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# Can we thrift shop biodiversity? Translocation of Topsoil from a Road Construction Site to Increase Native Biodiversity in an Urban Landscape

Graduate thesis in Natural Resources Management in biology

Supervisor: Bente J. Graae

Co-supervisor: Martijn L. Vandegehuchte

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Norwegian University of Science and Technology  
Faculty of Natural Sciences  
Department of Biology











## **Preface**

This master thesis has been written to fulfil the graduation requirements of the program Natural Resources Management in biology at the Norwegian University of Technology and Science. The seeds for this project were set in December 2021, and submitted in June 2023. In my previous years at NTNU, I started to develop my interests for global, ecosystem-based biology. During the pandemic, I found out I wanted to pursue my passion for restoration ecology, just not continuously through a computer. Therefore, I knew I wanted to pursue a project where I could be outdoors, elevate my statistical skills, work with plants and do something new. With the soil moving project GreenMove swooping in from the Department of Civil and Environmental Engineering and their multiple partners, these wishes became a reality.

When I was to choose my supervisor, I thought about which course I enjoyed the most, and deliberately chose Bente J. Graae. I want to thank you for the guidance, supportive conversations, for taking me to conferences and for providing me the challenges I needed to grow and improve as a student and soon-to-be biologist. New to NTNU at the time, my co-supervisor Martijn L.

Vandegheuchte was there to guide me through every step on the way; from the first glance of the study sites to the topsoil translocation, and to the very last revisions of the thesis. Your stress-free attitude, humour and range of knowledge made me enjoy statistics, push myself and see it through. Talking about support, I want to thank my family and friends for being there for me throughout my academic progression and making me see the big picture in a narrow field. Mom, thank you for giving me, both figuratively and literally, the guts to excel. Up until now, I never thought I could be a person who could move the earth from under our feet.

*To anyone who doubts.*

*Like a plant – it is only when it rains, that you grow,  
It is when you learn, you see all there is yet to know,  
It is when you are challenged, your traits show,  
With good relations by your side,  
Resiliently, to the conditions you abide.*

*Now the time has come to find your fate,  
For your roots, new substrates await,  
Against the odds, you translocate,  
Even though it may not go as planned,  
Like a seed, you will always land.*

*When excavated and ripped out of the ground,  
You bring the compounds you have bound,  
No matter the novel soil condition,  
You can always reach the target you envision.*

*From the known to the new, put together the pieces,  
Take the leap and hand in your thesis.  
Be true to yourself and see it through,  
As you will flourish and be restored anew.*

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## Abstract

Restoration of vascular plant diversity in species-poor urban areas is vital to remediate fragmented nature. For this, topsoil translocation can be a useful method, which can facilitate re-establishment of native plant communities via the introduction of entire plants, rhizome fragments, and seed banks together with the soil to which the plants are adapted. In this study we translocated topsoil from a forest and grassland at a road construction site to, respectively, a shaded and open site in an urban park. We examined the effects of translocating bulk soils and intact vegetation turfs on plant species communities. Effects of digging up turfs or mixing bulk soil were examined *in situ*, at the site of origin, *ex situ* when translocated into an urban park, and for original park vegetation. Vegetation responses in species richness, plant community composition and community-weighted mean Ellenberg indicator values were described one and three months after transfer. Turf translocations from forest to park restored species richness and plant community compositions, which was not found for bulk handling of soil where species richness decreased. Both turf and bulk soils could re-establish target species richness in the grassland. However, both presented novel community compositions compared to the reference grassland. In both habitats, *ex situ* bulk soils were colonized by dominant local park species, while turfs proved to be more resilient to immigration. In bulk soils translocated from the forest, about 70 % of all the species declined in abundance. This was true for half of the grassland species. Several species such as *Vaccinium myrtillus* were positively impacted by *ex situ* turf transfers. In both habitats, *ex situ* bulk soils and turf maintained its soil pH, moisture, nitrogen, and phosphorus, inferred from plant communities' Ellenberg values. Turf translocations appear to be a more useful restoration tool than bulk soil translocation as turfs contained the most species after one season; however, more long-term monitoring is necessary to better understand the effects of topsoil translocation, particularly regarding the eventual convergence of translocated bulk soil communities to the reference plant community.

## Sammendrag

Restaurering av plantemangfold i artsfattige urbane områder er avgjørende for å reparere fragmentert natur. Det kan utføres ved å translokere toppjord fra artsrike områder, hvor både frøbanken og plantenes tilpassede jordsmonn blir bevart, som kan fasilitere gjenvekst av lokale plantesamfunn. I denne studien undersøkte vi vegetasjonsresponsen ved jordforflytting fra en blandet skog og semi-naturlig eng fra artsrikt veiarbeidsområde, inn til et artsfattig urbant parkområde. To jordforflyttingsmetoder ble undersøkt – intakte vegetasjonsstykker på pall, og jordmasser med avbrutt vegetasjonsdekke lagt i poser. Effekten av håndtering av jordmasse og jordstykker undersøkt på opprinnelsesstedet (*in situ*), transplantert toppjord (*ex situ*) og for den lokale parkvegetasjonen. Artsrikhet, plantesamfunnsstruktur og gjennomsnittlige vektete Ellenberg-indikator verdier ble beskrevet én og tre måneder etter transplantering. For skogen fant vi at forflytting av jordstykker kunne gjenopprette artsrikhet og artssammensetning på lik linje som urørt vegetasjon både *ex situ* og *in situ*, mens transplanterte jordmasser var artsfattige og divergerte fra referansevegetasjonen. Enkelte arter som *Vaccinium myrtillus* økte i antall ved forflytting av jordstykker inn til den mer skyggefulle parken. I engen kunne forflyttede jordstykker og jordmasser kunne gjenopprette opprinnelig artsrikdom, men artssammensetningen divergerte fra urørt referansevegetasjon. For begge habitat ble jordmassene dominert av *Taraxacum sp.*, og *ex situ* jordmasser ble kolonisert av parkarter. I skogen var halvparten av alle artene mindre suksessfulle i jordmassene, som kun gjaldt for en tredjedel av artene i engen. Forflyttede jordmasser og jordstykker opprettholdt verdier for pH, fuktighet, nitrogen og fosfor sammenlignet med referansevegetasjonen i begge habitat, selv om parkvegetasjonen hadde gjennomgående høyere nivåer av næring pH. Vegetasjonsstykketranslokasjoner ser ut til å være et nyttig restaureringsverktøy og opprettholder plantesammensetningen uavhengig av mottakerområdets substrat, men mer langsiktig overvåking er nødvendig for å avdekke om de forflyttede jordmassene konvergerer til opprinnelsesstedet over tid.

## Introduction

As 75 % of all natural habitats have been altered by human activity, restoration of natural ecosystems is of global importance (Jones et al. 2018; IPBES 2022; Rumpel et al. 2022). The UN has declared the years 2021 to 2030 the Decade of Restoration (UNEA 2019), to emphasize its vital contribution to mitigating the current climate and nature crisis. The first step towards a long-term restoration process is access to healthy soil and native source plants to transform degraded ecosystems and grey infrastructure.

Road construction is one of the major land-use changes driving biodiversity loss, affecting whole landscapes through habitat fragmentation (Haddad et al. 2015; Razafindratsima et al. 2021) and on a local scale through pollution, hydrological changes, soil compaction and introduction of invasive species (Oddsdóttir et al. 2011; Marrs 2016; Balász et al. 2016; Vegdirektoratet 2016; Craven et al. 2018; Delgado-Baquerizo et al. 2020; Farrell et al. 2020; Skrindo & Mehlhoop 2021). These road conditions are found to homogenize roadside soil fauna diversity, which drives a bottom-up subsequent loss of plant species and functional diversity (El Mojahid et al. 2017; Oldén et al. 2021). As a result, roadside vegetation can be a battleground “won” by species with prerequisite traits for roadside survival, characterized as fast-colonizing, light-demanding, affinity for high soil pH, low moisture, and low organic matter content (Oldén et al. 2021). Meanwhile, road operation is dependent on vegetation and topsoil to create road verges with ecosystems providing erosion control, hydrological stability, snowdrift management, air quality and dust mitigation (Hagen & Skrindo 2010; Zhao et al. 2020). If the pre-construction plant communities are suitable to provide these services, revegetation of road verges by local topsoil masses is a desired method (Hagen & Skrindo 2010; Vegdirektoratet 2016; Skrindo & Mehlhoop 2021).

Topsoil (upper 10-30 cm layer) is the most biologically active part of the soil, contains fungal networks and favourable microfauna, and allows native seeds deposited in the soil to establish within its adapted soil conditions (Harris 2009, Reynolds et al. 2003, Riviera et al. 2014; Wubs et al. 2016). A common revegetation method is to strip and separate the soil layers and store the topsoil in the construction phase, either as intact vegetation mats as turfs or as stockpiled bulk soil (Skrindo & Pedersen 2003; Riviera et al. 2014; Jørgensen 2019; Skrindo & Mehlhoop 2021). Full regeneration of the original plant species is rarely the aim in road verges, as the ecological state of the area has been altered substantially (Vegdirektoratet 2016; Farrell et al. 2020). Commercial seed mixtures can be used to speed up succession, which can lead to competitive exclusion of native species in the area, with potential long-term negative effects (Aamlid et al. 2013; Hagen et al. 2014; Mehlhoop et al. 2022).

Therefore, road construction sites should be prime target for restoration efforts. Today’s protocols and practices for monitoring of revegetation practices are not sufficient to establish ecosystem-specific tools nor a unified method (Skrindo & Mehlhoop 2021; Mehlhoop et al. 2022). As the ecological



effects of roads are well-known, construction companies have an obligation to move valuable species away from the impact area on a permanent or temporary basis (Vegdirektoratet 2016). The protocols are mainly specific to Red Listed, historical or genetically valuable species (Vegdirektoratet 2016), and there is little focus on species-rich plant communities. Translocation of construction site soils and vegetation to other sites earmarked for restoration, with a higher potential for biodiversity development, present a real yet underutilized opportunity. Plant and soil communities can thus be salvaged from damaging construction and roadside disturbances and established in a suitable new habitat. Native seeds and soils can hence be sourced from areas where nature is about to be destroyed anyhow, as opposed to encroaching on, excavating, and translocating from pristine ecosystems (Aradottir 2012; McLean 2003, Bulot et al. 2017). Topsoil translocation can be viewed as “assisted dispersal” of plants and plant communities (Wade 1991), has been used to restore degraded ecosystems, such as mine quarries (Le Stradic et al. 2016), road fills (Riviera et al. 2014), oak forests (Douterlugne et al. 2018) and has proven to be able to recreate target ecosystems over time (Standen & Owen 1999). There are immense opportunities in the field yet to be discovered (Skrindo & Halvorsen 2008; Mehlhoop et al. 2022).

Translocation methods can have different outcomes, as turf translocations tend to be more successful than bulk soil spreading (Plywell 1995; Le Stradic et al. 2016). By translocating intact turfs, most of the living native vegetation can be maintained along with the soil and root structures, enabling rapid revegetation. The idea of bulk soil translocation is that plant propagules (seeds, rhizome fragments, etc.) can ensure regrowth of the original vegetation. In bulk topsoil transfers, the topsoil is transferred in bags or piles, and the initial vegetation usually consists of pioneer species with disturbance-tolerant traits, such as grasses and sedges (Aradottir 2012). On the other hand, bulk topsoil is spreadable over larger areas, requires less storage space, requires less care and fuel and is thus considerably less expensive (Good et al. 1999, Buisson et al. 2018, Piqueray et al. 2020). When comparing to a desired reference state, bulk soil spreading has been found to enable the system to converge to the desired state (Standen & Owen 1999; Bulot et al. 2014), but over longer time frames compared to turf translocations, especially in forest soil transfers (Zhao et al. 2020). Moreover, bulk topsoil transfers can be prone to drought, invasive and non-target species dominance, (Good et al. 1999; Ferreira et al. 2015; Ferreira & Vieira 2017; Buisson et al. 2018; Rezende et al. 2021), and germination failure due to seed burial, (Adjalla et al. 2022), which can hinder the success of ecosystem restoration.

Choice of receptor site can be tightly coupled to topsoil translocation success (Gerrits et al. 2023). When plant communities are introduced to a site with different management history, soil conditions, and biotic and abiotic factors, these can alter the donor vegetation (Manchester et al. 1999). All translocation projects can induce changes to the vegetation (Bullock 1998). Therefore, careful considerations are needed to direct the vegetation changes into a desired state. Meanwhile, challenges to restore habitats into a comparable intact state can be coupled to incompatible receptor site

characteristics in levels of soil moisture, pH and nutrients (Kiss et al. 2021), or lack of proper management (Tozer et al. 2012; Buckley et al. 2017, Buisson et al. 2018). The receptor site should therefore be selected to match the plant species' niches, as well as spatial limitations of machinery access and societal conflicts.

Restoration of forest species tends to be more challenging than that of grassland species in topsoil translocations, as grasslands are disturbance-induced habitats, dependent on mowing, grazing or fire regimes (Collins et al. 1998; Fynn et al. 2004). “Losers” of forest restoration can be those already negatively affected by soil disturbance and logging practices - specialist plant species with complex germination cues, long juvenile periods and transient seed banks such as dwarf shrubs, which tend to be less successful in both short- and long-term perspectives (Tozer et al. 2012; Aradottir 2012). Woody forest species are usually poorly represented in seed banks, (Douh et al. 2018, Adjalla et al. 2022), while herbaceous pioneer species can be four times more numerous (Adjalla et al. 2022). Therefore, moderate shading by nurse plants and irrigation has been provided in forest topsoil translocations, which can enhance species richness and woody species survival rates (Santiago-Garcia et al. 2008; Vloon et al. 2021; Zhao et al. 2022).

Most importantly, a topsoil translocation project must provide a substantial net gain of ecosystem services and biodiversity to the receptor site. Places in need of such services, are urban areas – which have doubled in land cover since 1992 and greatly expanded into forests and grasslands (IPBES 2019). Increasing the urban biodiversity has been increasingly emphasised over the last decades, aiming to benefit urban populations through recreational and educational values, mitigation of urban heat island effects (Akbari et al. 2004) and increased human well-being by access to green and diverse spaces (MacPherson et al. 1988; IPBES 2019). Nevertheless, urban areas can also be *ex situ* refuges for endangered species, as a complementary conservation effort to *in situ* conservation (Alvey 2006; Li & Pritchard 2009). Moreover, the variety of green urban spaces can meet the needs of specific species, (Godefroid et al. 2007; Fischer et al. 2013; Rudolph et al. 2017), but studies on introduction of endangered herb and shrub species into urban areas are lacking (Pan et al. 2019). Parks can be of special interest for enhancing species diversity, as frequently mowed lawns often originate from mass-produced seeds of the same diversity-poor mixtures used on a global scale (Thompson et al. 2004; Stewart et al. 2009; Rudolph et al. 2017).

The main national highway is under expansion in Norway, and in several locations, parallel to existing roads, natural habitats are used for the construction of high-speed and four-laned highways. As the state aims to enhance both connectivity and speed, creation of curvature is limited, and construction in species-rich areas is seemingly unavoidable. However, construction companies are obligated to revegetate and compensate for extensive losses of biodiversity and valuable ecosystems under the Nature diversity Act (Naturmangfoldloven). Nevertheless, efforts to date have had variable outcomes

(Hagen & Skrindo 2010; Statens Vegvesen 2021). The concept of topsoil translocation from construction sites can be a key element to the COP15 Montreal-Kunming agreement, as Norway is obligated to restore 30% of all degraded natural ecosystems by 2030 (EU 2022). According to the Red List of Threatened Species in Norway, 21% of species are threatened, while the majority are found in forests and semi-natural grasslands, which makes them key habitats for restoration research purposes (Artsdatabanken 2021).

The aims of this project are to examine plant community responses to bulk and turf topsoil translocation from a forest and grassland site and separate the effects of excavation or soil mixing and the effects of translocation to a novel receptor site. This separation of effects was achieved by excavating and placing back turfs and bulk soils within the donor site and within the receptor site, as well as translocating both turfs and bulk soils from donor to receptor site, and by using untouched sites in donor and receptor site as reference. To identify plant community responses associated with environmental changes, we chose to examine the effects of our handling and translocation treatments on plant community means for five Ellenberg indicator values. To elaborate on species specific responses, we focused on some native early successional species (*Taraxacum sp*, *Avenella flexuosa*), late successional species (*Vaccinium myrtillus*), species with affinity for nutrient-rich soils (*Dactylis glomerata*) and associated with nutrient-poor communities (*Briza media*). The study was designed to answer the following research questions: How many of the plant species can be restored in topsoil translocation from a grassland and forest into an urban landscape? How important for plant community composition is moving the vegetation in whole turfs versus in bulk soils? Which species survive soil handling and translocation to a novel habitat, and which do not? What change can be observed in Ellenberg indicator values in plant communities growing in excavated bulk soils and turfs, and how do they differ when being moved into a nutrient-rich receptor site?

# Methods

## Study sites

The topsoil receptor and donor sites are marked in Fig. 2.1 that were approximately 83 km apart, with the donor site located at Taubaneveien (62.82628, 010.03831) in Rennebu municipality with an elevation of 524 m.a.s.l.

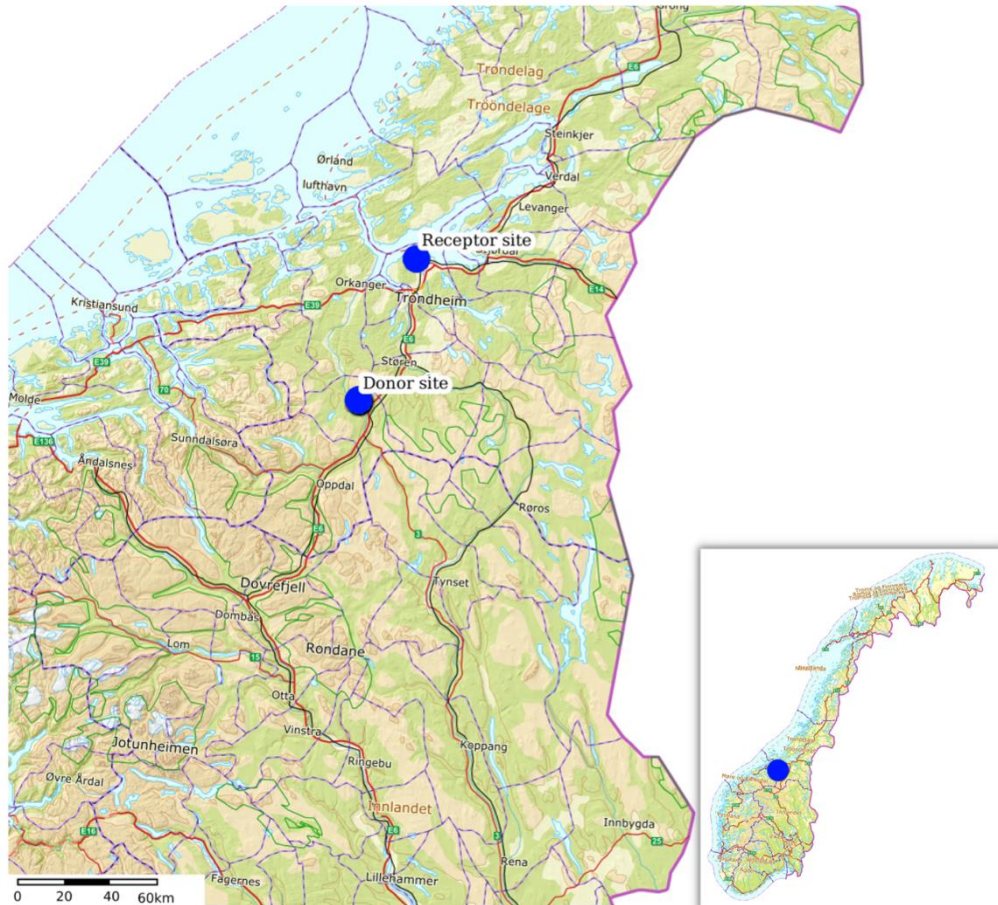


Figure 2.1: Both sites were located within the county of Trøndelag (Kilden, NIBIO 2023).

Rennebu has a mean annual temperature of 3,0 °C and a 750-1000 mm annual precipitation (30-year mean) (YR 2023a). Over 2022, minimum annual temperature spanned from -24 °C in December 2022 and to a maximum of 20 °C in July (YR 2023a). The site was within a 15-kilometer planned road transect, where all land was temporarily owned by the road construction company Nye Veier AS (Nye Veier 2022a; 2022b). The real-estate company and GreenMove-partner Proinvenia provided site access, and potential study sites were examined through maps of species occurrences, soil types and overall accessibility (Artsdatabanken 2022, NIBIO 2022). After site selection in the transect in early June, Taubaneveien was observed to have the highest biodiversity, best machinery access and had the closest proximity to Trondheim. Based on species maps, the near threatened grass *Briza media* were observed in the area (Artskart 2009), which is associated with the threatened traditional semi-natural grasslands (Solstad et al. 2021).

The chosen donor site (fig 2.2b) was approximately 210 m and 230 m wide, included a small hut, a gravel road for entrance, forest patches and grassland, with a farm and peatland outside of the experimental area. Construction of the road is planned to start in 2025, which will contribute as a part of the new national highway and will be ~2 kilometres away from the existing parallel road (Nye Veier 2022a; 2022b).

The area was in a pre-construction phase, as logging machines had cut almost all trees of *Sorbus aucuparia*, *Betula pubescens*, *Salix caprea*, and *Juniperus communis* shrubs in the spring of 2022. A fence divided the experimental area into two (see fig. 2.2), where the west side was dominated by understory forest species, such as *Vaccinium myrtillus*, *Anemona nemorosa* and *Avenella flexuosa*, while the east side was grazed by sheep until 2020 and dominated by *Rumex acetosa*, *Geranium sylvaticum* and *Ranunculus acris*.

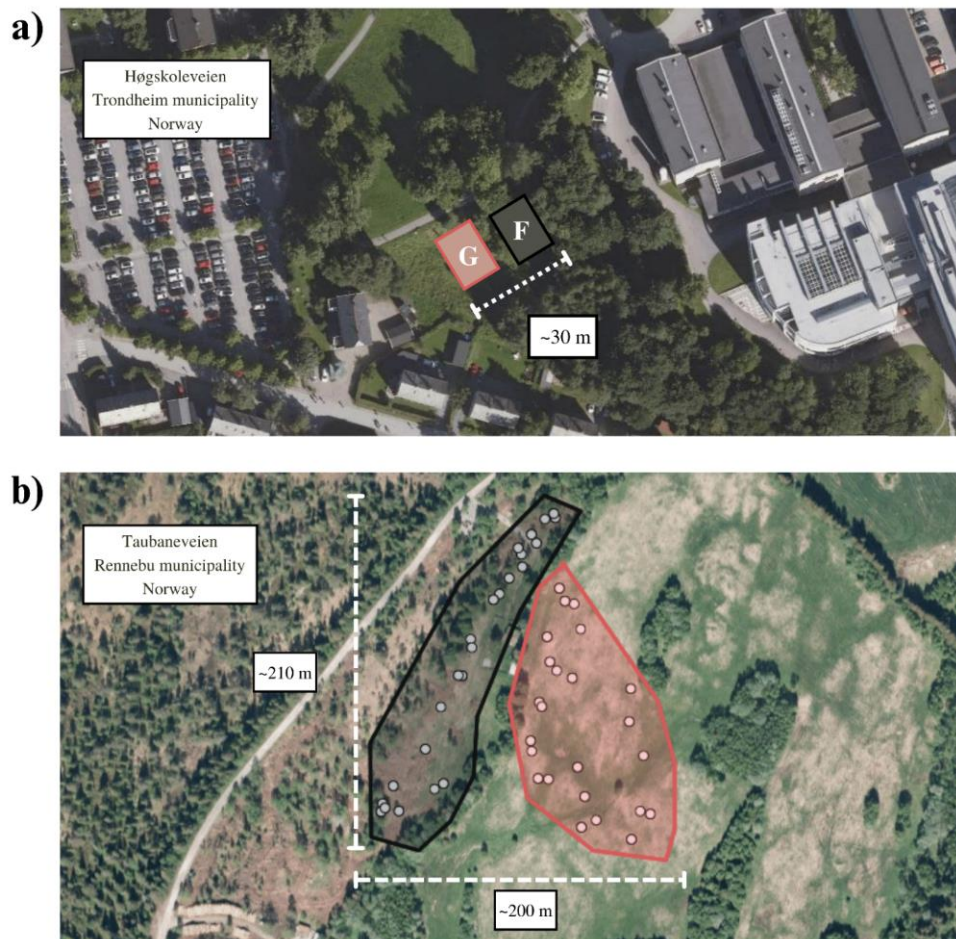


Figure 2.2: a) Receptor site in the campus park and. The campus park was separated into a shaded forest part, denoted as F in black and sun exposed grassland part denoted as G in red, in the chosen study area at Vestskråningen. The slope was located next to a paved and broad path connecting two parking lots, which allowed for excavator access. b) The donor site at Taubaneveien in Rennebu municipality. Each dot represents one of the 48 *in situ* plot coordinates for the forest (black) and grassland (red). The two habitats were separated by a fence. Constructed in GIS link (2023).

The receptor site, Høgskoleparken at campus Gløshaugen of The Norwegian University of Science and Technology (NTNU) (63.415671, 10.403098) is situated on a hill with densely constructed buildings, steep slopes, and grass-dominated lawns (see fig 2.2a). It has a mean annual temperature of 7,0 °C and a mean annual precipitation of 750-1000 mm (YR 2023b). The maximum temperature in 2022 spanned from 29,5 °C in June to a minimum of –18,7 °C in December (YR 2023b). Our study site covers approximately 200 m<sup>2</sup> of the park area called “Vestskråningen”, a south-east facing slope with an elevation of ~40 m.a.s.l. The slope displayed a moisture gradient with a dry top to wet bottom, and considered unsuitable for infrastructural development and will therefore not be directly affected by the planned campus extension plans. In addition, the site is not mowed, reflected by a tall and dense grassy vegetation cover, and has high ecological potential (SLA 2017).

Increasing the park biodiversity is emphasized in the current campus plan, as stated in NTNU’s Environmental Development Plan for 2020 to 2030 (NTNU 2019). The park has a total of 35 different species of vascular plants recorded between 2000 to 2023, and of these 9 were non-native, some with high invasive tendencies (Artsobservasjoner 2023). This project has been a part of a cross-disciplinary team from *Rewilding Campus*, with an aim to explore methods to rewild the park in line with NTNU’s biodiversity goals. The rewilding of Høgskoleparken could be a beacon for the future of sustainable technology, biodiversity conservation and urban ecology.

### **Experimental design**

The topsoil translocation process ran from 21-30 June. At the donor site, 40 bamboo flags were placed randomly in each of the two habitats (grassland and former forest) (fig. 2.2b). Of these, 16 in each habitat were marked as “*ex situ*” (eight turfs and eight bulk soils), 16 as “*in situ*” (eight turfs and eight bulk soils) and eight as “untouched donor” (Fig. 2.3). An excavator dug turfs and bulk soils in 120 x 80 cm rectangles and 20-25 cm deep. The “*ex situ*” turfs were placed on individual pallets with pallet collars covered with a plastic film, and the “*ex situ*” bulk soils were placed in plastic bags. As for “*in situ*” turfs, they were simply lifted up and put back. In “*in situ*” bulk plots, the turfs were flipped with the vegetation cover facing down and cut into four pieces (flipped). After translocation, one bulk “*in situ*” plot in the grassland was missing and was therefore remarked in a bulk “*ex situ*” plot, where only half of the bulk soil was translocated. The excavation and transportation were completed on 22 June. The 24 *in situ* forest and 24 grassland plots were spread out randomly within the two habitats (fig 2.3b).

Subsequently, *ex situ* bulk and turf were stacked in wooden boxes, transported and stored in the park for three days, during which brief irrigation was done on the turfs due to warm temperatures. For receiving the topsoil, two experimental grids of 12.5 × 8.0 m each were constructed; one in a sun exposed part, considered to be suitable for grassland topsoil (grid G), and the other in a part shaded by surrounding, large trees to receive forest topsoil (grid F) (see fig. 2.4). The construction of the park



grids was done using an excavator and hand-held shovels. Prior to the stripping of the park topsoil, the excavator extracted 8 turfs and 8 bulks from each grid (See Appendix A for park preparation). Park extractions of bulk and turf were conducted along the slope in each grid row. However, during extraction of turfs in the grassland, there was a risk of sinking in the moist soil conditions in the bottom and were therefore only extracted from the top of the slope. Additionally, 8 untouched park plots were marked along the slope at the same height as the grid rows in intact park vegetation, located one to three meters from each grid.

#### Experimental design N=128

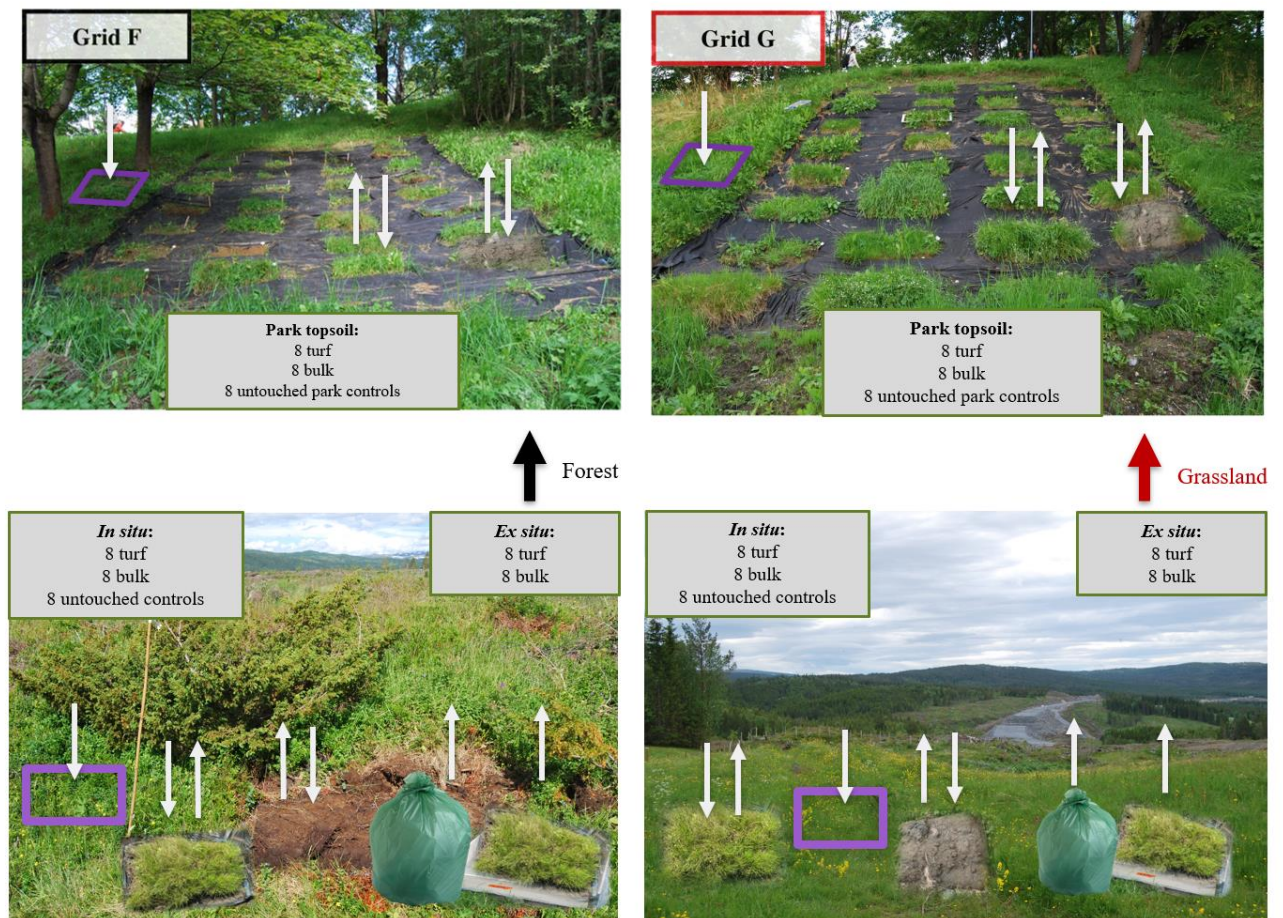


Figure 2.3: Experimental design at the donor site and in the park. In grid F, shadow was provided by *Betula pubescens*, *Salix caprea* and *Acer platanoides* located in between the grids and on the top of the slope. Both areas were dominated by *Alchemilla sp.*, *Dactylis glomerata*, *Rumex acetosa*, *Alopecurus pratensis*, and *Agrostis sp.* The bottom wet part of both grids was dominated by *Filipendula ulmaria* and *Cardamine pratensis ssp. paludosa*.

Each translocated piece of turf and bulk were placed in 15-20 cm deep holes and framed with soil-cloths with equal spacing (0,5 m), as illustrated in fig 2.4. The turfs were cut in half and placed in the designated hole, while bulk bags were emptied directly into the hole. Each treatment was represented

in each of the eight rows. Untouched controls in the park were marked one to two meters outside each grid. The topsoil was inserted into the grids from the 26<sup>th</sup> to the 30<sup>th</sup> of June with the help of 15 volunteers.

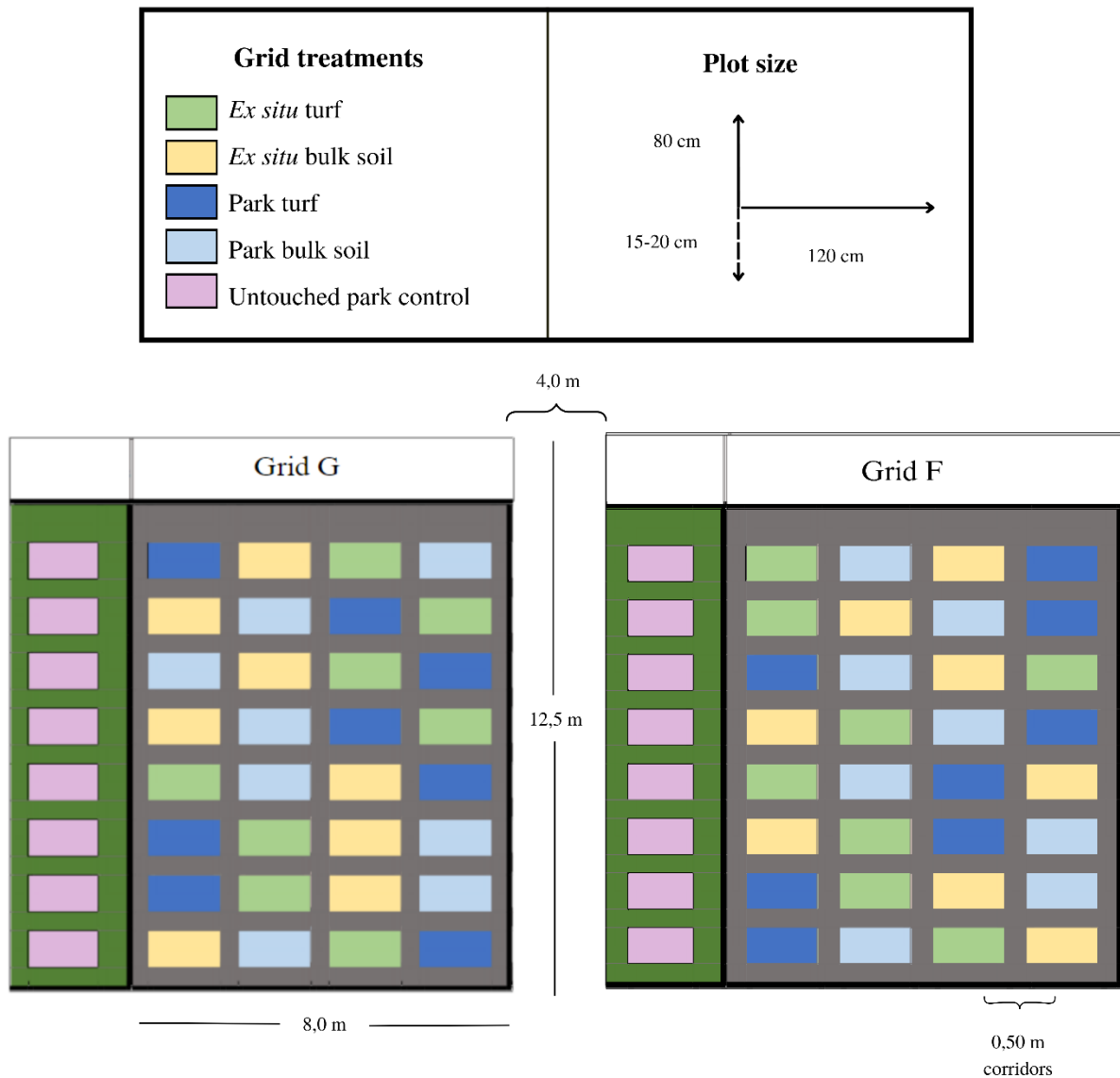


Figure 2.4: Experimental grids G (grassland) and F (forest) at the receptor site in Høgskoleparken. Each treatment was replicated in each row and row placements were randomized.

In summary the 8 treatments were: park bulk soil, park turf, untouched park control, soils *in situ* bulk soil, *in situ* turf, untouched donor control, *ex situ* turf, and *ex situ* bulk soil. These 8 treatments were replicated 8 times for each of the 2 habitat types (forest and grassland) for a total of  $8 \times 8 \times 2 = 128$  plots.



## Data collection

Two rounds of vegetation analysis on vascular plants were performed, the first in July and the second late in August to the beginning of September. All vascular plants were identified and counted within each plot with a 100 x 50 cm aluminum frame divided into 25 subplots (10 x 20 cm) as shown in figure 2.5. Presence-absence was recorded for individuals rooted within each subplot, yielding a frequency from 0-25 per plot per species. All plots except bulks were analyzed in the first round, as the bulk initially had no visible plant growth. Individuals in the genus *Taraxacum*, *Alchemilla*, and in some within *Hieracium* were not recorded to species. The early season analysis was done from the 1<sup>st</sup> of July and ended on the 19<sup>th</sup> of July, and late season from 18<sup>th</sup> of August and ended on the 12<sup>th</sup> of September. Succession of the vegetation cover can be found in Appendix A.

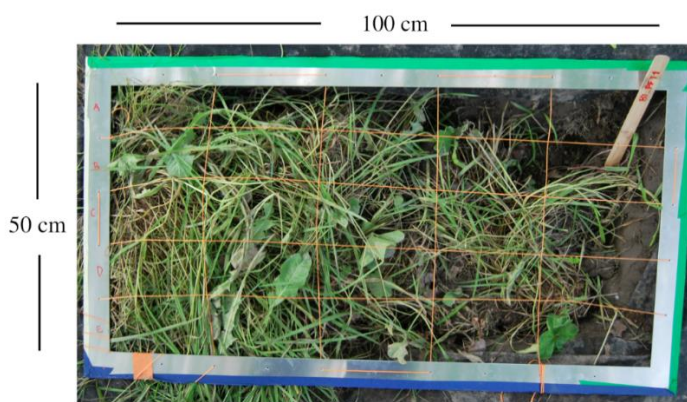


Figure 2.5: Aluminum vegetation frame used for vegetation analyses. Subplots were 10 x 20 cm.

## Ellenberg-values

Ellenberg indicator values (EIV) are species-specific values from 1-9 for plant fundamental niches, which can be used to reflect environmental conditions in e.g., the soil (Ellenberg 1974). Values for pH, phosphorous, nitrogen, moisture and light were found for 99 out of 108 observed species, most obtained from two Swedish studies (Tyler et al. 2020; 2021). The remaining 13 species were not included, as 4 species were unknown, 7 were only identified to genera and 2 had missing EVI's. Values for *Omatholeca norvegica* and *Sagina saginoides* were found through the Ecological Flora of Britain and Ireland (Fitter & Peat 1994). *Taraxacum sp.* was analysed as *Taraxacum officinalis*, and the pH-value was obtained from Lawesson (2003). EVI's for the subspecies *Cardamine pratensis ssp. paludosa* were only available for the main species *Cardamine pratensis*. The phosphorous indicator value for *O. norvegica* was missing, and therefore obtained by taking the average of its observed common observed neighbours in the collected data set. Moreover, community weighted means (CWMs) for each plot were calculated as such: each species abundance was multiplied with its EIV, these products were summed, and this sum was divided by the total abundance of species in the plot.

## Statistical analysis

All statistical analyses were done separately on forest and grassland data. To account for sampling bias and potential gradual increase of species richness over July and August, the data sets with vascular plant species occurrence were combined. This was done by selecting for each species and each plot the maximum number of subplots in which that species occurred from the July and August values. For example, if species X was observed in 7 of the 25 subplots in July and only in 3 subplots in August, the value for X was set to 7. To visualize “winners” and “losers” among the plant species translocated to the park, each species’ cumulative occurrence in untouched donor plots was subtracted from that in bulk soil or turf plots. In this way, values for species that decreased in abundance after translocation or excavation, compared to the untouched controls, were negative, while values for species that showed a relative increase were positive. The count data with the total species richness per plot was Poisson-distributed, had unequal residual variance and was not overdispersed, and was therefore analysed with a generalized linear model (GLM) with a Poisson distribution (O’Hara & Kotze 2010) with handling and translocation type and their interaction as fixed effects. Random effect of row position in the experimental grids on species richness was examined in a generalized linear mixed model (GLMER), and as the effect was found to be zero, did not improve the model fit and was therefore excluded from the analysis. Pairwise comparisons of species richness in the different treatment groups using a likelihood ratio test (LRT) for marginal means estimation (EMMs) and p-values were corrected with “False Discovery Rate”-adjustment (FDR) (Benjamini & Hochberg 1995). Ordination plots were generated using non-metric multidimensional scaling (NMDS) for species abundances, based on Bray-Curtis-dissimilarity and 999 permutations, using the “vegan”-package (Kruskal 1964a; 1964b; Faith et al. 1987; Oksanen 2016). We performed a permutational multivariate analysis of variance (PERMANOVA), on the Bray-Curtis dissimilarity matrix, with handling and translocation type, and their interaction as fixed effects (Clarke & Green 1988; Warton et al. 2012; Somerfield et al 2021), which was followed up by a post-hoc pairwise PERMANOVA (Martinez 2020). A test for homogeneity of species dispersions in each treatment group was done with the “betadisper()”-function (Anderson 2006), ANOVA and a post-hoc test with pairwise comparisons with FDR-correction. Furthermore, available Ellenberg indicator values for light, moisture, phosphorous, nitrogen and pH were used to calculate community weighted means (CWM) based on subplot occurrence. The CWM EIV’s for each plot were analysed with a NMDS and a PERMANOVA was conducted on the plotwise matrix. For each EIV, the CWMs were analysed with a two-way analysis of variance (ANOVA) with handling, translocation type and their interactions. Pairwise post-hoc comparisons were conducted with EMMs, and p-values were adjusted for multiple testing with FDR-correction. As some CWMs showed unequal variances across treatment groups, they were analysed with a Kruskal-Wallis-test and FDR-correction of p-values with the same predictors. Results were termed statistically significant for p-values  $\leq 0.05$ . Graphical illustrations were computed through “ggplot” (Wickham 2009). All statistical analyses were performed with R software, version 4.2.2 (RStudio Team 2020).

## Results

### Species richness

A total of 112 different vascular plant species were found, including 4 unknown species that due to very early life stage lacked diagnostic features. These had fewer than 4 observations across plots, and were excluded from all analysis, apart from their contribution to species richness. As shown in table 1, a total of 77 species were translocated from the forest into the park, of these were 46 new to the park area. Out of the 80 species found in treatment groups at the donor site, 15 were unique and thereby not translocated. In the grassland, 61 species were transferred through the *ex situ* treatments, where 30 species were novel to the park.

*Table 1:* Number of species found in the totally 256 vegetation analyses combined in the forest and grassland treatments for the different translocation types. Unique species were found only in the specified types. Grand total is all the species combined, when overlapping grassland and forest species are considered.

Translocation type	Forest	Unique species	Grassland	Unique species
<b>Park</b>	38	5	42	9
<i>Ex situ</i>	77	6	61	3
<i>In situ</i>	80	15	58	5
<b>Total in habitats</b>	<b>100</b>		<b>77</b>	
<b>Grand total</b>	<b>112</b>			

The highest number of species per plot was 26 species, found in forest *in situ* turf and *ex situ* turf, and secondly untouched forest donors with 25 species. Highest number of species in the grassland was 20 species in *in situ* turf and 17 in turf *ex situ*. In contrast, the lowest number of species was 3 species in *in situ* bulk in the forest, and 2 species in *in situ* bulk in the grassland. See Appendix B for the estimated means and groupwise species occurrences. Differences in species richness was driven by handling and translocation type and their interaction in both habitats, while translocation types in the grassland did not differ (Table 2).

Table 2: Effect on species richness based on the GLM-models for the forest and grassland.

Model	Factor	LR Chisq	Df	p-value	
<b>Forest</b>					
GLM	Handling type	92.60	2	<0.001	***
(Poisson)	Translocation type	39.22	2	<0.001	***
	Handling : Translocation	29.51	3	<0.001	***
<b>Grassland</b>					
GLM	Handling type	20.08	2	<0.001	***
(Poisson)	Translocation type	4.82	2	0.089	ns.
	Handling : Translocation	30.76	3	<0.001	***

In both habitats, the same level of plant species richness as untouched donors was found in excavated turfs put back at the donor site and turfs moved to the park (fig 3.1). Species richness in bulk did not differ when being moved into a novel receptor site, compared to excavated bulk at the donor site in both habitats. Park species richness did not change by topsoil handling in neither experimental grid, as the same level of species was found in untouched park controls as in turf and bulk. In the forest, the bulk treatments contained the fewest species (fig 3.1a, see Appendix B for raw data), as bulk excavation (*in situ*) and translocation (*ex situ*) had significantly fewer species than excavated and translocated turfs. In contrast, grassland species richness did not differ between untouched donors, translocated donor bulk and turfs (fig 3.1b), however excavated *in situ* bulk did not establish the same number of species compared to excavated and translocated turfs and untouched donors.

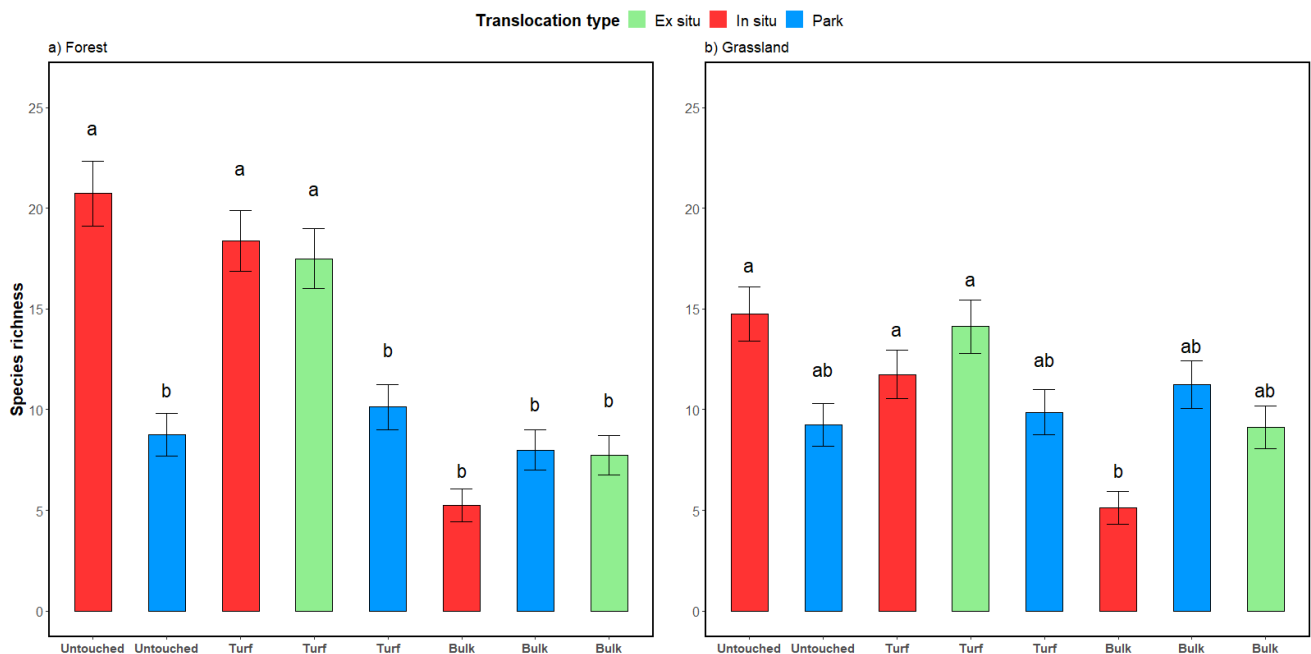


Fig 3.1: Number of species in the a) forest and b) grassland for each soil handling and translocation type. Error bars represent  $\pm$ SE and letters show significant post-hoc pairwise differences among treatment group after a two-way ANOVA was conducted.

## Community composition

Among all treatments with donor site origin, i.e., combinations of translocation and handling type, there was a significant difference in plant community compositions in both habitats (Table 3). In the forest, the only groups with similar plant community compositions were translocated turfs (*ex situ*), excavated turfs put back in place (*in situ*) and untouched donors. Meanwhile, excavated and translocated turfs in the grassland were similar but differed from the target communities. To display species correlations, five species were selected in terms of strong significance, vector direction and different growth forms (fig 3.2). Vectors were inflated for ease of visual interpretation.

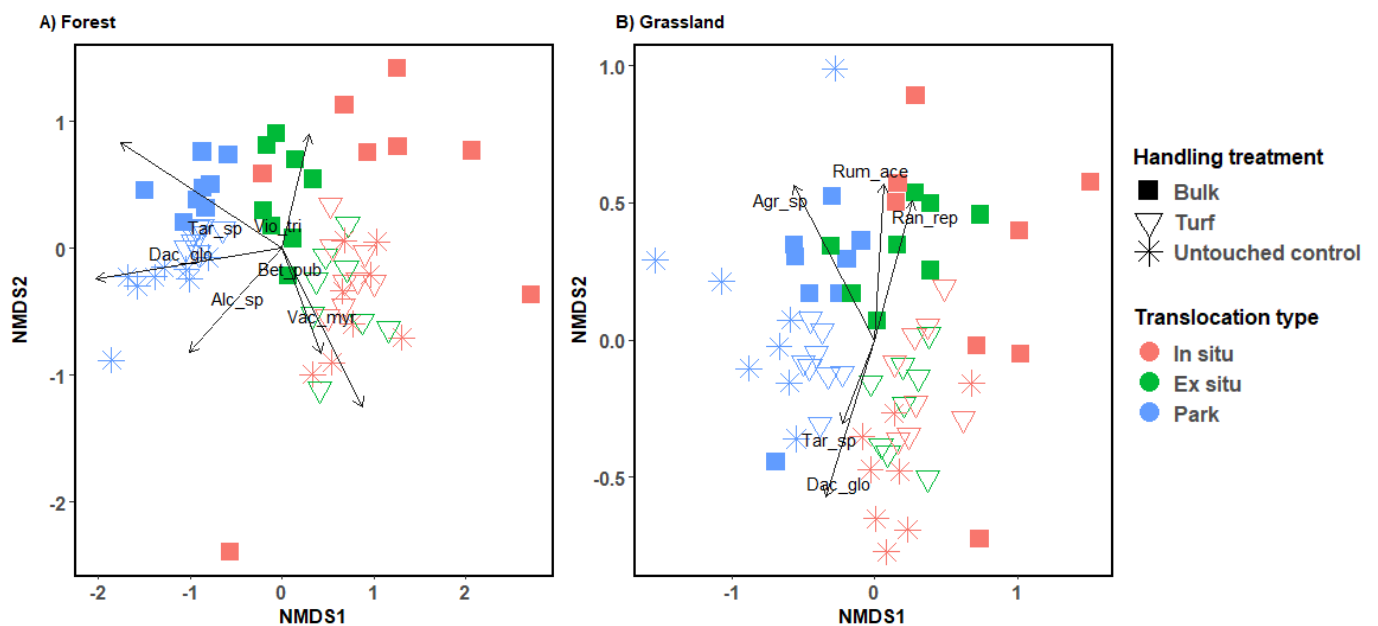


Figure 3.2: NMDS with species occurrences with vectors for selected significant species in the A) forest (stress=0.16) and B) grassland (stress=0.21). Species abbreviations: Alc\_sp: *Alchemilla sp.*, Bet\_pub: *Betula pubescens*, Vio\_tri: *Viola tricolor*, Dac\_glo: *Dactylis glomerata*, Tar\_sp: *Taraxacum sp.*, Vac\_myrr: *Vaccinium myrtillus*, Agr\_sp: *Agrostis sp.*, Rum\_ace: *Rumex acetosa*, Ran\_rep: *Ranunculus repens*, Phl\_alp: *Phleum alpina*, Luz\_mul: *Luzula multiflora*.

All forest park plots were densely clustered together along the NMDS1-axis. However, all bulk soils were separated from turfs and untouched control plots along the NMDS2-axis. There was a clear overlap among untouched donor plots, *in situ* turfs, and *ex situ* turfs. *In situ* bulk soils were the most spread-out, with more variation than *ex situ* bulk. Meanwhile, *ex situ* bulk soils had a plant community composition that was closer to that of all plots with a park origin, compared to all other groups originating from the donor site. *Alchemilla sp* pointed away from *in situ* bulk soils, while it was equally close to untouched donor and untouched park plots. *Dactylis glomerata* was associated most strongly with park plots, while *Betula pubescens* and *Vaccinium myrtillus* were directed towards plots

with donor site origin (untouched donor, *in situ* and *ex situ*). *Viola tricolor* was directed towards *in situ* and *ex situ* bulk soils, nearly in the opposite direction of *Alchemilla sp.*

In the grassland, *ex situ* bulk had some overlap with both park bulk and *in situ* bulk. The untouched park and bulk *in situ* groups showed the widest spread along the second axis. Like the forest, untouched donor, turf *in situ* and turf *ex situ* had an even overlap, but untouched donor plots in the grassland were even more separated. The vector for *Dactylis glomerata* was directed towards untouched plots, with a stronger relationship than *Taraxacum sp* and turfs, while *Ranunculus repens* and *Rumex acetosa* were correlated with bulk soils. *Agrostis sp* intersected between untouched park and park bulk, towards *ex situ* bulk, and away from untouched donors, *ex situ* and *in situ* turfs.

Table 3: PERMANOVA with 999 permutations for effects of species compositions for handling, translocation type and their interaction in the forest and grassland.

	Factor	ndf	ddf	Sum of squares	R <sup>2</sup>	F	p-value	
PERMANOVA Analysis of similarity	<b>Forest</b>							
	Translocation type	2	61	1.91	0.09	3.15	<0.0001 ***	
	Handling type	2	61	2.32	0.11	3.90	<0.0001 ***	
	Translocation : Handling	7	56	5.64	<b>0.27</b>	3.04	<0.0001 ***	
	<b>Grassland</b>							
	Translocation type	2	61	1.35	0.10	3.49	<0.0001 ***	
	Handling type	2	61	2.64	0.19	7.60	<0.0001 ***	
	Translocation : Handling	7	56	6.05	<b>0.45</b>	6.74	<0.0001 ***	

The homogeneity of variance on species compositions was tested to examine whether the community structures within treatment groups had equal plot-to-plot variance (see table 4). There was evidence of heterogeneity in community-structures between park and *ex situ* treatments in both forest and grassland treatments (see Appendix C). Meanwhile, the excavated donor bulk were in both habitats found to be the treatment with the most plot-to-plot variation, significantly different from *ex situ* turfs and untouched donors. In the grassland, all groups differed in variance from the *in situ* bulk, besides the untouched park and bulk *ex situ*.

Table 4: Permutational test of multivariate dispersion of species compositions in the grassland and forest, based on 999 permutations.

	<b>Factor</b>	<b>ndf</b>	<b>ddf</b>	<b>Sum of squares</b>	<b>F</b>	<b>p-value</b>	
Homogeneity of dispersion	<b>Forest</b>						
	Translocation type	2	61	0.28	6.52	0.002	**
	Handling type	2	61	0.05	1.71	0.18	ns.
	Translocation : Handling	7	56	0.91	12.08	<0.0001	***
	<b>Grassland</b>						
	Translocation type	2	61	0.05	2.78	0.07	ns.
	Handling type	2	61	0.10	10.62	0.0001	***
	Translocation : Handling	7	56	0.25	5.46	<0.0001	***

### Treatment effects reflected by community weighted means for pH, phosphorous, moisture, nitrogen and light

CWMs for light, pH, phosphorous and moisture EIVs showed a clear relation to topsoil origin for both habitats, but with some variations (see fig. 3.5 & 3.6). The differences in CWMs were best explained by the translocation type in the forest (R=0.42) (table 5)., while the differences in the grassland was best explained by the interaction effect between handling and translocation types (R=0.36)

Table 5: Community weighted mean Ellenberg indicator values and effect of handling, translocation type and interaction effect, based on a PERMANOVA with 999 permutations.

	<b>Factor</b>	<b>ndf</b>	<b>ddf</b>	<b>Sum of squares</b>	<b>F</b>	<b>R2</b>	<b>p-value</b>	
PERMANOVA Analysis of similarity	<b>Forest</b>							
	Translocation type	2	61	0.25	48.16	0.60	0.001	***
	Handling type	2	61	0.013	2.06	0.05	0.36	ns.
	Handling : Translocation	7	56	0.007	0.93	0.017	0.42	ns.
	<b>Grassland</b>							
	Translocation type	2	61	0.009	3.23	0.09	0.008	**
	Handling type	2	61	0.012	4.71	0.13	0.0007	***
	Handling : Translocation	7	56	0.035	4.68	0.36	0.0001	***

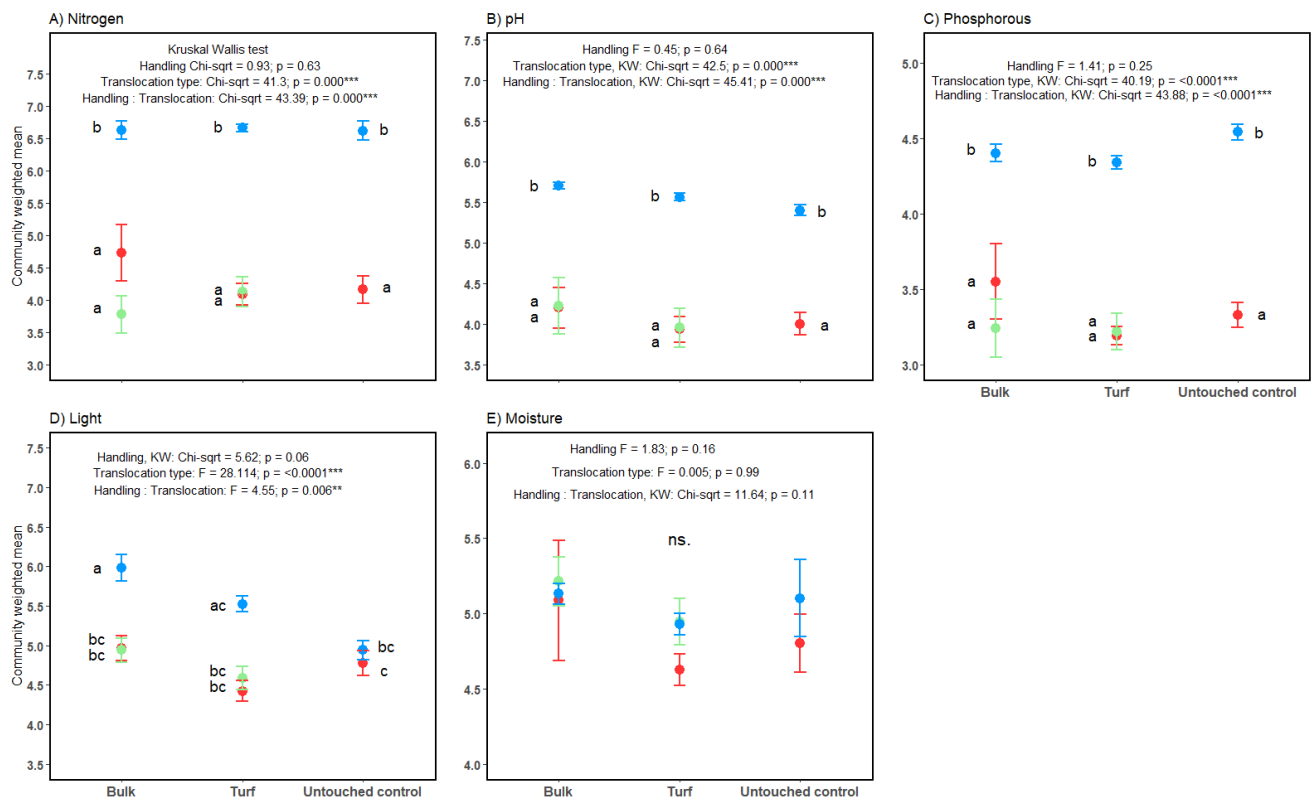


Fig 3.5: CWM Ellenberg values in the forest *in situ* (red), *ex situ* (green) and park (blue) treatments and significance levels for translocation, handling and in interaction. A two-way ANOVA and post-hoc pairwise comparisons (EMMs) were run if no other test is specified. Kruskal-Wallis tests were used for CWMs with unequal variances among treatment groups.

In the forest, CWM Ellenberg values were similar for all bulk soils, turf, and untouched donor plots with donor site origin (fig 3.5). Park treatments in the forest grid showed significantly higher CWM EIVs for nitrogen, pH and phosphorous than *in situ* and *ex situ* treatments. In the park bulk soils, the lack of vegetation cover led to an establishment of light-demanding species, in a higher level than bulk soils with donor site origin. There was no change or difference among groups for levels of moisture.

CWMs for the grassland had more groupwise overlap than for the forest (fig 3.6). The presence of nitrogen-demanding species did not differ between turf that was excavated and placed back *in situ*, untouched donors and *ex situ* turfs. *In situ* bulk soils had significantly more nitrogen-demanding species than untouched donors and park turfs. Plant communities had an affinity for significantly lower pH values in park turfs compared to untouched park and park bulk soils. CWM of pH was higher in the untouched park than the untouched donor and was the same among all in *ex situ* and *in situ* topsoil. CWM Ellenberg values for phosphorous did not differ among any treatment group, even though it was found to be significantly different in plant communities between treatment groups, no significant contrasts were found with FDR-p-value correction of multiple testing. In all bulk soils *in situ*, *ex situ* and from the park, community means for light were higher than untouched controls.



Community-weighted mean Ellenberg values for moisture did not differ between treatments, with only one significant contrast between the drier park turf and wet *in situ* bulk soil.

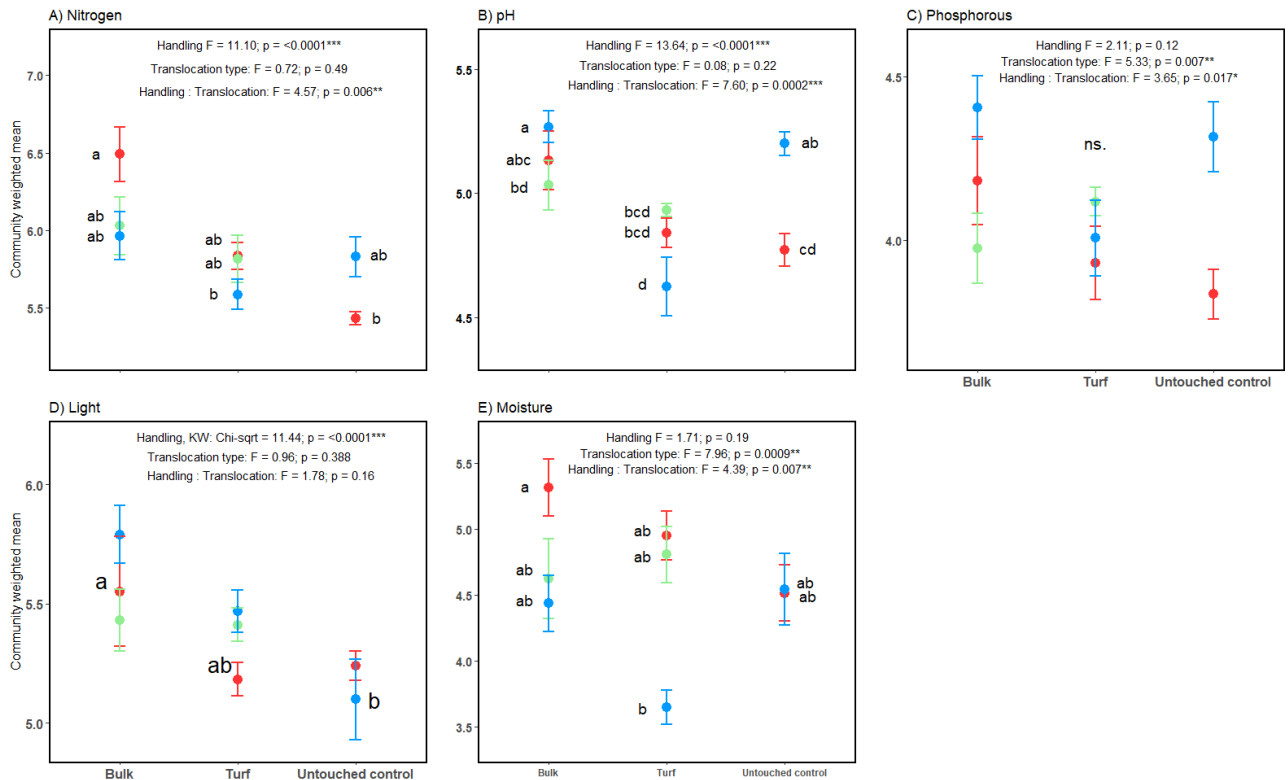


Figure 3.6: CWM Ellenberg indicator values in the grassland with colours for *in situ* (red), *ex situ* (green) and park (blue) treatments and significance levels for translocation, handling and in interaction based on ANOVA or Kruskal Wallis (KW) post-hoc tests.

In the NDMS for CWM EIVs, there was a clear distinction between park plots on the one hand, and both *in situ* and *ex situ* treatment groups on the other, especially for the forest plots (fig 3.7).

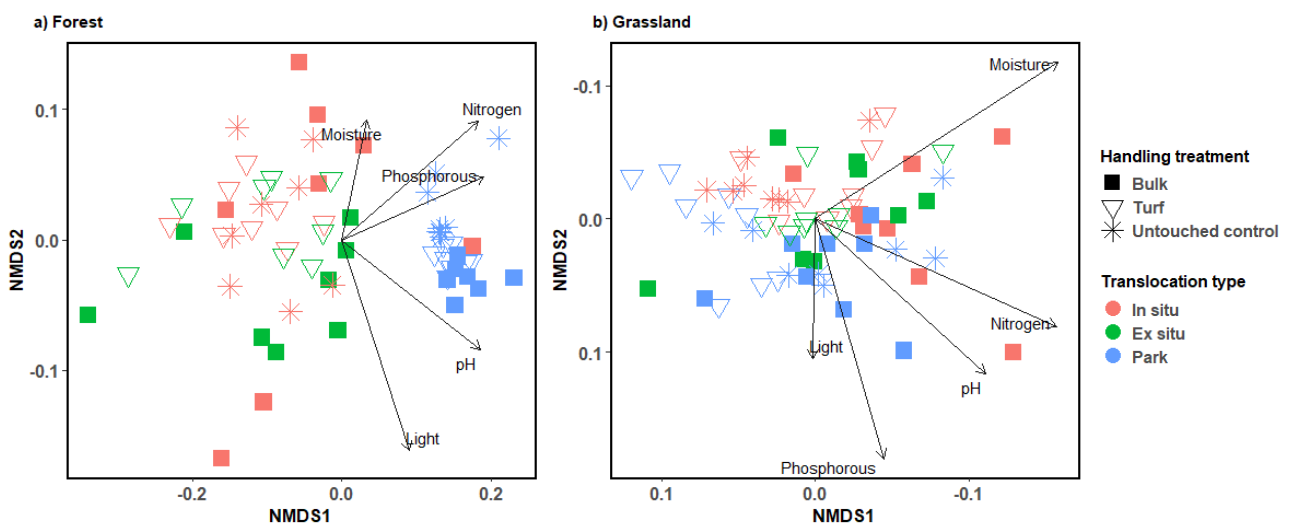


Figure 3.7: NMDS with community weighted means for A) forest (stress=0.07) and B) grassland (stress=0.12) for Ellenberg indicator-values for light, pH, phosphorus, moisture and nitrogen.

In the forest, high values for phosphorous, pH and nitrogen were correlated with the park treatments (fig. 3.7a). The CWM for light in the forest was directed towards bulk *in situ* plots and the cluster of bulk *ex situ* plots, away from excavated (*in situ*) and translocated (*ex situ*) turfs. In the grassland, the park topsoil types had more overlap with donor plots (fig. 3.7b). Vectors for moisture and pH were pointing away from turf park plots. Moisture was pointed towards *in situ* and *ex situ* turf, away from park treatments. Meanwhile, values for light and phosphorous were not significantly different among grassland treatment groups, the vector for light was shorter, indicating more similarity among the treatment groups. Nitrogen was only significantly different between untouched donor and bulk *in situ*, which can be seen in the vector directed towards bulk *in situ* plots.

## Winners and losers among plant species after translocation of bulk soil and turf

A set of 15 species in the forest and 5 in the grassland were only found in *in situ* plots, and not in *ex situ* plots. These “non-translocated” species in the forest were *Viola tricolor*, *Viola riviniana*, *Trisetum spicatum*, *Solidago virgaurea*, *Rubus saxatilis*, *Plantago major*, *Paris quadrifolium*, *Omalotheca norvegica*, *Fragaria vesca*, *Leucanthemum vulgare*, *Gymnocarpium dryopteris*, *Cirsium heterophyllum*, *Carex vaginata*, *Campanula rotundifolia* and *Briza media*. In the grassland, the non-translocated plant species were *Viola tricolor*, *Rhinanthus angustifolius*, *Pilosella aurantiaca*, *Campanula rotundifolia* and *Ajuga pyramidalis*. “Winners” are here determined as species that increased by turf or bulk translocations while “losers” were the species with lower occurrences, compared to the target communities. Values for species that showed a relative increase were positive in the green area while values for species with lower mean occurrences are shown in the red area, conducted for both *in situ* and *ex situ* plots, and for the forest and grassland separately (fig 3.8 and 3.9). To keep the focus on winners and losers, species with one or zero difference to the untouched donor in cumulative occurrences and all occurrences in park plots were excluded from the figures. The abundance of species in park treatments is described in Appendix A. See Appendix B for raw data.

Table 6: Number of winning (+), neutral (0) and losing (-) species in each handling and translocation type in the forest and grassland, based on the relative abundances to the untouched donors. Percentages were calculated based on the number of species found in the translocation type and rounded up to the closest integer.

Translocation type	Total number	Handling type	Winners	Neutral	Losers
<b>Forest</b>					
In situ	80	Bulk soil	5 % (4)	5 % (4)	73 % (58)
		Turf	18 % (14)	32 % (26)	20 % (16)
Ex situ	77	Bulk soil	10 % (8)	5 % (4)	71 % (55)
		Turf	23 % (18)	35 % (27)	18 % (14)
<b>Grassland</b>					
In situ	58	Bulk soil	4 % (2)	7 % (4)	53 % (31)
		Turf	21 % (12)	21 % (12)	22 % (13)
Ex situ	61	Bulk soil	23 % (14)	2 % (1)	44 % (27)
		Turf	33 % (20)	10 % (6)	23 % (14)

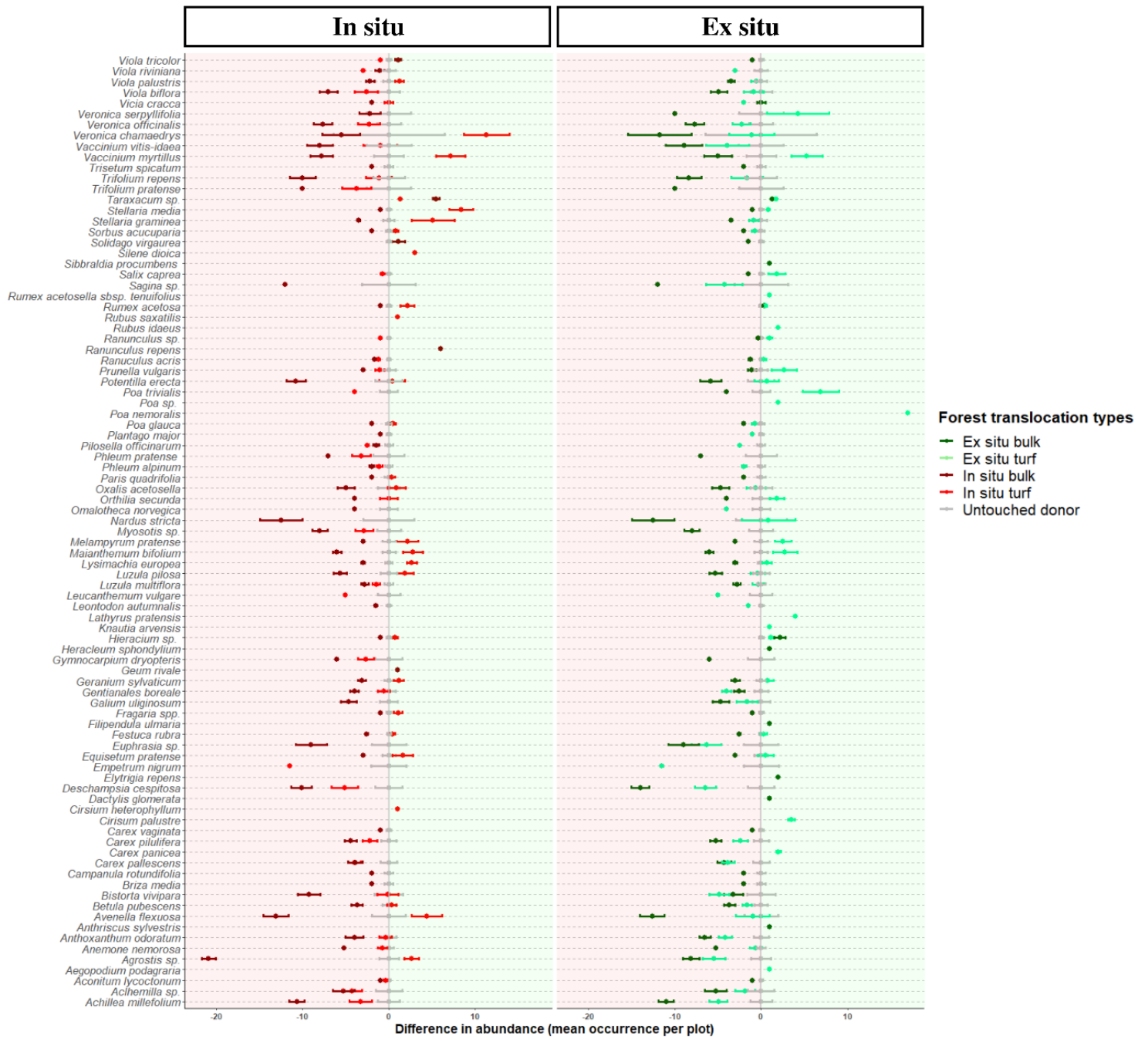


Figure 3.8: Mean change ( $\pm$ SE) in abundance compared to untouched controls for *in situ* and *ex situ* plots in the forest. Species found in park are not displayed. Mean abundance of the untouched donor is displayed with the line at  $y=0$ .

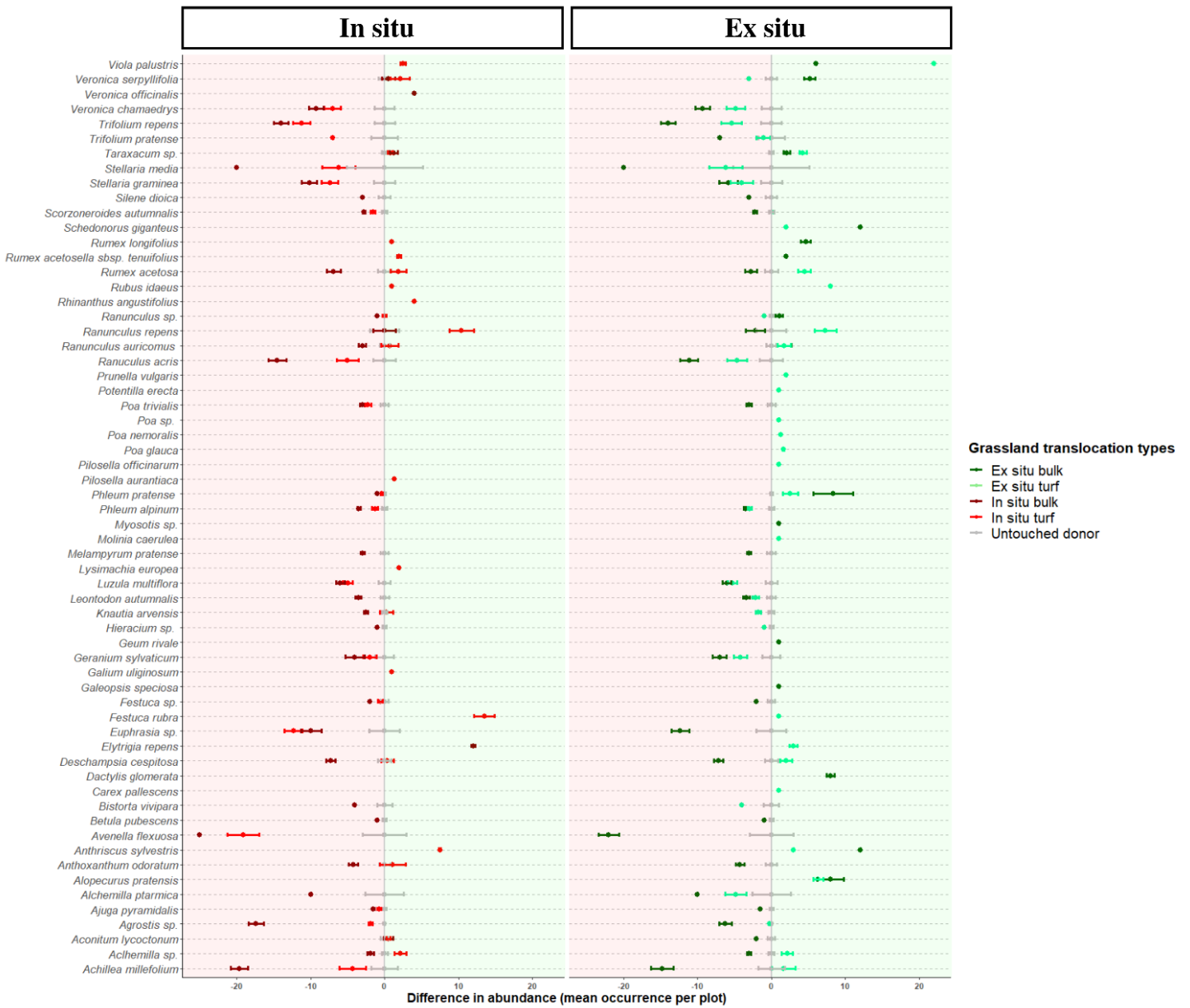


Figure 3.9: Mean change ( $\pm$ SE) in abundance compared to untouched controls for *in situ* and *ex situ* plots in the grassland. Species found in park are not displayed. The grey line at  $y=0$  with standard errors represents the untouched donor.

In the forest, 73 % of the species *in situ* plots had lower occurrence in bulk soils, and 71 % in *ex situ* bulk soils (Table 6, fig. 3.8). For both translocation types, bulk soils had a negative effect on the abundance of *Agrostis sp*, *Achillea millefolium*, *Deschampsia cespitosa*, *Trifolium repens*, *Veronica officinalis*, *Avenella flexuosa*, *Trifolium repens*, *Potentilla erecta*, *Nardus stricta*, *Sagina sp*, *Euphrasia sp*, and *Vaccinium myrtillus* i.e. the “losers”. Even though *Avenella flexuosa*, *Agrostis sp* and *Vaccinium myrtillus* (fig 3.10) were clearly negatively affected by the bulk soil treatment, they were more numerous in *in situ* turf transfers than in untouched donors. *V. myrtillus* increased in *ex situ* turf transfers as well.



Figure 3.10: The dwarf shrub *Vaccinium myrtillus*. Picture retrieved from Wikimedia Commons (2022).

The percentage of “winners” in bulk soil handling were twice as the amount in *ex situ* than in *in situ* forest bulk soils.. *Taraxacum sp* and *Hieracium sp*, had slightly higher occurrences in both *ex situ* and *in situ* bulk soils, while *Taraxacum sp* was found in all handling and translocation types, but not in untouched donor plots (fig. 3.11). *Ranunculus repens*, was in a single observation more abundant in *in situ* bulk than untouched donor plots.



Figure 3.11: *Taraxacum sp*. Picture retrieved from Wikipedia (2023).

Some forest species were just as negatively affected by turf as bulk soil handling in *in situ* soils, as found for *Aconitum lycoctonum*, *Phleum alpinum*, *Carex pallescens* and *Alchemilla sp*. The near threatened grass *Briza media* (fig 3.12) was not found in any treatments but untouched donors. Turfs in *in situ* plots yielded a higher abundance than untouched donors and bulk for *Poa trivialis*, *Poa nemoralis*, *Veronica serpyllifolia*, *V. chamaedrys* and *Maianthemum bifolium*.



Figure 3.12: *Briza media*. Picture retrieved from Wikipedia (2021).

Forest species that did worse in the *ex situ* treatments for both turfs and bulk soils were *Agrostis sp*, *Alchemilla sp*, *Bistorta vivipara*, *Euphrasia sp*, and *Genitiales boreale*, *Deschampsia cespitosa*, *Achillea millefolium*, *Carex pallescens*, *C. pilulifera*, *Anthoxanthum odoratum*, *Veronica officinalis*, *Betula pubescens*, *Bistorta vivipara*,



*Viola biflora*, *Luzula multiflora*, *Myosotis sp.*, *Euphrasia sp.*, *Vaccinium vitis-idea*, *Achillea millefolium* and *Myosotis sp.* Higher occurrences were found for *Knautia arvensis*, *Poa trivialis*, *Prunella vulgaris* and *Salix caprea.*, when being moved in turfs from the donor into the park.

In the grassland, among the species found in the bulk soils, 53 % had lower occurrences in *in situ* bulk soils and 44 % were less abundant in the *ex situ* bulk soils. Overall, in both translocation types, these were *Agrostis sp.*, *Achillea millefolium*, *Anthoxanthum odoratum*, *Euphrasia sp.*, *Ranunculus acris*, *R. auricomus*, *Trifolium repens*, *Poa trivialis*, *Melampyrum pratense*, *Phleum alpinum*, *Luzula multiflora*, *Geranium sylvaticum*, *Stellaria graminea* and *Deschampsia cespitosa*. *Avenella flexuosa* did not respond well to turf nor bulk translocations in either *in situ* or *ex situ* plots (fig. 3.13).



Figure 3.13. *Avenella flexuosa*.  
Picture retrieved from Lindsey Ecology of Commanster (2004).

*Dactylis glomerata* grew densely in the park (fig. 3.14). It was less abundant in park bulk soils and park turfs, than in untouched park vegetation. It was found to colonize *ex situ* bulk soils in both habitats, with highest number in the grassland, while it did not occur in *ex situ* turfs.

There were more “winners” in the grassland bulk soil in *ex situ* with 23%, and 4% in *in situ*. For both translocation types were *Taraxacum sp.*, *Elytrigia repens* and *Alopecurus pratensis* benefitting of the bulk treatment. *Ex situ* turfs had 8 more winners than *in situ*, while in both was an increase in abundance found for *Viola palustris*, *Rumex acetosa*, *Ranunculus repens*, *Rubus idaeus*, and *Festuca rubra*. The forb *Veronica officinalis* was not translocated and was one of the few species found only in *in situ* bulk soils. In bulk soil moved from the donor site to the park, the emergence of *Veronica serpyllifolia*, *Phleum pratense* and *Anthriscus sylvestris* were higher than their observed frequencies in all other treatments. *Rumex longifolius* was not found in untouched donors but was present in *in situ* turf and translocated bulk soil. There were more incidents of park plants immigrating into the *ex situ* soils, as *Alopecurus pratensis* was observed in the *ex situ* bulk soils and *Schedonorus giganteus* in *ex situ* turfs, species not present in any *in situ* treatment. *Ex situ* bulk soils had occurrences of *Galeopsis speciosa*, *Geum rivale*, *Myosotis sp.*, which were not found in any other treatment. Some species



Figure 3.14: *Dactylis glomerata*.  
Picture retrieved from Artsdatabanken Liebe (2023).

were only found in *ex situ* turfs, and not in any *in situ* treatment, which were *Molinia caerulea*, *Potentilla erecta*, *Prunella vulgaris*, *Poa nemoralis* and *P. glauca*.

## Discussion

In this study, we demonstrated how species-rich vegetation from a road construction site can be translocated into an urban species-poor park site. In the forest, there were limited receptor site effects on plant communities in translocated of turfs, as *ex situ* and *in situ* turfs re-established the same plant species richness and community compositions as in the original forest. Meanwhile, both *in situ* and *ex situ* grassland turfs restored the same species richness but did not recreate the same community composition as untouched donor vegetation. Bulk translocations did not recover plant community composition from either habitat, and significantly reduced forest understorey plant species richness. Meanwhile, the same number of plant species re-established in *ex situ* grassland bulk soils as untouched donor sites, but this was not found for the more species-poor *in situ* bulk soils. *Taraxacum sp* was most widespread in bulk soils in both habitats, while light and nutrient-demanding park grass *Dactylis glomerata* colonized *ex situ* grassland bulk soils, and in a lesser extent in forest bulk soils. In contrast to our expectations, *Vaccinium myrtillus* was positively affected when being moved into the urban area in turfs, which could be due to escape from competition in its original more sun exposed areas. Vegetation establishment in both *ex situ* bulk soils and turfs was not affected by the surrounding nutrient rich park topsoil and underlying subsoil, reflected by the community weighted mean Ellenberg indicator values.

### Handling responses in species richness differed between the forest and grassland

A total of 77 species were successfully translocated from the donor forest and 61 from the donor grassland into the park. In the forest, the number of species were equal for *ex situ* and *in situ* turfs and untouched donors but were lower in translocated bulk soils. Meanwhile, grassland bulk soils moved into the park re-established the same species richness. Grassland plant species are generally re-establishing after a disturbance (Aradottir 2012; Müller et al. 2014). Soil disturbances such as topsoil removal (Jentsch et al. 2009; Schnoor et al. 2015), tilling and ploughing (Valentin et al. 2016; Martin et al. 2021) in grasslands are used to reduce competitive species dominance to maintain and elevate species diversity (Pacala & Rees 1998). Meanwhile, logging and soil disturbances in understory forest vegetation can decrease the richness of wood-forming slow-developing species with less persistent seed banks (Carreño-Rocabado 2012; Vandewalle et al. 2010). Moreover, woody species can also have insufficient seed bank contributions and become diluted or buried in bulk soil transfers (Fowler et al. 2015). This corroborates our observation that bulk soil transfers had faster re-establishment of plant species for the grassland than for the forest.

In both habitats, *in situ* bulk soils did not re-establish the species richness as in untouched donor plots. In contrast, *ex situ* bulk soils in the grassland reached the same species richness as the untouched



target site. The cause might be explained by the lack of mixing, as the excavator flipped the *in situ* bulk soil and put it down, parts of the subsoil may have been exposed, where seed banks are scarce (Gramström 1982). In general, maintenance of biodiversity in grasslands does not tend to be limited by disturbances, but by availability of propagules (van Looy 2011; Müller et al. 2013; Buisson et al. 2018). Thereby, upper seed bank deposits could have been buried, which disabled seed bank germination. Contrastingly, this suggests that shuffling, airing and mixing of *ex situ* bulk soils could have allowed seeds from multiple soil depths to establish and stimulated seed germination of grassland species. The difference between *in situ* and *ex situ* bulk soils may also be due to the higher temperatures of the park. Although the *ex situ* bulk soils in the grassland restored target species richness, the richness was partly contributed by immigrating park species. Immigrations were observed for e.g. *Dactylis glomerata*, and if able to spread, might negatively affect the establishment of translocated native species in later stages (Li et al. 2016). Nevertheless, the number of species has a debatable relevance when it comes to measures of restoration outcomes (Brudvig et al. 2017; Rydgren et al. 2020), as this number does not reflect species identities or ecosystem functioning (Overbeck 2013).

### **Turfs restored the undisturbed community compositions in the forest, but not for the grassland communities**

In the forest, translocated *ex situ* and *in situ* turfs did not differ in species compositions from the untouched donor communities. In the grassland, there was no difference between *in situ* and *ex situ* turf excavation, but they did not recreate target plant community compositions. Disturbance associated with excavating and placing back the turf could have created a gap for grassland to germinate from seed banks or propagules, otherwise outcompeted in untouched donor vegetation, relating to the importance of disturbance (Grime 1973; Grime 1977; Collins et al. 1995; Guitet et al. 2018). Opposed to bulk soils, turfs in both habitats were less prone to immigrating park species, which can be due to that plant immigration can be hindered by microsite availability (Harper et al. 1965; Ross & Harper 1972; Grime 1977; Fowler 1986). While turfs are more resilient to changes in plant species composition (Le Stradic et al. 2016; Mùdrak et al. 2016), they still present novel plant communities that risk converging with the park vegetation over time because of changes to the composition of the transferred soil, such as nutrient influx, seed limitations (Douh et al. 2018, Adjalla et al. 2022), or competition from the local park vegetation (Auestad et al. 2015).

Transferred grassland and forest bulk soils diverged from the untouched donor plant communities; therefore, actions might be needed to reach the target plant community compositions. The plant communities of *ex situ* bulk soils were more similar to those of park plots than those of *in situ* bulk soils. As the *ex situ* bulk soil plant communities were isolated from their original habitat, competitive and dominant park species established where grassland species with nutrient-poor soil requirements, or forest shrubs and trees normally would. Absence of vegetation cover, nurse plants and abandonment

of management practices can lead to the dominance of a few competitive species in exposed bulk soils (Le Stradic et al. 2016), while poor seed bank content and insufficient seed deposition has been found to hamper long-term sustenance of other transferred communities (van Looy 2011; Buisson et al. 2018). Therefore, seeding or planting of native species, in the forest preferably shrubs or small trees, could be recommended to prevent transferred bulk soils from developing vegetation that converges with the vegetation of the receptor site, a successful application done in other studies (Zhou et al. 2019; Zhao et al. 2022).

Short-term monitoring after a disturbance such as topsoil translocation is merely an observation of early successional stages, while monitoring over multiple years can reveal the directional change and onset of functional change (Moog et al. 2005). Meanwhile, restored plant communities can become dominated by one functional group such as competitive grasses, which can persist over time without successional change (Zaloumis & Bond 2011). However, studies report that dominance of ruderals i.e., nutrient- and light-demanding species with fast life cycles, are found to decline over time after soil disturbance (Moog et al., 2005; Mudrak et al., 2017; Valko et al., 2022), such as tilling and excavation. This is because ruderals are generally poor competitors, and over time get replaced by more competitive species, which then gain dominance.

### **Introduced topsoil maintained its abiotic properties, a story told by Ellenberg indicator values**

Even though plant community compositions changed by handling treatments, there were minimal changes in community weighted mean Ellenberg indicator values for nitrogen, pH, phosphorous and moisture in both habitats. In the park, Ellenberg values of plants in the shaded area for nitrogen, pH and phosphorous were generally higher than for plant communities on forest topsoil, while the species in the donor grassland had more similar EIV's to those of the park grassland vegetation. The park and the grassland might reflect similar management schemes, as the park slope is mowed once a year (Gro Lefstad, personal communication), while the grassland was last grazed three years ago in 2020 (Knut Morten Odden, personal communication). When nutrient removal practices are abandoned, levels of phosphorous and nitrogen can accumulate in the soil, which also tend to be coupled with low species richness in grasslands (Wheeler 1988; Smith & Rushton 1994; Janssens et al. 1998). The grassland might be prone to a directional change in species composition toward plant species with stronger affinities for light and nutrients, which can explain why the donor grassland communities had more overlapping CWM EIV's with the park than with the forest communities.

Even though the park species had high EIV's for nutrients and light, there was no receptor site effect on either light or nitrogen affinity. The most successful species in an urban wasteland seedling study were found to be competitive species with the ability to grow tall, acquire both light and nutrients, while less competitive species requiring nutrient-deficient soil conditions were less successful (Fischer

et al 2012). However, CWM EIV's for light were not affected by handling or translocation types in the forest, while the bulk soil communities in the grassland presented higher CWM EIV's for light than untouched park vegetation. This suggests that the park vegetation developing on bulk soils is a light-demanding subset of the overall park species set. The shade in the area of the park into which forest soils were translocated might have created a poor arena for light-competitive park species to establish in *ex situ* bulk soils. The lack of change in turfs translocated into the park may be due to the lack of microsites for competitive park grass species to invade (Funk et al. 2008; Castro-Diez et al. 2016; Janneke et al. 2022).

The only difference in CWM EIV's for nitrogen in grassland donor soils was between the excavated *in situ* bulk soils with higher affinities for nitrogen than the untouched vegetation. As the *in situ* soils had limited seed bank germination as of the *ex situ* bulk soils, the plots were dominated by a few nitrogen-demanding colonizers, such as *Elytrigia repens*, *Taraxacum sp*, and *Geum rivale* (Tyler et al. 2021), inhabiting the partly exposed subsoil surface. Subsoil establishment can be less similar to target vegetation than topsoil establishment (Skrindo & Halvorsen 2008), which is likely why the nitrogen levels did not differ from the untouched vegetation in *ex situ* bulk soils,

Plant communities on park turfs in the grassland grid had very low Ellenberg indicator values for pH and moisture compared to the untouched park controls and park bulk soils. This could be an effect traced back to the preparation of the park grid. After the excavator had finished the forest grid, it had to abort midway during the turf and bulk extraction, due to wet soil conditions in the bottom of the slope. Initially, bulk soils and turfs were extracted from all parts of the slope. However, we were limited to extract turfs only from the top of the slope, while bulk soils were extracted from both ends. As ridges can be poorer in nutrients and moisture levels than adjacent depressions (Reader & Best 1989), it is likely that the topsoil on the top of the slope was prone to leaching, which can create acidic soils (Ball 1999). This suggests that a dry-to-wet slope gradient can affect nutrient levels and pH in the transferred topsoil in the years to come, even though there was no observed random effect of slope position.

### **More winners in the turfs, more losers in bulk soils**

Turfs projected almost an equal number of winners and losers in both habitats overall. Grassland turfs moved into the park even had more winners (33%) than losers (20%). As mentioned, this suggests that, by moving intact standing vegetation, survival can be facilitated by the presence of neighbouring plants and/or the minimizing of soil disturbance. Meanwhile, some species were only found in *ex situ* translocated bulk soils and turfs, but not at the donor site in both habitats. The translocation process could have created gaps for germination of fast-colonizing rather than competitive species from seed dispersal from the surrounding park vegetation or from the seed bank within the translocated soil

(Pacala & Rees 1998). Such species would otherwise have been outcompeted in untouched target vegetation (Schippers et al. 2001).

Bulk soils had more losers than any other treatment. In both *in situ* and *ex situ* soils, approximately three out of four forest species “lost” in bulk soils, while half lost in the grassland. As several species present at the donor site or the park did not appear in bulk soils, germination from seed banks in *ex situ* soils (Bissels et al. 2006; Skrindo & Mehlhoop 2021.) and seed dispersal and vegetative spread from surrounding park vegetation (Öster & Eriksson 2012) can be expected in the future. A winner in *ex situ* bulk soils was the competitive park grass *Dactylis glomerata*, which has no extensive vegetative spread (Grimes et al. 1988) but was fast to establish. This park grass has high demands for light and nitrogen and can inhibit the germination of other species present in its rhizosphere (Li et al. 2016). It was not found in *ex situ* turfs in both habitats and less abundantly in the shaded forest grid, as light was probably a limiting factor for *D. glomerata* to colonize. Meanwhile, *D. glomerata* can be inhibited by competitive allelochemical compounds produced by *Trifolium repens* (Macfarlane et al. 1982), which was common in both the shaded and open park grids, more abundant in the shaded grid which may also have contributed to reducing its establishment.

The number of winners and losers were almost equal in *ex situ* forest turfs (18 winners, 14 losers), and *in situ* forest turfs (14 winners, 16 losers). In contrast to what was expected, the dwarf shrub *Vaccinium myrtillus* increased in *in situ* and *ex situ* turfs compared to the untouched donor. Meanwhile, *Vaccinium myrtillus* decreased in bulk soil transfers as expected, as the old-growth structures did not re-establish rapidly after disruption, however small resprouts were observed in both *ex situ* and *in situ* forest bulk soils. In shaded areas, *V. myrtillus* can have a competitive advantage over its natural competitors *Avenella flexuosa* and *Agrostis capillaris* (Hester 1991). *Agrostis sp* and *Avenella flexuosa* increased in *in situ* forest turfs but were respectively decreasing and neutral in *ex situ* turf transfers. Thereby, the observed decrease of these two grasses when being moved from the logged sun-exposed forest area and placed in the shaded forest grid can be due to the increase of *V. myrtillus* when. Nevertheless, *A. flexuosa* and *Agrostis sp* were in general negatively affected by soil disturbances and decreased in both forest and grassland *in situ* and *ex situ* bulk soils. Grassland turfs (*in situ* and *ex situ*) projected a decrease of the two grasses, but in a lesser extent than in bulk soils. In interaction with shading, the presence of neighbours in turfs could have facilitated seedling recruitment and growth of dwarf shrubs and grasses, while the less sheltered bulk soil habitats may have enhanced seedling mortality rates, as found in other studies (Collins et al. 1995; Gough 2006; Eckstein 2011).

Translocation of species into urban areas can save species from regional or global extinction but does not necessarily save their ecological function (Luna et al. 2018). In this study, the threatened grass *Briza media* was not translocated or found in excavated plots. This is a grass species commonly found in nationally threatened nature types, such as hay meadows and coastal meadows. However, *B. media* was observed along the edges of one *ex situ* turf, but outside the frame used for vegetation analysis. This threatened grass performs poorly on phosphorous- and nitrogen-rich soils, (Solstad et al. 2021), does not have a persistent seed bank, (Grime et al. 1988) and has also been challenging to restore in other topsoil translocation studies (Vecrin & Muller 2003). It remains to be seen whether these few *B. media* will be able to survive or even thrive in the park on this translocated soil.

### **Limitations of the practice of soil translocation**

Over the project stages, there could be several factors that may have influenced the vegetation outcomes, as there are many caveats to equally treating 32 individual pieces of turf and bulk topsoil. Missing EIV-values may have limited the interpretation of the treatment responses to community weighted means, as there were no applicable values for species identified only to genus. A survey of all the plots treated as turfs and as bulk soils prior excavation could have provided more sampling practice, as well as knowledge about the initial plant community of the bulk or turf prior to excavation.

According to Bulot et al. (2017) and McLean (2003), *in situ* conservation is much more favourable than topsoil translocation and highlights the shortcomings of the method. Meanwhile, the most successful and needed translocations have been found for ecosystems with slow soil formation (Douterluge et al. 2018) and species with a higher competitive ability (Rudolph et al. 2017). Worthington & Helliwell (1987) argued that the expensive procedures of translocation efforts can develop a market and economic activity to mitigate impacts on nature. Nevertheless, the concept of topsoil translocation into urban areas should never justify infrastructural development in species-rich areas, but rather be a tool to compensate and tackle the dual problem of species-poor urban areas and adverse roadside effects on plant communities.

## Conclusion

This experiment described how two topsoil translocation methods can remediate species richness and introduce novel plant communities in an urban park slope. Translocation with bulk soils did not recover plant community compositions in either the forest or grassland four months after transfer, and significantly reduced forest understorey plant species richness. Forest understorey community composition did not change by turf translocation, both when inserted into a nutrient-rich urban park, or when remaining at the site of origin. Some species increased in abundance when being moved in turfs, which is suggested to be due to seed bank emergence or a positive effect of shade application for forest topsoil transferred to the park. Grassland turfs in situ and ex situ restored the same number of species as the untouched target sites, but presented novel plant communities, different from the untouched communities. Reflected by CWM Ellenberg indicator values, translocated donor topsoil communities were not affected by the higher levels of nutrients and pH reflected by the park vegetation. Meanwhile, light-demanding species were found to dominate in bulk soils excavated and put back at the donor site, while nutrient-demanding park species were found to immigrate into transplanted bulk soils, whereas turfs were less colonized. Our results suggests that turf transplants can be more resilient against receptor site immigrations, while both bulk soil and turf transplants can maintain donor site soil conditions, even in the face of receptor-donor mismatch of nutrients and pH levels. Future research should aim for long-term monitoring of restoration projects and explore species-specific and ecosystem-based approaches to broaden the toolbox of restoration ecology to remediate species-poor areas.

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## Appendix A

### Park preparation and park species

The topsoil in the park was removed to avoid resprouting of park species underneath the translocated donor soils (fig 1). The park species were not of main interest, but the overall treatment effect of bulk and turf transfers were examined to unravel what species were present in the park, both in the standing vegetation and available in the seed bank for germination (fig 2).



*Figure A1:* Stagewise procedure per grid: a) 8 park turf and 8 bulk were excavated. b) Remaining topsoil was stripped and excess soil from hole digging was placed alongside the grids, avoiding the control plot areas. c) Both grids were smoothed out with shovels, cleaned for rocks and remaining vegetation. The grid was covered with soil cloths four days later. d) Nine days in, all park and *ex situ* plots were inserted.

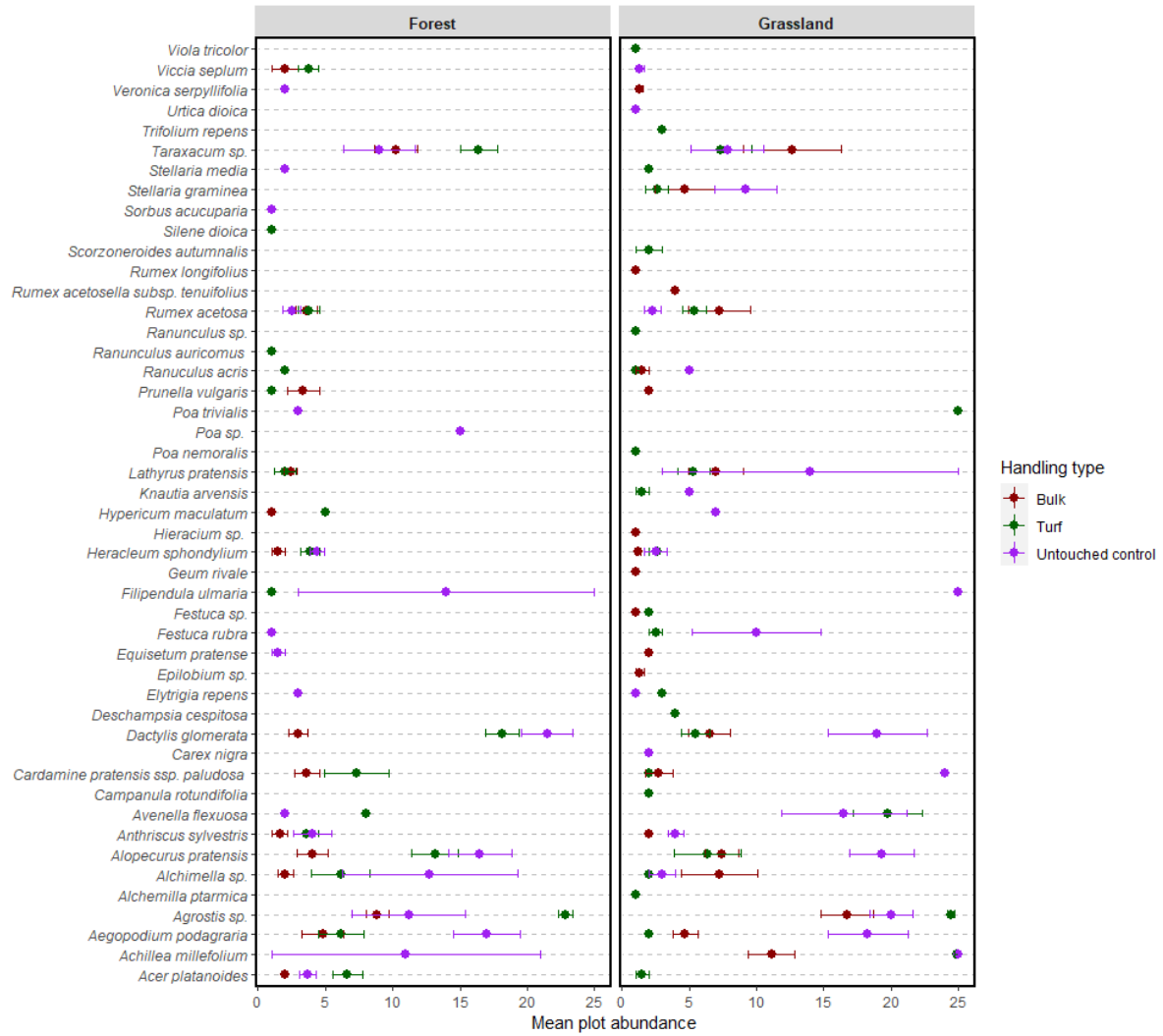


Figure A2: Mean species abundances of park species, presented as the observed lower or higher number of observations to untouched park plots in the forest and grassland park grids.

### Grassland grid bulk soils

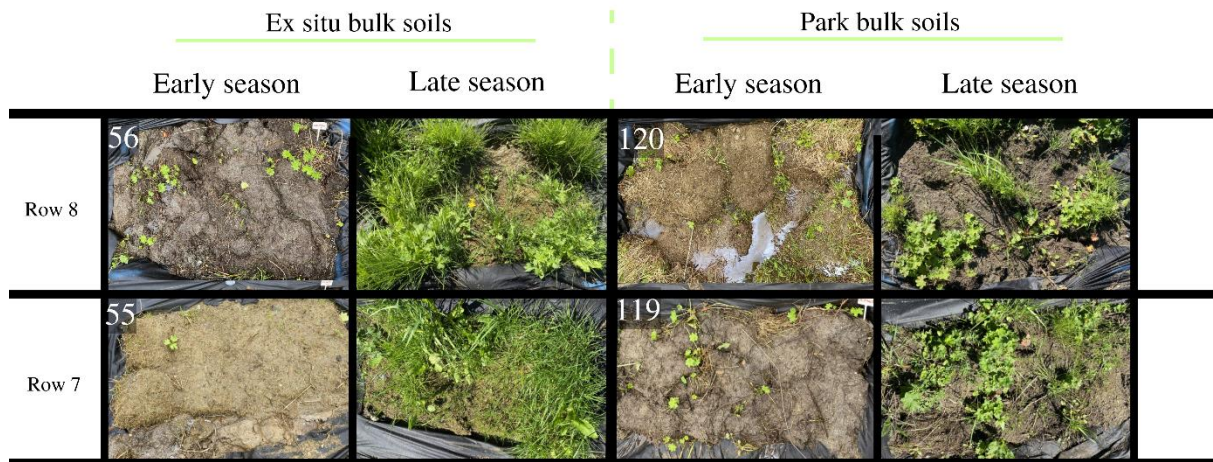


Figure A3: Example of establishment of vegetation cover in grassland bulk soils from early to late season.



## Appendix B

### Species occurrences and mean species richness

Table B1: Estimated mean, standard error (SE) and chosen model type for the number of species and accumulated number of occurrences per plot for the eight treatment groups in the forest and grassland. The maximum number of occurrences per species per plot was 25.

Model	Number of species	Forest			Grassland		
		Est. Mean	SE		Est. Mean	SE	
GLM	Bulk park	8,00	± 1,00		11,25	± 1,19	
	Bulk <i>in situ</i>	5,25	± 0,81		5,13	± 0,80	
Poisson	Bulk <i>ex situ</i>	7,75	± 0,98		9,12	± 1,07	
	Turf park	10,13	± 1,12		9,87	± 1,11	
	Turf <i>in situ</i>	18,38	± 1,52		11,75	± 1,21	
	Turf <i>ex situ</i>	17,50	± 1,48		14,13	± 1,33	
	Untouched donor	20,75	± 1,61		14,75	± 1,36	
	Untouched park	8,75	± 1,05		9,25	± 1,08	

Table B2: Total species abundances in the forest treatment groups with late and early season analyses combined. Species names follow Lid & Lid (2005).

Species	Growth form	Total abundance	<i>In situ</i> bulk	<i>Ex situ</i> bulk	Park bulk	<i>In situ</i> turf	<i>Ex situ</i> turf	Park turf	Un-touched donor	Un-touched park
<i>Acer platanoides</i>	Tree	77	-	-	2	-	-	53	-	22
<i>Achillea millefolium</i>	Forb	190	1	-	-	52	38	-	77	22
<i>Aconitum lycoctonum</i>	Forb	3	-	-	-	1	-	-	2	-
<i>Aegopodium podagraria</i>	Forb	187	-	-	24	-	1	43	-	119
<i>Agrostis sp.</i>	Grass	926	9	111	62	197	132	183	176	56
<i>Ajuga pyramidalis</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Alchemilla ptarmica</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Alchimella sp.</i>	Forb	135	1	1	6	2	11	43	20	51
<i>Alopecurus pratensis</i>	Grass	223	-	-	12	-	-	79	-	132
<i>Anemone nemorosa</i>	Forb	69	-	-	-	21	22	-	26	-
<i>Anthoxanthum odoratum</i>	Grass	100	14	-	-	36	11	-	39	-
<i>Anthriscus sylvestris</i>	Forb	64	-	1	8	-	2	25	-	28
<i>Avenella flexuosa</i>	Grass	324	8	8	-	125	83	8	90	2
<i>Betula pubescens</i>	Tree	27	-	-	-	13	3	-	11	-
<i>Bistorta vivipara</i>	Forb	97	-	16	-	36	8	-	37	-
<i>Briza media</i>	Grass	2	-	-	-	-	-	-	2	-
<i>Campanula rotundifolia</i>	Forb	2	-	-	-	-	-	-	2	-
<i>Cardamine pratensis ssp. paludosa</i>	Forb	62	-	-	18	-	-	44	-	-
<i>Carex nigra</i>	Sedge	0	-	-	-	-	-	-	-	-
<i>Carex pallescens</i>	Sedge	36	4	2	-	2	3	-	25	-
<i>Carex panicea</i>	Sedge	4	-	-	-	-	4	-	-	-

<i>Carex pilulifera</i>	Sedge	41	1	-	-	10	9	-	21	-
<i>Carex vaginata</i>	Sedge	1	-	-	-	-	-	-	1	-
<i>Cirsium heterophyllum</i>	Forb	1	-	-	-	1	-	-	-	-
<i>Cirsium palustre</i>	Forb	7	-	-	-	-	7	-	-	-
<i>Dactylis glomerata</i>	Grass	330	-	1	12	-	-	145	-	172
<i>Deschampsia cespitosa</i>	Grass	226	22	-	-	59	47	-	98	-
<i>Elytrigia repens</i>	Grass	5	-	2	-	-	-	-	-	3
<i>Empetrum nigrum</i>	Dwarf shrub	23	-	-	-	-	-	-	23	-
<i>Epilobium sp.</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Equisetum pratense</i>	Fern	16	-	-	-	6	4	-	3	3
<i>Euphrasia sp.</i>	Forb	31	-	-	-	-	4	-	27	-
<i>Festuca rubra</i>	Grass	60	-	-	-	21	20	-	18	1
<i>Festuca sp.</i>	Grass	0	-	-	-	-	-	-	-	-
<i>Filipendula ulmaria</i>	Forb	30	-	1	-	-	-	1	-	28
<i>Fragaria spp.</i>	Forb	4	-	-	-	3	-	-	1	-
<i>Galeopsis speciosa</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Galium uliginosum</i>	Forb	22	-	-	-	-	8	-	14	-
<i>Gentianales boreale</i>	Forb	15	-	1	-	6	-	-	8	-
<i>Geranium sylvaticum</i>	Forb	135	10	11	-	42	39	-	33	-
<i>Geum rivale</i>	Forb	1	1	-	-	-	-	-	-	-
<i>Gymnocarpium dryopteris</i>	Fern	7	-	-	-	1	-	-	6	-
<i>Heracleum sphondylium</i>	Forb	62	-	1	3	-	-	27	-	31
<i>Hieracium sp.</i>	Forb	16	-	7	-	3	5	-	1	-
<i>Hypericum maculatum</i>	Forb	6	-	-	1	-	-	5	-	-
<i>Knautia arvensis</i>	Forb	1	-	-	-	-	1	-	-	-
<i>Lathyrus pratensis</i>	Forb	31	-	-	17	-	4	10	-	-
<i>Leontodon autumnalis</i>	Forb	3	-	-	-	-	-	-	3	-
<i>Leucanthemum vulgare</i>	Forb	5	-	-	-	-	-	-	5	-
<i>Luzula multiflora</i>	Rush	31	-	-	-	5	12	-	14	-
<i>Luzula pilosa</i>	Rush	114	-	1	-	48	31	-	34	-
<i>Lysimachia europea</i>	Forb	60	-	-	-	32	16	-	12	-
<i>Maianthemum bifolium</i>	Forb	127	-	-	-	49	48	-	30	-
<i>Melampyrum pratense</i>	Forb	20	-	-	-	7	10	-	3	-
<i>Molinia caerulea</i>	Grass	0	-	-	-	-	-	-	-	-
<i>Myostis sp.</i>	Forb	34	-	-	-	10	-	-	24	-
<i>Nardus stricta</i>	Grass	53	-	-	-	-	28	-	25	-
<i>Omalotheca norvegica</i>	Forb	4	-	-	-	-	-	-	4	-
<i>Orthilia secunda</i>	Forb	17	-	-	-	4	9	-	4	-
<i>Oxalis acetosella</i>	Forb	63	1	2	-	25	16	-	19	-
<i>Paris quadrifolia</i>	Forb	5	-	-	-	3	-	-	2	-
<i>Phleum alpinum</i>	Grass	5	-	-	-	1	-	-	4	-
<i>Phleum pratense</i>	Grass	8	-	-	-	1	-	-	7	-
<i>Pilosella aurantiaca</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Pilosella officinarum</i>	Forb	6	1	-	-	-	-	-	5	-
<i>Plantago major</i>	Forb	1	-	-	-	-	-	-	1	-
<i>Poa glauca</i>	Grass	12	-	-	-	6	2	-	4	-
<i>Poa nemoralis</i>	Grass	17	-	-	-	-	17	-	-	-
<i>Poa sp.</i>	Grass	17	-	-	-	-	2	-	-	15



<i>Poa trivialis</i>	Grass	36	-	-	-	-	29	-	4	3
<i>Potentilla erecta</i>	Forb	284	2	34	-	83	85	-	80	-
<i>Prunella vulgaris</i>	Forb	24	-	1	10	1	8	1	3	-
<i>Ranunculus acris</i>	Forb	28	-	2	-	2	12	2	10	-
<i>Ranunculus auricomus</i>	Forb	1	-	-	-	-	-	1	-	-
<i>Ranunculus repens</i>	Forb	6	6	-	-	-	-	-	-	-
<i>Ranunculus sp.</i>	Forb	9	-	1	-	-	6	-	2	-
<i>Rhinanthus angustifolius</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Rubus idaeus</i>	Forb	2	-	-	-	-	2	-	-	-
<i>Rubus saxatilis</i>	Forb	1	-	-	-	1	-	-	-	-
<i>Rumex acetosa</i>	Forb	65	-	2	25	5	3	19	1	10
<i>Rumex acetosella subsp. tenuifolius</i>	Forb	1	-	-	-	-	1	-	-	-
<i>Rumex longifolius</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Sagina sp.</i>	Forb	16	-	-	-	-	4	-	12	-
<i>Salix caprea</i>	Tree	13	1	-	-	1	8	-	3	-
<i>Schedonorus giganteus</i>	Grass	0	-	-	-	-	-	-	-	-
<i>Scorzoneroides autumnalis</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Sibbaldia procumbens</i>	Forb	1	-	1	-	-	-	-	-	-
<i>Silene dioica</i>	Forb	4	-	-	-	3	-	1	-	-
<i>Solidago virgaurea</i>	Forb	12	6	-	-	3	-	-	3	-
<i>Sorbus acucuparia</i>	Tree	15	-	-	-	8	2	-	4	1
<i>Stellaria graminea</i>	Forb	32	-	-	-	21	4	-	7	-
<i>Stellaria media</i>	Forb	31	-	-	-	24	4	-	1	2
<i>Taraxacum sp.</i>	Forb	303	11	9	82	4	12	131	-	54
<i>Trifolium pratense</i>	Forb	13	-	-	-	3	-	-	10	-
<i>Trifolium repens</i>	Forb	134	-	2	-	42	40	-	50	-
<i>Trisetum spicatum</i>	Grass	2	-	-	-	-	-	-	2	-
<i>Urtica dioica</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Vaccinium myrtillus</i>	Dwarf shrub	233	3	17	-	92	79	-	42	-
<i>Vaccinium vitis-idaea</i>	Dwarf shrub	153	8	14	-	46	33	-	52	-
<i>Veronica chamaedrys</i>	Forb	130	5	3	-	76	21	-	25	-
<i>Veronica officinalis</i>	Forb	105	-	-	-	30	29	-	46	-
<i>Veronica serpyllifolia</i>	Forb	38	4	-	-	4	18	-	10	2
<i>Vicia sepium</i>	Forb	19	-	-	4	-	-	15	-	-
<i>Vicia cracca</i>	Forb	6	-	2	-	2	-	-	2	-
<i>Viola biflora</i>	Forb	89	-	5	-	19	29	1	35	-
<i>Viola palustris</i>	Forb	27	1	-	-	14	5	-	7	-
<i>Viola riviniana</i>	Forb	4	1	-	-	-	-	-	3	-

Table B3: Total species abundances in the grassland treatment groups with late and early season analyses combined. Species names follow Lid & Lid (2005).

Species	Growth form	Total abundance	In situ bulk	Ex situ bulk	Park bulk	In situ turf	Ex situ turf	Park turf	Un-touched donor	Un-touched park
<i>Acer platanoides</i>	Tree	6	-	-	-	-	-	6	-	-
<i>Achillea millefolium</i>	Forb	911	10	44	89	121	168	199	155	125
<i>Aconitum lycoctonum</i>	Forb	9	3	-	-	4	-	-	2	-
<i>Aegopodium podagraria</i>	Forb	163	-	-	33	-	-	2	-	128
<i>Agrostis sp.</i>	Grass	1114	61	15-	134	185	198	196	200	140
<i>Ajuga pyramidalis</i>	Forb	4	-	-	-	1	-	-	3	-
<i>Alchemilla ptarmica</i>	Forb	12	-	-	-	-	1	1	10	-
<i>Alchimella sp.</i>	Forb	132	6	-	29	33	34	6	18	6
<i>Alopecurus pratensis</i>	Grass	245	-	16	37	-	38	38	-	116
<i>Anemone nemorosa</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Anthoxanthum odoratum</i>	Grass	39	-	-	-	22	-	-	17	-
<i>Anthriscus sylvestris</i>	Forb	47	-	12	2	15	6	-	-	12
<i>Avenella flexuosa</i>	Grass	340	-	1	-	14	1	158	100	66
<i>Betula pubescens</i>	Tree	1	-	-	-	-	-	-	1	-
<i>Bistorta vivipara</i>	Forb	4	-	-	-	-	-	-	4	-
<i>Briza media</i>	Grass	0	-	-	-	-	-	-	-	-
<i>Campanula rotundifolia</i>	Forb	2	-	-	-	-	-	2	-	-
<i>Cardamine pratensis ssp. paludosa</i>	Forb	37	-	-	11	-	-	2	-	24
<i>Carex nigra</i>	Sedge	2	-	-	-	-	-	-	-	2
<i>Carex pallescens</i>	Sedge	1	-	-	-	-	1	-	-	-
<i>Carex panicea</i>	Sedge	0	-	-	-	-	-	-	-	-
<i>Carex pilulifera</i>	Sedge	0	-	-	-	-	-	-	-	-
<i>Carex vaginata</i>	Sedge	0	-	-	-	-	-	-	-	-
<i>Cirsium heterophyllum</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Cirsium palustre</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Dactylis glomerata</i>	Grass	208	-	16	26	-	-	33	-	133
<i>Deschampsia cespitosa</i>	Grass	203	4	5	-	60	73	4	57	-
<i>Elytrigia repens</i>	Grass	34	24	-	-	-	6	3	-	1
<i>Empetrum nigrum</i>	Dwarf shrub	0	-	-	-	-	-	-	-	-
<i>Epilobium sp.</i>	Forb	4	-	-	4	-	-	-	-	-
<i>Equisetum pratense</i>	Fern	2	-	-	2	-	-	-	-	-
<i>Euphrasia sp.</i>	Forb	38	1	-	-	-	-	-	37	-
<i>Festuca rubra</i>	Grass	84	-	-	-	27	2	5	-	50
<i>Festuca sp.</i>	Grass	6	-	-	1	1	-	2	2	-
<i>Filipendula ulmaria</i>	Forb	25	-	-	-	-	-	-	-	25
<i>Fragaria spp.</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Galeopsis speciosa</i>	Forb	1	-	1	-	-	-	-	-	-
<i>Galium uliginosum</i>	Forb	1	-	-	-	1	-	-	-	-
<i>Gentianales boreale</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Geranium sylvaticum</i>	Forb	58	10	-	-	15	5	-	28	-
<i>Geum rivale</i>	Forb	3	-	1	1	-	1	-	-	-

<i>Gymnocarpium dryopteris</i>	Fern	0	-	-	-	-	-	-	-	-
<i>Heracleum sphondylium</i>	Forb	24	-	-	6	-	-	8	-	10
<i>Hieracium sp.</i>	Forb	3	-	-	1	1	-	-	1	-
<i>Hypericum maculatum</i>	Forb	7	-	-	-	-	-	-	-	7
<i>Knautia arvensis</i>	Forb	33	-	2	-	11	2	3	10	5
<i>Lathyrus pratensis</i>	Forb	100	-	-	56	-	-	16	-	28
<i>Leontodon autumnalis</i>	Forb	31	1	2	-	1	7	-	20	-
<i>Leucanthemum vulgare</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Luzula multiflora</i>	Rush	44	-	-	-	5	3	-	36	-
<i>Luzula pilosa</i>	Rush	0	-	-	-	-	-	-	-	-
<i>Lysimachia europea</i>	Forb	2	-	-	-	2	-	-	-	-
<i>Maianthemum bifolium</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Melampyrum pratense</i>	Forb	6	-	-	-	-	-	-	6	-
<i>Molinia caerulea</i>	Grass	1	-	-	-	-	1	-	-	-
<i>Myostis sp.</i>	Forb	1	-	1	-	-	-	-	-	-
<i>Nardus stricta</i>	Grass	0	-	-	-	-	-	-	-	-
<i>Omalotheca norvegica</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Orthilia secunda</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Oxalis acetosella</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Paris quadrifolia</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Phleum alpinum</i>	Grass	35	-	-	-	12	2	-	21	-
<i>Phleum pratense</i>	Grass	37	-	25	-	1	9	-	2	-
<i>Pilosella aurantiaca</i>	Forb	4	-	-	-	4	-	-	-	-
<i>Pilosella officinarum</i>	Forb	3	-	-	-	-	3	-	-	-
<i>Plantago major</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Poa glauca</i>	Grass	5	-	-	-	-	5	-	-	-
<i>Poa nemoralis</i>	Grass	6	-	-	-	-	5	1	-	-
<i>Poa sp.</i>	Grass	2	-	-	-	-	2	-	-	-
<i>Poa trivialis</i>	Grass	35	-	-	-	1	-	25	9	-
<i>Potentilla erecta</i>	Forb	1	-	-	-	-	1	-	-	-
<i>Prunella vulgaris</i>	Forb	4	-	-	2	-	2	-	-	-
<i>Ranunculus acris</i>	Forb	402	19	46	3	95	98	1	135	5
<i>Ranunculus auricomus</i>	Forb	38	-	12	-	8	12	-	6	-
<i>Ranunculus repens</i>	Forb	323	41	26	-	119	96	-	41	-
<i>Ranunculus sp.</i>	Forb	6	-	3	-	1	-	1	1	-
<i>Rhinanthus angustifolius</i>	Forb	4	-	-	-	4	-	-	-	-
<i>Rubus idaeus</i>	Forb	9	-	-	-	1	8	-	-	-
<i>Rubus saxatilis</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Rumex acetosa</i>	Forb	584	45	78	58	115	136	43	100	9
<i>Rumex acetosella subsp. tenuifolius</i>	Forb	10	-	2	4	4	-	-	-	-
<i>Rumex longifolius</i>	Forb	16	-	14	1	1	-	-	-	-
<i>Sagina sp.</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Salix caprea</i>	Tree	0	-	-	-	-	-	-	-	-
<i>Schedonorus giganteus</i>	Grass	14	-	12	-	-	2	-	-	-
<i>Scorzoneroidea autumnalis</i>	Forb	41	-	2	-	5	14	6	14	-
<i>Sibbaldia procumbens</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Silene dioica</i>	Forb	3	-	-	-	-	-	-	3	-

<i>Solidago virgaurea</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Sorbus acucuparia</i>	Tree	0	-	-	-	-	-	-	-	-
<i>Stellaria graminea</i>	Forb	210	-	22	14	11	34	13	61	55
<i>Stellaria media</i>	Forb	28	-	-	-	3	3	2	20	-
<i>Taraxacum sp.</i>	Forb	319	21	29	76	19	47	59	13	55
<i>Trifolium pratense</i>	Forb	11	-	-	-	-	4	-	7	-
<i>Trifolium repens</i>	Forb	206	-	-	-	22	69	3	112	-
<i>Trisetum spicatum</i>	Grass	0	-	-	-	-	-	-	-	-
<i>Urtica dioica</i>	Forb	1	-	-	-	-	-	-	-	1
<i>Vaccinium myrtilis</i>	Dwarf shrub	0	-	-	-	-	-	-	-	-
<i>Vaccinium vitis-idaea</i>	Dwarf shrub	0	-	-	-	-	-	-	-	-
<i>Veronica chamaedrys</i>	Forb	108	-	-	-	14	29	-	65	-
<i>Veronica officinalis</i>	Forb	4	4	-	-	-	-	-	-	-
<i>Veronica serpyllifolia</i>	Forb	54	4	31	9	7	-	-	3	-
<i>Vicia sepium</i>	Forb	4	-	-	-	-	-	-	-	4
<i>Vicia cracca</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Viola biflora</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Viola palustris</i>	Forb	33	-	6	-	5	22	-	-	-
<i>Viola riviniana</i>	Forb	0	-	-	-	-	-	-	-	-
<i>Viola tricolor</i>	Forb	1	-	-	-	-	-	1	-	-



## Appendix C

### Pairwise comparisons of community compositions

Table C1: Pairwise PERMANOVA with 999 permutations for means of species compositions for handling, translocation type and their interaction in the forest and grassland.

Analysis of similarities (ADONIS)	Df.	Sums of squares	F	R <sup>2</sup>	p-value	
<b>Forest</b>						
Untouched control vs. Bulk	1	1,81	5,79	0,13	0,0001	**
Untouched control vs. Turf	1	0,51	1,90	0,05	0,09	ns.
Bulk vs. Turf	1	2,07	0,13	0,14	0,0002	**
Park to park vs. Donor to park	1	3,17	16,31	0,30	0,0002	**
Park vs. <i>In situ</i>	1	4,46	18,59	0,28	0,0002	***
<i>Ex situ</i> vs. <i>In situ</i>	1	0,79	2,78	0,07	0,002	**
<b>Grassland</b>						
Untouched control vs. Bulk	1	1,39	6,27	0,14	0,0002	***
Untouched control vs. Turf	1	0,52	3,46	0,08	0,005	***
Bulk vs. Turf	1	1,33	7,77	0,14	0,0002	**
Park to park vs. Donor to park	1	1,33	9,06	0,19	0,0002	**
Park vs. <i>In situ</i>	1	1,98	10,52	0,18	0,0002	***
<i>Ex situ</i> vs. <i>In situ</i>	1	0,28	1,47	0,04	0,13	ns.

Table C2: Post-hoc multiple comparisons for translocation and handling type on the variance within groups for species compositions in the two habitats.

Homogeneity of species dispersion	Difference	Lower	Upper	p-value	
<b>Forest</b>					
Untouched control vs. Bulk	0.004	-0.09	0.10	0.99	ns.
Untouched control vs. Turf	0.06	-0.03	0.15	0.27	ns.
Bulk vs. Turf	-0.06	-0.13	0.03	0.25	ns.
Park vs. <i>Ex situ</i>	-0.09	-0.21	0.16	0.11	ns.
<i>Ex situ</i> vs. <i>In situ</i>	-0.05	-0.16	0.05	0.49	ns.
Park vs. <i>In situ</i>	-0.15	-0.25	-0.05	0.001	***
<b>Grassland</b>					
Untouched control vs. Bulk	-0.03	-0.08	0.03	0.47	ns.
Untouched control vs. Turf	0.06	0.01	0.12	0.01	*
Bulk vs. Turf	-0.09	-0.13	-0.04	<0.0001	***
Park vs. <i>Ex situ</i>	0.01	-0.11	0.01	0.96	ns.
<i>Ex situ</i> vs. <i>In situ</i>	-0.06	-0.13	0.01	0.11	ns.
Park vs. <i>In situ</i>	-0.05	-0.11	0.01	0.12	ns.



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