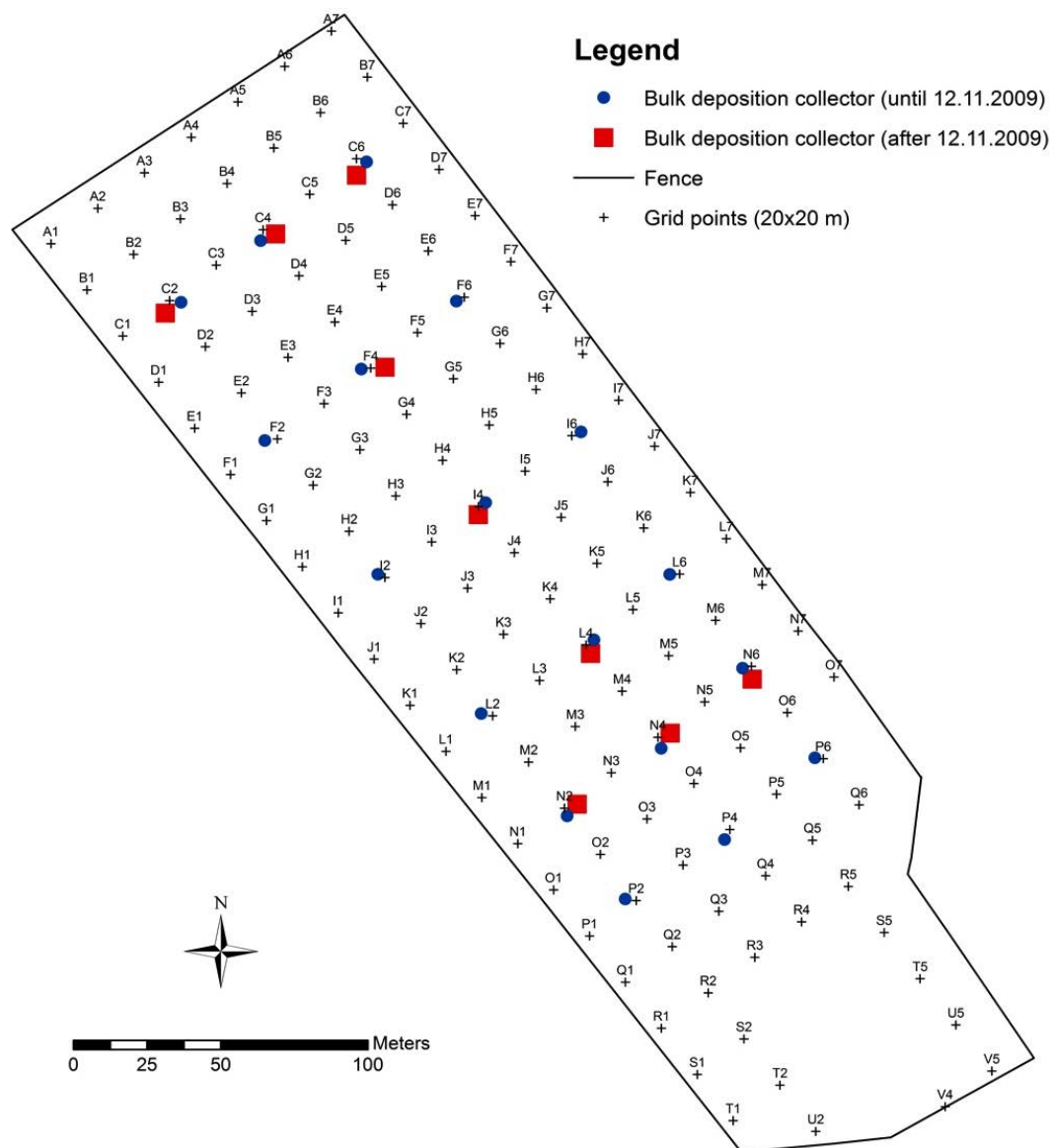

Fig. 4.1:   Position of bulk deposition samplers (old and new types) within the fenced area of 'Chicken Creek' catchment.

New sample collectors (type RS200 UMS) (Fig. 4.2) were installed and sampled biweekly. The collector surface of the new samplers is 314 cm^2 in 1 m height. Samples are stored in a 5 L bottle at the bottom of a tube. Further details on sample handling and analyses are described in Veste & Schaaf (2010).
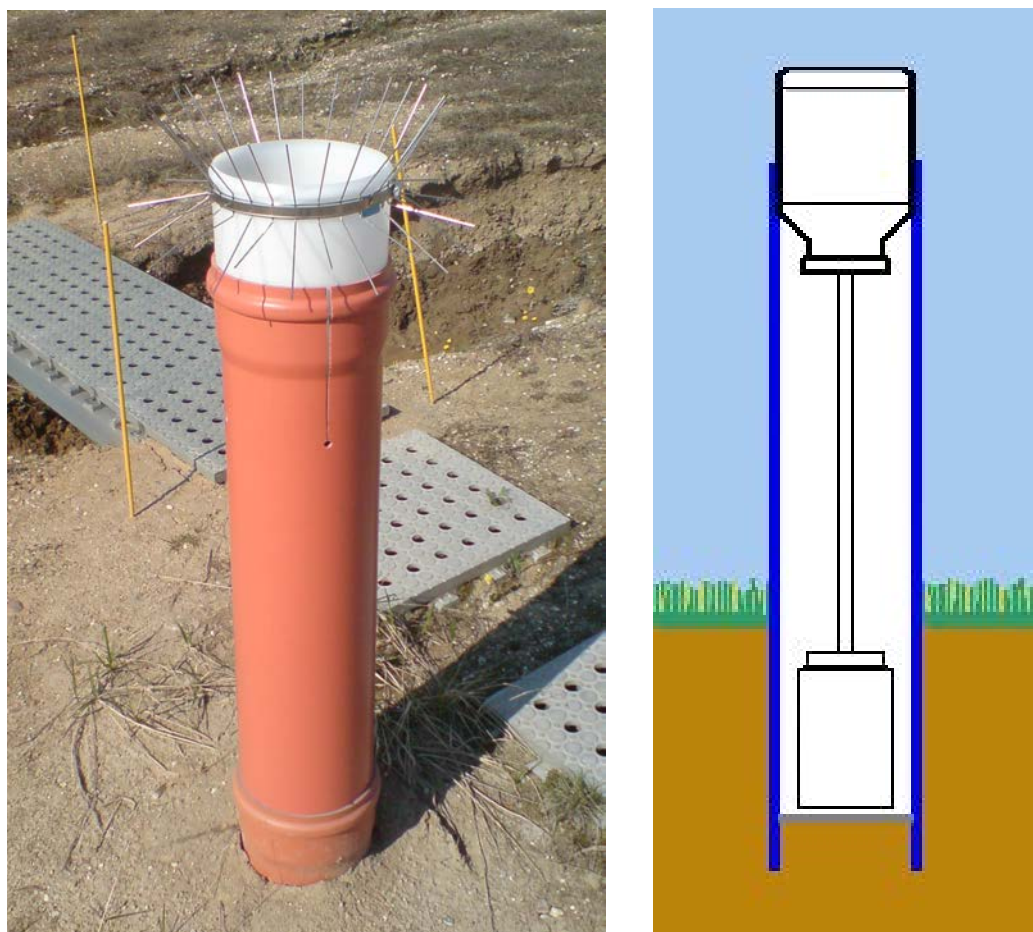

Fig. 4.2: Picture and construction (schematic) of the RS200 bulk deposition sampler equipped with bird protection rings.

Sampling at the two weirs (upper weir/weir 1, lower weir/weir 2) and one of the flumes (flume 2) (cf. chapter 3) and analyses of the water samples were continued as described by Biemelt et al. (2010). Sampling and analyses of soil solution was continued as described by Schaaf (2010). Due to the very high groundwater levels especially in 2010 (cf. chapter 3), the sampling at the lower depths had to be discontinued.

Element fluxes were calculated from amounts of precipitation and discharge (cf. chapter 3) and the corresponding elemental concentrations. Inputs to the catchment were calculated by multiplying the mean element concentrations (in mg L^{-1}) in bulk deposition samples with the mean rainfall amounts (in L m^{-2}) of the 18 or 9 samplers, respectively. For the catchment

output, pond discharge was calculated as L m^{-2} using the total surface area of the catchment (60,898 m^2). Element concentrations in the biweekly samples were then multiplied by the cumulated discharge amounts for the respective period. For element budgets all fluxes were summed and calculated as annual fluxes (kg ha^{-1} yr^{-1}) for hydrological years (November, 1st to October, 31st).

4.3   Results and discussion

Soil solution composition varied considerably between the four soil pits at the beginning of the observation period. Compared to these spatial variations, differences in soil depth were less pronounced. Main components of all sampled soil solutions were calcium (Ca), magnesium (Mg), bicarbonate (HCO$_3$) and sulfate (SO$_4$). Due to the carbonate content of the substrates, mean pH values varied between 7.0 and 8.4 in all samples (Fig. 4.3a).

The two parallel samples of each pit in 30 cm depth showed similar composition indicating that spatial heterogeneity was small at the scale of ~1 m. In general, ionic concentrations increased with soil depth. Concentrations of Ca (Fig. 4.3c), Mg (not shown) and SO$_4$ (Fig. 4.3d) decreased over the first year of measurements as reflected also in the electrical conductivity (EC) (Fig. 4.3b), and especially the spatial variability between the four pits decreased as shown by the error bars in Figure 4.3. Traces of gypsum in the substrates may be a source for both Ca and SO$_4$ in the initial phase of leaching. Thin white precipitations found at soil surfaces especially during dry periods were described as gypsum crusts (Schaaf et al., 2010). Dultz & Kühn (2005) described similar phenomena for Chernozems of Central Germany and explained the occurrence of gypsum by former high sulphur deposition due to unfiltered lignite-fired power plants during GDR times together with low precipitation and leaching. In contrast bicarbonate concentrations showed an overall trend of increasing values indicating decalcification (Fig. 4.3f). Compared to the very low C$_{org.}$ contents in the substrates dissoveld organic carbon (DOC) concentrations were at a high level (Fig. 4.3e) indicating a potential to mobilize this inherited organic carbon in the substrates.

A comparison of the ionic charge balances of soil solutions for the first year of measurements (2007/2008) to the comparable period in 2009/2010 visualizes the dramatic changes in soil solution composition (Fig. 4.4). Whereas in the first year, a strong gradient of increasing concentrations with depth was visible, overall ionic charge was reduced and similar for both depths in 2009/2010. As mentioned, this reduction is mainly due to decreases in Ca and SO$_4$ concentrations. If we assume gypsum as the major source for sulfate leaching, the data show that in the beginning gypsum dissolution accounted for 88% and 97% of the Ca leaching in the two soil depths, respectively. Only three years later these values were reduced to 43-48%. Therefore, increasing carbonate weathering became the dominating source for Ca as also indicated by the increasing bicarbonate concentrations.
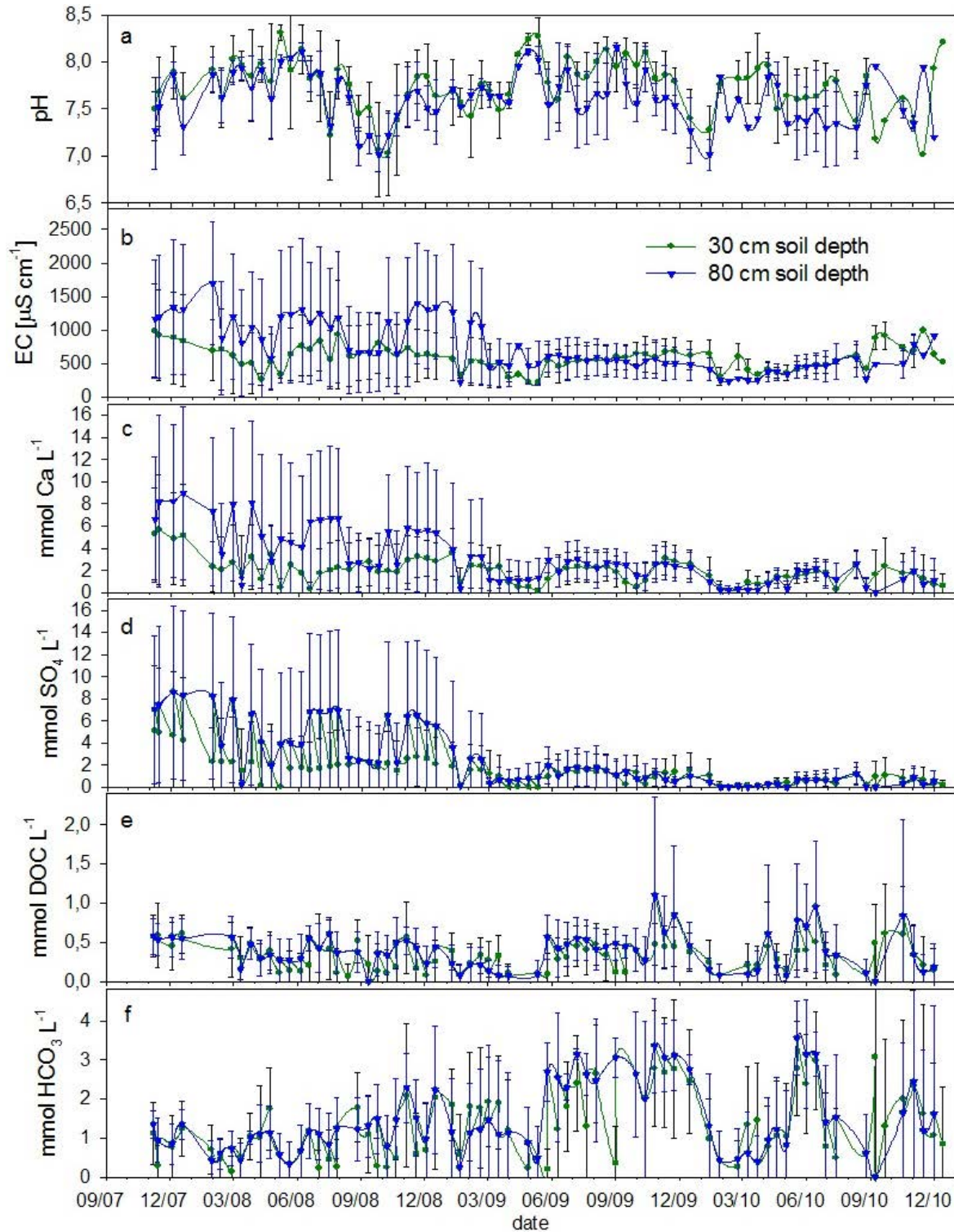
Fig. 4.3:   Soil solution composition in two soil depths (symbols represent mean values of the four soil pits, error bars indicate standard deviation).

pH values in all water samples from the two weirs and the flume varied between 6.8 and 8.4. Abrupt drops and increases in pH values of up to one pH unit per day could be observed (Fig. 4.5). This was most pronounced for the flume samples, but also the weir samples

followed similar trends. In general, ion concentrations in water samples from the three sampling sites were in the order flume (flume 2) > upper weir (weir 1) > lower weir (weir 2) as indicated by EC (Fig. 4.5). Whereas EC values at the lower weir (pond discharge) showed only little temporal variation, considerable short-term peaks in EC of up to +/- 1500 µS cm^{-1} per day were observed in samples from the upper weir and the flume. EC values of samples from the upper weir showed typical temporal patterns with increasing values during drought periods and sharp drops after precipitation events.
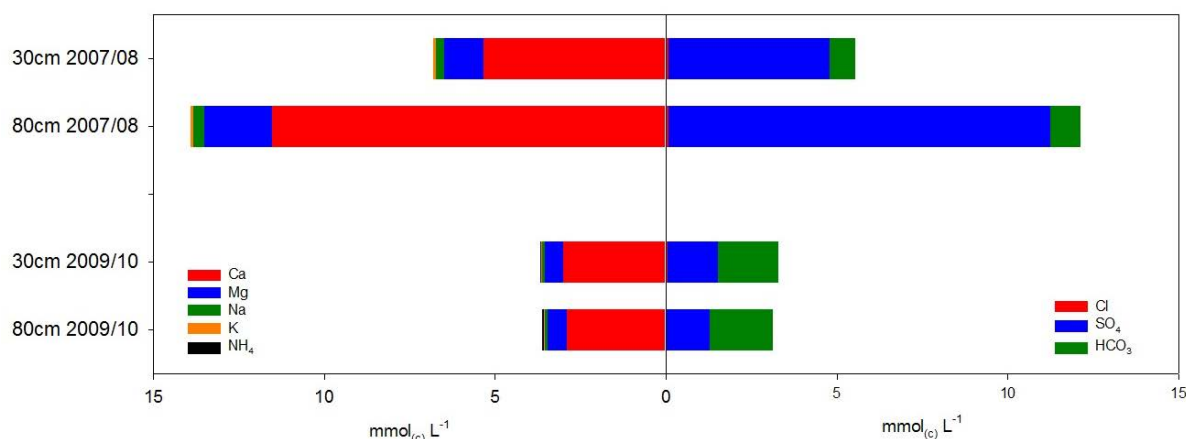

Fig. 4.4: Ionic charge balance for soil solution from 30 cm and 80 cm depth in two sampling periods (August 2007 – July 2008; August 2009 – July 2010).
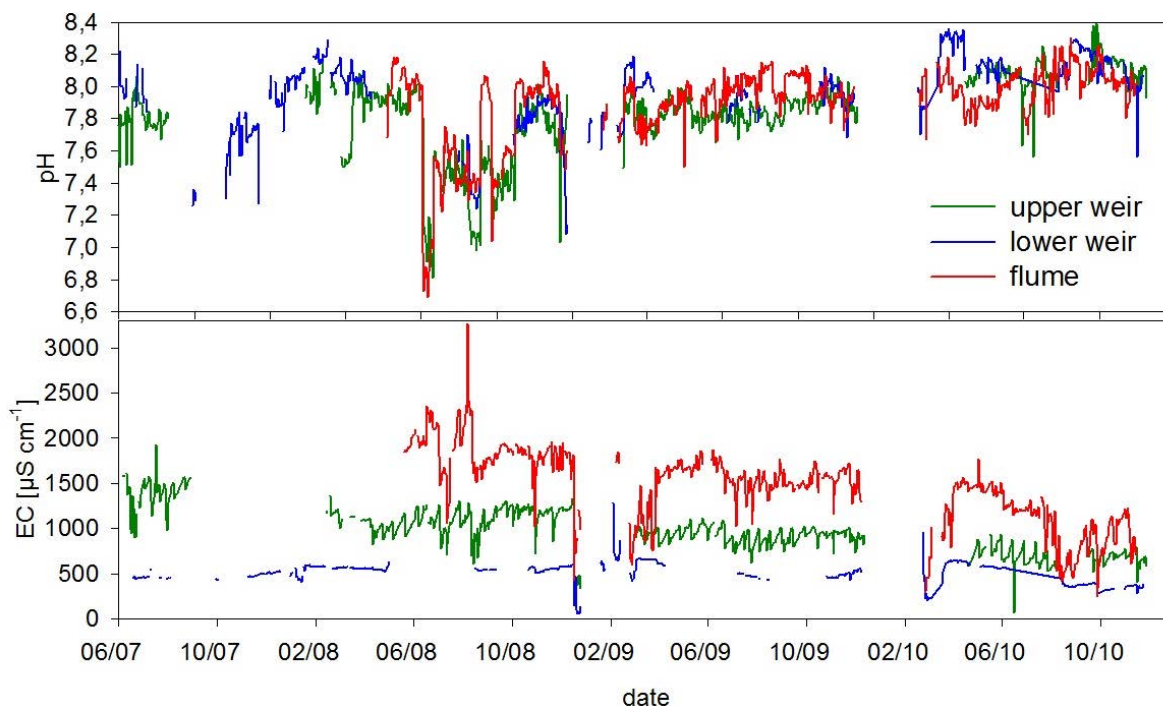

Fig. 4.5: Daily values of pH and electrical conductivity in water samples from the two weirs and the flume.

Similar to the soil solutions, the main components in all water samples collected at the different weirs were Ca, Mg, HCO_3, SO_4 and DOC (Fig. 4.6). Generally, no distinct temporal trends could be detected during the observation period (Fig. 4.7), except for a decline in Ca (Fig. 4.7a), Mg (Fig. 4.7b) and SO_4 (Fig. 4.7f) concentrations in the upper weir and flume samples, reflecting the same trends as observed for the soil solutions. Ammonium showed a strong peak in the flume samples in summer 2008 (Fig. 4.7d), but the overall concentrations were very low. At all sampling points, nitrate concentrations were continuously below detection limit (< 0.2 mmol NO_3 L^{-1}).
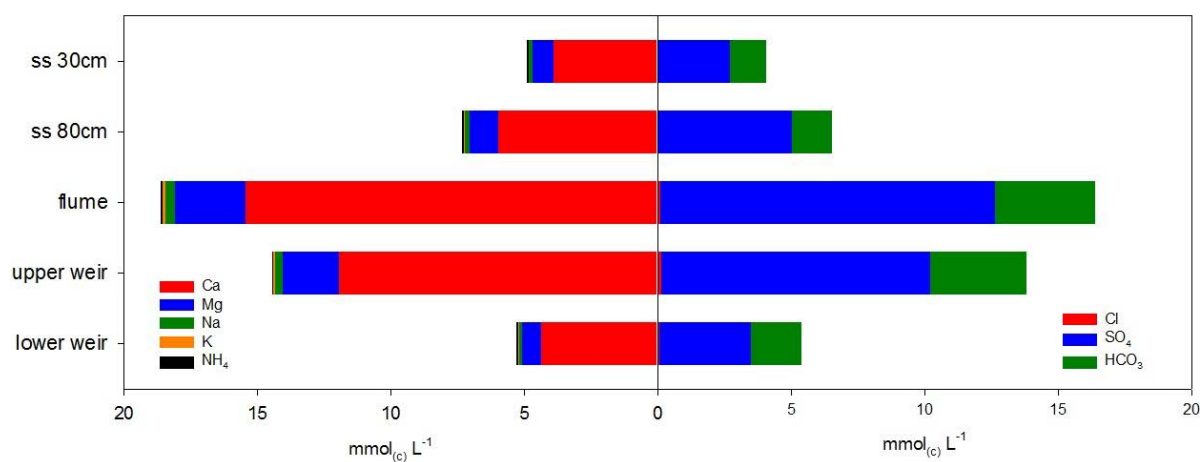

Fig. 4.6:   Ionic charge balance for water samples from the two weirs, the flume and soil solution from 30 cm (ss 30cm) and 80 cm (ss 80cm) depth (mean values for the whole observation period 2007 – 2010).

From element concentrations measured in bulk precipitation and pond discharge element budgets were calculated for the catchment (Tab. 4.1). Bulk deposition inputs varied relatively little over the years and were very well in the range reported by other authors for the region (Gast et al., 2001; Schaaf & Hüttl, 2005; Wellbrock et al., 2005).

Output rates indicate that the catchment acted as a net source for calcium, magnesium, sulfur and inorganic carbon released by carbonate weathering and gypsum dissolution, but as a strong sink for nitrogen. Output rates between the years varied considerably, mainly due to the changes in discharge rates as reflected by the chloride budget (Tab. 4.1). The increase in total inorganic carbon (TIC) output was a result of both discharge and bicarbonate concentrations increases.
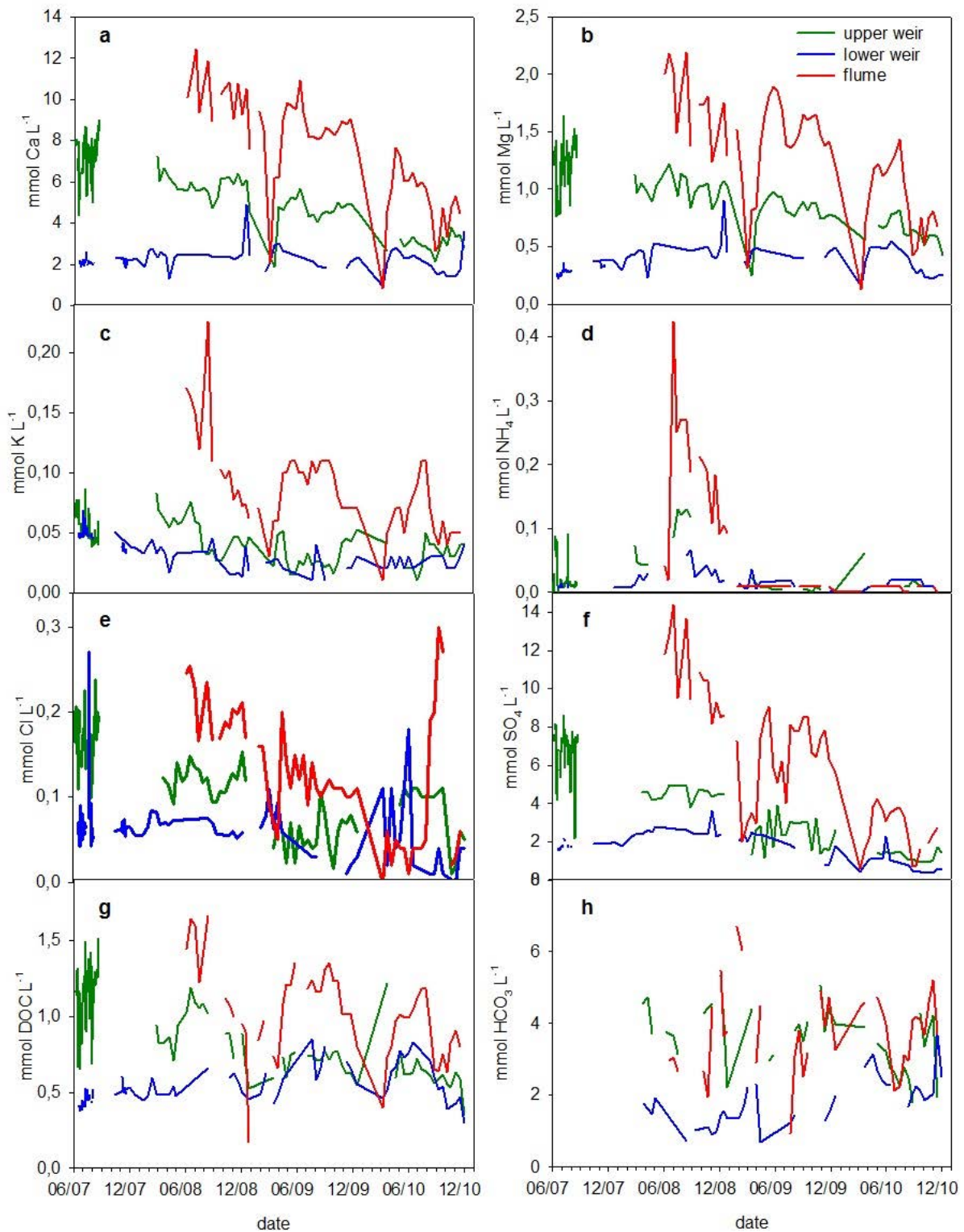
Fig. 4.7: Ion concentrations in biweekly mixed water samples from the two weirs and the flume.

Tab. 4.1: Element budgets of the catchment (values in (in kg ha^{-1} yr^{-1}) calculated from bulk deposition (IN) and pond discharge (OUT).

year	Ca	Mg	K	Cl	SO$_4$-S	NH$_4$-N	NO$_3$-N	DOC	TIC
2008*									
IN	7.2	1.2	3.9	11.4	16.1	13.5	8.7	18.2	3.5
OUT	76.5	8.2	1.1	1.9	61.3	0.3	b.d.	5.0	12.6
2009*									
IN	6.4	0.9	2.1	9.4	8.8	8.1	7.0	15.1	4.7
OUT	115.9	13.0	1.1	2.4	80.7	0.3	b.d.	9.2	30.4
2010*									
IN	7.5	0.8	2.3	5.3	10.2	10.7	6.8	14.6	7.6
OUT	290.5	33.2	4.5	9.3	107.4	0.7	b.d.	29.8	110.5

*hydrologic year, e.g., 2008 = November, 1st 2007 – October, 31st 2008; b.d. = concentrations below detection limit

4.4 Conclusions

Soil solution composition showed considerable changes over the investigation period 2007 – 2010. The large spatial variation (both horizontally between sampling sites and vertically with soil depth) in the first year of sampling indicates non-equilibrium conditions. With time, overall concentrations decreased significantly, mainly for calcium, magnesium and sulfate, whereas bicarbonate increased. The spatial variation strongly decreased over the investigation period, probably an indication for more equilibrium controlled conditions.

Dissolution of the gypsum contents of the parent material seems to be the dominating process controlling initial soil solution composition. Since the gypsum contents are very low, decreasing sulfate concentrations indicate most of the gypsum is dissolved and mobilized within a few years, although gypsum precipitates are still visible at the soil surface and along the walls of the stream corridors during dry summer periods. With increasing vegetation cover and litter input to the soil, carbonate weathering increased and became the main control for calcium concentrations in soil solution. This finding is supported by soil column studies with added litter in subproject A2 (Zimmermann & Schaaf, 2010).

In general, water chemistry of the weir and flume samples corresponds well to soil solutions and shows similar temporal trends for calcium, magnesium and sulfate. The overall concentrations at the flume and the upper weir are much higher compared to soil solutions and pond water at the lower weir.

The element budgets were strongly influenced both by the changes in water chemistry and in discharge amounts at the lower weir. Whereas the inputs to the catchment via bulk deposition did not vary too much over the years (mainly due to precipitation variation), the output from the catchment showed strong and continuous increases for most elements. This is mainly governed by the strong increases in discharge, especially in the very wet year 2010 (cf. chapter 3). The catchment is a strong source for calcium, magnesium, sulfur and both organic and inorganic carbon. Only for nitrogen, the catchment acts as a strong sink, typical for strongly nitrogen limited systems.

Acknowledgements

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5.   Vegetation patterning

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5.1   Introduction

The vegetation development in the artificial catchment 'Chicken Creek' in Northeast Germany (Gerwin et al., 2009) has been documented annually since the beginning of ecosystem development in autumn 2005. A regular net of permanent floristic monitoring plots with a distance of 20 m had been established in October 2005 (Zaplata et al., 2010, 2011). We recorded for each plot all the vascular plant species, for each vascular species the current cover, and the total bryophyte cover.

The vegetation in the catchment developed quickly, starting in 2005 with a few seeds of a few species in the soil seed bank (Zaplata et al., 2010) as well as with seeds of mainly anemochorous species invading from outside (Zaplata et al., 2010). In the course of time the total vegetation cover consistently increased with a maximum of about 60% in 2010 and a maximum cover of vascular species of about 39% in 2009. In 2010 a total of 149 vascular species occurred on the plots. Spatiotemporal species distribution patterns emerged and are in the focus here.

5.2   Materials and methods

Each plot of the regular grid net has a size of 5 m x 5 m with four subplots of 1 m² in the corners. For estimating the degree of vegetation cover we used the following classification scale: < 0.1%, 0.5%, from 1% to 10% in steps of 1%, from 10% to 30% in steps of 5%, and from 30% onwards in steps of 10%. Vegetation records were carried out first in October 2005 (immediately after the beginning of the succession, with preliminary 1 m² subplots) and were continued afterwards in each summer (mid-July till the end of August, 2006 to 2010).

Additionally, for *Brachypodium sylvaticum* the population was analyzed in 2010 by mapping young plants (progeny) around vigorously established (parent) individuals. Two groups of *B. sylvaticum* patches were differentiated: one with only adult plants (non-rejuvenating group) and one with adult as well as young plants growing in their vicinity (rejuvenating group). The maximum distance between neighbouring *B. sylvaticum* plants did not exceed 12 cm. Substrate parameters from mixed soil samples (0-25 cm depth, mixed from 10 soil cores) taken at the perimeter of a 1.2 m circle around a selected *B. sylvaticum* parent plant on

October 5th, 2010 were analysed for their carbonate content (according to Scheibler), pH(H$_2$O), soil water content (gravimetric determination by drying at 105 °C), and texture (by means of wet screening and fractionation with the Köhn method). Seed rain trapping was carried out in a second campaign along the border of the catchment between August 1st, 2008 and October 15th, 2009, using the same method as in 2005/2006 (Zaplata et al., 2010).

5.3   Results and Discussion

5.3.1   Results

Seed rain

Seed rain is the most important source for plant species colonizing new areas. At the beginning of the ecosystem development, seed input was high in terms of the amount of invading seeds, but low in terms of species numbers. In 2006 much more seeds were trapped at the eastern side than at the western side (Fig. 5.1). We found a strong positive correlation with wind direction indicating that the catchment started to serve as a source for *C. canadensis* seeds at that time (Zaplata et al., 2011).
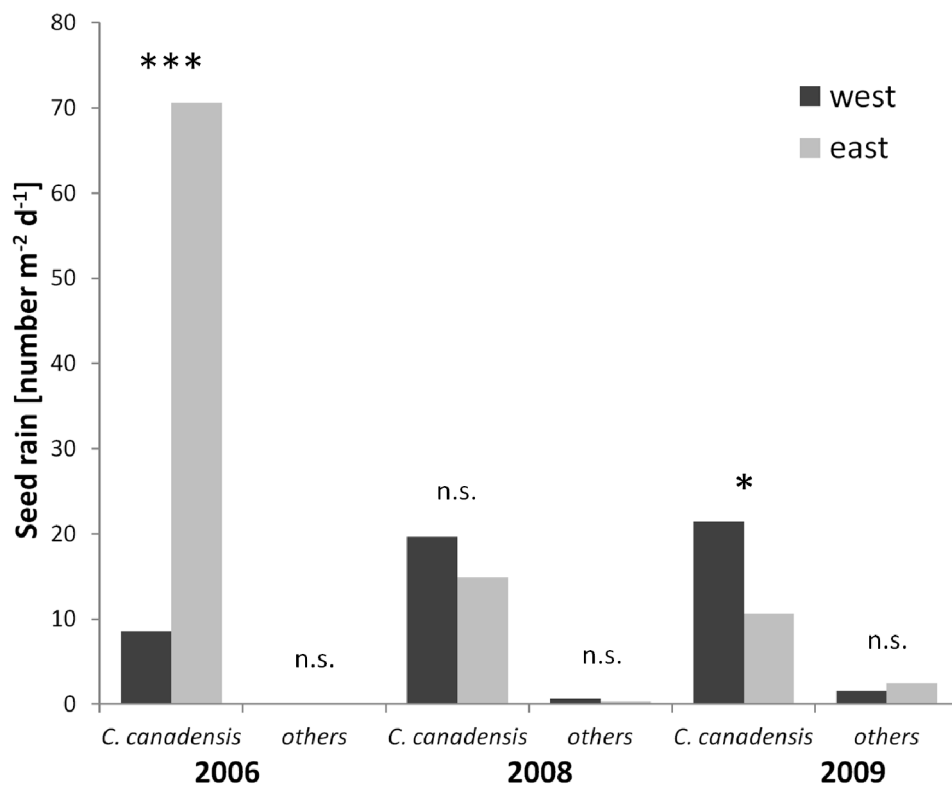

Fig. 5.1:   Development of seed rain (mean numbers per m² and day) at two catchment sides from 2006 to 2009: *Conyza canadensis* versus all other species (* $p < 0.05$, *** $p < 0.001$, n.s.: not significant).

From 2006 to 2008 and 2009 the amount of seed rain declined strongly due to a pronounced decrease of *C. canadensis* seeds. The number of seeds of other species increased, but on a rather low level. The seed rain pattern of *C. canadensis* changed markedly: from very high seed numbers along the eastern side, but very low along the western side in 2006 to lower quantities, but with certain dominance on the western side during the seed shedding periods of 2008 and 2009 (Fig. 5.1).

Vegetation development

The overall area covered by vascular plants increased substantially during the first years (Tab. 5.1) and reached a preliminary maximum of around 39% in 2009. The number of vascular species per 5 m x 5 m plot increased from almost zero in 2005 and around ten in 2006 to nearly 35 in 2010. On all permanent plots together the total number of vascular plant species increased from 18 in 2005 to 149 in 2010.

Tab. 5.1: Mean total vascular species cover and mean vascular species number per plot.

Year	Mean total cover (lower and upper value of the 95%-confidence interval)	Mean species number (lower and upper value of the 95%-confidence interval)
2005 (360 subplots)	**0.0** (0.0, 0.1)	**0.2** (0.1, 0.2)
2006 (119 plots)	**2.3** (1.7, 2.9)	**9.7** (9.0, 10.5)
2007 (120 plots)	**8.5** (7.3, 9.7)	**15.0** (14.0, 16.0)
2008 (119 plots)	**14.7** (12.8, 16.6)	**21.4** (20.4, 22.5)
2009 (119 plots)	**38.5** (34.2, 42.8)	**29.8** (28.3, 31.2)
2010 (119 plots)	**29.1** (27.1, 31.2)	**34.7** (33.3, 36.2)

Patterns of species distribution in space and time

Figures 5.2 to 5.16 show the distribution patterns of selected plant species, representing some common spatiotemporal patterns at the study area. The dot sizes represent the vegetation cover of the species on each single plot (1 m² in 2005, 25 m² from 2006 to 2010).

White goosefoot (*Chenopodium album*) (Fig. 5.2) is an annual ruderal species. Although only very few individuals were found at the plots in 2005, in 2006 this species was almost omnipresent, albeit with low cover values. In the following years *C. album* decreased in cover and almost disappeared in 2010.

Canadian horseweed (*Conyza canadensis*) (Fig. 5.3), being also an annual species, showed a similar behaviour. It reached maximum abundance one year later and with much higher vegetation cover. In 2010 it was still present at nearly all plots.

Cheat grass (*Bromus tectorum*) (Fig. 5.4) is an annual pioneer species typical of dry and nutrient poor sites. It did not colonize the catchment before 2007. Afterwards it appeared frequently in the study area.

Haresfoot clover (*Trifolium arvense*) (Fig. 5.5) is an annual pioneer species characteristic of open rough meadow communities and of dry places. Its dynamics were similar to *B. tectorum*, but it was totally missing during the first years at least at the monitoring plots. In the following years its dominance increased rapidly and steadily, reaching 100% frequency (all plots occupied) in 2009 and 2010 with the highest vegetation cover of all vascular plants within the catchment.

Wild carrot (*Daucus carota*) (Fig. 5.6) and Viper's bugloss (*Echium vulgare*) (Fig. 5.7), both biannual to short-living perennial species, started their establishment in 2005 with few individuals and increased steadily both in frequency and in vegetation cover reaching a maximum in 2010.

Evening primrose (*Oenothera parviflora* agg.) (Fig. 5.8) as well as Proliferous pink (*Petrorhagia prolifera*) (Fig. 5.9) showed a similar increasing tendency as to *D. carota* and *E. vulgare*, but on much lower levels.

Hairy vetch (*Vicia hirsuta*) (Fig. 5.10), Stinking hawk's-beard (*Crepis foetida*) (Fig. 5.11) and Spotted knapweed (*Centaurea stoebe*) (Fig. 5.12), although being annual or short-living perennial species, established not before the 3rd or 4th year of development with a certain level of frequency. *V. hirsuta* is one of the species that clearly shows a step by step (year to year) immigration from the surrounding area into the catchments.

Wood small-reed (*Calamagrostis epigejos*) (Fig. 5.13) is a perennial pioneer grass. Its spreading is supported by numerous lightweight anemochorous seeds as well as by rapidly growing root suckers. This species has been present on the plots since the very beginning and belongs to the most dominating species in the catchment in 2010.

False brome (*Brachypodium sylvaticum*) (Fig. 5.14), a characteristic species of European deciduous forests, appeared first in 2006. Its cover increased rapidly until 2010.

Hairy greenweed (*Genista pilosa*) (Fig. 5.15) established later, building dense patches due to lateral growth.

The pioneer tree species Silver birch (*Betula pendula*) (Fig. 5.16) occurred on the grit net plots for the first time after four years, whereas Scots pine (*Pinus sylvestris*) and Black locust (*Robinia pseudoacacia*) have been present on the plots since the beginning.

Fig. 5.2: Distribution and cover of *Chenopodium album* agg., 2005-2010; green dots depict covers between 0.1 and 4%.

Fig. 5.3: Distribution and cover of *Conyza canadensis*, 2005-2010; green dots depict covers between 0.1 and 10%.

Fig. 5.4: Distribution and cover of *Bromus tectorum*, 2005-2010; green dots depict covers between 0.1 and 3%.

Fig. 5.5: Distribution and cover of *Trifolium arvense* 2005-2010; green dots depict covers between 0.1 and 80%; scaling deviates from all other species presented.

Fig. 5.6: Distribution and cover of *Daucus carota*, 2005-2010; green dots depict covers between 0.1 and 6%.

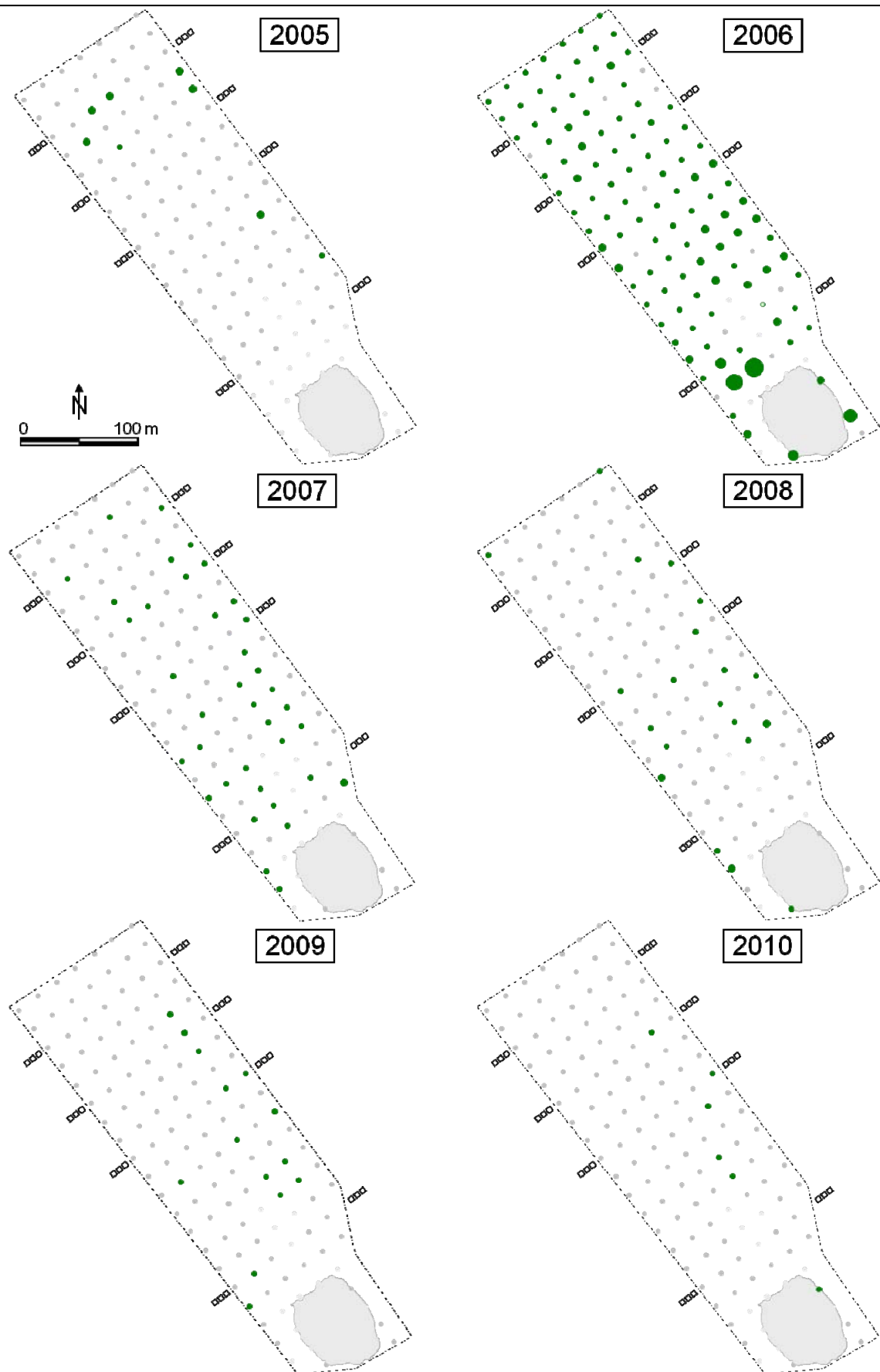Fig. 5.7: Distribution and cover of *Echium vulgare*, 2005-2010; green dots depict covers between 0.1 and 6%.
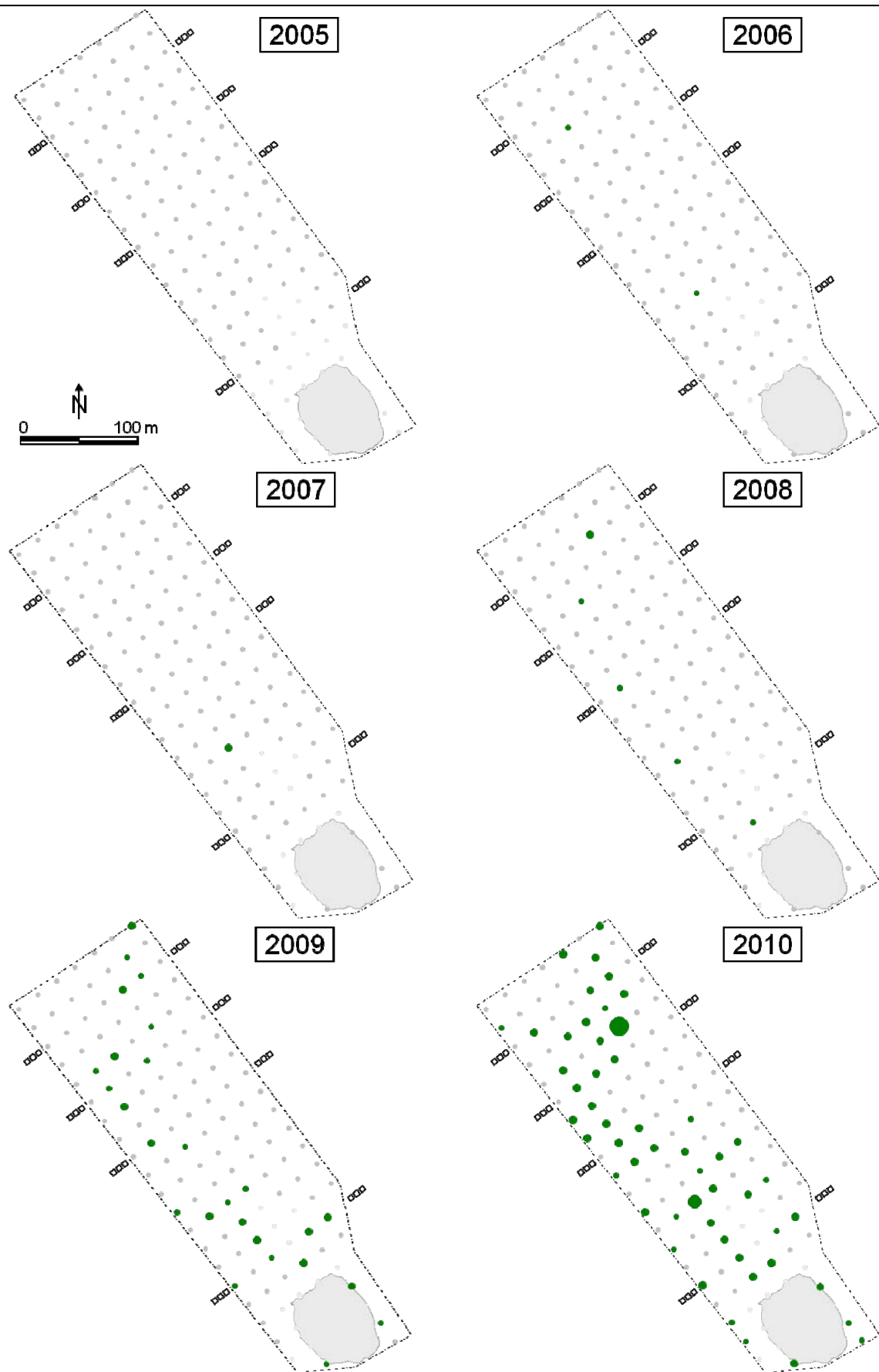
Fig. 5.8: Distribution and cover of *Oenothera parviflora* agg., 2005-2010; green dots depict covers between 0.1 and 4%.
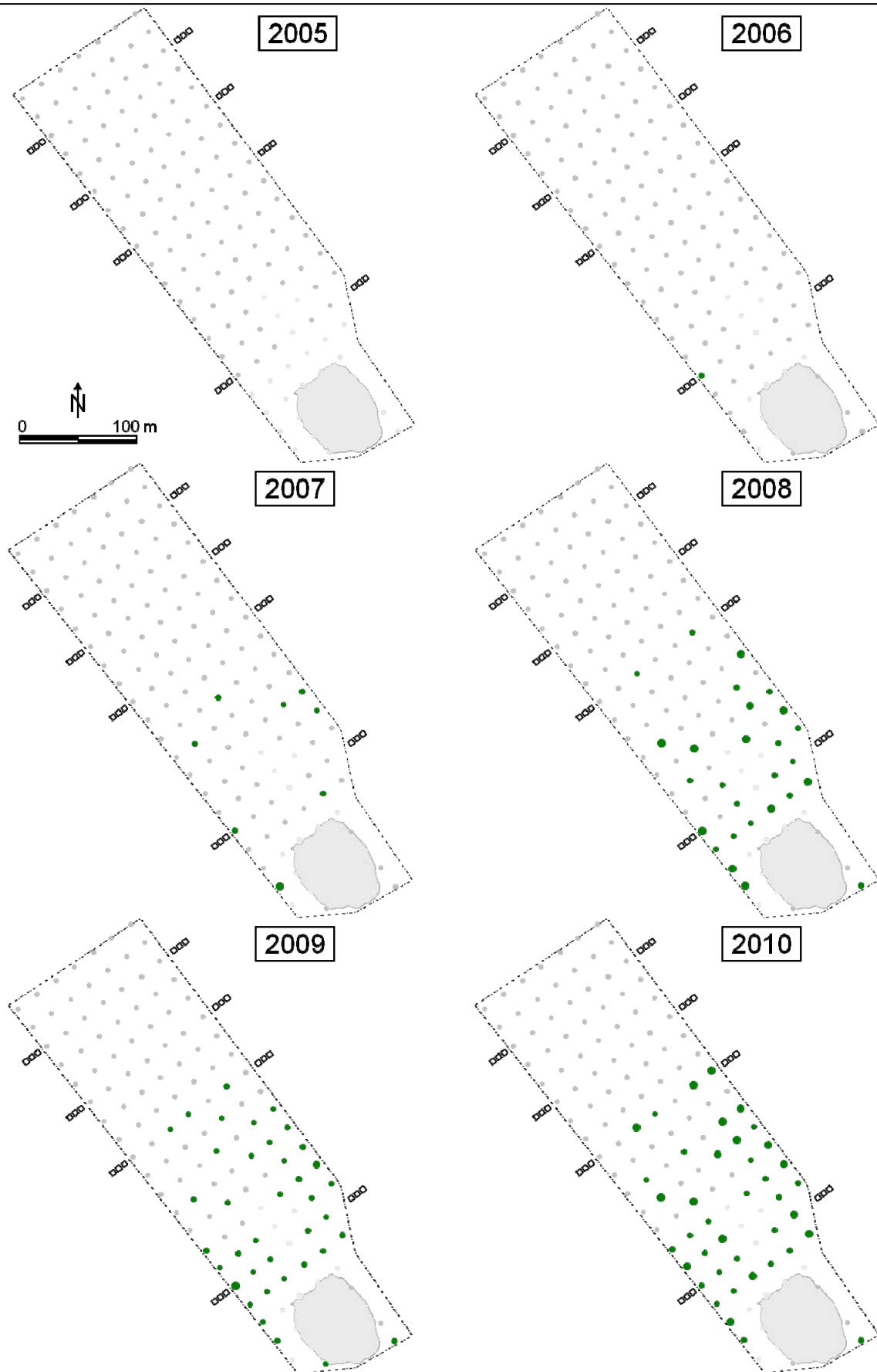
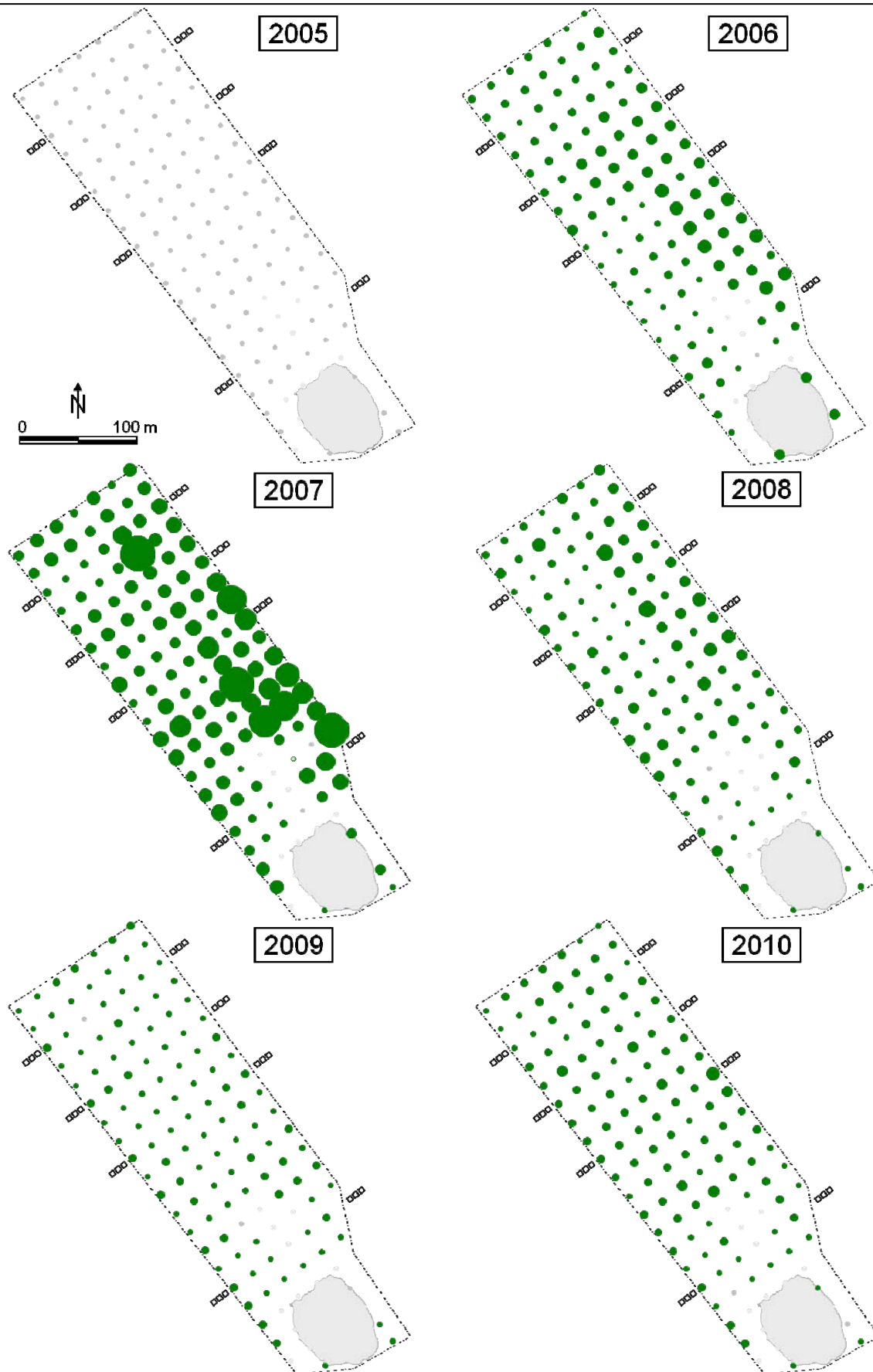Fig. 5.9: Distribution and cover of *Petrorhagia prolifera*, 2005-2010; green dots depict covers of 0.1 and 0.5%.
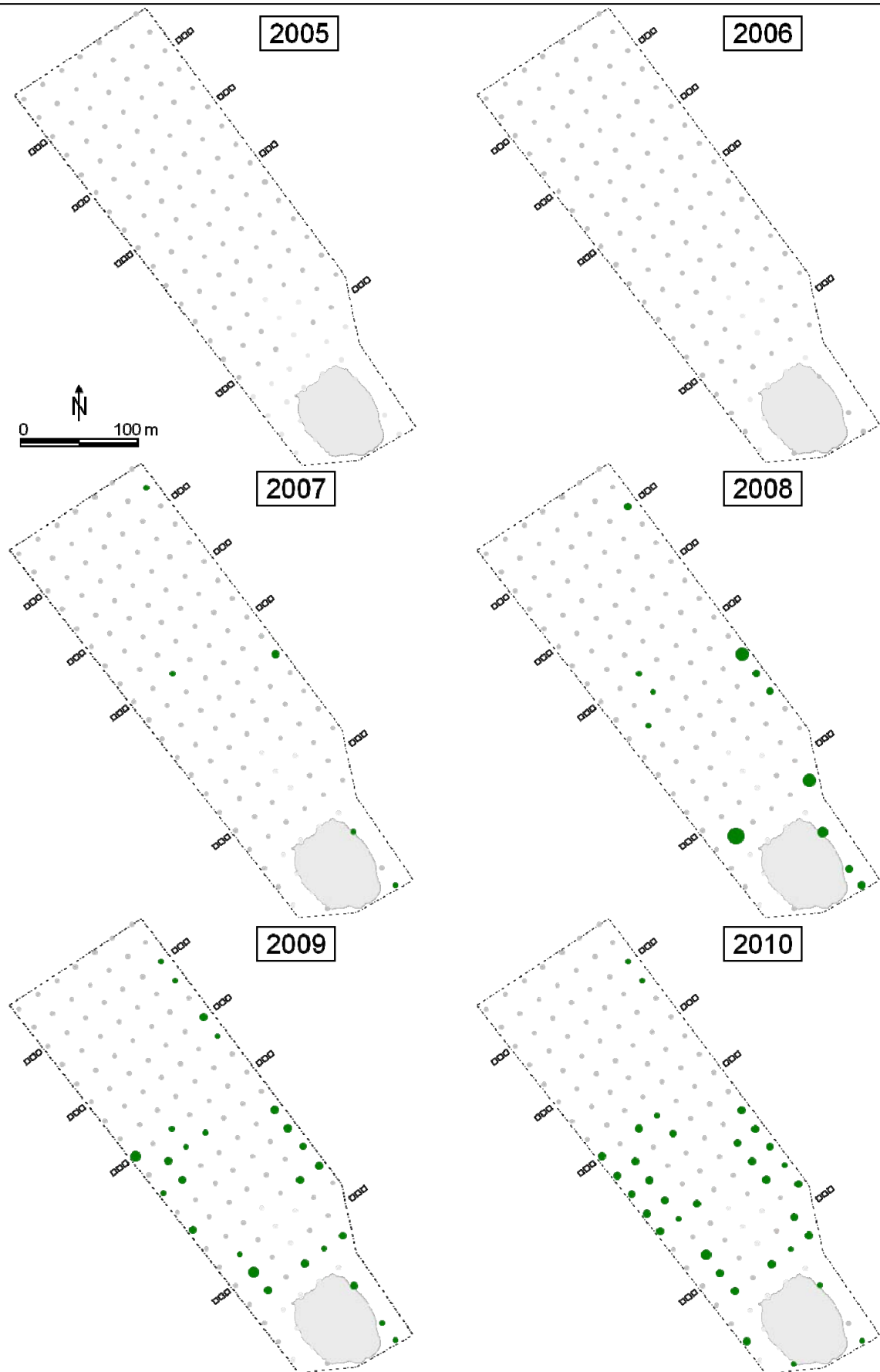
Fig. 5.10: Distribution and cover of *Vicia hirsuta*, 2005-2010; green dots depict covers between 0.1 and 3%.
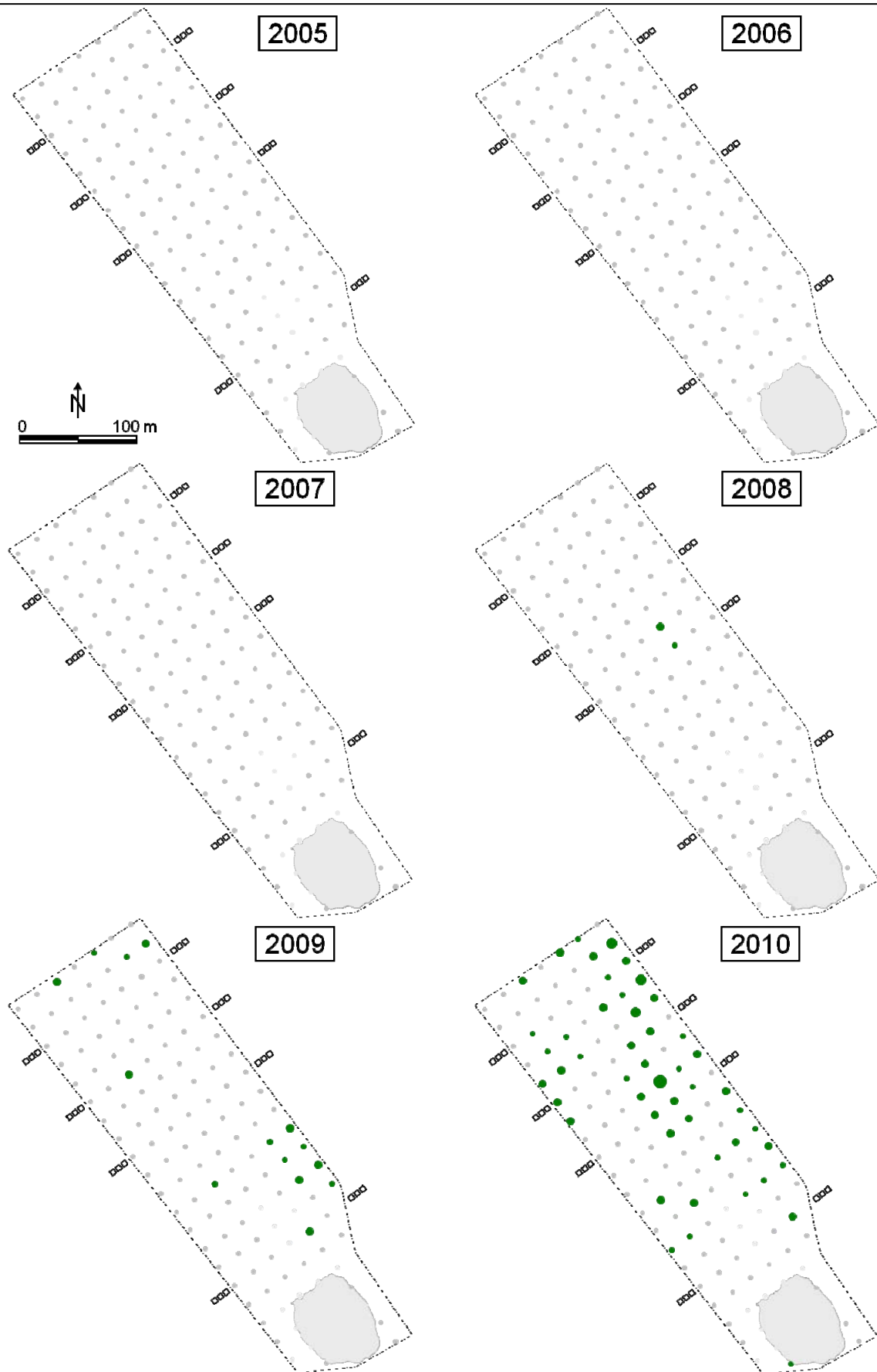
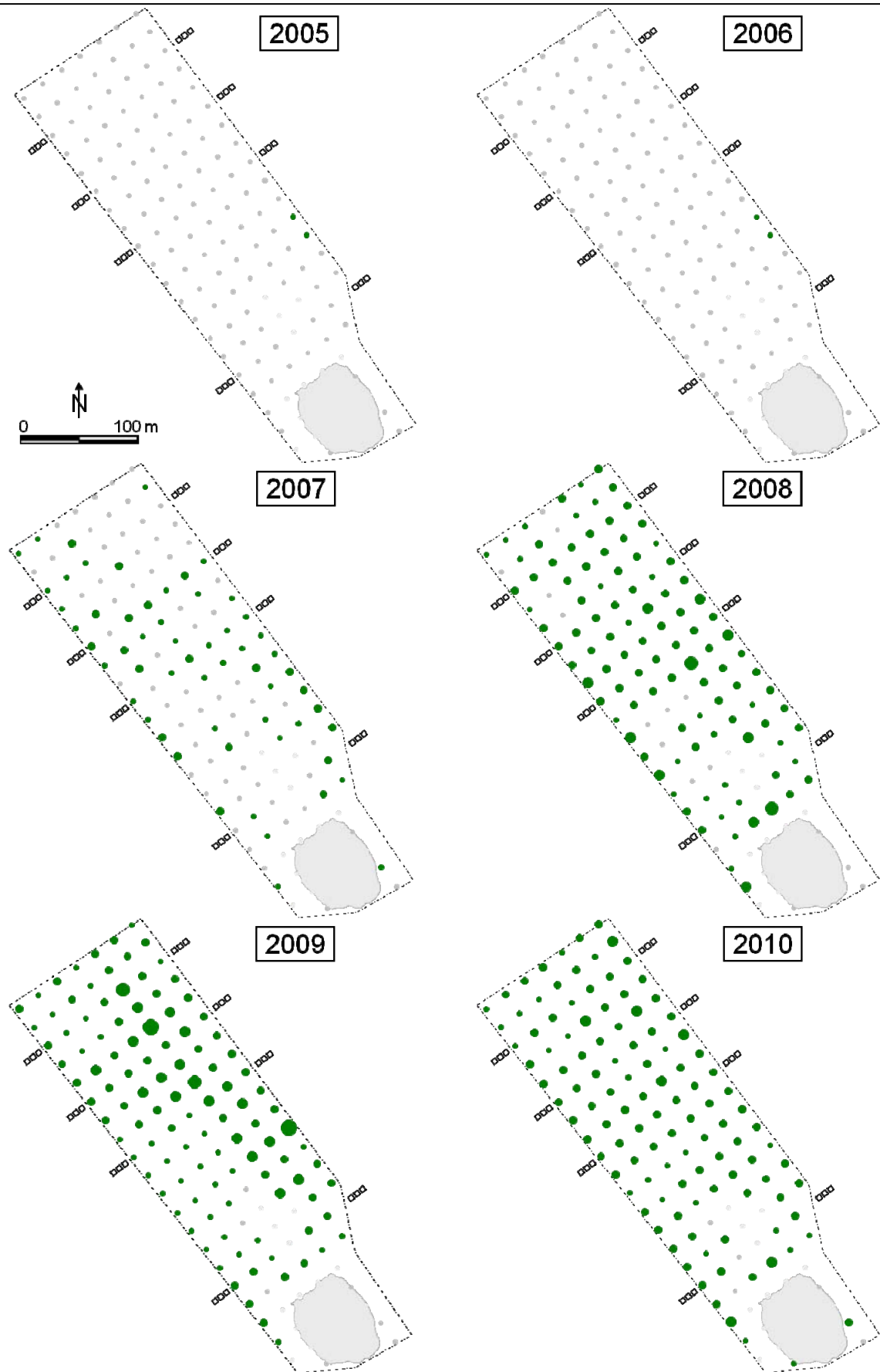Fig. 5.11: Distribution and cover of *Crepis foetida*, 2005-2010; green dots depict covers between 0.1 and 2%.

Fig. 5.12: Distribution and cover of *Centaurea stoebe*, 2005-2010; green dots depict covers between 0.1 and 4%.

Fig. 5.13: Distribution and cover of *Calamagrostis epigejos*, 2005-2010; green dots depict covers between 0.1 and 8%.

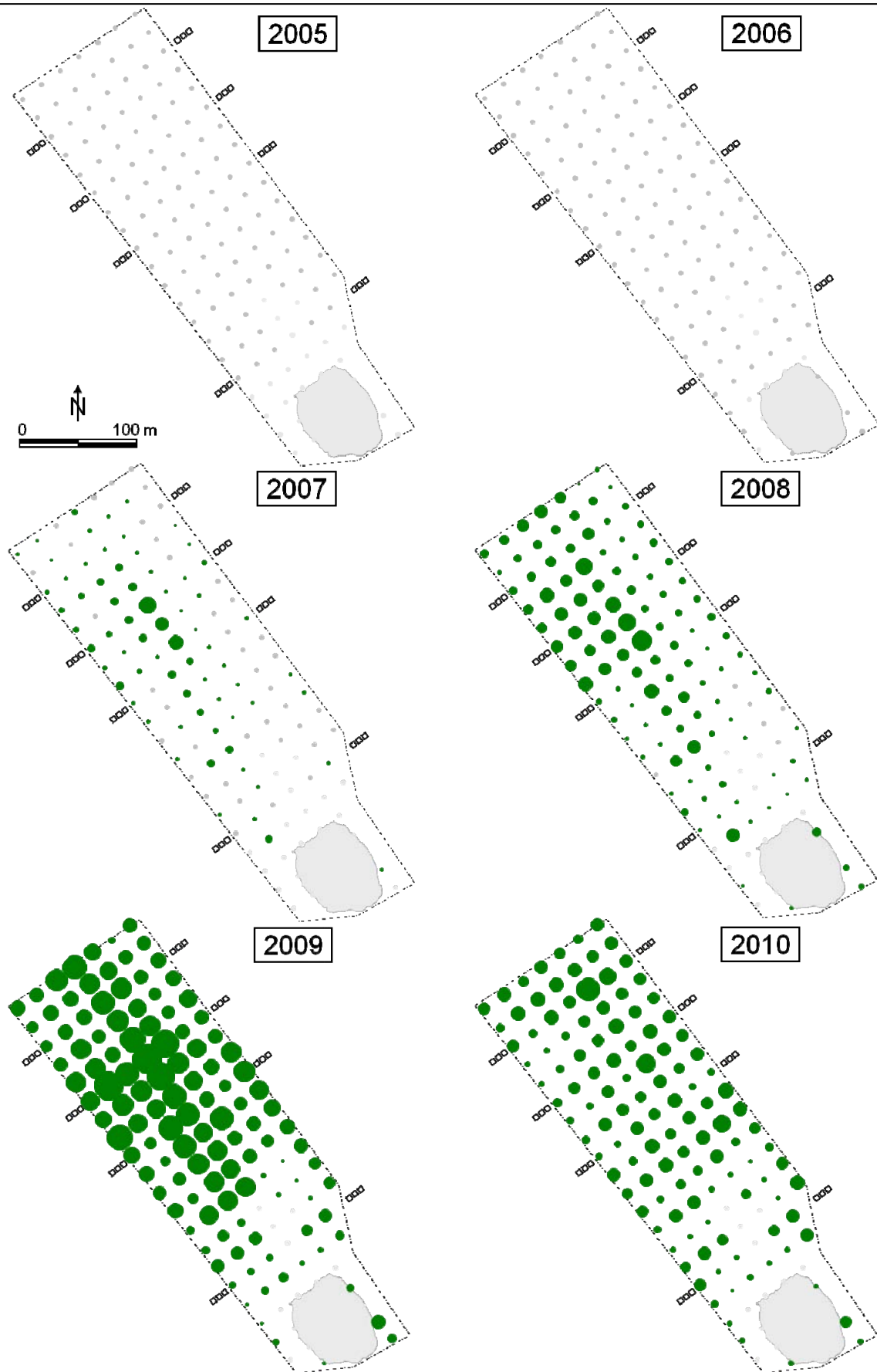Fig. 5.14: Distribution and cover of *Brachypodium sylvaticum*, 2005-2010; green dots depict covers between 0.1 and 4%.
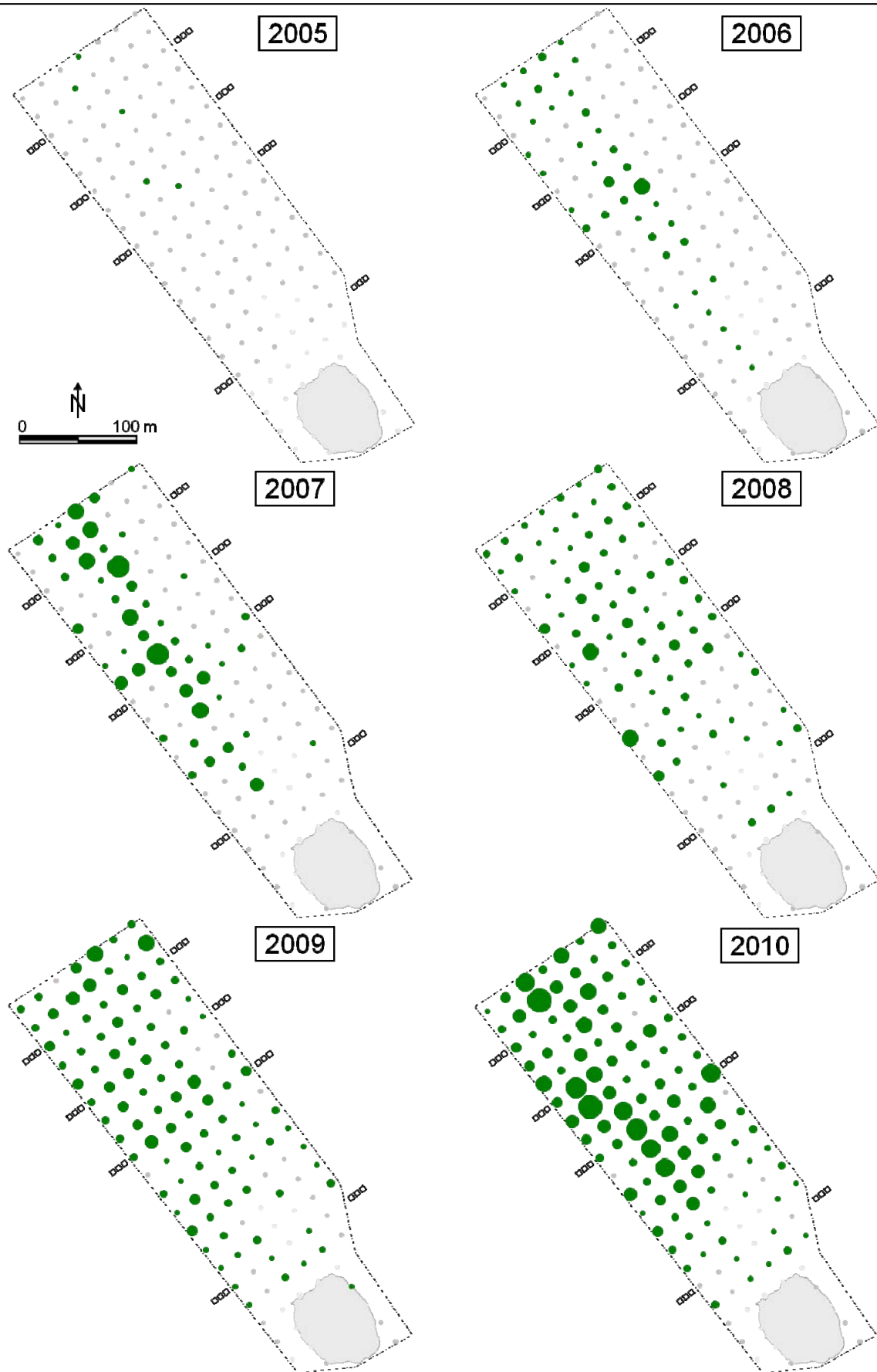
Fig. 5.15: Distribution and cover of *Genista pilosa*, 2005-2010; green dots depict covers between 0.1 and 5%.

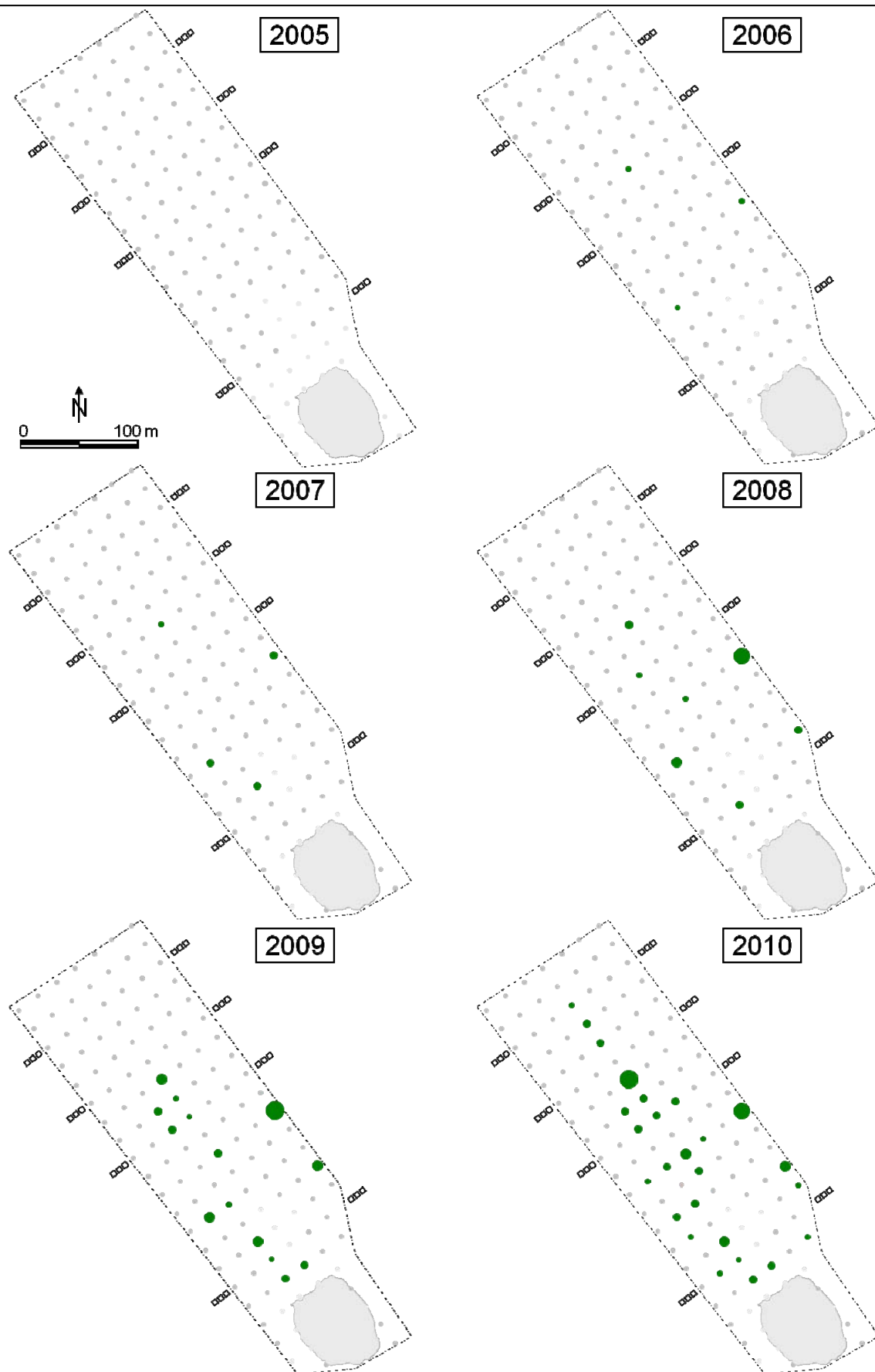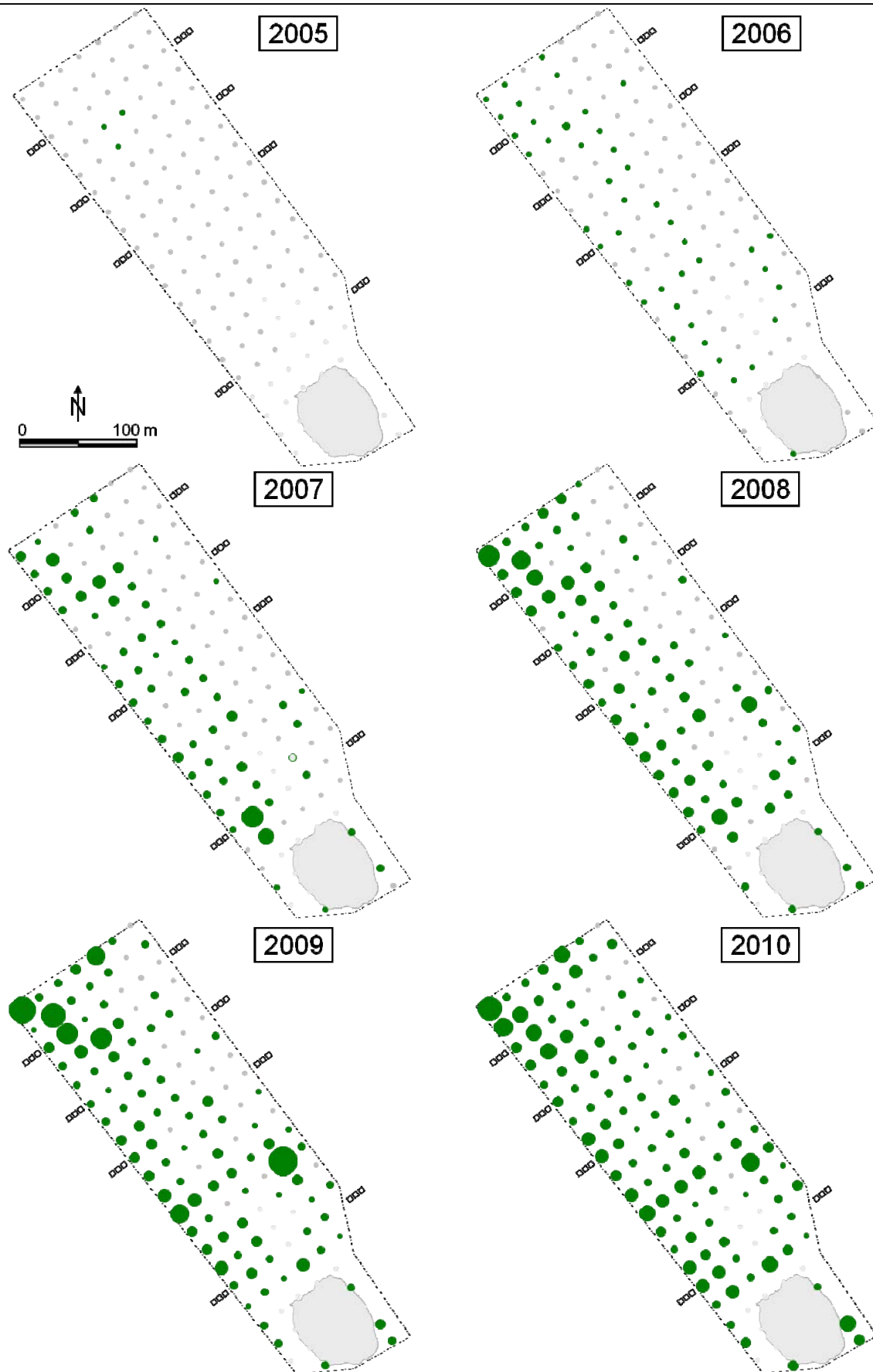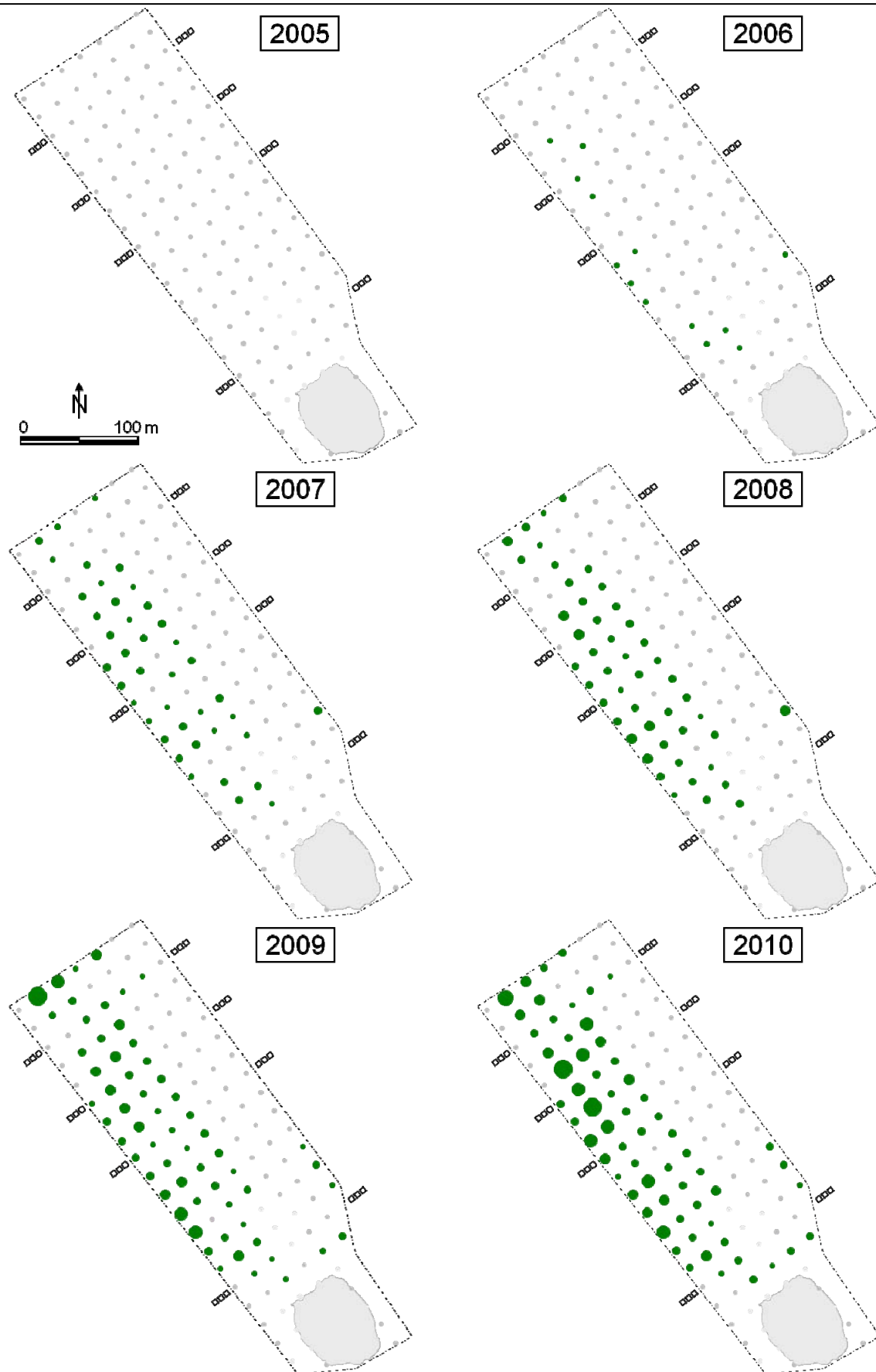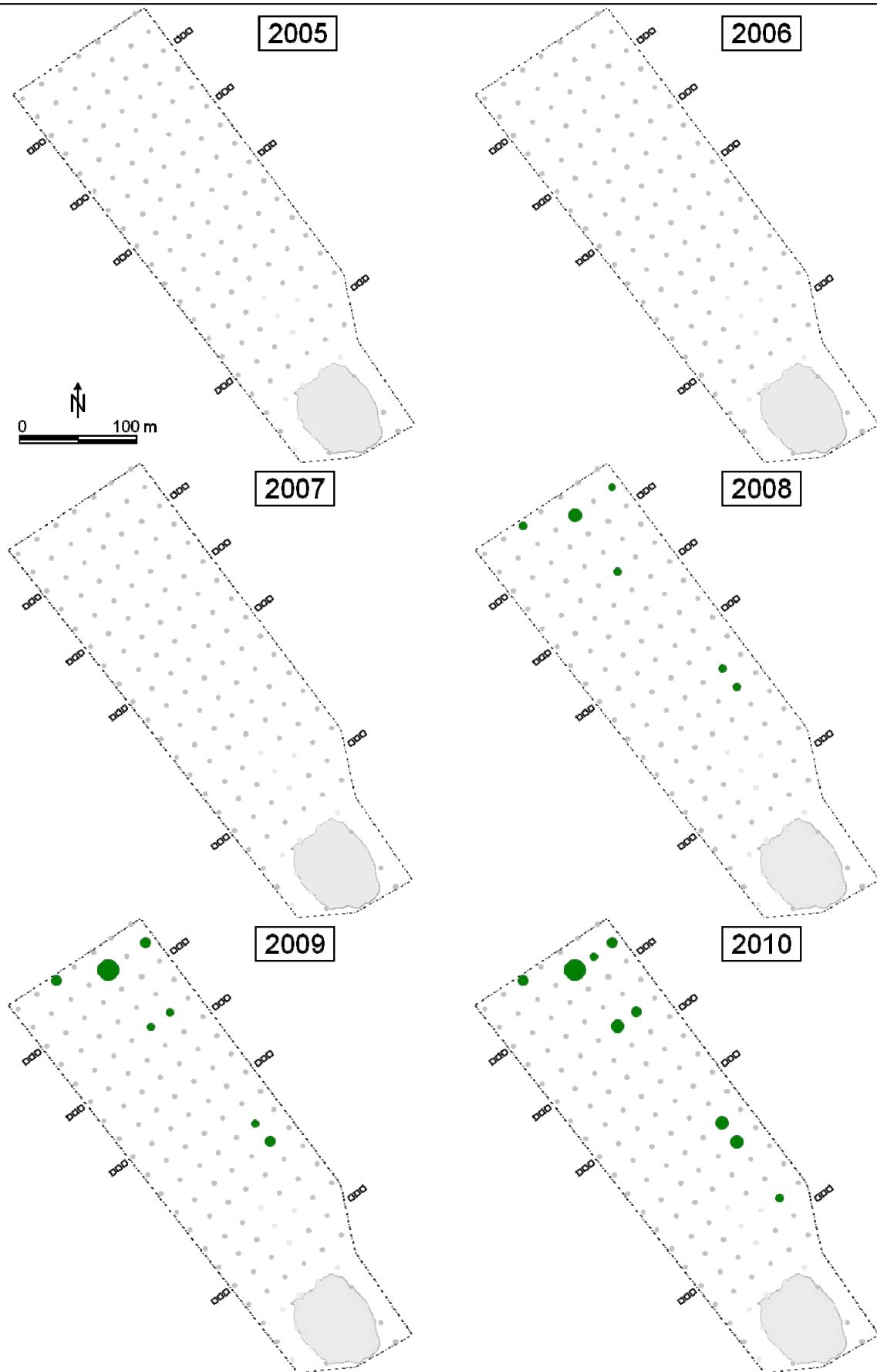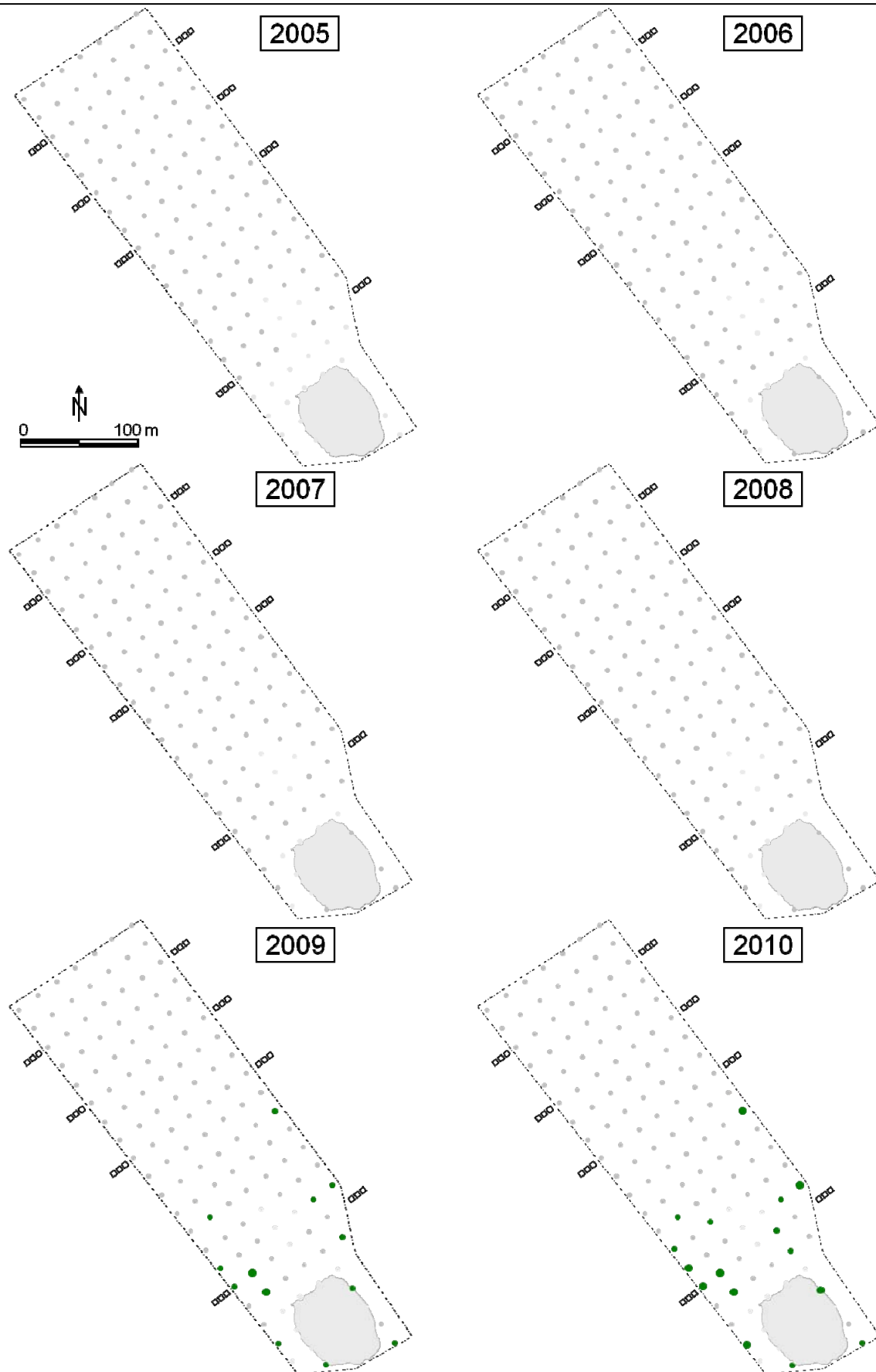Fig. 5.16:   Distribution and cover of *Betula pendula*, 2005-2010; green dots depict covers of 0.1 and 0.5%.

Vegetation patterns and site conditions

Several succession steps with regard to the geometry of patterns were identified: bare soil →
uniform, but sparse *C. canadensis* stands → dense spots consisting mainly of *T. arvense* →
increasing density of vegetation but still open places. More or less throughout the catchment a
bryophyte layer developed, particularly between 2009 and 2010. Some vascular species are
mainly restricted to the area close to the 'Chicken Creek' pond (*P. prolifera, V. hirsuta,
B. pendula*), where also an increasingly broader reed belt (*Phragmites australis*) has its origin.
However, the most important differentiation is evident between the sandier eastern and the
loamier western part of the catchment. Several species reflect this difference either during the
whole study period or during a certain part of that period: *C. canadensi*s, *T. arvense*,
E. vulgare, and *B. sylvaticum.*

In the western catchment part, the light green tussocks of *B. sylvaticum* – with sizable
distances between individual plants – established between 2006 and 2010 (Fig. 5.17). In 2010
it became evident that the population density of this species increased locally by progenies
starting from older tussocks (Fig. 5.18). The areas of dense *B. sylvaticum* patches (≤12 cm
distance between individuals) amounted to several square meters. Notably, carbonate, pH-
value, and texture did not differ significantly between the rejuvenating and the non-
rejuvenating group, neither in separate nor in multivariate analyses. Significant differences
were detected with regard to the water content of the fresh soil samples: Higher water content
($p < 0.01$, $N = 5$) was found in samples of the rejuvenating group. Thus, for the establishment
of young *B. sylvaticum* plants the soil water content seems to be of high importance, however,
the difference in water content could not be directly attributed to textural differences.

Fig. 5.17:   Tussocks of *Brachypodium sylvaticum* with a greenish-yellow colour in the
western part of the catchment (photo: Zaplata, June 2009).

An example of a very small-scaled spatial differentiation of the vegetation cover is the concentration of the tall herbaceous plant *Picris hieracioides* in the vicinity of grid point I1 in the western part of the backslope area.

Fig. 5.18: Young *Brachypodium sylvaticum* spread (dense, light-green lawn in the foreground) in close contact to an older individual (photo: Flade, August 2010).

5.3.2 Discussion

Seed rain

Seed rain characteristics changed within the first years of ecosystem development. Whereas the very first seeds were present in the substrate used for constructing the catchment, seeds from outside steadily invaded the catchment mainly from the west according to the main wind direction. The increasing *Conyza canadensis* population inside the 'Chicken Creek' catchment produced also a huge amount of seeds. In the seed shedding period of 2006 they were measured as seed rain mainly at the eastern side of the test area (Zaplata et al., 2011). In the following years the performance of *C. canadensis* within the catchment decreased (Fig. 5.3).

Also in the eastern surroundings of 'Chicken Creek' pioneer habitats decreased due to ongoing reclamation measures. As a result, the total amount of *C. canadensis* seeds in the overall seed rain decreased, and the differences between the western and the eastern part disappeared in 2008 and were reversed in 2009. An explanation for the latter might be the

advancing of the open cast mine and its reclamation areas in a westerly direction, creating large freshly dumped areas. Early plant colonizers like *C. canadensis* are dependent on such bare surfaces and therefore may follow the reclamation progress. Seed rain of other species than *C. canadensis*, mainly *Typha angustifolia* and *Calamagrostis epigejos,* increased.

Pattern formation

During the first years a differentiation in species composition occurred, mainly reflecting the two parts of the catchment area consisting of slightly different substrates. Further, a spatial differentiation between the backslope and the footslope area was obvious.

The vegetation could be classified as herb-grass stage of succession during the whole study period. We suppose that taller herbs like *Oenothera parviflora* agg. and *Picris hieracioides* have the potential to perform better in the near future. For the forest steppe zone of Western Siberia, to a certain degree comparable to the developing vegetation in 'Chicken Creek', 15 plant species had been reported from the initial stage of vegetation succession (solonchakous meadow) and 39 plant species from the second successional stage (mesohalophytic meadow) of a steppe succession series (Titlyanova, 1982). This suggests that also in 'Chicken Creek' catchment the vascular species number might further increase while progressing towards the next successional stage.

The decline of the by far dominant species *Trifolium arvense* in 2010 (Fig. 5.5) resulted in an overall decrease of the vascular species cover. This illustrates the initially low resilience of young ecosystems. In the course of further succession the number of species, which are able to compensate gaps in the vegetation layer, will increase. The significant changes of dominant plant species contributed to a differentiation into single successional phases of initial ecosystem development.

Woody species

Betula pendula, a tree species producing enormous quantities of seeds each year, was missing on the grid net plots until 2008, but occupied 11% of the 25 m² plots in 2009 (Fig. 5.16).

Robinia pseudoacacia has been present in the catchment area since September 2005 (though only one individual was present at one single plot); some other individuals, however, also existed outside the monitoring plots since 2005. The first individuals were found blossoming and fruiting for the first time in 2011, with an age of six years, which matches the present knowledge (http://neobiota.naturschutzinformationen-nrw.de). In 2008, *R. pseudoacacia* was found at five monitoring plots, while additional 83 individuals were counted outside the plots (Zaplata et al., 2010). A random distribution of 88 *R. pseudoacacia* individuals would lead to

4.7 occupied plots in an extrapolation. Therefore, the grid net seems to realistically represent the population density during the early succession. *R. pseudoacacia* occurred with a much higher population density in the eastern catchment part (Zaplata et al., 2010). Since late 2009 the population density increased rapidly due to successful reproduction by means of root suckers around some of the oldest individuals. This might confirm the hypothesis that succession reaches the woody stage earlier on nutrient-poor (sandy) than on more nutrient-rich substrate (Rebele, 1992). However, also a potential contamination of the substrate with seeds during the construction of the catchment could explain these spatial differences. The fact that this species was not detected in our initial soil seed bank analysis (Zaplata et al., 2010) might be the result of the limited quantity (volume) of substrate samples.

Patterns beyond grid-based monitoring

Certainly there is a need for permanent plot observations (Bakker et al., 1996) even so additional plotless investigations could count as considerable added value. Based on the example of *Brachypodium sylvaticum*, it is possible to show general succession principles. The primary colonization pattern of this species (only the western part of the catchment; Fig. 5.14) reflects only coarser variations in the physical environment. The subsequent local increase of patch size by progeny was merely explained by different soil water content, and is indicating increasing feedbacks between vegetation and environment (Baasch et al., 2009). Stochastic processes of colonization seem to vanish (e.g., Prach & Rehounkova, 2006). Thus, the predictability of certain vegetation patterns using environmental site factors has already increased during succession. In particular, the driving environmental site factor for the establishment of *B. sylvaticum* progenies seems to be the availability of water. This development is supposed to lead to a patchier occurrence of *B. sylvaticum* in the mid-term.

Rare species

Two rare species were found in the catchment: *Crepis foetida* (Fig. 5.19, 5.11) and Common centaury (*Centaurium erythraea*) (Fig. 5.20). *C. foetida* (besides *C. capillaris* and *C. tectorum* one of the members of the same genus present), is reported to be an extinct species in the State of Brandenburg (Ristow et al., 2006), critically endangered in Saxony (Schulz, 1999), and became extinct in the United Kingdom in 1980 (Walker, 2007). There is evidence for a secondary spread in the last decade, especially in the southern part of Brandenburg (Ristow, 2011, pers. comm.). The further development of these populations is still unclear. *C. erythraea*, the only member of the gentian family in the 'Chicken Creek' catchment, has a regional red list status (threatened; Ristow et al., 2006). In 2011, a remarkable population growth occurred with several specimens now settling on one plot (S5).

Fig. 5.19:   *Crepis foetida* exhibiting its white seed-heads, surrounded by *Trifolium arvense* (photo: Zaplata, July 2009).

Fig. 5.20:   Rosettes of *Centaurium erythraea* in the centre and at the bottom right; other rosettes are of *Prunella vulgaris* (photo: Zaplata, August 2011).

Ongoing monitoring offers the chance to study how these rare or threatened species may establish in new areas and perform over time. Real time series are an inestimable advantage as it might indicate best, which succession state and which community structures have which consequences for the population of a given rare species.

5.4   Conclusions

Both vegetation cover and vascular species numbers are increasing in the catchment, and vegetation patterns are developing. Exceptions to this trend like the 'setback case' of declined vascular species cover in 2010 might be 'unexpected', but can provide valuable hints to superior characteristics and principles of initial ecosystems.

The studied ecosystem is still 'young' and fluctuating. In future, setbacks and unpredictable incidents will steadily become less important, while resilience and predictability are expected to increase. With a continued vegetation recording on permanent plots we have the chance to document, analyse and quantify such changes.

The large set of data recorded from 2005 to 2010 meets the high today's statistical demands. It builds the basis for detailed analyses of the vegetation, e.g., how the community structure develops, whether succession proceeds in steps or continuously, and how vegetation interacts with both biotic and abiotic site conditions (cf. chapters 2, 3, 4, and 6). Analyses even might contribute to the scientific problem, whether it is inherent to certain species to be rare.

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6.   First five years of soil food-web development in 'Chicken Creek' catchment

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6.1   Introduction

Soil harbours the world's largest and most diverse faunal communities. The corresponding soil food webs are multi-connected networks with single soil food chains consisting of up to seven trophic levels, e.g., dead organic material – bacteria – flagellates – amoebae – predatory nematodes – predatory mites – predatory beetles (Hunt et al., 1987; Coleman et al., 2004; Moore et al., 2004). There are even more soil food chain levels when trophic interactions (loops) within taxonomic groups are considered (Scheu & Falca, 2000; Bardgett & Wardle, 2010). Through their feeding habits, the various components of soil food webs directly and indirectly affect microbial composition and accelerate bacterial activity, rates of litter decomposition, nutrient flows and plant growth (Lousier & Bamforth, 1988; Zwart & Brussaard, 1991; Wanner & Xylander, 2005). These complex living soil networks thus play key roles in decomposition processes and nutrient cycles. Still many unanswered questions remain, e. g. what purpose does this high diversity serve and from where does it originate? How do these highly complex food webs develop? Is there a regular order of developmental events or is it all random (first come first go) and chaotic?

Soil organisms interact intensely with each other and with their abiotic environment. Their impact on ecosystem processes is especially important in the early initial phase of primary succession (Bardgett & Wardle, 2010): first colonisers appear immediately after exposure of new land surfaces, facilitating the establishment of other biota and plant succession (Hodkinson et al., 2004; Wanner et al., 2008). Mining sites present highly applicable sceneries for studies of the primary succession of soil food webs (Dunger & Wanner, 2001; Frouz et al., 2007; Háněl, 2001; Topp et al., 2001). In opencast mining, dump substrates (= spoils), which in the beginning are mostly organism free, are taken from depths of several metres, where they had been buried for tens of thousands of years, and are brought to the surface where it takes decades for the spoil to develop into soils. To accelerate this process, post-mining sites are usually ameliorated and reclaimed. The present study of the artificial

catchment 'Chicken Creek' is the first investigation where no such measures and as little interference as possible occurred. After initial substrate deposition and site initiation in September 2005, the 'Chicken Creek' was left solely to natural succession (Gerwin et al., 2009, 2010).

We investigated the first five years of undisturbed primary succession, including major parts of the soil food webs commonly found in early succession into the study: testate amoebae, nematodes, tardigrades, collembolans, actinedid mites, gamasine mites, oribatid mites, and even some members of the above-ground fauna with special attention to carabid beetles. The results of these investigations will be discussed against the background of the plant community; or, in the words of Moore et al. (2004) "Progress towards answering fundamental ecological questions of the distribution and abundance of species can only be made by merging the (...) world of primary producers and the brown world (...) in a new integrative ecology."

6.2   Materials and methods

6.2.1   Sampling

The study was carried out at the artificial catchment 'Chicken Creek'. Sampling of soil fauna began in October 2005, only weeks after 'point zero' of ecosystem development (= initiation of the site) (Gerwin et al., 2009, 2010), and from then on took place at usual times of peak abundances of the soil fauna, namely in April/May and October of the following five years. At every sampling date, 27 soil cores each for microfauna (testate amoebae, Nematoda, Tardigrada) and mesofauna (Collembola, Acari) were taken, with 9 cores each in the upper, middle and lower region (cf. Elmer et al., 2010). Soil cores for microfauna (3.5 cm in diameter) and mesofauna (6.4 cm in diameter) were always taken to a depth of 5 cm and directly adjacent to one another to allow direct correlations between the different animal groups.

In May 2008 and August 2009, additional paired samples of bare ('uncovered') and vegetated ("covered") substrate were taken with a spatula in order to examine testate amoebae in dependence of vegetation cover (five samples per micro site, pooled, each sample with a surface of 0.5 cm² and a depth of 0.3 cm). The distance between covered and uncovered patches was only about 2 cm.

Adhesive seed traps (petri dishes with 15 cm diameter, mounted at 30 cm height, distance to next trap 20 m; cf. Zaplata et al., 2011) were sampled to detect wind-transported testate amoebae. On each petri dish, three cover slips (1.8 x 1.8 cm) were randomly applied (tap water as medium between). Additionally, pitfall traps were installed in summer 2008 in the upper, middle, lower, as well as in the semi-aquatic area around the pond.

6.2.2 Microfauna (testate amoebae, nematodes, tardigrades)

Testate amoebae were counted and determined at species level directly from soil suspensions using an inverted microscope (4% formaldehyde solution of 30-500 mg soil per sample). Aniline blue was added to differentiate between full (living) and empty (dead) shells (cf. Wanner, 1999). Seed trap cover slips were checked for testate amoebae under a light microscope at 200x magnification.

Nematodes and tardigrades were extracted for a maximum of five days (initiated on the sampling day) in modified Baermann funnels, from which they were removed daily, killed by heat, preserved in 4% formaldehyde and counted (cf. Hohberg, 2003, 2006; Elmer et al., 2010). Per sample, one hundred individuals of nematodes and of tardigrades were identified to species or genera level under an inverted microscope. Additionally, individual body lengths l [µm] and widths d [µm] were measured for each determined specimen. From these, fresh weights m [µg] of individual nematode (i) and tardigrade (ii) were calculated using the formulas:

i) $m = l * [d^2/(1.6 * 10^6)]$ (Andrássy, 1956),

ii) $m = l * (d/2)^2 * \pi * 1.04 * 10^{-6}$ (Hallas & Yeates, 1972).

Nematodes sharing the same feeding type (Yeates et al., 1993) and cp-value (Bongers, 1990) were assigned to "functional guilds" (Bongers & Bongers, 1998). The "weighted faunal analysis" concept was applied (Ferris et al., 2001), where functional guilds of nematodes are indicators of "basal" (b), "structured" (s) and "enriched" (e) conditions of the soil food web (cf. Hohberg, 2003; Elmer et al., 2010).

6.2.3 Mesofauna (Actinedida, Collembola, Gamasina, Oribatida)

Endogeic mesofauna were driven actively from the soil samples by means of an increasing temperature and desiccation gradient (modified Macfadyen extractor), killed in Törne fixative, preserved in 70% ethanol and counted (cf. Elmer et al., 2010). All individuals of the Actinedida, Gamasina, Oribatida and Collembola were determined to species or generic level under a differential-interference microscope at up to 1000× magnification. Species of Actinedida were assigned to the trophic groups "microbivorous", "omnivorous" and "carnivorous" according to Suski (1973), Zacharda (1978), Walter (1988), Greenslade & Clift (2004), Krantz & Walter (2009). Gamasina species were considered predatory as they predominantly feed on nematodes, eggs and larvae of insects (Walter & Proctor, 1999), and Collembola were all considered microbivorous, feeding predominantly on fungi and bacteria (Rusek, 1998; Castano-Meneses et al., 2004).

Finally, collembolan and actinedid species were assigned to different adaptation classes according to their occurrence in newly formed soils and hot and dry habitats, respectively (McDaniel & Bolen, 1981; Santos & Whitford, 1983; Dunger et al., 2004; Jesionowska, 2003, 2008), as these habitat parameters were characteristic for the initial study sites: initial soil inhabiting species: Actinedida: *Nanorchestes* sp., *Speleorchestes* sp., *Claveupodes delicates*, *Tydaeolus* sp., *Parallelorhagidia evansi*, *Coccotydaeolus* sp., Collembola: *Proisotoma minuta*, *Mesaphorura macrochaeta*; xero- / thermo- / psammophilous species: Actinedida: *Xerophiles ereynetoidalis*, *Cheletominus vescus*, *Hawaiieupodes thermophilus*, *Neonanorchestes ammolitoreus*, Collembola: *Folsomides parvulus*, *Pseudosinella octopunctata*, *Brachystomella parvula*.

6.2.4 Above-ground invertebrates

Pitfall traps were installed in May 2008 within the larger botanic permanent monitoring areas (cf. Elmer et al., 2010). Each trap (diameter 12 cm) was filled with a saturated benzoic acid solution and was emptied in two-weekly intervals for 13 months in order to collect the macroinvertebrates. In the laboratory, the specimens were transferred to a 70% ethanol solution and preserved until determination. Up to now, the carabid beetles (Coleoptera) of 19 periods (until May 2009, 54 weeks) have been determined to the genus level, the specimens of the sub-family Cicindelinae to the species level.

6.2.5 Abiotic soil parameters

Soil moisture was determined gravimetrically (as water per g dry soil) from the microfauna samples (cf. Elmer et al., 2010). Vegetation and moss coverage was assessed for all single samples during sampling in October 2007, with the classes 0% (no visible soil crusts, moss or vegetation), 10% (single pitches of soil crusts or mosses), 20% (scarce pitches of moss, soil crusts, single herbs) and 50% (approximately half of the substrate surface was covered by mosses, herbs or soil crusts).

6.2.6 Statistics

To test for significant temporal developments in the communities (densities species numbers), the data was submitted to a non-parametric Kruskal Wallis test with sampling date as the main factor. A post-hoc multiple comparison procedure tested for significant differences between sampling dates. Statistical analyses were performed using the Statistica 8.0 software package (StatSoft, Inc., Tulsa, USA).

6.3 Results and discussion

6.3.1 Testate amoebae

Up to now, 74 soil samples have been analysed from October 2005 to October 2008. A (more or less) continuous development of testate amoebal community was visible with respect to densities, biomasses and species inventory (Tab. 6.1). To reveal possible structural influences on the colonisation process, the May 2008 investigations (Tab. 6.1) distinguished between covered and uncovered microsites, each of these pairs situated close together.

Tab. 6.1:   Testate amoebae colonisation from 2005 to 2009; S = samples taken with a soil corer (d = 34 mm, 0-5 cm) from predominantly uncovered substrate; C, U = spatula samples from covered ("C") or uncovered ("U") microsites; N = number of samples analysed, (N') = soil samples containing testate amoebae shells (full and empty shells); Density: testate amoebae in individual per g dry mass substrate (full shells); Biomass: testate amoebae in µg wet mass per g dry mass substrate; standard deviations are given in parentheses; # = no amoebae or only single specimens; n.d. = not determined; moisture = % of substrate dry mass (air dried).

Date	Sample	N (N')	Taxa	Density	Biomass	pH$_{(KCl)}$	Moisture [%]
Oct 2005	S	6 (3)	3	#	#	7.7	9.6
May 2006	S	2 (1)	1	#	#	7.5	1.8
April 2007	S	12 (1)	1	#	#	7.1	1.8
April 2008	S	3 (3)	6	#	#	n.d.	11.2
May 2008	C	12 (12)	12	616 (892)	8.8 (12.7)	7.3	4.1
May 2008	U	12 (11)	7	64 (112)	0.8 (1.2)	7.6	2.9
Oct 2008	S	27 (12)	8	40 (84)	0.7 (1.8)	n.d.	5.1
Aug 2009	C	2 (2)	14	6801 (7985)	72.0 (29.0)	6.9	4.3

Almost all covered microsites revealed ten-fold higher densities of testate amoebae compared to the uncovered patches (Wanner & Elmer, 2009) (Fig. 6.1). Additionally, species inventory of the covered sites was always more diverse, including all species from the uncovered sites. For these effects, different explanations are possible. Plants may act as a carbon source for testate amoebae, since testate amoebae feed on bacteria occurring more frequently in plant patches. Thus, uncovered patches will be carbon limited in respect to amoebae. However, plants may also act as structural elements sheltering amoebae from being blown away by wind, and improving abiotic growth conditions for amoebae compared to bare soil. At vegetation patches wind-transported, nutrient-enriched fine materials are filtered and

accumulate. Further, the soil water regime under vegetated patches is more suitable for amoebal growth, because infiltration and water capacity are higher and temperatures are lower – effects which are well-known from desert ecosystems (Schlesinger et al., 1996). Following these explanations, uncovered patches are water and nutrient limited for amoebal growth.
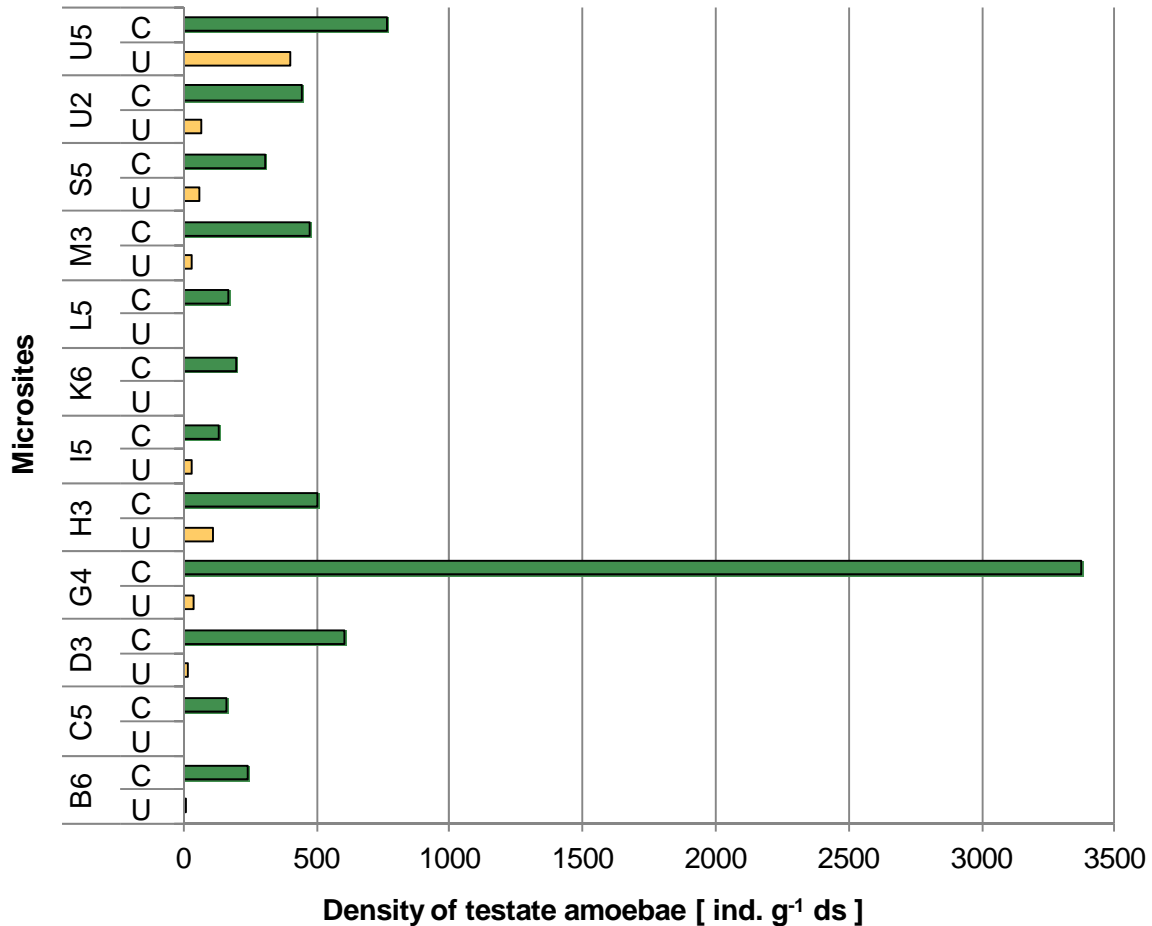
Fig. 6.1: Densities of living testate amoebae (full shells) at the 'Chicken Creek' on May 16th, 2008; patches covered by vegetation (C) versus vegetation-free patches (U) at different grid points B6 to U5.

Adhesive seed traps provided information on aerial "immigration" of testate amoebae. Up to now, 36 traps have been analysed (13 from August 17th, 2008 and 23 from September 3rd, 2009). At a first glance, immigration by wind appears to be of little importance, since only few testate amoeba shells were detected. However, in 12 out of 36 traps, at least one testate amoeba had been found, mainly (as reflected by catchment species inventory) xenosome-bearing shells such as *C. aerophila*, *C. sphagnicola*, *C. ambigua*, and *P. acropodia*. Over months and years, this small number of shells, however, may grow to a considerable number of testate amoebae blown by wind into the catchment area.

As shown by earlier studies dealing with wind-driven amoebal immigration and colonisation of soil microfauna (e.g., Wanner et al., 1998; Wanner & Dunger, 1999), wind dispersal may play a pivotal role in early colonisation processes. Still, the direction of migration (immigration, emigration) is not absolutely clear from seed trap studies, as was demonstrated by Zaplata et al. (2011), who found numbers of pappus fragments of *Conyza canadensis* in the seed traps, which were clearly originating from those plants already growing inside the catchment. Further analyses, which require a complete set of traps, will differentiate between the eastern (possible catchment output) and western set (possible input to the catchment) of seed traps (as shown in Zaplata et al., 2011).

6.3.2 Nematodes and Tardigrades

During the first two years of succession, nematodes and tardigrades occurred only sporadically and in rather small numbers (Fig. 6.2A, B). As of October 2007 densities steadily increased, and nematode numbers began reaching counts reported from mature soils ($1.6 \cdot 10^6$ individuals per m^2).

Also numbers of nematode species increased during succession (Fig. 6.2C), but were still low in October 2010, with 12 species per sample and 44 species in the total site. In the first six years of site succession, only 118 species and 48 genera were identified at 'Chicken Creek', indicating that the substrate is poor in terms of nutrient availability and microclimatic conditions (Hohberg, 2003, 2007). Mature soils of comparable latitudinal range (45° to 55°N) hold 40 to 60 nematode species per sample (Boag & Yeates, 1998).

The basal status of the 'Chicken Creek' substrate is further confirmed by the faunal profile, based on the indicator importance of functional guilds of nematodes ("weighted faunal analysis"). This suggests that the flow of resources into the food web system as well as the prevalence and abundance of higher trophic level organisms was dramatically low in 2005 and only slowly increasing during the initial ecosystem development (Fig. 6.3).

Nematode biomass mainly consisted of bacterial feeders in the beginning and, from October 2007 on, fungal feeders, omnivores and plant-root feeders gained importance, reflecting the presence of changing food sources during the succession in 'Chicken Creek' (Fig. 6.4A).

While after six years of succession the nematode fauna was still far from the species numbers and functional diversity of mature soils, the tardigrade fauna seems to have already passed its peak (Fig. 6.2B): the highest densities and biomass were measured between April 2008 and May 2009 (on average 281,900 to 349,700 individuals per m^2), whereby almost all individuals belonged to algal-feeding *Apodibius confusus* (Fig. 6.4B). From this point onwards, tardigrade numbers, and above all *Apodibius confusus*, significantly decreased (Fig. 6.2B).

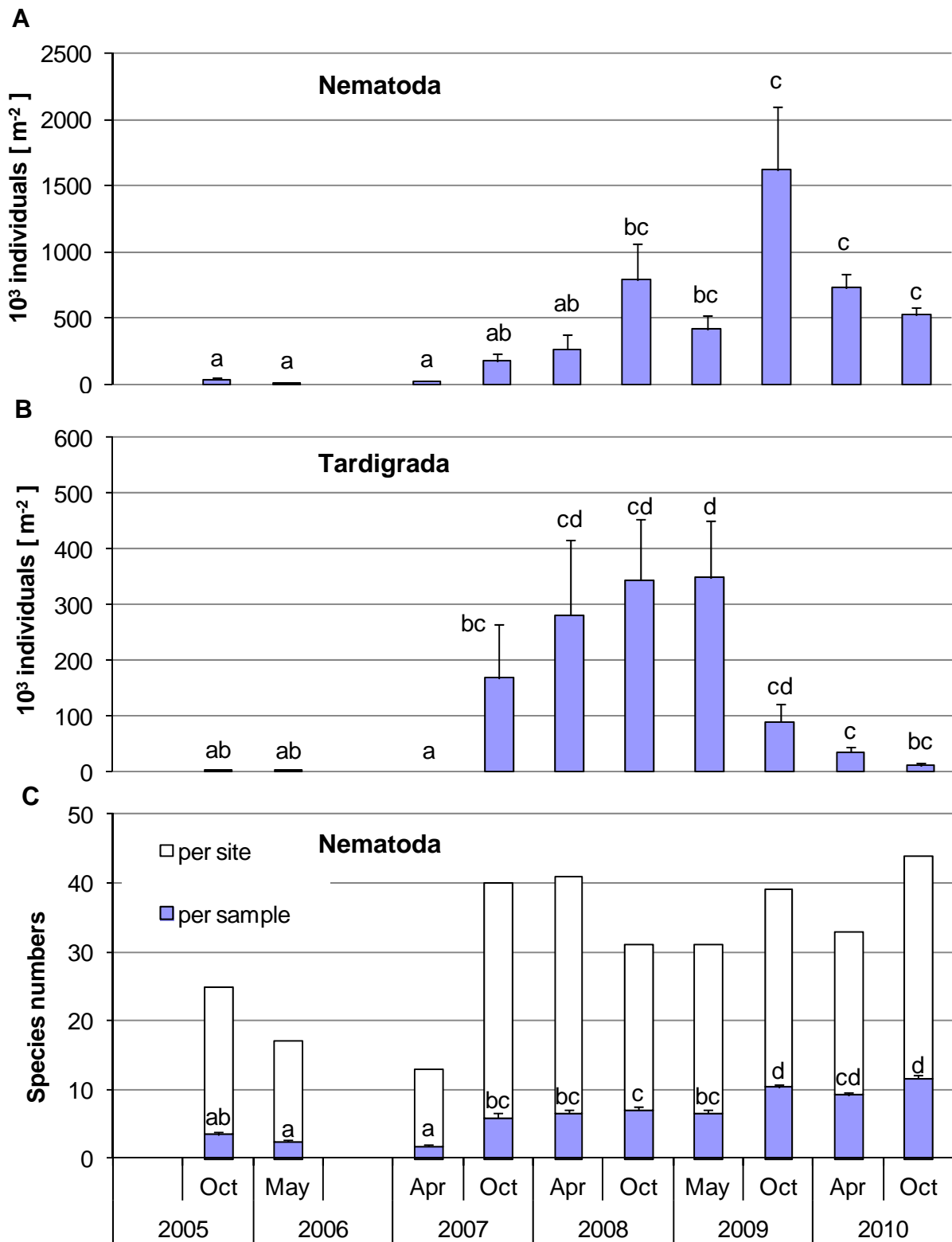Fig. 6.2: Faunal succession at the 'Chicken Creek' in the first six years of site development: Overall densities (mean ± 1 SE) of **A:** nematodes, **B:** tardigrades, **C:** Nematode species numbers in the total site (27 samples) and per sample (mean ± 1 SE); columns without a letter in common differ significantly from each other ($p < 0.05$, Kruskal Wallis multiple comparisons); note the different scales of the y-axes.
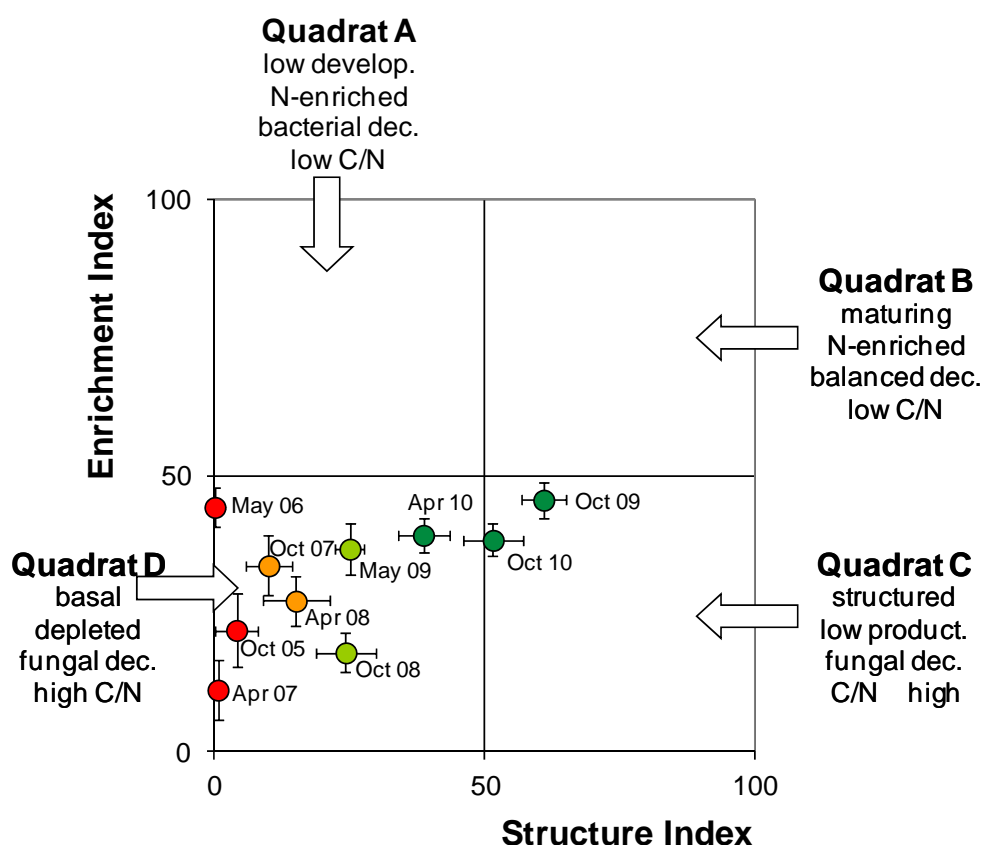
Fig. 6.3: Faunal profiles representing the food-web condition in relation to its structure (SI) and enrichment (EI) as indicated by the "weighted faunal analysis" (mean ± 1 SE, *n = 27*); Quadrat A: poorly developed or highly disturbed food web condition, N-enriched, bacterial decomposition channel, low C/N ratio, Quadrat B: maturing food web condition, disturbance low to moderate, N-enriched, balanced decomposition channel, low C/N ratio, Quadrat C: undisturbed, structured food web and relatively low primary production, fungal decomposition channel, moderate to high C/N ratio, Quadrat D: basal or degraded food web condition, depleted, fungal decomposition channel, high C/N ratio.

Tardigrades show a special affinity to young soils as was also reported from a post-mining site near Görlitz (Hohberg, 2006) and from young successional stages of German inland dunes (Russell et al., 1994). We predict that in the near future, tardigrades will only be found sporadically and in small numbers in the 'Chicken Creek' site: first, competitors (e.g., nematodes) and predators (e.g., predatory mites) will increase in number, limiting the previously rather unrestricted population growth of tardigrades. Second, the coarse dump substrate is slowly but steadily becoming combined (glued) to aggregates by organic debris and root exudates, thus decreasing the fraction of pores large enough for tardigrades to enter (Hohberg et al., 2011). And lastly, green algae, the food resource of the dominant tardigrade

of the present investigation, will decrease in the course of an increasing plant cover shading the previously bare substrates. In 2010, however, green algae were still an important food resource in 'Chicken Creek' substrate, but consumers switched from algal-feeding tardigrades to omnivorous nematodes (*Aporcelaimellus obtusicaudatus*, *Ecumenicus monohystera*) that were mainly feeding on green algae, as was easily recognizable from their green intestines (Fig. 6.4).
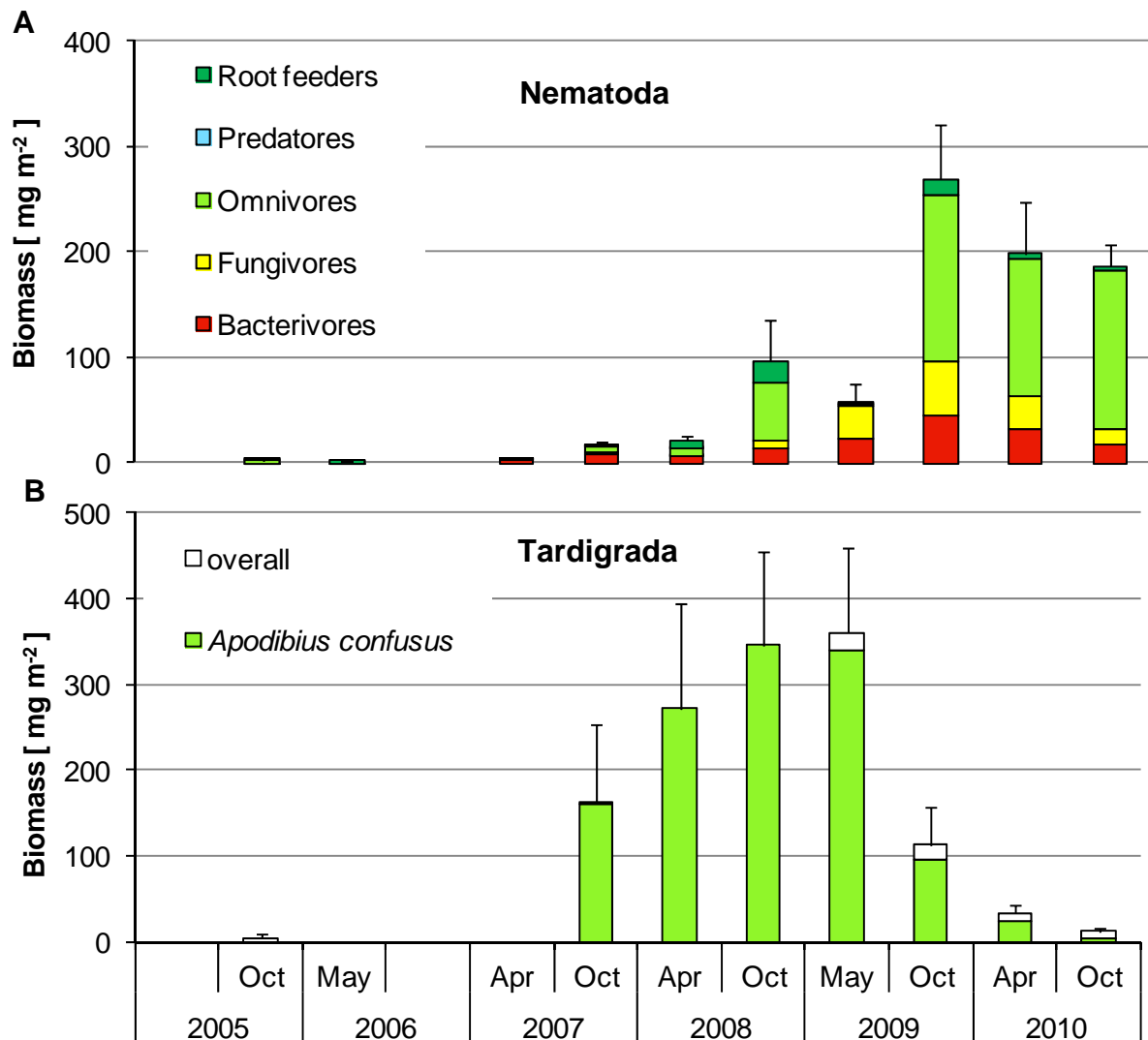
Fig. 6.4: Biomass of microfauna and its trophic compounds during the first six years of primary succession at the 'Chicken Creek'; **A:** Overall nematode biomass (mean ± 1 SE, $n = 27$) and biomass of the main nematode feeding types, **B:** Overall tardigrade biomass (mean ± 1 SE, $n = 27$) and the biomass of the most abundant species, the algal-feeding *Apodibius confusus*; note the different scales of the y-axes.

6.3.3 Soil microarthropods

Collembola and actinedid mites were present within months after exposition of the substrates, albeit in very low densities and only in sporadic samples (Fig. 6.5).
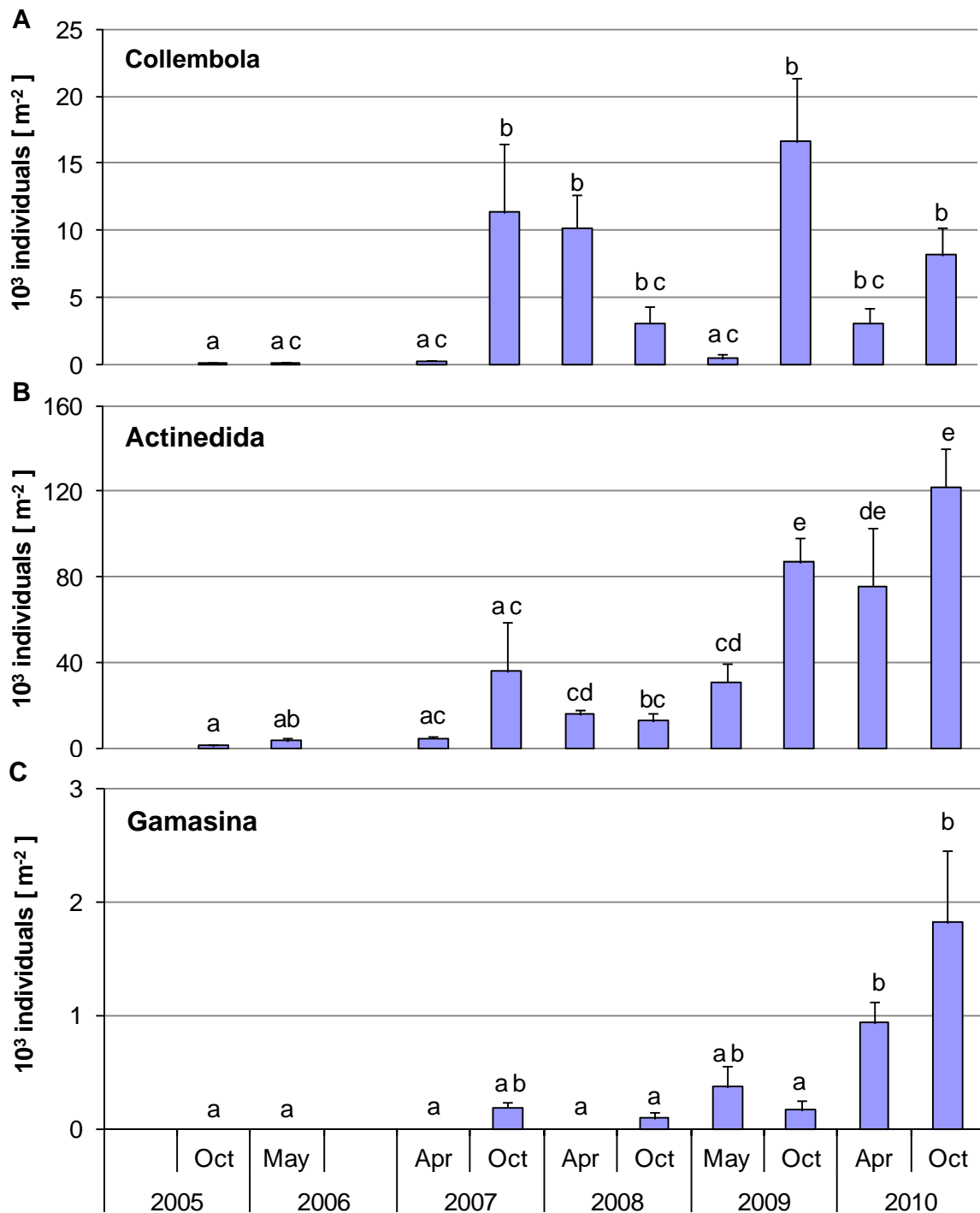
Fig. 6.5: Microarthropod succession: Overall densities (mean ± 1 SE) of **A:** Collembola, **B:** Actinedida **C:** Gamasina; columns without a letter in common differ significantly from each other (p < 0.05, Kruskal Wallis multiple comparisons); note the different scales of the y-axes.

During the entire study period, the microarthropod communities were strongly dominated by actinedid mites. Gamasine and oribatid mites (e.g., the euryoecious *Tectocepheus velatus sarekensis*) were found only sporadically and in single specimens in the substrates, where they had immigrated passively by the wind (Wanner & Dunger, 2002; Lehmitz et al., 2011) or had been phoretically transported by insects or birds (Athias-Binche, 1991; Walter & Proctor, 1999).

From April 2010 on, the first true colonisers within Gamasina (*Hypoaspis nolli*, *Asca bicornis*) developed populations in the 'Chicken Creek' mine spoils: males, females as well as nymphal and larval stages were found, indicating that in 2010 the environmental conditions and nutritional status of the mine spoils was finally favourable enough for these two gamasine species to reproduce.

Compared to Collembola and Actinedida, the slightly retarded colonisation by Gamasina and strongly retarded colonisation by Oribatida – even characteristic pioneer species, such as *Opiella nova* (Wanner & Dunger, 2002) were missing in the first five years of catchment development – concurs with the findings of Christian (1993, 1995) and Dunger & Wanner (2001) in young successional stages of a post-mining site near Berzdorf.

Initially, microarthropod species richness was extremely low, with only very few species present (Fig. 6.6). Densities and species richness increased steadily from year to year. The increases in density as well as species richness were statistically significant in Collembola and Actinedida as of late 2007, three years after site initiation, although they were still low compared to mature soils. The increase in densities of all groups was mainly due to expanding populations of only a few species. These few "regular" species also steadily increased their distribution throughout the site. Where they were initially found only in low densities in just a few samples, they then were registered in more samples in increasing densities and finally as of 2008/2009 in all samples (Fig. 6.7).

The increase in species richness, on the other hand, was caused by many "rare" species found in only sporadic samples and in very low individual numbers. These successional developments could be partly directly correlated with, i.e., increases in vegetation cover (Fig. 6.8), indicating a direct relation to ecosystem and soil maturation (Russell et al., 2010).

Those species, which were first present in the sites and were then most dominant in the communities (Fig. 6.9A) are all known from initial or nutrient-poor soils and can be considered primary colonisers (Santos & Whitford, 1983; Kinnear, 1991; Dunger et al., 2004). Three years after site initiation these species still strongly dominated the communities. However, already one year after site initiation, specialized species adapted to xero-thermophilous habitats were also present and slowly increased their populations and distribution throughout the study period.
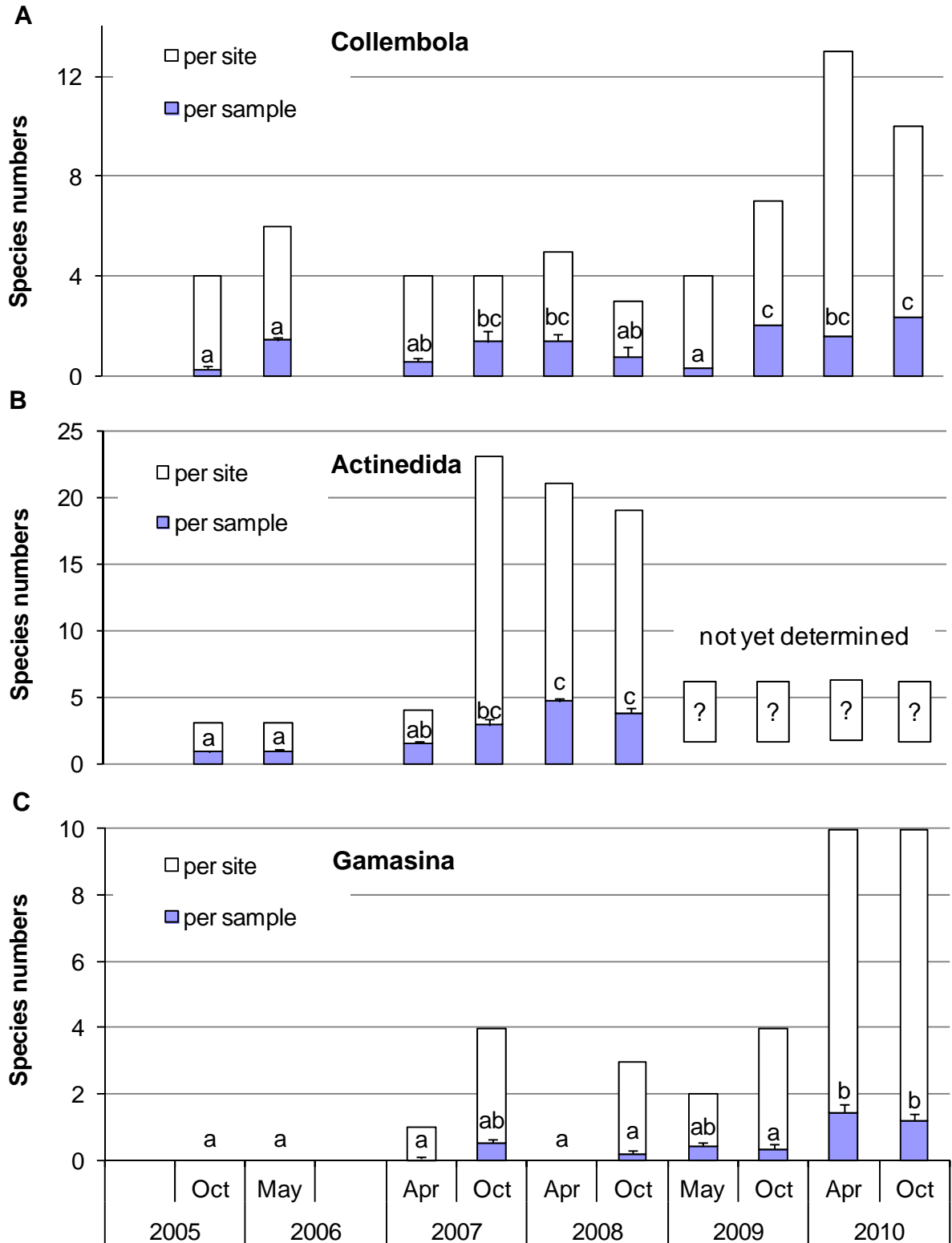
Fig. 6.6: Microarthropod species richness: **A:** Collembola **B:** Actinedida **C:** Gamasina, species numbers per site (27 samples) and per sample (mean ± 1 SE); columns without a letter in common differ significantly from each other (p < 0.05, Kruskal Wallis multiple comparisons); note the different scales of the y-axes.

Nanorchestes sp. (Acari, Actinedida)

Fig. 6.7: Increasing distribution and density of primary colonisers during initial 'Chicken Creek' site succession, for example the actinedid mite *Nanorchestes* sp., which was the most common microarthropod species in the present study; densities are given in density classes (individuals per 32 cm² soil sample).

Fig. 6.8: Densities of two actinedid mite species A: *Siteroptes* sp. and B: *Xerophilus ereynetoidalis* in dependence on moss/vegetation cover assessed for all single samples during sampling in October 2007; note the different scales of the y-axes.

Fig. 6.9: Development (mean densities per date) of microarthropod species groups based on **A**: ecological adaptation and **B**: feeding type.

Remarkable was the occurrence of species such as *Cheletomimus vescus*, *Hawaiieupodes thermophilus* and *Xerophiles ereynetoides*, which are very rare species adapted to nutrient-poor sandy soils (McDaniel & Bolen, 1981; Jesionowska, 2003, 2008). Increasing species richness, on the other hand, was caused by less-specialized species, which are known from mature soils, occurring in very low population sizes. In the course of early succession at the 'Chicken Creek' site, rare cases of species replacement were observed: For example, the thermophilous *Proisotoma minuta*, the second most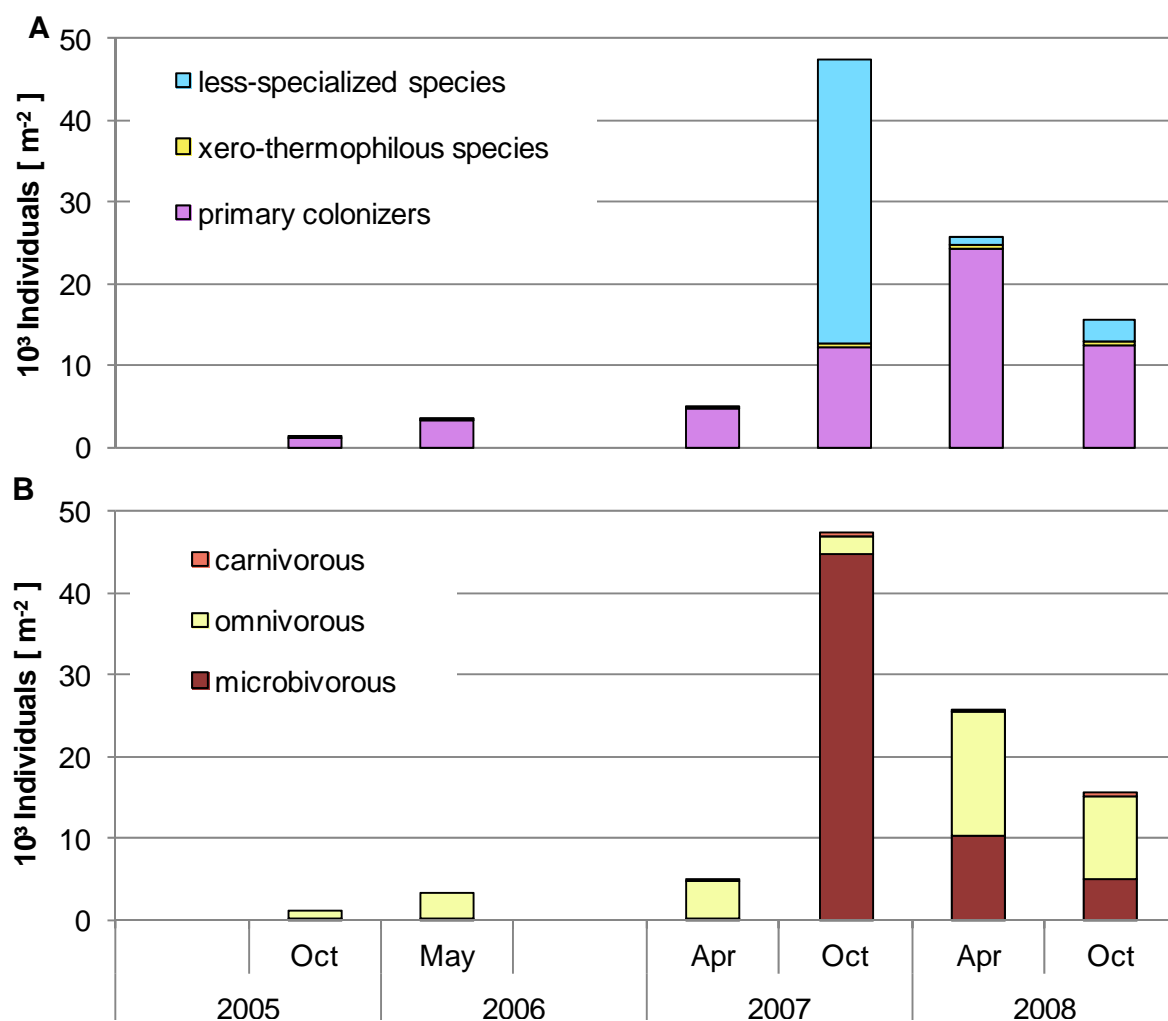 abundant collembolan species until October 2009 (on average 6,128 individuals per m^2, present in 56% of the samples), occurred only sporadically and in single specimens in 2010. It was replaced by *Proisotoma ripicola*, which newly appeared on the site in May 2010 and was present in 81% of the samples and amounted to on average 5,725 individuals per m^2 by October 2010.

Microarthropod communities were initially dominated by microbivores (Fig. 6.9B). During soil development throughout the study period, the communities became more dominated by

omnivores. Predatory species within the Gamasina and Actinedida were not present until 2007. Densities and species richness of these predatory taxa remained extremely low throughout the study and established populations of these taxa were not observed until 2009, four years after site initiation. These results indicate that soil food webs first develop from basal trophic levels and then increase in complexity during primary succession.

6.3.4 Above-ground invertebrates

From May 2008 to May 2009, 4,956 carabid beetles were captured, varying between 194 and 1,061 per trap. The activity-density of carabids varied between 4 and 20 individuals per trap and week, increasing significantly from the upper area to the semi-aquatic area. Up to now, 42 taxa have been observed (3 species, 39 genera). The eu-dominant species *Cylindera arenaria viennensis* (Schrank, 1781) had a share of 42% in total, with increasing importance in the lower area and the semi-aquatic area (Fig. 6.10).
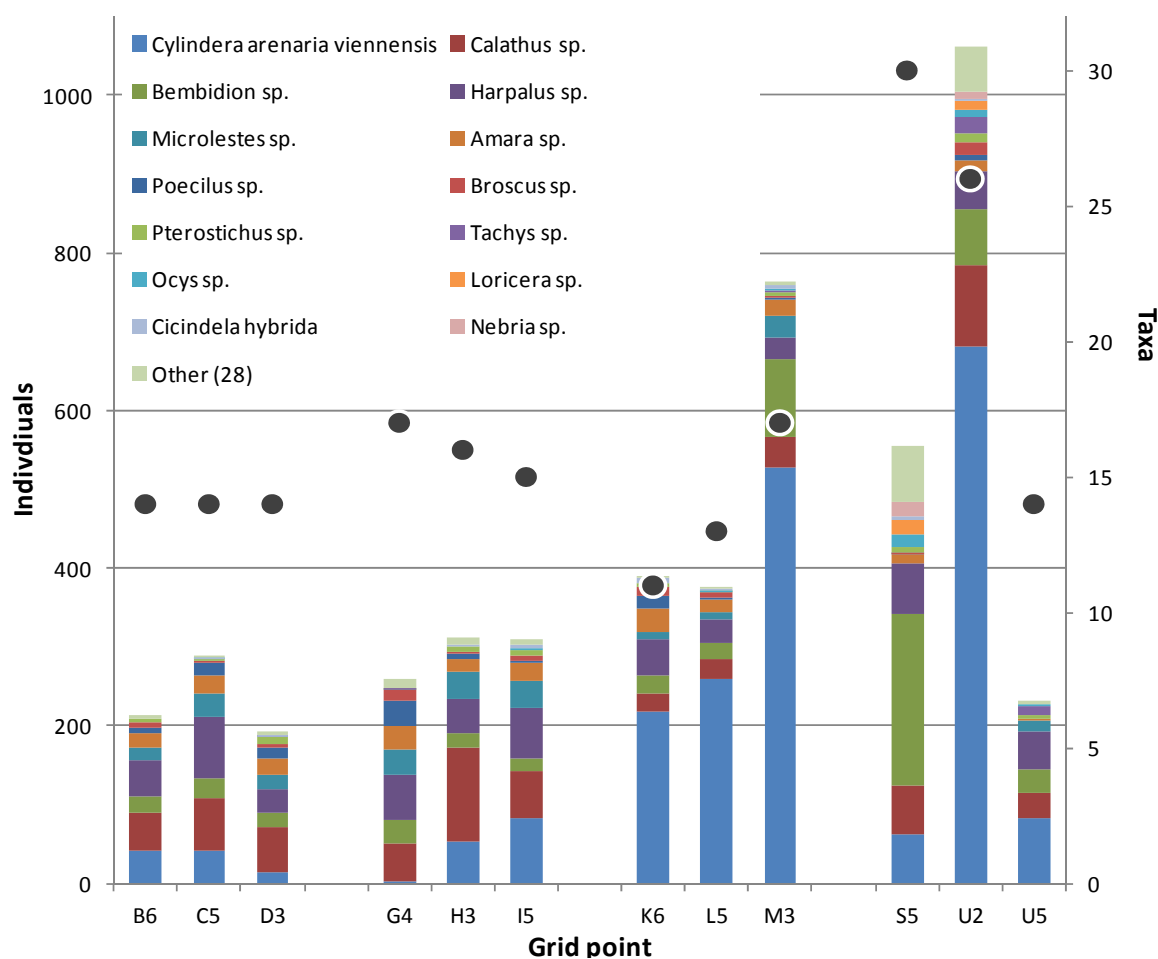

Fig. 6.10:   Carabid beetles found in the 'Chicken Creek' catchment from May 2008 to May 2009, most frequent taxa (bars) and total number of taxa (black dots).

This tiger beetle is a typical pioneer species next to ponds in the post-mining landscape and is threatened by the natural succession of these habitats (Scheffler et al., 1999). This is documented by our data as well: *C. arenaria viennensis* almost disappeared in spring 2009 (2 individuals), in accord with the distinct reduction of active larvae burrows in the catchment that year. In contrast, other taxa such as *Microlestes* sp. (2008: 0, 2009: 225) increased significantly in 2009, indicating the ongoing vegetation succession.

In general, the carabid communities are expected to increase further in diversity and density during the first ten years of primary succession. According to Brunk (2007), the key environmental parameters with regard to plot and catchment are assumed to be succession age, density and structure of the vegetation, soil moisture and pH-value.

6.4   Conclusions

One month after construction of the artificial catchment in autumn 2005, the initial substrates were already colonised by first representatives of nematodes, tardigrades, actinedid mites and collembolans. The 'initial nuclei' of these first colonisers arrived in the catchment site most probably passively by the wind (Wanner et al., 1998; Lehmitz et al., 2011) or phoretically by insects or birds (Athias-Binche, 1991; Walter & Proctor, 1999). It is highly unlikely that individuals actively immigrated since possible sources (developed soils) were several kilometres distant from the catchment. No matter how they arrived, specimens of those early colonising species found conditions good enough and nutrient resources sufficiently available to survive in the newly exposed surface materials, and from there some pioneer species spread almost immediately into individual-rich populations. During the first two years of primary succession, mainly algal and bacterial feeders lived in the 'Chicken Creek' mining substrate. This indicates that microbial and soil-algal communities were present and available from the very start in the newly formed soils.

The number of trophic links within the soil food web increased as of 2007. This concurs with the vegetation becoming richer and more diverse (112 vascular plant species in 2007) and the plant cover exceeding 10% for the first time and within large areas (cf. chapter 5), thus providing an increasing supply of nutrient resources and niches (Ferris et al., 2001; Hohberg, 2003; Russell et al., 2010). Testate amoebal colonisation patterns indicate a strong link between plant cover and testate amoebae. Even though testate amoebae do not directly feed on plant biomass (but predominantly on bacteria), vegetation structure and the development of soil amoebal communities are linked via carbon channels and water supply (Schlesinger et al., 1996). Thus, above-ground catchment development, i.e., the development of a plant cover, is mirrored and can thus be indicated by the development of the testate amoeba community (Wanner & Elmer, 2009). This holds true for carabid beetles as well: in accord with the distinct increase of plant biomass in 2009, a significant reduction of primary colonizers, such

as *C. arenaria viennensis*, was found. Hence, the soil food webs clearly developed in association with the plant community, which is typical for the initial phase of succession (Bardgett & Wardle, 2010). The correlation of above-ground (plant) and below-ground (soil fauna) succession predominantly and most directly holds true for root-feeding nematodes, which were registered from the 'Chicken Creek' catchment as of October 2007 and from then on made up a constant percentage of nematode densities.

Densities and species richness of soil-inhabiting carnivores, on the other hand, remained extremely low throughout the study and established populations were not found until 2009, four years after site initiation. Species numbers were still relatively low: in the fifth year of catchment development (October 2010), altogether 97 soil faunal species (44 Nematoda, 10 Gamasina, 1 Oribatida, 10 Collembola, and at least 18 Actinedida and 14 Testate Amoeba [counted in October 2008 and August 2009, respectively]) were reported from the site. A similar species number would be expected for nematodes and collembolans alone from a single sample of mature soil (20 cm²). We may accordingly expect that many more species remain to come. None of the soil faunal groups under investigation had yet reached its usual species numbers and densities, except for Tardigrada, which show a special affinity to young soils (Hohberg, 2006) and already passed their peak in May 2009.

The results of the present investigation indicate that soil food webs first develop from basal trophic levels and then increase in complexity during primary succession. All major trophic groups – first bacterial, fungal and algal feeders, then plant feeders and omnivores, and finally pioneer predators – may colonise new substrates within a few years, but species inventories of these early successional trophic groups consist of comparably few pioneer species. It will take several decades before the soil food web reaches a complexity and species richness expected from mature soils of comparable latitude and before the last arrivals, e.g., earthworms (Lumbricidae), a highly productive animal group in terms of ecosystem functions, can establish populations within the new substrates (Dunger & Wanner, 2001; Elmer et al., 2007).

Acknowledgements

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7. 'Chicken Creek' pond: aspects of six years of primary succession

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7.1 Introduction

Succession is a fundamental concept in ecology that encompasses the directional and continuous occurrence of a range of successional sequences over varying time-scales (Begon et al., 1990; Tilman, 1985). Ponds have already been in the focus of succession research for more than hundred years. E.g., Cowles (1901) described the primary succession of different ecosystems including ponds in Michigan sand dune systems, and Allee (1911) and Shelford (1913) investigated the animal inhabitation of ponds in different successional stages.

Among lentic water bodies, ponds are ideal subjects for primary succession studies due to relatively fast ecosystem changes. Under certain conditions, the whole lifespan of a pond from its formation to its complete silting-up can be observed within several decades. The succession of small and shallow standing water bodies from open waters to bogs or woodland is known as hydrosere (e.g., Tutin, 1941). Prominent stages within a hydrosere are the phytoplankton stage, the submerged macrophyte stage, the floating plants stage, the reed swamp stage, and the sedge-meadow stage (critical review of the concept by Klinger, 1996). An important basis of succession is the principle of facilitation: earlier successional species pave the way for later ones (Connell & Slatyer, 1977). Ponds are also ideal subjects for the research on land-water interactions because of their relatively long shore line in relation to the water volume.

The formation of an artificial pond in the experimental catchment 'Chicken Creek' offered the unique chance for research on primary succession of this system in dependence on the succession of its catchment area. The development of 'Chicken Creek' pond has more or less frequently been monitored since its formation in 2005, with increasing intensity since 2008. The data of the past six years (except 2007) allow the analysis of the changes in the physical, chemical and biological features of the system. The first four years of pond development and results of specific studies are described in detail by Lessmann et al. (2010a, b) and Lippert et al. (2010). The present report is based on results of the monitoring programme and aims to summarize some major more general limnological aspects of the long-term development of 'Chicken Creek' pond.

7.2 Materials and methods

7.2.1 Site description

Pond succession started in March 2005 when the depression in the clay layer at the lowest point of the catchment was filled by precipitation and surface run-off. The maximum absolute water level of 125.44 m was reached for the first time in January 2006 (cf. chapter 3).

During the first three years, pond morphology changed dramatically due to the high input of sediments, which led to a decrease in pond volume and depth (Lessmann et al., 2010a). As the comparison of the last two pond surveys shows, only minor changes in pond bathymetry occurred between 2008 (for map and morphometric data cf. Lessmann et al., 2010a) and 2010 (Fig. 7.1). This corresponds with the decrease in erosion processes (cf. chapter 2), the vegetation development in the catchment area (cf. chapter 5) and the increasing growth of macrophytes in the main inflow area, which act as an effective sediment trap. Changes in sedimentation rates and sediment characteristics are described in chapter 8.
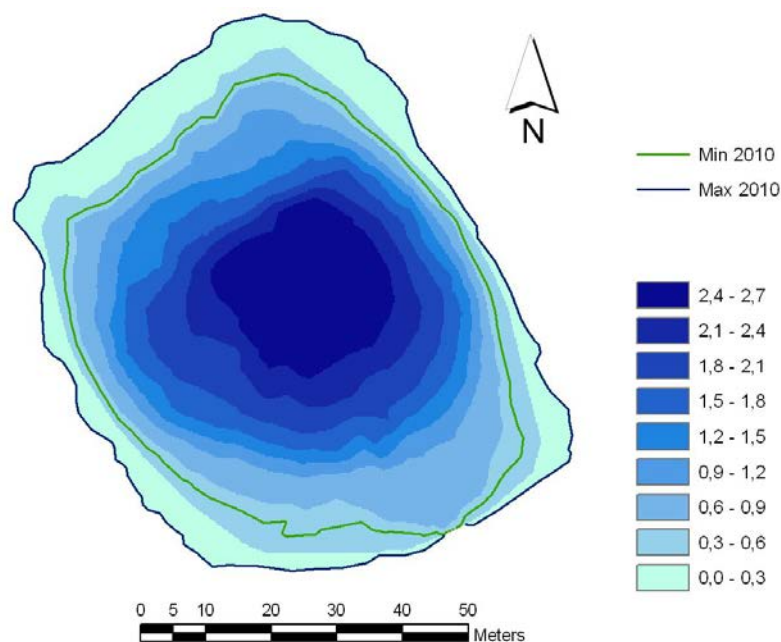
Fig. 7.1:   Bathymetric map of 'Chicken Creek' pond according to the survey of 2010 with minimum and maximum water level in 2010 (map: Nenov).

From the beginning of vegetation development, reed (*Phragmites australis*) and, with minor importance, cattail (*Typha latifolia*) were the dominant plants that surrounded the pond. High sediment accumulation was apparent along the northern shore within the inflow area. The dense vegetation is clearly visible in aerial photos (Fig. 7.2). With 2.4 m, maximum depth stayed constant in 2010 and 2011. However, a further significant decrease in depth can be

expected from sedimentation and accumulation of autochthonous organic material or from erosion during floods. The latter may cause a severe disturbance of the system.

Fig. 7.2:   Aerial photo of 'Chicken Creek' pond from June 29th, 2011 (photo: Nenov).

Submerged macrophytes became of increasing importance from year to year. Since 2008, macrophyte species composition has been stable and consists of three *Potamogeton* species (*P. pectinatus*, *P. lucens*, *P. natans*), *Myriophyllum spicatum*, and *Chara globularis*. Certain species use to inhabit specific areas of the pond (also visible in Fig. 7.2). In 2009, *P. pectinatus* was the most dominant macrophyte, followed by *P. lucens* and *M. spicatum*. *P. natans*, and *Ch. globularis* showed only small quantities. After rapid dispersion in spring, macrophyte growth was stagnant in early summer but increased again from August to the middle of October. Fluctuations were mostly due to *P. pectinatus* and *M. spicatum* (Fig. 7.3).

7.2.2   Methods

The sampling methods as well as the different analytical methods for the physical, chemical, and biological analyses are described in detail by Lessmann et al. (2010a). Since March 2009, monitoring has been continued by frequent, mostly monthly sampling by the same methods.
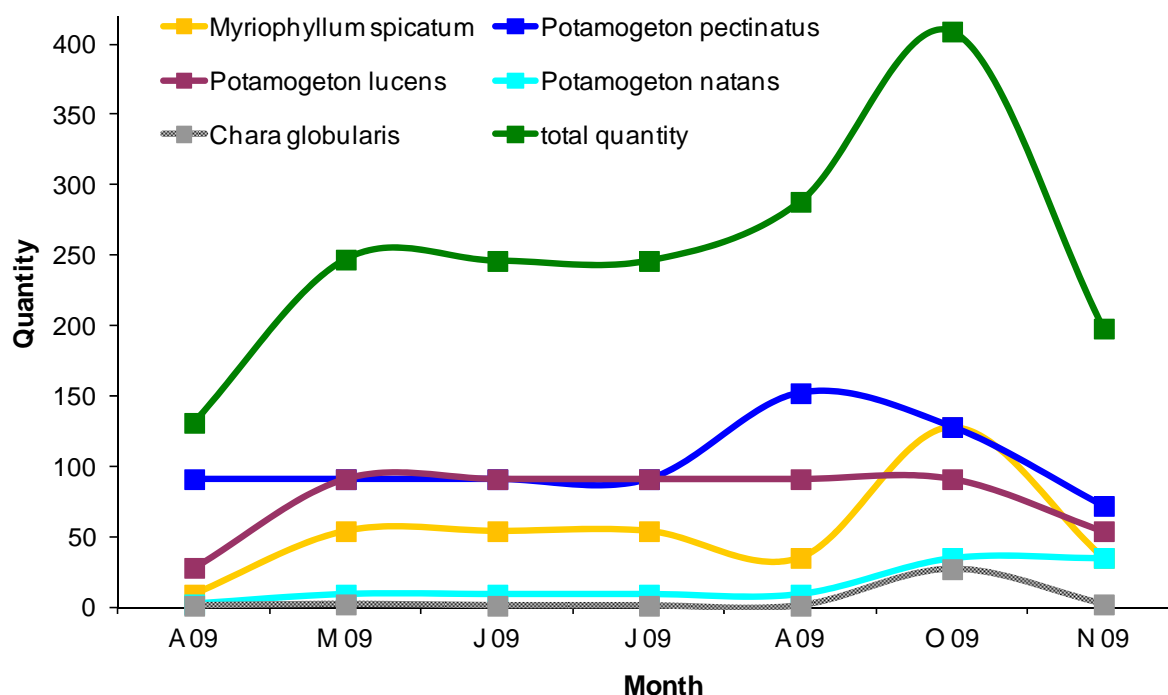
Fig. 7.3: Quantity of macrophyte coverage according to Melzer (1988) in 'Chicken Creek' pond from April to November 2009.

For this report, data were considered until December 2010 and June 2011, respectively. Plankton and macrophyte data were not available for all years; generally no data were available for 2007.

7.3 Results and discussion

7.3.1 Main ions

Except for a sharp decrease in 2006, total ionic concentration of the pond water was relatively stable as indicated by electrical conductivity (Fig. 7.4). Since 2008, the annual mean was between 550 and 600 μS cm^{-1}, which is within the range of hard water bodies. Nevertheless, anion proportions changed (Fig. 7.5). Due to continuously decreasing sulphate concentrations (Fig. 7.4) on the one hand and increasing bicarbonate concentrations on the other hand, bicarbonate replaced sulphate as the major anion. This change was correlated with an increase in alkalinity (cf. chapter 7.3.2) and indicates a maturation of the geochemical system of the catchment. This tendency directly refers to similar changes found in soil solution and water chemistry of the terrestrial parts of the catchment (cf. chapter 4). In contrast, cation contents were more or less stable with calcium as the major cation with about 80-95 mg L^{-1} (Fig. 7.4). In addition, only magnesium occurred in concentrations of more than 5 mg L^{-1}.
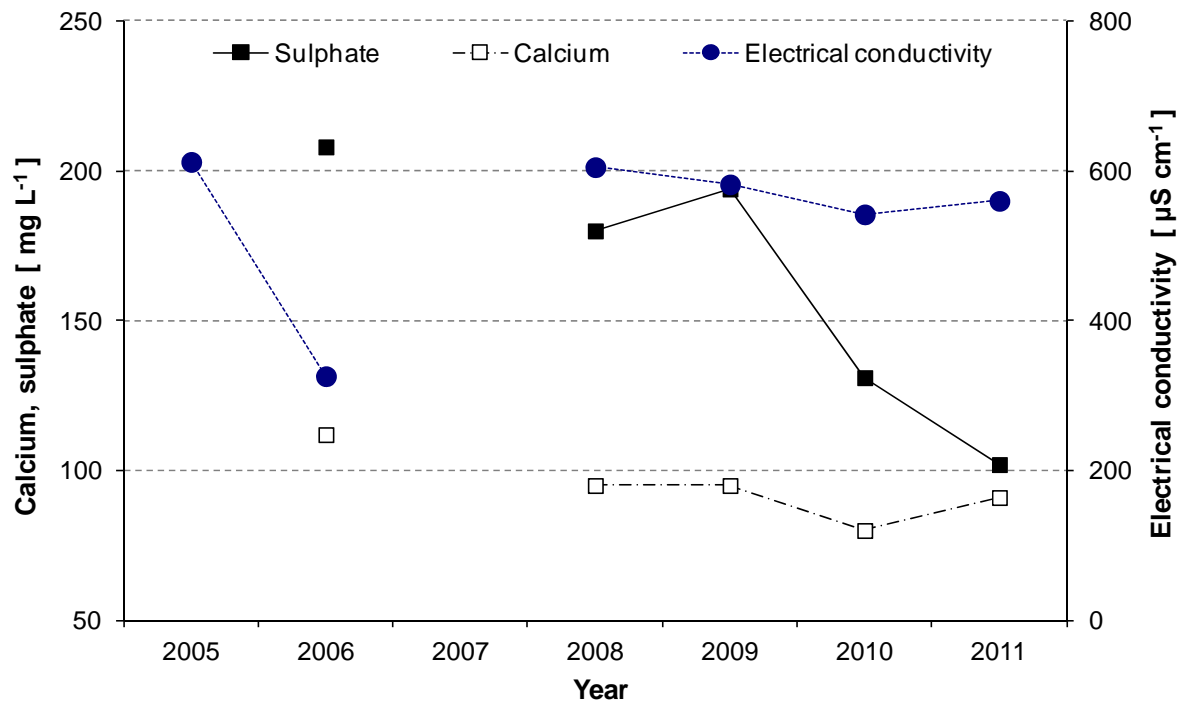
Fig. 7.4: Electrical conductivity and concentrations of calcium and sulphate in 'Chicken Creek' pond from 2005 to 2011.
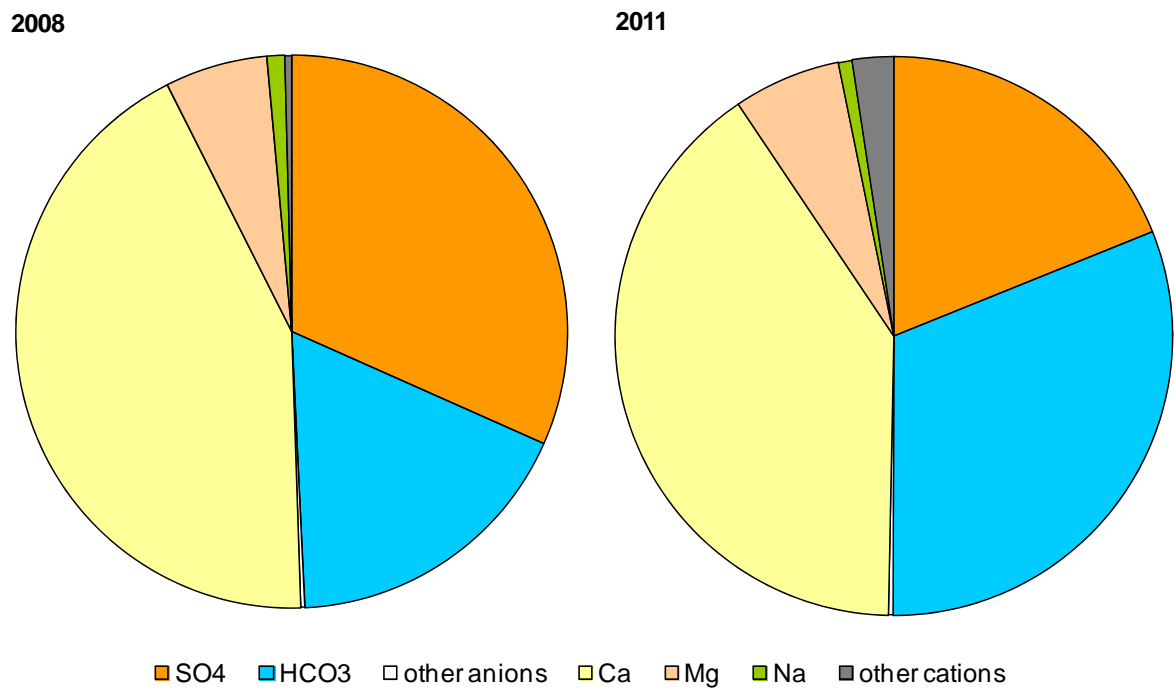

Fig.7.5: Annual mean proportion of different cations and anions of water from 'Chicken Creek' pond in 2008 and 2011.

7.3.2 pH and alkalinity/acid capacity $K_{S4.3}$

Right from the start, pH of the pond water was in the slight to moderate alkaline range with the highest pH in 2006. Since 2008, fluctuations were between 7.8 and 8.2 (Fig. 7.6). The pH was stabilized by increasing alkalinity due to the increase in bicarbonate ions. Between 2009 and 2011, acid capacity ($K_{S4.3}$) data more than doubled from 1.5 to 3.5 mmol L^{-1} (Fig. 7.6), which indicates the formation of a strong bicarbonate buffering system.
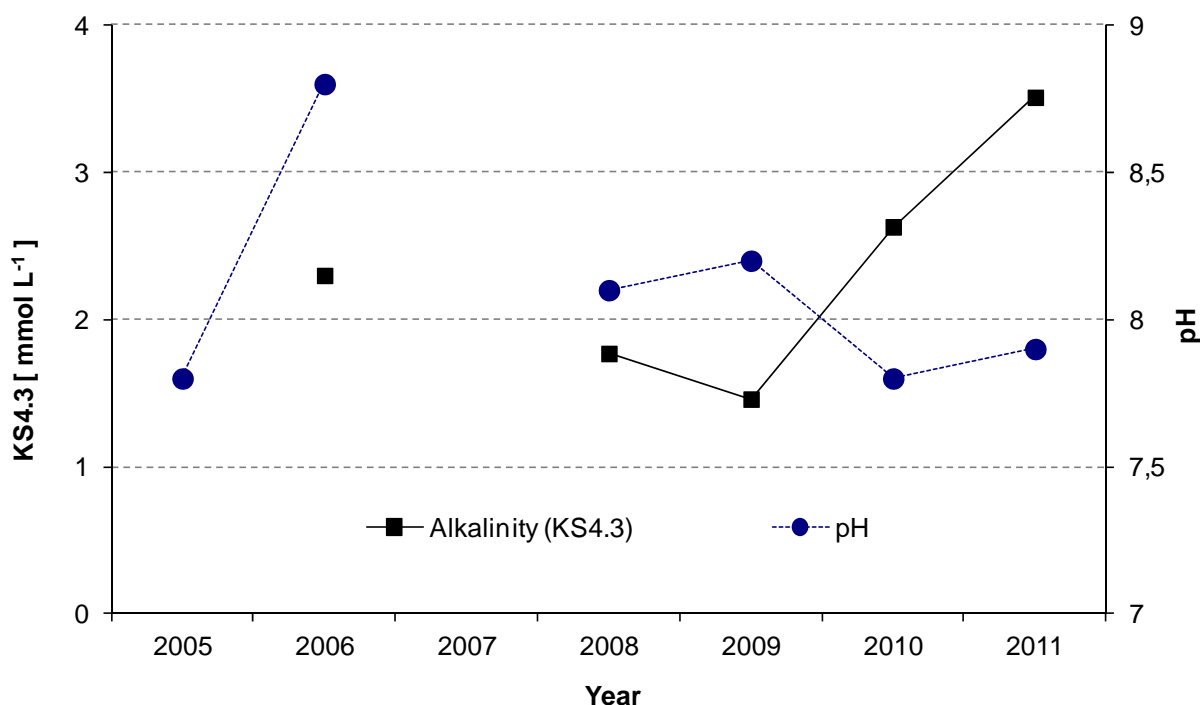
Fig. 7.6:   Annual mean pH and alkalinity/acid capacity $K_{S4.3}$ of water from 'Chicken Creek' pond from 2005 to 2011.

7.3.3 Nutrients

Since phosphorus is the regulator of pelagic primary production in 'Chicken Creek' pond (Lessmann et al., 2010a, b), the development of phosphorus concentrations is of special importance for the whole aquatic system. Continuously low dissolved inorganic phosphorus (DIP) concentrations below 10 μg L^{-1} (Fig. 7.7) show that the available phosphorus was always rapidly taken up by organisms or bound by geochemical processes. The effectiveness of P uptake seemed to increase, because data showed a continuous slight decrease in DIP concentrations since 2005. Great differences occurred in total phosphorus (TP) concentrations. The annual mean in 2005 and 2006 was about 45 μg L^{-1} and considerably higher than in the period from 2008 to 2010 with 19-33 μg L^{-1} (Fig. 7.7), which can be attributed to the increasing macrophyte growth and the resulting more permanent binding of P

in their biomass (the role of macrophytes on the development of the phosphorus cycle of the pond is intensively discussed in chapter 8). With regard to TP concentrations the pond always was in the mesotrophic range.
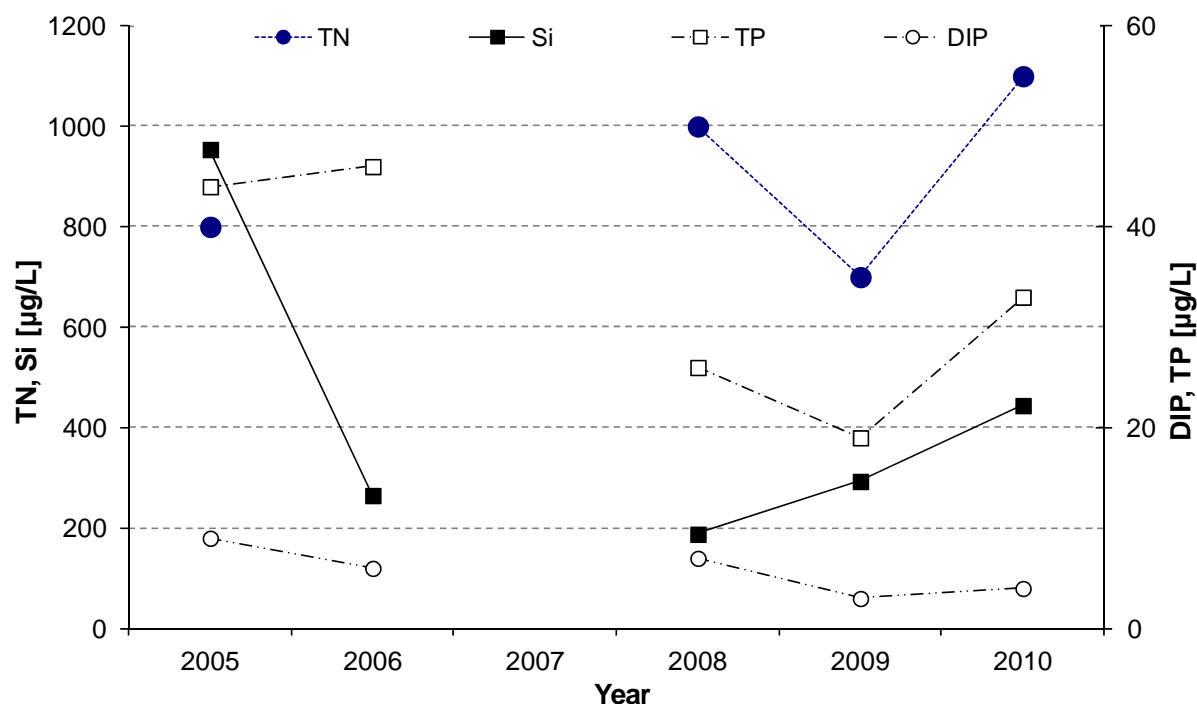

Fig. 7.7:   Annual mean concentrations of phosphorus (DIP: dissolved inorganic phosphorus, TP: total phosphorus), nitrogen (TN: total nitrogen), and silicon (Si) in water from 'Chicken Creek' pond from 2005 to 2011.

Continuous high fluctuations also were typical for nitrogen concentrations (Fig. 7.7). Annual mean concentrations always were low in the range between 0.7-1.1 mg L^{-1}. For diatoms, silicon concentrations also are important. They completely depend on the input from the catchment area. After high concentrations in the first year of pond development, dissolved silica was scarce in 2006. Since 2008, a continuous increase occurred from about 200 µg L^{-1} to about 450 µg L^{-1} in 2010 (Fig. 7.7).

7.3.4   Oxygen

With regard to the annual mean oxygen saturation (Fig. 7.8) the pond seemed to be a net-heterotrophic system since 2008, which means that oxygen consumption by respiration exceeded oxygen production by photosynthesis in total. But the oxygen saturation data allow only a rough estimation, only the measurement of production and respiration processes can prove that. Lippert et al. (2010) discuss production processes in the pond in 2009 in detail.
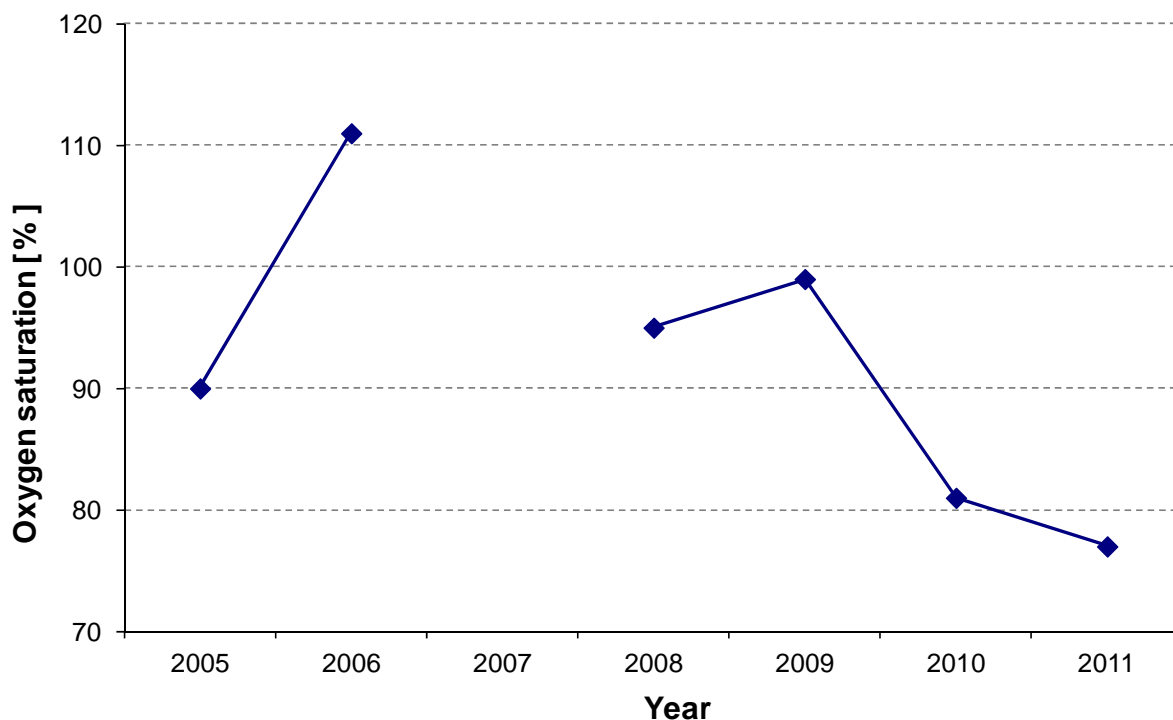
Fig. 7.8: Annual mean oxygen saturation in the water column of 'Chicken Creek' pond from 2005 to 2011.

7.3.5 Phytoplankton

Species number

Phytoplankton data are available for comparison for the period from 2008 to 2010. The approximate total number of phytoplankton taxa increased from 70 in 2008 to 100 in 2009 and decreased afterwards to 90 in 2010 (Fig. 7.9). 43 taxa occurred in every year, 13 taxa only in 2008, 35 in 2009, and 36 in 2010, respectively. The number of species that occurred in two consecutive years was low with 11 and 12 taxa, respectively. Only one taxon was an inhabitant of the pond in 2008 and 2010 but missing in 2009. Several studies proofed that primary inhabitation and to a certain degree also primary succession of relatively isolated systems as 'Chicken Creek' pond can be considered as strongly stochastic with a high importance of random events (e.g., Talling, 1951; Margalef, 1963; Titus et al., 1999).

Chlorophyll-*a*

The intensity of phytoplankton biomass and pelagic primary production is reflected by chlorophyll-*a* concentrations. While annual mean concentrations were only 5-7 µg L^{-1} in 2005, 2008, and 2009, an increase to 25 µg L^{-1} occurred in 2010. Also in 2011, the first semi-

annual mean already reached 15 µg L^{-1}. The comparison of sampling date based chlorophyll-*a* data since 2008 showed high fluctuations within short periods and no typical seasonal pattern of phytoplankton development up to now (Fig. 7.10).
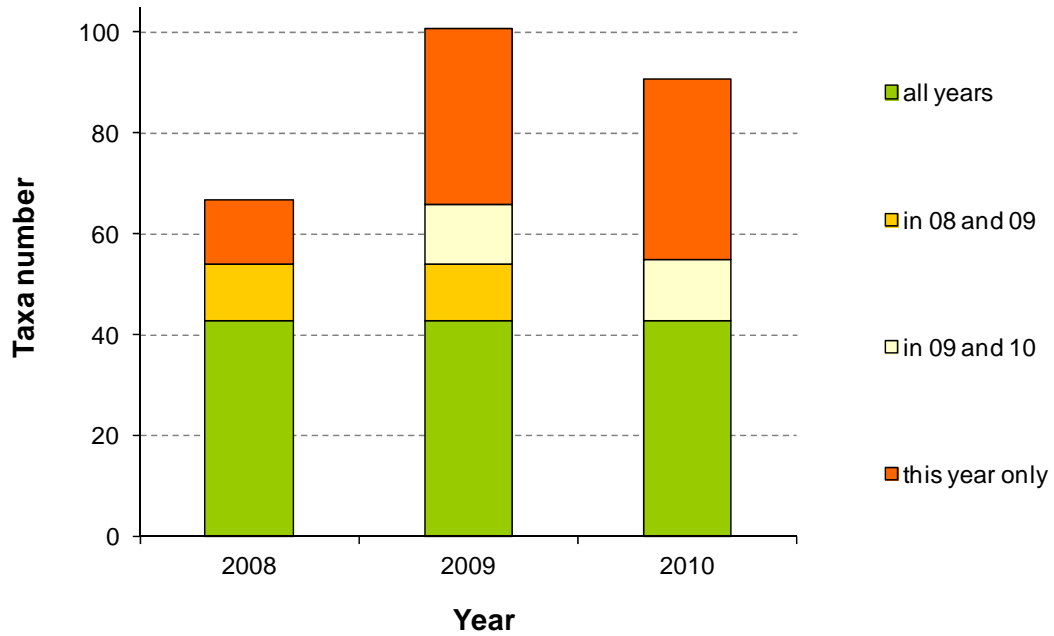

Fig. 7.9:   Number of phytoplankton taxa in 'Chicken Creek' pond from 2008 to 2010 with the frequency of their occurrence.
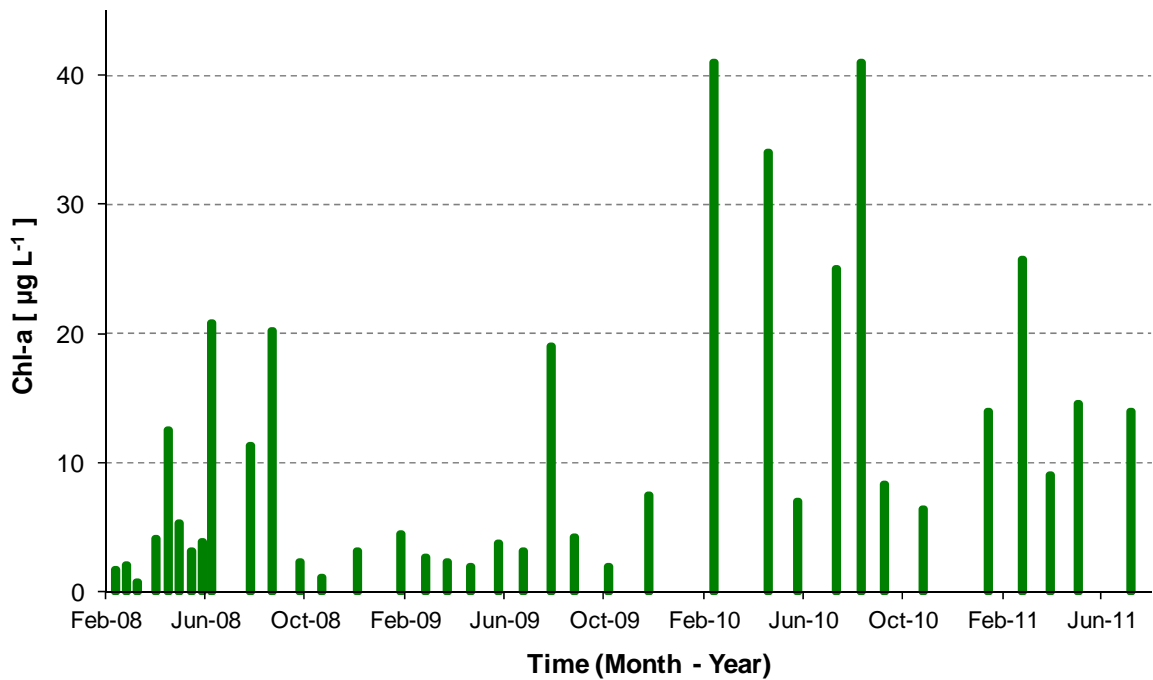
Fig. 7.10:   Mean chlorophyll-*a* concentrations of the water column of 'Chicken Creek' pond on the sampling dates from February 2008 to June 2011.

In 2008, chlorophyll-*a* concentrations increased during the summer from June to August. In 2009, only one single chlorophyll-*a* peak occurred in August. The rest of the year, concentrations were always lower than 5 µg L^{-1}. That can be attributed to the low nutrient concentrations (P and N) in that year; macrophyte growth also has to be considered. In contrast, in 2010 and continuing in 2011 chlorophyll-*a* reached high concentrations nearly continuously from already February to August with a short interruption in May, which can be a hint for a clearwater phase caused by zooplankton grazing.

Species and biovolumes

In correspondence with chlorophyll-*a* concentrations, phytoplankton biovolumes showed a similar development from 2008 to 2010 (Fig. 7.11).
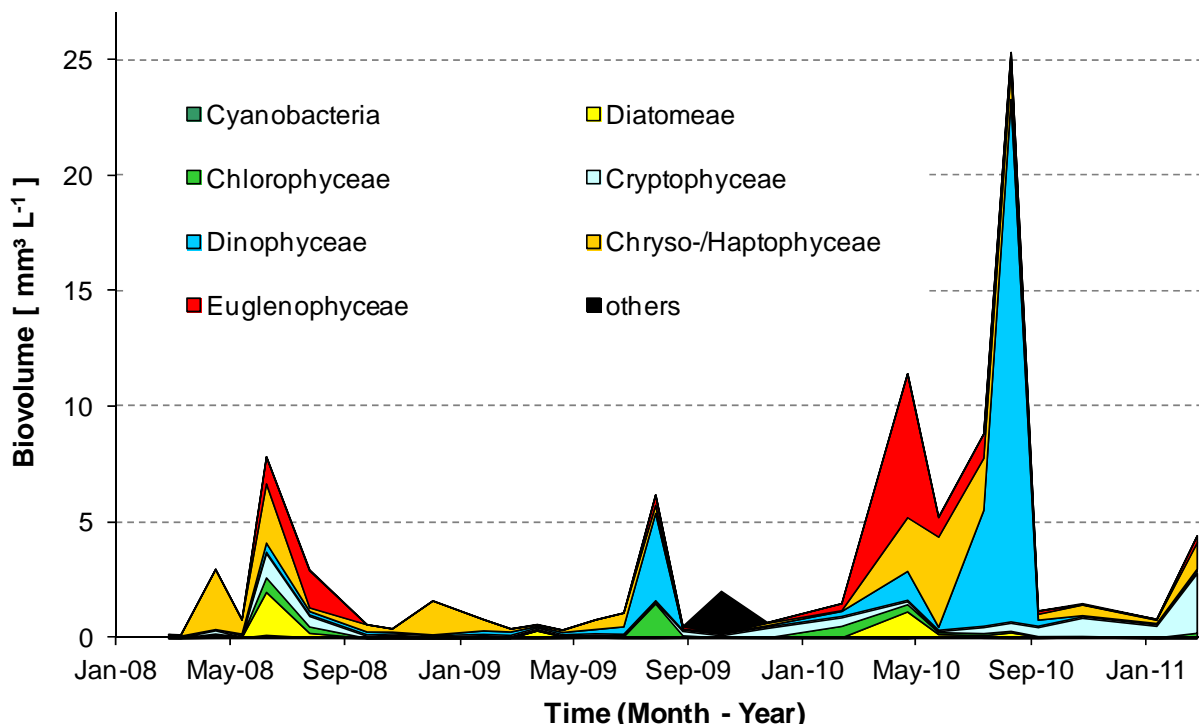
Fig. 7.11: Class based cumulative biovolume of phytoplankton in 'Chicken Creek' pond from February 2008 to February 2011.

The biovolume maximum in June 2008 was formed by *Pseudopedinella erkensis* (Chrysophyceae), *Fragilaria* sp. (Bacillariophyceae), and *Euglena* sp. (Euglenophyceae). Different Chrysophyceae also were the most dominant species at other times of the year. The year 2009 was characterized by two smaller phytoplankton biovolume peaks, which were due to *Ceratium hirundinella* (Dinophyceae) in July and *Spyropgyra* sp. (Conjugatophyceae) at

the beginning of October. 2010 spring and summer blooms were formed by *Euglena gasterosteus* (Euglenophyceae) in April and by the large *Ceratium hirundinella* (Dinophyceae) in July and August, while in May *Dinobryon sociale* (Chrysophyceae) was the most dominant species. In August 2010, *Ceratium hirundinella* contributed 85% to total biovolume, although several smaller species were much more abundant. Such blooms of *Ceratium* often can be observed in lakes in late summer. The bloom stops suddenly, when the organisms simultaneously form cysts and disappear from the water column.

Statistics show, that mean and maximum phytoplankton biovolumes were in a similar range in 2008 and 2009 with an annual mean of 1.9 and 1.3 mm³ L^{-1} and maxima of 7.8 and 6.2 mm³ L^{-1}, while the annual mean increased to 7.8 mm³ L^{-1} and the maximum to 25.3 mm³ L^{-1} in 2010.

7.4   Conclusions

Ecological succession is a series of transitions in the development of ecosystems that are linked with changes in environmental conditions and control biological inhabitation. Such changes can result from changing external factors and/or be controlled by internal processes (Begon et al., 1990). With regard to the succession of ponds and lakes, catchment succession and changes in internal matter cycling determine the velocity of primary succession. Milner et al. (2007) differentiate between different phases in ecosystem development in Glacier Bay/Alaska after glacial retreat and the start of succession in this newly formed landscape. In their studies, lake succession is mainly driven by catchment succession through soil formation and weathering as well as vegetation development. Most important determinants are physical and chemical changes in the terrestrial part of the catchment, which lead to changing inputs of sediment, major ions, nutrients, and organic carbon. The results of the monitoring programme of 'Chicken Creek' pond are similar to their findings.

Up to now, 'Chicken Creek' pond underwent two phases of succession. The first phase comprised the first two to three years of pond development (2005-2007). That period was dominated by major morphological changes due to high sediment input from the catchment area, which reduced depth and volume and increased turbidity. In the second phase of pond development since 2008, the increasing abundance of macrophytes in the terrestrial parts of the catchment led to a decrease of erosion rates. Together with the formation of dense reed stands in the semi-aquatic parts of the pond and especially in the inflow area, sediment transport from the catchment area into the pond was much lower. Those changing conditions fostered the growth of aquatic macrophytes in the second phase of pond development.

In the period from 2008 to 2010, phytoplankton reached a remarkable total of ca. 150 taxa. The proportion of taxa that occurred only in single years was relatively high. Together with

the continuous occurrence of only 43 taxa in all three years, a relative instability of the composition of the phytoplankton coenosis was indicated. From matured Lake Stechlin, for a 15-year period 212 species are reported, 97 to 122 species occurred per year (Padisak et al., 2010). Nevertheless, the high number of species clearly shows that a new isolated water body can already be inhabited by a wide variety of algae within a few years. This guarantees high flexibility and efficiency in the use of resources also in young ecosystems.

With regard to primary production, the first phase was dominated by pelagic algae with a control of algae biomass by phosphorus and light. Ponds with their shallow water and only slight motion of water are ideal habitats for macrophytes. Therefore, it took only a short period until submerged macrophytes biomass exceeded phytoplankton biomass. In summer 2009, macrophytes primary production was up to ten times higher than phytoplankton production (Lippert et al., 2010). Nevertheless, in 2010 a longer lasting sharp increase in phytoplankton biomass occurred. The bloom of the large dinoflagellate *Ceratium* is a common phenomenon, which is not correlated with a parallel increase in phytoplankton abundance. In 'Chicken Creek' pond also other species contributed to the difference from previous years. This deviation needs further investigations, because it cannot be explained e.g., by nutrient availability.

Nutrient availability for primary producers especially changed with regard to phosphorus. While phosphorus concentrations reached the upper end of the mesotrophic state in 2005/2006, they were much lower in the period from 2008 to 2010. This corresponds with the increased growth of submerged macrophytes, which extract higher amounts of phosphorus from the water column.

Chemical characteristics of pond water are determined by hydrological conditions, inflowing water quality, and internal processes, which predominantly occur at the sediment-water interface. Water data of 'Chicken Creek' pond showed a continuous change in anion proportions: a steady decrease in sulphate and increase in bicarbonate. Total ion concentrations stayed more of less stable without clear trend. Processes in the soils of the catchment area and in the sediments of the pond have to be considered for explanations (cf. chapters 4, 8, and 9).

Altogether, monitoring proved the significance of catchment succession for the development of the pond, but also showed the increasing importance of internal processes. If the external influence remains stable and shifts occur only slowly, it can be expected that further pond development will mainly be regulated by internal development. Larger changes in the catchment by, e.g., severe meteorological conditions will act as disturbance and can, in the extreme case, cause an interruption of primary succession and result in secondary succession.

Acknowledgements

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8. Sediment accumulation and impact of aquatic macrophyte decomposition on sedimentary nutrient and metal mobilization in initial ecosystem development

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8.1   Introduction

To study processes and structures of initial ecosystem development an artificial catchment area was created (Gerwin et al., 2009). This catchment discharges into a man-made experimental pond which was created in 2005 (Kleeberg et al., 2010a). Recent studies on pond development showed, that the sediment accumulation rates were very high and catchment soil properties still determine the chemical sediment quality and low reactivity (Kleeberg et al., 2010b). The comparison of nutrient-poor pond sediments with those of nutrient-rich shallow lakes revealed that a sedimentary phosphorus (P) cycle, crucial for the future pond development, was dominated by P accumulation representing an initial stage of development. However, we hypothesize that the initial high sedimentation rate will decrease and that an enhanced emersed and submersed macrophyte growth and decomposition cycle will control further pond succession by concurrently intensifying the internal P cycling.

Pioneer colonization, growth and decomposition of macrophytes can considerably influence nutrient cycling and energy flow in aquatic ecosystems (e.g., Barko & James, 1998; Dinka et al., 2004). During growing season, macrophytes accumulate nutrients from both sediment and water (e.g., Pelton et al., 1998). When the macrophytes die, decomposition process starts, which in turn can substantially regulate the recycling of nutrients in fresh water ecosystems for a long time (e.g., Pieczyńska, 1993; Shilla et al., 2006). In initial ecosystem development, a key issue of discussion is, whether the biota at a site determine their own future by modifying their own environment, or if the development of an ecosystem is simply a response to the external environment (Mitch & Gosselink, 2000).

Weathering of P-containing minerals in the catchment area is usually a slow process (e.g., Sharpley et al., 1999). Thus, only little P was mobilized and accumulated in the pond (Kleeberg et al., 2010b). Pond-internal sedimentary processes, however, could considerably enhance P mobilization. Living macrophytes can alter sediment biogeochemistry resulting in varying pore water P, solid-phase P, and metal levels (Wigand et al., 1997). Decomposing macrophytes can supply organic matter (Bärlocher & Graca, 2005) increasingly stimulating microbial and redox processes (e.g., Longhi et al., 2008) in the pond.

It has been hypothesized that higher nutrient concentrations are an important factor in controlling decomposition rates (Xie et al., 2004; Shilla et al., 2006). The reasons are that nitrogen and particularly P demands associated with decomposer activity often exceeds P supply from decomposing material. In the pond, pore water P concentrations were relatively low. Hence, until now it remains elusive how this type of sediment behave.

Moreover, it can be also assumed that metals concurrently released during plant decomposition could contribute to P binding, lowering direct P availability to decomposers and the potential for being released to overlying water and/or other biota. This applies particularly to iron (Fe) and manganese (Mn) known to be efficient P binding partners. P is co-precipitated with oxidized Fe and Mn species, and adsorbed to their amorphous oxyhydroxides (e.g., Christensen et al., 1997; Martynova, 2010). However, in studies on aquatic macrophyte decomposition P binding partners have not been regarded yet. Moreover, the availability of elements such as Fe can influence (further) macrophyte species composition (Lamers et al., 2002).

This present study reports the repetition of a previous study on sediment accumulation which was complemented by a sediment trap campaign. The decomposition of three aquatic macrophytes already thriving in the pond (*Phragmites australis* (Cav.) Trin. ex Steudel, *Potamogeton natans* L., and *P. pectinatus* L.) was investigated with regard to the implications on initial ecosystem development, and pond succession. In particular, the extent of P regeneration as directly related to metal mobilization should be evaluated. We chose these plant species to answer the question whether or not their decomposition contributes to larger fluxes of P through this immature ecosystem.

8.2 Materials and methods

8.2.1 Site description

'Chicken Creek' pond is part of the artificially established and hydrologically defined same-named catchment in the post-mining landscape, which was developed by a Collaborative Research Centre (SFB/TR 38, Gerwin et al., 2009). The pond is situated on the south east of an artificial hillslope 'Chicken Creek' pond, with a crater-shaped bed, was created in 2005. The asymmetric round hollow basin (originally about 55-65×3 m) with a rectangular bottom area was already partly filled in August 2005 and completely filled after the snowmelt in January 2006 (Kleeberg et al., 2010b). Details of the pond morphology and basic water chemistry are described in chapter 7 (this volume).

8.2.2 Sampling and analysis of sediment and settling particulate matter

To follow the sediment accretion and changes in quality the campaign on August 14th, 2008 (Kleeberg et al., 2010b) was repeated on August 10th, 2010. For determining the extent of accumulated sediment layers, transects were taken from inflow to the outflow (north-south transects) and crossways (west-east transects). Along each transect undisturbed sediment cores (Ø 6 cm) were taken every 10 m – until the bottom clay layer was reached – by means of a sediment corer with a telescope bar (UWITEC©, Mondsee, Austria).

The sediment cores were the sliced into layers of various thickness (0-0.5, 0.5-1, 1-2, 2-3, 3-4, 4-6, 6-8 cm etc.) down to the bottom clay layer. Aliquots of sediment fresh weight (fw) were used for the determination of dry weight (DW, 105 °C, 8 hrs). Based on the dry weight the proportion of organic matter (OM) was determined as loss on ignition (450 °C, 3 hrs).

Settling particulate matter was trapped in two sedimentation traps, made of transparent PVC pipes with a length of 60 cm and an inner diameter of 8.7 cm. These pipes were mounted on a rectangular PVC base plate each, which were positioned on the sediment surface in the inflow and outflow area at about 2 m water depth. After removing larger particles such as snail shells, the entrapped material was dried and analyzed in the same manner as the sediment material.

8.2.3 Laboratory experiment on plant decomposition and phosphorus mobilization

To study the effect of decomposing plants on P mobilization a lab experiment was carried out with three macrophyte species indigenous in the pond. At the end of growing season, on September 20th, 2009, fresh leaves and stem material of Common Reed *Phragmites australis* (Cav.) Trin. ex Steudel (= *Phau*) was sampled above water surface. Whole plants of the Floating Pondweed *Potamogeton natans* L. (= *Pona*) and the Fennel Pondweed *Potamogeton pectinatus* L. (= *Pope*) were harvested by a commercial garden rake from a water depth of about 1.2 m.

The plant material was thoroughly washed with pond water. After dripping off most of the water the plants were hackled using a double-edged chopping-knife. Hackled plant parts were used because of the different chemical composition of different reed organs and the differences in their decomposition rate and activity of microbial decomposers (Dinka et al., 2004). About 30 sediment cores were taken simultaneously by a gravity corer (Ø 60 mm; UWITEC, Mondsee, Austria). Only the upper 5 cm sediment layers were pooled. This sediment slurry was mixed with an increasing mass of hackled plant fresh weight (Tab. 8.1).

Tab. 8.1: Plant species and dry weight of sediment and plant biomass [g] from 'Chicken Creek' pond mixed and used in the decomposition experiment from September 30th, 2009 to January 30th, 2010.

Identification of samples	*Phragmites australis* (= *Phau*)		*Potamogeton natans* (= *Pona*)		*Potamogeton pectinatus* (= *Pope*)	
	sediment	plant	sediment	plant	sediment	plant
Control (= Con)	262.6	÷	276.6	÷	286.4	÷
1	259.4	8.1	261.6	2.6	333.9	3.3
2	259.7	16.2	261.4	5.0	261.8	9.7
3	259.6	32.4	261.7	10.1	261.6	19.3
4	259.3	64.9	194.5*	20.1	129.6*	38.6

* *Pona₄* and *Pope₄*: sediment diluted 1:1 with pond water in advance.

These sediment-plant slurries were incubated at 10 °C in the dark for 124 days. For pore water sampling two 18 ml glass vessels were pushed into each mixture. These vessels were covered by a dialysis membrane (polysulfone, 0.2 µm HT-Tuffryn 200®, PALL GmbH, Germany) and incubated until equilibration (8-63 days).

8.2.4 Pore water analysis

Concentration of soluble reactive P (SRP) and ammonium (NH_4^+) of pore water were photometrically determined by the molybdenum-blue method at $\lambda = 880$ nm (Murphy & Riley, 1962) and by a modified indophenole method at $\lambda = 660$ nm (Krom, 1980) using a segmented flow analysis (SFA, Skalar San^plus, Skalar Analytical B.V., De Breda). The concentrations of dissolved iron (DFe) and dissolved manganese (DMn) were determined by AAS (Perkin Elmer 3300, Rodgau-Juegesheim). For details cf. Zwirnmann et al. (1999).

8.2.5 Data handling

The statistical differences among the mean values of the sediment constituents of consecutive years were determined by respective t- and Welch-tests, and relationships among environmental parameters and the sedimentation rate were tested using a linear model (Sachs, 1988).

8.3 Results and discussion

8.3.1 Sediment accumulation and composition

In concordance with the different periods of sediment accretion (short-term up to 30 cm yr^{-1}) the proportion of organic matter (LOI) continuously increased with time and towards the sediment surface, respectively (Fig. 8.1). For a detailed description of the event-related sediment accumulation between 2005 and 2008 see Kleeberg et al. (2010b). The accumulation of sedimentary organic matter was compared in two campaigns in 2008 and 2010. Although there is an increase in LOI for the three areas of the pond previously distinguished, only the increase in LOI for the inflow by 1.7% DW was significant ($n = 8$, $p = 0.05$), whereas that for the basin and the outflow by only 0.8% DW and 0.6% DW was not.
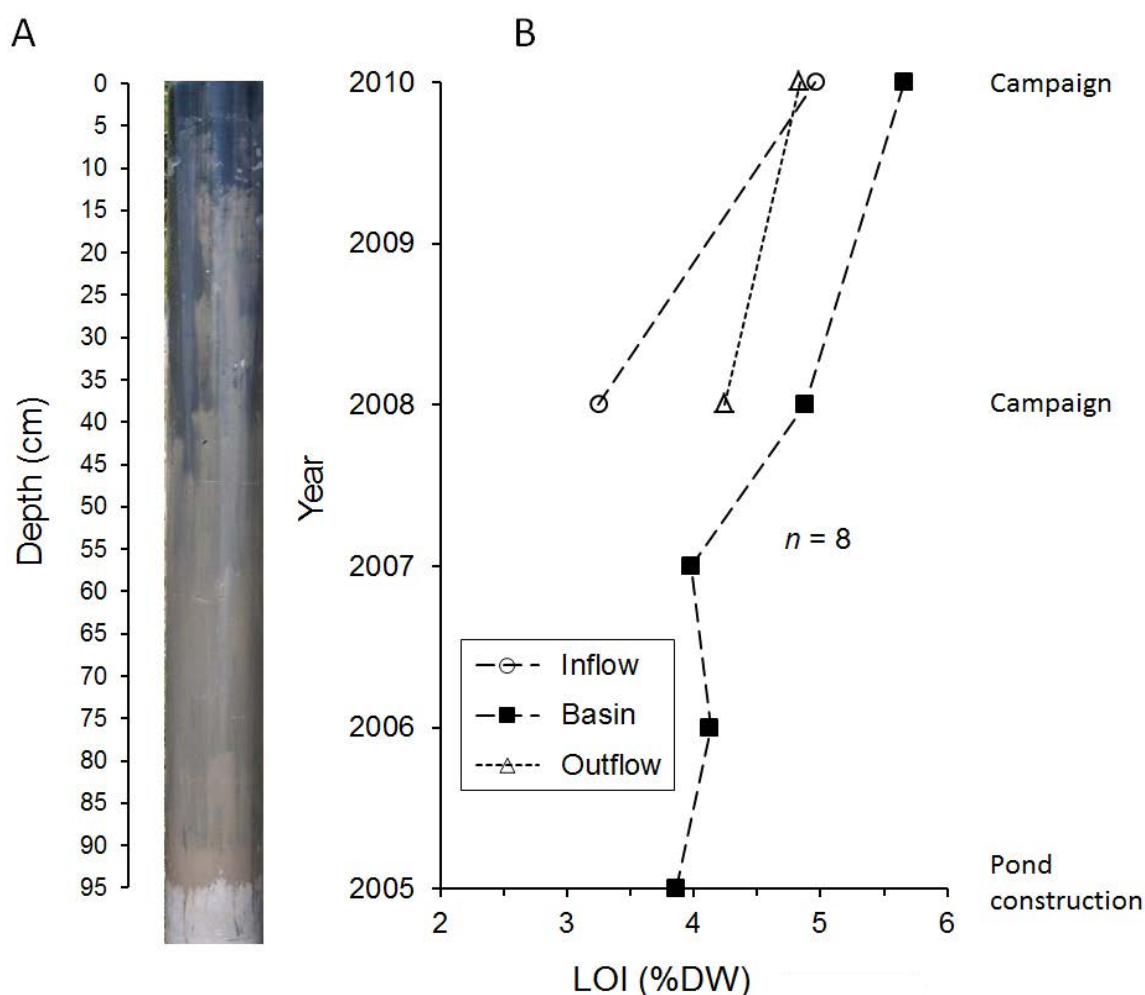

Fig. 8.1:   A: Photograph of an undisturbed sediment core from the deepest site of the experimental pond 'Chicken Creek', August 10th, 2010; B: Mean values of loss on ignition (LOI) of surface sediment (0-5 cm) of the different parts of the pond in August 2010 in comparison to those of a campaign in August 2008, and to data from a previous study (Kleeberg et al., 2010b).

The vertical profile of sediment stratigraphy reveals that the sediment accumulation rate decreased to about 17 cm yr^{-1} between 2008 and 2010. The slow increase in LOI reflects the slowly increasing supply of organic carbon both from the catchment and from autochthonous sources, the biomass of various plants (see below).

8.3.2 Sedimentation

The air temperature showed a regular seasonal pattern except July 2011 where a peak in precipitation caused a decrease in temperature compared to July 2010 (Fig. 8.2).
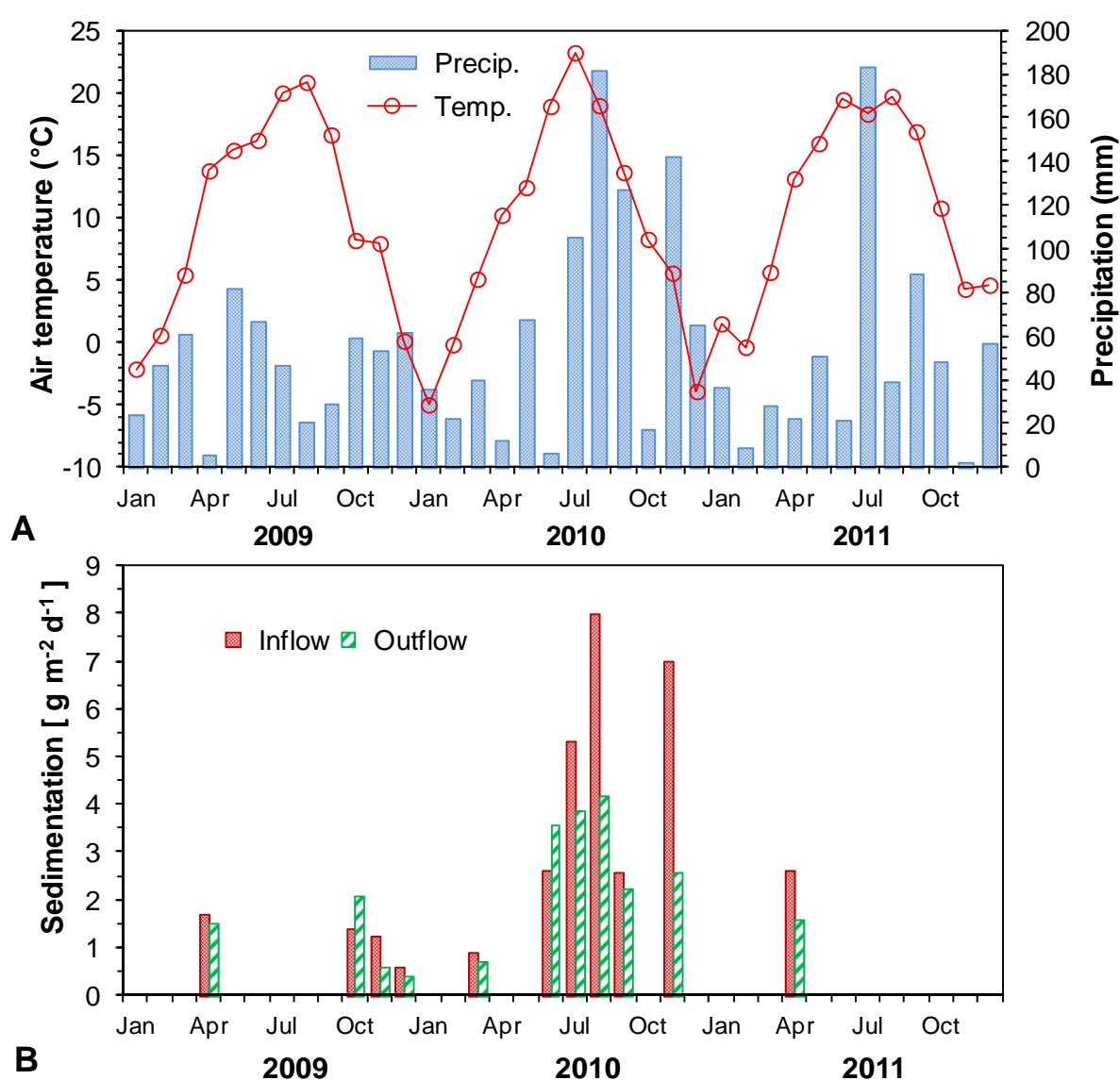

Fig. 8.2: A: Annual course of mean values of air temperature (Temp.) and of sum of monthly precipitation (Precip.); B: Course of sedimentation rate (dry weight) at the inflow and outflow region of the experimental pond 'Chicken Creek'; air temperature and precipitation were directly measured in the catchment.

A significant positive linear relationship could be found between air temperature and sedimentation rate of the inflow trap ($p = 0.05$, $R^2 = 0.70$, $n = 9$) and the outflow trap ($p = 0.05$, $R^2 = 0.68$, $n = 11$) indicating that most material settled during growing season, i.e., at higher water temperatures.

In concordance with the maxima of precipitation the sedimentation rate in the pond increased respectively, particularly in the inflow area. The linear relationship between precipitation and sedimentation rate was only significant for the inflow trap ($p = 0.05$, $R^2 = 0.65$, $n = 11$). It means that although a reed stand of more than 2 m width surrounds the pond, periods of higher rain intensity and heavy rain events, respectively, still cause an enhanced matter input from the terrestrial catchment area. However, it is reasonable to assume that in the near future this still expanding reed belt will retain more material than in the previous years.

8.3.3 Nutrient and metal mobilization during plant decomposition

SRP concentration in the control (pure sediment, $n = 6$) decreased from initially 0.13 mg L^{-1} during the first 8 days to 0.05 mg L^{-1}, and remained afterwards constant at 0.07 ± 0.03 mg L^{-1}, $n = 8$ occasions (Fig. 8.3). Particularly at the highest plant amendments ($Phau_4$, $Pona_4$, $Pope_4$) the SRP concentration increased within the first 20-30 days indicating a rapid initial leaching of plant constituents. In $Phau_4$, SRP increased from 0.13 to 2.96 mg L^{-1}, the absolute maximum during incubation. In $Phau_1$ to $Phau_3$, such a clear increasing trend was not visible. In contrast, a constant increase of SRP occurred in $Pona_1$ to $Pona_3$, whereas in $Pona_4$ SRP increased initially and decreased to the level of the other treatments. In $Pope$, SRP increased with increasing amendments, and most distinctly in $Pope_4$.

NH_4^+ concentration in the control remained ab initio low at 1.51 ± 0.43 mg L^{-1}, $n = 8$ (Fig. 8.4). Depending on the amount of OM added, NH_4^+ was considerably leached. $Phau_4$ reached a maximum of 135.9 mg L^{-1} after 61 days. In contrast, in $Pona_4$ and $Pope_4$ NH_4^+ concentration reached peaks after 12 and 20 days, and a minimum after 61 days, but increased again. In $Pona_2$, $Pona_3$, $Pope_2$, and $Pope_3$ NH_4^+ concentration increased constantly.

DFe concentration in the control decreased from initially 14.4 mg L^{-1} to 3.7 mg L^{-1} during the first 8 days, and was afterwards constant at 3.88 ± 0.87 mg L^{-1}, $n = 7$ (Fig. 8.5). In $Phau_1$, DFe increased from 14.4 to 94.4 mg L^{-1}. In $Phau_{2-4}$, DFe concentration steadily increased to maxima of 508.5 mg L^{-1} and 640.2 mg L^{-1} $Phau_4$ and $Phau_2$. In $Pona_1$ to $Pona_4$, DFe concentration increased from 14.4 mg L^{-1} to 34.1 mg L^{-1}, 108.9 mg L^{-1}, 83.8 mg L^{-1}, and 125.6 mg L^{-1} at day 124. In $Pope_1$ to $Pope_4$, DFe concentration increased from 14.4 mg L^{-1} to 26.8 mg L^{-1}, 49.9 mg L^{-1}, 112.5 mg L^{-1}, and 122.0 mg L^{-1} at day 124.

Fig. 8.3: Mean value (± *SD*) of soluble reactive P (SRP) concentration in pore water of sediment increasingly amended by *Phragmites australis* (*Phau*), *Potamogeton natans* (*Pona*), and *P. pectinatus* (*Pope*) material. (control Con with $n = 6$, and *Plants* 1 to 4 with $n = 2$).

Fig. 8.4: Mean value ($\pm$ *SD*) of ammonium (NH_4^+) concentration in pore water of sediment increasingly amended by *Phragmites australis* (*Phau*), *Potamogeton natans* (*Pona*), and *P. pectinatus* (*Pope*) material. (control Con with $n = 6$, and *Plants* 1-4 with $n = 2$).
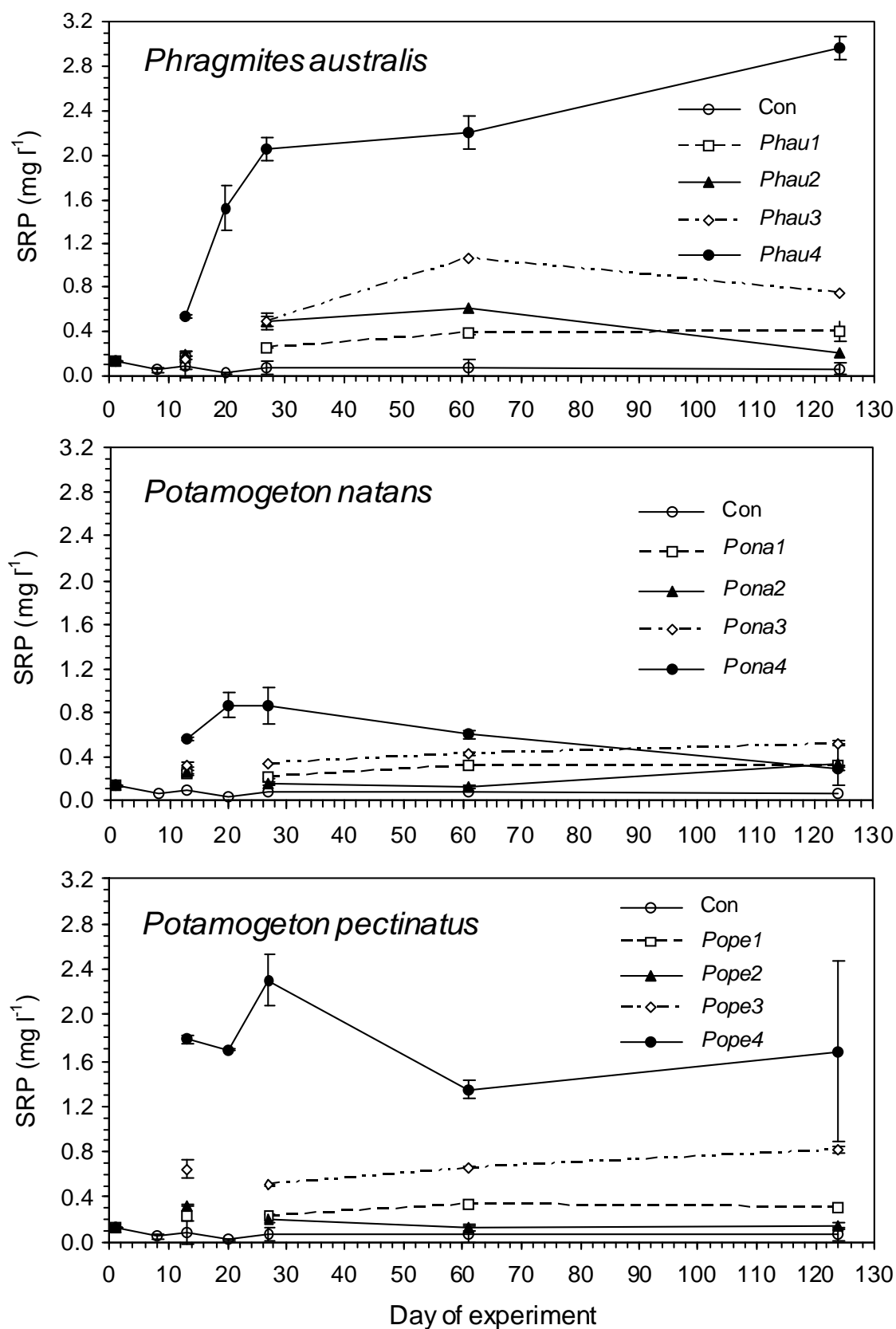
Fig. 8.5: Mean value (± *SD*) of dissolved Fe (DFe) concentration in pore water of sediment increasingly amended by *Phragmites australis* (*Phau*), *Potamogeton natans* (*Pona*), and *P. pectinatus* (*Pope*) material (control Con with *n* = 6, and *Plants* 1-4 with *n* = 2).
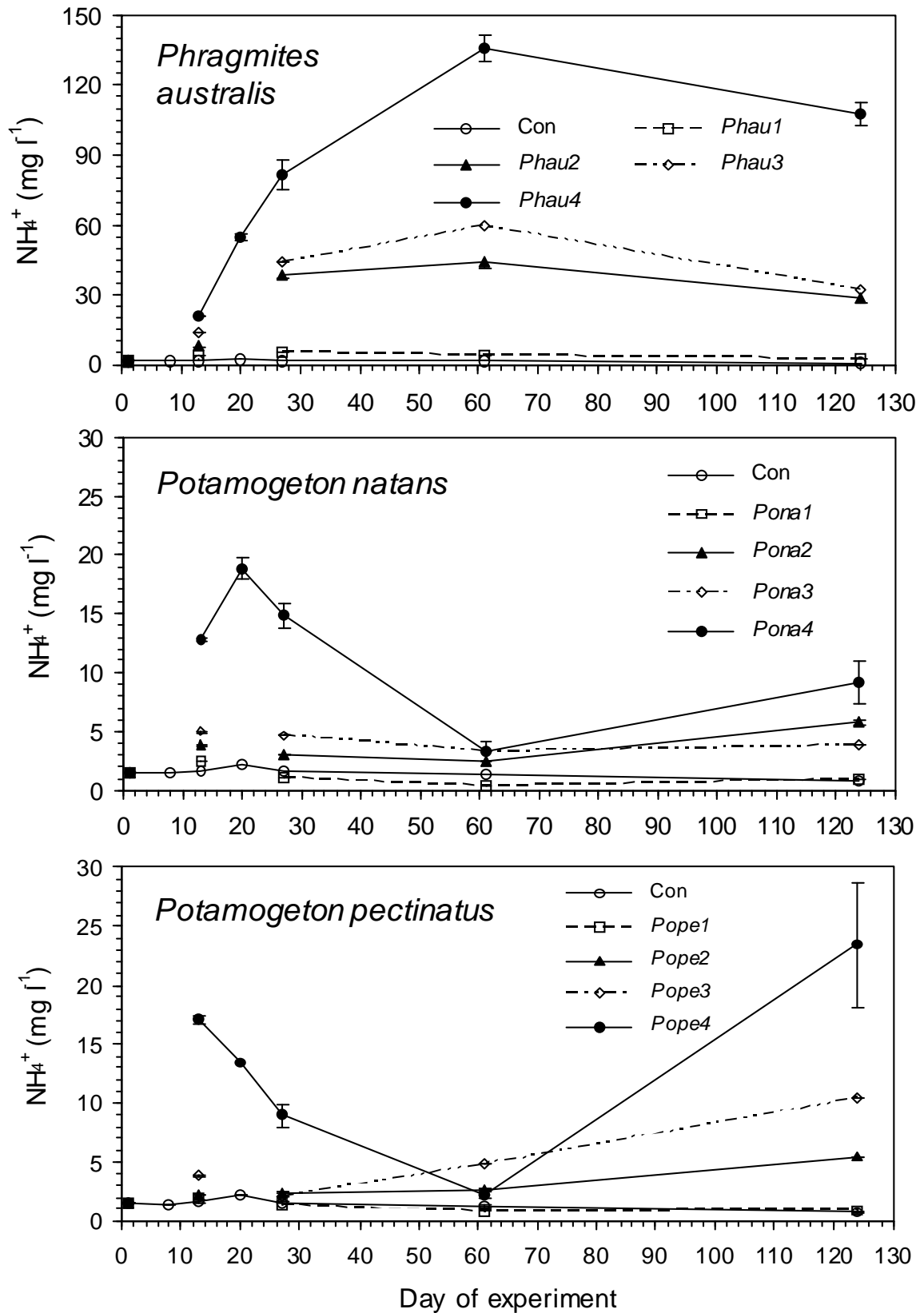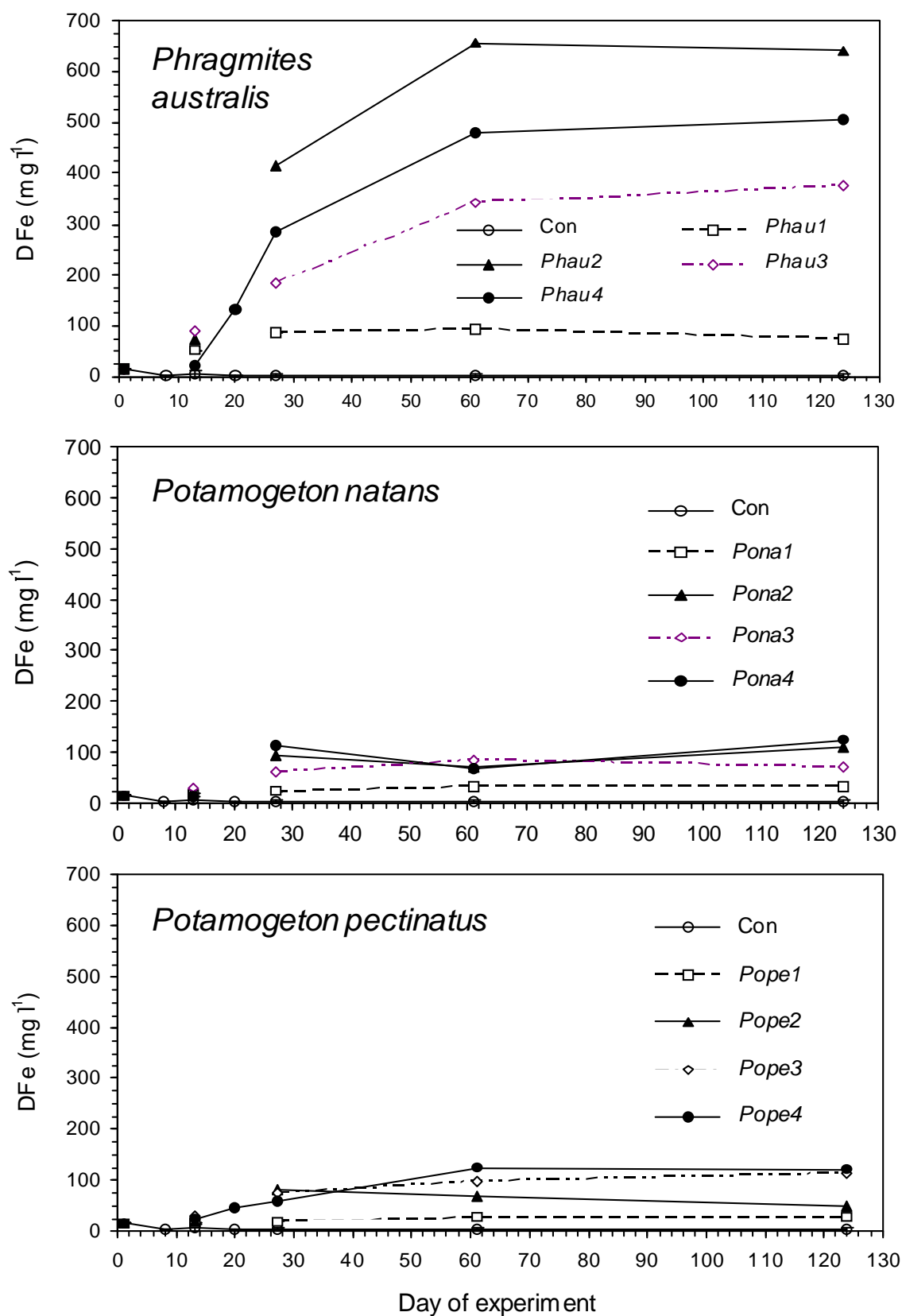
DMn concentration in the control decreased from initially 9.0 mg L^{-1} to 6.0 mg L^{-1}, during the first 8 days, and remained afterwards constant at 4.31 ± 0.76 mg L^{-1}, $n = 6$ (Fig. 8.6). In $Phau_1$ to $Phau_4$, there was an erratic increase in DMn concentration within the first 27 days, and except $Phau_1$ a further slighter increase thereafter to absolute maxima of 75.3 mg L^{-1} ($Phau_4$) and 95.8 mg L^{-1} ($Phau_2$). In $Pona_1$ to $Pona_4$, a similar trend became obvious, i.e., a rapid increase in DMn which peaked at day 27 with 38.3 mg L^{-1} ($Pona_4$). Afterwards DMn decreased in all cases. Also in $Pope_1$ to $Pope_4$ a rapid increase in DMn occurred which peaked at 64.3 mg L^{-1} ($Pope_4$). Afterwards DMn decreased ($Pope_2$ and $Pope_4$) or remained rather constant ($Pope_1$ and $Pope_3$).

Methodologically, our cup approach has various advantages: (i) it allows to completely balance the system, (ii) dialysis pore water sampling enables a small number of cups since sacrificing of cups omits, i.e., the present approach needs a lower number of cups, (iii) using the dialysis technique no filtration step is necessary and thus no oxygen input occurs into redox-sensitive elements. However, equilibrating of tubules needs ≥ 10 d time so that an intense sampling at the beginning of incubation is only possible to a limited extent.

At the beginning of incubation, the already low in situ SRP concentration decreased 3-fold obviously due to sediment oxidation, and Fe~P co-precipitation. NH_4^+ increased 9-fold due to adding the plant material (already leaching?), whereas higher metal concentrations remained unchanged. The rapid increase from the first five days onward is attributed to leaching – the 'washing out' of soluble constituents (Fig. 8.3 to 8.6). The further erratic increase was obviously due to conditioning, a modification of the leaf matrix by microorganisms as a result of enzymatic activities (e.g., Dinka et al., 2004; Shilla et al., 2006). However, it became obvious that because of sediment texture and the respective elemental inventory (Kleeberg et al., 2010b) a certain amount of plant material is necessary to notice any net effects of changes in dissolved pools, e.g., SRP of $Pona_3 > Pona_1 > Pona_2$ (Fig. 8.3) or DFe of $Phau_2 > Phau_4$ (Fig. 8.5).

In conjunction with plant decay the coincident release of polyphenolics could contribute to the decrease of DMn (Fig. 8.6). Polyphenolics, particularly tannins are known to precipitate proteins and complexing metals (e.g., Scalbert, 1991). Because of the coincidence of DFe and DMn (Fig. 8.5, 8.6) it is reasonable to assume that Mn is also involved in Fe^{2+} oxidation. In the upper sediment layer of the Mozhaisk reservoir, Russia, the $Fe^{2+} : Fe^{3+}$ ratio was directly (nonlinearly) related to mobile Mn concentration suggesting its involvement in Fe^{2+} oxidation (Martynova, 2010).

Figures 8.3 to 8.6 reveal that although only small proportions of plant solid phase are mobilized within 124 days of incubation with increasing amendments of macrophytic material (Fig. 8.7) the pore water concentration may considerably rise having a distinct influence on nutrient turnover if compared to the initial sediment.
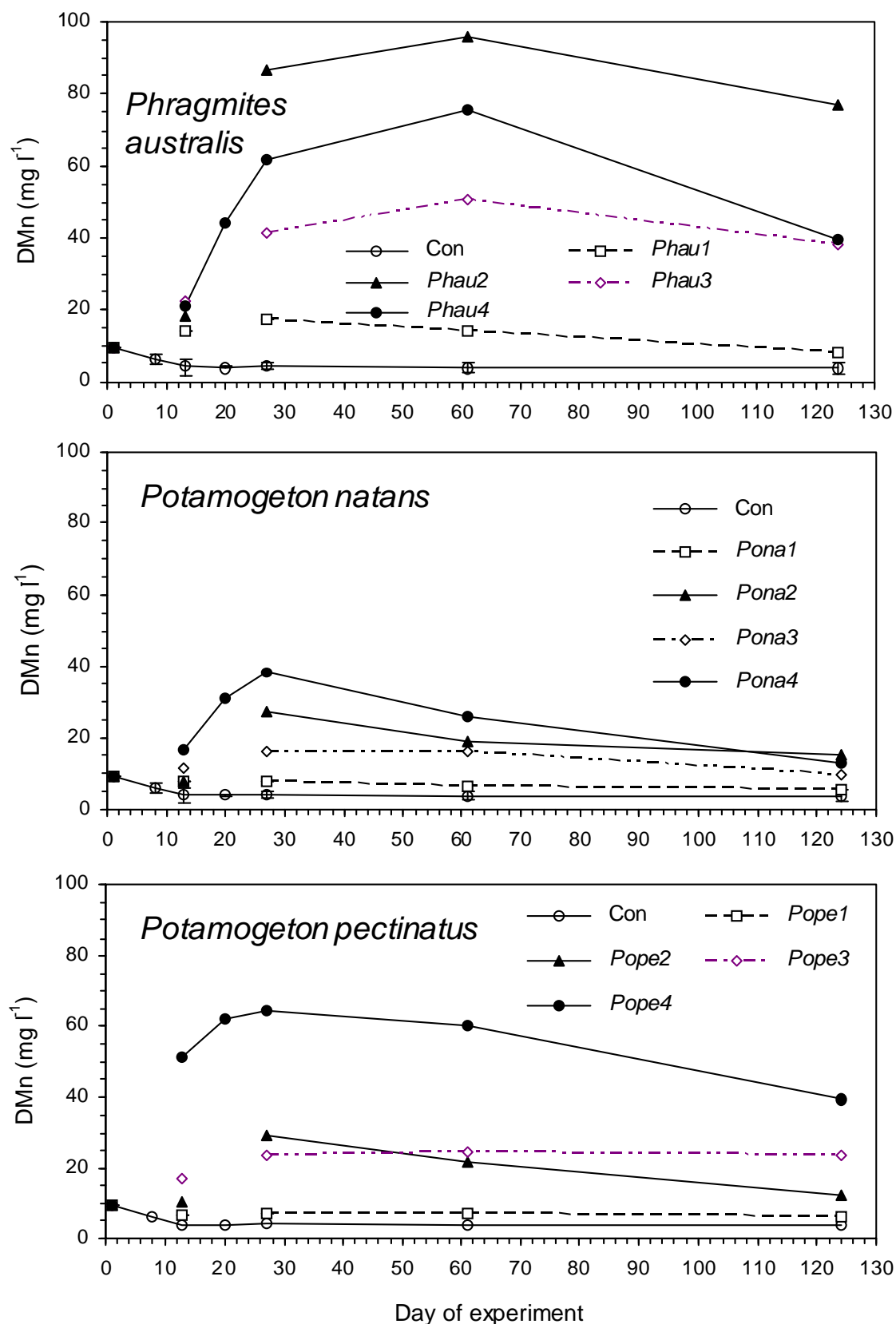
Fig. 8.6:   Mean value (± *SD*) of dissolved Mn (DMn) concentration in pore water of sediment increasingly amended by *Phragmites australis* (*Phau*), *Potamogeton natans* (*Pona*), and *P. pectinatus* (*Pope*) material (control Con with *n* = 6, and *Plants* 1-4 with *n* = 2).

Fig. 8.7: Percentage of dissolved nutrients (NH_4^+, SRP) and metals (DFe, DMn) released from the initial solid phase (TN: total nitrogen, TP: total phosphorus, TFe: total iron, TMn: tota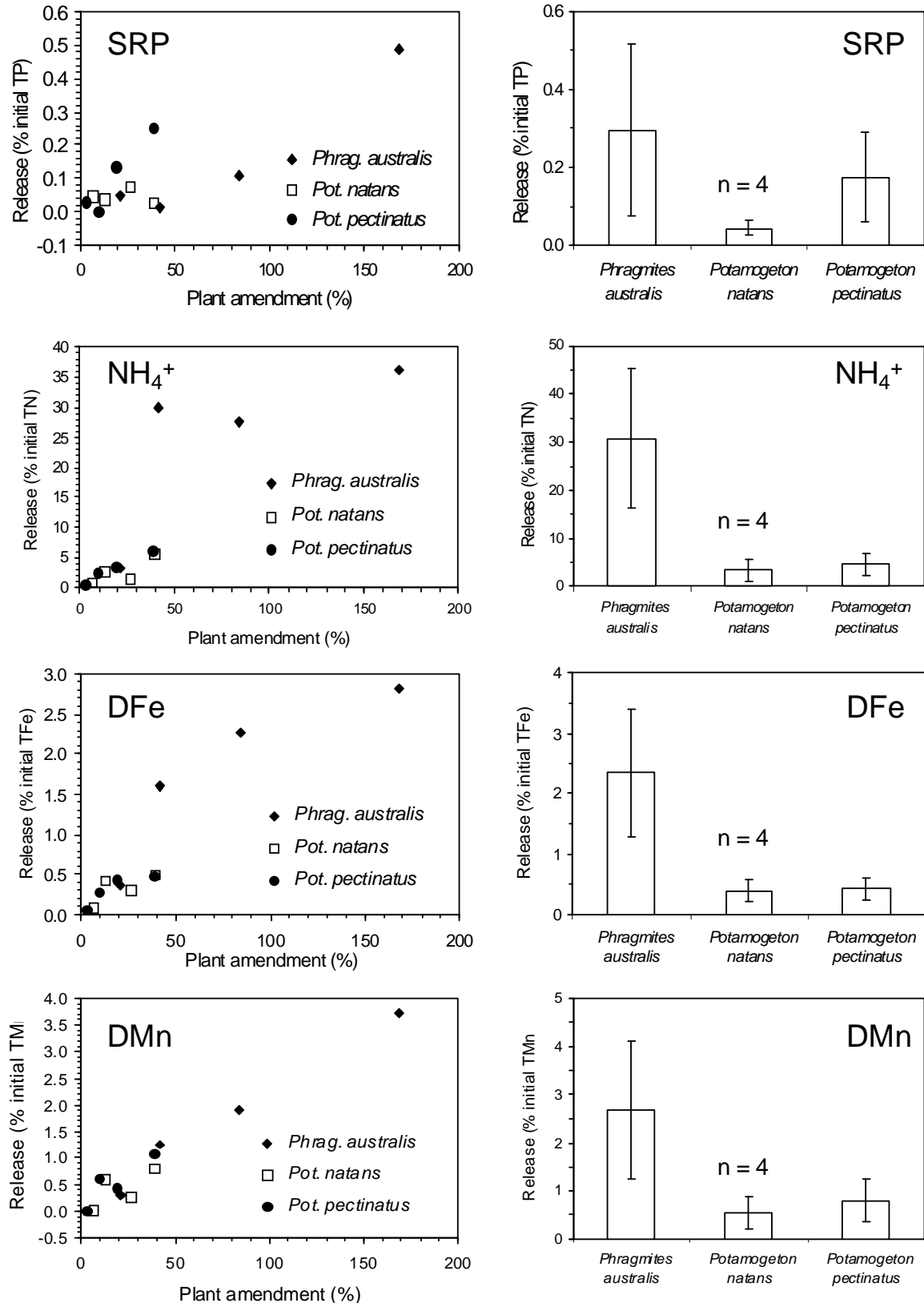l manganese) into pore water of sediment slurry increasingly amended by *Phragmites australis*, *Potamogeton natans*, and *P. pectinatus* (125 days incubation).

Plant decomposition-mediated regeneration of nutrients and metals (Fig. 8.3 to 8.6) is a function of sediment binding capacity and plant elemental content where plant species differently contributed on a short-term mobilization of SRP and NH_4^+ (Fig. 8.3, 8.4) as well as DFe and DMn (Fig. 8.5, 8.6). However, based on plant mass added, there were no differences among the species in the extent of enrichment of pore water SRP (51- to 60-fold) and DMn (19- to 26-fold). Whereas there was a clear sequence for NH_4^+ and DFe: *Phau > Pona > Pope* in a span of 86- to 25-fold and 239- to 85-fold. In all cases the extent of Fe mobilization exceeded that of P as indicated by an increasing Fe:P ratio. Regardless of the origin of solid Fe, it suggests a tendency towards mobile redox-sensitive Fe. Consequently, at an oxidized sediment surface mobilized P can subsequently co-precipitated and trapped or taken up again by growing macrophytes. However, a further acceleration of macrophyte growth and OM accumulation could lower redox potential, stimulating both anaerobic reduction of sulphate, at an already elevated level (Kleeberg et al., 2010b), and the formation of iron sulfides (FeS_x) (e.g., Roden & Edmonds, 1997). During anoxic conditions under ice, considerably increasing concentrations of nutrients and metals in the water column of the pond were detected as net effects of plant decomposition at the sediment surface. For example, on 9th March 2010, at 0.1 m depth SRP and NH_4^+ concentrations were still below detection limit (< 0.003 mg L^{-1} and 0.04 mg L^{-1}), but were 0.378 mg L^{-1} and 1.23 mg L^{-1} at 2 m depth. DFe and DMn concentrations were < 0.1 mg^{-1} L^{-1} and 0.64 mg L^{-1} at 0.1 m depth, whereas 0.1 mg L^{-1} and 5.8 mg L^{-1} at 2 m depth.

8.4   Conclusions

Only long-lasting rainy periods or heavy rainfall events contribute to an enhanced material input as revealed by the annual sediment development in the inflow area of the pond. The extent of surface soil erosion at the previously vegetation-free catchment obviously decreased as reflected by a decreasing sediment accumulation. Nevertheless, the sediment is still an organic-poor freshwater sediment indicating an immature succession state of the pond. However, macrophytes can accelerate the transition from a rather less geochemically reactive to a more biogeochemically active pond sediment (a certain priming effect) and essentially enhance the mobilization and translocation of phosphorus to other biota. At present, the relative iron surplus determines the equilibrium between release and binding of phosphorus after plant decomposition shifts towards redox-sensitive phosphorus binding indicating a progressive phosphorus mobilization and availability in the 'Chicken Creek' pond. A mass balance of the whole pond vegetation proportionally considering standing stock of all plant species, i.e., their potential to mobilize, can provide the necessary information on their contribution to intensification of pond-internal nutrient cycling that is crucial in initial ecosystem development.

Acknowledgements

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9.   Conclusions

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The artificial catchment 'Chicken Creek' was constructed to analyze initial structures and related processes of ecosystem development under well-defined boundary conditions to better understand the behavior of more complex mature systems. Volume 1 of this series (Gerwin et al., 2010) gave detailed insight into the construction process and initial properties of the 'Chicken Creek' catchment. A first comprehensive report on the monitoring program, installations, and methods is given in the second volume of this series (Schaaf et al., 2010). This volume summarizes the monitoring results for the period 2005 to 2010.

9.1   Hydro-geomorphic development

The very first phase of ecosystem development is characterized by a still more or less abiotic system controlled by initially existing structures and external drivers (Fig. 9.1, right). The most relevant structures in the very first period of development were slope and exposition of the surface, initial morphology, and textural properties of the substrates. These structures are regarded as fundamental initial conditions resulting from the construction process.

Fig. 9.1:   'Chicken Creek' catchment in August 2006 (right) and July 2010 (left).

During the initial years of 'Chicken Creek' catchment development, surface runoff processes, e.g., after episodic precipitation or snow melt events, dominated catchment hydrology inducing severe sheet and gully erosion. Hence, the increasing water level of the pond was dominated by surface runoff. As a consequence, the geo-system was considerably altered by geomorphic processes, e.g., sediment relocation (cf. chapter 2). The hydro-geomorphic development was characterized by vertical rill incision, growth of the rill network, and a high dependence on abiotic initial conditions. The emergence of erosional and depositional structures resulted in rapid surface differentiation. After the first phase of rapid growth and vertical incision, evolution of the erosion rill network was later found to be predominantly characterized by lateral erosion. Finally, the areal extent of actively eroding rills decreased, mainly due to the establishment of plants in formerly more instable regions (cf. chapter 5).

Results from wind erosion modeling and aerial image analysis suggest that aeolian sediment relocation was about two magnitudes lower than that caused by water erosion. The soon starting and further proceeding consolidation of surfaces, e.g, by crusts and vegetation (cf. chapter 5), will probably completely inhibit wind erosion processes at the catchment, while aeolian sediment import from the surrounding mining landscape could remain relevant well beyond this very first phase.

Besides geomorphic processes, which rapidly altered the structure of the juvenile catchment, hydrologic processes like surface runoff and the primary groundwater recharge defined the first stage of ecosystem evolution (cf. chapter 3). However, only little interactions between groundwater and the hydrological conditions at the surface were observed in the first years of ecosystem development. With rising groundwater levels, however, the main erosion gullies developed to locally perennial streams draining the groundwater in the footslope.

The filling of the substrate pore volume and the establishment of a groundwater body needed the time span of several years. This period is defined by a constantly increasing trend of groundwater levels on top of the aquiclude to very high levels at the end of the observation period. Exceptional high amounts of rainfall in 2010 led to an almost complete saturation of the aquifer. Surface runoff together with basic flow in gullies fed from groundwater discharge resulted in high discharge rates as reflected in a respective accumulation of eroded matter in the pond (cf. chapters 7, 8; Kleeberg et al., 2010). Overall, the external impacts described above exceeded trends caused by the development of the catchment itself. For example, a trend of decrease in total discharge due to increased transpiration by the increase of vegetation biomass (cf. chapter 5) was not observed during the study period.

Due to the importance of the hydrological development of the system much attention was paid to a comprehensive monitoring of hydrological components. The well determined water balance components in this monitoring concept are precipitation, the storage of water in the saturated zone, and total discharge. This is not true for the actual evapotranspiration and the

storage in the unsaturated zone, for which an exact quantification is difficult. For the future, additional measurements to calculate values for actual evapotranspiration are needed to decrease the uncertainty in the water balance of the catchment.

Weathering and soil formation processes led to considerable alterations in the water chemistry of the catchment (cf. chapter 4) and in the pond (cf. chapter 7). The monitoring program took account of these expected changes with the installation of adequate measuring devices at relevant structures. The element budgets were strongly influenced by both changes in water chemistry and in discharge rates mentioned above. Whereas the inputs to the catchment via bulk deposition did not vary too much over the years, the output from the catchment showed strong and continuous increases for most elements. This was mainly governed by the strong increases in discharge, especially in the very wet year 2010. The catchment was a strong source for calcium, magnesium, sulfur and both organic and inorganic carbon. Only for nitrogen, the catchment acted as a strong sink, typical for strongly nitrogen-limited systems. With regard to water chemistry of the weir and flume samples, concentrations decreased significantly, mainly for calcium, magnesium and sulfate, whereas bicarbonate increased. The overall concentrations at the flume and the upper weir were much higher compared to soil solutions, concentrations at the lower weir, and pond water.

Soil solution composition showed considerable changes over the investigation period (cf. chapter 4). The large spatial variation in the first year of sampling indicates non-equilibrium conditions. With time, concentrations corresponded well to water chemistry and showed similar temporal trends for calcium, magnesium and sulfate. The spatial variation strongly decreased over the investigation period, probably an indication for more equilibrium controlled conditions. Dissolution of gypsum of the parent material seems to be a dominating process controlling initial soil solution composition. Since the gypsum contents are very low, decreasing sulfate concentrations in both soil water and pond water (cf. chapter 7) indicate that most of the gypsum has been dissolved and mobilized within a few years. The rapid increase in sulphate concentration will have consequences on the pond's sediment phosphorus mobility enhancing its availability for various biota (Kleeberg et al., 2010). With increasing vegetation cover (cf. chapter 5) and litter input into the soil, carbonate weathering is increased and represents the main control for calcium concentrations in soil solution.

9.2 Establishment of populations

The development of the hydro-geomorphic system significantly affected the establishment of plant and animal populations. For both vegetation and soil fauna species numbers and biomasses increased substantially during the first years. The initial spatial differentiation of substrate characteristics led to an early patterning of species composition and therefore to early colonization patterns.

The total number of vascular plant species increased quite continuously and nitrogen-fixing plant species became a major component of the establishing vegetation (cf. chapter 5). Species immigration by wind was crucial and led to a quite uniform dominance of *Conyza canadensis* in the first years (Fig. 9.1, right). In 2009, legumes showed maximum cover throughout the catchment that was significantly higher than the sum of all other vascular plants. In 2010, however, a general decline in legume cover was observed due to a marked reduction of *Trifolium arvense*. Thereby the spatial patterns changed with regard to legume cover: A significant increase in the eastern part was accompanied by a reduction in the originally more favoured western part. The extensive decline of particular plant species or the increasing dominance of other species, as exemplified by the development of *T. arvense*, may enable the delimitation of succession phases as subunits of progressive succession. In this regard, the significant cover decline of a single plant species can be regarded as a characteristic feature of early successional stages. The results indicate a low resilience during the observed years.

In water-limited regions of the world, a stable vegetation pattern formation is observed in order of increasing precipitation amounts: bare soil < spots < bands < vegetation with holes < uniform vegetation. These patterns are mainly induced by feedback mechanisms between plants and water availability (cf. chapter 3). In the 'Chicken Creek' catchment, we observed all types of vegetation patterns listed above. However, these patterns were not stable, transitions occurred rapidly, and pattern emergence appeared to be mainly species-driven. In disfavoured areas, slower colonisation, transition, and a more distinct pattern formation was observed. We expect that these small-scale patterns will disappear in further development.

One month after construction in autumn 2005, the initial substrates were already colonised by first representatives of the soil food web (cf. chapter 6). Specimens of those early colonising species found adequate conditions and nutrient resources sufficiently available to survive in the newly formed substrate, and from there some pioneer species spread almost immediately into individual-rich populations. However, overall species numbers were still relatively low in the fifth year of catchment development and many more species are expected to come in the future. None of the soil faunal groups under investigation had yet reached its usual species numbers and densities, except for Tardigrada, which already passed their peak in 2009.

During the first two years of primary succession, mainly algal and bacterial feeders lived in the 'Chicken Creek' substrate. This indicates that microbial and soil-algal communities were present and available from the very beginning in the newly formed soils. The number of trophic links within the soil food web increased as of 2007. This concurs with the vegetation becoming more diverse and the plant cover rising, thus providing an increasing supply of nutrient resources and niches. In accord with the distinct increase of plant biomass in 2009, a significant reduction of primary colonizers was found for carabid beetles. At the same time, root-feeding nematodes and testate amoebae were positively affected by the above-ground

succession. Hence, the soil food webs clearly developed in association with the plant community, which is typical for the initial phase of succession. Densities and species richness of soil-inhabiting carnivores, on the other hand, remained extremely low throughout the first five years and established populations were not found until 2009.

Hence, soil food webs first develop from basal trophic levels and then increase in complexity during primary succession. All major trophic groups may colonise new substrates within a few years, but species inventories of these early successional trophic groups consist of comparable few pioneer species. It will take several decades before the soil food web reaches a complexity and species richness expected from mature soils. Overall, with regard to above- and below-ground colonization, the system is still immature and fluctuating after five years of development. However, external events causing retrogression are expected to become less important in the future, while resilience, stability, and predictability are supposed to increase. Larger animals and plants (e.g., earthworms, trees) will establish and become more dominant.

9.3   Pond succession

The 'Chicken Creek' catchment offers the unique chance to analyze the initial development and intensification of matter cycling and habitat structuring in the experimental pond in dependence of the concurrently proceeding development of the catchment. Monitoring results illustrate the close dependencies between the catchment and the pond in terms of water chemistry, nutrient regime and sediment accumulation. The terrestrial hydro-geomorphic and biotic development is a key factor for the velocity of primary pond succession, as it controls the input of sediment, major ions, nutrients, and organic carbon. 'Chicken Creek' pond so far underwent two phases of succession: The first phase (2005-2007) (Fig. 9.1, right) was dominated by major morphological changes due to high sediment input from the terrestrial area (cf. chapter 2), which reduced depth and volume and increased turbidity. Longer lasting rainy periods and episodic precipitation events contributed to an enhanced material input as revealed by the annual sedimentation rate in the inflow area of the pond (cf. chapter 8). In the second phase (2008-2010) (Fig. 9.1, left), the increasing plant biomass in both terrestrial and aquatic areas, particularly in the riparian reed zone of the inflow area of the pond, led to considerably decreasing sediment accumulation in the pond and enhanced growth of submerged macrophytes.

Phytoplankton reached a high species richness within a few years (cf. chapter 7). However, the high share of taxa that occurred only in single years and the low number of steady taxa indicates a low stability of the phytoplankton community. Nevertheless, the high diversity documents that a spatially isolated water body can be colonized by a wide variety of algae within a few years. This guarantees a high efficiency in the use of resources also in immature ecosystems. With regard to primary production, the first phase was dominated by pelagic

algae with a control of algae biomass by phosphorus and light. Ponds with their shallow water and only slight motion of water are ideal habitats for macrophytes. Therefore, it took only a short period until submerged macrophyte biomass exceeded phytoplankton biomass. In 2010, a sharp increase in phytoplankton biomass occurred, one year after the distinct rise in terrestrial plant biomass (cf. chapter 5).

These changes correspond corresponds to an increasing extraction of nutrients from the surface sediments by thriving plants and from water column. Availability of phosphorus for primary producers decreased: While concentrations reached the upper end of the mesotrophic state in 2005 and 2006, they were much lower in the period 2008 to 2010, which corresponds to the increased submerged macrophyte growth. The chemical composition of pond water is determined by inflowing water quality and by internal processes that occur predominantly at the sediment-water interface. Water data of 'Chicken Creek' pond showed a continuous change in anion proportions: a steady decrease in sulphate and increase in bicarbonate, in concordance with the result for soil solution and flume water composition (cf. chapter 4).

After five years of pond development, the sediment is still organic-poor indicating an immature succession state. However, the observed acceleration of macrophyte establishment will contribute to the transition from a rather less geochemically reactive to a more biogeochemically active pond sediment and essentially enhance the mobilization and translocation of phosphorus to other biota. At a current relative iron surplus the equilibrium between release and binding of phosphorus after plant decomposition shifts towards redox-sensitive phosphorus binding indicating a progressive phosphorus mobilization and availability in 'Chicken Creek' pond.

Overall, a significant influence of the terrestrial succession on pond development could be detected. However, internal processes gained increasing importance. If the external influence stays stable and shifts occur only slowly, further pond development is expected to be regulated mainly by internal processes. Larger changes of the terrestrial compartment, however, could act as disturbance and cause retrogression.

Further studies will provide insights into the initiation of matter fluxes at both terrestrial and aquatic interfaces, especially concerning the source function of various substrates for nutrients. Thus, the spatial and temporal dynamics of processes like turnover rates in the soil and the pond sediment will be investigated simultaneously. Finally, future research will provide empirical and conceptual knowledge on the changes in structure-function relationships of a pond, as the results allow for a transfer on a significant number of small inland water bodies most vulnerable to global change phenomena.

9.4   Synthesis

Ecological succession is a series of transitions in the development of ecosystems, which are linked to changes in environmental conditions and affect biological colonization. These changes may be caused by changing external factors and/or may be controlled by internal processes (Begon et al., 2005). During the first five years we observed considerable changes within the 'Chicken Creek' catchment. Both internal and external factors could be identified as driving forces for the formation of structures and patterns.

Internal structures formed by the construction process and initial substrate characteristics were decisive for distribution and flow of precipitation water as well as for biotic succession. External factors like episodic precipitation events triggered erosion and dissection of the surface during this first phase, promoted by the low vegetation cover and the unconsolidated sandy substrate. These processes resulted in transport and redistribution of water and sediment within the catchment and the formation of new structural elements. Invasion of flora and fauna was another important external process influencing colonization patterns. As a result, we observed an overall differentiation of the study area into parts with abrasion or accumulation processes dominating and parts with stable surfaces. During further development, both external factors and processes within the catchment continued to influence the site. The dissection and stability of surfaces was an important factor for the formation of biological soil crusts, vegetation patterns, soil food webs, and pond biocoenoses.

After five years, several parts of the catchment still remained sparsely colonized by animals and plants. Initially established structures like soil crusts obviously influenced vegetation patterns by altering soil surface properties, and by promoting surface runoff and erosion. Redistribution and sedimentation of transport material, both in terrestrial and aquatic parts of the catchment, seemed to be important for plant and animal colonization. Vegetation establishment preferentially started along the linear structures of erosion channels, probably due to better growing conditions with respect to water and nutrient availability. The transformation of the initial geo-system into areas with evolving terrestrial or aquatic ecosystem characteristics and from a very episodic to a more permanent stream network and discharge, together with the observed biotic dynamics increased site diversity and heterogeneity with respect to water and nutrient availability and transformation processes.

Odum (1969) proposed trends of ecological processes to be expected in the development of ecosystems. These trends can be applied to the model of ecosystem development suggested by Fath et al. (2004): The distinct increase in the amount, number, and size of biota in the catchment corresponds to 'structural growth'; this holds true for all presented groups, terrestrial and aquatic as well as plants and animals, and especially since 2009. On the other hand, indications for low resilience and stability in different compartments of the system, for low below-ground species diversity, and for the minor role of feedback mechanisms clearly

support the assessment that the 'Chicken Creek' catchment is still in a very early, initial state. However, the decrease in spatial variation of the soil solution, the increasing complexity of food webs, and the rising importance of larger plants (i.e., trees, macrophytes) reflect slight network and information growth according to Fath et al. (2004), indicating a trend of increasing system organization and efficiency.

For the future, we expect (i) a short-term quantitative increase in ecosystem biomass, capturing more of the incoming solar energy, (ii) further growth of connectivity both between and within the compartments raising the internal organization in the mid-term, and (iii) a delayed qualitative growth to more conservative, energetically efficient patterns. Finally, we suppose that feedback mechanisms (e.g., primary producers and groundwater development) will intensify with the increasing number of effective structures within the system in the near future. Therefore, investigations of these interactions between different compartments of the 'Chicken Creek' catchment need to be intensified and adapted to the system transitions.

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10. Zusammenfassung

Das künstliche Einzugsgebiet Hühnerwasser wurde 2004 und 2005 in enger Kooperation von Brandenburgischer Technischer Universität Cottbus (BTU) und der Vattenfall Europe Mining AG (VEM) im Tagebau Welzow-Süd erbaut. Wissenschaftliches Ziel ist die Untersuchung initialer Strukturen und der mit ihnen verbundenen Prozesse der Ökosystementwicklung unter klar definierten Rahmenbedingungen. Dies soll dazu beitragen, die Zusammenhänge auch in reiferen Systemen besser verstehen zu können. Band 1 dieser Schriftenreihe (Gerwin et al., 2010) stellte den Entstehungsprozess und die Ausgangsbedingungen des Einzugsgebietes im Herbst 2005 vor. Zu diesem Zeitpunkt startete die kontinuierliche Erfassung meteorologischer und hydrologischer wie auch bodenkundlicher und geomorphologischer Parameter. Zusätzlich wurden die ober- und unterirdische Besiedlung durch Flora und Fauna sowie die Entwicklung in Wasserkörper und Sediment des Teichs im Gebiet untersucht. Band 2 der Schriftenreihe stellt die Aktivitäten dieses Umweltmonitorings, die angewandten Methoden und erste Ergebnisse übergreifend vor (Schaaf et al., 2010). Dieser dritte Band präsentiert die Entwicklung des Einzugsgebietes Hühnerwasser für die Jahre 2005 bis 2010, bewertet sie für die unterschiedlichen Kompartimente und ordnet sie durch den Vergleich mit reiferen Systemen ein.

10.1 Geo-Hydro-System

Die ersten Jahre der Entwicklung waren durch ein mehr oder weniger abiotisches System gekennzeichnet, das durch die initialen Eigenschaften und externe Faktoren gesteuert wurde. Die wichtigsten Strukturen waren dabei die Hangneigung und die Exposition der Oberfläche, die initiale Morphologie und die Textur der Substrate. Diese Strukturen werden als grundlegende Startbedingungen angesehen, die aus dem Bau des Gebietes resultierten.

Die hydrologische Entwicklung des Einzugsgebietes wurde zu Beginn vor allem durch Oberflächenabflüsse und ereignisbedingte, starke Erosion dominiert, z.B. nach episodischen Niederschlagsmengen oder der Schneeschmelze (s. Kapitel 2). Diese Ereignisse veränderten das Geosystem deutlich und verursachten starke Schwankungen des Teichpegels. Durch Erosion und Sedimentablagerung erfolgte eine Ausdifferenzierung der terrestrischen Oberfläche. Die hydro-geomorphologische Entwicklung zeichnete sich im Folgenden durch eine Vertiefung der Rinnen sowie die Ausbildung des Gerinne-Netzwerkes aus und zeigte insgesamt den großen Einfluss der Ausgangsbedingungen. Später nahm die Bedeutung lateraler Erosion deutlich zu und der Flächenanteil aktiver Rinnen nahm ab, vor allem aufgrund der Festlegung ehemals instabiler Bereiche durch die Vegetation. Winderosion war von signifikant geringerer Bedeutung und wird in Zukunft noch weiter zurückgehen, wohingegen der äolische Staubeintrag in das Gebiet weiterhin von Relevanz sein dürfte.

Die initiale hydrologische Ausprägung des Gebietes wurde demnach vor allem durch externe Faktoren beeinflusst, weniger durch endogene Prozesse im Einzugsgebiet selbst (s. Kapitel 3). Die Auffüllung des Porenvolumens im Substrat und der damit verbundene primäre Grundwasseranstieg bis in oberflächennahe Bereiche dauerte mehrere Jahre. Schließlich entwickelten sich Teile der Hauptrinnen zu ausdauernden Bächen, die auch den Grundwasserkörper entwässerten. Im Beobachtungszeitraum konnten jedoch nur geringe Interaktionen zwischen dem Grundwasser und den hydrologischen Bedingungen an der Oberfläche beobachtet werden (z.B. Einfluss der Vegetation über die Verdunstung). Für die Bilanzierung des Wasserhaushaltes konnten die Niederschläge, die Speicherung in der gesättigten Zone sowie der Gesamtabfluss aus dem Gebiet gut quantifiziert werden. Die Untersuchungen zur Speicherung im ungesättigten Substrat und zur realen Verdunstung sollten in Zukunft intensiviert werden, um die Bilanzierung weiter zu verbessern.

Prozesse der Bodengenese führten zu Veränderungen der Wasserqualität von Einzugsgebiet (s. Kapitel 4) und Teich. In der Bodenlösung wurden zu Beginn große räumliche und zeitliche Unterschiede im Gebiet festgestellt. Die Angleichung über die vorgestellte Periode deutet auf zunehmende Gleichgewichtsbedingungen hin. Die niedrigen Calciumsulfat-Gehalte und die abnehmenden Sulfat-Konzentrationen im Boden- und Teichwasser weisen auf eine weitgehende Lösung von offenbar im Substrat vorhandenem Gips in den ersten Jahren hin. Für die Zukunft ist durch die vermehrte Pflanzenbiomasse verbunden mit größerem Streueintrag eine vermehrte Carbonatverwitterung zu erwarten. In den Abflüssen sanken die Konzentrationen von Calcium, Magnesium und Schwefel deutlich, wohingegen die Carbonatgehalte anstiegen. Die Gesamtkonzentrationen in den Teichzuflüssen waren deutlich höher als in der Bodenlösung, dem Wasserkörper sowie dem Abfluss aus dem Teich. Der Stoffhaushalt wurde wesentlich durch diese Veränderungen und durch den Gesamtabfluss reguliert: Während der Eintrag die Luftdeposition relativ konstant war, stieg der Austrag im Laufe der Jahr für die meisten Elemente deutlich an – vor allem aufgrund des deutlich gestiegenen Gesamtabflusses in 2010. Das Einzugsgebiet diente als Quelle für Calcium, Magnesium, Schwefel sowie organischen und anorganischen Kohlenstoff, für Stickstoff dagegen als Senke. Letzteres ist typisch für junge Stickstoff-limitierte Systeme.

10.2   Besiedlung

Die Charakteristika und die Entwicklung des hydro-geomorphologischen Systems beeinflussten die Besiedlung des Einzugsgebietes durch Pflanzen und Tiere in den ersten Jahren der Entwicklung signifikant. Die initialen Unterschiede in den Substrateigenschaften führten zu einer Ausdifferenzierung der Artenzusammensetzung und damit zu frühen Besiedlungsmustern. Darüber hinaus erhöhten sich die Artenzahlen und die Biomassen beider Gruppen im beobachteten Zeitraum deutlich.

Die Zahl höherer Pflanzenarten stieg nahezu kontinuierlich an (s. Kapitel 5). In den ersten Jahren war die Einwanderung durch den Wind von großer Bedeutung und führte zu einer Dominanz des Kanadischen Berufkrautes (*Conyza canadensis*). Stickstoff-fixierende Leguminosen wurden ein wesentlicher Bestandteil der sich ausbildenden Vegetation. Im Jahr 2009 wiesen sie den bisher größten Anteil auf, der über dem aller anderen Pflanzen lag. Durch den Rückgang des Hasenklees (*Trifolium arvense*) als der Art mit der größten Deckung in 2010 ging die Bedeutung der Leguminosen jedoch wieder leicht zurück – vor allem im westlichen Teil des Einzugsgebietes. Dies kann auf eine neue Phase der Entwicklung der Gemeinschaften hindeuten und zeigt die geringe Stabilität des Systems in der beobachteten Periode. Im Gebiet wurden alle kleinräumigen Vegetationsmuster beobachtet, die aus Wasser-limitierten Regionen der Erde bekannt sind. Hier waren sie jedoch nicht stabil und die Übergänge vollzogen sich in kurzen Zeiträumen. Deshalb ist zu erwarten, dass diese Muster im Einzugsgebiet nur von kurzer Dauer sein werden.

Bereits im Herbst 2005 waren die initialen Substrate durch erste Vertreter der Bodenfauna besiedelt und diese Pionierarten konnten zum Teil große Siedlungsdichten entwickeln (s. Kapitel 6). Allerdings waren die Artenzahlen auch nach fünf Jahren noch relativ gering: Bis auf die Tardigraden hatte keine Gruppe in diesem Zeitraum ihre üblichen Artenzahlen und Siedlungsdichten erreicht. Die Dominanz von Algen- und Bakterienfressern in den ersten beiden Jahren deutet auf entsprechende Nahrungsgrundlagen von Beginn an hin. In der Folge (ab 2007) stiegen die trophischen Verknüpfungen parallel zur zunehmenden Vegetation und mit dem deutlichen Anstieg der Pflanzenbiomasse in 2009 sank die Bedeutung der epigäischen Pionierarten. Gleichzeitig profitierten wurzelfressende Nematoden und Amöben von der beschriebenen Entwicklung. Artenzahlen und Dichten räuberischer Vertreter der Bodenfauna blieben dagegen im gesamten Untersuchungszeitraum äußerst niedrig.

Demnach ist nicht damit zu rechnen, dass die ober- und unterirdischen Nahrungsnetze kurzfristig eine Komplexität und Artenzahlen aufweisen, wie sie von reiferen Systemen zu erwarten sind. Weiterhin ist mit bedeutenden Interaktionen zwischen Vegetation und Bodenfauna zu rechnen, typisch für die Entwicklung von jungen, instabilen Ökosystemen. Deshalb werden diese Wechselwirkungen wie auch die Interaktionen mit den abiotischen Standortsfaktoren im Fokus zukünftiger Untersuchungen stehen.

10.3 Teichentwicklung

Die terrestrische Entwicklung des Einzugsgebietes ist ein Schlüsselfaktor für die Entwicklung des Teichs im Gebiet, weil sie den Eintrag von Sediment sowie dessen Wasserchemie und Stoffhaushalt steuert. Die Monitoring-Ergebnisse zeigen diese engen Zusammenhänge, wobei die Entwicklung der ersten Jahre in zwei Phasen einzuteilen ist: Von 2005 bis 2007 waren morphologische Änderungen im Zuge des starken Sedimenteintrages aus dem terrestrischen

Bereich von großer Bedeutung. In der zweiten Phase (2008-2010) dagegen reduzierte die deutlich ansteigende Pflanzenbiomasse im terrestrischen und semi-aquatischen Bereich den Sedimenteintrag, verbunden mit einem stärkeren Wachstum submerser Wasserpflanzen.

Innerhalb weniger Jahre entwickelte sich eine große Phytoplankton-Diversität (s. Kapitel 7). Diese schnelle, artenreiche Besiedlung eines räumlich isolierten Wasserkörpers zeigt die hohe Effizienz in der Ressourcennutzung schon in den ersten Jahren. Die große Fluktuation der Arten zwischen den Jahren deutet jedoch auf eine geringe Stabilität der Gemeinschaften hin. Die erste Phase wurde dabei durch pelagische Algen dominiert, die durch die Verfügbarkeit von Phosphor und Licht gesteuert wurden. Da Teiche mit stabilen Flachwasserbereichen jedoch ideale Habitate für submerse Wasserpflanzen darstellen, wiesen diese schon bald eine größere Biomasse auf.

Diese Entwicklung führte in der zweiten Phase der Teichentwicklung zu einem verstärkten Entzug von Nährstoffen aus dem Teichsediment und einer geringeren Verfügbarkeit von Phosphor für die Primärproduzenten (s. Kapitel 8). Die große Bedeutung der Zuflüsse für die chemische Zusammensetzung des Wasserkörpers zeigte sich an der beständigen Abnahme von Schwefel sowie der kontinuierlichen Zunahme von Hydrogencarbonat. Die geringen Gehalte an organischer Substanz im Sediment zeigen den jungen Entwicklungszustand des Teiches an. Die beschriebene Besiedlung durch submerse Wasserpflanzen wird jedoch eine zunehmende Verfügbarkeit von Phosphor zur Folge haben.

Die Ergebnisse zeigen die große Bedeutung der terrestrischen Sukzession für die initiale Teichentwicklung, allerdings gewinnen interne Prozesse an Bedeutung. Wenn größere externe Störungen ausbleiben, wird die weitere Entwicklung des Hühnerwasser Teichs vermehrt durch diese internen Prozesse gesteuert werden. In Zukunft wird der Stofffluss an terrestrischen und aquatischen Grenzflächen intensiver untersucht werden, unter anderem im Hinblick auf die Quellenfunktion der Substrate für Nährstoffe.

10.4   Synthese

Ökologische Sukzession ist definiert als Serie von Übergängen in der Entwicklung von Ökosystemen, die mit Veränderungen der Umweltbedingungen verbunden sind und die biologische Besiedlung beeinflussen. Diese Veränderungen können von externen Faktoren hervorgerufen werden oder durch interne Prozesse kontrolliert werden (Townsend et al., 2005). In den ersten fünf Jahren der Entwicklung des Einzugsgebietes Hühnerwasser beobachteten wir bedeutende Veränderungen, die sowohl auf interne wie auch auf externe Faktoren zurückgeführt werden konnten.

Interne Strukturen und initiale Substrat-Eigenschaften waren entscheidend für die Verteilung und den Fluss von Niederschlagswasser und für die biologische Sukzession (s. Kapitel 9).

Externe Faktoren wie episodische Niederschlagsereignisse erzeugten in Verbindung mit lückiger Vegetation und lockerem Substrat Erosion und Zerschneidung der Oberfläche. Diese Prozesse führten zur Umlagerung von Wasser und Sediment innerhalb des Einzugsgebietes und zur Ausbildung neuer Strukturelemente. Die Folge dieser Entwicklung war die Unterteilung des Gebietes in Bereiche mit Abtragung oder Akkumulation sowie Bereiche mit stabilen Oberflächen. Im Folgenden beeinflussten sowohl externe Faktoren als auch Prozesse innerhalb des Einzugsgebietes die Entwicklung. Dabei war die Oberflächenausbildung ein wichtiger Faktor für die Ausbildung von biologischen Bodenkrusten, Vegetationsmustern, Nahrungsnetzen im Boden sowie der Gemeinschaften im Teich.

Nach fünf Jahren waren Teile des Gebietes noch spärlich mit Pflanzen und Tieren besiedelt. Initiale Strukturen wie biologische Bodenkrusten beeinflussten diese Muster durch die Veränderung von Oberflächeneigenschaften sowie die Erzeugung von Oberflächenabfluss und Erosion. Ein Beispiel dafür war die Ablagerung von Sediment in den Erosionsrinnen, die dort die Wasser- und Nährstoffverfügbarkeit erhöhte und dadurch bessere Wuchsbedingungen für die Vegetation erzeugte. Der Übergang von einem Geo-System in Bereiche mit terrestrischen oder aquatischen Charakteristika und von einem episodischen zu einem permanenten Abfluss erhöhte zusammen mit der biologischen Dynamik die Heterogenität des Gebietes im Hinblick auf die Verfügbarkeit von Wasser und Nährstoffen.

Die beschriebenen Prozesse erlauben mit Hilfe des Modells von Fath et al. (2004) eine integrierte Einordnung der Entwicklung des Einzugsgebietes: Aufgrund der gestiegenen Diversität, Menge und Größe der Organismen (Pflanzen und Tiere, terrestrisch und aquatisch) stieg die Zahl effektiver Strukturen deutlich an. Andererseits weisen die Ergebnisse auf eine geringe Stabilität in mehreren Kompartimenten des Systems, eine geringe unterirdische Diversität und auf eine geringe Bedeutung von Rückkopplungs-Mechanismen hin. Dies unterstützt die Einordnung, dass sich das Gebiet noch in einem sehr jungen Stadium der Entwicklung befindet. Allerdings spiegeln die gesunkene räumliche Varianz der Bodenlösung, die zunehmende Komplexität der Nahrungsnetze und die ansteigende Bedeutung größerer Pflanzen (Bäume, Wasserpflanzen) bereits eine leicht zunehmende Organisation und Effizienz – und damit Weiterentwicklung – des Systems wider.

Für die Zukunft erwarten wir deshalb (i) kurzfristig ein weiteres strukturelles Wachstum, d.h. eine Zunahme der Biomasse, (ii) mittelfristig eine weitere Zunahme der Konnektivität zwischen den Bereichen und innerhalb der Kompartimente sowie (iii) zeitlich versetzt ein qualitatives Wachstum hin zu mehr effizienten Mustern im Einzugsgebiet Hühnerwasser. Schließlich gehen wir davon aus, dass Rückkopplungs-Mechanismen in naher Zukunft an Bedeutung gewinnen werden, hervorgerufen durch die steigende Zahl effektiver Strukturen im Gebiet. Daher sollten diese Wechselwirkungen zukünftig verstärkt analysiert und die Untersuchungen an die weiteren Veränderungen des Ökosystems angepasst werden.

Danksagung

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