Norwegian University
of Life Sciences

Master's Thesis 2019 60 ECTS

Faculty of Environmental Sciences and Natural Resource Management (MINA)

Evaluation of restoration success for semi-natural and flower meadows in the Oslo municipality

Alexius Folk

Ecology

Acknowledgements

This thesis has been an exercise in familiarizing myself with the academic environment of Norway, as well as learning to navigate the array of region-specific resources available online and off. I have also gained a great deal of practical experience and competence in using statistics programs and in identifying Norwegian flora.

I would like to thank my advisor, Dr. Jonathan Colman, for his guidance and for allowing me to take on this project, as well as for helping me find appropriate people to contact for information and permissions, for including me in the wider process of designing this study and collaborating with the other researchers and organizations involved, and – when necessary – giving me the space to find my own solutions. Thanks also to Dr. Diress Tsegaye Alemu, who helped extensively with the statistics for this project and who dedicated several afternoons to the work of teaching me how to use R by myself. I must also thank Leif Ryvarden for his help in giving me a starting point from which to start identifying Norwegian plants, and especially for helping with the really difficult plants that consisted of only a few leaves.

I am grateful also to the soil science laboratory of NMBU, whose technicians regularly assist graduate students in learning how to perform their own soil analyses as well as providing a substantially discounted rate for them to do so. For some reason MINA and Jordfaget had not been working together before this, so we're excited to have started collaborating again.

I would also like to thank Bymiljøetaten, especially Bård Bredesen, who suggested some of the sites and gave us information about them for this study. Additionally, thank you to Tor Øystein Olsen of Maridalens Venner, Torbjørn Hansen from Bærum Vest, and the other people involved in Bærum kommune's park network, who answered all of our questions on the history and upkeep of their meadows, and who also have excellent information about their work available online. Additionally, I would like to thank the people of Fornebu S, Stovner Senter, and the apartment block at Lillestrøm for allowing us access to their private properties for research, and for answering questions about their maintenance regimes.

Finally, I must thank my friends and family for their cheerleading and researcher-herding, especially my partner Mikkjel, who occasionally acted as my secretary; our friend Trond Moe, who constructed and donated the wooden frames used for plant surveying; and finally Jennifer Robson-Trønnes and my parents, for investing in my education.

Thank you all,

Alexius C.D. Folk
Oslo, Norway

Table of Contents

Abstract

While cities are often overlooked as targets for conservation, ecologists have argued that the urban environment is actually a good candidate for re-creating semi-natural meadows, since these nutrient-poor grasslands prefer dry, well-drained soils and are adapted to human disturbance. Although semi-natural meadows are potentially vulnerable to eutrophication and domination by non-target plant species, they are also capable of becoming very species-rich. A number of these meadows have therefore been created or restored in Oslo, Norway in recent years, and this study evaluates the success of these projects.

We surveyed vascular plants (excluding Poaceae) at 11 sites, including 3 calcareous meadows, 3 hay meadows, and 5 generic flower meadows. Despite a severe drought in the growing season of 2018, the majority of our sites appeared to be relatively resilient. On the other hand, the extreme impact to one rooftop meadow showed the importance of adaptive management even for meadows that are expected to be self-sufficient after establishment. We found that, in line with other studies, undisturbed soil was less likely to recruit non-native plants. Commercial soil was not worse than soil from local donor meadows in this regard, but local soil was potentially better able to recruit target species through the preserved seed bank. Non-native plants were additionally associated with high pH and phosphate. Phosphate in particular was high in some sites, possibly due to soil geology, which indicates that pre-restoration soil testing may be necessary.

We found that re-created meadows were generally much more alkaline than reference meadows, suggesting that the project organizer over-estimated the target pH for these sites. We also found that the reference sites were less biodiverse than new sites, possibly due to the exclusion of mosses and lichens from the data, insufficient sampling, or because new sites will lose species over time. Plant family abundances showed clear divisions based on the project organizer responsible for the site, demonstrating the importance of this top-down effect, but were still differentiated based on meadow type. Importantly, we found that the techniques used in this study were straightforward and useful, with some suggestions for improvement.

Introduction

The urban environment as a conservation target

The proportion of humanity living in urban areas worldwide grew to more than 50% in 2008, closer to 80% in more developed countries, and is predicted to rise further (United Nations 2012). Urbanization is a threat to biodiversity in many ways, including severe land-use changes, habitat loss, and fragmentation, as well as invasive species introduction from gardens and imports, widespread impermeable surfaces restricting water supply and plant growth, and an abundance of pollutants (Hansen *et al.* 2005). The urban environment, however, is also home to surprising biodiversity (Breuste *et al.* 2008), and while some of this is due to the wide variety of introduced species, the majority of life in the urban environment actually consists of species native to the region (Pyšek 1998). This is because humans tend to build cities in areas of pre-existing biodiversity (Kowarik 2011), and also because the fragmented nature of cities results in a highly heterogenous environment in a constant state of successional tension, which tends to increase species richness (Pyšek 1998, Connell 1978).

Landscape-level factors that impact animal biodiversity are not yet well-understood, and differ depending on taxa (Lin & Fuller 2013), with birds and some insect groups over-represented in studies compared to others (Braaker *et al.* 2014). That said, it appears that the plant community in the urban environment, like elsewhere, is the foundation for animal life – avian biodiversity is connected to native trees (Fontana *et al.* 2011, Paker *et al.* 2014), and general floral diversity is tied to that of insects (Shwartz *et al.* 2013, Steffan-Dewenter & Tscharntke 2002). Native plants generally support native animals, and likewise for non-native plants and animals, so regionally-informed urban habitats show promise as refuges for struggling native species (Goddard *et al.* 2010).

There is some debate about the best way in which to conserve habitat in the context of the urban environment, and whether we should focus on sequestering human impacts in dense areas of modification away from more natural areas or whether a more diffuse and integrated landscape can be viable as a conservation goal (Lin & Fuller 2013). Studies have found that humans benefit from the proximity of nature, both through ecosystem services such as oxygen production and recreation (Lin & Fuller 2013), as well as psychologically (Lin & Fuller 2013, Loder 2014), particularly when those green spaces have higher biodiversity (Fuller *et al.* 2007). Increased property values are also associated with the quality of green spaces (Breuste *et al.* 2008).

Among ecologists, the debate is largely centered on what kind of habitat is worth conserving (Kowarik 2011, Goddard *et al.* 2010). Cities, however, are already home to biodiversity, and just as space for building elsewhere is limited, so are our choices for where to compensate for habitat loss (Lin & Fuller 2013). It's already not uncommon for abandoned urban areas to revert to species-rich grasslands without human interference (Klaus 2013). There's also a growing push to construct green roofs, which have historically been intended for cooling and water management (Madre *et al.* 2014), and lately also for air quality improvement and urban heat reduction (Nagase & Dunnett 2010), but also have the potential to greatly increase the total area of habitat in a city (Madre *et al.* 2014). Provided that our goal is not to push for a hypothetical "pristineness" of nature (Kowarik 2011), we have a toolset to promote native urban biodiversity with complex communities, multiple trophic levels, and intact ecosystem functions (Lin & Fuller 2013).

Semi-natural meadows as candidates for urban ecosystems

Nutrient-poor and semi-natural grasslands are home to what may be the highest species richness of any class of temperate grassland (Bobbink & Willems 1991, Steffan-

Dewenter & Tscharntke 2002, Evju *et al.* 2015). These meadows promote the proliferation of specialist species due to the constraints they impose on plant growth, including the lack of nutrients, shallowness of the soil, the resulting dryness, and the regular disturbance encouraged by normal management (Bakker & Berendse 1999, Willems 2001, Willems 1987 in Willems 2001, Evju *et al.* 2015). Historically, nutrient-poor grasslands such as calcareous and hay meadows have been a part of the European landscape for thousands of years. As reviewed by Eriksson *et al.* (2002), these grasslands were used for the grazing of farm animals, hay production, and manure production for use as fertilizer in more arable fields, activities that gradually remove biomass and nutrients and relocate it elsewhere. These habitats are described as "semi-natural" due to their dependence on regular human interference to prevent encroachment by bushes, trees, and dominating grasses (Cousins & Eriksson 2001).

The main threat to these nutrient-poor habitats began with the invention of artificial fertilizer (Eriksson *et al.* 2002). The utility of meadows for hay and grazing became overshadowed by their profitability for crop production, and many meadows were converted into arable land or turned into plantation forests if unsuitable for crops, effectively destroying the soil seed bank and their ability to recover (Eriksson *et al.* 2002). Meanwhile, increased demands for space for roads and buildings created another threat to semi-natural meadows (Pyšek 1998, Evju *et al.* 2015), especially since many meadows are naturally found in open, sunny areas attractive for development (Bratli *et al.* 2005).

In the last half century and lately in Norway, emphasis has been placed on the conservation of these meadows due to both an increased interest in the preservation of history (the "cultural landscape") (Auestad *et al.* 2015) and in the high species richness found in these habitats (Direktoratet for Naturforvaltning 2009, Bratli *et al.* 2005). Ironically, one of the most promising areas of focus for calcareous and hay meadow conservation is arguably the urban environment (Klaus 2013). Roadsides and urban wastelands are often seen as being marginal habitat with little value or potential for conservation, but their tendency to have shallow soil and to be mowed during routine maintenance makes them prime habitat candidates for calcareous and hay meadow plants. Additionally, green roofs seem to be well-suited for the creation of new semi-natural grasslands, since rooftop habitats are characterized by the shallow soils, abundant light, and dryness (Madre *et al.* 2014) that semi-natural meadows require (Bobbink & Willems 1991, Willems 2001). The presence of a diverse plant community, in turn, leads to a diverse insect community, and encourages diversity in resident vertebrates even in urban habitats (Fontana *et al.* 2011, Paker *et al.* 2014, Steffan-Dewenter & Tscharntke 2002).

The urban matrix does present significant, although not insurmountable, challenges for calcareous and hay meadows. Nitrate deposition from fossil fuel burning is a major threat to meadow biodiversity worldwide (Maskell *et al.* 2010), but especially in the urban environment (Knapp *et al.* 2009). This can acidify the soil at high levels (Maskell *et al.* 2010) or promote the proliferation of generalist plants that out-compete specialists at moderate levels (Eriksson *et al.* 2002). However, since one of the functions of semi-natural meadows is to relocate nutrients elsewhere, this may be a self-fixing problem, provided that the risk is considered during the planning and maintenance phases and restoration goals are adjusted accordingly (Bakker *et al*. 2002). Nitrogen deposition is also lower in Norway than in much of the rest of Europe (European Environmental Agency 2017), and therefore potentially less of a concern.

Some of the most successful non-native species in the urban environment are those that are particularly good at establishing themselves in disturbance and marginal areas, a characteristic that, in plants, is referred to as being "ruderal" (Braaker *et al.,* 2014, Donath *et al.* 2003). Newly-restored, disturbed, and/or unmanaged meadows are particularly vulnerable to invasion by ruderal plants, which grow faster than target meadow species and outcompete them for light (Bobbink & Willems 1991, Donath *et al.* 2003). Invasion by ruderals is usually managed through mowing and hay removal, which prevents this shading and removes nutrients (Eriksson et al. 2002), but this requires active yearly maintenance. Unified conservation strategies in cities are rare due to the patchwork of management boundaries and private properties (Goddard *et al.* 2010), which can result in source patches for ruderal and invasive species to spread from. The ultimate aim is that a well-managed semi-natural meadow will be species-rich enough that it becomes resistant to domination by unwanted invaders, plant or otherwise (Auestad *et al.* 2015).

Several studies have found that the plants native to semi-natural meadows are typically poor dispersers (Butaye *et al.* 2005, Bakker & Berendse 1999, Kiehl & Pfadenhauer 2007, and Knapp *et al.* 2009), which is potentially problematic in a landscape characterized by fragmentation. On the other hand, in the Oslofjord region of southern Norway, it seems that calcareous meadows were always at least somewhat fragmented (Evju *et al*. 2015, Bratli *et al.* 2005). Adaptation to this geological circumstance may be why connectivity of remaining meadow patches was shown by Evju *et al.* (2015) to have a significant effect on species richness in those patches, in contrast to analyses in other European countries that found no effect of connectivity because all patches were already too isolated for effective migration (Kiehl & Pfadenhauer 2007). Therefore, dispersal may be inherently less of a problem when restoring semi-natural meadows in southern Norway's urbanized landscape, especially since seeds tend to be collected locally and by hand (Naturrestaurering A.S., *pers. comm.*; Naturhistorisk Museum *pers. comm.*). This ensures that plants are of regional genetic stock, maximizing the anticipated success of target species' germination in light of small differences in climate preferences (Auestad *et al.* 2015).

The challenges necessarily to ensure success seem worth the effort in in areas where semi-natural meadows fit into the historical landscape and native specialists can make use of available niches (Goddard *et al.* 2010). The probability of success for a restored or re-created meadow can be increased in the planning stage while also safeguarding against future changes in management, but in order to know how best to plan, we first need a better understanding of how well current efforts have worked.

Project background

Since these meadows are of high conservation value in Norway (Direktoratet for Naturforvaltning 2009, Bratli *et al.* 2005), interest in the restoration and re-creation of hay and calcareous meadows has peaked locally, both among state-associated organizations (Direktoratet for Naturforvaltning 2009, Bratli *et al.* 2005, City of Oslo 2019) and private restoration consultants. Examples of these include the company Naturrestaurering A.S., co-founded by this researcher's graduate advisor, and Bymiljøetaten, the state-led organization responsible for the physical environment of the Oslo municipality, including ecological restoration projects on public properties. Accordingly, Naturrestaurering A.S. contracted to create a number of semi-natural meadows on private properties in Oslo over the past few years, both on the ground and as green roofs. As a point of comparison, several flower meadows that had been restored or created by Bymiljøetaten were put forward by that organization for analysis. In order to fine-tune and adapt techniques, these project organizers have requested an evaluation of the meadows' status and relative success.

This project focuses on success as measured by high plant biodiversity, low abundances of non-native species, similarity of plant species communities between created sites and reference sites, low soil macronutrient levels, and similar alkalinity to reference sites. We also describe the sites in terms of site area, the origin of the soil, the site maintenance regime (specifically weeding and watering), the site's age, the amount of exposed ground, and the grain size composition of the sites' soil. These characteristics were contrasted against selected plant family abundances, the prevalence of non-native species, and observed species richness and Shannon diversity.

Since the majority of meadow restoration in Oslo has occurred in the past few years, these results will primarily address recent management decisions. In addition to evaluating the success of restoration techniques, we are also testing the suitability of the techniques used in this study for future projects and long-term monitoring.

Hypotheses

Since this was in some ways an exploratory study, we were interested in many variables both explanatory and dependent. Our hypotheses were therefore fairly complex and numerous, so we will not go into an itemized discussion of their results in this paper, but rather pick out the results that were of most relevance to this study's stated goals of evaluating 1.) the restoration/re-creation techniques employed by Naturrestaurering A.S. and Bymiljøetaten, 2.) the usefulness of the methods used in this study for later research, and 3.) further directions for research based on our findings.

In general, our predictions are as follows:

- Plant biodiversity (as measured by Shannon diversity and species richness) will be **higher** in sites that…

  …are watered, are in the "reference" treatment category, are older, have a higher percentage of bare soil, have a higher pH, and that use local or unmoved soil

- Meanwhile, plant biodiversity will be **lower** in sites that…

  …are in the in the "roof" and "planted" treatment category, are in the "flower" meadow type category, are higher in macronutrients, are lower in pH, and that have a higher abundance of non-native plants

- The soil of restored and re-created meadows will contain higher amounts of nitrate, ammonium, and phosphate than reference meadows, and also be lower in pH. We also expect that commercially-obtained soil will be higher in macronutrients, although relatively neutral in pH.
- We expect to see differences in the grain size composition of soil between sites, treatments, meadow types, and soil origins, possibly with a significantly high proportion of sand in semi-natural meadows in general and the reference meadows in particular.
- Some plant families will be more associated with some meadow types than others. We also expect that some plant families will be more or less common in the reference sites than in the re-created semi-natural meadows and may show preferential use by different project organizers. We also expect that plant families will show different responses to soil and site variables.
- The abundance of non-native plants may be higher or lower in sites that were weeded the year prior. We also expect the abundance of non-native plants to increase with greater macronutrient values, lower pH, and lower barrenness of the soil. We expect that sites in the "reference" treatment category and sites with unmoved soil will have a lower abundance of non-native species.

Methods

Site selection

The experimental sites used in this study were chosen based on recommendation by the organizations responsible for their creation or restoration. All hay and calcareous meadows were re-created by Naturrestaurering A.S. Since previous research by Naturrestaurering A.S. used the calcareous meadows at Kalvøya and the hay meadows at Maridalen as reference communities and seed collection sources for those sites, they were also used in this study (*pers. comm.*). Since both reference locations encompass a number of smaller meadows, specific sites were chosen based on accessibility, size, and the presence of community-specific plants.

The remainder of the sites were recommended by Bymiljøetaten. These sites were all restored or created with the intent of becoming flower meadows and providing food resources for pollinators, and were part of a simultaneous study using the same sites to compare resource allocation between honeybees (*Apis mellifera*), wild bees, and other insect pollinators in Oslo (Nielsen *et al.*, *unpublished manuscript*). They were included in this study as both a contrast to the semi-natural meadows and also to compare the effectiveness of restoration methods used by Bymiljøetaten to those of Naturrestaurering A.S., regardless of meadow subtype.

Site boundaries were determined during surveying based on physical characteristics, avoiding areas under dense canopy or bushes, areas managed as lawn, fences, and concrete boundaries. A visual overview of the sites can be viewed in Supplementary Figure 1.

Site characterization

Treatment

Sites were divided into the *Treatment* categories "reference", "restored", "created", "roof", and "planted". "Restored" sites were defined to be those where a previously-established patch of land was altered in order to increase ecosystem function in line with a target habitat type, and in this study consists of flower meadow sites where a variety of characteristic plant species were sown or planted and where foreign species are discouraged through weeding and mowing. "Planted" sites are similar, but were primarily sown with red clover (*Trifolium pratense*) and a select few unspecified flowering plants in order to provide resources for pollinators. It should be noted that all sites in these two categories are defined as flower meadows.

The "created" and "roof" categories are essentially identical in definition except that sites in the "roof" category are located on top of buildings. Otherwise, "created" sites have been made in places that had not previously had plant life, such as a new hillside or a concrete berm filled with substrate. The Mustad site was placed into this category since, unlike the restored sites, the original topsoil there was removed to bedrock and either replaced with donor soil from a nearby calcareous meadow (including seed bank) or left bare and with seeds from target species raked into the remaining soil. It is also the only "created" or "roof" site that shares continuous soil with other wild plant communities. "Roof" sites were made with intentionally varying substrate depths, consisting predominantly of a flat, relatively thin expanse with scattered mounds rising approximately 30cm higher than the surroundings. The predominant vegetation structure of the thinner substrate area was muscinal at Fornebu S and herbaceous at Lillestrøm, following the definitions put forward by Madre *et al.* (2014).

The "reference" sites are considered to be typical examples of their habitat types, and were rehabilitated from relatively undamaged states in recent history. The hay meadows in Maridalen were partially overgrown by trees and bushes prior to the resumption of annual mowing and hay removal starting in 1991, when the interest group Maridalens Venner ("Friends of Maridalen") began a systematic restoration plan for the area (Olsen, *n.d.*). The specific meadow used in this study was Tørrenga, but this paper will refer to the site simply as "Maridalen" hereafter. Maridalen is located in the more rural landscape to the northeast of urban Oslo.

According to park manager Torbjørn Hansen (*pers. comm.*, 2019), the island Kalvøya was used for sheep grazing until 1955, when charlock mustard (*Sinapis arvensis*) farming began and horses and cows took over grazing. A bridge connected Kalvøya to the mainland in 1962, after which it was used as a park. Park management began to cut the grass short in 1972. Reestablishment of the meadows began in 2010 by ceasing lawn maintenance in several areas, cutting down small trees and bushes and resuming annual hay removal. The area used in this study is not one of the meadows that was specifically reestablished, but was selected based on the presence of characteristic calcareous meadow species such as *Galium verum* and *Hylotelephium maximum*.

Table 1. Summary of site characteristics.

Site	Treatment	Type	Soil Origin	Organization	Year of project start	Age	Area (m^2)	% sampled	Weeded (2017)	Watered (2018)
Maridalen	Reference	Hay	Unmoved	Reference	1991	27 *	1290	0.54 %	No	No
Kalvøya	Reference	Calcareous	Unmoved	Reference	2010	8 *	2120	0.33 %	No	No
Mustad	Created †	Calcareous	Local	Naturrestaurering	2017	1	440	1.59 %	Yes	No
Stover S	Created	Hay	Commercial	Naturrestaurering	2016	2	520	1.35 %	Yes	Yes
Fornebu S	Roof	Calcareous	Local	Naturrestaurering	2015	3	1540	0.45 %	No ‡	No ‡
Lillestrøm	Roof	Hay	Commercial	Naturrestaurering	2017	1	480	1.46 %	Yes	No §
Akershus	Restored	Flower	Unmoved	Bymiljøetaten	2015	3	480	1.46 %	Yes	No
Brynseng ¶	Restored	Flower	Unmoved	Unknown	2011	7	1410	0.50 %	No ‡	No ‡
Langkaia	Created	Flower	Commercial	Bymiljøetaten	2017	1	60	11.67 %	No	Yes
Elgsletta	Planted	Flower	Unmoved	Bymiljøetaten	2017	1	720	0.97 %	No	No
Filipstadveien	Planted	Flower	Unmoved	Bymiljøetaten	2017	1	270	2.59 %	No	Yes

* Not quite equivalent to the ages of the other plots; see "Treatment" above.
† "Created" in this case means that the topsoil was removed; see "Treatment" above.
‡ Assumed based on field observations, but unable to contact site management.
§ Unclear – not deliberately watered according to management, but may have benefited from runoff from nearby cultivated plants on the same roof.
¶ Information on project variables obtained secondhand from Bymiljøetaten, through aerial photos, and from field observations.

Other factors

Meadows were determined *a priori* to be either hay, calcareous, or flower meadows according to the restoration goals at project start. The reference sites are in addition officially registered as their respective habitat types by Miljødirektoratet (2018). The *Soil Origin* variable differentiates between projects that either did not move the original soil away from the restoration site ("unmoved"), that used commercially-obtained garden soil without added fertilizer ("commercial"), or that used natural topsoil taken from a local donor site where a meadow of the same type had been excavated for construction ("local").

The year of project start was recorded as the last year in which sowing or planting occurred. In the case of the Brynseng site, for which no direct project information was available, aerial photos were used to determine the date (Terratec 2011, Kartverket 2016). Site areas were measured using the mapping service from Kartverket (2019) by drawing a polygon around the border of each site and recording the area given. For more complex site shapes, multiple polygons were measured and then added or subtracted to each other as necessary. Features such as dense bushes, strands of trees, skylights, built paths, and large trampled areas were either not included or else measured and then subtracted from the site area. Area measurements were then rounded to the nearest $10m^2$.

The managers responsible for the maintenance of their respective sites were surveyed about their practices, including whether the sites were weeded or mown the year before (in 2017), whether the sites were watered in the year of the study (2018), and whether herbicides or fertilizers were ever used. Since only one site used herbicides (Akershus, to prevent plants from growing over the edge of the concrete berm), only one or two sites were not mown (Fornebu S, possibly Brynseng), and no sites used fertilizers, these factors were discarded as variables. In cases where managers could not be contacted or did not respond (i.e. Brynseng and Fornebu S), management practice was inferred based on field conditions. The absence of last year's cut stems was assumed to indicate that the sites had not been mowed, and the widespread presence of wilted plants sprouted that season in addition to a lack of watering implements was assumed to indicate that the sites had not been watered.

Plant surveying

Plant species at each site were surveyed based on their presence or absence within a sample plot. A sample plot consisted of a wooden frame measuring 0.5 x 0.5m on the inside edge laid onto the ground, and each site was sampled 14 times on the same day, once in the beginning of summer (June 13th – July 3rd) and once toward the end of summer (August 11th – September 15th) for a total of 28 times (or $7m^2$ total). The plots were distributed semi-evenly across any given site by counting steps between plots (number dependent on the size of the site), then tossing the wooden frame blindly in order to avoid selectively including plant specimens. Human-related features such as paths and trampled regions were actively avoided, while bare rock was not. Because plots were not marked it's possible that some plants were resampled in the second round of 14 surveys, but due to the relatively small area sampled compared to site size for all but the smallest site (see Table 1, "% sampled") this was not considered to be a significant source of error.

Each angiosperm plant emerging from the ground within the frame's area was identified to species if possible, and in cases where species was uncertain, a botanist was consulted. If a plant's species was still unknown, or if the species of a surveyed plant was later called into question, it was instead assigned to a genus. Grasses (Poaceae) were initially included in the survey but later excluded since some sites were mown partway through the season, removing the identifying organs of most grass species. Each species (or genus) was recorded a maximum of once per plot. Plant species were compared against the Norwegian Red List (Solstad & Elven 2015) to determine their conservation status. In order to make some additional statistical analyses possible, plant species and genera were further condensed into either their family or into the "non-native" category regardless of family, since invasive plants are thought to have some common traits inclining them toward invasiveness (Jogesh *et al.* 2008). Plants were labelled as non-native based on their presence on the Norwegian Foreign Species List (Artsdatabanken 2018) regardless of their relative risk evaluations.

In addition to surveying the plant species, the estimated percentage of barren ground was also recorded for each plot, defined as ground not covered by living plant life (including grasses and mosses).

Soil collection and preparation

Soil samples from each site were collected over a period of approximately 4 hours on Nov. 7, 2018, using a spade to remove 200-250g topsoil from five locations distributed evenly across the meadow. For roof locations, sampling was distributed between areas of deep substrate and areas of shallow substrate. On the day of collection, the Blindern weather station in Oslo reported an average temperature of 6.5°C and 0.1mm precipitation (Meteoroloisk Institutt, 2019). Samples were stored separately in labelled plastic bags, grouped by site, and stored in a freezer at -20°C until prepared for analysis. For each site, one soil sample was randomly excluded from the analysis, leaving 4 soil samples per site for testing (44 in total).

We separated approximately two-thirds of each sample into cartons, dried them at 55°C for more than 72 hours. These were then sifted through a 2.0mm sieve, and the portion of the sample larger than 2.0mm was discarded. The remaining one-third of each sample was kept frozen until analysis.

Soil analyses

pH analysis

For every sample, we proportioned 10ml by volume of dried, sifted soil and 25mL deionized water into 50mL plastic test tubes following Øien & Krogstad (1987). Analysis used a MeterLab PHM210 Standard pH Meter calibrated using 4.00 and 7.00 buffer solution and then tested against a standard water sample after every ten test samples and at the end of the series to ensure accuracy.

Phosphate analysis

In Norway, the standard test for phosphate concentration in water is the AL-method (Øien & Krogstad, 1987), so that was also used in this study. For each sample, 2g of dried, sifted soil were proportioned into HCl-rinsed 100mL glass bottles along with 40mL AL-solution (0.1mol/L ammonium lactate + 0.4mol/L acetic acid). Samples were shaken for 90 minutes and then filtered into 50mL plastic test tubes through 0.2µm Whatman ashless Blue Ribbon filter paper that had been pre-wetted with AL-solution. These samples were analyzed spectrophotometrically at 700nm using the molybdenum blue method. The molybdenum reagent was formulated after Norwegian standard NS 4725 (Miljødirektoratet 2019). Outputs were converted into measurements of mg/kg PO_4-P following Øien & Krogstad (1987).

Dry matter content

From each frozen sample, we scraped a known mass of soil into a numbered ceramic crucible. These were baked in an oven overnight, then re-weighed to calculate the percentage of dry matter in each of the original samples. This was done primarily to correct the nitrate and ammonium measurements for water content, but since the samples were all collected within a short window and under similar weather conditions, this seemed like a potentially useful approximation of soil water retention at the sites. Therefore, we included this measure in the final analysis as well.

Nitrate and ammonium analysis

Frozen samples were thawed for approximately 5 minutes prior to proportioning 10g of soil and 25mL 2M KCl into numbered 100mL glass bottles, along with four additional blank controls containing only KCl. Small stones and large pieces of plant matter were avoided where possible. After agitation, each sample and blank control was filtered into 50mL plastic test tubes through 0.2µm Whatman ashless Blue Ribbon filter paper pre-wetted with 2M KCl. These were then analyzed spectrophotometrically using a FIAstar 5000 Analyzer set up for NO_3/NO_2 and NH_4 analysis. Outputs were corrected for the percentage dry matter of the original samples. Measurements below the minimum observable concentration 0.020 mg/kg NO_3 were reported as 0.020 mg/kg NO_3 for the analysis.

Particle size distribution analysis

Two of the four samples from each site (22 samples) were selected for grain size distribution analysis. From these, 10mL of dry, sifted soil were separated into labelled 1L beakers. These were further prepared and analyzed according to the methods described in Krogstad *et al.* (2018) by the NMBU soil science laboratory, along with two additional soil samples acting as internal standards. During the course of the preparation, organic material was oxidized out of the samples. Clay (<0.002mm) and silt (0.002-0.060mm) particle percentages were determined through sedimentation fractionation and the pipette method, while sand particles (0.060-2.000mm) were fractionated by sieving. Although more precise size categories were returned from the analysis, we restricted our data to the broader percentage categories of clay, silt, and sand.

Statistical analysis

We used general linear models (GLM) to analyze our data, differentiating between data with continuous values (which default to a gaussian distribution) and data with interval values (i.e. species richness and plant family counts), for which we specified a Poisson distribution. Explanatory variables included *Site*, *Treatment* (e.g. “created”, “planted”, etc.), *Type* (“flower”, “hay”, and “calcareous” meadow), *Soil Origin* (“unmoved”, “commercial”, and “local” soil), and *Organization* (“Naturrestaurering A.S.”, “Bymiljøetaten”, “unknown”, and “reference”). All data with integer and interval values were potential response variables for these, except for Julian date. In addition, some of those data (e.g. all soil values, % barren, and Julian date) along with the categories *Weeded* (“yes” or “no”) and *Watered* (“yes” or “no”) were used as potential explanatory variables affecting Shannon diversity and richness.

In addition, we performed principal component analyses (PCAs) on the abundances of plant family and non-native plant groups with the explanatory variables *Site*, *Treatment*, *Type*, *Soil Origin*, and *Organization*. The groups found to have the most directionality were then selected for a new set of PCAs using the same explanatory variables, then tested against the explanatory variables listed in the paragraph above using GLM, with the addition of the three classes of mean percent grain size composition (“% clay”, “% silt”, and “% sand”). Analyses were performed in R version 3.4.1 (RCoreTream 2017).

Results

Plant survey

In total, we identified 116 unique plant species plus 6 genera, with a total observation count n = 1401. Most sites had fewer species when sampled later in the year except for Fornebu S (+11) and Langkaia (+4); despite that, all sites gained species in the second round of surveys that were not present during the first. Species accumulation curves suggest that, for most sites, additional samples would have been required to account for most of the species present. The most numerous species and genera were *T. pratense* (n = 126), *Leucanthemum vulgare* (n = 89), *Achillea millefolium* (n = 81), *Galium* spp. (n = 78), *Taraxacum* spp. (n = 73), and *Silene vulgaris* (n = 68). For a full list, see Supplementary Table 3.

Only three plant species present on the Norwegian Red List were observed, two of which were trees and therefore not strictly meadow species (*Fraxinus excelsior* (VU) and *Ulmus glabra* (VU)). The only red-listed herbaceous species was *Veronica spicata* (VU), a calcareous meadow indicator species observed at only the Kalvøya site. Due to the low numbers, an analysis of red-listed species was not performed.

Table 2. Summarized species/genera count data.

Site	# species 1st round	# species 2nd round	Difference in sample	Species gained	% new species gained	Total # species	# foreign species	Total counted	Total foreign counted	% foreign species
Maridalen	22	20	-2	6	30.0 %	28	0	109	0	0.0 %
Kalvøya	22	21	-1	8	38.1 %	30	1	119	1	0.8 %
Mustad	36	33	-3	8	24.2 %	44	3	191	21	11.0 %
Stover S	25	24	-1	7	29.2 %	32	3	202	6	3.0 %
Fornebu S	6	17	11	13	76.5 %	19	4	72	7	9.7 %
Lillestrøm	31	27	-4	9	33.3 %	40	4	208	17	8.2 %
Akershus	24	21	-3	7	33.3 %	31	4	117	8	6.8 %
Brynseng	25	22	-3	11	50.0 %	36	3	117	3	2.6 %
Langkaia	12	16	4	9	56.3 %	21	2	78	2	2.6 %
Elgsletta	10	7	-3	2	28.6 %	12	1	63	1	1.6 %
Filipstadveien	16	14	-2	2	14.3 %	18	1	125	3	2.4 %

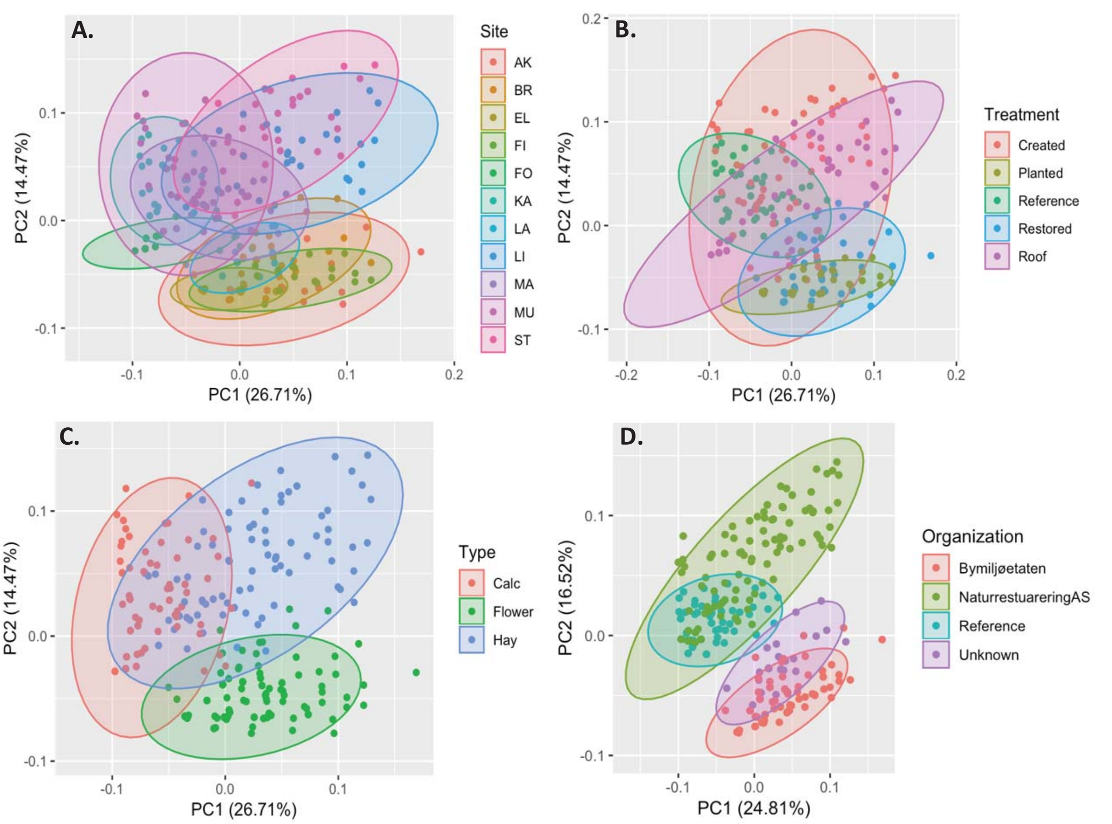


Figure 1. PCA plots of selected plant families and non-native plant observation sorted according to the variables: **A.** *Site*, **B.** *Treatment*, **C.** *Type* of meadow, and **D.** *Organization* responsible for the project. The site abbreviations used in plot **A** correspond to the first two letters of each site's name. Note that all of the plots show the same data, but with clusters assigned according to different variables.

We observed a total of 28 native plant families. The most-observed families were Asteraceae (n = 388), Fabaceae (n = 256), Caryophyllaceae (n = 121), Rubiaceae (n = 111), Rosaceae (n = 88), Crassulaceae (n = 70), and Plantaginaceae (n = 56). Full a full family list, see Supplementary Table 4. In addition, 18 plant species were assigned to the "non-native" category, including 8 species rated as severe risk (SE), 1 species rated as high risk (HI), 6 species rated as potentially high risk (PH), 1 species rated as low risk (LO), 1 species for which there was no known risk (NK), and 1 species that had not been rated (NR) (Artsdatabanken 2018). Of the last three categories, only one species was observed more than once: *Malus ×domestica* (NR), 8 times at Lillestrøm (and indeed, these were all new sprouts from nearby parent trees).

At least one site (Mustad) was weeded over the course of the sampling season and is therefore missing some species that were previously present on site, including the additional two invasive species *Melilotus officinalis* (SE) and *Lupinus polyphyllus* (SE), and greater numbers of both native and non-native unwanted species that were counted as less abundant (*Alliaria petiolata* (LC), for example). Although the site Lillestrøm was also reported as having been weeded, management was primarily for native species in the genera *Taraxacum* and *Urtica*.

PCA analyses comparing counts of plant families and non-native plants to the variables *Treatment*, *Site*, and *Type* revealed that the eight most numerous groups (including non-native plants and those listed above) as well as the family Brassicaceae (n = 31) contributed to the most difference between site communities. Therefore, these nine groups were picked out for visualization in another set of PCAs (Fig. 1), and also analyzed more closely for relationships to other explanatory variables in our study.

Soil characteristics including grain size

Results are for the most part summarized in Table 3 A&B and Supplementary Table 1 A&B. Although pH is often reduced by NO_3 deposition, we did not find a significant correlation between the two in this study (p = 0.0535). The grain size distribution was only significantly related to the variables *Site* (Fig. 2) and *Treatment*, and was not related to *Type* or *Soil Origin*. Only the sites Maridalen, Kalvøya, and Elgsletta had a significantly higher clay content than the others (p = 0.02793, 0.00264, & 0.00420, respectively), as well as a significantly lower sand grain content (p = 0.04589, 0.00261, & 0.00727). Kalvøya and Elgsletta also had significantly higher silt content than the other sites (p = 0.00260 & 0.00945), while Maridalen did not (p = 0.05596). Of the treatments, only the "reference" category was significantly different, with a higher clay content (p = 0.039488) and lower sand content (p = 0.0485) than the other treatments.

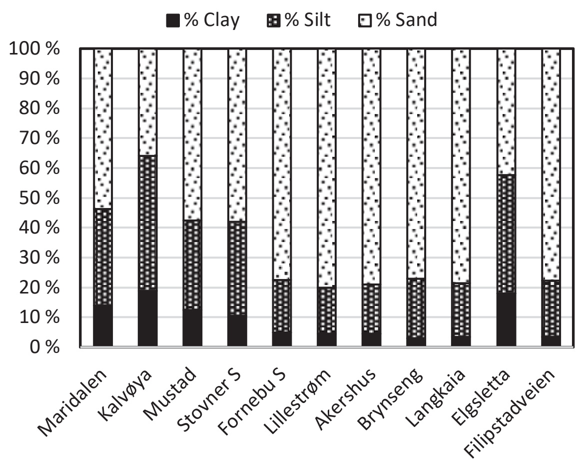


Figure 2. Mean grain size distribution of study areas.

Neither Shannon diversity nor richness were related to grain size in any size category. Some plant families, however, had a significant relationship to grain size (see Table 4).

Table 3 A & B. All mean measures for soil values except for grain size distribution, as well as averages of Shannon diversity and species richness for samples arranged by: **A.** *Site* and **B.** *Treatment*. P-values displayed in italicized gray except for a comparative category, which all other values are in comparison to. **Legend:** Solid orange down-arrow = low value (two = very low); solid blue up-arrow = high value; gray circle = medium value.

A.

Site	Other variables	pH	PO_4-P (mg/kg)	NO_3 (mg/kg)	NH_4 (mg/kg)	Dryness (%)	% Barren	Shannon diversity	Richness
Maridalen	Reference Soil unmoved Hay	↓ 5.25 5.49e-10 ***	↓ 62.00 9.13e-05 ***	○ 11.08 0.0718	↑ 38.50 0.000204 ***	↓ 65.76 0.000445 ***	↓ 15.5 0.03077 *	○ 0.8945 0.865352	○ 4.89 0.633839
Kalvøya	Reference Soil unmoved Calcareous	↓ 5.59 2.22e-08 ***	↓↓ 16.50 1.04e-05 ***	○ 5.35 0.44736	○ 17.00 0.170053	↓ 72.04 0.030655 *	↓ 14.8 0.02427 *	○ 0.9392 0.824545	○ 5.25 0.906828
Mustad	Created Local soil Calcareous	○ 6.74 0.007960 **	↑ 478.00 0.027016 *	○ 4.48 0.55113	○ 4.60 0.845271	○ 73.94 0.10999	○ 19.6 0.13962	↑ 1.2624 0.002450 **	↑ 7.82 0.000117 ***
Stovner Senter	Created Commercial Hay	↑ 7.24 0.401201	↓ 53.00 5.96e-05 ***	○ 0.96 0.94684	○ 5.18 0.903221	○ 81.26 0.785987	↓↓ 2.4 3.27e-05 ***	↑ 1.3599 0.000114 ***	↑ 8.21 1.36e-05 ***
Fornebu S	Roof Local soil Calcareous	↑ 7.05 0.120609	↓ 72.50 0.000150 ***	○ 0.16 0.82676	○ 5.38 0.923521	○ 78.53 0.63506	↑ 70.2 2.40e-09 ***	↓ 0.6310 0.013691 *	↓ 3.57 0.004257 **
Lillestrøm	Roof Commercial Hay	↑ 7.38 0.756393	○ 265.50 0.26129	○ 0.56 0.89	○ 1.65 0.567512	○ 84.35 0.281454	↑ 47.1 0.00460 **	↑ 1.3828 5.09e-05 ***	↑ 8.43 3.91e-06 ***
Akershus	Restored Soil unmoved Flower	↑ 7.46 NA	○ 335.75 NA	○ 1.32 NA	○ 6.13 NA	○ 80.21 NA	○ 29.2 NA	○ 0.9139 NA	○ 5.18 NA
Brynseng	Restored Soil unmoved Flower	○ 6.68 0.004723 **	↓ 42.00 3.53e-05 ***	○ 0.51 0.879	○ 3.75 0.761277	○ 81.55 0.734385	○ 40.2 0.08149	○ 0.9216 0.945785	○ 5.18 1
Langkaia	Created Commercial Flower	○ 6.25 4.06e-05 ***	↓ 55.50 6.71e-05 ***	○ 0.31 0.84891	○ 2.29 0.624129	○ 85.43 0.160087	○ 17.2 0.05794	↓ 0.6693 0.032810 *	↓ 3.79 0.014221 *
Elgsletta	Planted Soil unmoved Flower	○ 6.66 0.003571 **	↓ 152.50 0.005357 **	↑ 20.40 0.00093 ***	○ 3.91 0.776898	○ 78.28 0.587727	○ 23.4 0.35776	↓ 0.5871 0.004469 **	↓ 3.25 0.000495 ***
Filipstadveien	Planted Soil unmoved Flower	○ 6.37 0.000164 ***	↓ 97.50 0.000477 ***	○ 1.75 0.93405	○ 10.35 0.59	○ 73.97 0.096485	↓ 11.8 0.00612 **	○ 0.9728 0.605695	○ 5.46 0.643098

B.

Treatment	Sites	pH	PO_4-P (mg/kg)	NO_3 (mg/kg)	NH_4 (mg/kg)	Dryness (%)	% Barren	Shannon diversity	Richness
Created	Mustad Langkaia Stover S	○ 6.74 NA	○ 195.50 NA	○ 1.92 NA	○ 4.02 NA	○ 80.33 NA	↓ 13.0 NA	↑ 1.0972 NA	↑ 6.61 NA
Planted	Elgsletta Filipstadveien	○ 6.51 0.2791	○ 125.00 0.337198	↑ 11.08 0.0189 *	○ 7.13 0.552	○ 76.13 0.12058	↓ 17.6 0.269	↓ 0.7780 0.000193 ***	↓ 4.36 5.95e-08 ***
Reference	Kalvøya Maridalen	↓ 5.42 1.66e-07 ***	↓ 39.25 0.037517 *	○ 8.21 0.1003	↑ 27.75 4.72e-05 ***	↓ 69.00 0.00012 ***	↓ 15.2 0.599	○ 0.9169 0.032726 *	○ 5.07 0.000288 ***
Restored	Akershus Brynseng	○ 7.07 0.1216	○ 188.88 0.927713	○ 0.91 0.7901	○ 4.94 0.861	○ 80.88 0.8392	○ 34.8 4.30e-07 ***	○ 0.9178 0.033604 *	○ 5.18* 0.000773 ***
Roof	Fornebu S Lillestrøm	↑ 7.21 0.0286 *	○ 169.00 0.716905	○ 0.36 0.6794	○ 3.51 0.922	○ 81.38 0.69658	↑ 57.54 < 2e-16 ***	↑ 1.0069 0.283788	↑ 6.00 0.163164

Trends in plant families & non-native species

The effects of variables on the abundances of individual plant families and non-native species are summarized in Table 4. Additionally, differences between variables with more than two categories are shown in Fig. 3 A-E. Differences in abundances between sites are not shown here due to the visual clutter of the data, but raw counts can be viewed in Supplementary Table 4, and sites that may have contributed to bias are discussed below.

Table 4. Coefficients and estimates of effect between explanatory variables and abundances of 8 selected plant families and non-native species. Values generated using GLM assuming a Poisson distribution. P-values displayed in italicized gray. **Legend:** Solid orange down-arrow = significant negative effect; solid blue up-arrow = significant positive effect; gray circle = no significant effect.

Variable	Asteraceae	Brassic-aceae	Caryophyll-aceae	Crassul-aceae	Fabaceae	Plantagin-aceae	Rosaceae	Rubiaceae	Non-native
Weeded (yes)	↑ 0.4254 2.91E-05 ***	○ 0.4951 0.168	↑ 1.5825 1.62E-14 ***	↓ -2.0053 1.55E-05 ***	↑ 0.37156 0.00308 **	↑ 0.8473 0.0017 **	↑ 0.6506 0.0023 **	↑ 0.7221 0.00015 ***	↑ 1.6776 1.92E-09 ***
Watered (yes)	↑ 0.37536 0.00041 ***	○ -0.2513 0.559	↑ 0.698 0.000144 ***	○ -18.1394 0.986	↑ 0.6657 1.44E-07 ***	○ -0.5452 0.118	↓ -0.865 0.00536 **	○ -0.361 0.123	↓ -0.6817 0.0382 *
pH	↑ 0.3539 1.68e-05 ***	↑ 0.8964 0.008963 **	↑ 1.6733 4.55e-14 ***	○ 0.1498 0.414	↑ 0.28834 0.00373 **	○ -0.294 0.118	↓ -0.619 2.09e-05 ***	↓ -0.3099 0.0199 *	↑ 1.1929 2.35e-06 ***
PO_4-P (mg/kg)	↑ 1.31E-03 0.00122 **	2.41E-03 0.1257	○ 5.77E-05 0.9412	○ -8.99E-05 0.954733	○ -5.03E-04 0.336	↑ 3.68E-03 0.000547 ***	○ 6.58E-04 0.4243	○ 1.30E-03 0.086	↑ 3.61E-03 0.000852 ***
NO_3 (mg/kg)	↓ -3.11E-02 0.01851 *	○ 5.36E-02 0.0531	↓ -1.15E-01 0.0106 *	↓ -8.81E-01 1.5e-05 ***	○ 8.30E-03 0.444	○ -4.91E-02 0.237816	○ -5.84E-02 0.0725	↓ -6.36E-02 0.0437 *	↓ -1.16E-01 0.038005 *
NH_4 (mg/kg)	↑ 4.94E-02 1.53e-05 ***	○ -3.81E-02 0.5754	○ 1.45E-02 0.7395	↑ 1.96E-01 0.011113 *	○ -1.85E-02 0.24	○ -1.43E-02 0.624115	↓ -5.87E-02 0.0141 *	↑ 4.60E-02 0.0189 *	○ -1.56E-01 0.051122
Area (m^2)	↓ -5.91E-04 1.19e-09 ***	↓ -1.06E-03 0.00815 **	↓ -5.62E-04 0.001119 **	↑ 1.37E-03 2.63e-13 ***	↓ -5.83E-04 1.03e-06 ***	○ -3.70E-04 0.122	↑ 7.12E-04 8.16e-06 ***	↑ 4.46E-04 0.00201 **	↓ -7.10E-04 0.00302 **
Dryness (%)	↑ 9.50E-02 2.44e-06 ***	8.01E-02 0.3232	↑ 9.88E-02 0.0253 *	↓ -1.89E-01 0.000788 ***	○ 1.79E-02 0.481	↓ -1.49E-01 0.008421 **	↓ -2.58E-01 1.81e-07 ***	○ 7.86E-03 0.8303	○ -6.52E-02 0.259242
% Barren	○ -1.71E-03 0.339	○ 6.89E-03 0.228	○ -2.91E-03 0.374	↑ 2.98E-02 1.59e-14 ***	↓ -1.08E-02 2.31e-05 ***	○ -4.78E-03 0.335	↓ -1.23E-02 0.00629 **	↓ -9.87E-03 0.00969 **	○ 6.01E-03 0.126
% Clay	↓ -0.0686 2.7e-11 ***	-0.04596 0.178	↓ -0.07004 0.000156 ***	↓ -0.04718 0.0382 *	○ -0.01539 0.163	↑ 0.04946 0.024 *	↑ 0.1404 3.33e-13 ***	↑ 0.07412 2.11e-06 ***	○ -0.03574 0.108
% Silt	↓ -0.03965 6.77e-11 ***	○ -0.03916 0.06777	↓ -0.02969 0.00407 **	↓ -0.05262 0.000626 ***	○ -0.00263 0.678	○ 0.02034 0.112	↑ 0.07836 9.79e-14 ***	↑ 0.04219 2.20e-06 ***	↓ -0.03697 0.00926 **
% Sand	↑ 0.02579 3.51e-11 ***	↑ 0.02224 0.0962	↑ 0.02112 0.00163 **	↑ 0.02661 0.00395 **	○ 0.003195 0.433	○ -0.01537 0.0591	↓ -0.05172 8.36e-14 ***	↓ -0.02765 1.47e-06 ***	↑ 0.01883 0.0308 *

Differentiating Fabaceae trends from T. pratense

Given that *T. pratense* comprised almost half of all Fabaceae observations (126/256) but was also deliberately and preferentially planted at some sites, especially the flower meadows, we felt it would be useful to also analyze Fabaceae trends excluding this species. For most variables, Fabaceae minus *T. pratense* demonstrated similar trends. Differences include *site area*, which did not have an effect on Fabaceae-Tp (p = 0.121); *nitrate*, which had a significant negative effect on Fabaceae-Tp (estimate = -0.1009, p = 0.0053); and *pH*, which lost its effect (p = 0.08973). Categorical variables that changed included *Treatment*, where "restored" and "planted" sites were now significantly less abundant in Fabaceae-Tp than "created" sites and more similar to other sites (estimates -0.5534 and -1.2910, p = 0.02404 and 8.28e-05, respectively); *Type* of meadow, which left flower meadow sites now much poorer in Fabaceae-Tp than hay meadows (estimate = -1.0061, p = 7.31e-07); and of course *Site*, which will not be summarized here, but the raw counts can be seen in Supplementary Table 4.

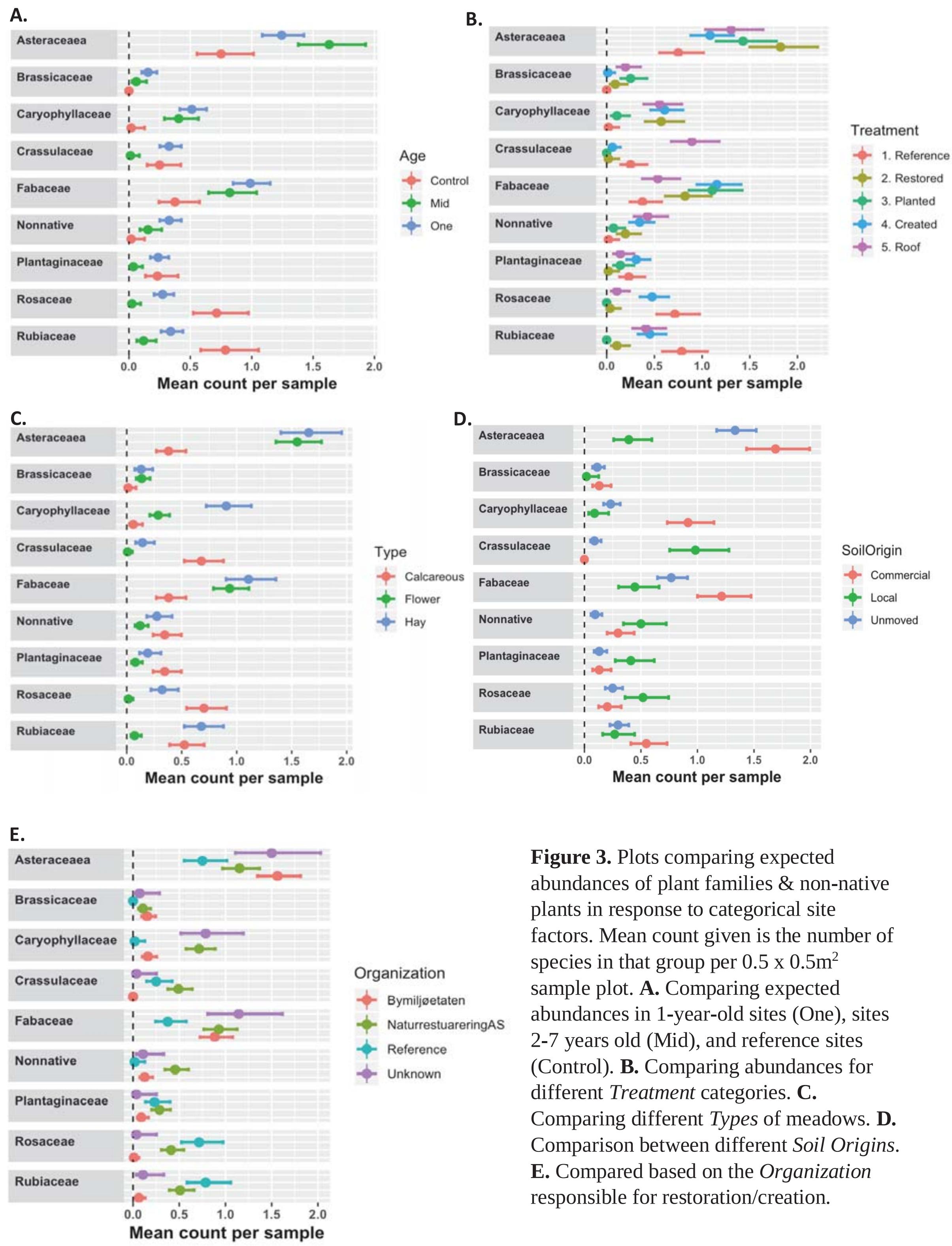


Figure 3. Plots comparing expected abundances of plant families & non-native plants in response to categorical site factors. Mean count given is the number of species in that group per 0.5 x 0.5m^2 sample plot. **A.** Comparing expected abundances in 1-year-old sites (One), sites 2-7 years old (Mid), and reference sites (Control). **B.** Comparing abundances for different *Treatment* categories. **C.** Comparing different *Types* of meadows. **D.** Comparison between different *Soil Origins*. **E.** Compared based on the *Organization* responsible for restoration/creation.

Percent barren ground

The percent of barren ground visible in a sample plot was strongly dependent on the *Site* variable, and also significantly related to the *Treatment*, *Type*, and *Soil Origin* variables (Table 3 A&B, Supplementary Table 1 A&B). The mean barren ground of a site was not related to the dryness of the soil samples taken from that site ($p = 0.134$), nor was the percent barren ground of a sample significantly affected by the Julian date ($p = 0.0501$). Some groups of plants, however, showed significant trends relating to the barrenness of a plot (see Table 4). It should be noted that it was possible for a site to have 0 species counted and 0% barrenness if the plot contained only grasses or mosses.

Site age

In general, the *Age* variable was not significantly related to the majority of factors in this study, especially since the structure of the data required a categorical analysis rather than a linear regression. Shannon diversity and richness were significantly lower for the one site in the 2-year category (Stover Senter, $p = 0.0292$, $p = 0.01020$, respectively). Richness was significantly higher in the one-year category (n=5) than other ages of sites ($p = 0.00486$).

The abundances of all selected plant families as well as non-native plants were significantly related to the *Age* variable. However, since there was not a clear pattern between the two-, three-, and seven-year categories, these were condensed into a "mid" category (n=4) and tested against the one-year category and the reference/control category (n=2). These results are shown in Fig. 3-A.

Discussion

A note on an unusual growing season

The growing season of 2018 in Oslo was highly unusual. Spring arrived late, with a mean March temperature of -2.3°C (2.1°C below average), as well as precipitation 29% of normal that month (Meteorologisk Institutt 2019). This contributed to a later-than-normal growing season, especially since precipitation remained low (87% of normal in April, then below 60% the next two months) while temperatures increased to above-normal levels (+2.0-5.3°C). The warm, dry weather developed into a severe drought in July, when the mean temperature reached 5.8°C above normal and Oslo received a mere 20% of its typical precipitation. Although temperatures dropped to 1.1 and 1.7°C above normal in August and September, drought conditions did not cease until after the majority of sampling had already concluded in September. In general, Shannon diversity, but not richness, decreased as the season progressed, which is likely related.

In some ways, the extreme weather is an interesting stress-test of our sites, since semi-natural meadows are supposed to be characterized by drought-tolerant species growing on thin, well-drained soil (Evju *et al.* 2015). Therefore, the survival of these plants this summer should bode well for their utility in the harsher environments of green roofs and urban spaces more generally, especially if they can thrive without additional intervention by property managers. On the other hand, there's a real risk that this has skewed our data toward those drought-tolerant plants, especially at sites that might have faced a more significant threat from non-native species if not for their specialist advantage. Since this researcher was responsible for much of the weeding at the Mustad site over the summer, I can report that, anecdotally, some of the non-target species did not appear to be thriving by comparison – specimens of *M. officinalis* in particular appeared to be in poor health in drier parts of the meadow, while this species had been especially over-dominant the year prior. Similarly, the native but easilydominating species *Aegopodium podagraria* was nearly eradicated from the drier side of the meadow, as were a majority of *Rubus idaeus* there.

By contrast, the roof at Fornebu S was an example of how a project can fail due to extreme weather and lack of adaptive maintenance. Here, the site managers were not instructed to provide any additional care. This is likely a significant contributing factor in why the species diversity and richness were both very low at Fornebu S, as well as why it was one of the only two sites to increase in species richness later in the year after a few events of significant precipitation. Additional factors that likely made this site particularly vulnerable to the drought were that the majority of the substrate appeared to be much thinner than the other roof site, Lillestrøm, with only 5-7cm substrate thickness above gravel. Lillestrøm is also more shielded from direct sunlight and wind by taller parts of the building to the south and north (see Supplementary Figure 1), while Fornebu S is entirely exposed to the elements.

"*Watered*: yes" was surprisingly not a strong variable in our study, despite the drought. Part of the lack of statistical significance may be due to the binary yes/no assignment of this variable, while the reality may have been more complicated – although we know that some sites were watered based on manager surveys, we do not know how often or how much, and in addition we cannot say for certain whether the meadow at Lillestrøm (categorized as un-watered in this study) benefited from the watering of plants on the same roof since we do not know the precise dynamics of water there. Stovner Senter's meadow seems to have been handled rather like a garden and watered regularly. Langkaia and Filipstadveien, on the other hand, were reportedly given supplementary watering, but we do not know how often. It's conceivable, for instance, that frequent indirect watering near the roof-based meadow at Lillestrøm could have had a stronger effect on that site than infrequent but direct watering at Langkaia and Filipstadveien.

That said, watering was associated with higher diversity and species richness among semi-natural meadows only (Supplementary Table 2, rightmost side). It also seems obvious that in extreme cases, such as the roof at Fornebu S, watering should be used at the site manager's discretion in order to save a vulnerable site that has not necessarily had time to become established, or which may be situated in an environment harsher than ideal. The subject is one that Naturrestaurering A.S. plans to investigate more in the coming years, and in the meantime, the recommendation for supplementary watering in times of severe drought has already been added to the care instructions for newly-established meadows.

Why were the reference sites less biodiverse than expected?

We were surprised that our reference sites were only of medium biodiversity, especially when compared to entirely artificial sites in the "created" and "roof" *Treatment* categories. Although it's possible that these particular reference sites truly are less rich than believed, we have developed a number of alternate hypotheses. One factor that we believe to be contributing is that, while freshly-created sites are initially very species-rich, they may lose species over time as species out-compete each other and are lost from the patch (Breuste *et al.* 2008). We may expect, for example, that the meadow at Stovner Senter will favor species that benefit from regular watering and that are faster-growing or more shade-tolerant, that the meadow at Mustad will continue to be shaped by the site's high phosphate values and the favorable conditions that creates for unwanted species, and that the meadow at Lillestrøm may come to support more shade-tolerant species.

Another explanation that we suggest is that, since these sites are older, they may have a greater degree of niche partitioning between microhabitats. This would require a higher density of sample plots in order to capture a representative count of species, and indeed, species accumulation curves for the reference sites suggested that additional samples would have returned a higher number of species. Finally, we know that a significant proportion of the biodiversity in semi-natural meadows is among moss and lichen species (Bratli *et al.* 2005, Butaye *et al.* 2005), as well as the grasses, which we did not survey. Mosses and lichens are among the first species to be impacted by environmental disturbance and increased nutrient levels, and are also highly sensitive to pH (Maskell *et al.* 2010), so we may expect these to be more biodiverse in the reference sites than in newer meadows and with different species depending on soil chemistry.

Factors affecting non-native plant abundance

As one would expect, non-native plants were most common in one-year-old sites (p = 0.00394), and more common in 2-7-year-old sites than in control sites (p = 0.03744). The *Soil Origin* variable was probably the strongest determinant of non-native plant abundance among the categorical variables – sites with undisturbed soil tended to have fewer non-native species than those where open soil had provided colonization opportunities for unwanted plants, in line with our predictions.

Sites with high pH tended to have more non-native species, and high phosphate was also highly correlated with a high abundance of these plants – referring to Tables 2 and 3A, the sites with the highest percentages of non-native plants were a close match for the sites with the highest phosphate values. Since the sites were not deliberately fertilized, this implies that there may be some factor in the local soil and/or rock resulting in higher phosphate, making this condition potentially difficult to avoid. However, it is possible to be forewarned, and adjust management expectations accordingly. Additionally, project planners can avoid causing overly high pH.

Two sites with particularly high counts of non-native species require some additional explanation: Fornebu S and Mustad. At Fornebu S, five of the seven non-native plants counted were invasive succulents that probably arrived as part of the succulent mat installed there, so the method of colonization for these was somewhat different. In addition to the factors above, Mustad was also the only "created" site that has no physical separation from the surrounding plant community. Invasive species were weeded only up to the inside edge of the perimeter fence; beyond that, they grew unimpeded and in great abundance.

The "*Weeded*: yes" variable corresponded with higher abundances of non-native species – presumably only sites that already had invasive species needed weeding in 2017, and at least some of these survived to the year of our survey. Note that the explanation for weeded sites tending to be more biodiverse is likely the reverse – that factors increasing the biodiversity of a site (e.g. recent establishment, high pH, and high phosphate) were related to higher abundances of unwanted species.

Soil strategies

Our study confirms that working with the original soil was the best option for avoiding invasion by unwanted species, but leaving the soil intact is not always an option in site design or due to soil chemistry. Comparing commercially-acquired soil to topsoil taken from local donor sites, we found that using commercial soil did not result in significantly fewer non-native plants than using local soil, while local soil did seem to provide more target plant species at Mustad compared to other semi-natural meadows. When local topsoil is available, suitably low in macronutrients, and can be stored correctly in order to preserve the seed bank, it is preferable to commercial soil for the chance to recruit target species (Brenneisen 2003). That said, commercial soil was not shown to be inherently worse, provided that it does not contain unwanted additives, particularly fertilizer.

The specific soil strategy should be determined on a site-by-site basis and preferably after some simple soil tests, especially for nitrate and phosphate. Local factors such as agriculture and high vehicle traffic should be considered. We believe that the high nitrate values at Elgsletta (20.40 mg/kg, p = 0.00093) may be caused by its proximity to a highly-trafficked intersection, especially since the slope may encourage deposition. In our study, species richness was negatively impacted by higher nitrate even when Elgsletta's high value was excluded from the analysis (Supplementary Table 2, rightmost column). There was also only one site with a significantly high measure of ammonium, Maridalen at 38.50 mg/kg (p = 0.000204), probably due to agriculture. Otherwise, it appears that ammonium was not a significant macronutrient in this study.

We were particularly surprised by the low macronutrient values of the Stovner Senter site – although it's not quite captured by the plant survey measurements, the vegetation at that site was incredibly dense and tall. There were a number of coconut fiber mats used at Stovner S due to its steep incline, and there was concern from Naturrestaurering A.S. that this acted as a source of nutrients. Instead, we found that all macronutrient values there were low. Presumably the plant density is instead due to the combination of extreme heat in the summer of 2018 and regular watering, since the property owners installed drip hoses at the top of each meadow terrace. Whether that was a good management strategy, however, is debatable – tall and dense vegetation is at odds with a semi-natural meadow's characteristic low-growing, light-craving species (Bobbink & Willems 1991).

For pH especially, we expected a high value at the calcareous meadow Kalvøya and instead found the second-lowest (5.59, p = 2.22e-08), and the lowest at Maridalen (5.25, p = 5.49e-10). We did not find that calcareous meadows in our study had an especially high pH, nor was it significantly different from the hay or flower meadows. Instead, it seems likely that sites created by Naturrestaurering A.S. generally over-estimated the target pH (estimate = +0.4181 compared to other project organizers, p = 0.0171). It appears, in fact, that the normal range of pH for Norwegian soil is 4.5-5 or lower (Land Resources Management Unit 2010), suggesting that the low values found in this study are well within normal. The critical threshold for acidification and aluminum toxicity for meadow species was given as ≤ 4.5 pH by Maskell *et al.* (2010) – a value which none of our measurements reached.

Even though we found that vascular plant biodiversity increased with higher pH (Supplementary Table 2), it's probably better to aim for a closer match to our reference sites and native soil values, even if they are relatively acidic. Some vital macronutrients become less bioavailable in very alkaline soils, such as iron, manganese, and zinc (Aasen 1992). Additionally, since mosses and lichens are an important part of the meadow community and their species composition is altered significantly by soil pH (Maskell *et al.* 2010), sometimes resulting in thick mats that impede target plant growth (Butaye *et al.* 2005, Auestad *et al.* 2015), this is an important issue.

Although we measured the grain size composition of soils in this study, we ultimately felt that this did not quite capture the information we were most interested in. Since the test procedure for grain size composition removes the organic component of the soil, we cannot know how this factor varied between sites. During collection, it did appear as if some sites – especially the two reference sites – had more organic content than others. Additionally, some sites may have had an over-representation of gravel, but since the grain size composition analysis only measures particles under 2mm, this information was not captured. Water retention (reported as "dryness", i.e. percent dry matter) did seem to be related somewhat to high clay content, since the soils at Maridalen and Kalvøya were both significantly wetter and more fine-grained, though the same was not true of Elgsletta. Soil dryness was only significantly related to biodiversity in a sample set restricted to semi-natural sites, however (Supplementary Table 2, rightmost column), and was highest in commercially-acquired soils (Supplementary Table 1-B).

Diversity and richness were negatively impacted by an increased percentage of barren ground (Supplementary Table 2), which was not in line with our hypothesis, but which was not entirely surprising. Sample plots at the reference sites usually had at least some form of plant life covering most of the area as opposed to bare earth, so it does not appear to be a feature of the habitat based on our results. It may simply be the case that the newer sites need more time for succession to occur in these rocky areas.

Successes and failures of green roofs

The high degree of similarity between the community composition at Lillestrøm and Stovner Senter (Figure 3-A) is both encouraging and scientifically interesting: since these sites were both created by the same organization, using similar methods, with the same habitat goal, and within a year of each other, they represent a nearly one-to-one comparison between creating a meadow on the ground and attempting to create that same meadow on a green roof. The initial results are promising – both of these sites had very high biodiversity, with a good potential for species composition to change over time to suit the site. These sites should be monitored over the coming years to track changes, but our initial impression is that this represents a success, even though the height of the vegetation at Stovner Senter remains a concern.

To account for further differences between Lillestrøm and Fornebu S, it would be useful to find out exactly what the minimum substrate depth was at both locations, since substrate depth is known to determine the kind of plant community supported by a green roof (Gabrych *et al.* 2016; Madre *et al.* 2014). According to Naturrestaurering A.S. (*pers. comm.*), the architects associated with Fornebu S initially reported that the minimum substrate depth would be 15-20cm deep, but it appeared to actually be 5-7cm, while the substrate at Lillestrøm may be 20-30cm deep. The underlying gravel at Fornebu S also did not appear to be different from the typical gravel used on roofs for water drainage, as opposed to the local substrate used for the mounds. Even though Norwegian semi-natural meadows are adapted for thin soils on rocky outcrops, these results appear to be in line with previous findings showing that roof substrate optimized for water drainage supports a more restricted community compared to the same substrate depth of local rock (Brenneisen 2003).

Matching plant communities to project goals

In our study, the organization responsible for a project appears to be the primary top-down variable determining the plant community of a site followed by the type of meadow (Figure 3), and therefore success in matching plant communities is dependent largely on that organization's research and goals. The high abundances of some families in the restored and re-created sites, especially species in Asteraceae, Fabaceae, and Caryophyllaceae, may be a result of selection for their aesthetic contribution, the relative ease of collecting seeds, or for pollinator resources (which was specifically the case for *T. pratense* - Bymiljøetaten, *pers. comm.*).

The calcareous meadow at Mustad matched the community at Kalvøya fairly well and had a good representation of plants in the Plantaginaceae family, which seems to be characteristic for these meadows along with the presence of plants in the Crassulaceae family. Through the combination of using seed-bearing donor soil from a previous meadow and sowing collected plants based on the desired community, the top-down effect of plant selection seems to have been effective. However, time will ultimately determine whether this will overcome the bottom-up pressures at this site caused by high phosphate and the surrounding landscape's abundance of non-native plants.

Reference sites had a significantly higher abundance of Rosaceae and Rubiaceae species than other semi-natural meadow sites. These families were also well-represented in the re-created semi-natural meadows contrasted against the flower meadows, but could perhaps be emphasized even more in future projects. Although we did not measure grasses in this study, it's likely that we would find this plant family under-represented at these same sites in abundance and perhaps also variety. This may be due to biases during seed collection, or due to an intentional desire for a more flower-rich site favored by humans and/or pollinators.

That said, accuracy does not always have to be the main goal of a project in order to see good results for biodiversity (Kowarik 2011). In fact, even though we found a lower species richness for planted flower meadows than for restored, the simultaneous study by Nielsen *et al.* (unpublished manuscript) found that the planted meadows were nonetheless heavily utilized by native bee species, suggesting that these sites are still important contributors to conservation and biodiversity. It does not seem like an ineffective strategy to add some species back and then simply stop maintaining it as a lawn, in line with Klaus (2013) and Kowarik (2011). It also seems like hay removal at roadside sites like Elgsletta is a valid method of removing accumulated nitrate deposition (Bakker *et al.* 2002) while also providing pollinator resources.

Age trends between selected plant families (Figure 3-A) are not uniform, and may be split into families affected changing project organizer goals and families affected by the age of the site. Families that showed a potential successional trend of decreasing over time were Brassicaceae, Caryophyllaceae, and Fabaceae, although we cannot say definitively that this was not caused by the way the data were pooled into categories. Additional research will probably be required to see whether the reduction in these families over time is real or due to random effects.

Usefulness of techniques for research

One of the purposes of this study was to determine the suitability of these techniques for use in future studies conducted in conjunction with Naturrestaurering A.S. Overall, the techniques used in this study were relatively simple, quick, and repeatable. The plant survey's greatest time requirement is probably the travel between sites (especially when using public transit), whereas survey plots recording plant species presence/absence take at most a few minutes each in places where species grow very densely. Plants that are difficult to identify without additional resources (i.e. image searches online or consulting an expert) can be documented extensively in the field and then identified later, assuming the surveyor knows what to document. The major drawback of the survey method is that it requires someone who is capable of identifying those plants, or who is willing to invest the time needed to learn how. We also recommend that this surveying method be expanded to include some of the more identifiable bryophytes, although that increases the difficulty.

The soil tests are relatively straightforward, inexpensive, and are conducted according to Norwegian standard, so they are also widely applicable and can be performed in most labs.

Further directions for research

Rather than characterizing our sites based on measuring the site itself and then using the plant survey primarily to measure biodiversity, we could instead use our plant species data to get Ellenberg R values and describe our sites based on those. Since Ellenberg R values describe the preferred habitat characteristics of plant species (Maskell *et al.* 2010), we could calculate the approximate characteristics of our sites and compare them in that way, as well as on a plot-by-plot basis to account for small variations within a site. Since Ellenberg R values corrected for Norwegian flora have been calculated and published (Vevle 1985), this remains a possibility for future analysis of the same data used in this study. Additionally, we could try using a partial redundancy analysis (RDA) rather than a pure principal component analysis, as in Madre *et al.* 2014. A significant advantage of these techniques is that they preserve the species-level data of our observations instead of forcing us to combine species to their families or downgrading even further to raw species richness.

Another direction for research is to directly compare the species sown in a site to the species that grew there. Since Naturrestaurering A.S. presently uses a hand-collected selection of seeds, it should be possible to calculate their success in growing these target species as well as record which new species colonize a site given different treatment conditions. This is in essence a project that has been planned for the summer of 2019, which will also attempt to more directly measure the effect of establishment-year watering on target species success.

Lastly, long-term monitoring will be necessary in order to record changes over the coming years. We can probably expect that the newly-created, high-diversity sites will become less species-rich and differentiate more in the coming years, while the least species-rich sites in this study will recruit more specimens of either target or non-target plants. Although we did not measure them in this study, we may also expect that those same sites could become richer in moss and lichen over time. It could even be the case that we find that restoration success for these meadows is best measured using mosses and lichens, requiring an expanded plant survey in the future.

It will be interesting to see how nutrient levels change (or do not) over the coming years, and how management efforts continue to direct the species compositions we see. We are also interested in seeing how our results during this year of severe drought compare to future non-drought years, especially since these kinds of extreme weather events are predicted to become more common in Europe as climate change progresses (Schär *et al.*, 2004). Since the plant survey techniques used in this study are relatively simple and the sites are mostly accessible, we're hopeful that we can indeed continue monitoring them in a consistent and thorough manner.

Conclusion

We have successfully identified examples of both successful and unsuccessful techniques used by Naturrestaurering A.S. and Bymiljøetaten, and have made recommendations here based on our findings. Our results, especially regarding soil pH and failure to adaptively manage during extreme weather events, demonstrate the importance of monitoring project success in the years after project completion as well as determining target soil values directly from reference sites. We have also shown that semi-natural meadows can be surprisingly resilient during severe drought, at least over the relatively short period of a year.

We have tested the monitoring techniques described in this paper and found that, for the most part, they are repeatable and effective. We have also described additional avenues of research and ways in which the methods used here could be improved in order to capture more detailed information. Overall, our results are encouraging for the future of semi-natural meadow restoration and re-creation in urban Oslo, as well as for research on the subject.

References

Aasen, I. (1992) Mikronæringsstoff, magnesium og svovel i jordbruket: oversikt over tilhøva i Noreg. Ås, Norway. Statens fagtjeneste for landbruket.

Artsdatabanken (2018) Fremmedartslista 2018. <https://www.artsdatabanken.no/fremmedartslista2018> Accessed April 2, 2019.

Auestad, I., Austad, I. and Rydgren, K. (2015) Nature will have its way: Local vegetation trumps restoration treatments in semi-natural grassland. *Applied Vegetation Science*, 18(2), pp. 190–196.

Bakker, J. P. and Berendse, F. (1999) Constraints in the restoration of ecological diversity in grassland and heathland communities. *Trends in Ecology and Evolution*, 14(2), pp. 63–68.

Bakker, J. P., Elzinga, J. A. and De Vries, Y. (2002) Effects of long-term cutting in a grassland system: Perspectives for restoration of plant communities on nutrient-poor soils. *Applied Vegetation Science*, 5(1), pp. 107–120.

Bobbink, R. and Willems, J. H. (1991) Impact of different cutting regimes on the performance of *Brachypodium pinnatum* in Dutch chalk grassland. *Biological Conservation*, 56(1), pp. 1–21.

Braaker, S., Ghazoul, J., Obrist, M. K., and Moretti, M. (2014) Habitat connectivity shapes urban arthropod communities: the key role of green roofs. *Ecological society of America*, 94(4), pp. 1010–1021.

Bratli, H., Evju, M. and Stabbetorp, O. E. (2005) *Kalkberg – et hotspot-habitat*.

Brenneisen, S. (2003) Space for urban wildlife: designing green roofs as habitats in Switzerland. *Urban Habitats*, 4(1), pp. 27–36.

Breuste, J., Niemelä, J. and Snep, R. P. H. (2008) Applying landscape ecological principles in urban environments. *Landscape Ecology*, 23(10), pp. 1139–1142.

Butaye, J., Adriaens, D. and Honnay, O. (2005) Conservation and restoration of calcareous grasslands: a concise review of the effects of fragmentation and management on plant species. *Biotechnologie, Agronomie, Société, Environnement*, 9(2), pp. 111–118.

City of Oslo (2019). Application form for the European Green Capital Award 2019. *European Green Capital Award*, European Commission. <http://ec.europa.eu/environment/europeangreencapital/wp-content/uploads/2017/06/City_Introduction.pdf> Accessed April 19, 2019.

Connell JH (1978) Diversity in tropical rainforests and coral reefs. *Science*, 199, pp. 1302–1310.

Cousins, S. A. O. and Eriksson, O. (2001) Plant species occurrences in a rural hemiboreal landscape: effects of remnant habitats, site history, topography and soil. *Ecography*, 24(4), pp. 461–469.

Direktoratet for Naturforvaltning (2009) *Handlingsplan for slåttemark*.

Donath, T. W., Hölzel, N. and Otte, A. (2003) The impact of site conditions and seed dispersal on restoration success in alluvial meadows. *Applied Vegetation Science*, 6(1), pp. 13–22.

Eriksson, O., Bruun, H. H. and Cousins, S. A. O. (2002) Land-Use History and Fragmentation of Traditionally Managed Grasslands in Scandinavia. *Journal of Vegetation Science*, 13(5), pp. 743–748.

European Environmental Agency (2017) Pressure from critical loads from nitrogen [map]. <https://www.eea.europa.eu/data-and-maps/figures/pressure-from-critical-loads-from-nitrogen> Accessed April 29, 2019.

Evju, M., Blumentrath, S., Skarpaas, O., Stabbetorp, O. E., and Sverdrup-Thygeson, A. (2015) Plant species occurrence in a fragmented grassland landscape: the importance of species traits. *Biodiversity and Conservation*, 24(3), pp. 547–561.

Fontana, S., Sattler, T., Bontadina, F., and Moretti, M. (2011) How to manage the urban green to improve bird diversity and community structure. *Landscape and Urban Planning*. Elsevier B.V., 101(3), pp. 278–285.

Fuller, R. A., Irvine, K. N., Devine-Wright, P., Warren, P. H., and Gaston, K. J. (2007) Psychological benefits of greenspace increase with biodiversity. *Biology Letters*, 3(4), pp. 390–394.

Gabrych, M., Kotze, D. J. and Lehvävirta, S. (2016) Substrate depth and roof age strongly affect plant abundances on sedum-moss and meadow green roofs in Helsinki, Finland. *Ecological Engineering*. Elsevier B.V., 86, pp. 95–104.

Goddard, M. A., Dougill, A. J. and Benton, T. G. (2010) Scaling up from gardens: biodiversity conservation in urban environments. *Trends in Ecology and Evolution*, 25(2), pp. 90–98.

Hansen, A. J., Knight, R. L., Marzluff, J. M., Powell, S., Gude, P. H., and Jones, K. (2005) Effects of Exurban Development on Biodiversity: Patterns, Mechanisms, and Research Needs. *Ecological Applications*, 15(6), pp. 1893–1905.

Jogesh, T., Carpenter, D. and Cappuccino, N. (2008) Herbivory on invasive exotic plants and their non-invasive relatives. *Biological Invasions*, 10(6), pp. 797–804.

Kartverket (2016) Norge i Bilder [map]. 1:50,000. "Norge i Bilder." <https://norgeibilder.no> Accessed February 2, 2019.

Kartverket (2019) Norgeskart [map]. 1:50,000. "Norgeskart." <https://norgeskart.no> Accessed January 28, 2019.

Kiehl, K. and Pfadenhauer, J. (2007) Establishment and persistence of target species in newly created calcareous grasslands on former arable fields. *Plant Ecology*, 189(1), pp. 31–48.

Klaus, V. H. (2013) Urban grassland restoration: A neglected opportunity for biodiversity conservation. *Restoration Ecology*, 21(6), pp. 665–669.

Knapp, S., Kühn, I., Stolle, J., and Klotz, S. (2009) Changes in the functional composition of a Central European urban flora over three centuries. *Perspectives in Plant Ecology, Evolution and Systematics*, 12(3), pp. 235–244.

Kowarik, I. (2011) Novel urban ecosystems, biodiversity, and conservation. *Environmental Pollution*. Elsevier Ltd, 159(8–9), pp. 1974–1983.

Krogstad, T., Børresen, T., and Almås, Å. R. (2018) Field and laboratory methods for the analysis of soil. *Norwegian University of Life Sciences*.

Land Resources Management Unit (2010) Map of soil pH in Europe [map]. Institute for Environment & Sustainability, European Commission Joint Research Centre. <https://esdac.jrc.ec.europa.eu/public_path/ph2.png> Accessed April 29, 2019.

Lin, B. B. and Fuller, R. A. (2013) FORUM: Sharing or sparing? How should we grow the world's cities? *Journal of Applied Ecology*, 50(5), pp. 1161–1168.

Loder, A. (2014) "There's a meadow outside my workplace": A phenomenological exploration of aesthetics and green roofs in Chicago and Toronto. *Landscape and Urban Planning*. Elsevier B.V., 126, pp. 94–106.

Madre, F., Vergnes, A., Machon, N., and Clergeau, P. (2014) Green roofs as habitats for wild plant species in urban landscapes: First insights from a large-scale sampling. *Landscape and Urban Planning*, 122, pp. 100–107.

Maskell, L. C., Smart, S. M., Bullock, J. M., Thompson, K., and Stevens, C.J. (2010) Nitrogen deposition causes widespread loss of species richness in British habitats. *Global Change Biology*, 16(2), pp. 671–679.

Meteorologisk Institutt (2019) Været som var (detaljert) – Oslo (Blindern) målestasjon, Oslo (Oslo). Yr.no <https://www.yr.no/sted/Norge/Oslo/Oslo/Oslo_(Blindern)_målestasjon/detaljert_statistikk.html> Accessed on April 3, 2019

Miljødirektoratet (2018) Naturbase Kart. <http://kart.naturbase.no> Accessed March 31st 2018.

Miljødirektoratet (2019) Analysemetode. *Vannmiljø - Visning av importformat og kodeverk*. <https://vannmiljokoder.miljodirektoratet.no/AnalysisMethod.aspx> Accessed April 29, 2019.

Nagase, A. and Dunnett, N. (2010) Drought tolerance in different vegetation types for extensive green roofs: Effects of watering and diversity. *Landscape and Urban Planning*. Elsevier B.V., 97(4), pp. 318–327.

Nielsen, A., Reitan, T., Sydenham, M., Folk, A., Flydal, K., Stange, E., Dahle, B., Jørgensen, R. R., Fuchs, D., Asmervik, I. F., and Colman, J. E. (no date) Pollinators in the City - Inclusive solutions in practice. *Unpublished manuscript*.

Olsen, T. T. (no date) Slåttemark i Maridalen. *Maridalens Venner*. <http://www.maridalensvenner.no/slaattemark-i-maridalen.162833-25771.html> Accessed February 26, 2019.

Paker, Y., Yom-Tov, Y., Alon-Mozes, T., and Barnea, A. (2014) The effect of plant richness and urban garden structure on bird species richness, diversity and community structure. *Landscape and Urban Planning*. Elsevier B.V., 122, pp. 186–195.

Pyšek, P. (1998) Alien and Native Species in Central European Urban Floras: A Quantitative Comparison. *Journal of Biogeography*, 25(1), pp. 155–163.

R Core Team (2017) R: A language and environment for statistical computing. Vienna, Austria: R Foundation for Statistical Computing. <http://www.R-project.org/>

Schär, C., Vidale, P. L., Lüthi, D., Frei, C., Häberli, C., Liniger, M. A., and Appenzeller, C. (2004) The role of increasing temperature variability in European summer. *Nature*, 427(January), pp. 332–336.

Solstad H and Elven R (2015) Karplanter (Pteriodophyta, Pinophyta og Magnoliophyta). Norsk rødliste for arter 2015. Artsdatabanken <http://www.artsdatabanken.no/Rodliste/Artsgruppene/Karplanter> Accessed March 9, 2019

Steffan-Dewenter, I. and Tscharntke, T. (2002) Insect communities and biotic interactions on fragmented calcareous grasslands - A mini review. *Biological Conservation*, 104(3), pp. 275–284.

Terratec (2011) *Oslo omløp 2011* [air photo]. 1:50,000. Photo LOM0002, Lysaker, Norway.

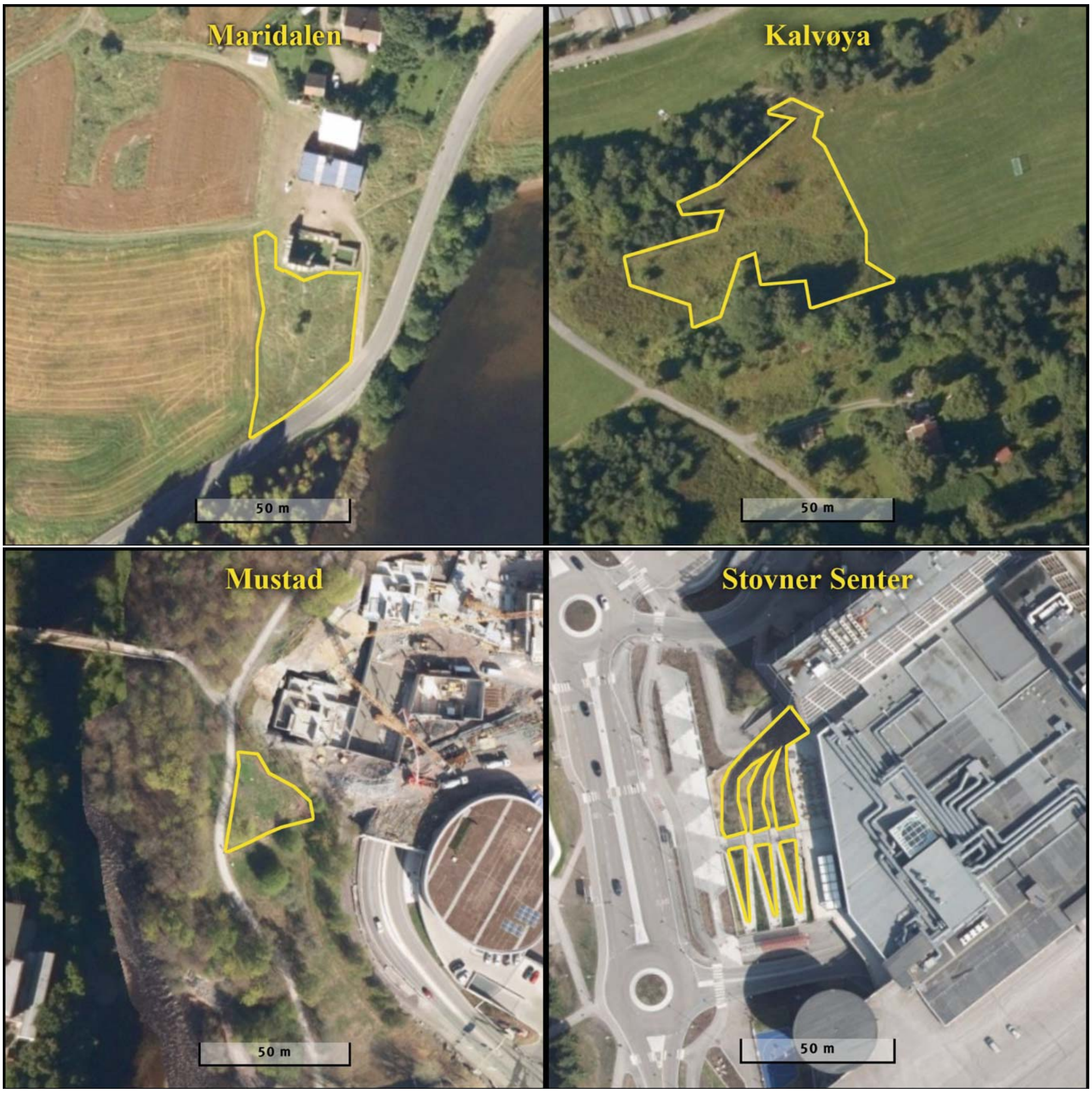
United Nations. (2012) World urbanization prospects: the 2011 revision. *United Nations*, New York, New York, USA.

Vevle, O. (1985) Økologiske faktortal for norske karplantar: arbeidsnotat. Bø, Norge.
Willems, J. H. (1987) Kalkgrasland in Zuid-Limburg. Wetenschappelijke Mededelingen KNNV, Hoogwoud, Nr. 184 (with summary in English).
Willems, J. H. (2001) Problems, approaches, and results in restoration of Dutch calcareous grassland during the last 30 years. *Restoration Ecology*, 9(2), pp. 147–154.
Øien, A. and Krogstad, T. (1987) Øvelser i jordanalyser. *NLH*.

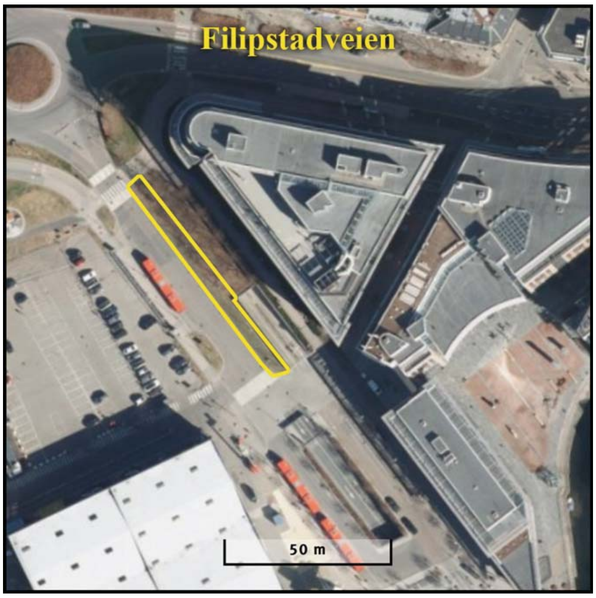
Supplementary material

Supplementary Figure 1

An aerial overview of each site used in the study with meadow boundaries highlighted in yellow. Scale is approximately comparable between images. Maps © Kartverket 2019.




Fornebu S
50 m
Lillestrøm
50 m
Akershus
50 m
Brynseng
50 m
Langkaia
50 m
Elgsletta
50 m



Supplementary Table 1

All mean measures for soil values except for grain size distribution, as well as averages of Shannon diversity and species richness for samples arranged by: **A.** *Type* of meadow and **B.** *Soil Origin*. P-values displayed in italicized gray except for a comparative category, which all other values are in comparison to. **Legend:** Solid orange down-arrow = low value; solid blue up-arrow = high value (two = very high); gray circle = medium value.

A.

Type	Sites	pH	PO_4-P (mg/kg)	NO_3 (mg/kg)	NH_4 (mg/kg)	Dryness (%)	% Barren	Shannon diversity	Richness
Flower	Akershus Brynseng Elgsletta Filipstadveien Langkaia	○ 6.68 *NA*	○ 136.65 *NA*	○ 4.86 *NA*	○ 5.29 *NA*	○ 79.90 *NA*	○ 24.3 *NA*	○ 0.8129 *NA*	○ 4.57 *NA*
Calcareous	Fornebu S Kalvøya Mustad	○ 6.46 *0.427*	○ 189.00 *0.386556*	○ 3.33 *0.6468*	○ 8.99 *0.4665*	○ 74.92 *0.0588*	↑ 34.9 *0.00935 **	↑ 0.9442 *0.0437 *	↑ 5.55 *0.00148 **
Hay	Maridalen Lillestrøm Stovner S	○ 6.62 *0.829*	○ 126.83 *0.870443*	○ 4.20 *0.8434*	○ 15.11 *0.0583*	○ 77.17 *0.2925*	○ 21.9 *0.54928*	↑↑ 1.2124 *2.25e-09* ****	↑↑ 7.18 *1.84e-15* ****

B.

Soil origin	Sites	pH	PO_4-P (mg/kg)	NO_3 (mg/kg)	NH_4 (mg/kg)	Dryness (%)	% Barren	Shannon diversity	Richness
Commercial	Langkaia Lillestrøm Stovner S	○ 6.96 *NA*	○ 124.67 *NA*	↓ 0.61 *NA*	○ 3.04 *NA*	↑ 83.67 *NA*	○ 22.5 *NA*	↑ 1.1373 *NA*	↑ 6.81 *NA*
Local	Fornebu S Mustad	○ 6.89 *0.847*	↑ 275.25 *0.03795 *	○ 2.32 *0.015436 *	○ 4.99 *0.7545*	○ 76.38 *0.015436 *	↑ 45.1 *6.14e-06* ****	○ 0.9467 *0.0234 *	○ 5.70 *0.0106 *
Unmoved	Akershus Brynseng Elgsletta Filipstadveien Kalvøya Maridalen	↓ 6.33 *0.017 *	○ 117.71 *0.89882*	↑ 6.73 *0.000582* ****	↑ 13.27 *0.0389 *	○ 75.33 *0.000582* ****	○ 22.5 *1*	↓ 0.8715 *5.25e-05* ****	↓ 4.87 *7.55e-10* ****

Supplementary Table 2

Coefficients and estimates of effect of site-wide and sample-specific variables on Shannon diversity and richness calculated by GLM. Columns compare the strength of effects when looking at all sites, at only recently-created sites (3 years or younger), or only sites belonging to the hay meadow or calcareous meadow categories. P-values displayed in italicized gray. **Legend:** Solid orange down-arrow = significant negative effect (two = stronger negative effect); solid blue up-arrow = significant positive effect (two = stronger positive effect); gray circle = no significant effect.

Variable	All sites		New sites (≤ 3 years) only		Semi-natural meadows only	
	Diversity	Richness	Diversity	Richness	Diversity	Richness
pH	↑ 1.24E-01 0.00285 **	↑ 1.77E-01 2.24e-06 ***	↑↑ 3.32E-01 2.87e-05 ***	↑↑ 3.85E-01 7.69e-09 ***	↑ 1.55E-01 0.00174 **	↑ 1.98E-01 4.91e-07 ***
PO_4-P (mg/kg)	↑ 7.49E-04 0.000157 ***	↑ 9.18E-04 8.09e-09 ***	↑ 8.69E-04 0.000203 ***	↑ 9.89E-04 5.32e-08 ***	↑ 8.52E-04 0.000589 ***	↑ 9.30E-04 8.24e-08 ***
NO_3 (mg/kg)	↓ -1.66E-02 0.000386 ***	↓ -2.22E-02 1.19e-06 ***	↓ -1.87E-02 0.000438 ***	↓ -2.44E-02 2.45e-06 ***	○ -1.81E-02 0.0936	↓ -2.48E-02 0.00309 **
NH_4 (mg/kg)	○ -2.34E-03 0.4	○ -4.91E-03 0.0501	○ -7.50E-03 0.589	○ -1.54E-02 0.174	↓ -9.22E-03 0.00394 **	↓ -1.27E-02 2.32e-06 ***
Dryness (%)	○ 2.79E-03 0.5838	○ 3.22E-02 0.22	○ 1.59E-04 0.985	○ 6.92E-04 0.92163	↑ 2.16E-02 0.00105 **	↑ 2.57E-02 4.76e-07 ***
% Barren	↓ -3.34E-03 0.000641 ***	↓ -2.23E-03 0.00975 **	↓ -4.26E-03 0.000414 ***	↓ -2.61E-03 0.01 *	↓ -5.32E-03 9.19e-05 ***	↓ -3.66E-03 0.00058 ***
Julian date	↓ -7.69E-03 <2e-16 ***	○ -4.75E-04 0.539	↓ -7.16E-03 6.17e-12 ***	○ 9.37E-05 0.916	↓ -8.76E-03 3.89e-12 ***	○ -9.04E-04 0.369
Site area	↓ -1.039e-04 0.0243 *	↓ -1.462e-04 0.000344 ***	↓ -2.396e-04 0.00419 **	↓ -2.791e-04 0.000159 ***	↓↓ -3.502e-04 1.81e-08 ***	↓↓ -4.173e-04 1.67e-15 ***
Weeded (yes)	↑ 4.28E-01 3.01e-14 ***	↑ 5.02E-01 <2e-16 ***	↑ 5.15E-01 1.78e-15 ***	↑ 6.12E-01 <2e-16 ***	↑ 5.13E-01 3.77E-11 ***	↑ 5.79E-01 <2e-16 ***
Watered (yes)	○ 5.91E-02 0.353	○ 6.66E-02 0.213	○ 4.52E-02 0.528	○ 2.99E-02 0.603	↑ 3.38E-01 0.00204 **	↑ 3.14E-01 0.000101 ***

Supplementary Table 3

All counts of plant species, as well as genera were species could not be determined with certainty. Conservation status is also listed. The 18 non-native species included in the "Non-native" group for analysis are highlighted in yellow on the right, while red-listed species are highlighted in green.

Species	Maridalen	Kalvøya	Mustad	Stovner Senter	Fornebu S	Lillestrøm	Akershus	Brynseng	Langkaia	Elgsletta	Filipstad-veien	Total	Status
Acer platanoides	0	0	0	0	0	0	0	1	0	0	0	1	LC
Achillea millefolium	20	5	5	3	1	10	8	7	1	15	6	81	LC
Achillea ptarmica	0	0	0	2	0	0	0	0	0	0	0	2	LC
Acinos arvensis	0	8	0	0	0	0	0	0	0	0	0	8	LC
Aegopodium podagraria	0	0	10	0	0	0	0	1	0	0	0	11	LC
Agrimonia eupatoria	0	2	2	0	0	0	0	0	0	0	0	4	LC
Alchemilla monticola	1	0	0	0	0	0	0	0	0	0	0	1	LC
Alliaria petiolata	0	0	1	0	0	0	3	0	0	0	0	4	LC
Allium oleraceum	0	1	0	0	0	0	0	0	0	0	0	1	LC
Allium schoenoprasum	0	0	0	0	2	0	0	0	0	0	0	2	LC
Anchusa officinalis	0	0	0	0	0	0	1	0	0	0	0	1	LC
Anemone ranunculoides	0	0	0	0	0	0	0	1	0	0	0	1	LC

Species	Maridalen	Kalvøya	Mustad	Stovner Senter	Fornebu S	Lillestrøm	Akershus	Brynseng	Langkaia	Elgsletta	Filipstad-veien	Total	Status
Anthriscus sylvestris	2	0	1	0	0	0	4	0	0	0	0	7	LC
Arctium tomentosum	0	0	0	0	0	0	0	0	0	1	0	1	SE
Artemisia vulgaris	0	0	3	3	2	1	9	1	0	1	6	26	LC
Ballota nigra	0	0	0	0	0	0	1	0	0	0	0	1	N/A
Barbarea vulgaris	0	0	0	1	0	3	6	0	0	0	0	10	SE
Berteroa incana	0	0	0	0	1	5	0	0	0	0	0	6	SE
Campanula rapunculoides	0	0	1	0	0	0	0	0	0	0	0	1	PH
Campanula rotundifolia	4	0	0	0	0	0	1	0	4	0	0	9	LC
Capsella bursa-pastoris	0	0	0	0	0	1	0	2	0	2	7	12	LC
Carum carvi	0	0	0	0	0	0	1	0	0	0	0	1	LC
Centaurea scabiosa	0	0	1	0	0	1	0	0	1	0	0	3	LC
Chaenorhinum minus	0	0	0	0	0	0	0	1	0	0	0	1	PH
Chamerion angustifolium	0	0	1	0	0	1	0	0	0	0	0	2	LC
Chenopodium album	0	0	1	0	1	2	0	3	0	0	0	7	LC
Cirsium arvense	0	5	2	13	0	0	3	3	0	0	0	26	LC
Convolvulus arvensis	0	0	6	3	0	0	0	0	0	0	0	9	LC
Cota tinctoria	0	0	2	7	0	19	0	0	0	0	0	28	LC
Dianthus deltoides	1	0	0	0	0	0	0	1	0	0	0	2	LC
Digitalis purpurea	0	0	0	0	0	1	0	0	0	0	0	1	LC
Epilobium spp.	0	1	0	0	0	12	0	1	0	0	0	14	LC
Erodium cicutarium	0	1	0	0	0	0	0	0	0	0	0	1	LC
Filipendula ulmaria	0	1	0	0	0	0	0	0	0	0	0	1	LC
Fragaria vesca	0	10	0	0	0	0	1	0	0	0	0	11	LC
Fraxinus excelsior	0	0	3	0	0	0	1	0	0	0	0	4	VU
Galinsoga quadriradiata	0	0	0	0	1	0	0	0	0	0	0	1	PH
Galium boreale	3	1	0	0	0	0	0	0	0	0	0	4	LC
Galium spp.	1	16	15	17	0	20	3	3	3	0	0	78	N/A
Galium verum	11	12	0	2	0	3	0	0	1	0	0	29	LC
Geranium robertianum	0	0	1	0	0	0	4	0	0	0	0	5	LC
Geum urbanum	0	0	9	0	0	3	0	1	0	0	0	13	LC
Glechoma hederacea	0	8	4	0	0	0	0	0	0	1	3	16	LC
Gnaphalium uliginosum	0	0	0	0	0	1	0	0	0	0	2	3	LC
Hieracium murorum agg.	0	0	0	0	0	0	0	1	0	0	0	1	NE
Hieracium umbellatum	7	0	0	0	0	0	0	0	1	0	0	8	NE
Hylotelephium maximum	12	2	5	0	2	0	0	1	0	0	0	22	LC
Hypericum perforatum	2	7	8	2	0	0	1	0	4	0	2	26	LC
Hypochaeris	0	0	0	0	1	1	0	0	0	0	0	2	N/A
Juncus articulatus	0	0	0	0	0	1	0	0	0	0	0	1	LC
Knautia arvensis	3	0	0	0	0	0	0	0	0	0	0	3	LC
Lamium album	0	0	0	0	0	0	0	0	1	0	0	1	LC
Lathyrus pratensis	0	0	2	17	0	0	0	1	1	0	0	21	LC
Lathyrus sylvestris	3	0	0	0	0	0	0	0	0	0	0	3	LC
Lavandula angustifolia	0	0	0	0	0	0	0	0	1	0	0	1	NK
Lepidotheca suaveolens	0	0	0	0	0	1	0	0	0	0	3	4	PH
Leucanthemum ×superbum	0	0	0	0	0	0	1	0	0	0	0	1	LO
Leucanthemum vulgare	1	0	0	13	0	18	18	11	27	0	1	89	LC
Linaria vulgaris	3	2	7	0	0	0	0	0	0	0	0	12	LC
Lipandra polysperma	0	0	0	0	0	0	0	1	0	0	0	1	PH
Lotus corniculatus	0	1	0	0	0	0	0	14	2	0	1	18	LC
Lychnis flos-cuculi	0	0	0	0	0	0	3	0	0	0	0	3	LC
Malus ×domestica	0	0	0	0	0	8	0	0	0	0	0	8	NR
Medicago lupulina	0	0	7	0	1	0	0	1	1	0	4	14	LC
Melilotus officinalis	0	0	0	2	0	0	0	0	0	0	0	2	SE
Parthenocissus quinquefolia	0	0	0	0	0	0	1	0	0	0	0	1	HI
Phedimus hybridus	0	0	0	0	1	0	0	0	0	0	0	1	SE
Phedimus spurius	0	0	0	0	4	0	0	0	0	0	0	4	SE
Pimpinella saxifraga	4	2	0	0	0	0	0	0	0	0	0	6	LC
Plantago lanceolata	4	0	0	0	0	0	0	0	0	0	0	4	LC
Plantago major	0	0	0	0	0	7	0	1	2	1	7	18	LC
Polygonatum odoratum	0	0	1	0	0	0	0	0	0	0	0	1	LC
Polygonum aviculare	0	0	0	0	0	0	0	1	0	2	4	7	LC
Potentilla anserina anserina	0	0	0	11	0	0	0	0	0	0	0	11	LC
Potentilla argentea	0	3	5	0	0	0	0	0	0	0	0	8	LC

Species	Maridalen	Kalvøya	Mustad	Stovner Senter	Fornebu S	Lillestrøm	Akershus	Brynseng	Langkaia	Elgsletta	Filipstad-veien	Total	Status
Potentilla erecta	1	0	0	0	0	0	0	0	0	0	0	1	LC
Potentilla norvegica norvegica	0	3	0	0	0	2	0	0	0	0	0	5	LC
Potentilla thuringiaca	0	0	13	0	0	0	0	0	0	0	0	13	PH
Ranunculus acris acris	0	0	0	9	0	0	1	0	0	0	0	10	LC
Ranunculus repens	1	0	1	2	0	1	0	0	0	0	0	5	LC
Rorippa palustris	0	0	0	0	0	8	0	0	0	0	0	8	LC
Rorippa sylvestris	0	0	0	0	0	2	0	0	0	5	0	7	LC
Rosa dumalis	8	1	0	0	1	0	0	0	0	0	0	10	LC
Rosa mollis	0	1	0	1	0	0	0	0	0	0	0	2	LC
Rubus idaeus	0	9	12	0	0	0	0	0	0	0	0	21	LC
Rumex acetosa	3	3	0	0	0	0	0	0	0	0	0	6	LC
Rumex longifolius	0	0	0	6	0	1	0	0	0	2	0	9	LC
Rumex obtusifolius	0	0	0	2	0	0	0	0	0	0	0	2	LC
Scorzoneroides autumnalis	0	0	0	0	0	5	0	2	0	0	15	22	LC
Scrophularia nodosa	0	1	2	0	0	0	0	0	0	0	0	3	LC
Sedum acre	0	0	0	0	24	0	0	0	0	0	0	24	LC
Sedum album	0	0	0	0	24	0	0	0	0	0	0	24	LC
Seseli libanotis	0	2	7	0	0	0	1	0	0	0	0	10	LC
Silene dioica	0	0	0	2	0	6	1	1	1	0	0	11	LC
Silene vulgaris	0	0	1	23	0	22	5	16	1	0	0	68	LC
Solidago canadensis	0	0	7	3	0	0	0	1	1	0	0	12	SE
Sonchus arvensis	0	0	0	0	1	0	0	0	0	0	0	1	LC
Sonchus oleraceus	0	0	0	0	1	0	1	0	0	0	3	5	LC
Stachys sylvatica	0	0	0	1	0	0	0	0	0	0	0	1	LC
Stellaria graminea	0	0	3	20	0	2	0	3	0	0	0	28	LC
Stellaria media	0	0	0	0	1	0	1	0	0	1	5	8	LC
Succisa pratensis	0	0	0	0	0	1	0	0	0	0	0	1	LC
Symphytum officinale	0	1	0	0	0	0	0	0	0	0	0	1	SE
Tanacetum vulgare	0	0	0	0	0	0	0	2	0	0	0	2	LC
Taraxacum spp.	1	0	0	1	2	3	19	11	5	4	27	73	N/A
Thymus pulegioides	1	0	0	0	0	0	0	0	0	0	0	1	LC
Trifolium aureum	0	0	0	0	0	3	0	0	0	0	0	3	LC
Trifolium hybridum hybridum	0	0	2	2	0	0	0	0	0	0	0	4	LC
Trifolium medium	4	0	0	0	0	0	0	0	0	0	0	4	LC
Trifolium pratense	1	0	4	15	1	13	10	13	18	28	23	126	LC
Trifolium repens	3	1	1	3	0	11	4	1	0	0	5	29	LC
Tripleurospermum inodorum	0	0	0	0	0	2	1	0	0	0	0	3	LC
Tussilago farfara	0	0	1	0	0	3	1	4	0	0	0	9	LC
Ulmus glabra	0	0	1	0	0	0	0	2	0	0	0	3	VU
Urtica spp.	0	0	9	1	0	3	2	0	1	0	0	16	N/A
Verbascum nigrum	0	0	12	1	0	0	0	0	0	0	0	13	LC
Verbascum thapsus	0	0	1	0	0	0	0	0	0	0	0	1	LC
Veronica chamaedrys	0	0	4	0	0	0	0	0	0	0	0	4	LC
Veronica spicata	0	4	0	0	0	0	0	0	0	0	0	4	VU
Vicia spp.	3	5	7	14	0	1	0	2	1	0	1	34	N/A
Viola spp.	1	0	0	0	0	0	0	0	0	0	0	1	N/A
Viscaria vulgaris	0	0	0	0	0	0	0	1	0	0	0	1	LC

Supplementary Table 4

All counts of plant family and non-native plants regardless of family.

Family	Maridalen	Kalvøya	Mustad	Stovner Senter	Fornebu S	Lillestrøm	Akershus	Brynseng	Langkaia	Elgsletta	Filipstad-veien	Total
Amaranthaceae	0	0	1	0	1	2	0	3	0	0	0	7
Amaryllidaceae	0	1	0	0	2	0	0	0	0	0	0	3
Apiaceae	6	4	18	0	0	0	6	1	0	0	0	35
Asparagaceae	0	0	1	0	0	0	0	0	0	0	0	1
Asteraceae	32	10	14	42	8	65	60	42	35	20	60	388
Boraginaceae	0	0	0	0	0	0	1	0	0	0	0	1
Brassicaceae	0	0	1	0	0	11	3	2	0	7	7	31
Campanulaceae	4	0	0	0	0	0	1	0	4	0	0	9
Caryophyllaceae	1	0	4	45	1	30	10	22	2	1	5	121
Convolvulaceae	0	0	6	3	0	0	0	0	0	0	0	9
Crassulaceae	12	2	5	0	50	0	0	1	0	0	0	70
Fabaceae	14	7	23	51	2	28	14	32	23	28	34	256
Fabaceae minus *T. pratense*	13	7	19	36	1	15	4	19	5	0	11	130
Geraniaceae	0	1	1	0	0	0	4	0	0	0	0	6
Hypericaceae	2	7	8	2	0	0	1	0	4	0	2	26
Juncaceae	0	0	0	0	0	1	0	0	0	0	0	1
Lamiaceae	1	16	4	1	0	0	1	0	1	1	3	28
Oleaceae	0	0	3	0	0	0	1	0	0	0	0	4
Onagraceae	0	1	1	0	0	13	0	1	0	0	0	16
Plantaginaceae	7	6	23	1	0	8	0	1	2	1	7	56
Polygonaceae	3	3	0	8	0	1	0	1	0	4	4	24
Ranunculaceae	1	0	1	11	0	1	1	1	0	0	0	16
Rosaceae	10	30	28	12	1	5	1	1	0	0	0	88
Rubiaceae	15	29	15	19	0	23	3	3	4	0	0	111
Sapindaceae	0	0	0	0	0	0	0	1	0	0	0	1
Scrophulariaceae	0	1	3	0	0	0	0	0	0	0	0	4
Ulmaceae	0	0	1	0	0	0	0	2	0	0	0	3
Urticaceae	0	0	9	1	0	3	2	0	1	0	0	16
Violaceae	1	0	0	0	0	0	0	0	0	0	0	1
Non-native	0	1	21	6	7	17	8	3	2	1	3	69

Norges miljø- og biovitenskapelige universitet
Noregs miljø- og biovitskapelege universitet
Norwegian University of Life Sciences

Postboks 5003
NO-1432 Ås
Norway