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HOW TO BEST CREATE OR RESTORE WILD MEADOWS IN AN URBAN SETTING

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Ecology

Acknowledgments

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ABSTRACT

Anthropogenic activities associated with increasing influx of people in cities globally and the capital city of Norway, Oslo, has expansively fragment and decreased urban green space, reducing the connectivity between the remaining remnant green patches. Consequently, this contributes to decreasing the biodiversity and increasing the spread of invasive species in Oslo and cities worldwide. Oslo has begun the establishment of native meadow patches on and off green roofs as steppingstone to mitigate anthropogenic encroachment in Oslo, and hence, improve biodiversity in the city. However, native meadow assemblages on small patches, such as roofs are affected by multiple negative factors, for instance, extreme environmental conditions and production of high volumes of minerals, which causes intensive drought conditions. To mitigate these effects and minimise the creation of ecological traps and sinks, we conducted an experimental field test to investigate methods for creating native meadows on green roofs in an urban setting (case study; Oslo city Norway). An exact number of native meadow plant seeds was sown in various plots, the effect of three different soil treatments (100% topsoil(organic soil from Oslo compost), 50% topsoil and 50% sand, 95% topsoil and 5% silt), and two substrate depths (18 cm and 30 cm) were tested, on the sprouting success and biomass of sown meadow species, colonizers, and sprouting number of invasive species. The sprouts and biomass of sown species appeared higher for 30 versus 18 cm for all soil types, but the pattern was not statically significant. Averagely, topsoil and 30 cm depth had the highest number of sprouts and dry weight of sown species and topsoil mixed with sand and 18 cm depth had the lowest number of sprouts and dry weight. Colonizing species (those that appeared but we did not sow in ourselves) exhibited a significantly lower number of sprouts in 30 versus 18 cm for all soil types, with highest in topsoil and lowest in topsoil mixed with silt. The biomass of colonizing species revealed similar results with highest in 18 cm substrate depth of topsoil and lowest in 30 cm depth of topsoil mixed with sand. The number of sprouts of invasive species appeared to be higher in 30 compared to 18 cm for topsoil and topsoil mixed with silt and decreased in 30 versus 18 cm for topsoil mixed with sand, though the patterns were not statically significant. The biomass of invasive species showed a higher amount in 30 compared to 18 cm, but lower for topsoil mixed with sand and the patterns was also not statically significant. There was no statistically significant interaction between substrate depth and soil treatments on the sprout and biomass of all meadow species. Meaning that both substrate depth and soil treatment influenced the sprouts and biomass of these species in our experiment independently. These results indicate that soil quality was the main driver for the total number

of sprouts and biomass combined for all species registered in this field experiment. Meanwhile, substrate depth has little role to play in determining the number of sprouts and biomass of sown meadow species and invasive species yet influenced the colonization of our site. Additional factors such as surrounding green space, shading from trees, seed banks, phenological differences between species, seed dispersal by birds, insects, and wind, could be potential sources of colonization, influencing sprouting, growth, and ultimately, gene flow and genetic rescue in our experiment.

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INTRODUCTION

Over the past decades, the human population has observed a tremendous increase globally, most pronounced in urban areas. The welling human population density simultaneously drives an escalating use of urban lands for the construction of buildings, roads, railways, and more, leading to a degradation of the urban matrix into fewer, smaller and more isolated green patches (Van Rossum, 2010). Consequently, the biodiversity and native plant communities in urban areas worldwide is decreasing (Kendal et al., 2012, Pataki, 2015, Williams et al., 2009). This also promotes species that are adapted to disturbance, allowing for their dominance over other, perhaps native urban plant communities (Kinlocka et al., 2016).

The capital city of Norway, Oslo, experienced a 21% increase in population over the last 10 years, with an estimated increase of 33% of the population by 2040 (Oslo Kommune, 2011). The government decided to protect green areas surrounding the city of Oslo (forest and farmland) as a measure of minimizing urban sprawl, thereby distributing urban growth inside the city area (Nordh and Østby, 2013). Thus, there is an increase of anthropogenic pressure on the remaining remnant green patches in Oslo, including for instance, fragmentation of calcareous meadows native to the region (Evju et al., 2015a). Fortunately, and like few other capital cities around the world, Oslo still maintains numerous green patches (forest, agricultural land, lawns, garden, parks), relatively evenly distributed throughout the urban gradient (Fig.1) (Arnott, 2015). However, given their urban existence, these green patches are confined and constrained within fixed and ecological hostile outer boundaries. Such fixed outer boundaries coupled with increasing demand for infrastructure results in the susceptibility of green patches to, for example, disturbance, destruction and spread of invasive species (Oslo Kommune, 2015). Consequently, negative impacts on green patches, their species composition, size, distribution, and connectivity further influence species of insects (for example, pollinators) and birds, underscoring their value within the larger perspective of “urban ecology”. Conserving existing green patches and establishing new ones can thus have positive consequences for native flora and fauna of the Oslo urban environments. For the background of this thesis, I have gone a deeper dive into urban ecological connectivity for Oslo. The focus of my thesis is the experimental field study presented here, thus, the discussion of the urban matrix in Oslo is presented in a separate section in the appendix.

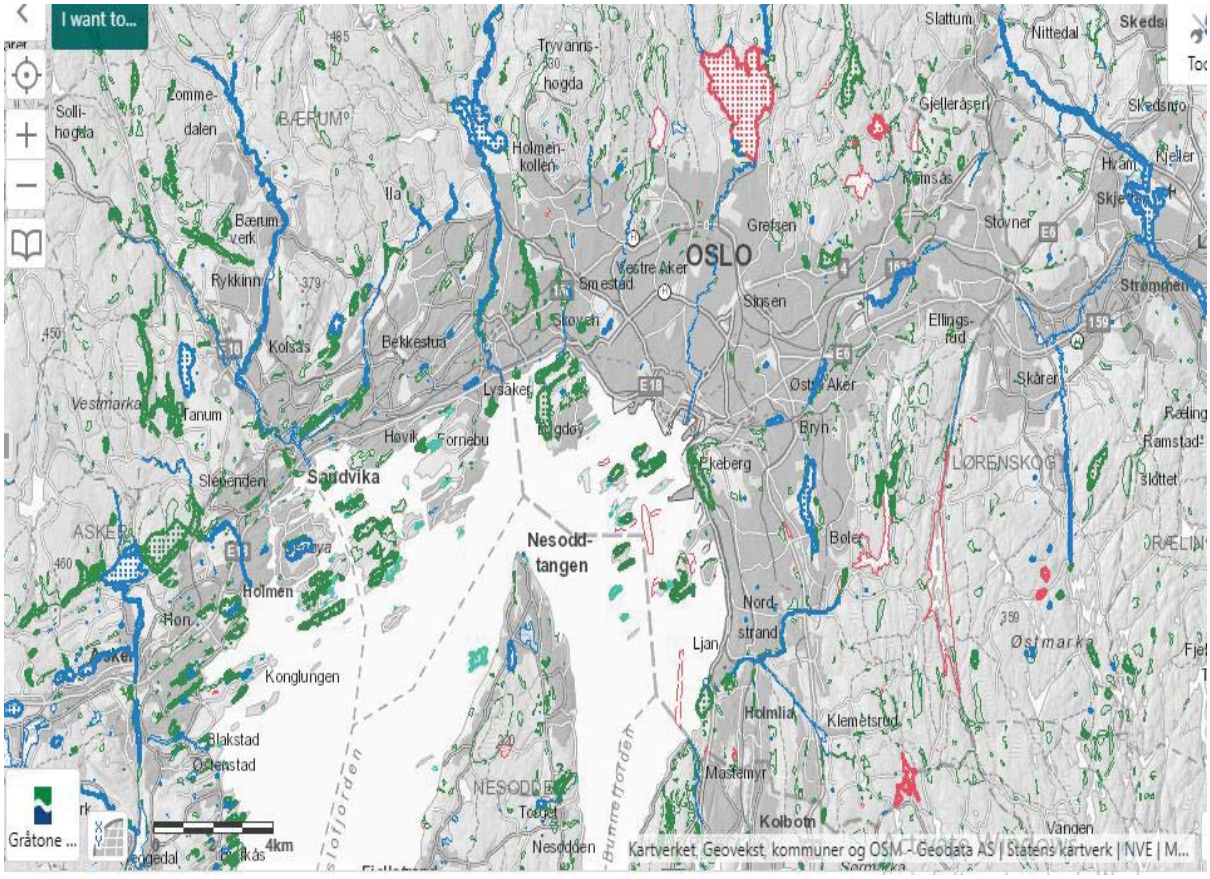

Figure 1: The Oslo region, showing important nature types in and around the city of Oslo, Norway. With numerous dark green patches representing forest, agricultural land, parks, lawn, and gardens. Blue coloured patches for freshwater and wetlands, pale green for coast and sea beach, red for other important nature types (MILJØDIREKTORATET, 2020).

Facilitating the movement of species between urban green patches, sub-urban reservoirs, and the colonisation of new habitats, corridors and/or steppingstones need to be restored or established (Ahern, 2013). One way of creating steppingstones in urban areas is through the establishment of green roofs. This has been practised for many years to improve patch connectivity and mitigate the loss of urban green space (Mayrand and Clergeau, 2018). The establishment of more green roofs will decrease the distance between Oslo urban green patches and increase the interaction between species and within ecosystems (see appendix 3) (Braaker et al., 2014). In this recent era, green roofs have been classified as intensive (>20 cm), simple-intensive (15 to 20 cm) and extensive (6 to 15 cm) based on the depth of their substrates (Landschaftsentwicklung-Landschaftsbau, 2008). Most green roofs are built as extensive green roofs (EGRs) with a characteristic shallow growing medium (depth<15 cm), because of demands concerning building statics and costs (Schroder et al., 2019). Furthermore, the use of green roofs with shallow substrate layers is less expensive, requires little maintenance and incur

less harm on the roof tops (Dunnett and Kingsbury, 2008, Schroder et al., 2019). However, shallow substrate layers often result in extreme environmental conditions, such as limited water availability due to low water retaining capacity, rapid changes in temperature, and increased exposure to wind and solar radiation (Bengtsson et al., 2005). EGRs also produce high volumes of minerals, which causes intensive drought conditions (Dunnett and Kingsbury, 2008, Oberndorfer et al., 2007).

Green roof restorationists have adopted strategies for how to mitigate these extreme drought conditions on EGRs. For instance, the use of different depths of growth medium or meadow seed mixtures (Lundholm et al., 2010, Nagase and Dunnett, 2010, Wolf and Lundholm, 2008). Burghardt et al. (2010) suggest the use of a wide range of native meadow species which are adapted to drought conditions on EGRs. These native species will promote the interactions between specialist native flora and fauna which will increase their species richness on these roofs, thereby, reducing the risk of colonization by invasive species (Cook-Patton and Bauerle, 2012). Bates et al. (2015) found that an optimized established growth medium will reduce drought effects and improve meadow species survival on green roofs. Moreover, Lu et al. (2015) concluded that the interaction between the total amount of water, depth of growth medium, and water retention(storage) layer, are of great significance for biodiversity of meadows on green roofs.

Few studies have investigated means of improving sprout and growth conditions on EGRs, such as comparing substrate depth and soil composition (Molineux et al., 2017, Nagase and Dunnett, 2013). Accordingly, we conducted an experimental field test which tested methods for creating native meadows on green roofs in an urban setting (case study; Oslo city Norway) to improve overall native biodiversity in this area. Sowing an exact number of native meadow plant seeds in various plots, we tested the effect of three different soil treatments (100% topsoil(organic soil from Oslo compost), 50% topsoil and 50% sand, 95% topsoil and 5% silt), and two substrate depths (18 cm and 30 cm), on the sprouting success and biomass of sown meadow species, colonizers, and sprouting number of invasive species. Seeds from native meadow species were collected from local meadows a year in advance. Colonizers were defined as species that sprouted but were not sown in by us. Invasive species were defined according to fremmedartslista from Artdatabanken as species occurring outside their natural range or found to be foreign, i.e. outside the area where the species natural dispersal potential indicates where it should be found (Fremmedartslista, 2018).

Hypothesis

The main purpose of this field experiment was to investigate the effect of soil types and substrate depth on the sprout and biomass of sown meadow species, colonizers, and the sprout of invasive species in each treatment in the first growing season. Registering over additional growing seasons was out of the scope of this study. The background for this is based on the search for improved methods when establishing new green roofs in Oslo to alleviate the effect of fragmentation and improve biodiversity in the urban landscape. The following hypotheses were tested:

1. The 18 cm substrate depth will have high mineral and low water retention capacity, thus leading to drought conditions and will have lower number of sprouts and biomass than the 30 cm depth.

2. The three soil treatments will hold water in different capacities, with the 100% topsoil holding moderate water, 50% sand and 50% topsoil holding the least water, and 95% topsoil with 5% silt holding the most water. Accordingly, 95% topsoil and 5% silt will have the highest number of sprouts and biomass, 100% topsoil will support a moderate number of sprouts and biomass, and 50% sand and 50% topsoil will have the least number of sprouts and lowest biomass.

3. The three soil treatments provide different amounts of organic matter, and hence, there will be higher number of sprouts and biomass in 100% topsoil, moderate number of sprouts and biomass in 95% topsoil and 5% silt and the least number of sprouts and biomass in 50% sand and 50% topsoil.

MATERIALS AND METHODS

Study site and species

This experiment was carried out at Geitmyra garden east of Ullevål sykehus in Oslo, Norway. The garden (matkultursenter) is found at the centre of the city and is partly surrounded by human structures to the south (roads and buildings). One third of the garden`s upper portion borders the Ullevål graveyard, a comparative vegetated area acting as a resource patch for some organisms from the garden (Fig.2). Fifteen seeds from each of twenty-five native Norwegian meadow plant species (Table.1) were collected in august 2018, at Bygdøy and Kalyøya about

10 km west of Oslo. Seeds were selected based on the species with a high amount of seeds that year. The seeds were stored at variable temperatures -4-16^{o}c below freezing for approximately 4 months (ca. 15. November to the 15. March) simulate stratification and vernalisation.

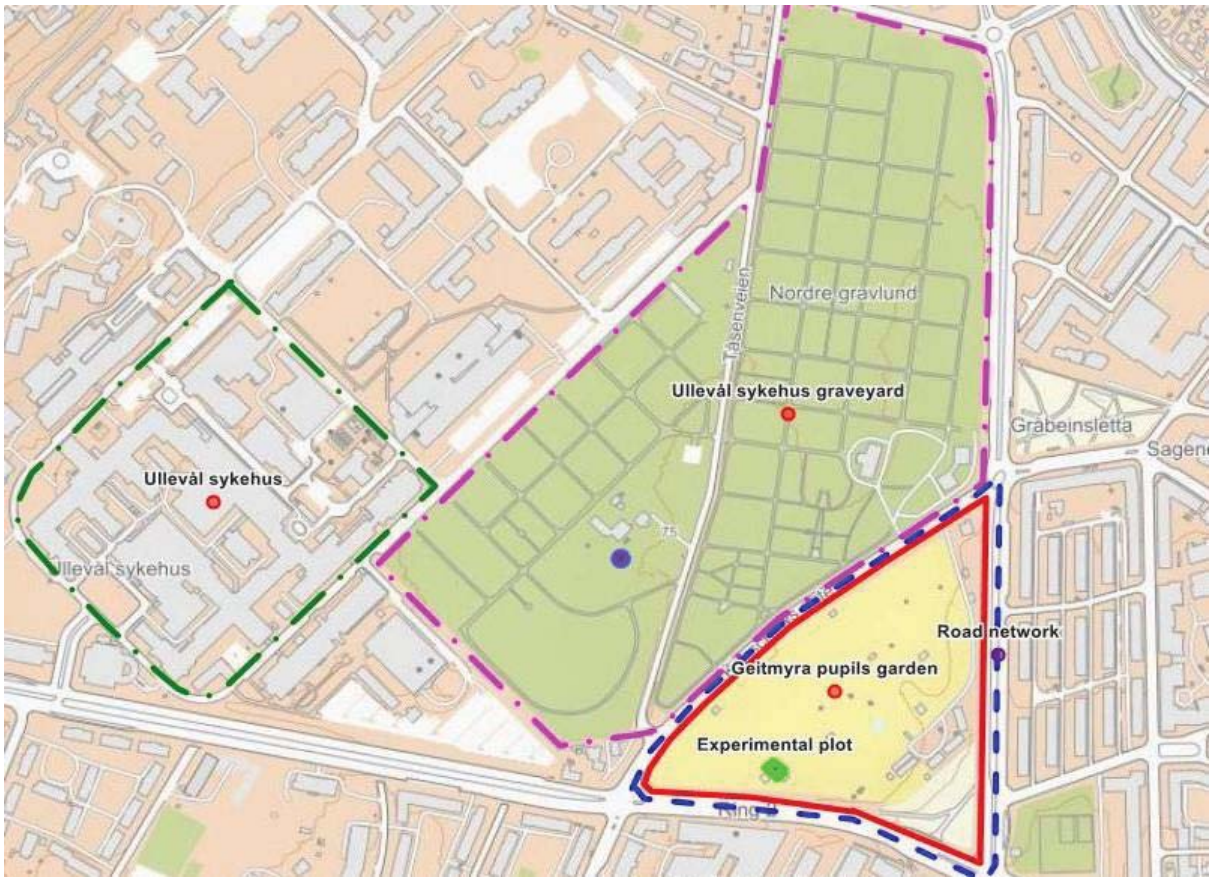

Figure 2: The location of the experimental area and its surroundings in an urban matrix (Geitmyra, Oslo. Pupils garden (matkultursenter))

Table.1: Species list showing family and growth form. We sowed 15 seeds from each species in each plot

Genus	Family	Growth form
Barbarea vulgaris	Brassicaceae	biennial
Festuca ovina	Poaceae	perennial
Hieracium umbellatum	Asteraceae	perennial
Leucanthemum vulgare	Asteraceae	perennial
Phleum pratense	Poaceae	perennial
Silene dioica	Caryophyllaceae	perennial
Trifolium arvense	Fabaceae	annual/biennial
Tripleurospermum inodorum	Asteraceae	annual/biennial
Anthoxanthum odoratum	Poaceae	perennial
Dianthus deltoides	Caryophyllaceae	perennial
Galium verum	Rubiaceae	perennial
Hypericum perforatum	Hypericaceae	perennial
Luzula multiflora	Juncaceae	perennial
Pimpinella saxifraga	Apiaceae	perennial
Trifolium pratense	Fabaceae	perennial
Viscaria vulgaris	Caryophyllaceae	perennial
Lathyrus sylvestris	Fabaceae	perennial
Allium oleraceum	Amaryllidaceae	perennial
Angelica sylvestris	Apiaceae	annual/perennial
Scrophularia nodosa	Scrophulariaceae	perennial
Seseli libanotis	Apiaceae	biennial/perennial
Trifolium medium	Fabaceae	perennial
Pilosella officinarum vaill	Asteraceae	perennial
Solidago virgaurea	Asteraceae	perennial
Trifolium hybridum	Fabaceae	perennial

Experimental design

A 10 m by 4 m plot was established by removing vegetation manually and making the surface flat like that of a rooftop with the use of shovels. A mat the same size as the plot was placed to prevent weeds from growing into the plot. We then used 18 wooden pallets 115 x 75

x 19.5 cm (L x W x H) internal size, 120 x 80 x 19.5 cm (L x W x H) external size placed on the mat. The pallets were placed 40 cm apart to provide space for moving between them and to avoid interactions amongst them and their surroundings. To simulate the drainage system on green roofs, plastics with undulated surfaces and of sizes 111 x 71 cm (L x W) were placed in the bottom of each pallet, allowing approximately 4.5 cm between the plastic edge and the pallet edge (Fig.3). The three soil treatments were 100% topsoil (topsoil), 50% sand mixed with 50% topsoil (topsoil.sand), 95% topsoil mixed with 5% silt (topsoil.silt). The organic topsoil formed the base, and then sand and silt were mixed in with an electric cement mixer before being poured into their respective pallets. These treatments were then allocated into two substrate depths:18 cm (n = 9) and 30 cm (n= 9), with three replicates for each treatment (total number of treatments: 18). All treatments were distributed randomly.

Fifteen seeds from each of the 25 meadow species were sown into each pallet on the 7th of June 2019. A hand rake was used to facilitate "opening" the soil (not more than 5 cm deep) and 15 X 25 seeds were sprinkled evenly on the loose soil. The soil was then lightly compacted by hand to allow contact between the soil and the seeds. We also kept a 10 cm edge from the sides of the pallets without seeds to prevent edge effects within each treatment due to differences in heat intensity and moisture and variation between day and night. A wire grid of 2 mm in diameter was used to prevent cats and predatory birds from defecating, playing, and picking seeds from the pallets. The temperature and moisture content of the treatments were measured using Stevens Hydra probe® water sensor every 5th day. The pallets were watered with 5 litres of water each if the moisture content was below a predefined standard average of 0,120 water fractions by volume (wfv or m^3m^3), measured for three random pallets. Watering was also conducted according to the amount of rainfall. To register the number of individuals for each species that sprouted independent of our experiment, an additional reference pallet was established at our site in which 15 seeds for each meadow species was sown and tagged with their respective numbers and names.

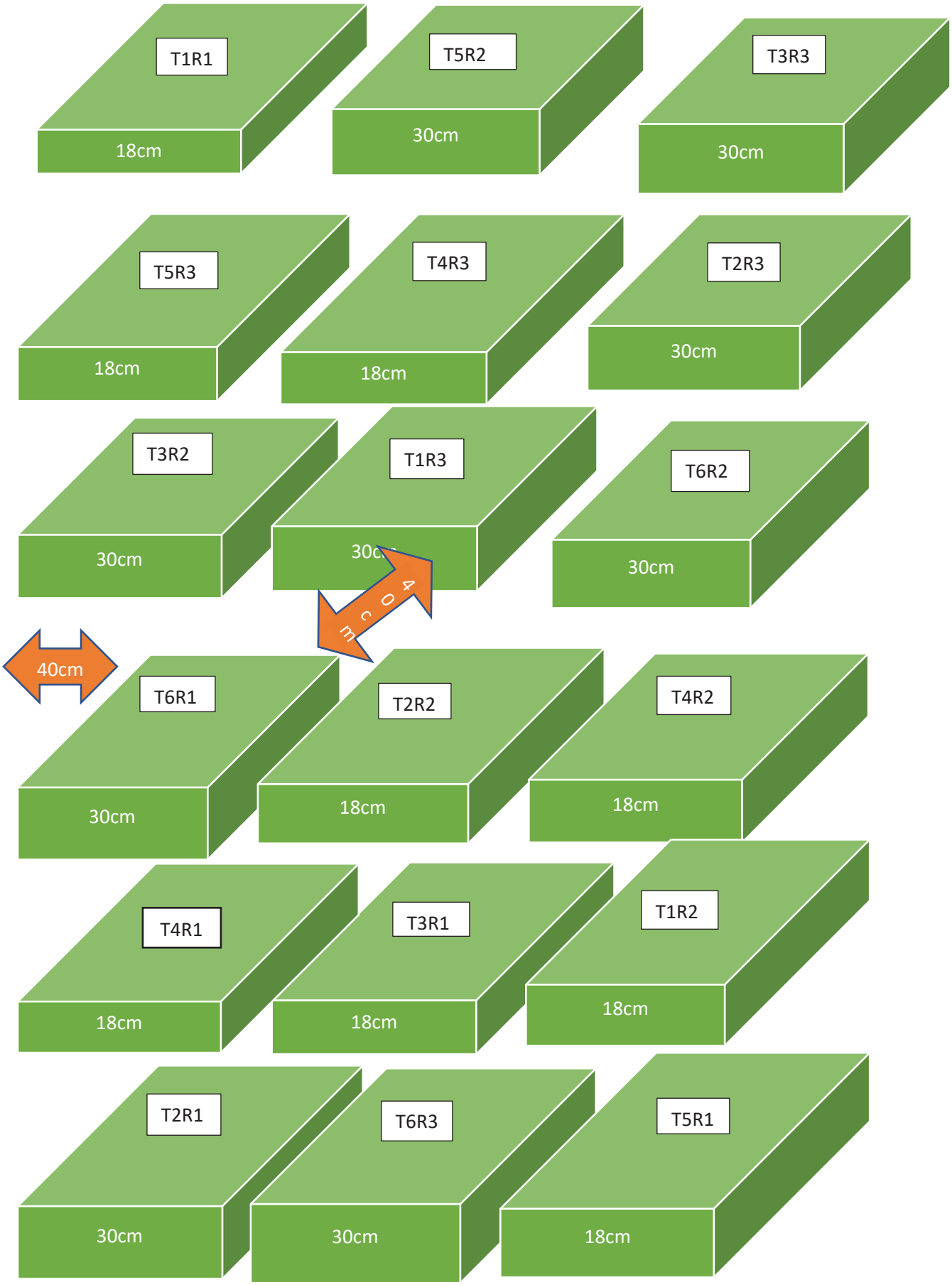

Figure 3: Experimental design with 18 pallets with 40 cm gaps in-between and at the edge used to test two substrate depths (18 and 30 cm), and three soil types: T= treatment (depth and soil type) and R= replicate, randomly placed in a 10 x 4 meter plot.

Field Sampling

Using toothpicks and adhesive tapes in different colours (describing different dates), we tagged, numbered, and dated each sprout for all species every 5th day starting on the 18th of June 2019. By 8th of august 2019, we manually removed *Barbarea vulgaris*, an alien species as defined by the Species Data Bank (2007, 2012, 2018), with the category very high risk (SE) and *Lipandra polysperma* also an invasive species. Individuals of both species were removed manually including the root, thus preventing sprouting from root fragments. Since these two species were removed a few months after recording their sprouts, their biomass was not analysed. On the 24 of October 2019, all the vegetation was cut above ground level and gathered in coded paper bags for each treatment, separate for each replicate. The vegetation in each bag was dried in a drying oven at a temperature of 60^{0}c for 72 hours. The contents were then emptied onto a tared silver dish with an A3 paper beneath. After carefully removing soil particles from the vegetation, the dried weight of each treatment and respective replicates was measured in grams (g).

Soil Analysis

Three 10 litter containers were filled up each with a respective soil type; topsoil, topsoil.sand, and topsoil.silt. These were analysed at the soil science laboratory at Norwegian University of Life Science. On the 7th of November 2019, with the help of prof. Trond Børrensen and Tore Knapstad, we began the measurement of soil water retention characteristics (pF analysis). Four sub-samples were taken from each soil type in the laboratory, along with two additional sub-samples acting as internal standards. The soil samples were saturated and the pF values at 10cm suction (pF 1.0), 0.1 bar pressure (i.e. field capacity, 100 hpa, pF 2), and 1 bar pressure (1000 hpa, pF 3) were analysed following the methods described in Krogstad et al. (2018). The analysis from pF 1.0 to pF 3 were determined according to the description of soil physical methods using a sandbox in the compendium and the weights before and after each pF-value were recorded (appendix 1). The wilting point(15 000 hpa, pF 4,2 or 15 bar), was measured following the description of soil physical methods using a pressure chamber. Weights were recorded at pF 4.2 (appendix 2). Different pF values (soil moisture tension) were calculated by subtracting the weight of a soil sample from the dry weight of the same soil sample. The percentage moisture in the soil was then calculated using the following equation:

[100*(weight of water content at various pF values (g) – weight of the ring(with sample, B and RB)(g)) / (weight of the ring(with sample, B and RB)(g) – weight of the ring, B and RB(g)].

The pore volume was found by adding the water and air volume at pF2. The bulk density was obtained by taking the gross weight of the soil sample minus the weight of the empty cylinder (cylinder weight + two red lids), and then divided with the volume of the cylinder (100 cm^3). The wilting point (pF 4.2) was first calculated as weight % water [100*(weight before drying – weight after drying) / net weight after drying]. The wilting point as vol % was found by taking water content in weight % and multiplying by the dry bulk density of the same soil. These results were presented as percentage categories of pF2 – pF3 (field capacity), pF3 – pF4.2 (plant available water), pF<4.2 (permanent wilting point) of the various soil types.

In collaboration with Irene Dahl from the same soil science laboratory, we sampled the various particle sizes and organic matter in the three treatments. From the three-soil mixtures, we separated approximately two-thirds of each sample into cartons, and then dried them at 55°C for at least 72 hours. These were then sifted through a 2.0mm sieve, and the portion of the sample larger than 2.0mm was discarded. 10 mL of dry, sifted soil were separated into labelled 1 litre beakers. The sifted soil from the beakers, along with two additional soil samples acting as internal standards were then prepared and analysed according to the methods described in Krogstad et al. (2018) by the NMBU soil science laboratory. 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. In accordance with the soil types used in the experiment, we separated the detailed results from the analysis into three broader percentage categories of clay, silt, and sand.

For the organic matter of the three soil mixtures, we weighted 3 to 5g of soil into a previously weighed crucible and dried these in a drying cabinet for at least 6 hours at 105+/-5ºC. This was cooled in an exciccator for 30 minutes and then weighted. The loss on ignition was determined by placing the crucible with the dried soil in a calcinating oven and calcinated for at least 3 hours at 550+/-25ºC. This was again cooled in the exciccator for 30 minutes and weighted. The percentage loss on ignition was then calculated according to the following equation:

[100* (weight of crucible with sample after drying – weight of crucible and sample after calcination / weight of crucible with sample after drying – weight of crucible].

The organic matter content was given as % loss on ignition.

Statistical Analysis

To test for the effect of growth medium depth and soil type on the number of sprout and biomass of sown meadow species, colonizers, and sprouts of invasive species, separate generalized Linear regression models (GLM) were used for each treatment (Crawley, 2012). The number of sprouts and biomass were used as the response variables and soil type and substrate depth as the categorical predictor variables. The number of sprouts and biomass for all species were aggregated with respect to their pallets, prior to any analysis. To test if the sprouts and biomass of sown species, colonizers and sprouts of invasive species differed between substrate depth, soil type and among the pallets, separate linear mixed effect models (LME) (Aiken et al., 1991) were constructed using the nlme package with pallet as random effect (Pinheiro and Bates, 2000). QQ plots were generated for visual inspection of normalities of residual. All statistics were carried out in R environment version 3.5.2 (2018-12-20) "Eggshell Igloo" Copyright (C) 2018.

Map Analysis

To show the green areas in Oslo that are preserved for the conservation of biodiversity, I used the layer function in Naturebase.no to create a map displaying areas mapped in accordance with DN handbook 13 (Fig.1). The map showing the project area (Fig.2) and its surrounding urban matrix was created using Fylkesatlas.no and Artsdatabanken.no was used to check if a species was invasive and their invasive categories (Species Data Bank, 2018). Additional map-work is presented in appendix 3.

RESULTS

One growing season (20 weeks) after sowing 25 native Norwegian meadow species, with 15 seeds per species, we recorded 1533 sprouts from sown seeds, 1326 sprouts of colonizers (*Rorippa sylvestris* was the most common colonizer, and spreads mainly vegetatively), 317 sprouts that we could not identify (excluded in all analyses) and 91 sprouts of invasive species [89 sprouts were *Barbarea vulgaris* (SE) and 2 were *Lipandra polysperma* (PH)] across all pallet collars. Out of the sown species, 5 of them (*Scrophularia nodosa, Lathyrus Sylvestris L., Angelica sylvestris L., Trifolium medium L., Allium oleraceum L)* did not germinate. A total of 43 sown individuals flowered and 9 of these developed seeds.

Dry Weight and Sprout Success of Sown Seeds

On average, the number of sprouts for sown species was higher in 30 compared to 18 cm substrate depth for all soil types, though this pattern was not statistically significant (β =

4.00, P = 0.74, Fig.4A). Topsoil had the highest number of sprouts of sown species, and topsoil.silt had slightly fewer, but not significantly differ from topsoil (β = -17.00, P = 0.18). Topsoil.sand had a significantly lower number of sprouts than both Topsoil(β = -51.33, P = 0.001) and topsoil.silt (β = -17.00, P = 0.18). There was little variation in the number of sprouts amongst pallets for all soil types and substrate depths except for 30 cm depth of topsoil, showing variation in the number of sprouts (Fig.4A).

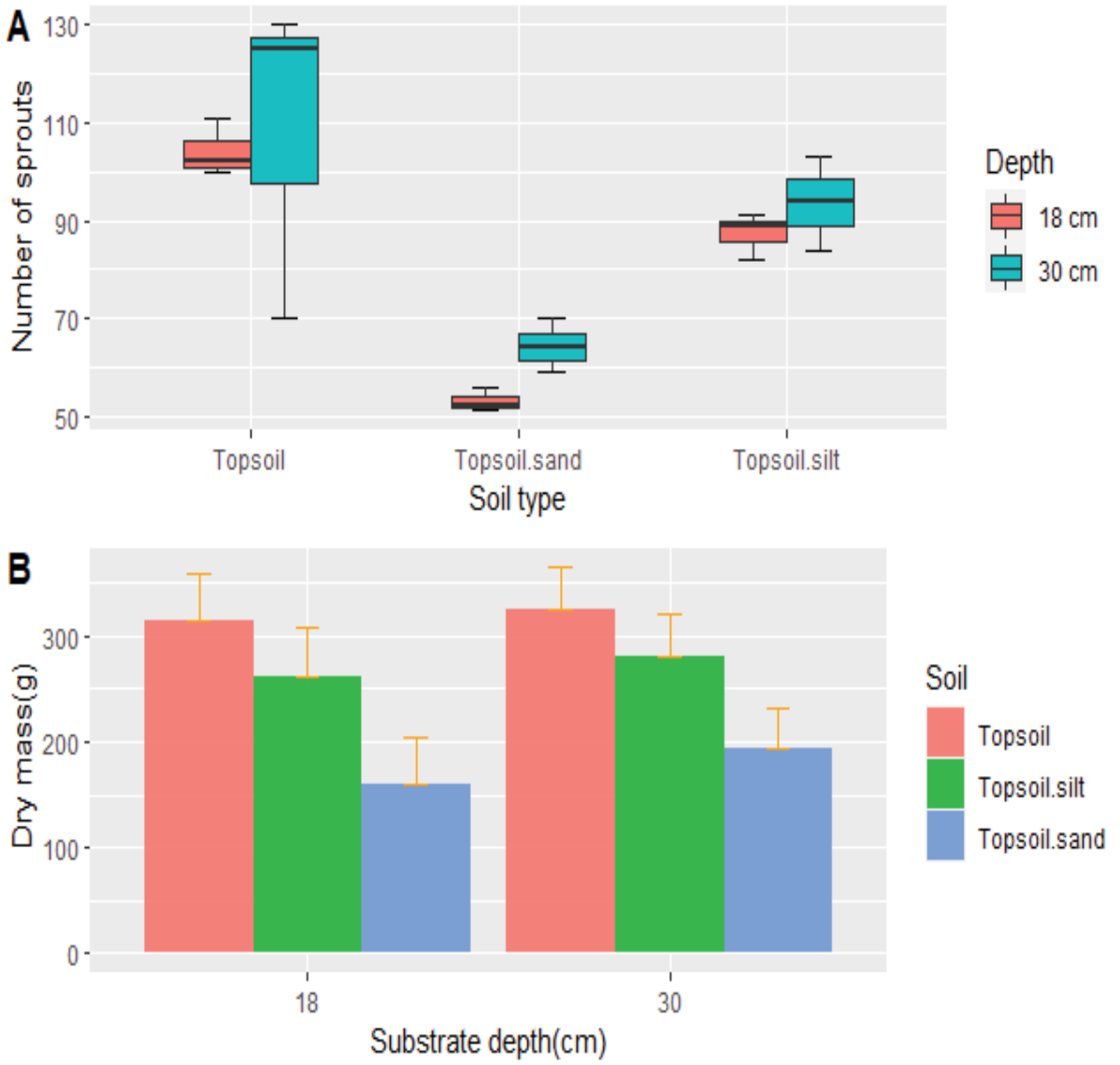

Figure 4: The effect of different soil types and substrate depths on **(A)** sprouts of sown meadow species, and **(B)** dry mass of sown meadow species; Topsoil = 100% topsoil(organic soil), Topsoil.silt = mixture of 95% topsoil and 5% silt, and Topsoil.sand = mixture of 50% topsoil and 50% sand

Biomass of sown seeds was higher in 30 versus 18 cm substrate depth for all soil types, though the pattern was statistically insignificant (β = 4.00, P = 0.74, Fig.4B). When averaged, topsoil and 30 cm depth had the highest amount of dry weight of sown species, while topsoil.sand and 18 cm depth had the lowest amount of dry weight.

The main effect of 30 cm depth (t = 0.33, P = 0.74) was not a significant predictor of the sprout of sown species (table 2). There were no significant interactions between 30 cm depth, topsoil.sand (t=0.43,P= 0.67) and topsoil.silt (t=0.14,P= 0.89) on the sprout of sown species (table 2). Looking at the interactions and the main effect of 30 cm depth, the sign on the coefficients are positive, indicating that the relationship between number of sprouts of sown species in all soil types and substrate depths was stronger.

Table 2: Coefficients and estimates of effect between categorical independent variables, soil type and substrate depth and number of sprouts of sown species.

	Value	Std.Error	DF	t-value	p-value
(Intercept)	104.333	8.474	12	12.311	0.000
Depth30cm	4.000	11.984	12	0.333	0.744
SoilTopsoil.sand	-51.333	11.984	12	-4.283	0.001
SoilTopsoil.silt	-17.000	11.984	12	-1.418	0.181
Depth30cm:SoilTopsoil.sand	7.333	16.948	12	0.432	0.672
Depth30cm:SoilTopsoil.silt	2.333	16.948	12	0.137	0.892

Dry Weight and Sprout Numbers of Colonizers

The sprout of colonist species was lower in 30 versus 18 cm substrate depth for all soil types (β = -149.66, P = 0.015, Fig.5A). Topsoil.silt had a significant lower number of colonizer sprouts than topsoil (β = -158.33, P = 0.01), and topsoil.sand had the lowest number of colonizer sprouts (β = -179.33, P = 0.005). In other words, on average, topsoil and 18 cm depth had the highest number of colonizer sprouts and topsoil.sand and 30 cm depth had the lowest number of colonizer sprouts. There were consistencies in the sprouts of colonist species on all soil types of 30 cm substrate depth, while topsoil and topsoil.silt had a wide variation of sprouts on substrate depth of 18 cm, with the exception of topsoil.sand with little variation (Fig.5A). Similarly, the dry weight of colonizers was lower in 30 versus 18 cm substrate depth for all soil types (β = -149.66, P = 0.02, Fig.5B). In other words, on average, there was a high biomass of colonizers in 18 cm depth of topsoil, while topsoil.sand of 30 cm depth had the lowest dry weight (β = -179.33, P = 0.005).

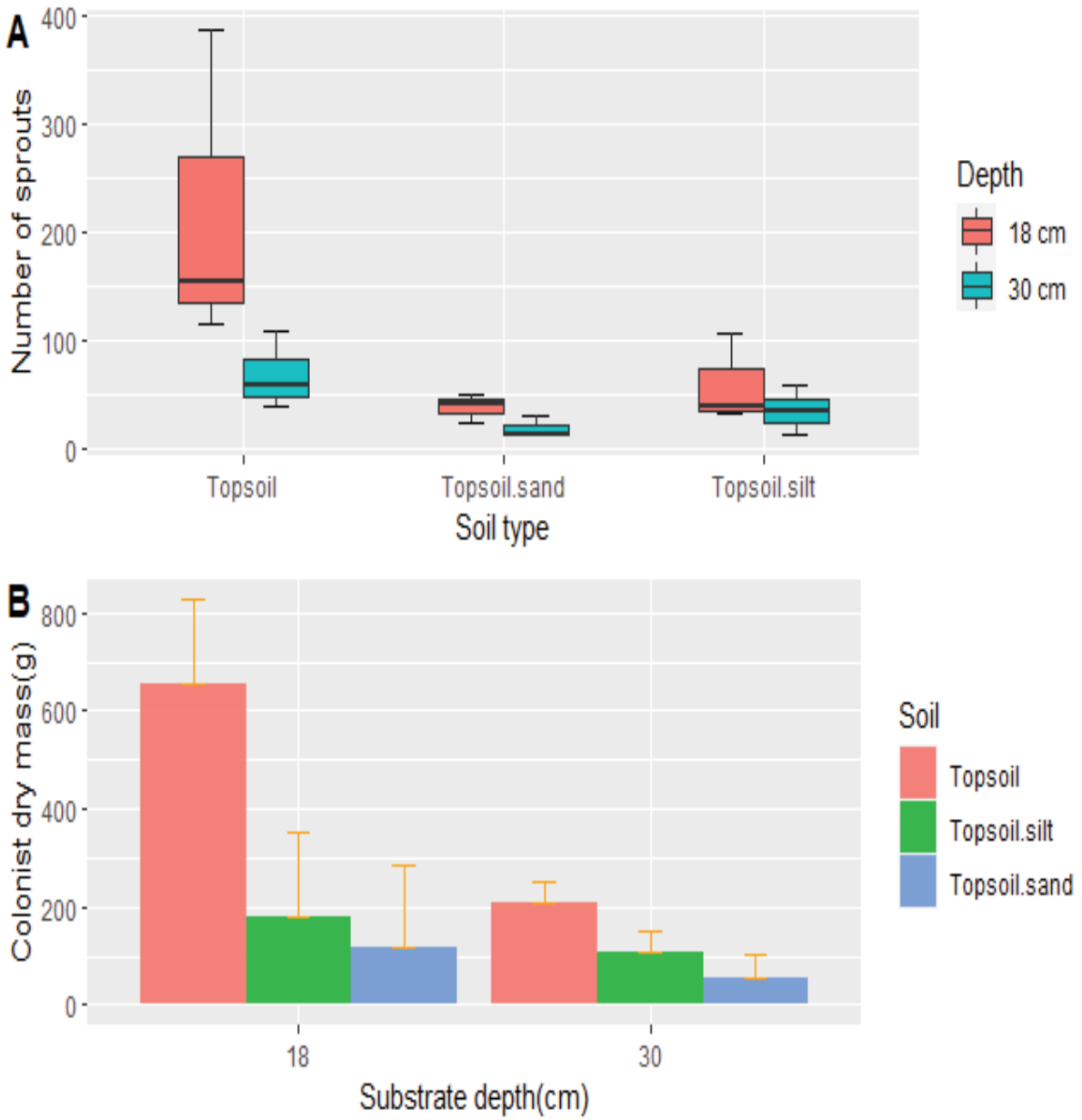

Figure 5: The effect of different soil types and substrate depths on **(A)** number colonizer sprouts, and **(B)** dry weight of colonizers; Topsoil = 100% topsoil(organic soil), Topsoil.silt = mixture of 95% topsoil and 5% silt, and Topsoil.sand = mixture of 50% topsoil and 50% sand.

The main effect of 30 cm depth (t = -2.83, P = 0.015) was a significant predictor, while 18 cm depth (β = -149.66) was a weak predictor for the sprout of colonizers (table 3). There were no significant interactions between 30 cm depth, topsoil.sand (t = 1.73, P = 0.11) and topsoil.silt (t = 1.67, P = 0.12) on sprout of colonizers. Looking at the interactions, the sign on the coefficients are positive, indicating that the relationship between the number of sprouts of colonizers in all soil types was stronger.

Table.3: Coefficients and estimates of effect between categorical independent variables, soil type and substrate depth on sprouts of colonist species.

	Value	Std.Error	DF	t-value	p-value
(Intercept)	218.666	37.437	12	5.840	0.0001
Depth30cm	-149.666	52.944	12	-2.826	0.015
soilTopsoil.sand	-179.333	52.944	12	-3.387	0.005
soilTopsoil.silt	-158.333	52.944	12	-2.990	0.011
Depth30cm:soilTopsoil.sand	129.333	74.875	12	1.727	0.109
Depth30cm:soilTopsoil.silt	125.000	74.875	12	1.669	0.120

The Number of Sprouts of Invasive Species

The sprout of invasive species was higher in 30 compared to 18 cm substrate depth for topsoil and topsoil.silt, though this pattern was not statistically significant (β = 0.66, P = 0.67). Topsoil.sand had a lower number of sprouts of invasive species in 30 versus 18 cm depth, though the pattern was statistically insignificant (β = -2.33, P = 0.15, Fig.6A). In addition, topsoil.silt had a non-significantly higher number of sprouts of invasive species compared to topsoil (β = 0.67, P = 0.67).Topsoil.sand had a non-significantly lower number of sprouts compared to topsoil (β = -2.33, P = 0.15).

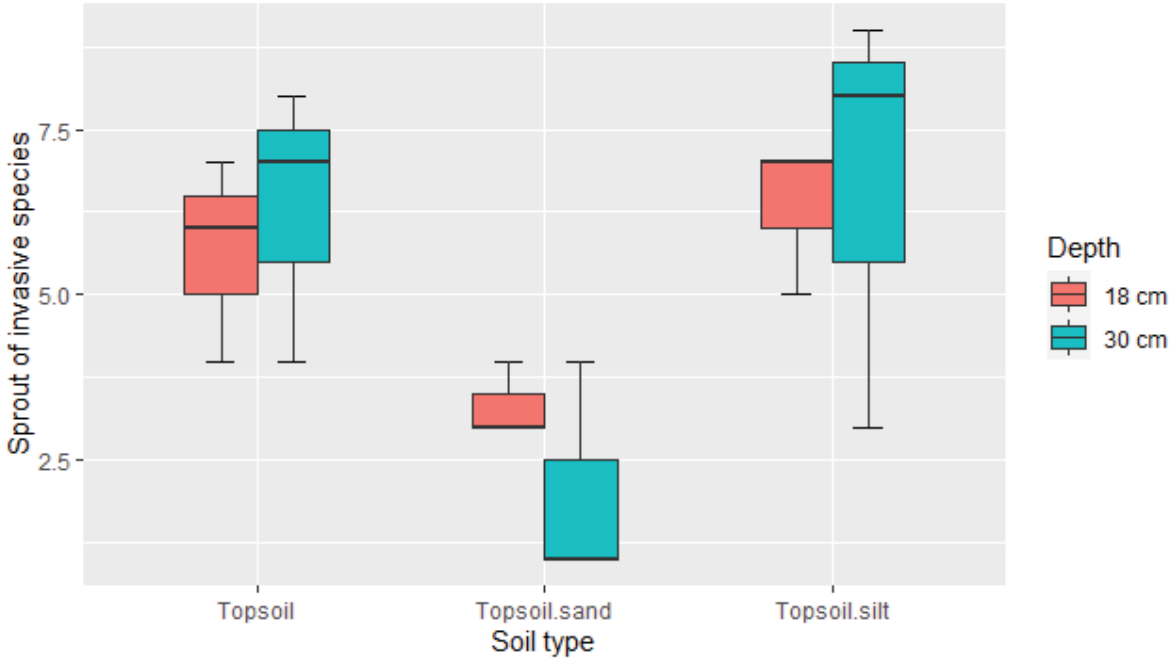

Figure 6: The effect of different soil types and two substrate depths on the sprout number of invasive species; Topsoil = 100% topsoil(organic soil), Topsoil.silt = mixture of 95% topsoil and 5% silt, and Topsoil.sand = mixture of 50% topsoil and 50% sand.

On average, topsoil.silt and 30 cm depth had the highest number of sprouts of invasive species and topsoil.sand and 30 cm depth had the lowest number of sprouts of invasive species. There was little variation in the sprouts of invasive species in topsoil.sand and topsoils.silt for 18 cm depth, but 30 cm depth had more variation, while topsoil had a slight variation in the sprouts of invasive species in all substrate depths (Fig.6).

The main effect of 30 cm depth ($t = 0.43$, $P = 0.67$) was not a significant predictor, while a strong relationship with 18 cm depth ($\beta = 0.66$) was found on the sprout of invasive species (table 4). There were no significant interactions between 30 cm depth, topsoil.sand ($t= -0.91$,$P= 0.38$) and topsoil.silt ($t=0.15$,$P= 0.88$) on the sprout of invasive species. Looking at the interactions, the sign on the coefficients are negative, indicating that the relationship between the number of sprouts of invasive species in all soil types was weak.

Table.4: Coefficients and estimates of effect between categorical independent variables, soil type and substrate depth on the number of sprouts of Invasive species.

	Value	Std.Error	DF	t-value	p-value
(Intercept)	5.666	1.097	12	5.164	0.0002
Depth30cm	0.666	1.551	12	0.429	0.675
soilTopsoil.sand	-2.333	1.551	12	-1.503	0.158
soilTopsoil.silt	0.666	1.551	12	0.429	0.675
Depth30cm: soilTopsoil.sand	-2.000	2.194	12	-0.911	0.380
Depth30cm:soilTopsoil.silt	-0.333	2.194	12	-0.151	0.881

Soil Types

Topsoil.sand was made up of 47.6% of sand, 31.3% of air, 1.7% of clay, 1.8% of silt and only 2.9% of organic matter (Fig.7). Topsoil (organic soil from Oslo compost) had 29.9% of sand, 30.7% of air, 1.2% of clay, 3.6% of silt and 9.7% of organic matter. Topsoil.silt had 32.6% of air, 30% of sand, 3.5% of silt, 0.7% of clay and 8.6% of organic matter. The percentage of water that was available for the plants (pF3 – pF4) in topsoil (14.2) and topsoil.silt (12.5) was slightly different. Meanwhile, the percentage of water available for the plants in topsoil.sand was approximately half the amount (6.5) in topsoil and topsoil.silt. The amount of sand in topsoil.sand was 20% higher than that of topsoil and only 2.9% of organic matter. There was approximately 30% sand in topsoil before the start of the experiment.

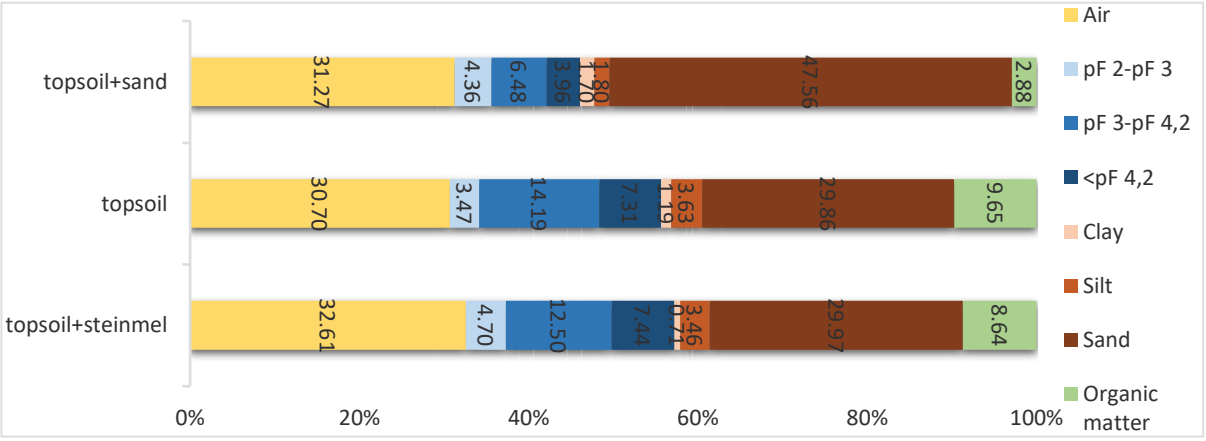

Figure 7: Horizontal bar graph showing the percentage categories of clay, silt and sand, % of organic matter, % air volume, and % categories of soil moisture present in the various soil type prior to seeding; topsoil = 100% topsoil (organic soil), topsoil+steinmel = mixture of 95% topsoil and 5% silt, topsoil + sand = mixture of 50% topsoil and 50% sand, pF2 – pF3 (field capacity), pF3 – pF4.2 (plant available water), and pF<4.2 (permanent wilting point).

DISCUSSION

Soil quality, as discussed in more detail below, was the main driver for the total number of sprouts and biomass combined for all species registered in this field experiment. Surprisingly, the two soil depths supported overall similar numbers of sprouts and biomass, while depth became more influential with lower soil quality. We found no significant interaction between soil quality and substrate depth, indicating soil quality and substrate depth were acting independently. This also applies to the number of sprouts found for invasive species. Thus, according to our study, the three soil types and two substrate depths allow for similar colonization of invasive species. However, and with that said, where there were higher numbers of native, sown species, there were lower numbers of invasive species, as discussed below.

Substrate depth

The two substrate depths showed no significant difference in total number of sprouts and biomass of sown species and number of sprouts of invasive species. For the colonist species, there was a slightly lower number of sprouts and biomass in 30 cm compared to 18 cm. The surprising similarities between the two substrate depths might be reflective of only sampling the first growing season after establishment and sowing. Differences between the two depths are likely to appear over time and with additional growing seasons. Although this was out of the scope of this thesis, we suggest continued sampling in the years to come.

For colonists, our results were in line with previous studies showing a decrease in biomass of colonizers with increasing growing medium depth (Aloisio et al., 2019). However, Aloisio et al. (2019) found an increase in biomass of colonizers with increasing growth medium for only one of their treatments and emphasized that their pattern was statistically insignificant. Contrary to our result, Dunnett et al. (2008), found an increase in the number of sprouts of colonizers with increasing substrate depth.

We hypothesized that shallower depths might have accumulated high mineral content and retain less water, thus increasing the drought conditions. Simultaneously, these conditions are likely less pronounced with increasing substrate depths (Oberndorfer et al., 2007). There was a significant effect of 30 cm depth on the sprouts of colonizers, supported by Aloisio et al. (2019). Colonization of our 30 cm pallets could favour meadow species based on the characteristic of their roots, with the growing medium depth showing an effect with species having deeper roots (perennials). The 30 cm substrate depth will provide space for root expansion, while the 18 cm depth might allow for more fibrous root systems (annuals) (Monterusso et al., 2005, Wolf and Lundholm, 2008).

Soil types

The total number of sprouts and biomass of all sown species combined was highest in topsoil and lowest in topsoil.sand, with topsoil.silt slightly lower than topsoil, but not significantly different. Furthermore, the number of sprouts and biomass of colonist species combined was again highest for topsoil and lowest in topsoil.sand. Like substrate depth, no difference was found between soil types for the number of sprouts for invasive species.

Little variation between the total number of sprouts and biomass for topsoil and topsoil.silt for sown species was supported by similarities amongst the tested indices of "soil quality". For instance, the proportion of soil particles (clay, silt, and sand) were consistent with our expectations and hypothesis. Moreover, the proportion of soil particles in topsoil and topsoil.silt were different from the particles in topsoil.sand. Congruous to our second hypothesis and keeping in mind that the soils were constantly saturated (pF 2 – pF3 = field capacity; amount of water the soil can hold, was approximately the same for all soil types), topsoil.sand became dryer quicker than the other two soil types. This is because of a high amount of larger particle sizes (sand) allowing water to run between them. These larger particle sizes also influenced how topsoil.sand made water available for the plants (pf 3 – pf 4.2). Thus,

a combination of less water in the soil (pf <4.2) together with less available water for the plants (maintain soil moisture, Fig.7.) result in overall less sprouts and biomass in topsoil.sand.

The addition of silt to reduce the pore size of the soil particles to improve water retention was unsuccessful. This resulted in approximately the same amount of pF-values and soil particle sizes compared to topsoil. This again likely influenced the lack of differences in the total number of sprouts and biomass for topsoil and topsoil.silt. Our results are supported by Lu et al. (2015), who found that the interaction between general availability of water, substrate depth and water storage layer is necessary for the development of meadows on green roofs. The similar total number of sprouts and biomass of sown species and colonizers are also in accordance with Bates et al. (2015) idea that, though the essential factor defining meadow growth on green roofs is drought, the quality of the substrate (proportion of particles) also has a significant part to play. In our experiment, adding silt did not bring any advantages compared to "regular" topsoil, and adding sand was actually a disadvantage, as discussed more below.

Soil organic matter

The distribution of soil organic matter was lowest in topsoil.sand and highest for topsoil, with topsoil.silt slightly lower than topsoil. Topsoil already had approximately 30% sand, not 100% compost. By mixing 50% sand with 50% topsoil that already had 30% sand, made topsoil.sand 20% more sandy and with about 3% organic matter. Topsoil and topsoil.silt had about 10% and 9% organic matter, respectively, and 30% sand. The higher amount of organic matter certainly supported the higher total number of sprouts and biomass of sown species and colonists in topsoil and topsoil.silt, as well as the slightly lower results for topsoil.silt compared to topsoil.

Like soil treatments, no difference was found between sprouts of invasive species between the three soil types. A limitation in our study was that we did not analyse the soil prior to sowing. These results could be explained by the findings of Madsen (2005) that organic matter has a varying number of detritivores and microorganisms that recycle nutrients from the organic matter differently. The activities of these microorganisms will further produce different amounts of readily available forms of nutrient to the meadows (Trasar-Cepeda et al., 2007), hence lower sprouts and biomass in topsoil.sand and higher in topsoil. These readily available forms of nutrients were shown by Comas et al. (2013) to be important, since they will also enhance root development by providing beneficial effects to water holding capacity and water uptake by plants. The higher amount of compost retained more water by increasing stability of

the soil aggregates (Celik et al., 2004, Graceson et al., 2013). This supports our third hypothesis, with the total number of sprouts and biomass being lower in topsoil.sand and higher in topsoil due to different amounts of organic matter in the different soil treatments.

Altogether, the no effect of substrate depth, soil treatments, and amount of organic matter on the sprouts of invasive species could be that the native species are adapted to drought conditions in our site (Burghardt et al., 2010). This is in accordance with Bates et al. (2015), who found that an optimized established growth medium will reduce drought effects and improve meadow species survival on green roofs. Hence, sprout in great amounts will reduce the chances of invasive species to sprout. This is true for previous studies which shows invasiveness at the local scale to be oppositely symmetrical to resident species amount or abundance (Levine et al., 2004). Furthermore, although not tested and since our study was not on green roof per se, the use of indistinguishable soil treatment throughout our pallet collars resulted in the presence of surrounding meadows and moth larva in our pallets proving that their colonization was influenced by surrounding plant species. This was contrary to the study of Madre et al. (2014) and consistent with Aloisio et al. (2019), indicating that the surrounding green space does not only influences the colonization of meadows on green roofs, but also contributes to configure the arthropod assemblages on these roofs.

Though the pallets were randomised with the assumption that the effects from the microclimate condition were equally distributed, shading from surrounding trees, seeds from seed banks, phenological differences between species, and seeds dispersal by birds and wind could be potential sources influencing colonization in our field experiment (Aloisio et al., 2019, Braaker et al., 2017, Kowarik, 2011). Also, the seed bank could be a means through which an invasive species is introduced to the green roof prior to sowing, e.g. we had mistakenly collected *Barbarea vulgaris,* an alien meadow species in Norway discovered after sowing. We recommend a proper check of the seed bank, characteristics of habitat specific species, and inspection of roofs after seeding in future studies. This is to minimise the spread of alien species not only to the created or restored patches, but to the entire community. The removal of invasive species during the early stages of meadows propagation on green roofs will further reduce invasiveness on green roofs (Kinlocka et al., 2016). For instance, we removed some invasive species on our site when they were 10 cm leaf coverage and identifiable at the species level. Thus, the measurements and tests presented here represent a minimum of invasive species.

Non-Sprouting of seeds

After one growing season of 20 weeks and sowing 25 wild Norwegian meadow species of 15 seeds each, 5 of the species did not sprout. This is associated with the fact that *Angelica sylvestris L.* take long to sprout (i.e. above 3 months) and their seeds may sometimes wait for spring before emerging (Roberts, 1979). Also, the seeds of *Scrophularia nodosa* can persist in the soil for many years before sprouting (Thompson and Grime, 1979), requires light for their sprouting (Vranckx and Vandelook, 2012), and increase in temperature induces secondary dormancy (Karssen, 1980). 90% of the seeds of *Lathyrus Sylvestris L* require acid scarification before they can sprout (Wright, 1985). *Trifolium medium L* has the reputation of poor seed-setting (Robertson and Armstrong, 1964). Finally, the seeds of *Allium oleraceum L* are sterile, and they entirely reproduce asexually (Konvička, 1972, LEVAN, 1933). The characteristics of seed germination should be taken into consideration before sowing seeds to be certain about their sprouts and method of propagation (Nagase and Dunnett, 2013). We propose pretesting of seeds in a nursery before sowing to determine their germination success. We also suggest continued registrations of our pallets in the years to come for gathering new data.

CONCLUSION

The aim of this thesis was to investigate the effect of substrate depth and soil treatments on the sprouts and biomass of native meadows, colonizers, and sprouts of invasive species on green roofs. This investigation provided results upon which green roofs can be established in Oslo as stepping-stone, thus, enhancing the connectivity between fragmented patches of native meadows (see appendix 3). The novelty of this study is that we tested the number of sprouts and biomass of an exact number of sown Norwegian native meadow species in a field experiment simulating a green roof environment. Altogether, the results from this study corroborate the positive relationship between the quality of the substrate layer and the total presence of water. In addition, this study indicates that high organic content and soil treatment are a key to construct successful green roofs, as it was best suited for the growth of sown and colonizing native species. This study has provided valuable input on important soil related factors to consider when constructing or restoring green roofs. From a management perspective, and when working on a low budget, it is better to reduce soil depth than to mix in more sand. To improve the biodiversity in Oslo with the use of green roofs, not only soil related factors are important, but other factors influencing colonization and invasiveness on these roofs should be integrated in the planning and construction of new green roofs. For example, the abundance and composition of native meadows species, surrounding vegetation, seeds from

seed banks, pollination, and wind and animal dispersal of seeds (Aronson et al., 2014, Hanski, 1998, Kowarik, 2011, Rudnick et al., 2012). While green roofs can act as stepping-stones and foraging grounds for some species, their combined patch sizes, quality, connectedness, and the amount of green area within Oslo are essential for biodiversity benefits. This should be considered when establishing green roofs to mitigate anthropogenic encroachment of urban areas, as a means of improving urban biodiversity. Continuous research could be carried out at the site and the species composition for each year is compared with subsequent years.

REFERENCES (including those in appendix 3)

AHERN, J. 2013. Urban landscape sustainability and resilience: the promise and challenges of integrating ecology with urban planning and design. *Landscape Ecology,* 28**,** 1203-1212.

AIKEN, L. S., WEST, S. G. & RENO, R. R. 1991. *Multiple regression: Testing and interpreting interactions*, Sage.

ALOISIO, J. M., PALMER, M. I., GIAMPIERI, M. A., TUININGA, A. R. & LEWIS, J. D. 2017. Spatially dependent biotic and abiotic factors drive survivorship and physical structure of green roof vegetation. *Ecological Applications,* 27**,** 297-308.

ALOISIO, J. M., PALMER, M. I., TUININGA, A. R. & LEWIS, J. D. 2019. Plant Colonization of Green Roofs Is Affected by Composition of Established Native Plant Communities. *Frontiers in Ecology and Evolution,* 6.

ARNOTT, D. 2015. *General recommendations for the implementation of a stepping-stone corridor between Nøkkelvann and sognsvann in the city of Oslo, Norway.* landscape ecology term paper, Norwegian university of life science.

ARONSON, M. F., LA SORTE, F. A., NILON, C. H., KATTI, M., GODDARD, M. A., LEPCZYK, C. A., WARREN, P. S., WILLIAMS, N. S., CILLIERS, S. & CLARKSON, B. 2014. A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences,* 281**,** 20133330.

BALDOCK, K., GODDARD, M., HICKS, D., KUNIN, W., MITSCHUNAS, N., OSGATHORPE, L., POTTS, S., ROBERTSON, K., SCOTT, A. & STONE, G. 2015. Where is the UK's pollinator biodiversity? Comparing flower-visitor communities between cities, farmland and nature reserves using visitation networks. *Proc. R. Soc. B,* 282**,** 10.1098.

BÅRD BREDSEN 2020. data containing the coordinate of existing native meadow patches in Oslo.

BATES, A. J., SADLER, J. P., GRESWELL, R. B. & MACKAY, R. 2015. Effects of recycled aggregate growth substrate on green roof vegetation development: A six year experiment. *Landscape and Urban Planning,* 135**,** 22-31.

BENGTSSON, L., GRAHN, L. & OLSSON, J. 2005. Hydrological function of a thin extensive green roof in southern Sweden. *Hydrology Research,* 36**,** 259-268.

BENINDE, J., VEITH, M. & HOCHKIRCH, A. 2015. Biodiversity in cities needs space: a meta-analysis of factors determining intra-urban biodiversity variation. *Ecology letters,* 18**,** 581-592.

BENVENUTI, S. 2014. Wildflower green roofs for urban landscaping, ecological sustainability and biodiversity. *Landscape and Urban Planning,* 124**,** 151-161.

BRAAKER, S., GHAZOUL, J., OBRIST, M. & MORETTI, M. 2014. Habitat connectivity shapes urban arthropod communities: the key role of green roofs. *Ecology,* 95**,** 1010-1021.

BRAAKER, S., OBRIST, M. K., GHAZOUL, J. & MORETTI, M. 2017. Habitat connectivity and local conditions shape taxonomic and functional diversity of arthropods on green roofs. *Journal of Animal Ecology,* 86**,** 521-531.

BURGHARDT, K. T., TALLAMY, D. W., PHILIPS, C. & SHROPSHIRE, K. J. 2010. Non-native plants reduce abundance, richness, and host specialization in lepidopteran communities. *Ecosphere,* 1**,** 1-22.

BUTLER, C., BUTLER, E. & ORIANS, C. M. 2012. Native plant enthusiasm reaches new heights: Perceptions, evidence, and the future of green roofs. *Urban Forestry & Urban Greening,* 11**,** 1-10.

CELIK, I., ORTAS, I. & KILIC, S. 2004. Effects of compost, mycorrhiza, manure and fertilizer on some physical properties of a Chromoxerert soil. *Soil and Tillage Research,* 78**,** 59-67.

CHASE, J. M. 2003. Community assembly: when should history matter? *Oecologia,* 136**,** 489-498.

COLLINGE, S. K. 2009. *Ecology of fragmented landscapes*, JHU Press.

COMAS, L., BECKER, S., CRUZ, V. M. V., BYRNE, P. F. & DIERIG, D. A. 2013. Root traits contributing to plant productivity under drought. *Frontiers in plant science,* 4**,** 442.

COOK-PATTON, S. C. & BAUERLE, T. L. 2012. Potential benefits of plant diversity on vegetated roofs: A literature review. *Journal of Environmental Management,* 106**,** 85-92.

CRAWLEY, M. J. 2012. *The R book*, John Wiley & Sons.

DONNELLY, R. & MARZLUFF, J. M. 2004. Importance of reserve size and landscape context to urban bird conservation. *Conservation Biology,* 18**,** 733-745.

DUNNETT, N. & KINGSBURY, N. 2008. *Planting green roofs and living walls*, Timber press Portland, OR.

DUNNETT, N., NAGASE, A. & HALLAM, A. 2008. The dynamics of planted and colonising species on a green roof over six growing seasons 2001–2006: influence of substrate depth. *Urban Ecosystems,* 11**,** 373-384.

DVORAK, B. & VOLDER, A. 2010. Green roof vegetation for North American ecoregions: A literature review. *Landscape and Urban Planning,* 96**,** 197-213.

EVJU, M., BLUMENTRATH, S., SKARPAAS, O., STABBETORP, O. E. & SVERDRUP-THYGESON, A. 2015a. Plant species occurrence in a fragmented grassland landscape: the importance of species traits. *Biodiversity and conservation,* 24**,** 547-561.

EVJU, M., HANSSEN, O., STABBETORP, O. E., BRATLI, H. & ØDEGAARD, F. 2015b. Strandeng–et hotspot-habitat. Sluttrapport under ARKO-prosjektets periode III.

FARGIONE, J., BROWN, C. S. & TILMAN, D. 2004. Community assembly and invasion: An experimental test of neutral versus niche processes (vol 100, pg 8916, 2003). *Proceedings of the National Academy of Sciences of the United States of America,* 101**,** 414-414.

FATTORINI, S., MANTONI, C., DE SIMONI, L. & GALASSI, D. M. P. 2018. Island biogeography of insect conservation in urban green spaces. *Environmental Conservation,* 45**,** 1-10.

FLETCHER JR, R. J., ORROCK, J. L. & ROBERTSON, B. A. 2012. How the type of anthropogenic change alters the consequences of ecological traps. *Proceedings of the Royal Society B: Biological Sciences,* 279**,** 2546-2552.

FOLEY, J. A., DEFRIES, R., ASNER, G. P., BARFORD, C., BONAN, G., CARPENTER, S. R., CHAPIN, F. S., COE, M. T., DAILY, G. C. & GIBBS, H. K. 2005. Global consequences of land use. *science,* 309**,** 570-574.

FREMMEDARTSLISTA. 2018. *What is an alien species?* [Online]. Artdatabanken. Available: https://www.artsdatabanken.no/fremmedearter [Accessed 18 of August 2020].

FUKAMI, T. 2015. Historical Contingency in Community Assembly: Integrating Niches, Species Pools, and Priority Effects. *In:* FUTUYMA, D. J. (ed.) *Annual Review of Ecology, Evolution, and Systematics, Vol 46.*

GRACESON, A., HARE, M., MONAGHAN, J. & HALL, N. 2013. The water retention capabilities of growing media for green roofs. *Ecological Engineering,* 61**,** 328-334.

HANSKI, I. 1998. Metapopulation dynamics. *Nature,* 396**,** 41-49.

HEIM, A. & LUNDHOLM, J. 2016. Phenological complementarity in plant growth and reproduction in a green roof ecosystem. *Ecological Engineering,* 94**,** 82-87.

HILLEBRAND, H., BENNETT, D. M. & CADOTTE, M. W. 2008. Consequences of dominance: a review of evenness effects on local and regional ecosystem processes. *Ecology,* 89**,** 1510-1520.

KARSSEN, C. 1980. Patterns of change in dormancy during burial of seeds in soil. *Israel Journal of Plant Sciences,* 29**,** 65-73.
KENDAL, D., WILLIAMS, N. S. G. & WILLIAMS, K. J. H. 2012. A cultivated environment: Exploring the global distribution of plants in gardens, parks and streetscapes. *Urban Ecosystems,* 15**,** 637-652.
KINLOCKA, N. L., SCHINDLER, B. Y. & GUREVITCH, J. 2016. Biological invasions in the context of green roofs. *Israel Journal of Ecology and Evolution,* 62**,** 32-43.
KÖHLER, M. 2006. Long-term vegetation research on two extensive green roofs in Berlin. *Urban Habitats,* 4**,** 3-26.
KONVIČKA, O. 1972. Cytotaxonomische Studien von vier sterilen Arten der GattungAllium. *Biologia Plantarum,* 14**,** 62-70.
KOWARIK, I. 2011. Novel urban ecosystems, biodiversity, and conservation. *Environmental pollution,* 159**,** 1974-1983.
KROGSTAD, T., BØRESEN, T. & ALMÅS, Å. 2018. Field and laboratory methods for the analysis of soil Compendium. *Faculty of environmental science and natural resource management: Norwegian University of Life sciences*.
KUUSSAARI, M., SAARINEN, M., KORPELA, E. L., PÖYRY, J. & HYVÖNEN, T. 2014. Higher mobility of butterflies than moths connected to habitat suitability and body size in a release experiment. *Ecology and Evolution,* 4**,** 3800-3811.
LANDSCHAFTSENTWICKLUNG-LANDSCHAFTSBAU, F. 2008. *Richtlinie für die Planung, Ausführung und Pflege von Dachbegrünungen: Dachbegrünungsrichtlinie; Ausgabe 2008*, FLL.
LEVAN, A. 1933. CYTOLOGICAL STUDIES IN ALLIUM, III ALLIUM CARINATUM AND ALLIUM OLERACEUM. *Hereditas,* 18**,** 101-114.
LEVINE, J. M., ADLER, P. B. & YELENIK, S. G. 2004. A meta-analysis of biotic resistance to exotic plant invasions. *Ecology letters,* 7**,** 975-989.
LU, J., YUAN, J. G., YANG, J. Z., CHEN, A. K. & YANG, Z. Y. 2015. Effect of substrate depth on initial growth and drought tolerance of Sedum lineare in extensive green roof system. *Ecological Engineering,* 74**,** 408-414.
LUNDHOLM, J., MACIVOR, J. S., MACDOUGALL, Z. & RANALLI, M. 2010. Plant species and functional group combinations affect green roof ecosystem functions. *PloS one,* 5.
MACARTHUR, R. & LEVINS, R. 1967. The limiting similarity, convergence, and divergence of coexisting species. *The American Naturalist,* 101**,** 377-385.
MACARTHUR, R. H. & WILSON, E. O. 1967. *Island biogeography*, Princeton.
MACIVOR, J. S., MARGOLIS, L., PUNCHER, C. L. & MATTHEWS, B. J. C. 2013. Decoupling factors affecting plant diversity and cover on extensive green roofs. *Journal of environmental management,* 130**,** 297-305.
MADRE, F., VERGNES, A., MACHON, N. & 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**,** 100-107.
MADSEN, E. L. 2005. Identifying microorganisms responsible for ecologically significant biogeochemical processes. *Nature Reviews Microbiology,* 3**,** 439-446.
MAYRAND, F. & CLERGEAU, P. 2018. Green roofs and green walls for biodiversity conservation: a contribution to urban connectivity? *Sustainability,* 10**,** 985.
MILJØDIREKTORATET. 2020. *The Oslo region, showing important nature types in and around the city of Oslo, Norway* [Online]. Available: https://Kart.naturbase.no [Accessed 18 of August 2020].
MODERNIZATION, M. O. G. A. 2014. *Framtidens Byer* [Online]. Available: http://www.regjeringen.no/nb/sub/framtidensbyer/forside.html?id=551422 [Accessed 18 of august 2020].
MOLINEUX, C. J., GANGE, A. C. & NEWPORT, D. J. 2017. Using soil microbial inoculations to enhance substrate performance on extensive green roofs. *Science of the Total Environment,* 580**,** 846-856.

MONTERUSSO, M. A., ROWE, D. B. & RUGH, C. L. 2005. Establishment and persistence of Sedum spp. and native taxa for green roof applications. *HortScience,* 40**,** 391-396.
NAGASE, A. & DUNNETT, N. 2010. Drought tolerance in different vegetation types for extensive green roofs: effects of watering and diversity. *Landscape and urban planning,* 97**,** 318-327.
NAGASE, A. & DUNNETT, N. 2013. Establishment of an annual meadow on extensive green roofs in the UK. *Landscape and urban planning,* 112**,** 50-62.
NAGASE, A., DUNNETT, N. & CHOI, M.-S. 2013. Investigation of weed phenology in an establishing semi-extensive green roof. *Ecological engineering,* 58**,** 156-164.
NORDH, H. & ØSTBY, K. 2013. Pocket parks for people–A study of park design and use. *Urban forestry & urban greening,* 12**,** 12-17.
OBERNDORFER, E., LUNDHOLM, J., BASS, B., COFFMAN, R. R., DOSHI, H., DUNNETT, N., GAFFIN, S., KÖHLER, M., LIU, K. K. & ROWE, B. 2007. Green roofs as urban ecosystems: ecological structures, functions, and services. *BioScience,* 57**,** 823-833.
OLLY, L. M., BATES, A. J., SADLER, J. P. & MACKAY, R. 2011. An initial experimental assessment of the influence of substrate depth on floral assemblage for extensive green roofs. *Urban Forestry & Urban Greening,* 10**,** 311-316.
OSLO KOMMUNE. 2011. *Byøkologisk Program 2011 - 2026: Oslo kommune.* [Online]. Available: https://www.oslo.kommune.no [Accessed 12 of April 2020].
OSLO KOMMUNE. 2015. *Byøkologisk Program: Improved management of biodiversity in oslo* [Online]. oslo: Stian Berger Røsland ,Guri Melby. Available: https://www.oslo.kommune.no/politikk-ogadministrasjon/statistikk/befolkning/befolkningsfremskrivinger/ [Accessed 15 of April 2020].
PATAKI, D. E. 2015. Grand challenges in urban ecology. *Frontiers in Ecology and Evolution,* 3.
PINHEIRO, J. & BATES, D. 2000. Mixed-Effects Models in S and S-PLUS Springer-Verlag. New.
ROBERTS, H. 1979. Periodicity of seedling emergence and seed survival in some Umbelliferae. *Journal of Applied Ecology***,** 195-201.
ROBERTSON, R. & ARMSTRONG, J. 1964. Factors affecting seed production in Trifolium medium. *Canadian Journal of Plant Science,* 44**,** 337-343.
RUDNICK, D., RYAN, S. J., BEIER, P., CUSHMAN, S. A., DIEFFENBACH, F., EPPS, C., GERBER, L. R., HARTTER, J. N., JENNESS, J. S. & KINTSCH, J. 2012. The role of landscape connectivity in planning and implementing conservation and restoration priorities. Issues in Ecology.
SCHRODER, R., MOHRI, M. & KIEHL, K. 2019. AMF inoculation of green roof substrate improves plant performance but reduces drought resistance of native dry grassland species. *Ecological Engineering,* 139.
SHANAHAN, D. F., MILLER, C., POSSINGHAM, H. P. & FULLER, R. A. 2011. The influence of patch area and connectivity on avian communities in urban revegetation. *Biological Conservation,* 144**,** 722-729.
SMITH, M. D., WILCOX, J. C., KELLY, T. & KNAPP, A. K. 2004. Dominance not richness determines invasibility of tallgrass prairie. *Oikos,* 106**,** 253-262.
SPECIES DATA BANK. 2018. *Fremmedartslista 2018* [Online]. Available: https://www.artsdatabanken.no/fremmedartslista2018 [Accessed 20 of August 2020].
THOMPSON, K. & GRIME, J. P. 1979. Seasonal variation in the seed banks of herbaceous species in ten contrasting habitats. *The Journal of Ecology***,** 893-921.
TRASAR-CEPEDA, C., GIL-SOTRES, F. & LEIRÓS, M. 2007. Thermodynamic parameters of enzymes in grassland soils from Galicia, NW Spain. *Soil Biology and Biochemistry,* 39**,** 311-319.
VAN ROSSUM, F. 2010. Reproductive success and pollen dispersal in urban populations of an insect-pollinated hay-meadow herb. *Perspectives in Plant Ecology, Evolution and Systematics,* 12**,** 21-29.
VRANCKX, G. & VANDELOOK, F. 2012. A season- and gap-detection mechanism regulates seed germination of two temperate forest pioneers. *Plant Biology,* 14**,** 481-490.

WILLIAMS, N. S. G., SCHWARTZ, M. W., VESK, P. A., MCCARTHY, M. A., HAHS, A. K., CLEMANTS, S. E., CORLETT, R. T., DUNCAN, R. P., NORTON, B. A., THOMPSON, K. & MCDONNELL, M. J. 2009. A conceptual framework for predicting the effects of urban environments on floras. *Journal of Ecology,* 97**,** 4-9.

WOLF, D. & LUNDHOLM, J. T. 2008. Water uptake in green roof microcosms: Effects of plant species and water availability. *Ecological Engineering,* 33**,** 179-186.

WRIGHT, S. 1985. Germintion, nodulation and early growth response of a hard-seeded legume, Lathyrus sylvestris L.(flatpea). *Plant and soil,* 88**,** 455-459.

APPENDIX

Appendix 1

Table showing the weights measured at various soil pF analysis of the different soil mixture.

Soil type	Topsoil				Topsoil + sand				Topsoil + silt			
Sample number.	1	2	3	4	5	6	7	8	9	10	11	12
Weight of cylinder [g]. **A**	145.55	145.04	145.35	145.54	133.74	133.59	133.55	133.21	140.71	138.19	138.07	138.27
saturated Weight [g] (soil,cylinder, lid,RB,B)	296.25	294.32	294.30	292.63	302.71	300.47	300.10	301.20	294.92	295.03	290.34	291.48
Weight at 10 cm/10 hpa, (pF 1.0) [g] (soil,cylinder, lid,RB,B)	286.25	288.24	289.66	285.97	290.45	286.20	288.12	289.76	275.99	276.94	276.30	276.30
Weight at 30 cm/30 hPa, [g] (soil,cylinder, lid,RB,B)	266.99	267.79	268.03	265.98	273.69	271.61	272.56	273.27	263.64	263.49	262.57	263.18
Weight at 0,1 bar/100 hpa, (pF 2)[g] (soil,cylinder, lid,RB,B)	262.29	264.54	263.94	263.93	270.58	268.87	269.71	270.26	260.80	260.36	259.43	260.74
Weight at 1 bar/1000 hpa, (pF 3) [g] (soil,cylinder, lid,RB,B)	257.03	262.35	260.60	260.84	266.87	262.25	266.10	266.76	256.02	255.23	254.10	257.20
Dry weight of soil [g] (with cylinder and paper sheet). **B**	237.39	239.00	240.26	238.15	255.49	253.82	255.58	255.31	236.62	236.44	235.07	234.67
Dry weight of soil [g]. **C = B - A**	91.84	93.96	94.91	92.61	121.75	120.23	122.03	122.10	95.91	98.25	97.00	96.40

Appendix 2

Table showing the weights measured during analysis of wilting point of the different soil mixture (15 000 hpa, pF 4,2 or 15 bar).

Sample number	ring number (small)	Box number	tare Box [g]	weight at 15 bar [g] (and box)	weight at 15 bar [g](without box)	weight dry [g] (with box)	weight dry [g] (without box)
Topsoil							
1	166	64	17.85	32.96	15.11	31.94	14.09
2	138	67	17.86	33.37	15.51	32.25	14.39
3	140	78	17.72	35.06	17.34	33.63	15.91
4	152	97	17.83	34.88	17.05	33.72	15.89
Topsoil + sand							
5	107	103	17.86	36.98	19.12	36.44	18.58
6	124	87	17.85	36.66	18.81	35.99	18.14
7	143	83	17.96	37.09	19.13	36.51	18.55
8	137	57	17.94	38.17	20.23	37.52	19.58
Topsoil + silt							
9	134	109	17.98	32.42	14.44	31.56	13.58
10	142	76	17.81	33.59	15.78	32.33	14.52
11	117	59	17.71	33.68	15.97	32.51	14.8
12	128	107	17.85	31.45	13.6	30.47	12.62
Standard							
	62	71	17.92	29.75	11.83	28.45	10.53
	61	95	17.96	29.76	11.8	28.55	10.59

Appendix 3

Urban connectivity of native meadows in Oslo

The susceptibility and fragmentation of Oslo`s green space can be conceptualized as a set of habitat islands (green patches surrounded by human structures) for many of the organisms that inhabit them. According to island biogeography theory, the species richness on such patches are influenced by the distance between these patches from one another, immigration and colonization, extinction, and size and shape of the patches (MacArthur and Wilson, 1967). In this light, the biological communities of the city`s green patches are

equivalent to those of a group of islands (Fattorini et al., 2018). In Oslo, many of these fragmented green patches represent remnant habitat for native species. In accordance with island biogeography, these species depend on other existing and potentially new patches to avoid extinction (Donnelly and Marzluff, 2004). For instance, Oslo presently has the highest number of observed native species in Norway (Norwegian Biodiversity information centre gave a record of 11.554 species in June 2013). Furthermore, about 60% of the national Red listed species were recorded within the municipality of Oslo (Oslo Kommune, 2015).

Though there is an understanding about the patch size that can sustain certain species of organism, a combination of many smaller patch sizes (and their quality) will generate more green space in urban areas, that in turn, is necessary for the maintenance of biodiversity of plants, birds, insects and other organisms (Aronson et al., 2014, Beninde et al., 2015). Generally, to enhance biodiversity of fragmented urban landscapes, connectivity between urban green patches has been proven to be the way forward (Shanahan et al., 2011). The connectivity between these patches can be achieved through the establishment of corridors and/or steppingstones. Where corridors are linear strips of vegetation that link otherwise isolated fragments, steppingstones are series of small non-connected habitats in-between large habitat fragments (Collinge, 2009). Steppingstones enhance the movement, foraging, nesting, and reduce the distances between patches for plants, pollinators, birds, and other organisms. One way of creating steppingstones in urban areas is through the establishment of green roofs. This has been practised for many years to improve patch connectivity and mitigate the loss of urban green space (Mayrand and Clergeau, 2018).

Oslo has used the steppingstone concept through the establishment of native meadow patches both on roofs and on the ground (example by Oslo Bymijløetaten at Strømsveien 102) (Fig.8)(Modernization, 2014). These patches serve as a means of mitigating the loss of connectivity, interaction within and between individuals, populations, and communities of insects, birds, and pollinators in general, hence, increasing ecosystem interactions (Foley et al., 2005). Though the distances between the established native meadow patches seems vast for some pollinator species, for example, the Lepidoptera macro moths, that rarely move further than 63 m (Kuussaari et al., 2014). Consequently, the establishment of more green roofs will decrease the distance between Oslo urban green patches and increase the interaction between species and within ecosystems (Braaker et al., 2014). These shorter distances will promote the transfer of pollen and dispersal of seed by pollinators and birds and provide important ecosystems functions advantageous for urban development (Hanski, 1998). Increasing the

interactions between species and within ecosystems in the city of Oslo will also reinforce the transfer of nutrients and energy, increase gene flow, which will significantly increase habitat heterogeneity and plant species diversity (Kowarik, 2011). This will create a diverse community of organisms such as wild bees and Lepidoptera macro moth (Baldock et al., 2015).

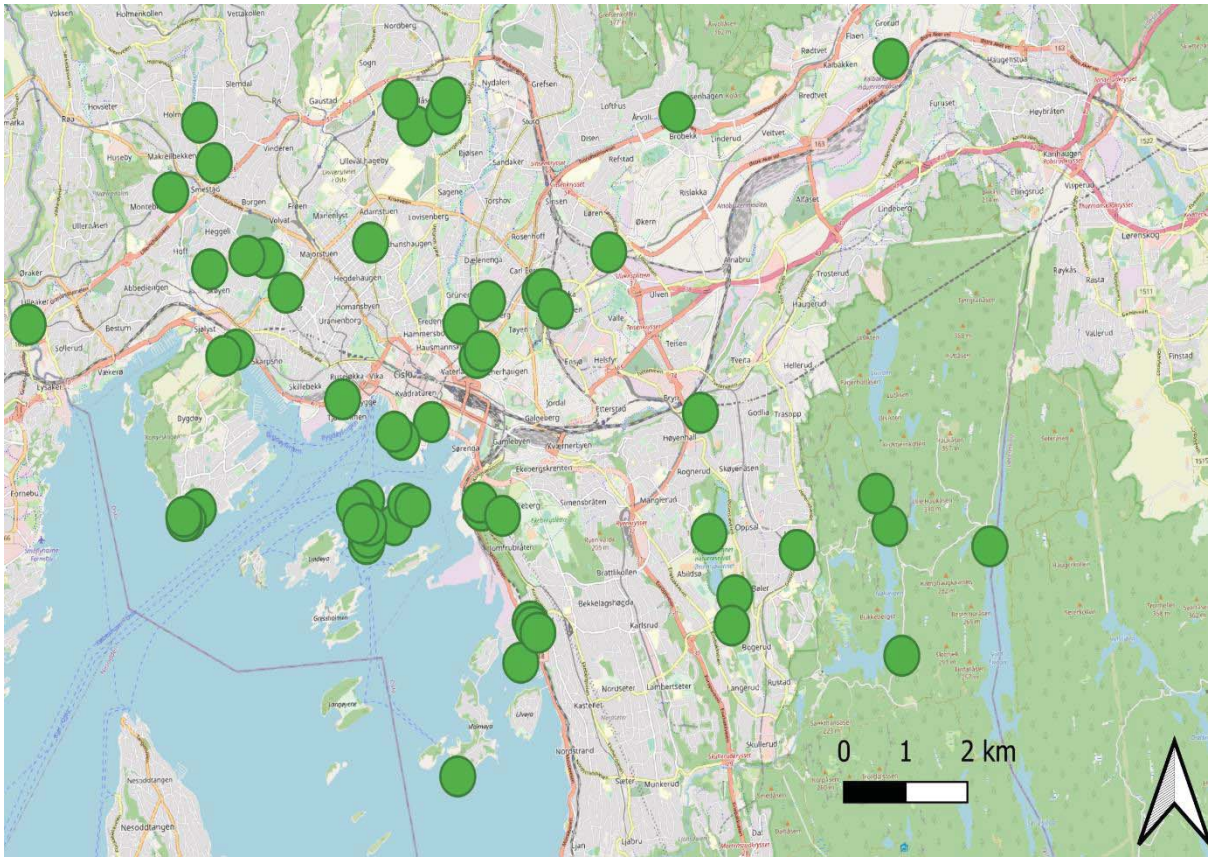

Figure 8: Existing native meadow patches, in and around Oslo, Norway (Bård Bredsen, 2020).

Although native meadow species on green roofs can alleviate anthropogenic encroachment and decreases the distance of isolation amongst urban green patches, their assemblages are affected by multiple factors (Aloisio et al., 2017, Butler et al., 2012, Dvorak and Volder, 2010). For example, colonization of green roofs by species that were not sown or part of the seed mixture and which are evaluated by the expert committee in fremmedartslista from Artdatabanken and found to occur outside their natural range or found to be foreign in Norway i.e. invasive species (Dunnett et al., 2008, Köhler, 2006, Madre et al., 2014, Nagase et al., 2013, Olly et al., 2011). Despite this, restorationists are not conscious of the fact that green roofs favour the colonization of beneficial and detrimental species of flora and fauna when establishing green roofs (Nagase et al., 2013). These invasive meadow species and their ability to dominate native plant communities, negatively affect the propagation of native plant families

by forming monocultures on some patches and green roofs (Hillebrand et al., 2008). Furthermore, Green roofs can be located at different points in the urban area, separated by anthropogenic and geologic features which will cause differences in meadow species pools due to differences in the number of species and microclimatic conditions across the city (Aloisio et al., 2019). Substrate depth may be considered as a factor influencing the composition of colonizers on green roofs, since native meadows species may decrease with decreasing substrate depth (Fargione et al., 2004, MacArthur and Levins, 1967, Smith et al., 2004). The colonization of green roofs may be promoted by species surrounding the roofs, wind and animal dispersal of seeds, abundance, and composition of native meadows species (Aloisio et al., 2019, Chase, 2003, Dunnett et al., 2008, Fukami, 2015). In fact, meadow community composition on green roofs changes between years and during one growing season making factors that influence their colonization difficult to understand (Benvenuti, 2014, Heim and Lundholm, 2016, Köhler, 2006, MacIvor et al., 2013). Moreover, from the evolutionary and meta-population dynamics of novel ecosystems, there may always exist the creation of ecological traps and sinks in urban areas which might cause the extinction of native populations (Fletcher Jr et al., 2012).

GIS Analysis

To show the connectedness and green space dynamics in Oslo, it was necessary to know the position and the distances between existing native meadows patches in and around Oslo. I generated a map of the existing native meadows patches in and around Oslo (Fig.8), showing distances between these patches (Fig.9), using QGIS 3.12 Bucuresti. This was created by uploading data with coordinates of the areas containing existing Norwegian meadow patches in and around Oslo into QGIS (Bård Bredsen, 2020). The background map and street map were searched by using the coordinate reference system as WGS84/UTMzone33N, with an authority ID as ESPG 32633N. The distances between these patches were calculated using the distance parameter in QGIS.

Distances Between Oslo Urban Green Patches

The lowest distances between the established native meadow patches in Oslo was 0.1 km and the highest was 0.7 km (Fig.9).

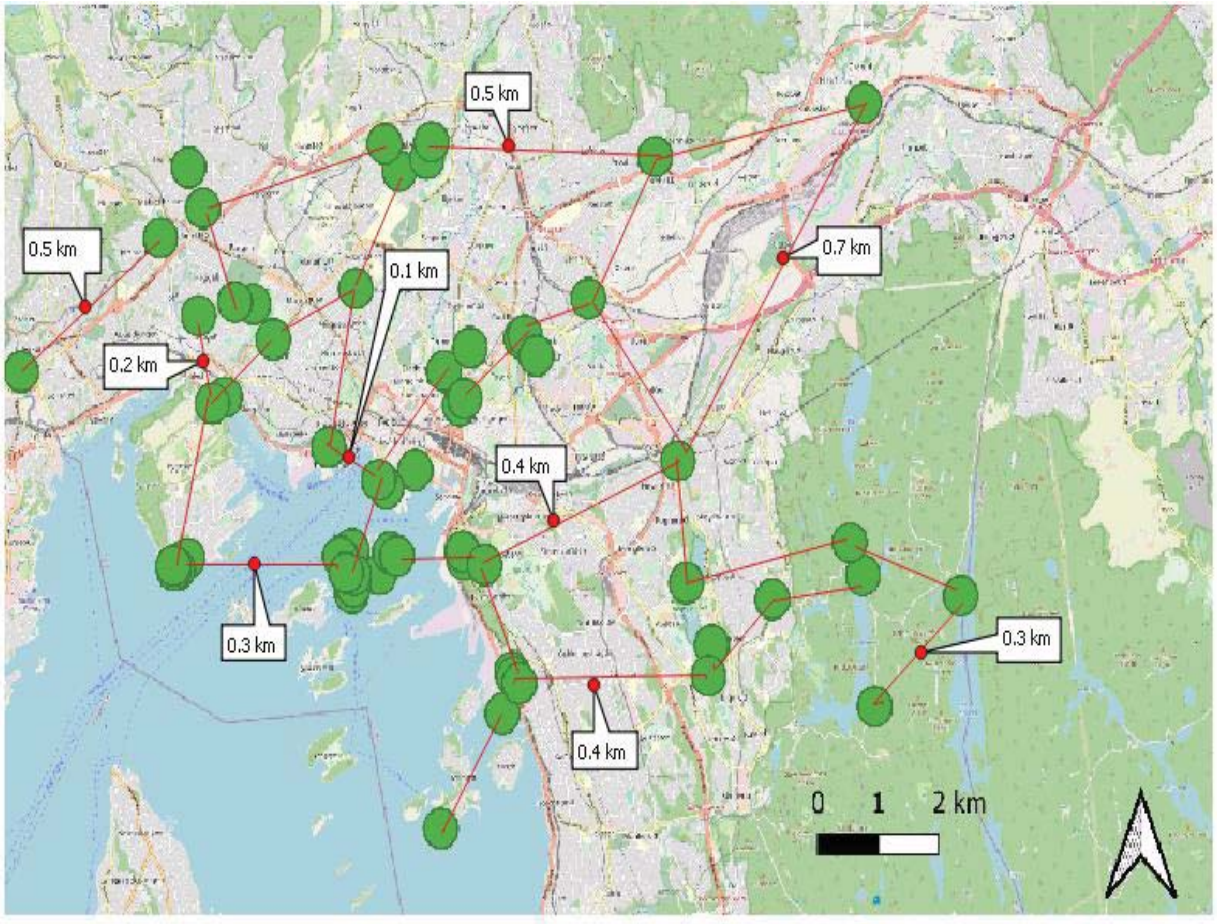

Figure 9: Existing native meadow patches, and the distances between them in and around Oslo, Norway (Bård Bredsen, 2020).

Improving Oslo urban biodiversity

Ultimately, the results from our experimental field test can provide improved methods when creating new green roofs that promote the growth of native meadow vegetation. This in turn will increase the connectivity between fragmented patches and enhance biodiversity in Oslo. With the results from our study we will establish more native Norwegian meadows patches as stepping-stones and juxtaposition them on/off green roofs between the remaining large remnant patches in Oslo`s urban area. These intermediate patches will link the large green patches with each other and with the surrounding forested areas. According to island biogeography theory, the size of these fragmented patches influences species richness (MacArthur and Wilson, 1967). Therefore, the small area of green roofs will reduce the number and type of species found in such habitats. Moreover, these roofs will be acting as stepping-

stones and increasing the connectivity between the remaining remnant green parches, thereby reducing the effects of fragmentation on the calcareous meadows in Oslo fjord region in Southern Norway (Evju et al., 2015a, Evju et al., 2015b). An increase in connectivity will facilitate immigration and colonization of native organisms from the surrounding forested areas, reducing the risk of extinction of native Norwegian flora and fauna in the Oslo region. Furthermore, a combination of these newly created or restored green patches will create an overall larger green space in Oslo urban landscape (Beninde et al., 2015). A larger greenspace will increase the movement and create more foraging grounds for pollinators and birds and a diversity of habitat type needed for their life cycles, hence, improving the biodiversity in the Oslo urban area (Braaker et al., 2014). Improving the movement of pollinators and birds within the Oslo urban matrix will enhance the transfer of pollen, dispersal of seeds, transfer of nutrient and energy and bring about the demographic rescue of these remnant green patches, thus, maintain the biodiversity in Oslo (Aronson et al., 2014, Hanski, 1998, Kowarik, 2011, Rudnick et al., 2012). Finally, our results will reduce the likelihood of creating ecological traps and sinks, reduce inbreeding depression, and increase the colonization of unoccupied habitat types in Oslo. The establishment or restoration of green roofs in Oslo will improve the biodiversity of flora and fauna in Oslo area generally and minimise the creation of ecological traps and sinks.

Appendix 4

Figure 10: Pictures showing the propagation of sown wild Norwegian meadows at our site.

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