Microclimatic and Diversity Controls on UK Grassland Carbon Cycling

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Declaration

I declare that the work produced for this thesis is my own, and has not been presented to obtain any other degree. Collaborations with other researchers are fully acknowledged.

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Statement of authorship

This thesis is prepared in the alternative format, as a series of four papers, intended for submission to peer reviewed journals. All papers have several authors. Their contributions to each paper are detailed below, and have been approved by my supervisors. Chapters 1 and 6 include an introduction and discussion respectively, and are not intended for submission.

Chapter 2 is intended for publication as: HL Stott, NJ Ostle, J Whitaker, AB Armstrong (2017) Interactive effects of solar radiation and temperature on carbon dioxide fluxes in high and low diversity grasslands,

HLS designed the experiment, collected and processed the samples, analysed the data and prepared the manuscript. ABA also collected samples and contributed significantly to the experimental design and revised manuscript. NJO and JW contributed significantly to the experimental design and revised manuscript

Chapter 3 is intended for publication as: HL Stott, NJ Ostle, J Whitaker, AB Armstrong (2017) Solar farms reduce the temperature sensitivity of leaf litter decomposition and alter microbial community composition

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HLS designed the experiment, collected and processed the samples, analysed the data and prepared the manuscript. ABA, NJO and JW contributed significantly to the experimental design and revised manuscript.

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Abstract

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The deployment of solar farms in the UK and Europe is accelerating in response to the need to decarbonize energy supplies and the increasing cost competitiveness of photovoltaic (PV) systems. In the UK, solar farms are generally installed on low grade agricultural grasslands. However, UK grasslands are important carbon stores, and the impact of the microclimatic changes induced by the presence of PV arrays, on the myriad of biological, chemical and physical processes which govern carbon cycling and ultimately carbon storage is uncertain. To understand how changes in temperature, soil moisture and solar radiation induced by the presence of PV arrays, affect grassland carbon cycling, these unique spatial and temporal microclimatic changes and their direct and indirect effects on grassland carbon cycling must be disentangled. Using a combination of field and laboratory experiments, this thesis investigates how the microclimatic changes imposed by solar farms affect ecosystem productivity and decomposition processes in temperate grasslands. Results show that the effects of warming and shading on productivity and decomposition processes were determined by the diversity of the vegetation community, with high diversity grassland communities more resistant to warming and shading than low diversity grasslands. Further, we provide some of the first evidence to show how decomposition in temperate grasslands may be affected by changes in solar radiation receipts, with UV-B exposure facilitating subsequent microbial decomposition. Both decomposition and productivity in grassland directly under the PV arrays were suppressed. Further, under PV arrays a greater proportion of biomass was invested as below ground biomass, which is a more stable pool of carbon and more likely to become part of stable carbon stores. Determining the impacts of solar energy farms on grassland carbon cycling could be used to develop

management strategies to help maximise the benefits of this renewable energy technology.

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1. Introduction: Climate change, grassland C cycling and land-based renewables

1.1 Climate Change

Since the agricultural and industrial revolutions, the clearing of natural vegetation and the burning of fossil fuels, has resulted in the loading of carbon-based greenhouse gases (GHGs) in the atmosphere (*IPCC*, 2013). The increasing concentrations of GHGs in the atmosphere has enhanced the greenhouse effect, causing the earth to warm by 0.85 ^{o}C since 1880 (*IPCC*, 2013). This rate of warming is unprecedented and poses a major threat to ecosystem functioning and the services that those functions provide (Shaw *et al.*, 2011; Montoya and Raffaelli, 2010; Mooney *et al.*, 2009). As global temperatures rise, weather patterns are disrupted, and the frequency and severity of extreme weather events i.e. drought and flooding increases (Greg, 2009; Konisky, Hughes and Kaylor, 2016). Through changes in cloud cover, ozone depletion and atmospheric particles, solar radiation receipts are changing (Wild, 2009). Changes in climatic variables such as temperature, moisture availability and radiation receipts affect C cycling, potentially leading to changes in GHG emissions to the atmosphere, creating feedbacks (Zepp *et al.*, 2011; Cox *et al.*, 2000; Zhang *et al.*, 2013). Despite this, our understanding of the effects of climate change on C cycling is incomplete, with significant knowledge gaps surrounding the effects of solar radiation on terrestrial C cycling. Consequently, there is clearly a pressing need to reduce GHG emissions and mitigate the effects of climate change.

1.2 UK Grassland C Cycling and Storage

Grasslands are important C stores: in the UK they store 2097 teragrams of C to a depth of 1 m and cover around a third of the UK land surface (Ward *et al.*, 2016). However, over the last 50 years, the intensification of agricultural systems through land management practices such as, fertiliser application, overgrazing and tillage, have perturbed C cycling in grasslands (Ostle *et al.*, 2009; Ward *et al.*, 2016). Further, changes in land management practices often have a strong influence on the grassland vegetation community, a key driver of C cycling in grasslands (De Deyn *et al.*, 2009; Steinbeiss *et al.*, 2008).

Land management is a strong driver of grassland ecosystem functions (Spurgeon *et al.*, 2013). During the 20th century technological developments and an expanding population drove changes in the way grassland systems in the UK (Jones *et al.*, 2013). There was primarily a shift

towards the intensification of land management, the application of fertilisers, lime and pesticides, draining land amongst other practices in an effort to improve the soil and crop productivity to allow for greater intensity of livestock production (Lambin and Meyfroidt, 2011). However, this intensification in grassland management has led to the degradation of grasslands and their soils, reducing the ability of the land to support livestock production and the loss of other ecosystem services (Soussana and Lemaire, 2014). This creates a negative feedback whereby further

In addition to land management practices, grassland C cycling is known to be highly dependent on climate (Brockett, Prescott and Grayston, 2012; Cao and Woodward, 1998). Under climate change predictions of increased temperatures and extreme precipitation patterns, there is a potential that vegetation communities may interact with climate change to exacerbate or mitigate the effect on soil C stores (Conti and Díaz, 2013). With an understanding of the biotic and abiotic effects and interactions on these processes, it may be possible to manipulate land management strategies in a way which will promote the storage of C and in turn mitigate climate change (Armstrong, Ostle and Whitaker, 2016).

Grassland C cycling is controlled through a variety of processes including, photosynthesis and ecosystem respiration (Figure 1) (Cao and Woodward, 1998). Photosynthesis is the biological process by which atmospheric CO_2 is converted to organic compounds (Raven and Karley, 2006). Around half of photosynthesised C compounds are lost in autotrophic respiration: the C remaining after respiration represents the C stored in the biosphere (Raven and Karley, 2006). Ecosystem respiration is the production of CO_2 in an ecosystem by autotrophs and heterotrophs (Gianelle *et al.*, 2009). C compounds in rhizodeposits (organic C release from the roots into the surrounding soil) and leaf litter, are oxidised by soil heterotrophs, predominantly bacteria and fungi, releasing CO_2 back into the atmosphere (Jones, Nguyen and Finlay, 2009).

1.2.1 Climatic Controls on Grassland C Cycling

Rates of photosynthesis and ecosystem respiration, and subsequently the balance between these processes controlling ecosystem C sequestration, are highly dependent on climatic controls such as temperature, soil moisture availability and radiation receipts (Lloyd and Taylor, 1994; Briones *et al.*, 2014; Foereid *et al.*, 2011). The

response of ecosystems to changes in climate have the potential to create positive or negative feedbacks, exacerbating or mitigating atmospheric C loading (*IPCC*, 2013; Davidson and Janssens, 2006; Dufresne *et al.*, 2002). However, given the potential for feedbacks and implications for climate sensitivity, our understanding of them is limited, specifically solar radiation and its interactions with temperature and soil moisture in temperate ecosystems.

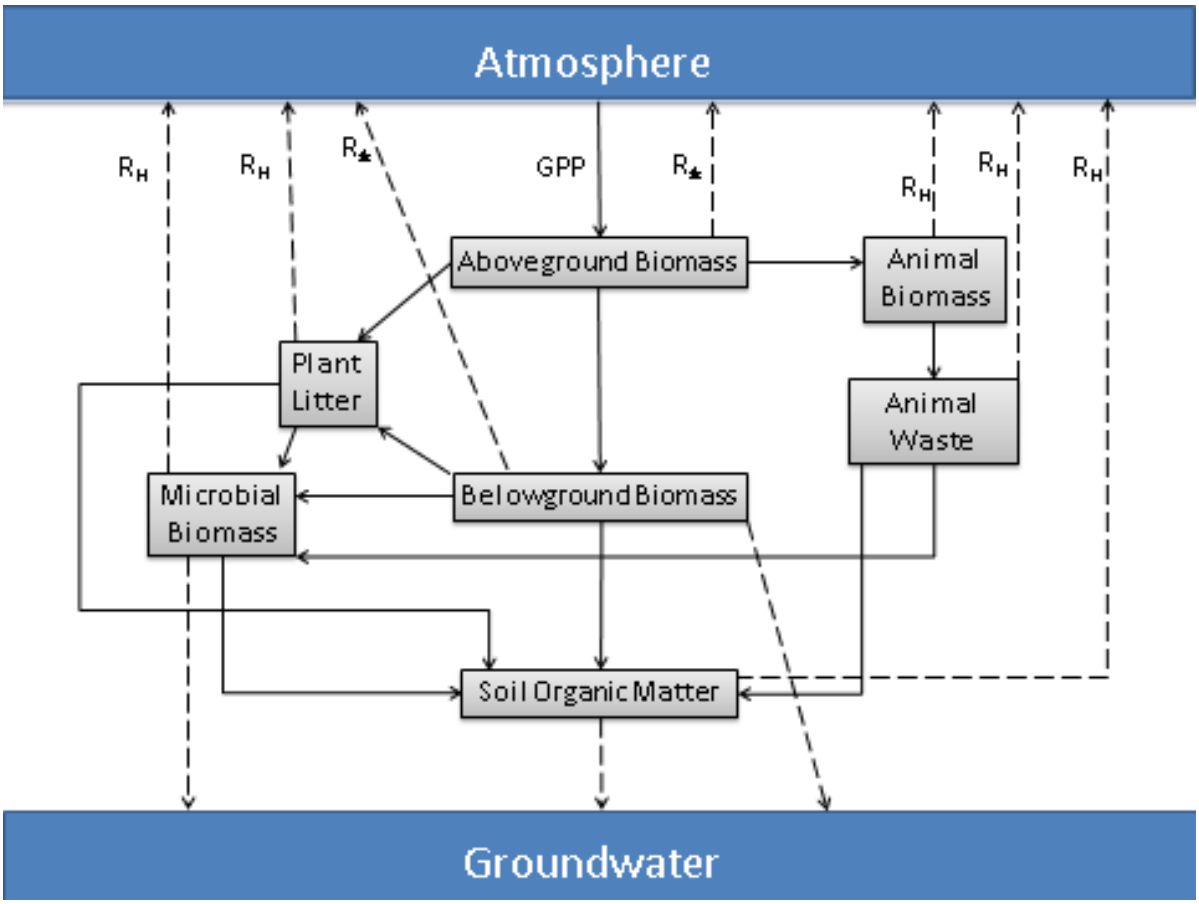


Figure 1. Flow diagram of the grassland C cycle. Boxes represent C pools, solid arrows represent fluxes into and within the ecosystem and dashed arrows represent fluxes out of the ecosystem (GPP, Gross Primary Production; R_A, Autotrophic Respiration; R_H, Heterotrophic Respiration)

Temperature

Temperature is an important control in terrestrial carbon cycling (Zhang *et al.*, 2013). Directly, temperature controls rates of biological activity of enzymes involved in production and decomposition processes (Bradford, 2013). Short-term experiments

have shown that soil microbial respiration increases with temperature (Davidson and Janssens, 2006) and that the rate of photosynthesis also increases with temperature (Mathur, Agrawal and Jajoo, 2014b). This may mean that as ecosystems become warmer the balance of production and decomposition processes is affected. If decomposition occurs at a greater rate there may be a decrease in C storage (Kirschbaum, 2010), whereas if production processes are more responsive to the change in temperature C storage increases (Raven and Karley, 2006). However, a meta-analysis of experiments from a variety of ecosystems found that warming of 0.3-6 ^{o}C over 2-9 years, increased soil respiration rates by 20 %, whilst plant productivity by 19 % (Rustad *et al.*, 2001). This indicates that decomposition processes are slightly more sensitive to changing temperatures than productivity processes, resulting in a loss of ecosystem C storage capacity. Changes in temperature may also affect soil microbial community composition further impacting soil respiration rates, with higher temperatures increasing soil respiration to a threshold (Zhang *et al.*, 2013). Indirectly, changes in productivity and vegetation community composition induced by temperature change may result in changes to the quality and quantity of organic matter inputs to the soil in the form of leaf litter or rhizosphere deposits (Jones, Nguyen and Finlay, 2009). These leaf litter and rhizodeposits inputs are important controls on soil microbes affecting community composition and activity, which in turn affects ecosystem C storage (Bray, Kitajima and Mack, 2012). Further, it is likely that these direct and indirect effects may interact to affect soil microbes. However, considerable knowledge gaps, in our understanding of the effect of temperature on biological C cycling exist.

Soil Moisture

Soil moisture availability is an important control on terrestrial C cycling affecting productivity and decomposition processes (Ise and Moorcroft, 2006). Changes in soil moisture induced through climate and or land-use change have the potential to increase or decrease terrestrial C stores and create climate feedbacks, where more C is released in to the atmosphere further enhancing the greenhouse effect and leading to further warming (Zhang *et al.*, 2013). Rates of photosynthesis are generally greater in wetter climates, however thresholds exist (Hall, 1994). Low soil moisture inhibits primary productivity through two processes. Firstly, water is an essential reactant in photosynthesis and without it, the process cannot happen (Hall, 1994). Secondly, when water which is lost through transpiration a pressure gradient is created which draws

moisture up from the soil through the xylem, if there is insufficient water in the soil, the stomata are closed to prevent further water loss. However, when the stomata are closed gaseous exchange is also inhibited. Without the import of CO_2 into the plant, photosynthetic rates rapidly fall (Hall, 1994).

In temperate ecosystems, the decomposition of organic matter is dominated by microbes (Swift, 1979). Microbial activity is extremely sensitive to soil moisture availability (Brockett, Prescott and Grayston, 2012). Generally, as soil moisture increases rates of decomposition also increase. Water facilitates the movement of nutrients and the exchange of O_2 and CO_2 at the cell wall surface, which is essential for respiration and the growth of the community (Madigan and Brock, 2009). However, in very high water content soils microbial activity is suppressed. This is due to a lack of oxygen in the soil which prevents the breakdown of phenols by phenol oxidase (Freeman, Ostle and Kang, 2001). The subsequent build-up of phenols in the soil inhibits microbial activity, suppressing aerobic decomposition processes (Freeman, Ostle and Kang, 2001).

Solar Radiation

In temperate ecosystems, lower levels of photosynthetically active radiation (PAR) generally reduce photosynthesis particularly in light demanding species, reducing C inputs to soil, and potentially C storage (Raven and Karley, 2006; Tkemaladze and Makhashvili, 2016). Root production in temperate grasslands is associated with radiation flux (Edwards *et al.*, 2004). Specifically, root biomass, length, birth rate, number and turnover are reduced by reductions in PAR (Rozema *et al.*, 1997). Radiation has been found to be a driver of decomposition (Austin and Vivanco, 2006). Radiation induced changes to leaf litter and rhizodeposits affect microbial decomposition processes, through alterations to the leaf litter chemistry and thus the lability of leaf litter (McLeod, Newsham and Fry, 2007). In addition to the changes in leaf litter and rhizodeposit inputs, radiation can exert controls on decomposition processes through changes to substrate chemical properties and microbial activity (King, Brandt and Adair, 2012; Bahn *et al.*, 2013). In arid ecosystems, the physical degradation of organic matter by photons (photodegradation) leads not only to the direct decomposition of organic matter but the change in the substrate chemical properties may facilitate microbial decomposition processes (Foereid *et al.*, 2010; Gallo *et al.*, 2009), however this is poorly resolved in temperate ecosystems. Additionally, shading may interact with

warming and plant communities and buffer C losses through reduced decomposition in warmer climes.

1.2.2 Biodiversity Controls on Grassland C Cycling

Plant, microbial and faunal diversity have all been shown to directly affect and interact with each other to impact grasslands with feedbacks to their carbon cycling (Yeates *et al.*, 1997; Kristin *et al.*, 2016; Stevens, 2018).

Plant diversity is central to the carbon cycle, soil C storage and GHG emissions (De Deyn *et al.*, 2009). Generally, there is a positive relationship between plant diversity and carbon storage, with ecosystem C storage falling as result of plant diversity losses (Cornwell *et al.*, 2008). There are two bodies of theory to explain this relationship: changes to leaf litter quality and quantity, and changes to the microbial community. Changes to vegetation community diversity may affect growth rates and cycles, which can lead to changes in the quantity and timing of leaf litter inputs and rhizodeposits (Bardgett and Shine, 1999; Meier and Bowman, 2008). Changes to leaf litter inputs can also affect soil coverage; with reductions in soil coverage generally increasing decomposition rates (Henry, Brizgys and Field, 2008). In addition, the chemical composition of the leaf litter and rhizodeposits inputs can change, which can play a key role in determining decomposition rates (Jones, Nguyen and Finlay, 2009; Aerts, 1997). For example, if a species or a group of species whose litter contains a larger proportion the recalcitrant chemical lignin is lost from a vegetation community, decomposition rates are likely to increase as the microbes are no longer inhibited by the lignin (Talbot and Treseder, 2012). Changes in plant communities have been directly associated with changes in the soil microbial community, which affects decomposition rates (Zak *et al.*, 2003; Garbeva, van Elsas and van Veen, 2008). Furthermore, plant diversity has been shown to mediate the response of carbon dynamics to climate change (Steinbeiss *et al.*, 2008) (Conti and Díaz, 2013) (Wood, Cavaleri and Reed, 2012). Generally, a high diversity community is more ecologically resistant: it is more likely to possess the trait necessary to adapt to changes in climate to remain in the same ecosystem state with the same functions (Fornara and Tilman, 2008) (Tilman, Wedin and Knops, 1996). However, there is limited understanding of how the variation in plant functional traits responds to changes in climate and affects grassland carbon dynamics. With regards to plant diversity- carbon cycling relationships in grasslands, it has been shown that it is not the number of species per say but the composition of forbs and legumes.

Specifically, soil C and N pools were enhanced by the presence of *Lotus corniculatus* and *Trifolium repens* (De Deyn *et al.*, 2009). Productivity in grasslands is also in part determined by the presence of legumes, with strongest effects in low fertility soils (Grime, 1988; Fornara and Tilman, 2008).

The diversity of the microbial community has also in part been found to correlate to plant diversity (Reese *et al.*, 2018). Mycorrhizal relationships have been shown to enhance plant diversity through supporting the growth of subordinate plant species. In grasslands arbuscular mycorrhizal fungi (AMF) form symbiotic relationships with plants, where the fungal mycelium enhances the root network, playing an important role in nutrient and in particular P acquisition. However, in return the plant loses some energy to the fungus. There is some evidence to suggest that in addition to this mutualistic lifestyle, during times when photosynthesis is limited and energy inputs from the plant to the mycorrhizae decrease, mycorrhizal fungi can adapt to saprophytic lifestyles thereby playing a direct role in the decomposition of soil carbon stores (Moore *et al.*, 2015). Further, AMF have been found to suppress soil respiration rates, playing a important role in soil carbon stabilisation; a mechanism that became more important under extreme climatic conditions (Bingwei *et al.*, 2016).

The balance of bacterial and fungal energy channels in grassland ecosystems is hypothesised to be a major factor in determining grassland carbon cycling. There is an increasing body of evidence to suggest that the sustainability of agro-ecosystems requires the soil community to be dominated by fungi as opposed to bacteria (Ushio, Miki and Balser, 2013). It is the asymmetry of bacterial and fungal energy channels which imparts stable soil functions, thereby enhancing resistance and resilience to perturbations. In most natural terrestrial systems, soil communities are dominated by fungi. The mineralisation of organic matter through the slower fungal energy channel, promotes the cycling of carbon and nutrients in the soil, enhancing soil conditions for sustainable crop production. Many intensive agricultural practices have shifted the soil fungal to bacterial ratio, resulting in more carbon being cycled through the fast bacterial channel, subsequently degrading soil fertility. One theory proposes that deep and heavy tilling, breaks up the mycelium network restricting fungal activity. Further, increases in soil N through the application of inorganic fertilisers, strongly correlates to the loss of fungal activity. However, when organic fertilizers are utilised which balance the C and N being added to the soil system, fungal communities are not negatively affected and

crop yields often increase, due to more efficient nutrient cycling and fewer nutrient losses.

The diversity of the soil faunal community is also plays an important role in in soil C and N cycling (Lavelle *et al.*, 1995; Briones *et al.*, 2009). The abundance of predators indirectly affects soil C and N cycling by controlling the populations of fungal grazing soil organisms such as mites and collembola (Wardle *et al.*, 1995). Intensive land management practices have been associated with reductions in soil predator populations, which allows for the unsustainable rise in fungal grazing populations (Filser *et al.*, 2002; Filser, 2002). As the populations of fungal grazers increase the fungal population decreases resulting in the aforementioned effects of fungal community decline on soil C cycling.

1.3 Biotic and Abitoic Decomposition Agents

Decomposition in temperate ecosystems is thought to be dominated by microbial processes, however, recently it has been shown to be driven by photodegradation in arid ecosystems, although our understanding is limited in more mesic systems (Austin, Méndez and Ballaré, 2016).

Microbial decomposition is the breakdown of organic matter by microbes, whereas photodegradation is the abiotic process by which solar irradiance breaks down the compounds of organic matter(King, Brandt and Adair, 2012). Microbial decomposition and photodegradation are controlled by three main factors: climate, litter quality and soil organisms (Prescott, 2010; Butenschoen, Scheu and Eisenhauer, 2011). Debate exists, as to which factors exert the dominant control over microbial decomposition and photodegradation (Bradford *et al.*, 2016; Gaxiola and Armesto, 2015). Factors such as climate, microbial communities and soil properties have been found to be the best predictors of microbial decomposition (Aerts, 1997). Climate modulates microbial decomposition, through temperature and moisture availability (Davidson and Janssens, 2006) which regulate soil microbial activity (Gaxiola and Armesto, 2015). Generally, microbial decomposition is greatest in warmer wetter environments (Ise and Moorcroft, 2006). However, thresholds exist where excess moisture or high temperatures start to inhibit microbial activity (Freeman, Ostle and Kang, 2001; Briones *et al.*, 2014; Bradford, 2013). There is an increasing body of evidence to suggest that vegetation communities exert a dominant control over microbial decomposition rates (Bakker,

Carreno-Rocabado and Poorter, 2011; Boyero *et al.*, 2014). Specifically, it may be the presence or absence of a key species or plant functional type from a community which controls decomposition rates (Cornwell *et al.*, 2008). Differences in the vegetation community composition, affect microbial decomposition through changes in leaf litter and rhizodeposits chemistry and quantity, the allocation of C above and below ground, and microbial communities (Ward *et al.*, 2015; Wood, Cavaleri and Reed, 2012; Lamb, Kennedy and Siciliano, 2011; Drenovsky *et al.*, 2010). Litter quality and climate modulate microbial activity and are fundamental controls on litter decomposition in terrestrial ecosystems and subsequently mediate ecosystem carbon storage (Aerts, 1997; Meentemeyer, 1978; Bakker, Carreno-Rocabado and Poorter, 2011; Gaxiola and Armesto, 2015). Litter quality variables crucial to decomposition processes include nutrient concentrations, carbon-to-nitrogen ratios (C: N) and lignin content (Parton *et al.*, 2007; Melillo, Aber and Muratore, 1982; Talbot and Treseder, 2012; Talbot *et al.*, 2012). Specifically, increases in nutrient availability and lower C:N ratios increase microbial decomposition, and increases in lignin decrease microbial decomposition.

Home field advantage refers to the theory that leaf litter are decomposed at a faster rate in their native ecosystems in contrast to foreign ecosystems (Jacob *et al.*, 2010; Ayres *et al.*, 2009). This is due to the interactions between plants and their co-adapted food webs, which contain niche biotic decomposition agents specially adapted to be able to decompose complex species specific compounds (Palozzi and Lindo, 2018). This allows the soil organisms to efficiently decompose the plant material, allowing for the plant to reuptake the nutrients lost in leaf litter (Veen *et al.*, 2018). Home field advantage may further benefit plants, through the fast processing of leaf litter which can increase the risk of plant pathogens (Austin *et al.*, 2014). Due to the predominant nature of microbial decomposition, home field advantage studies focus mainly on microbial communities, however, it can also refer to microbial and soil fauna communities and their interactions (Milcu and Manning, 2011).

Soil fauna, are generally classified by size. Microfauna (< 0.1 mm), refer to protists, rotifers and nematodes, these generally feed on bacteria, however a few are fungivores, detrivores and predators (Bezemer *et al.*, 2010). Mesofauna (0.1 - 2mm) include arthropods such as collembola and mites, are important fungal and bacterial grazers indirectly affecting SOM processing (Amorim *et al.*, 2012). Further, soil mesofauna consume degraded organic matter: subsequently their excretions containing this organic

matter are more readily available to fungi and bacteria to breakdown. Soil Macrofauna (> 2 mm) include macroarthropods, annelids and gastropods (Wall and Bardgett, 2012). Earthworms, a member of the annelids, are ecosystem engineers- they are a major control on the physical, chemical and biological properties of their habitat, impact interactions with other organisms and regulate ecosystem functions (Forey *et al.*, 2018). They are major controls on ecosystem functions such and decomposition, nutrient cycling, soil structure, water infiltration, soil food web, disease suppression, and primary productivity (Briones and Álvarez-Otero, 2018). Depending on their life strategy earthworm are classified as either (1) epigeic, (2) endogeic or (3) anecic. Epigeic earthworms feed on and dwell in leaf litter layers. The abundance of earthworms positively correlates the leaf litter decomposition rates (Muys, Lust and Granval, 1992). Earthworms do not mineralise most of the carbon in leaf litter layers, the vast majority is excreted back to the soil or humus layer. Leaf litter which has been passed through earthworms is more labile, and can be more readily transformed in to soil humus by the action of soil microbes (Cortez, 1998).

1.4 Land-based renewables

In response to the pressures of climate change and increasing energy demands, policies have been devised to promote the development of the renewable energy infrastructure. By 2020, the UK has committed to producing 15 % of its energy from renewable resources. Land-based renewables such as solar farms, wind farms and bioenergy crops offer a potential solution to meeting energy demands in a sustainable manner. However, land-based renewables represent a substantial land-use change due to their low energy density and the rapid expansion on a global scale (Armstrong *et al.*, 2014; Turney and Fthenakis, 2011). Changes in microclimate and land management practices induced by land-based renewables have the potential to affect plant-soil processes controlling ecosystem services such as C cycling (Figure 2). It is essential that our understanding of the effects of land-based renewables on hosting ecosystem C cycling and C storage is developed to determine the true C cost and sustainability of these renewable energy technologies. The UK government signed international agreements pledging to generate 15% of its energy demands from renewable sources (DECC, 2014). Land-

based renewables in the UK, in the form of solar, wind and bioenergy crops have in response accelerated (Montag, Parker and Clarkson, 2016; Popp *et al.*, 2014). However, this sea change in the UK's energy infrastructure, accounts for a large shift in land use and has potential impacts on ecosystem functioning (Armstrong *et al.*, 2014). The pertinent question being, do these strategies to reduce C emissions have their own unintended impacts on ecosystem C cycling and storage amongst any other environmental impacts?

Bioenergy crops result in the direct change in vegetation type and land management(Keith *et al.*, 2015). This can result in changes in ecosystem services, with changes in soil carbon stocks being the primary determinant of whether the establishment of a bioenergy crop on land was positive or negative in terms of net GHG emissions (Popp *et al.*, 2014). One major factor in determining the net effect on GHG emissions was the soils initial carbon stock: with soils originally high in carbon, having greater losses when established as Miscanthus or Short Rotation Coppice (SRC) sites(Rowe *et al.*, 2016). On lower C soils such as those commonly found in arable land, net GHG emissions dropped, when land management changes to bioenergy production (Parmar *et al.*, 2014).

Whilst the impact of wind energy on bird and bat populations is relatively well studied (Pearce-Higgins *et al.*, 2009; Newson *et al.*, 2017), current research surrounding the impacts on plant soil interactions is extremely limited. It has been predicted that wind turbines affect surface meteorology, namely wind speed, turbulence and mixing, subsequently resulting in changes in energy distribution, and exchange at the land surface (Armstrong *et al.*, 2014), this could result in changes in temperature and humidity, which are important controls on plant soil processes.

Studies surrounding the impact of solar farms on carbon cycling and other ecosystem services are limited, however due to increases in their deployment globally, new research on this area is emerging (Raúl *et al.*, 2018). Some of this research has focused on effects on crop production, however it was found that only small changes in cropping practices in agriPV systems would be required, and that any changes in management should focus on minimising the effects of light reduction through PV design and the selection of shade tolerant plants (Marrou *et al.*, 2013) (Raúl *et al.*, 2018). Microclimatic differences induced by the presence of PV arrays and land management, have been

observed at UK solar farms and are of a magnitude known to affect plant-soil processes such as photosynthesis and net ecosystem exchange, however these effects vary spatially and temporally (Armstrong, Ostle and Whitaker, 2016). This clearly demonstrates the value to this PhD research to better understand how solar farms can impact ecosystem services.

1.4.1 Solar Farms

Solar photovoltaic (PV) electricity generation in the UK has expanded at an unprecedented rate since 2010, from a capacity of 32 MW to over 10 GW in 2016. This trend is reflected globally, with 73 GW installed in 2016, bringing the total global capacity to 310 GW (Focus, 2016). The growth of solar PV is expected to continue and by 2050 solar PV could be the dominant renewable energy source globally (IEA, 2014).

A substantial proportion of PV is installed as ground-mounted solar farms (EPIA, 2016). Solar farms in Europe are predominantly located on grasslands or on land previously managed for arable purposes but after the installation of PV arrays a grassland plant community is established. In the UK up to April 2016, the total operational capacity of large-scale solar PV deployment (predominantly ground mounted systems) had reached an operational capacity of 10,967 MW (DBEIS, 2016), covering approximately 222 km^2 of land (Burke, 2015). Solar farms, therefore, present an interesting and unique change in grassland land management: generally, a move away from intensive grassland management practices such as overgrazing, fertiliser application and tillage, towards less intensive management regimes. In addition to energy production, many solar farms are managed to provide and enhance multiple grassland ecosystem services such as biodiversity and food production.

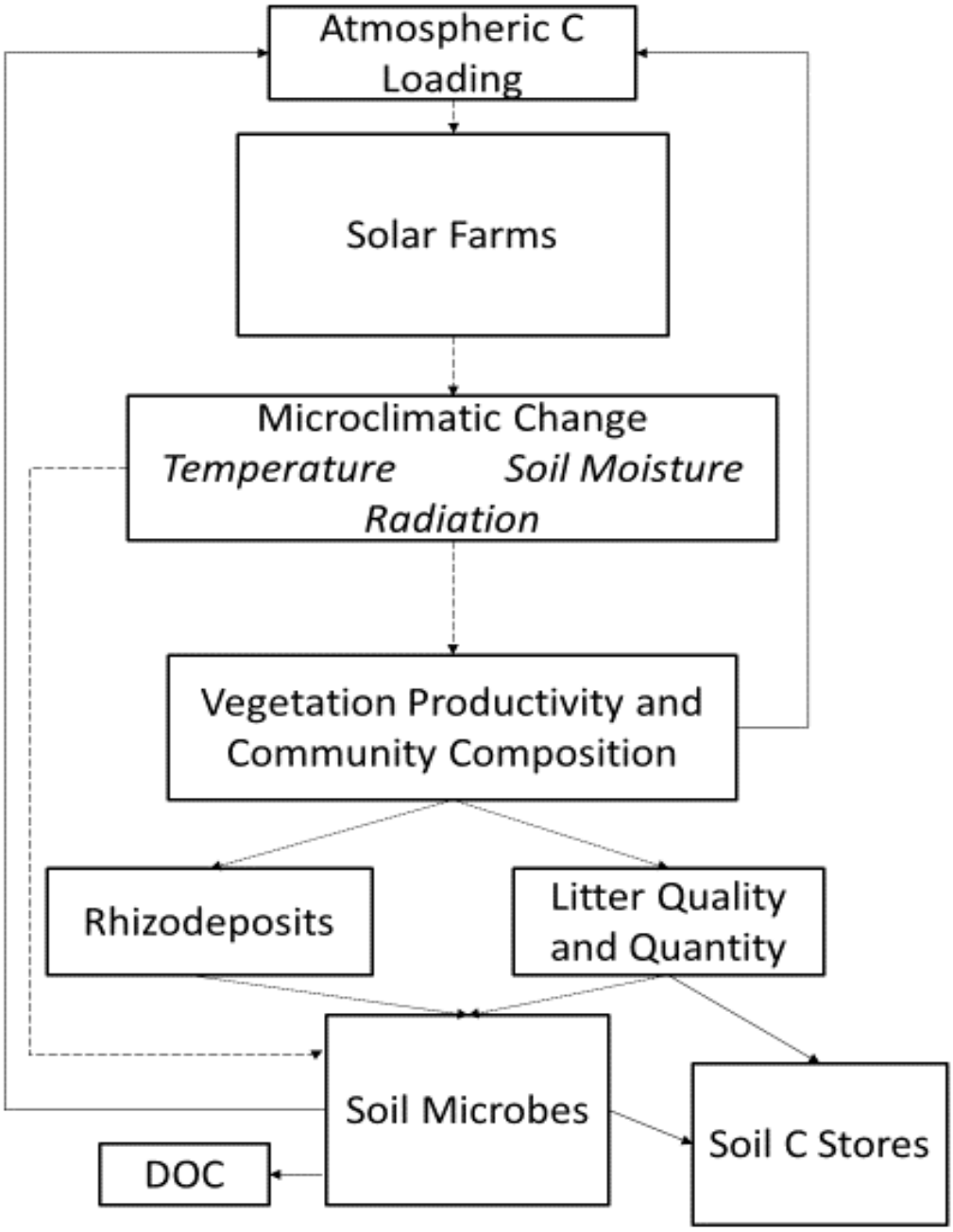


Figure 2. Direct (solid lines) and indirect (dashed lines) effects of solar farms on grassland carbon cycling

1.4.1.1 Solar Farm Induced Microclimatic Effects

The presence of PV (photovoltaic) arrays has been shown to alter the microclimatic conditions both above and below ground (Armstrong, Ostle and Whitaker, 2016; Marrou *et al.*, 2013) (Figure 3). Solar farms may alter grassland C cycling and C storage, through changes to radiation receipts, temperature and soil moisture. The presence of PV arrays has been found to reduce the total photosynthetically active radiation

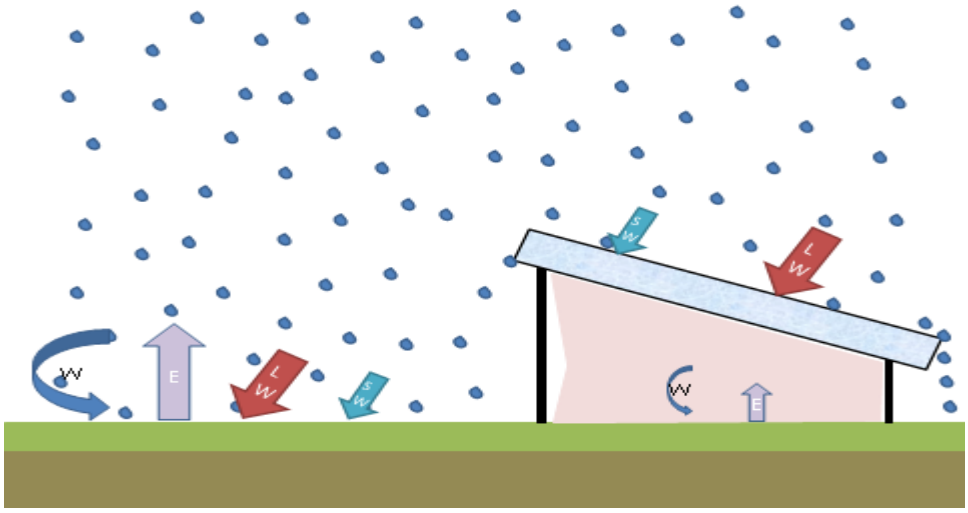
reaching the grassland surface by 92 % and alter the proportion of radiation which is diffuse by 11 % under the PV arrays (Armstrong, Ostle and Whitaker, 2016). This has potential implications for C cycling as there may be a change in plant productivity and in turn decomposition processes, if the quantity of leaf litter inputs changes (Zhang and Wang, 2015).

In the summer the presence of PV arrays has been found to suppress temperatures under the PV arrays by as much a 5.2 °C (Armstrong, Ostle and Whitaker, 2016). Temperature is a major control on grassland C cycling, affecting productivity and decomposition processes (Bradford, 2013; Davidson and Janssens, 2006; Day, Ruhland and Xiong, 2008). However, the effects of changes in temperature due to the presence of PV arrays, on grassland C cycling are uncertain (Armstrong *et al.*, 2014).

The impact of solar farms on soil moisture is unclear(Armstrong *et al.*, 2014). The design of many solar arrays often causes the water to be funnelled, potentially creating a mosaic of varying soil moisture (from high to low) under the panels. However, the extent of variation in soil moisture may be affected by factors such as root infiltration, soil type, and soil structure (Bell *et al.*, 1980). It is uncertain how changes in soil moisture caused by solar farms may impact carbon cycling.

Microclimatic changes, over time, may lead to changes in species composition, which has the potential to alter the function of the vegetation community. Plant diversity has been shown to mediate response to climate change (Steinbeiss *et al.*, 2008; Conti and Díaz, 2013; Wood, Cavaleri and Reed, 2012). Solar farmed systems are increasingly being managed to promote biodiversity (BRE, 2014), it is, therefore, necessary to assess whether changes to grassland plant diversity could mitigate impacts of warming and shading on C cycling

A



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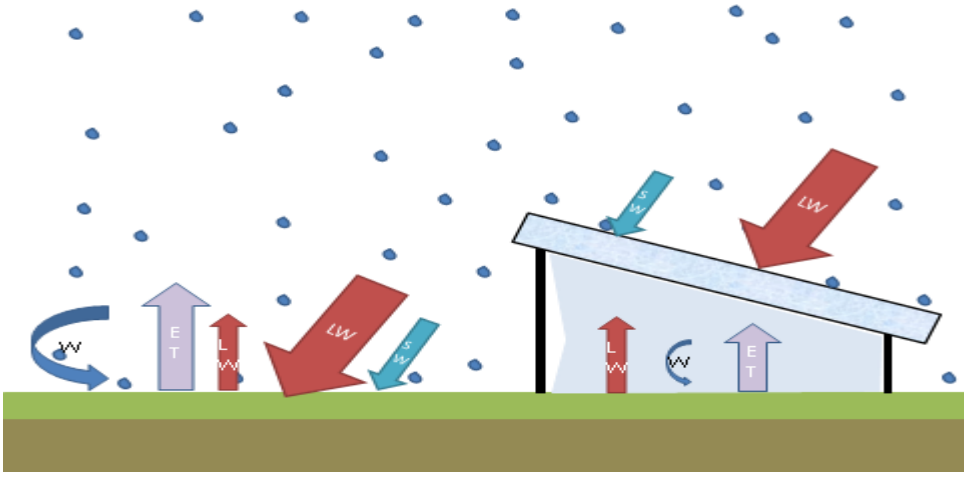

Figure 3. Microclimatic effects of PV arrays, which intercept shortwave radiation (SW), longwave radiation (LW) and precipitation. Microclimatic effects are dependent on current climatic conditions. In the winter (a) the area directly under the PV array is warmer due to the lower ambient temperatures and the sheltering effect, which reduces wind speed (W) and traps LW from the ground. The reduced wind speed under the PV array in winter reduces evaporation under the panel and due to the greater precipitation rates and the movement of water throughout the soil, soil moisture is greater under the PV arrays. In summer (b), the area directly under the PV arrays in cooler, and soil moisture levels are lower. The higher ambient air temperature means that any sheltering effect of the PV arrays is not great enough to make up for the deficit in LW receipts. Soil moisture under the PV arrays is lower, due to the interception of precipitation in addition to relatively high evapotranspiration rates due to the plant community under the PV arrays.

1.4.1.2 Solar Farm Management Strategies

On solar farms, the move away from traditional agricultural practices in addition to the implementation of site appropriate biodiversity action plans have been found to have a positive impact on biodiversity (Hernandez *et al.*, 2013). Solar farms managed for biodiversity often have a short period of grazing over winter, which helps maintain a habitat which promotes flora and fauna diversity. However, many solar farms in the UK have continued use for year-round agricultural purposes, predominantly through sheep grazing, providing another dual land-use scenario (BRE, 2014). As part of the Solar trade Associations "10 commitments" of good practice, solar farm developers actively encourage multi-purpose land-use, through the implementation of land management plans which aim to support multiple ecosystem service production (biodiversity-food-energy) to maximise the benefits of this technology through careful land management strategies (BRE, 2014).

1.5 • The function of different light wave-lengths in biological systems, role of shade in influencing plant growth.

The electromagnetic spectrum is made up of electromagnetic radiation of varying wavelengths. The spectrum is divided in to different bands depending of wavelengths, these bands are known as radio waves, microwaves, terahertz waves, infrared, visible light, ultraviolet, x rays and gamma rays. In biological systems the light of wavelengths between 400-700 nm utilised by plants for photosynthesis, is referred to as photosynthectically active radiation (Edwards *et al.*, 2004). Chlorophyll a is the primary photosynthetic pigment, whereas, chlorophyll b is an accessory pigment that captures energy for transfer to chlorophyll a (Van Gaalen, Flanagan and Peddle, 2007).

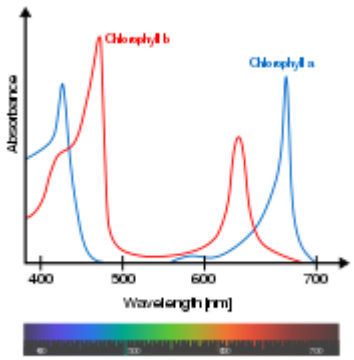

Figure 4. Absorbance spectra of chlorophyll a and chlorophyll b

UV radiation acts as an eco-physiological factor which plays an important role in plant growth and photosynthesis (Ballaré *et al.*, 2011). A plants response to UV is regulated by the presence of protective pigments, adaptive leaf morphology and antioxidant responses (Klem *et al.*, 2012). The balance between UV and PAR absorbance has been found to be an important factor in controlling effects of UV on plant growth. Specifically, when PAR levels are high, UV-B has been found to have no negative effect and on occasion a positive effect on plant growth/ photosynthesis (Klem *et al.*, 2012).

Much attention has focused on the competition by plants for resources of light, however, the facilitative effect of shading on plant growth and plant communities are poorly resolved (Semchenko *et al.*, 2012). Studies have found that the presence of shade can be beneficial for shade tolerant species and provide protection from UV rays (Paoletti, 2005). The growth response of drought or poor nutrient tolerant grassland species to shade is greater than non-stress tolerant species (Semchenko *et al.*, 2012).

1.6 Plant chemistry: plant cell walls and rhizoexudation

Plant cell walls are protective and supporting structures which control the passage of substances in and out of the cell (Sarkar, Bosneaga and Auer, 2009). Cell walls are primarily composed of cellulose together with other matrix polysaccharides such as hemicellulose (Smith, 1977). Cellulose is a linear polymer made up of hundreds of glucose molecules, these polymers aggregate to form microfibrils and most abundant in the secondary lamella layers (Rose, 2003). Hemicellulose is the most abundant carbohydrate in the compound middle lamella and binds strongly to cellulose microfibrils(Rose, 2003). Unlike other cell wall polymers, lignin is the only non-carbohydrate based compound as a cross linked phenolic polymer (Sarkar, Bosneaga and Auer, 2009). Lignin functions to strengthen and waterproof plants cells, however, its deposition varies dependant on cell function, for example xylem tracheids, vessel elements and sclereid cells all have high cell wall lignin levels (Rose, 2003).

Rhizodeposition is defined as material lost from plant roots in to the soil system: this includes exudates, secretions of insoluble materials, lysates, dead fine roots and gas efflux (Lynch and Whipps, 1990). Rhizoexudates are water soluble and predominantly formed of sugars, amino acids and organic acids (Jones, Nguyen and Finlay, 2009). Once in the pedosphere, these rhizoexudates are often rapidly metabolised by fungi and

bacteria, and are thought to represent up to a fifth of all carbon fixed by plants through photosynthesis (Churchland and Grayston, 2014). The quantity and quality of rhizoexudates, determines microbial niche and subsequent ecological functions (Huang *et al.*, 2014). In soils with fungal dominated microbial communities, rhizoexudates may be primary energy source for soil food webs, however this research is ongoing to challenge the traditional view that the primary energy source in to these food webs is derived from leaf litter inputs.

1.7 Methods for determining the soil microbiological community.

Soil microbial communities are intricately linked to ecosystem functions, playing integral roles in carbon and nutrient cycles (de Vries and Shade, 2013). However, our understanding of these soil microbial communities and associated functions is extremely limited (Malik *et al.*, 2016; Thomson *et al.*, 2015). A range of techniques to deduce soil microbial community composition and function are discussed below.

1.7.1 Molecular Techniques

Molecular techniques for analysing the soil microbial community in-situ have emerged over the last few decades (Griffiths *et al.*, 2000). The studying of microbial genes in-situ allows for the analysis of the environmental context and the microbial networks the microbes form and their functional roles (de Vries *et al.*, 2018). However, very large gaps in our knowledge persist and molecular techniques are expensive and slow. In the coming years molecular techniques are predicted to increase in prevalence as new technological developments such as DNA chips emerge (Levy and Myers, 2016).

1.7.2 PLFA

Phospholipid fatty acids (PLFAs) are key components of bacteria and eukarya cell membranes (Buyer and Sasser, 2012). PLFAs are classified by different carbon chain lengths and structure (Heng *et al.*, 2017). PLFAs play a key role in cell-membrane architecture and cellular function (Yeagle, 1989). In soil science, PLFAs are used to assess the composition of fungi and bacteria communities (Quideau *et al.*, 2016). Carbon chain lengths of between 14-20 atoms are considered to be the dominant cell membrane PLFA components for fungi and bacteria (Malik *et al.*, 2016). PLFAs cannot be used to identify individual species of fungi or bacteria, however, the identification of specific PLFAs and their relative abundance to other PLFAs can be used as a marker of that soils function (Quideau *et al.*, 2016). On cell death PLFAs are rapidly degraded

in the environment, therefore PLFA analysis is considered to be a true snapshot in time of the microbial community, which isn't contaminated by historic populations (Buyer and Sasser, 2012).

Commonly, in soil PLFA analysis, the differing PLFAs are grouped into fungi, bacteria, gram positive bacteria and gram negative bacteria, in addition to a measure for total PLFAs (Frostegård, Tunlid and Bååth, 2011). However, the relative abundance of individual PLFAs can also be assessed. Total PLFAs can in addition be converted to a measure of microbial biomass. The ratio of fungi to bacteria is hypothesised to be indicative of the slow versus fast energy pathways. However, using PLFAs as markers for specific microbial groups can be contentious (Frostegård, Tunlid and Bååth, 2011). For example 18:1u9 is in some soils considered to be a good fungal indicator, however in soils where fungal levels are low this is a poor indicator, due to its presence in bacteria and plants (Kaiser *et al.*, 2010).

Despite advances, rapidly increasing the processing ability of PLFA samples, there is a shift towards molecular techniques. However, when comparing treatment effects on a specific geographic location PLFA is a greater indicator of shifts in microbial community functioning in response to perturbation.

1.7.3 Ergosterol

Ergosterol plays an important role in the fugal growth and is found in the cell membrane (Stahl and Parkin, 1996). Ergosterol can be used to determine living fungal biomass in soils (Montgomery *et al.*, 2000). However, current methods are slow and require large volumes of reagents. Further, conversion factors of ergosterol to fungal biomass are disputed and wide ranging (Wallander *et al.*, 2013).

1.7.4 Catabolic profiles

The catabolic profile of a soil microbial community can be analysed using a multi-substrate induced respiration test (Campbell *et al.*, 2003). This involves testing the ability of a soil to metabolise a range of compounds, by analysing CO_2 evolution, giving an indication of the function of the microbial community i.e. the relative abundance of microbes able to metabolise lignin.

1.7.5 Extracellular enzyme activity

Microbial functions can also be assessed via extracellular enzyme activities (Meyers and Edwards 2014). Extracellular enzymes are secreted by microbes directly in to the environment and work to breakdown organic materials in to more labile compounds which can then be up taken by the microbes to acquire nutrients and energy .

Methods

This thesis will employ a range of methods to assess biological, physical and chemical soil and plant properties to elucidate how microclimatic changes induced by solar farms may impact ecosystem functioning.

In chapter 2 I used a three factor warming, shading and diversity experiment to simulate solar farm climatic conditions to see how carbon cycling is affected in these ecosystems. Initial visits to Collymore farm in October 2013, were made to get a first-hand look at a solar farm, and potentially identify areas of the farm with contrasting management histories that could be used for experimental purposes. On this visit, two proximate fields were identified with vegetation representative of intensive (low diversity) and extensive (high diversity) management regimes. In March 2014, intact cores were removed from these fields and transported to Hazelrigg Research Station, Lancaster, UK. These cores were then transplanted into the grassland, bound by their extraction tube, to isolate it from the transplantation environment. The native grassland was then cut and maintained to a height of < 5cm throughout the duration of the experiment. Shade tents were erected at two different densities to remove light. The three levels of shading were tested using PAR meters over the course of a week and were found to block out 0 %, 74 % and 90 % of PAR. Open top chambers applied a passive warming treatment, however, the size of the warming effect was dependent on ambient temperature.

The experiment consisted of 5 replicate blocks, with a fully factorial design to allow for the analysis of effects and interactions on ecosystem functions.


Chapter 3 used a classic decomposition study method: litter bags. The litter bags used in this study were filled with recently senesced *Poa annua* leaf litter collected from an area of undisturbed, monoclimatic grassland at Hazelrigg Research Station, Lancaster, UK, in October 2014 and oven dried at 60 $^{\circ}$C. The litter was placed in litter bags at a mass to area ratio of (0.5 g: 25 cm^2). The litter bags were constructed from a 1 mm PVC coated fibre glass mesh. To ensure accurate mass loss measurements litter bags were transported to the field in sealed bags to ascertain how much litter was lost in transit. On the 31/3/2015 four litter bags were placed at each of the ambient and warmed treatments, in each plot at the soil surface under any existing litter layer.

Litter bags were recovered after 3 months (26/6/2015), 6 months (29/9/15), 9 months (7/12/15) and 12 months (29/3/16,) and were stored at 4 $^{\circ}$C until processed. For the purposes of data analysis (seasonal average temperatures), in this experiment we defined the seasons as the period between the sampling dates i.e. 31/3/15 to 26/6/2015 (spring), 27/6/15- 29/9/15 (summer), 30/9/15- 7/12/15 (autumn) and 8/12/15- 29/3/16 (winter).

Large roots and soil were removed by hand from the outside of the recovered litter bags, which were then carefully rinsed with deionised water over a 1 mm sieve. Litter bags were then cut open and any remaining roots and soil were removed using tweezers. The cleaned litter was dried at 60 $^{\circ}$C to a constant weight (Ward et al., 2015).

The mass loss data was then used to calculate the decomposition rate of the litter bags under the different location and warming treatments, by using the formula:-

$X_{(t)} = e^{-kt}$

where $X_{(t)}$ is the proportion of the original litter remaining and t is the time in days since the litter bags were installed.

Chapter 4 follows up on the findings of chapter 3 which found differences in decomposition rates under solar panels. Specifically, chapter 4 looks at the potential role of UV-B in the facilitation of microbial decomposition. Briefly, three different senesced leaf litters of contrasting chemistry were collected, half exposed to high levels of UV radiation, then subsequent effects on microbial decomposition assessed over time. Chapter 4 assess data from two experiments, experiment A and experiment B:-

Experiment A assessed the effects of UV-B pre-exposure on the decomposition of three litter types (grass, conifer and broadleaf). The factorial design combined two levels of radiation treatment (UV-B pre-exposed and unexposed) and three litter types (grass broadleaf, conifer) with five replicates of each treatment combination destructively harvested at four time points (120 microcosms in total). All litter used in experiment A was initially decomposed for 2 months at 20 $^{\circ}$C[1]. After the initial decomposition period, mass loss was determined on one set of replicates, for the remaining sampling sets, the soil in each petri dish was replaced with fresh soil before the secondary decomposition period, at 15 $^{\circ}$C. In the secondary decomposition period mass loss was measured over 6 months, at two monthly intervals[1].

Experiment B used only the UV-B pre-exposed and unexposed grass litter. This experiment assessed the impact of UV-B pre-exposure on short-term decomposition, as grass litter was found to have a large mass loss in the 0-2 month decomposition period in experiment A. The same decomposition set up was used as previously described, however this time 0.5 g of litter was used and mass loss at 15 ^{o}C was measured after 1, 2, 4 and 10 weeks. For each time point, there were five replicates.

This unconventional approach was undertaken as the initial temperature of 20 °C had caused the soil to dry out. Therefore, the soil was replaced and the petri dishes were then incubated at 15 °C. After this a decision was made to investigate short term decomposition. Ideally, all three litter types would have been used in this analysis, however, there was insufficient quantities of broadleaf and conifer litters. Consequently, experiment B only includes the grass leaf litter.

Chapter 5 assessed the effects of solar farms on vegetation, and aim to assess how management practices and climate may contribute to the success of the solar farm in terms of ecosystem service provision. To assess this above and below-ground biomass, species composition, vegetation height and leaf C: N in grasses, under photovoltaic (PV) arrays, in gaps between PV arrays and in control areas, was measured at 17 sites across England and Wales, in June 2016.

The sites selected for this study had a wide ranging climate, land use history and current land management practices alongside the sites role as a solar farm. Although there are no site replicates in this experimental design, the gradients method of analysis is similar to Bahn's suggestion that ecological studies should be performed on a gradient to disentangle relationships. Due to the difficulties being granted access to solar farms, it was not possible to use the traditional replication design i.e. grouping sites by climate type, such as 600-650 mm of rainfall p.a. The following pictures demonstrate the wide ranging properties of the solar farms used in this study.







At all of the 17 solar farms, five plots (1 m x 1 m) were randomly selected within each of the three designated treatment areas: under the PV arrays, in the gaps between the PV arrays and in the control area, giving a total of 15 plots per site. Control measurements were taken away from the PV arrays, whilst still remaining in the enclosed area of the solar farm to ensure management strategies were similar. Gap measurements were taken directly in the centre of the area between two rows of arrays (Figure 17). Under the PV arrays, measurements were taken in the centre of the panel

away from areas where water may be channelled through gaps between individual panels. At all of the sampling locations, we identified all of the distinct species present, using a 1 x 1 m open quadrat (Klimek et al. 2007) and collected soil cores for below ground biomass analysis, and soil and grass samples for C:N analysis (leaf, root and soil). In addition, at sites which had not been grazed in the three months prior to sampling we also measured total above ground biomass, above ground biomass by plant functional type and vegetation height.

1.9 Research Objectives

The over-arching aim of this research was to assess the effect of solar farms on grassland C cycling in the UK. This deliver this aim to objectives were devised and assessed using a combination of field and laboratory experiments:-

1. How does plant diversity and solar radiation affect decomposition processes? (Chapter 2, Chapter 3 and Chapter 4)
2. How does plant diversity and solar radiation affect productivity processes? (Chapter 2 and Chapter 5)

In chapter 2, I used a triple factor warming and shading experiment, to test whether CO_2 fluxes in high and low diversity grasslands are contrastingly affected by these factors. Briefly, intact grassland cores extracted from high and low diversity grasslands, and transplanted under warmed and ambient temperature conditions created using open top chambers, and three levels of shading using neutral density shade cloth. Over the course of one growing season, weekly measurements of ecosystem respiration, photosynthesis and net ecosystem exchange to determine effects on carbon cycling. In addition, climatic, soil and vegetation measurements were taken to be analysed in conjunction with the CO_2 flux data.

Chapter 3 and 4 assessed the effect of solar farms on decomposition processes. Firstly, Chapter 3 uses a field experiment which aimed to assess whether microclimatic changes induced by the presence of PV arrays affected the decomposition of leaf litter. Here, litter bags, sampled at three monthly intervals over the course of a year were used to determine decomposition rates. In addition, we analysed soil microbial community composition and monitored temperature, radiation receipts and soil moisture as variables to explain differences in decomposition rates. Chapter 4, investigates whether in temperate ecosystems where microbial decomposition dominates, changes in solar

radiation, specifically, the UV-B fraction, can influence decomposition. In this dual factor experiment, I assessed how UV-B exposure prior to microbial decomposition could affect the decomposition of three litter types with varying litter chemistry characteristics. Grass, broadleaf and conifer leaf litter was collected and sterilised and half was exposed to a UV-B treatment for 6 months. After the UV-B treatment, the exposed and unexposed leaf litters were rewet and inoculated with a loamy clayey soil collected from a field at Hazelrigg Research Station. Mass loss was measured at 2 monthly intervals, over a period of 8 months to calculate decomposition rates for the different treatments.

In the final experimental chapter (Chapter 5), I investigated the effects of solar farms on vegetation. To assess this above and below-ground biomass, species composition, vegetation height and leaf C: N in grasses, under photovoltaic (PV) arrays, in gaps between PV arrays and in control areas, was measured at 17 sites across England and Wales, in June 2016.

Finally, Chapter 6 pulls together the work from the previous chapters into a holistic examination of this research within a wider context. A general synthesis in provided of the studies contained within the thesis as well as a discussion of the lessons learned and suggested future research.

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2. Interactive effects of solar radiation and temperature on carbon dioxide fluxes in high and low diversity grasslands

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Data referred to in this chapter can be viewed in full in the appendices presented at the end of this thesis.

2.1 Abstract

Understanding controls on carbon (C) cycling in grasslands is critical given their role as important C stores. Climate is a key driver of C cycling in grasslands, however, the impacts of solar radiation are relatively poorly understood. Further, there is growing evidence that plant diversity can modulate climate change effects on carbon cycling.

To investigate the interactions between grassland diversity, photosynthetically active radiation (PAR) and temperature, we took intact cores from high and low diversity grasslands and exposed them to passive warming and shading treatments, and measured the effects on microclimate, CO_2 fluxes, vegetation and soil properties.

We found that grassland diversity, PAR, and temperature affected grassland CO_2 fluxes with plant diversity being the most influential factor. Above ground biomass, C: N and vegetation community composition were affected by changes in temperature and PAR and their interactions with diversity. Also, low diversity grasslands were generally more sensitive to changes in temperature and PAR, than high diversity grasslands.

These results demonstrate the need to resolve the importance of PAR and its interactions with temperature and diversity in temperate grassland C cycling to predict the impacts

of environmental change, including changes in solar radiation receipts due to changes in cloud cover, atmosphere and land-use change.

2.2 Introduction

Carbon (C) stores in terrestrial ecosystems are important to help mitigate the impacts of CO_2 loading in the atmosphere, however, these stores are threatened by land-use change, vegetation removal, pollution and climate change (*IPCC*, 2013). Grasslands are globally significant C stores. In the UK grasslands store around a tonne of C per hectare, in above ground biomass and around 30-100 g of C per kg of soil below ground; this equates to one-third of total UK soil carbon stores (Ostle *et al.*, 2009; Bellamy *et al.*, 2005). Consequently, environmental change that affects C cycling may increase or decrease carbon storage and could significantly impact greenhouse gas (GHG) emissions (Guo and Gifford, 2002). The consequences of temperature and moisture changes on grassland C cycling are relatively well understood (Briones *et al.*, 2009; Flanagan and Johnson, 2005), however, there has been relatively little research on the effect of solar radiation receipts, namely photosynthetically active radiation (PAR), and its interactions with temperature (Mercado *et al.*, 2009a; Rutledge *et al.*, 2010)

Globally, PAR receipts are changing with cloud cover, ozone depletion, atmospheric particle loading, vegetation removal and land-use change. Under various climate change scenarios, as the atmosphere warms, more moisture will evaporate from the land and water into the atmosphere and in some places is likely to result in increased cloud cover (Zhu *et al.*, 2013). Specifically, the Clausius-Clapeyron equation means that a 1^{o}C increase in global temperature will increase the atmospheric water-holding capacity by 7 % (Allaby, 2013). However, in other regions, if weather patterns are disrupted, and precipitation levels fall, cloud cover may be reduced. Changes in cloud cover or atmospheric particle loading through various other processes (natural or anthropogenic), will directly affect solar radiation receipts in both quantity and type (diffuse/direct) (Mercado *et al.*, 2009b).

The effects of changes in PAR and temperature are likely to be observable in ecosystems undergoing microclimatic changes through land-use changes such as the construction of solar farms. Terrestrial ecosystems are increasingly hosting renewable energy production, (namely bioenergy, wind farms and solar farms) to mitigate against

the effects of global environmental change (Armstrong *et al.*, 2014; Hernandez *et al.*, 2013; Ostle *et al.*, 2009). In the UK solar PV represents a substantial land-use change, as the number and size of solar energy farms continue to grow (DECC, 2012; DECC, 2014). The microclimatic changes imposed by a solar farm such as changes in the spatial distribution of precipitation reaching the ground, soil and air temperature, and radiation receipts, means that this technology has a high potential to affect biogeochemical processes, the ecosystem services that these functions govern, with the potential to create positive or negative environmental change feedbacks (Armstrong *et al.*, 2014). Specifically, the presence of PV arrays has been found to reduce the total photosynthetically active radiation reaching the grassland surface by 92 %, and reduce temperature by up to 5.2 ^{o}C (Armstrong, Ostle and Whitaker, 2016). In addition, solar farms are increasingly being managed to promote biodiversity (BRE, 2014), it is, therefore, necessary to assess whether changes to grassland plant diversity could mitigate impacts of warming and shading on C cycling.

Temperature is a major control on grassland C cycling, affecting productivity and decomposition processes (Nie *et al.*, 2015; Bradford, 2013; Davidson and Janssens, 2006; Day, Ruhland and Xiong, 2008). This may mean that as ecosystems become warmer decomposition occurs at a greater rate, resulting in a decrease in C storage (Kirschbaum, 2010), or that production processes are more responsive to changes in temperature and C storage increases through greater vegetation inputs (Raven and Karley, 2006). Short-term experiments have shown that soil microbial respiration, a key component of ecosystem respiration increases with temperature (Davidson and Janssens, 2006) and that the rate of photosynthesis also increases to a threshold dependent on abiotic and biotic factors, with temperature (Mathur, Agrawal and Jajoo, 2014a). A meta-analysis of experiments from a variety of ecosystems found that warming of 0.3-6 ^{o}C over 2-9 years, increased soil respiration rates by 20 %, whilst above ground plant productivity increased by 19 % (Rustad *et al.*, 2001). However, the response of ecosystem respiration and plant productivity was dependent on ecosystem type. Specifically, in grassland ecosystems, the response of soil respiration to warming was generally lower, than in other ecosystems. However, despite much research, there is currently no consensus on the effect of warming on ecosystem C storage (Davidson and Janssens, 2006). Further, the direct and indirect effects of warming on grassland C cycling are likely to be complex and display spatial and temporal variation.

Plant photosynthesis tends to increase with PAR receipts (Raven and Karley, 2006). However, recent theoretical and observational studies have demonstrated that photosynthesis is also more efficient under diffuse light conditions where maximum PAR receipts are suppressed (Mercado *et al.*, 2009b).

In temperate ecosystems lower levels of PAR generally reduce photosynthesis particularly in light demanding species, reducing C inputs to soil, and potentially C storage (Raven and Karley, 2006). In addition, PAR can exert controls on C cycling through changes to temperature, microbial activity and substrate chemical properties, such as leaf litter N and lignin contents, affecting productivity and decomposition processes (King, Brandt and Adair, 2012; Foereid *et al.*, 2011; Bahn *et al.*, 2013; Foereid *et al.*, 2010). PAR has been found to be a driver of decomposition, extensively in arid ecosystems, as well as non-water limited ecosystems such as Southern Hemisphere peatlands (Austin and Vivanco, 2006; Williamson *et al.*, 2014; Foereid *et al.*, 2011). Additionally, the effects of shading may be dependent on the vegetation community, specifically leaf litter chemistry and buffer C losses through reduced decomposition in warmer climes (Austin, Ballaré and Schlesinger, 2010). It is unclear as to the overall effect of this projected change in PAR on plant productivity and the land carbon sink, and how it may interact with atmospheric warming.

Plant communities are central to the carbon cycle (De Deyn *et al.*, 2009). Different functional traits such as litter quality and growth cycles alter the rates and input of organic matter for decomposition altering C cycling in communities (Bardgett, De Deyn and Ostle, 2009). Plant diversity has been shown to mediate response to climate change (Steinbeiss *et al.*, 2008; Conti and Díaz, 2013; Wood, Cavaleri and Reed, 2012; Zak *et al.*, 2003). Generally, a high diversity community is more ecologically resilient: it is more likely to adapt to changes in climate i.e. drought, extreme precipitation and warming, to remain in the same ecosystem state with the same functions (Fornara and Tilman, 2008; Tilman, Wedin and Knops, 1996). Moreover, given the complexities in C cycling, climatic variables, soil conditions and plant diversity may mediate or exacerbate the effect of each other (Wood, Cavaleri and Reed, 2012)

Under the pressures of climate change and land-use change, there is clearly a need to resolve our understanding of the effects of changes in temperature, PAR on grassland

carbon cycling. Further, there is an increasing body of evidence to suggest that the diversity of the grassland vegetation community is a factor controlling the response of grassland ecosystem C cycling to changes in climate.

The aim of this study was to investigate how differences in PAR and temperature interact to affect grassland C cycling in low and high diversity plant communities. To address this aim we took intact cores from high and low diversity grasslands and applied factorial passive warming and shading treatments for 6 months, measuring the effects on CO_2 fluxes. We hypothesised that ecosystem respiration, net ecosystem exchange and photosynthesis in high diversity grasslands would be more resistant to warming and shading than in low diversity grasslands.

2.3 Materials and Methods

2.3.1 Study site and experimental design

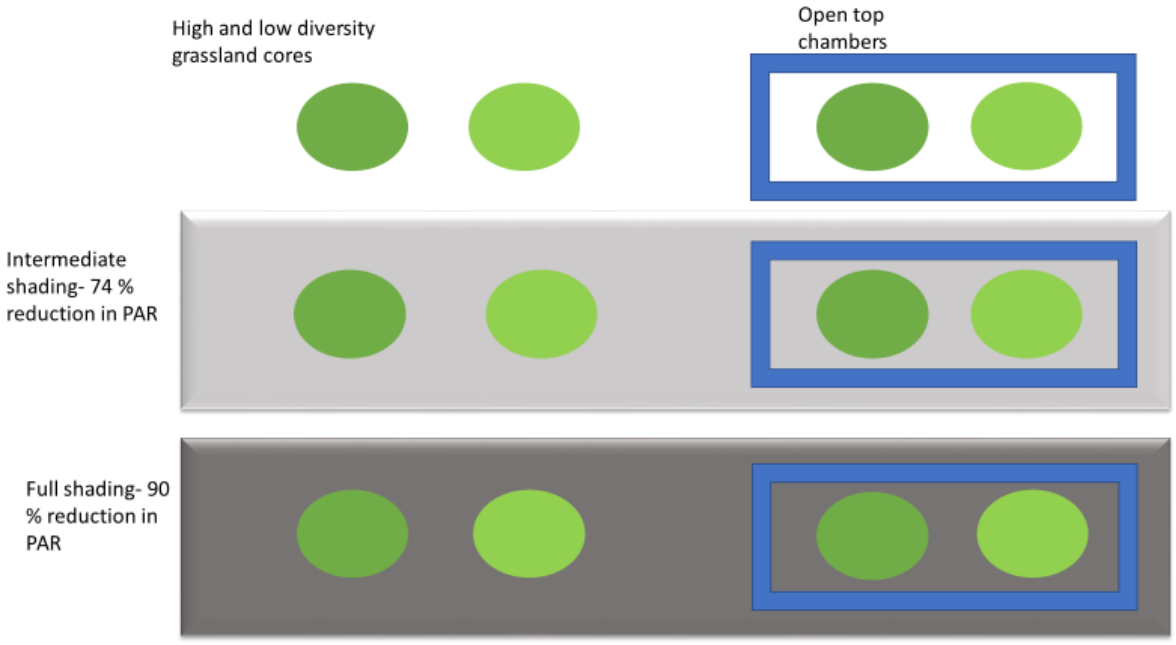
This was a fully factorial experiment, with three factors: PAR (3 levels, full PAR, intermediate PAR, low PAR), diversity (2 levels, high diversity, low diversity) and temperature (2 levels, ambient, warmed). This resulted in a total of 12 treatments, replicated in five blocks (figure 4).

Soil cores with intact vegetation (60 in total: 100 mm diameter x 220 mm height) were extracted from two fields approximately 0.5 km apart at Colleymore Farm, Oxfordshire (51.621241, -1.6575408: 112 m): one, a highly diverse grassland community, and the other, a low diversity grassland community. Colleymore Farm is located in an area of downland and limestone grassland, with freely draining shallow lime-rich soils over chalk or limestone (Soilscapes soil type 3) (Cranfield University 2017) . The high diversity community was a forb-rich calcareous grassland community (including *Avena fatua, Poa pratensis, Festuca rubra, Trifolium pratense, Ranunculus sardous, Equisetum pratense, Anacamptis pyramidalis*). In comparison, the low diversity community comprised of *Lolium perenne* and *Trifolium repens*. A species richness survey was conducted at the two sites in July 2014 to validate differences in the vegetation community. The cores were extracted using PVC-U tube (100 mm diameter x 220 mm height), which was hammered into the ground, until it was level with the ground, then dug out. Additionally, 6 soil cores from each site (3 cm diameter, 10 cm depth) were extracted to test field soil properties and check the spatial variability

between the two field sites (Appendix 1). There was no significant variability between the soils at the two field sites.

The study was conducted from May 2014 to November 2014, at Hazelrigg Research Station, Lancaster University, Lancashire, UK (54 0 57, -2 46 23: 95 m). At Hazelrigg the climate is temperate, with a mean annual (1985-2014) temperature of 9.3^{o}C, and a mean annual precipitation of 1133 mm (MetOffice, 2016). The high and low diversity intact cores, complete with the PVC-U extraction tube, were transplanted (dug into the ground- level with the surrounding vegetation) at Hazelrigg in April 2014, under permanent imposed warming and shading treatments (Figure 4). The cores were retained, in the extraction tubes, to maintain a barrier between the transplanted grassland cores and the native ecosystem at the study site. The shading treatment was implemented through the use of neutral density shade cloth made from 1 mm PVC coated fibre glass mesh (Streme), the intermediate PAR level which removed 74 % of PAR, was created using one layer of the cloth, whilst the low PAR level which removed 90 % of PAR, was created using two layers of the cloth. The cloth was suspended over the necessary cores using four wooden stakes in a rectangular formation at a height of 50 cm, and attached to the ground at all four sides using pegs. Open top chambers were used to passively warm half of the cores (Acrylic: 550 mm x 200 mm x 200 mm). The vegetation around the transplanted cores was maintained to a height of <5 cm to minimise the impact of competition from the surrounding plant community.

Figure 4. Experimental set up to investigate the effects of warming and shading on high and low diversity grasslands. This block set up was repeated five times.



2.3.2 Microclimate

To determine warming and shading effects on the microclimatic conditions, air temperature (°C) (10 cm above the soil surface with radiation shields) and soil temperature (°C) (5 cm below the soil surface) were recorded every 5 minutes from 12/5/14 15.00 GMT to 14/11/14 9.00 GMT (Tempcon UK: HOBO 8K Pendant® Temperature/Alarm (Waterproof) Data Logger). From this daily average temperatures were calculated for analysis In order to minimise disturbance to the transplanted cores, the soil and air temperature sensors were installed under each of the temperature and PAR treatment combinations in the native soil. In addition, at one of the replicated blocks, PAR was sampled every 5 minutes and the hourly average recorded, under the three distinct shading treatment levels (Tempcon, UK: HOBO Micro Station with S-LIA-M003). PAR data was recorded as µ Einstein's which is equal to µ mol m^{-2}.

To ensure that the shading and warming treatments were not influencing soil moisture, soil moisture was measured weekly (Delta-T Devices, ML3 Theta Probe Soil Moisture Sensor) in the soil next to the transplanted cores in each of the temperature and PAR treatment combinations in all replicate blocks.

2.3.3 CO_2 fluxes

CO_2 flux monitoring took place between 10.30 and 14.30, at approximately seven day intervals between May and November 2014 giving a total of 25 sampling points over the growing season. Using a EGM4 portable IRGA (PP systems, USA) coupled to a customised chamber lid 10 cm diameter and 20 cm height, made from opaque Perspex acrylic for the ecosystem respiration measurements and transparent Perspex acrylic for the net ecosystem exchange measurements, sealed to the transplanted core shell using synthetic clay, measurements of CO_2 exchange were made over 120-s intervals. The CO_2 fluxes in the dark and light chambers were used to determine ecosystem respiration and net ecosystem exchange respectively (Ward *et al.*, 2007). Ecosystem respiration and net ecosystem exchange data were used to generate calculated values for photosynthesis using the following equation: -

Photosynthesis = Net Ecosystem Exchange - Ecosystem Respiration

Negative NEE values are indicative of a carbon sink, whilst positive values represent an ecosystem where there is a net release of carbon to the atmosphere.

2.3.4 Vegetation

To gain a measure of the treatment impacts on above ground plant biomass and leaf C: N, each core was harvested in July. In addition to above ground biomass, the composition (% graminoids, % legumes, % forbs) was also determined by dry mass. To determine above ground total C and N, a 30 mg subsample of oven dried (60 °C) and ground (using a ball mill for 5 minutes) grass (from the dominant species from the two communities- smooth meadow for the high diversity cores and perennial rye grass from the low diversity cores), was analysed using a LECO Truspec CN Analyser (LECO, USA) and C:N was calculated (Carter, 2007).

2.3.5 Soil properties

In November 2014, the transplanted cores were removed and soil samples taken for analysis. Soil from each of the transplanted cores, along with the initial field cores, was analysed for total C and N, pH, soil moisture, NH_4^+, NO_3^- and PO_4^{3-}. Total C and N were determined on a 30 mg subsample of homogenised and oven dried (60 °C) and ground soil, using a LECO Truspec CN Analyser (LECO, USA) and C:N was calculated

(Carter, 2007). Soils were analysed for extractable NH_4^+ and NO_3^- using potassium chloride (KCl) extractions (Ward *et al.*, 2007). Briefly, fresh soil samples (8 g) were mixed with 1 mol/L KCl on an orbital shaker (model KS501 digital, IKA, Werke, Germany) and then filtered using Whatman no.1 paper. Concentrations of NH_4^+ and NO_3^- in filtrate were determined by colorimetric technique (Ross, 1992), on a continuous-flow stream autoanalyser (Autoanalyser 3, Bran Luebbe, Norderstedt, Germany). A 0.5 M sodium bicarbonate ($NaHCO_3$) solution at a pH of 8.5, was used to extract PO_4^{3-} from fresh soil (Rowell, 1994). The procedure, as described for the NH_4^+ and NO_3^- extraction and analysis, was used to determine PO_4^{3-} concentrations. Soil moisture content was determined by drying the sub samples (cores 3 cm x 5 cm) at 105^{o}C to a constant mass. Soil pH was determined through the use of a probe and meter (Seven Compact pH meter, Mettler Toledo) (soil: H_2O, 1: 2.5 w: v).

2.3.6 Statistical analysis

All data were processed and analysed using R studio (RStudio Team, 2015) and *p* values < 0.05 deemed significant. Differences in microclimate (soil and air temperature, and sunrise to sunset PAR receipts- defined from the first full hour after sunrise and full hour before sunset) were assessed using a two way ANOVA, with warming and shading and their interactions as factors. Whilst, soil properties (soil total C and N, pH, soil moisture, NH_4^+, NO_3^- and PO_4^{3-}) and vegetation (above ground plant biomass and leaf total C and N) were tested for significance using a three way ANOVA with shading, warming and diversity as factors. Tukey's pairwise comparisons post hoc tests were used in conjunction with the two and threeway ANOVAs.

The influence of solar radiation and temperature on carbon dioxide fluxes in the high and low diversity grassland cores, was tested using linear-mixed effects models with lme4 and lmerTest (to derive p values) packages in the R statistical program. A model was developed which tested the effects and two way interactions between PAR, temperature and diversity on CO_2 fluxes, including random intercept terms for block, temperature and PAR nested in date. In this model, the fixed effects of temperature and radiation were input as continuous variables (temperature: ^{o}C, Solar radiation: PAR µ mol m^2), whilst diversity was input as two levels (high and low). Model assumptions were scrutinised using fitted values versus residuals plots and QQ plots, and post hoc

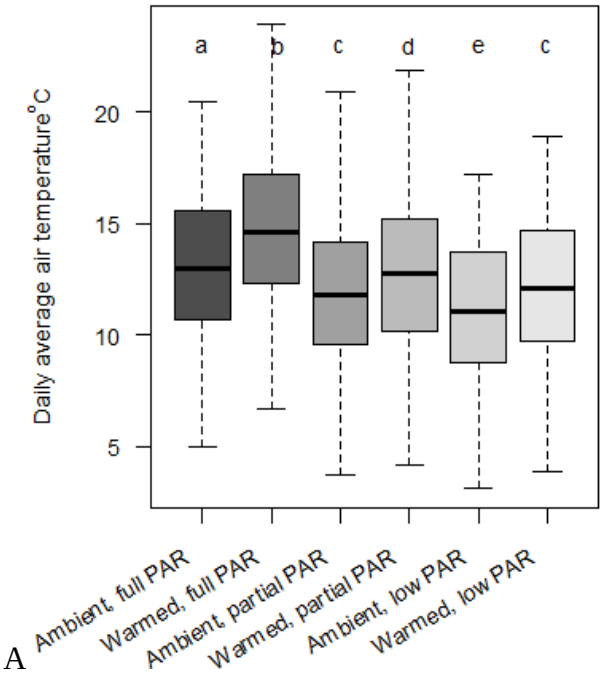
comparisons were made using Tukey's (Zuur, 2010). Coefficients of fixed effects for the model were used to determine effects size.

2.4 Results

In the following section, we outline the effects of the warming and shading treatments on microclimate, soil properties, biomass, vegetation community composition and CO_2 fluxes in high and low diversity grasslands.

2.4.1 Microclimate effects

For the duration of the experiment (6 months- growing season), warming and shading treatments significantly affected average air temperature ($df = 1$, $F = 73.9$, $p < 0.01$, Figure 5) and soil temperature ($df = 1$, $F = 91.8$, $p < 0.01$, Figure 5). It was observed that the warming effect on soil temperature was dampened by the significant interaction with the shading treatments ($df = 1$, $F = 8.5$, $p < 0.05$). Specifically, with 100% PAR exposure soil temperature increased by 3.1 ^{o}C, in comparison to warming of 2.5^{o}C for plots with 74 % of PAR removed and warming of 1.7 ^{o}C for plots with 90 % of ambient PAR removed.



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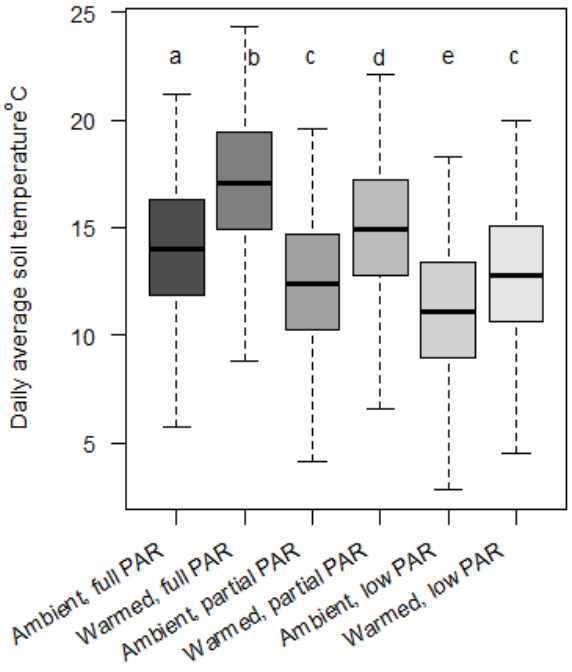

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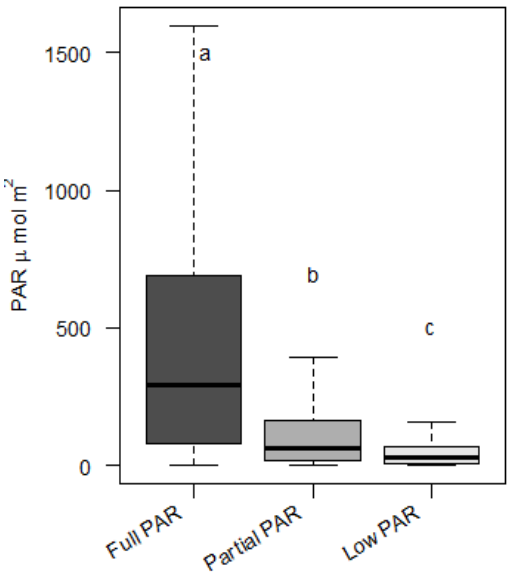

C

Figure 5 . Differences in growing season (a) daily average air temperature, (b) daily average soil temperature and (c)sunrise-sunset PAR receipts, between shading and warming treatments. The middle line in the boxes represents the median, and the box edges the 25th and 75th percentile. Error bars represent standard error. The box plots with different letters had significantly different values

2.4.2 Vegetation and soil properties

Above ground biomass was affected by the warming and shading treatments (*df* = 1, *F* = 33.4, *p* < 0.01) (*Table 1).* Across all cores warming by 2.4 ^{o}C (soil temperature) increased average above ground biomass by 10 % in high diversity cores and 13 % in low diversity cores (df = 1, F = 9.5, p < 0.05). For above ground biomass, there were no significant interactions between warming and diversity, and warming and PAR,

however there was an interaction between diversity and PAR, affecting above ground biomass. Specifically, in high diversity cores, reductions in PAR resulted in a smaller decrease in above ground biomass than in low diversity cores ($df = 1$, $F = 4.3$, $p < 0.05$). The vegetation harvest in July, revealed that the high diversity cores contained a greater proportion of biomass made up of forbs and legumes than the low diversity cores which were dominated by graminoids and a small proportion of legumes (Table 1). There were differences found in vegetation composition (percentage graminoids, forbs, legumes) in response to warming and shading (Table 1, Table 2). Diversity was found to affect the response of graminoid, forb and legume, % biomass composition to the warming and shading treatments. In the low diversity grassland cores, the warming and shading treatments did not affect the biomass of graminoids and legumes (Table 2). Specifically, graminoid, forb and legume, composition in low diversity plots was unaffected by temperature, whereas, in high diversity plots graminoid ($df = 1$, $F = 6.5$, $p < 0.05$ and legume ($df = 1$, $F = 5.7$, $p < 0.05$) composition increased in response to warming), and forb composition decreased in response to warming ($df = 1$, $F = 13.4$, $p < 0.05$). Further, in low diversity plots shading did not affect graminoid, forb or legume composition, whereas in high diversity cores shading decreased the amount of biomass made up from graminoids ($df = 1$, $F = 148$, $p < 0.01$) and legumes ($df = 1$, $F = 12$, $p < 0.05$) and increased forb composition ($df = 1$, $F = 5$, $p < 0.05$). However, it should be noted that legume composition was only affected by the 90 % reduction in PAR in the high diversity cores.

Above ground vegetation total carbon to total nitrogen ratio was found to be affected by diversity, warming and shading (Table 1, Table 2). Shading, reduced the C: N ratio, by 7 % with 74% of ambient PAR removed, and by 36 % with 90 % of ambient PAR removed ($df = 1$, $F = 316$, $p < 0.01$). Additionally, shading caused a greater relative decrease in leaf C: N in low diversity (50 % decrease) grasslands in comparison to high diversity grasslands (35 % decrease) ($df = 1$, $F = 6.2$, $p < 0.05$). There were no significant differences in soil total C, total N, pH, NH_4^+, PO_4^{3-}, NO_3^- or soil moisture between the warming and shading treatments, at the end of the experiment (*Table 3*).

Table 1. Effects of diversity, temperature and radiation treatments on vegetation composition and properties, mean ± standard deviation (n=5)

Diversity	Temperature	% Ambient PAR	Total biomass (kg) dwt per m^2	Graminoid (%)	Forb (%)	Legume (%)	Grass leaf C: N
Low	Ambient	100	1.79 ± 0.02	98.5 ± 1.9	0	1.5 ± 1.9	14 ± 0.9
Low	Warmed	100	2.07 ± 0.04	97.25 ± 0.5	0	2.8 ± 0.5	12 ± 1.6
Low	Ambient	26	1.56 ± 0.10	98.0 ± 0.8	0	2 ± 0.8	11 ± 1.5
Low	Warmed	26	1.78 ± 0.07	97.25 ±1.5	0	2.8 ± 1.5	10 ± 2.1
Low	Ambient	10	0.87 ± 0.03	97.5 ± 1.3	0	2.5 ± 1.3	7 ± 0.7
Low	Warmed	10	0.92 ± 0.08	98.75 ± 1.5	0	1.2 ± 1.5	5 ± 1.1
High	Ambient	100	1.56 ± 0.05	34.2 ± 2.2	54.6 ± 4.3	11.2 ± 4.0	20 ± 1.4
High	Warmed	100	1.70 ± 0.02	37.0 ± 2.3	47.4 ± 3.4	15.6 ± 1.8	18 ± 1.2
High	Ambient	26	1.33 ± 0. 14	29.6 ± 2.6	58.4 ± 4.1	12.0 ± 2.2	16 ± 2.6
High	Warmed	26	1.46 ± 0.07	31.6 ± 2.1	54.2 ± 4.0	14.2 ±2.7	15 ± 2.1
High	Ambient	10	0.73 ± 0.07	16.4 ± 2.7	74.6 ± 1.8	9.0 ± 1.6	15 ± 0.6
High	Warmed	10	0.87 ± 0.03	19.1 ± 1.5	69.2 ± 1.6	11.8 ± 1.9	13 ± 1.3

Table 2. Statistical analysis (ANOVA) for the effects of and interactions between, warming, shading and diversity on vegetation properties. The degrees of freedom value = 1 for each analysis.

Source of variation	*F*	*P*				
Above ground biomass						
Temperature	9.5	<0.05*				
Diversity	22.7	<0.05*				
PAR	265	<0.01**				
Temperature: Diversity	0.1	>0.05				
Temperature: PAR	1	>0.05				
Diversity: PAR	4.3	<0.05*				
Species Composition	*Graminoids*		*Forbs*		*Legumes*	
Temperature	2.4	>0.05	13.4	<0.05*	12.4	<0.05*
Diversity	13320.5	<0.001***	6095.7	<0.001***	350.7	<0.01**
PAR	144.9	<0.01**	124.4	<0.01**	5.7	<0.05*
Temperature: Diversity	6.5	<0.05*	13.4	<0.05*	5.7	<0.05*
Temperature: PAR	0.9	>0.05	0.2	>0.05	3.1	>0.05
Diversity: PAR	148.2	<0.01**	124.4	<0.01**	5.0	<0.05*

C: N

Temperature	24.2	<0.05*
Diversity	461.0	<0.01**
PAR	316.0	<0.01**
Temperature: Diversity	0.0	>0.05
Temperature: PAR	0.0	>0.05
Diversity: PAR	6.2	<0.05*

Table 3. Field soil properties for the high and low diversity grasslands, means ± standard deviation, n = 6

Property	High Diversity	Low Diversity
Soil C: N	11.28 ± 0.6	11.69 ± 0.9
pH	7.06 ± 0.1	7.82 ± 0.2
NO_3^- (mg/g of soil)	0.2 ± 0.1	0.8 ± 0.4
NH_4^+ (mg/g of soil)	1.0 ± 0.4	0.5 ± 0.1
PO_4^{3-} (mg/g of soil)	1.2 ± 0.3	2.0 ± 0.2
Soil moisture %	25 ± 2.5	28 ± 1.7

2.4.3 CO_2 fluxes

Plant diversity, PAR, and temperature were shown to effect ecosystem respiration, net ecosystem exchange. Diversity and PAR were found to effect photosynthesis. Further,

there were interactions between diversity*PAR for all CO_2 fluxes, and diversity*temperature for ecosystem respiration and photosynthesis (Table 4.) Temperature and PAR did not interact to affect ecosystem respiration, net ecosystem exchange or photosynthesis. The estimates of fixed effects from the model showed that for ecosystem respiration and photosynthesis, diversity was the most influential factor and PAR receipts the second most influential. Temperature was determined to have the largest effect on net ecosystem exchange, with diversity being the second most influential factor. For the interactions, diversity*PAR was the most influential factor for ecosystem respiration and net ecosystem exchange, whereas diversity*temperature was the most influential interaction for photosynthesis.

Table 4. Linear mixed effects model analysis, for the effects of and interactions between, temperature, PAR receipts and diversity on CO_2 fluxes

	F	***P***	***Estimates of Fixed Effects Coefficient***
Ecosystem Respiration			
Diversity	795.9	<0.001***	-85.59
Temperature	31.2	<0.01**	14.04
PAR	48.0	<0.01**	0.33
Diversity*Temperature	12.9	<0.05*	-1.62
Diversity*PAR	40.8	<0.01**	-0.07
PAR*Temperature	0.1	>0.05	0.01
Net Ecosystem Exchange			
Diversity	116.7	<0.01**	-116.20
Temperature	187.7	<0.01**	18.62
PAR	67.3	<0.01**	-0.28

Diversity*Temperature	0.9	>0.05	5.74
Diversity*PAR	40.0	<0.01**	-0.09
PAR*Temperature	1.7	>0.05	-0.01
Photosynthesis			
Diversity	268.2	<0.01**	-30.61
Temperature	0.3	>0.05	-5.42
PAR	66.5	<0.01**	-0.75
Diversity*Temperature	27.6	<0.01**	7.37
Diversity*PAR	2.51	<0.05*	-0.01
PAR*Temperature	0.0	>0.05	0.01

Reductions in PAR, had differential effects on ecosystem respiration in low versus high diversity grasslands, with shading resulting in a greater reduction in respiration in low diversity grassland *(Diversity*PAR, $p < 0.01$)* (Figure 6). A 74 % reduction in PAR resulted in a 34 % decrease in respiration in low diversity plots, compared to 31 % in high diversity plots. Further, when PAR was reduced by 90 %, respiration in low diversity plots decreased by 60 %, compared to a 54 % reduction in high diversity plots. Warming caused a greater relative increase in ecosystem respiration in low diversity compared to high diversity grasslands (*Diversity*Temperature, $p < 0.05$*). Warming, was associated with a 14 % increase in ecosystem respiration in low diversity plots, compared with 11 % in high diversity plots.

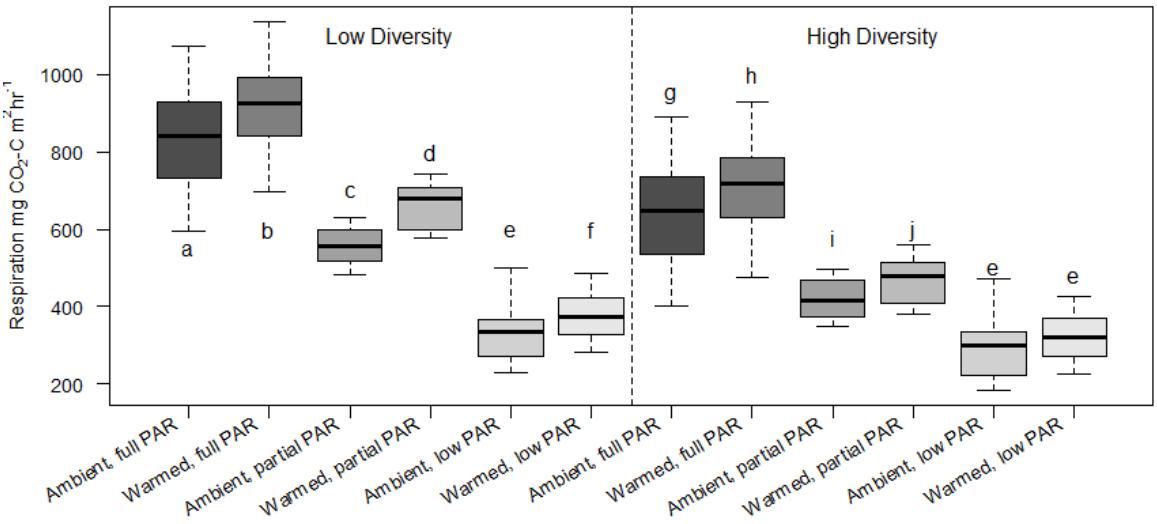

Figure 6. Differences in growing season average ecosystem respiration between the different diversity, warming and shading treatments. The middle line in the boxes represents the median, and the box edges the 25th and 75th percentile. Error bars represent standard error. Only data over the limit of detection are included. Bars with different letters had a significantly different flux.

Diversity modulated the effect of PAR on net ecosystem exchange, with shading resulting in a greater reduction in C uptake in low diversity plots compared to high diversity plots (Diversity*PAR, $p < 0.01$) (Figure 7). Specifically, a 74 % reduction in PAR reduced C uptake by 65 % in low diversity plots, compared to 57 % in high diversity plots. Under very low PAR conditions (90 % removal), there was no difference in NEE between the high and low diversity plots. Further, under these conditions, the grassland cores were at times found to be a small source of carbon to the atmosphere (+ve net ecosystem exchange values). However, the response of the low and high diversity plots to 90 % PAR reduction differed. Specifically, in low diversity plots net ecosystem exchange was reduced by 94 % and in high diversity plots by 100 %. In addition, warming was found to decrease C uptake (Temperature, $p < 0.01$).

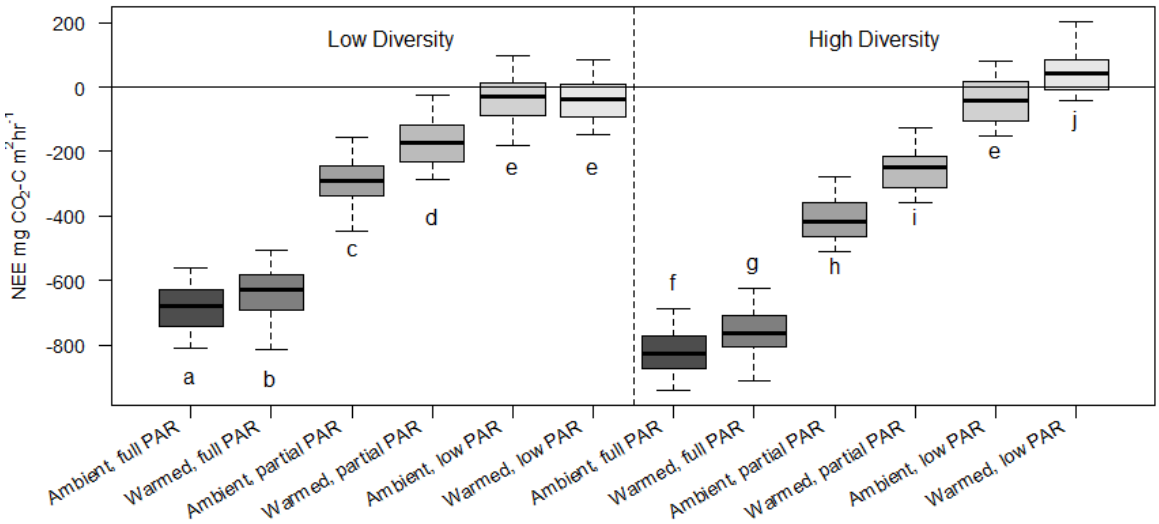

Figure 7. Differences in growing season average net ecosystem exchange between the different diversity, warming and shading treatments. The middle line in the boxes represents the median, and the box edges the 25th and 75th percentile. Error bars represent standard error. Only data over the limit of detection are included. Bars with different letters had a significantly different flux.

Reductions in PAR had differential effects on photosynthesis in low versus high diversity grasslands, with shading resulting in a greater reduction in photosynthesis in low diversity grasslands (Diversity*PAR, $p < 0.05$) (Figure 8). More specifically, a 74 % reduction in PAR, reduced photosynthesis by 45 % in low diversity plots, compared to 42 % in high diversity plots. Further, a 90 % reduction in PAR, reduced photosynthesis by 75 % in low diversity plots, compared to 79 % in high diversity plots. In addition, warming increased photosynthesis in low diversity plots, whereas warming decreased photosynthesis in high diversity plots (Diversity*Temperature, $p < 0.01$). Specifically, warming in low diversity plots increased photosynthesis by 3 %, whereas, in high diversity plots warming decreased photosynthesis by 6 %

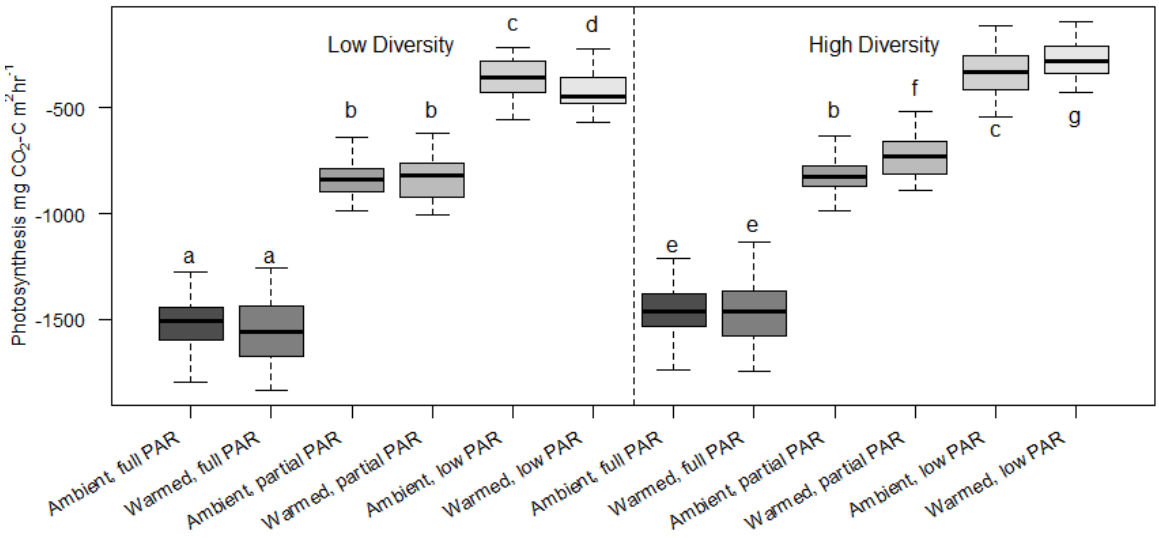

Figure 8. Differences in growing season average photosynthesis between the different diversity, warming and shading treatments. The middle line in the boxes represents the median, and the box edges the 25th and 75th percentile. Error bars represent standard error. The boxes represent the 25th percentile, the median, and the 75th percentile. Only data over the limit of detection are included. Bars with different letters had a significantly different flux.

2.5 Discussion

Our findings show that the effects of reduced PAR on grassland C cycling varied with grassland diversity. Further, temperature is an important control, with warming having differential effects on grassland C cycling in low and high diversity vegetation communities. Specifically, C cycling in high diversity grasslands was repeatedly found to be less sensitive to warming and shading than low diversity grasslands, supporting our hypothesis.

2.5.1 Effects of warming in high and low diversity grasslands

Diversity significantly interacted with temperature to affect ecosystem respiration and photosynthesis. Warming increased ecosystem respiration: in low diversity grasslands, the response of ecosystem respiration to warming was greater than in the low diversity grasslands. The results reveal that warming increases photosynthesis in low diversity grasslands, whereas, in high diversity grasslands, warming decreases photosynthesis.

The interaction between warming and diversity could be explained by the presence or absence, of plant functional groups in the low or high diversity communities (Fornara and Tilman, 2008; Fry *et al.*, 2013). In ecological stability theory, high diversity communities are inherently more resistant: remaining essentially unchanged when subjected to disturbances e.g. warming (Grimm and Wissel, 1997; Allison, 2004). High diversity grassland communities are more resistant/less sensitive to disturbances than low diversity communities, because in theory high diversity communities will possess a greater number of functional traits which will allow the community to adapt to the new conditions, whilst retaining the functions of the ecosystem (McCann, 2000; Allison, 2004).

The photosynthesis results, do not support the theory that high diversity grasslands will be more resistant to warming. Previously, warming has been found to decreases photosynthetic capacity in multispecies grasslands, however, this study there were no effects in low diversity grasslands with one or two species grasslands(De Boeck *et al.*, 2007). The negative effect of warming was attributed to midday stress (Mathur, Agrawal and Jajoo, 2014b): this effect may only be present in high diversity grasslands, as forbs are more susceptible to midday stress than graminoids dominated community in low diversity grasslands. In addition, the reduction in photosynthesis in high diversity grasslands may be due to the change in vegetation composition, with a loss of forbs and an increase in graminoids and legumes in response to warming. However, an analysis of the biomass data, reveals that warming increased above ground biomass, unlike in the photosynthesis analysis. In addition, although not significant to $p < 0.05$, there is an apparent trend between the effect of warming on above ground biomass in low and high diversity grasslands (Diversity*Temperature, $p = 0.06$), with warming increasing above ground biomass by 10 % in high diversity grasslands and by 13 % in low diversity grasslands, which would support the theory that the response of high diversity grasslands to warming is lower than the response of low diversity grasslands. The paradox between the photosynthesis and above ground biomass data suggests that there may have been methodological limitations, particularly as photosynthesis was calculated from the ecosystem respiration and net ecosystem exchange measurements, which were unable to be taken concurrently.

Another explanation for the differences in the warming response of ecosystem respiration and photosynthesis, in high and low diversity grassland communities might be due to differences in plant functional groups, which differ in the rate that they allocate photosynthetic carbon below ground and the carbon allocation response to warming also differs (Sevanto, Dickman and Way, 2015; Ward *et al.*, 2009). Consequently, differences in allocation are likely to affect the quality and quantity of root exudates to the soil, thereby affecting the composition and activity of microbial communities (Bardgett and Shine, 1999; Wardle *et al.*, 2011; Bardgett *et al.*, 2013). Also, the calculated differences in the photosynthetic response to warming in high and low diversity communities are likely to have altered carbon flux to roots and rates of root exudation, thereby further contributing to shifts in the composition and activity of microbial communities affecting net ecosystem exchange and ecosystem respiration.

The effect of experimental warming on ecosystem respiration and net ecosystem exchange is potentially explained as a direct effect of increased soil and air temperature on microbial activity. This view is supported by the positive correlation between soil temperature and carbon dioxide fluxes, with CO_2 fluxes consistently being greatest in the warmest plots and lowest in the coolest plots. This finding is consistent with other studies, where the positive effects warming on ecosystem respiration and net ecosystem exchange are reported (Piao *et al.*, 2008; Wang *et al.*, 2014; Suyker and Verma, 2001; Klumpp *et al.*, 2011; Conant, Haddix and Paustian, 2010). However, in this study, it is striking that the overall experimental warming (2.4 ^{o}C soil, 1.2 ^{o}C air) increased ecosystem respiration by 14 %, in low diversity grasslands and 11 % in high diversity grasslands, and increased net ecosystem exchange by 24 % and 16 % in low and high diversity grasslands, respectively. This suggests that in grasslands, ecosystem respiration and net ecosystem exchange are highly sensitive to rising temperatures, and that with climatic warming, grassland carbon stores may decrease (Davidson and Janssens, 2006).

2.5.2 Effects of changes in PAR in high and low diversity grasslands

Reductions in PAR typically reduce CO_2 fluxes, but ecosystem respiration, net ecosystem exchange and photosynthesis are generally more sensitive to changes in PAR recipients in low diversity grasslands than in high diversity grasslands. Net ecosystem exchange, ecosystem respiration and photosynthesis, displayed a greater reduction in

CO_2 fluxes in response to a 74 % reduction in PAR receipts in low diversity grasslands than in high. However, with a 90 % removal of ambient PAR, high diversity grasslands became more sensitive in their CO_2 fluxes than low diversity grasslands. This is likely due to the fact that high diversity grasslands could be more resistant to change, due to the more complex community possessing a higher number of functional traits than our low diversity community (Holling, 1973). The high diversity community in this study was highly species diverse as well as structurally diverse. This structural diversity could mean that the basal species, naturally present in grassland communities, could be adapted to/ more tolerant of lower light levels, due to shading from taller plants in the community (Roscher *et al.*, 2011; Grubb, 2015), as is observable to forest ecosystems (Bohn and Huth, 2017). Therefore, high diversity grasslands could maintain ecosystem functioning in the 74 % shading level better than the low diversity species. However, a 90 % reduction in shading for both high and low diversity communities meant that the grassland even at the height of the growing season was often a net source of CO_2. This is probably explained though the fact that the level of shading was too high for ecosystem functions to be maintained. Light availability has been found to control the strength of diversity effects on primary productivity, and how this response depends on plant functional types (Siebenkäs, Schumacher and Roscher, 2016). Specifically, when legumes were present the effect of shading on above ground biomass was less than when legumes were absent (Siebenkäs, Schumacher and Roscher, 2016).

The response of CO_2 fluxes to reductions in PAR receipts could be explained through changes in CO_2 uptake and release through biological mechanisms (Mercado *et al.*, 2009b; Wild, 2009). Specifically, PAR receipts have been found to alter rates of photosynthesis, the allocation of photosynthetic carbon (above/ below ground) and the quantity and quality of plant metabolites (Bahn *et al.*, 2013; Rutledge *et al.*, 2010). These changes subsequently affect roots exudates and litter inputs to the system, which are major controls on soil microbial community composition and activity, effecting ecosystem respiration and net ecosystem exchange. Further, the effect of changes in PAR receipts on root exudates and leaf litter quality is likely to depend on the vegetation community, with potential implications for carbon cycling in high and low diversity grasslands.

Shading was found to affect a variety of vegetation properties which could have resulted from or in the changes on CO_2 fluxes in high and low diversity grasslands. Specifically, the effects of changes in PAR receipts on the grassland communities resulted in changes to the quality and quantity of plant-C inputs, reflected in our above ground biomass and leaf C: N data. This research indicates that the change in above ground biomass is directly affected by PAR receipts. When PAR receipts were cut by 74 % and 90 %, total biomass dramatically decreased: this can be attributed to a reduced rate of photosynthesis (Raven and Karley, 2006). Changes in total above ground biomass directly and indirectly affect carbon dioxide fluxes. Directly, total above ground biomass positively correlates to photosynthetic capacity, controlled by chloroplast totals (Raven and Karley, 2006). Indirectly changes to the quantity of above biomass and production of root exudates will alter the inputs available to soil heterotrophs, and consequently ecosystem respiration (Jones, Nguyen and Finlay, 2009). The variation in above ground biomass response to shading between high and low diversity grasslands, is potentially explained through the fact that the high diversity community is more likely to contain shade tolerant species, with their growth unaffected by the shading treatments, leading to a smaller effect of shading on above ground biomass in the high diversity communities. A decrease in leaf C: N was associated with a decrease in PAR receipts, however, in low diversity grasslands this effect was stronger than in high, reflected by the respective 50 % and 35 % decreases. It is likely that the decrease in leaf C: N under the PV arrays is the result of changes in plant C and N allocation under the microclimatic changes imposed by solar PV arrays. Other studies have found that deep and sustained shading also resulted in a decrease in leaf C: N ratio (Ma *et al.*, 2015). Shading strongly reduces total C in shots but not roots (Bahn *et al.*, 2013). This decrease in C in shade is attributed to a decrease in photosynthetic capacity, and also results in a decrease in the C: N ratio. In addition to changes in C, leaf N has been found to increase under shaded conditions, due to the translocation of N to photosynthetic tissue (Ma *et al.*, 2015). This would again result in a decrease in leaf C: N. The differential effects of C: N in response to shading for high and low diversity grasslands in this study may be due to a species specific response, as smooth meadow for the high diversity cores and perennial rye grass for the low diversity cores was tested. Further, vegetation communities have been found to influence the shading stress resistance of grasses, with high diversity communities enhancing resistance to shading, through facilitative interspecific interactions between plants (Gustafsson and Bostrom, 2013).

Our results highlight that high diversity grasslands have the potential to be greater net sinks of C when subjected to warmed conditions, due to changes in productivity and decomposition processes. As the main terrestrial exchange of carbon from grasslands is as CO_2, quantifying net ecosystem exchange of CO_2 allows us to get a measure of the net ecosystem carbon balance in high and low diversity grasslands in response to warming (Cao and Woodward, 1998)

Our results are in line with an increasing body of evidence which suggests that plant diversity is a key driver of ecosystem CO_2 fluxes (Steinauer *et al.*, 2015; Strecker *et al.*, 2015; Van Haren *et al.*, 2010). However, measurements were taken during a single growing season, so implications for long-term changes in grassland biogeochemical cycling-the collective processes that determine carbon dioxide fluxes, should be inferred with caution. Diversity was found to affect ecosystem respiration to a greater extent than warming, which could result in changes to ecosystem respiration and net ecosystem exchange (Steinauer *et al.*, 2015). In addition plant diversity has also been found to be a control on primary productivity in grasslands (Tilman, Wedin and Knops, 1996). C storage a result of the balance between production and decomposition processes has been found to be 600 % greater in high diversity grasslands (Fornara and Tilman, 2008).

Our data demonstrates the effects of changes in temperature and PAR on ecosystem functions specifically, CO_2 fluxes, above ground biomass, C: N and vegetation community composition, and how these responses are determined by the diversity of the vegetation community. These findings are of increasing importance under the pressures of climate change resulting atmospheric warming, and the potential changes in PAR receipts through climate change effects on cloud cover, atmospheric particles and land-use change such as the installation of solar farms. Generally, our findings support our hypothesis that high diversity grasslands are more resistant to changes in temperature and PAR than low diversity grasslands.

2.6 Conclusion

The results presented here reveal for the first time, the differential effects of warming and shading on CO_2 fluxes, in high and low diversity grasslands. Generally, we found

that CO_2 fluxes are more resistant to warming and shading, in high diversity grasslands than low diversity grasslands. We recognise that the findings of this study are limited: our results are from one growing season, and the photosynthesis data apparently contradicts the above ground biomass data with regards to plant productivity, potentially revealing methodological deficiencies in sampling and calculation, although we did not measure below ground biomass. As such, a cautious interpretation of these results in the effects of warming and shading on carbon cycling in high and low diversity grasslands is required. Further work should focus on identifying the response of plant functional types, key species or interactions between species, to warming and shading, and increasing spatial and temporal scales. However, the findings presented in this paper extend our understanding of controls on grassland carbon cycling and how changes to temperature and PAR recipients due to land-use change and climate change might affect ecosystem function, with the potential to create feedbacks.

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3 Solar farms reduce the temperature sensitivity of leaf litter decomposition and alter microbial community composition

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Data referred to in this chapter can be viewed in full in the appendices presented at the end of this thesis.

3.1 Abstract

Due to the necessity to develop sustainable energy sources solar farms are being deployed at an unprecedented rate, however, the effects on hosting ecosystems are unclear. Solar farms have the potential to affect decomposition processes in UK grasslands and subsequently carbon stores, through microclimatic changes such as air and soil temperature, soil moisture and PAR receipts. It is unclear how the changes in microclimate under PV arrays will affect decomposition, but in order to elucidate the true impact of PV arrays on hosting ecosystems, this needs to be resolved. Our study demonstrates for the first time that the presence of PV arrays may suppresses the decomposition of leaf litter, mainly due to temperature differences, and that the presence of PV arrays reduces the temperature sensitivity of leaf litter decomposition. We found that mass loss was lowest in the areas directly under the PV arrays, intermediate in the gaps between two rows and greatest in the areas away from the PV arrays. This study also reveals that the soil fungal community was reduced under the PV arrays. In addition, using open top chambers to create a warming treatment, we show that the effects of warming on litter decomposition under the PV arrays were reduced by approximately 40 %. The relationship between leaf litter decomposition, microclimatic variables and soil microbial community suggests that the presence of PV arrays may drive changes in in grassland C cycling directly through changes to the microclimate and indirectly by altering the composition of the soil microbial

community. The reduced decomposition rates under the PV arrays have the potential to increase grassland C stores, however, further research will be needed to balance this with any changes in productivity experienced on grasslands managed as solar farms.

3.2 Introduction

In order to mitigate against the effects of climate change and meet increasing global energy demands, terrestrial ecosystems are increasingly hosting renewable energy technologies, namely bioenergy crops, wind farms and solar farms (Armstrong *et al.*, 2014; Hernandez *et al.*, 2013; Ostle *et al.*, 2009). Solar energy is the fastest growing energy sector, with 73 GW installed in 2016, bringing the total installed globally to 310 GW, a yearly increase of 31 % (Focus, 2016) This trend is expected to continue and by 2050 solar PV could be the dominant renewable power source globally (IEA, 2014), as policies such as the US's Clean Power Plan and China's Golden Sun, have put solar energy at the forefront of future energy infrastructure (UNEP, 2015; EPA, 2015). A substantial proportion of PV is installed as ground-mounted solar farms (EPIA, 2016). Due to the relatively low energy density of solar farms (Mackay, 2013), solar PV represents a significant land use change (DECC, 2014). In the UK up to April 2016, the total operational capacity of large-scale solar PV deployment (predominantly ground mounted systems) had reached 10,967 MW (DBEIS, 2016), covering approximately 222 km^2 of land (Burke, 2015). ,

In Europe, ground-mounted solar farms are commonly located on grasslands (Armstrong, Ostle and Whitaker, 2016). Grasslands are important carbon (C) stores, with the C stored in grassland soils contributing to more than 32% of the total C stored in British soils (Ostle et al., 2009)(Ostle *et al.*, 2009) and thus any land use or management changes could lead to significant impacts on decomposition processes (Guo and Gifford, 2002)(Guo and Gifford, 2002), soil moisture, and solar radiation receipts under the panels(Armstrong et al., 2016)(Armstrong, Ostle and Whitaker, 2016)soil moisture and radiation receipts imposed by solar farms have the potential to be of a magnitude that would significantly affect decomposition processes. Therefore, it is increasingly important to elucidate the impact of PV arrays on decomposition processes and the wider grassland C cycle and C storage potential.

The flow of C through an ecosystem is partly controlled through the decomposition of leaf litter (Bardgett, 2005; Wardle, 2002). The decomposition of leaf litter, including

rhizodeposits contributes to approximately 70 % of the total C flux (77 Pg C) from soils globally (Raich and Schlesinger, 1992). The main factors controlling decomposition are climate, litter quality and soil organisms (Butenschoen, Scheu and Eisenhauer, 2011; Prescott, 2010). Whilst the impacts of temperature and soil moisture on decomposition are relatively well understood (Ise and Moorcroft, 2006), relatively little is known about the effects of solar radiation and its interactions with temperature and soil moisture in temperate grassland decomposition (Rutledge *et al.*, 2010). However, changes in solar radiation receipts have been shown to affect decomposition either through changes in litter quality, photodegradation of litter, and direct and indirect effects on microbial communities.

Decomposition in temperate ecosystems is dominated by soil microbes (Portillo-Estrada *et al.*, 2015) . Soil microbial activity and community composition is predominantly driven by litter quality and quantity, soil temperature and moisture availability (Cornwell *et al.*, 2008; Zhang and Wang, 2015). With regards to climatic variables, microbial activity and subsequently decomposition rates, are generally greater in warmer wetter conditions (Wieder, Cleveland and Townsend, 2009). Bacteria and fungi in soils degrade plant residues differently and have different roles in nutrient cycling (Churchland and Grayston, 2014). Fungi are generally more efficient at assimilating and storing nutrients than bacteria (Manzoni *et al.*, 2012). The ratio of fungi to bacteria is a good indicator of environmental change in the soil (Malik *et al.*, 2016). The presence of PV arrays has the potential to affect microbial community structure, through factors which have direct and indirect effects, and the interactions between these variables. Soil moisture has the potential to be affected by the presence of PV arrays (Armstrong *et al.*, 2014). Soil moisture has been found to be a major factor influencing microbial community structure i.e. bacterial: fungal, gram +ve: gram –ve bacteria (Drenovsky *et al.*, 2010). Low soil moisture levels are known to limit microbial activity and alter community structure (Castro *et al.*, 2010). The presence of PV arrays which promote microclimatic changes therefore has the potential to affect microbial activity and community composition. Further, the changes in microclimate may also promote changes in vegetation growth and community composition, which in turn may alter litter quality, which has been suggested to be the most important control on leaf litter decomposition (Cornwell *et al.*, 2008).

The aim of this research was to investigate the effect of a typical solar farm on grassland decomposition due to the microclimatic effects of PV arrays. We hypothesised that:-

1. *The presence of PV arrays will reduce litter decomposition rates and sensitivity in response to warming.*
2. *The variation in litter decomposition will be accounted for to a greater extent by the change in abiotic variables as opposed to biotic variables.*

3.3 Methods

3.3.1 Study site and experimental design

This study was conducted at Westmill Solar Farm, UK (51° 37' 03" N 01° 38' 45" W, http://www.westmillsolar.coop/). Westmill Solar Farm has a 5 MW operational capacity and was installed in 2011. The site since has been managed as a species rich grassland to promote biodiversity, with winter grazing and yearly mowing to prevent shading of the PV arrays. The plant community included five forb species (Leucanthemum vulgar, Plantago lanceolata, Achillea millefolium, Ranunculus acris, Ranunculus repens), four legume species (Trifolium pratense, Trifolium repens, Lotus corniculatus, Onobrychis viciifloia) and four grass species (Phleum pratense, Poa spp. (pratensis or annua), Brachypodium sylvaticum, Festuca rubra).

Distinct treatment areas were identified: the areas under the PV arrays (under), in the gaps between the PV arrays (gap) and control areas away from the influence of the PV arrays, but still in the enclosed site area (control). In each of the three treatment areas four sampling plots were randomly selected and sheep exclusion fencing was erected. At each plot, warmed conditions were simulated using open-top chambers (0.4 x 0.4 x 0.15 m) constructed from polyethylene, and a open area of equal size desingnated for litter bags in ambient temperature conditions. At each of the ambient and warmed treatments in each plot, decomposition, soil temperature and moisture, PAR, total C, total N, NH_4^+, NO_3^- and PO_4^{3-} were measured.

3.3.2 Microclimate

At each of the ambient and warmed treatments in each plot soil temperature was measured every 2 hours from 31/03/15 04:00:00 PM to 29/03/16 10:00:00 PM using (Tempcon UK: HOBO 8K Pendant® Temperature/Alarm (Waterproof) Data Logger) placed at 5 cm below the soil surface. At one replicate for each treatment PAR was

sampled every minute and the hourly average recorded (Tempcon, UK: HOBO Micro Station with S-LIA-M003).

3.3.3 Litter decomposition

Litter decomposition was determined by measuring the mass loss of litter bags as detailed in (Ward *et al.*, 2015). Recently senesced *poa annua* litter was collected from an area of undisturbed grassland at Hazelrigg Research Station, Lancaster, UK, in October 2014 and oven dried at 60 ˚C. The litter was then stored in airtight containers at 4 ˚C until March 2015. The litter was placed in litter bags at a mass to area ratio of (0.5 g: 25 cm^2). The litter bags were constructed from a 1 mm PVC coated fibre glass mesh. To ensure accurate mass loss measurements litter bags were transported to the field in sealed bags, with the bags weighed before and after to ascertain how much if any litter was lost in transit. On the 31/3/2015 four litter bags were placed at each of the ambient and warmed treatments, in each plot at the soil surface under any existing litter layer.

Litter bags were recovered after 3 months (26/6/2015), 6 months (29/9/15), 9 months (7/12/15) and 12 months (29/3/16,) and were stored at 4 ˚C until processed. For the purposes of data analysis (seasonal average temperatures), in this experiment we defined the seasons as the period between the sampling dates i.e. 31/3/15 to 26/6/2015 (spring), 27/6/15- 29/9/15 (summer), 30/9/15- 7/12/15 (autumn) and 8/12/15- 29/3/16 (winter).

Large roots and soil were removed by hand from the outside of the recovered litter bags, which were then carefully rinsed with deionised water over a 1 mm sieve. Litter bags were then cut open and any remaining roots and soil were removed using tweezers. The cleaned litter was dried at 60 ˚C to a constant weight (Ward *et al.*, 2015).

The mass loss data was then used to calculate the decomposition rate (k) of the litter bags under the different location and warming treatments, by using the formula:-

$$X_{(t)} = e^{-kt}$$

where $X_{(t)}$ is the proportion of the original litter remaining and t is the time in days since the litter bags were installed.

3.3.4 Soil properties

For each litter bag, a soil corresponding sample was taken from directly beneath where the litter bag had been, and analysed for soil moisture, pH, total C, total N, C:N, NH_4^+, NO_3^- and PO_4^{3-}. Soils were homogenised by hand and stones and roots were removed. 30 mg subsamples of oven dried and ground (ball mill) soil, were used to determine total C and total N using a LECO Truspec CN Analyser (LECO, USA): the soil C:N ratio was subsequently calculated and used in the statistical analysis (Carter, 2007). Soils were analysed for extractable NH_4^+ and NO_3^- using potassium chloride (KCl) extractions (Ward *et al.*, 2007). Fresh soil samples (5 g) were mixed with 1 mol/L KCl on an orbital shaker for 1 hour (model KS501 digital, IKA, Werke, Germany) and then filtered using Whatman #1 filter paper. Concentrations of NH_4^+ and NO_3^- in the filtrate were determined by colorimetric technique (Ross, 1992), on a continuous-flow stream autoanalyzer (Autoanalyzer 3, Bran Luebbe, Norderstedt, Germany). A 0.5 M sodium bicarbonate ($NaHCO_3$) solution at a pH of 8.5, was used to extract PO_4^{3-} from fresh soil (Rowell, 1994). The procedure, as described for the NH_4^+ and NO_3^- extraction and analysis, was used to determine PO_4^{3-} concentrations. Soil moisture content was determined by drying the sub-samples at 105^{o}C to a constant mass. pH was determined through the use of a pH probe and meter (soil: H_2O, 1:2.5 w:v) (Rowell, 1994).

Effects of warming and shading on bacterial and fungal community composition were determined using phospholipid fatty acid (PLFA) analysis according to (Bardgett, Hobbs and Frostegård, 1996), on soil samples taken from directly under the litter bag in March 2016. Phospholipids were extracted using the Bligh and Dyer method, from 2.0 g soil fresh weight and analysed using a gas chromatograph. Gram-positive bacteria were identified by the terminal and mid-chain branched fatty acids (15:0i, 15:0a, 16:0i, 17:0i, 17:0a), and cyclopropyl saturated and monosaturated fatty acids (16:1ω7, 7,cy-17:0, 18:1ω7, 7,8cy-19:0) were considered indicative of gram-negative bacteria (Rinnan and Baath, 2009). The fatty acids 18:2ω6,9 and 18:1ω9 were considered to represent saprotrophic and ectomycorrhizal fungi (Kaiser *et al.*, 2010; De Deyn *et al.*, 2009). Total PLFA concentration was calculated from all identified PLFAs (15:0, 14:0, 16:1, 16:1ω5, 16:0, 17:1ω8, 7Me-17:0, br17:0, br18:0, 18:1ω5, 18:0, 19:1; plus those listed above)(Whitaker *et al.*, 2014). The ratios of fungal to bacterial (F:B) PLFA and gram-positive to gram-negative (GP:GN) PLFA were taken to represent the relative abundance metrics of these groups (Whitaker *et al.*, 2014).

3.3.5 Data analysis

All data were checked for normality and log transformed where necessary. The effects and interactions of experimental warming and location on litter decomposition, soil properties and microbial community composition were analysed using linear-mixed effects models in the R studio statistical program using the nlme package (RStudio Team 2015).

A multiple linear regression was conducted to analyse the relative contributions to changes in decomposition rate against independent variables selected (soil temperature, soil moisture, PAR, fungal PLFAs, total PLFAs and bacterial: fungal PLFAs) and all 2 way interactions, based on the significant results of linear-mixed effects models. The independent variable data were centred on the mean and standardised using the standard deviation. To create the interaction variables, the centred independent variables were multiplied, and then the data was standardised using the standard deviation of the product term The use of the standardised variables meant that the regression coefficients could be used to determine relative effects size (Cohen, 1977).

3.4 Results

In the following section, we outline the variation in leaf litter decomposition and associated abiotic and biotic variables in the solar farm.

3.4.1 Abiotic Variables

There were differences in soil temperature for the location and warming treatment combinations (Table 6). On average experimental warming increased soil temperature by 0.3 °C ($df = 1$, $F = 19.6$, $p < 0.05$), whereas the presence of PV arrays reduced soil temperature by 1.5 °C ($df = 2$, $F = 488$, $p < 0.001$)(Table 6). Furthermore, warming had larger positive effects on soil temperature in control > gap > under treatments ($df = 2$, $F = 207$, $p < 0.01$(Table 6). Specifically, in control areas experimental warming increased soil temperature by 0.4 °C, in comparison to warming of 0.3 °C for gap areas and 0.1 °C for under areas.

Seasonal variations in soil temperatures across the warming and location treatments were observed (Table 5). Soil temperatures in the spring were 15 % lower under the PV arrays than in the control areas and in the summer, temperatures were 17 % lower (Table 5). During the autumn, seasonal average soil temperature for the gap and under treatments did not differ, whereas, soil temperatures in the control areas were 6 %

warmer (Table 5). Interestingly, during the winter we found that seasonal average soil temperatures were cooler by 6 % in the gaps areas in comparison to the areas under the PV arrays and the control areas (Table 5).

Table 5. Differences in yearly and seasonal average temperatures and standard deviations between the location (control, gap and areas under the PV arrays on a solar energy farm) and warming treatment combinations. Different letters denote temperatures which differ significantly from each other.

Location-solar farm	Temperature	Yearly average temperature	Spring average temperature	Summer average temperature	Autumn average temperature	Winter Average Temperature
Control	Ambient	11.9 ± 4.7	11.9 ± 2.7[a]	17.3 ± 2.6[a]	11.7 ± 2.8[a]	6.7 ± 2.9[a]
Control	Warmed	12.3 ± 5.4	12.8 ± 4.1[b]	18.2 ± 5.4[b]	11.7 ± 2.5[a]	6.8 ± 2.7[a]
Gap	Ambient	11.6 ± 4.8	11.6 ± 2.5[c]	17.3 ± 2.4[c]	11.1 ± 2.6[b]	6.3 ± 3.3[b]
Gap	Warmed	11.9 ± 5.2	12.5 ± 3.3[d]	17.8 ± 3.4[b]	11.2 ± 2.3[b]	6.5 ± 2.5[c]
Under	Ambient	10.4 ± 3.6	10.1 ± 2.1[e]	14.5 ± 1.7[d]	10.7 ± 2.0[c]	6.6 ± 2.0[d]
Under	Warmed	10.9 ± 3.7	10.5 ± 2.0[f]	15.1 ± 1.8[e]	11.1 ± 1.9[b]	7.0 ± 1.9[a]

Table 6. Linear mixed effects model analysis for the effects of and interactions between warming and solar farm location on soil temperature.

	Df	F	P

Location	2	488.0	< 0.001
Warming	1	19.6	< 0.05
Location*Warming	2	207.4	< 0.01

Under the PV arrays daytime PAR receipts over the course of the year were 92% lower than in the control areas. PAR receipts in the gap areas were similar to the control areas (Figure 9).

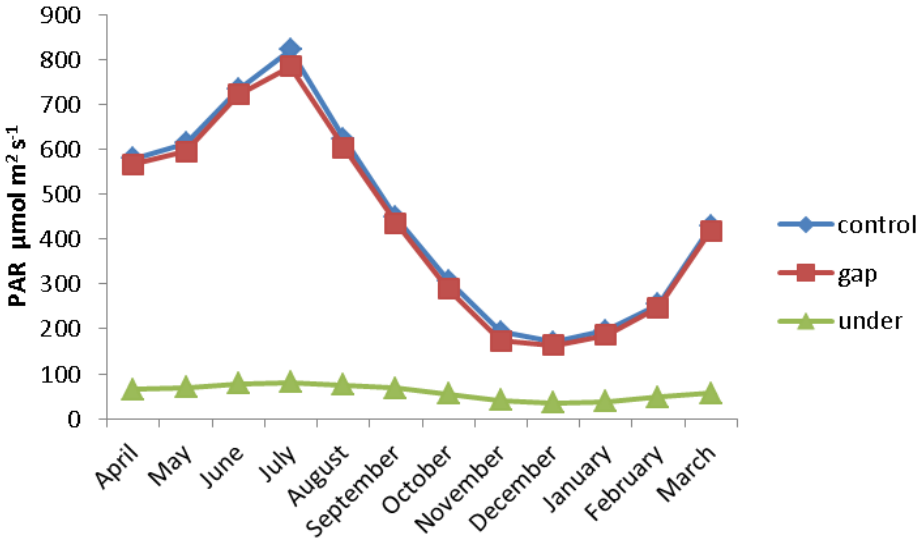


Figure 9. Average monthly daytime PAR receipts (µmol m^2 $s^{-1)}$ at control, gap and under treatments

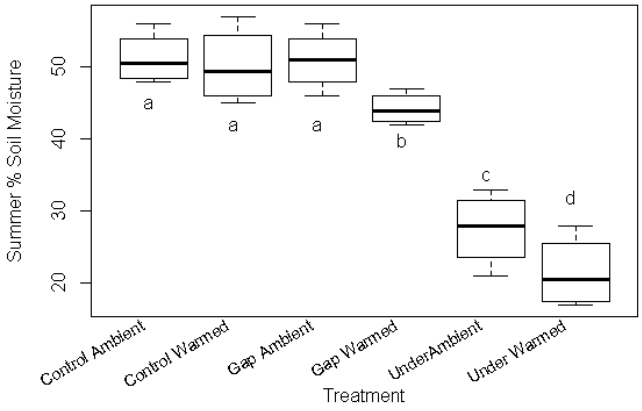
Soil moisture was affected by the presence of PV arrays throughout the year, and the warming treatment in spring, summer and winter (Table 7) (Figure 10). There was no interaction between warming and location on soil moisture at any time during the year on soil moisture (Table 7). In the spring and summer under the PV arrays, soil moisture was lower than in the control and gap areas (Table 7) (Figure 10). In the spring the presence of PV arrays reduced soil moisture by 48 % (df = 2, F = 84, p< 0.05) and in the summer the difference increased to 53 % (df = 2, F = 85, p< 0.05) (Table 7). In autumn and winter, the effect of PV arrays on soil moisture was reversed with higher soil moisture under the PV arrays than in the control and gaps areas (df = 2, F = 18, p < 0.05) (df = 2, F = 34, p < 0.05)(Table 7) (Figure 10). The warming treatment reduced soil moisture in the spring, summer and winter, under the PV arrays, and in the spring,

summer and autumn in the gap areas(Table 7) (Figure 10). The warming treatment did not affect soil moisture in the control areas at any of the seasonal sampling points (Figure 10).

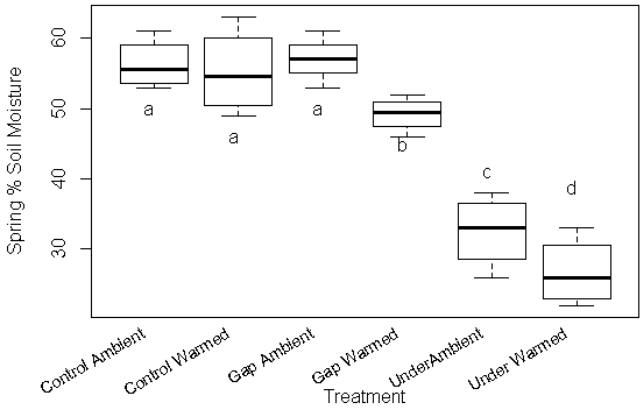
Soil pH, total C, total N, C:N, NH_4^+, NO_3^- and PO_4^{3-} were not affected by the presence of PV arrays or warming treatments.

Table 7. Linear model analysis for the effects and interactions of warming and solar farm location on % soil moisture by season.

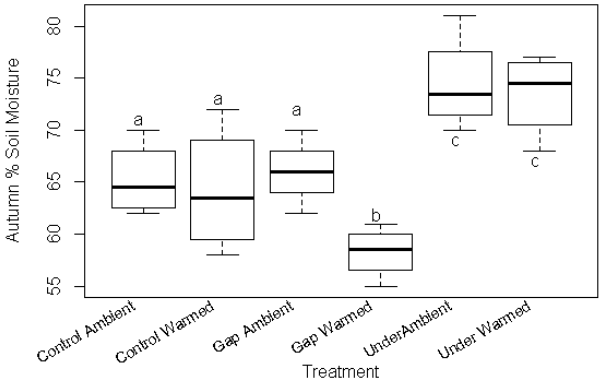
	Df	*F*	*P*
Spring			
Location	2	84.1	< 0.05
Warming	1	7.1	< 0.05
Location*Warming	2	1.2	> 0.05
Summer			
Location	2	84.5	< 0.05
Warming	1	6.4	< 0.05
Location*Warming	2	1.0	> 0.05
Autumn			
Location	2	17.7	< 0.05
Warming	1	3.6	> 0.05
Location*Warming	2	1.7	> 0.05
Winter			
Location	2	33.9	< 0.05
Warming	1	4.2	< 0.05
Location*Warming	2	0.1	> 0.05


Summer % Soil Moisture
20
30
40
50
a
a
a
b
c
d
Control Ambient
Control Warmed
Gap Ambient
Gap Warmed
UnderAmbient
Under Warmed
Treatment

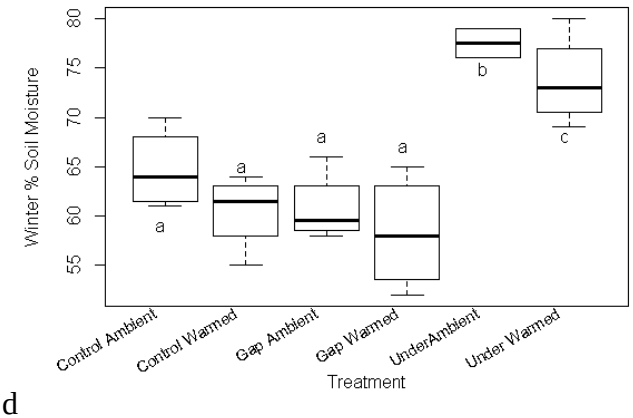
a


Spring % Soil Moisture
30
40
50
60
a
a
a
b
c
d
Control Ambient
Control Warmed
Gap Ambient
Gap Warmed
UnderAmbient
Under Warmed
Treatment

b


Autumn % Soil Moisture
55
60
65
70
75
80
a
a
a
b
c
c
Control Ambient
Control Warmed
Gap Ambient
Gap Warmed
UnderAmbient
Under Warmed
Treatment

c



d

*Figure 10. Differences in (a) spring, (b) summer, (c) autumn and (d) winter % soil moisture, between the location (control, gap and areas under the PV arrays on a solar energy farm) and warming treatment combinations. The top and bottom lines of the rectangle are the 25th and 75th quartiles and the midline represents the median. The error bars represent the 25th and 75th quartile -/+ 1.5*interquartile range, and circles outlying data. The box plots with different letters are significantly different from one another.*

3.4.2 Biotic Variables

Fungal abundance decreased under the PV arrays (df = 2, F = 6, $p < 0.05$), however, fungal abundance was unaffected by experimental warming (Table 8) (Figure 11). The change in fungal abundance under the PV arrays also affected total PLFAs (df = 2, F = 9, $p < 0.05$) and the fungal: bacterial ratio (df = 2, F = 12, $p < 0.05$) (Table 8) (Figure 11). Bacterial abundance was unaffected by the presence of PV arrays or experimental warming or the interactions (Table 8) (Figure 11). Gram +ve, gram –ve and the ratio of gram +ve: gram –ve PLFAs was unaffected by the warming treatment and the presence of PV arrays (Table 8) (Figure 11).

Table 8. Linear model analysis for the effects and interactions of warming and solar farm location on soil microbial community composition.

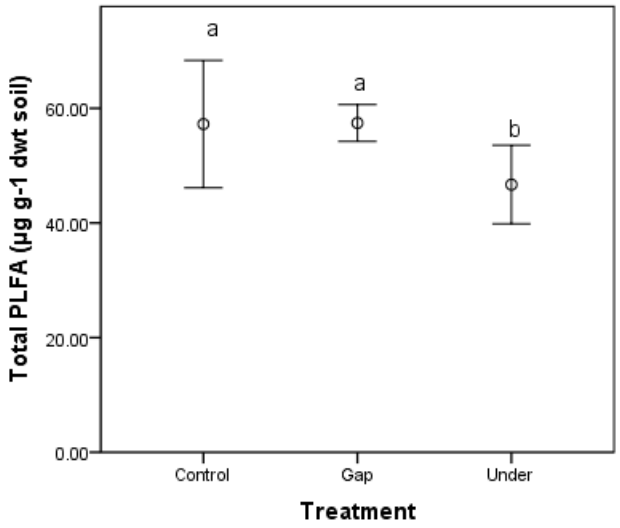
	df	*F*	*P*
Total PLFA			
Location	2	8.6	< 0.05

Warming	1	0.5	> 0.05
Location*Warming	2	1.5	> 0.05
Total Fungal			
Location	2	5.7	< 0.05
Warming	1	0.7	> 0.05
Location*Warming	2	1.4	> 0.05
Total Bacterial			
Location	2	1.1	> 0.05
Warming	1	0.5	> 0.05
Location*Warming	2	1.8	> 0.05
Fungal: Bacterial			
Location	2	12.1	< 0.05
Warming	1	0.3	> 0.05
Location*Warming	2	0.7	> 0.05
Total Gram +ve			
Location	2	1.7	> 0.05
Warming	1	0.2	> 0.05
Location*Warming	2	1.3	> 0.05
Total Gram -ve			
Location	2	0.1	> 0.05
Warming	1	0.8	> 0.05
Location*Warming	2	1.8	> 0.05

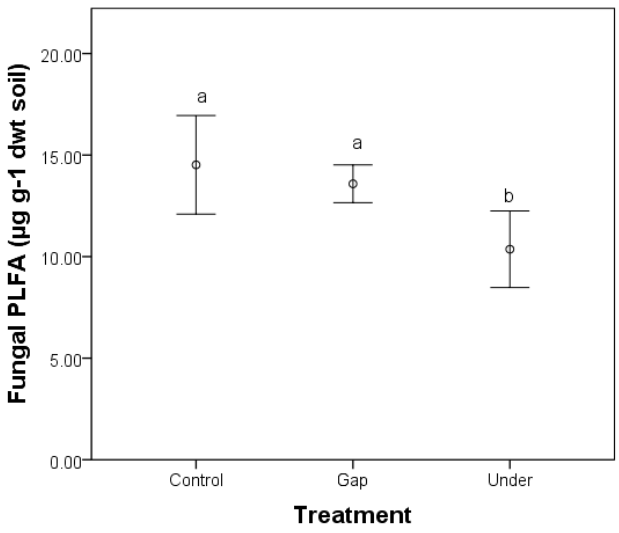
Gram +ve: Gram -ve

Location	2	0.5	> 0.05
Warming	1	0.3	> 0.05
Location*Warming	2	1.2	> 0.05

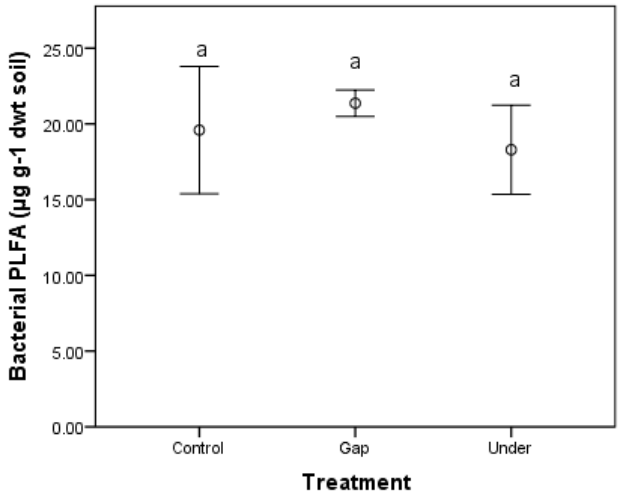
A



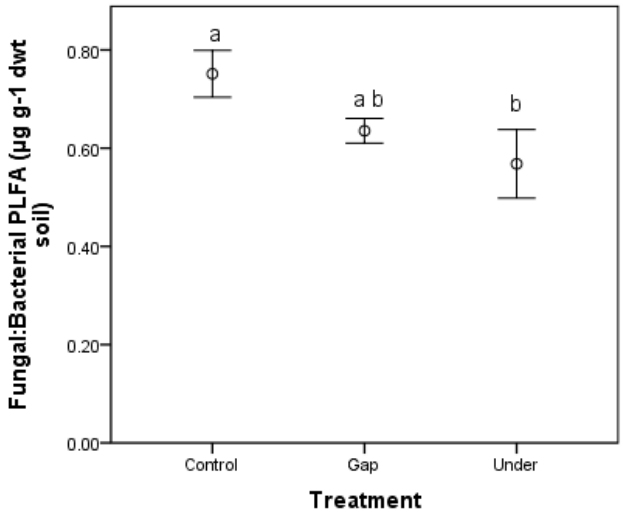
B



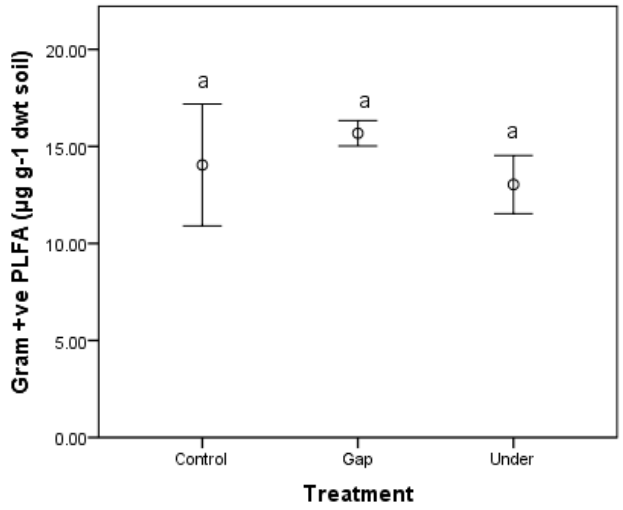
C


Bacterial PLFA (µg g-1 dwt soil)
25.00
20.00
15.00
10.00
5.00
0.00
a
a
a
Control
Gap
Under
Treatment

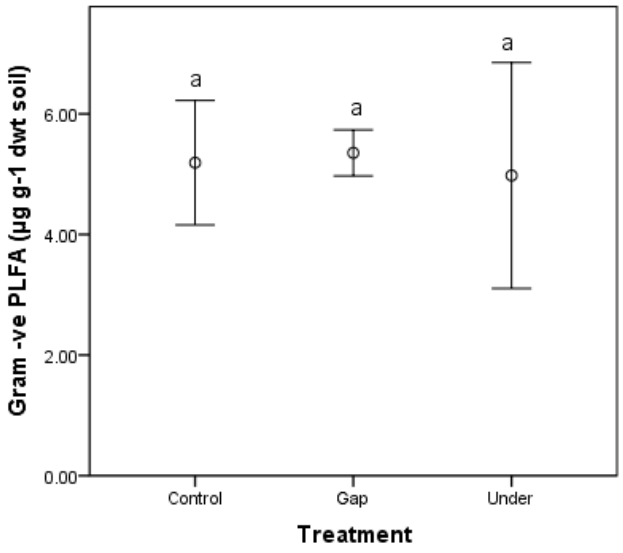
D


Fungal:Bacterial PLFA (µg g-1 dwt soil)
0.80
0.60
0.40
0.20
0.00
a
a b
b
Control
Gap
Under
Treatment

E


Gram +ve PLFA (µg g-1 dwt soil)
20.00
15.00
10.00
5.00
0.00
a
a
a
Control
Gap
Under
Treatment

F



G

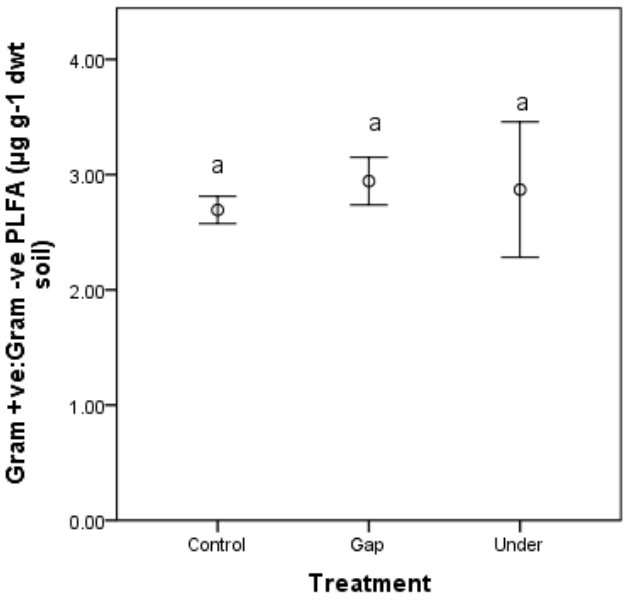


Figure 11. Differences in (A) total PLFAs, (B) total fungal, (C) total bacterial, (D) fungal: bacterial, (E) total gram +ve, (F) total gram –ve, (G) gram +ve: gram -ve for the location treatment (control, gap and areas under the PV arrays on a solar energy farm) The error bars represent the standard error. The points with different letters are significantly different from one another.

3.4.3 Decomposition

Rates of litter decomposition exhibited differences based on warming and location (Table 9), further there were interactions between location*warming for litter decomposition.

At the end of the in-field incubation period (365 days), the mean mass loss percentage in ambient plots was 63 ± 0.6 % for control treatments, 50 ± 1.3 % for gap treatments and 40 ± 1.1 % for under treatments (Table 11). For the warmed treatment combinations, the values were 73 ± 1.0 %, 59 ± 0.6 % and 44 ± 0.6 % for control, gap and under treatments respectively (Table 11). The lowest decomposition rate was measured in the ambient under treatments, here the yearly average temperature was the lowest, and the effect of shading from the solar panels is greatest (Figure 12). The highest rate of decomposition was observed in the warmed control treatments, here the yearly average temperature was the highest, and there were no solar panels affecting radiation receipts. Initially, the differences in mass loss between the control, gap and under treatments were small, but as time progressed the difference between the control and, gap and under treatments became greater.

Mass loss was affected by location in relation to solar panels i.e. control (away), gap (between two rows of panels), and under (the ground directly underneath the panels) (Table 10): percentage mass loss was 24 % greater in control areas than gap areas, and 31 % greater in gap areas than under areas (Table 11). Experimental warming was shown to affect leaf litter mass loss (Table 10). The overall percentage mass loss was 15 % greater in warmed plots in comparison to ambient plots (Table 11).

Table 9. Linear mixed effects model analysis for the effects of and interactions between warming and solar farm location on litter decomposition rates (k).

	Df	F	P
Location	2	374.5	< 0.05
Warming	1	127.7	< 0.05
Location*Warming	2	17.6	< 0.05

Table 10. Linear mixed effects model analysis for the effects of and interactions between, warming and solar farm location on litter mass loss %, by sampling date.

Source of variation	*Df*	*F*	*P*
3months			
Location	2	39	< 0.05
Warming	1	33	< 0.05
*Location*Warming*	2	2	> 0.05
6 months			
Location	2	400	< 0.001
Warming	1	111	< 0.01
*Location*Warming*	2	5	< 0.05
9 months			
Location	2	434	< 0.001
Warming	1	156	< 0.01
*Location*Warming*	2	5	< 0.05
12 months			
Location	2	320	< 0.01
Warming	1	96	< 0.01
*Location*Warming*	2	6	< 0.05

Table 11. Effects of experimental warming on litter mass loss (mean ± standard deviation) at quarterly sampling points throughout a year, at Westmill Solar Farm, Oxfordshire UK

	Litter mass loss %	
Solar Farm Location	Ambient	Warmed
3 months		
Control	19 ± 0.5	23 ± 1.0
Gap	16 ± 0.6	20 ± 0.9
Under	14 ± 0.4	16 ± 0.6
6 months		
Control	45 ± 1.0	52 ± 1.4
Gap	35 ± 0.6	41 ± 0.5
Under	30 ± 1.1	33 ± 1.1
9 months		
Control	58 ± 1.1	67 ± 0.9
Gap	46 ± 0.6	54 ± 1.0
Under	38 ± 1.1	43 ± 1.4
12 moths		
Control	63 ± 0.6	73 ± 1.0

Gap	50 ± 1.3	59 ± 0.6
Under	40 ± 1.1	44 ± 0.6

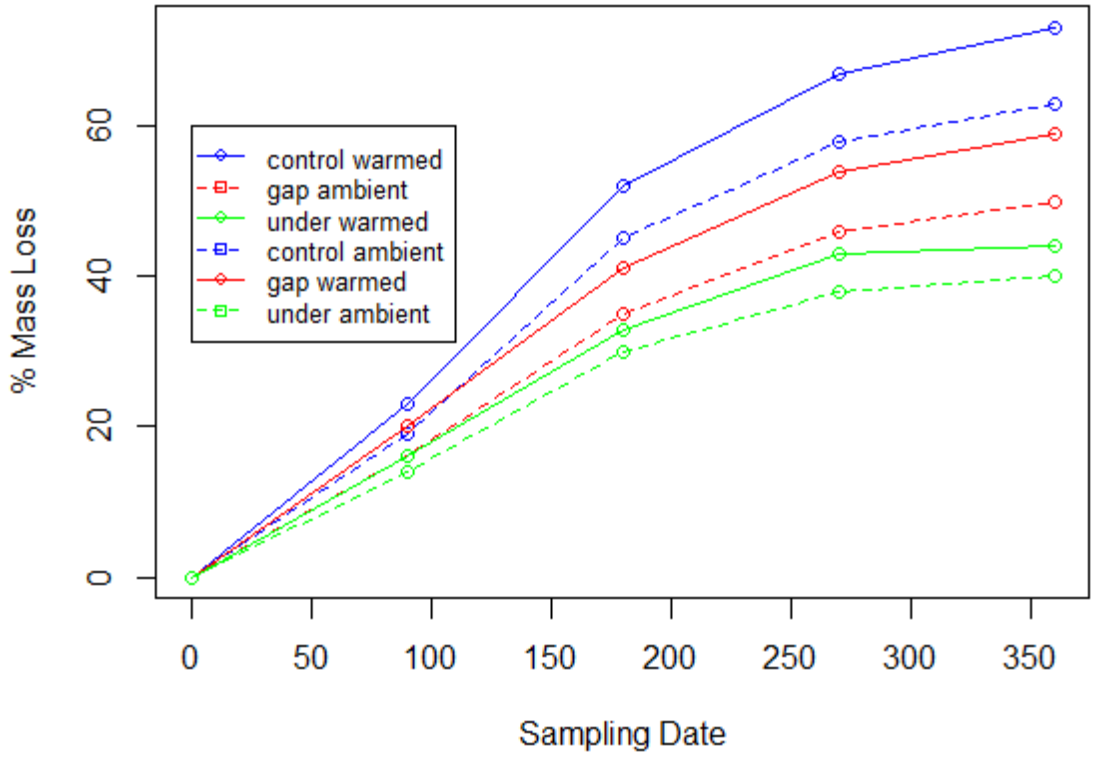


Figure 12. Differences in percentage mass loss of leaf litter, between the location (control, gap and areas under the PV arrays on a solar energy farm) and warming treatment combinations, n = 96.

Warming has differential effects on decomposition rates in control versus gap versus under, with warming resulting in a greater increase in decomposition in control areas, and the lowest increase in decomposition in under areas (Table 9) (Table 10) (Table 11) (Figure 12). Specifically, experimental warming was associated with a 16 % increase in decomposition in control plots, compared to an 18 % increase in decomposition in gap plots and 10 % increase for under plots (Table 9).

The importance of temperature, soil moisture, PAR and fungal PLFAs (the significant controls, with total PLFAs and fungal: bacteria PLFAs were removed due to collinearality) in explaining the variation in litter decomposition rates between location and warming treatments was determined using a multiple linear regression analysis

(Table 12). 93 % of the variation in % mass loss could be explained by the variables included in the analysis. Of the variables found to be significant in the regression analysis, the abiotic variables of soil temperature and PAR had larger individual effects sizes than the biotic variable tested- fungal PLFAs. The change in soil temperature had the greatest effect on % mass loss, the effects of changes in PAR followed closely, and fungal PLFAs resulted in the lowest change in % mass loss in our data set. Increases in temperature and PAR increased % mass loss. Whereas we found that there was a negative relationship between total fungal PLFAs and mass loss. The regression analysis revealed positive interactions for soil temperature* PAR, soil moisture* PAR and PAR* fungal PLFAs, and a negative interaction between soil temperature* fungal PLFAs. The coefficient interactions revealed that the relationship between soil temperature and fungal PLFAs on % mass loss had the greatest relative effects size in this study followed closely by the interactions between fungal PLFAs* PAR, and soil temperature* PAR. Although changes in soil moisture were not revealed to be a significant variable in this model, soil moisture did interact with PAR, although the relative effects size was at least 50 % lower than the effects sizes of the other interactions.

From assessing the interactions, we found that:-

1. Higher soil temperatures, increases the effect of higher PAR receipts on % mass loss.
2. As fungal PLFAs increase the effect of increasing soil temperature on mass loss is reduced.
3. We found that as soil moisture increases, increasing PAR receipts have a greater effect on % mass loss.
4. In unshaded areas, increases in fungal PLFAs result in a greater increase in mass loss, than in shaded areas.

In addition to the analysis using the standardised variables, an unstandardized analysis was conducted using the centred variables. The coefficients from the unstandardized analysis revealed that a yearly average 1^{o}C increase in soil temp increased % mass loss by 23. A yearly 100 unit increase in PAR increases % mass loss by 8.5. Therefore the ~ 300 unit increase in PAR between the under and control areas would account for an increase in % mass loss by 25. A 1 unit increase in fungal PLFAs decreases % mass

loss by 1.7. Fungal PLFA values had a range of 10, therefore the change in fungal PLFAs in this model could is associated with a 17 point reduction in % mass loss.

Table 12. Multiple linear regression analysis of the effects of soil temperature, growing season soil moisture, PAR and total fungal PLFAs (bacteria: fungal PLFAs, and total PLFAs were excluded due to collinearity) and all 2 way interactions, on leaf litter % mass loss. Variables were centred and standardised prior to analysis. The coefficients for the variables were taken from the model to indicate relative effects size and direction, and R^2 to determine the goodness of fit of the model.

	Coefficients
Soil Temperature	15.19792
Soil Moisture	NS
PAR	14.67482
Fungal PLFAs	-4.52088
Soil Temperature* Soil Moisture	NS
Soil Temperature* PAR	16.12458
Soil Temperature*Fungal PLFAs	-18.0587
Soil Moisture* PAR	8.131233
Soil Moisture* Fungal PLFAs	NS
PAR* Fungal PLFAs	17.42718
R^2	0.93

3.5 Discussion

Our first hypothesis that the presence of PV arrays will reduce litter decomposition rates and sensitivity in response to warming was supported. Soil temperatures across the solar farm were similar during the winter decomposition period; however, soil temperatures in the summer under the PV arrays were on average 3 ^{o}C lower than in the control and gap areas (Table 9). Temperature differences of this magnitude are supported by other studies which have found similar microclimatic effects in UK solar farms (Armstrong, Ostle and Whitaker, 2016). The reduction in PAR under the panels was also found to correlate to decomposition rates. Lower levels of solar radiation including UV radiation as indicated by the PAR data, have been found to directly through the physical breakdown of leaf litter or indirectly through effects on microbial communities in

ecosystems with high solar radiation levels (Foereid *et al.*, 2010; Johnson, 2003). There is evidence to suggest, although limited, that even in temperate ecosystems where radiation receipts are relatively ambient it is likely that changes in radiation will affect litter decomposition (Rutledge *et al.*, 2010). In addition, solar radiation receipts affect root exudate quality which is closely related to photosynthetic rates. Changes in root exudates may effect microbial communities and subsequently decomposition, through effects on the solubility, movement and absorption of nutrients key to microbial communities and pH (Yang and Cai, 2006). Soil moisture was found to be affected by the presence of PV arrays. This change in soil moisture under the PV arrays is likely to be a factor controlling decomposition (Prescott, 2010). Low levels of soil moisture inhibit decomposition through reduced microbial activity (Briones *et al.*, 2014). However, under the PV arrays during the winter months soil moisture was very high, although this may also inhibit decomposition through reduced microbial activity due to lower the lower gas exchange potential of the soil (Freeman, Ostle and Kang, 2001). The response of microbial respiration is dependent on other soil properties, so it is not possible to be certain that at ~ 70-80 % soil moisture under the PV arrays that microbial activity would be limited. The effects of PV arrays on soil moisture are seasonal: in the winter months, soil moisture is greater under the PV arrays than in the control and gap areas, whereas in the summer soil moisture is lower under the PV arrays than in the control and gap areas. The observed differences in microclimate across the solar farm in the control gap and areas under the PV arrays (PAR and soil temperature) are of a magnitude known to affect litter decomposition (Mercado *et al.*, 2009a; Ise and Moorcroft, 2006).

Soil microbial communities were affected by the presence of PV arrays: specifically, fungal community abundance was affected by the presence of PV arrays, with lower total fungal PLFAs in soils under the PV arrays in comparison to the control and gap areas. This change in the fungal community also affected total soil PLFAs and the ratio of fungi: bacteria. The effect of PV arrays on soil microbial communities is likely due to direct changes in microclimate and indirectly through changes in vegetation communities (Bradford, 2013; Zhang *et al.*, 2013; Waldrop and Firestone, 2006; De Deyn *et al.*, 2009). The vegetation community at this site is known to differ between the different treatment locations, this is mainly due to different seeding regimes, although there is a possibility that some of the vegetation differences particularly in

above ground biomass could be attributed to climatic variables (Armstrong, Ostle and Whitaker, 2016). Studies have found that fungal abundance is promoted by both drought and wetter soil conditions (Brockett, Prescott and Grayston, 2012; Yuste *et al.*, 2011; Briones *et al.*, 2014). It is, therefore, unclear how the observed changes in fungal abundance, may be related to changes in soil moisture on solar farms. Fungal abundance decreases under the PV arrays most likely due to the lower summer temperatures. Multiple studies have found that fungal abundance positively correlates to temperature (Crowther and Bradford, 2013; Briones *et al.*, 2009; Castro *et al.*, 2010). Although there are seasonal differences in the microclimatic effects of the PV arrays, temperature is most likely to affect microbial communities in the summer when mass loss and subsequently activity rates and population growth is greatest. Changes in the fungal: bacterial ratio as observed under the PV arrays have been shown to be important determinants for processes of C and N cycling i.e. decomposition. Increases in fungal community abundance have been found to have more efficient C and N cycling (Malik *et al.*, 2016).

This study demonstrated that warming and location had interactive effects on litter decomposition processes. This suggests that there is potential for the response to global warming to be exacerbated or mediated by variations in radiation receipts across solar farms. We found that the presence of PV arrays dampened the effect of warming on decomposition processes. This may be due to lower soil moisture under the panels during the peak decomposition period, the effect of shading, and the change in soil microbial community amongst other factors such as the vegetation community composition. The temperature sensitivity of the decomposition of soil organic matter in grasslands has been found to positively correlate with soil moisture (Bradley-Cook *et al.*, 2016). This means that the lower soil moisture contents under the PV arrays during the peak decomposition period could have mitigated the effect of warming on the decomposition of leaf litter in our study. Studies have found that shading may reduce the temperature sensitivity of decomposition due to the direct and indirect effects of solar radiation on decomposition processes (Zepp *et al.*, 2007b; Williamson *et al.*, 2014). Exposure to UV radiation has been shown in some circumstances to stimulate the decomposition of leaf litter. Directly, photodegradation, the physical breakdown of the leaf litter, will be lower under the PV arrays due to reduced radiation receipts. Indirectly, the degradation of leaf litter by exposure to UV, will affect microbial

decomposition, by altering the mineralization of nutrients which microbial communities need to function (Zepp *et al.*, 2011). In addition, changes in solar radiation receipts can promote differences in vegetation community affecting the quality and quantity of litter inputs affecting decomposition (Bahn *et al.*, 2013; Lin, Scarlett and King, 2015). These effects have the potential to interact with climate warming. The effect of warming on decomposition under the PV arrays may be reduced due to the above stated direct and indirect effects of solar radiation on decomposition. Fungal diversity controls the temperature sensitivity of soil organic matter decomposition under drought conditions such as under the PV arrays during summer (Yuste *et al.*, 2011; Louis *et al.*, 2016). The diversity of the fungal community was found to be the best predictor of the sensitivity of soil organic matter decomposition to temperature (Yuste *et al.*, 2011).

Our second hypothesis that the variation in litter decomposition will be accounted for to a greater extent by the change in abiotic variables (temperature, soil moisture, PAR) as opposed to biotic variables (fungal plfa) was supported. Through a multiple regression analysis of our decomposition data, we revealed that of our measured variables, temperature had the largest effects size. Second to temperature was PAR and finally total fungal PLFAs. Climate and decomposer organisms (including soil microbes), along with litter quality are recognised as the dominant factors regulating litter decomposition rates (Palosuo *et al.*, 2005). Of the two fundamental controls, we included in our experiment (litter quality was constant), climate was assumed to be the most important group of factors, as the activity of decomposer organisms is regulated by climate (Castro *et al.*, 2010; Classen *et al.*, 2015). In addition to climatic controls on decomposer organisms, in solar farms the change in vegetation may also account for changes in the microbial community (Churchland and Grayston, 2014). The mycorrhiza which fungi form with plants means that, fungi may only be present or active if a particular plant species is present (Finlay, 2008). Of the variables analysed, it is notable that the change in PAR receipts on a solar farm in the UK effected decomposition, due to the relatively low ambient solar radiation receipts at the latitude of the study site and the fact that most assessments of C cycling in temperate grasslands do not include solar radiation. There is a strong positive correlation between temperature and PAR during the spring and summer, however, there is a difference in temperature between the control and gap areas and subsequently decomposition, even though there is no difference in PAR receipts between these two areas. In temperate ecosystems leaf litter

decomposition generally positively correlates with temperature and precipitation(Portillo-Estrada *et al.*, 2015). Our data supports this as under the PV arrays during the spring and summer when decomposition rates were greatest, the lower soil temperatures and soil moisture, which is closely related to precipitation, resulted in suppressed decomposition rates. There is some debate about whether temperature or moisture has the largest control over decomposition rates in temperate ecosystems. Here, our evidence supports the theory that of the climatic variables, temperature has the largest effect in determining decomposition rates (Aerts, 1997). Soil moisture, which was averaged over the year, was not found to be a significant variable in the multiple regression analysis, this is likely to be due to the seasonal variation the effect of PV arrays on soil moisture. In contrast other studies have shown that soil moisture is generally the limiting factor in microbial decomposition(Cortez, 1998). The greater influence of soil moisture over PAR, which correlations strongly to spring and summer day time temperatures, on litter decomposition is supported by an experiment which found that litter decomposition on forest edges in the UK was lower than in the interior of the forest, mainly due to lower water availability at the forest edges, even though shading was greater in the forest interior (Riutta *et al.*, 2012).

3.6 Conclusion

Solar farms in the UK represent a significant land use change, and through changes in microclimate and management have the potential to affect ecosystem functions, however, understanding of these impacts is limited. This study has demonstrated how solar farms may impact upon decomposition processes in UK solar farms, which could increase or decrease the climate change mitigation potential of this renewable energy technology, through changes in ecosystem C storage.

The presence of PV arrays has significant impacts on soil temperature, soil moisture, PAR receipts, leaf litter decomposition rates and the soil microbial community. Here, lower decomposition rates under the PV arrays, appear to be associated with changes in soil temperature, soil moisture, PAR receipts, fungal community composition, specifically affecting the fungal component. We also show that of these variables, the change in temperature under the PV arrays is the main factor in determining decomposition rates. This study also demonstrated how the presence of PV arrays reduced the temperature sensitivity of litter decomposition, potentially through changes in soil moisture, PAR receipts and the soil microbial community. Given the effects of

PV arrays on decomposition, a critical component of C cycling in UK grasslands, it is essential to fully assess the impact of solar farms on ecosystem services such as C storage. The reduced decomposition under the PV arrays, and the reduced temperature sensitivity of decomposition under the PV arrays, has the potential to increase or decrease ecosystem C storage now and under climate warming. However, there is further need to assess the impacts of PV arrays on other ecosystem functions.

3.7 References

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4. UV-B exposure facilitates microbial decomposition of leaf litter in mesic systems

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Data referred to in this chapter can be viewed in full in the appendices presented at the end of this thesis.

4.1 Abstract

Globally, UV-B receipts are changing with cloud cover, ozone depletion, atmospheric pollutants, and land-use change, with potential consequences for decomposition in terrestrial ecosystems. UV-B has been found to be a driver of decomposition in arid ecosystems, however, our understanding of effects in mesic systems is extremely limited. In addition, UV-B may interact with leaf litter chemistry to affect plant litter decomposition. Here we investigate how pre-exposure to UV-B affects the decomposition of litter from three plant functional types: grass- *Agrostis capillaris*, conifer- *Picea abies* and broadleaf- *Acer pseudoplanatus*, by analysing differences in decomposition rates. This analysis reveals an interaction between leaf litter chemical traits and UV-B pre-exposure on leaf litter decomposition, with UV-B facilitating microbial decomposition of recalcitrant leaf litter. A negative correlation was observed between decomposition rates and the effect of UV-B exposure on decomposition rates: the conifer litter had the lowest levels of decomposition; however, pre-exposure to UV-B had the greatest effect on decomposition rates in the conifer litter, increasing decomposition rates by 40 %. The effect of UV-B on microbial decomposition was greatest in litter with high lignin and low nitrogen, cellulose and hemicellulose contents. Our findings demonstrate the need to consider how changes in UV-B receipts may impact decomposition rates with potential consequences for carbon storage in mesic ecosystems. Further, the effect of changes in UV-B on decomposition processes may

dependent on ecosystem type, with decomposition in conifer forests affected to a greater degree than grasslands.

4.2 Introduction

Globally, UV-B receipts are changing with cloud cover, ozone depletion, atmospheric pollutants, and land-use change, with potential consequences for carbon (C) cycling. C stores in terrestrial ecosystems are increasingly important to help mitigate the impacts of climate change. It is, therefore, important that we develop our understanding of controls on carbon storage i.e. production and decomposition processes in terrestrial ecosystems (Crowther and Bradford, 2013). Decomposition is thought to be dominated by microbial processes, however, recently it has been shown to be driven by photodegradation in arid ecosystems, although our understanding is limited in more mesic systems (Austin, Méndez and Ballaré, 2016).

Microbial decomposition is the breakdown of organic matter by microbes, whereas photodegradation is the abiotic process by which solar irradiance breaks down the compounds of organic matter(King, Brandt and Adair, 2012). Microbial decomposition and photodegradation are controlled by three main factors: climate, litter quality and soil organisms (Prescott, 2010; Butenschoen, Scheu and Eisenhauer, 2011). Debate exists, as to which factors exert the dominant control over microbial decomposition and photodegradation (Bradford *et al.*, 2016; Gaxiola and Armesto, 2015). Factors such as climate, microbial communities and soil properties have been found to be the best predictors of microbial decomposition (Aerts, 1997). Climate modulates microbial decomposition, through temperature and moisture availability (Davidson and Janssens, 2006) which regulate soil microbial activity (Gaxiola and Armesto, 2015). Generally, microbial decomposition is greatest in warmer wetter environments (Ise and Moorcroft, 2006). However, thresholds exist where excess moisture or high temperatures start to inhibit microbial activity (Freeman, Ostle and Kang, 2001; Briones *et al.*, 2014; Bradford, 2013). There is an increasing body of evidence to suggest that vegetation communities exert a dominant control over microbial decomposition rates (Bakker, Carreno-Rocabado and Poorter, 2011; Boyero *et al.*, 2014). Specifically, it may be the presence or absence of a key species or plant functional type from a community which controls decomposition rates (Cornwell *et al.*, 2008). Differences in the vegetation community composition, affect microbial decomposition through changes in leaf litter and rhizodeposits chemistry and quantity, the allocation of C above and below ground,

and microbial communities (Ward *et al.*, 2015; Wood, Cavaleri and Reed, 2012; Lamb, Kennedy and Siciliano, 2011; Drenovsky *et al.*, 2010). Litter quality and climate modulate microbial activity and are fundamental controls on litter decomposition in terrestrial ecosystems and subsequently mediate ecosystem carbon storage (Aerts, 1997; Meentemeyer, 1978; Bakker, Carreno-Rocabado and Poorter, 2011; Gaxiola and Armesto, 2015). Litter quality variables crucial to decomposition processes include nutrient concentrations, carbon-to-nitrogen ratios (C: N) and lignin content (Parton *et al.*, 2007; Melillo, Aber and Muratore, 1982; Talbot and Treseder, 2012; Talbot *et al.*, 2012). Specifically, increases in nutrient availability and lower C:N ratios increase microbial decomposition, and increases in lignin decrease microbial decomposition.

Photodegradation in arid environments is thought to be influenced by factors such as exposure of soil, the thickness of the litter layer, litter chemistry, soil microbial films and exposure to UV radiation (Song *et al.*, 2013). Solar radiation has been demonstrated to affect decomposition processes, through the photodegradation of leaf litter i.e. photochemical mineralisation (oxidation of carbon to CO_2, CO, CH_4), effects on soil microbial communities and litter quality and quantity (Figure 13) (Austin and Vivanco, 2006; Rutledge *et al.*, 2010; King, Brandt and Adair, 2012; Brandt *et al.*, 2010; Schade, Hofmann and Crutzen, 1999). Laboratory experiments have directly linked UV-B exposure to the breakdown of cellulose (Schade, Hofmann and Crutzen, 1999) and partial or complete degradation of lignin (Austin and Ballaré, 2010).The effects of solar radiation exposure on microbial decomposition are poorly characterised. Some studies show UV-B exposure decreases microbial decomposition, reporting a direct negative effect of UV-B radiation on soil microbial activity and abundance (Belnap *et al.*, 2008). Whereas, other research has demonstrated that when vegetation ground cover is sufficient to shade the soil and microbial communities, senesced leaf litter which is effectively pre-exposed to UV-B prior to microbial decomposition, exhibits greater decomposition rates (Gaxiola and Armesto, 2015). This means that in ecosystems where there is little exposed soil, such as in grasslands, UV-B exposure may increase decomposition through microbial facilitation.

Differences in leaf litter inputs, through changes in quantity and quality (i.e. lignin content, C:N, cellulose and hemicellulose), can alter the rate of microbial and photodegradation processes leading to changes in C storage (Xu, Liu and Sayer, 2013; Talbot *et al.*, 2012). Plant residues with high lignin and low N and P are not as readily

broken down through extracellular digestion, and thus can inhibit microbial decomposition leading to enhanced C storage (Austin, Ballaré and Schlesinger, 2010). UV-B exposure predominantly affects the lignin fraction of plant litter, causing the biologically recalcitrant material to breakdown when exposed to wavelengths in the UV-B, UV-A and short wave visible range (Zepp *et al.*, 2007a). However, the UV-B fraction has the highest impact on photodegradation per photon flux, compared to other the wavelengths (Ballaré et al., 2011). For example, enhanced UV-B (30% above ambient) has been shown to increase decomposition of *Quercus robra* leaf litter by 27% (Newsham et al., 1997). Lignin is more sensitive to photodegradation, as it contains photosensitive compounds which have a maximum absorbance in the UV-B range (Day, Zhang and Ruhland, 2007). Some studies have found a link between leaf litter lignin content and the effect of UV-B exposure (Austin and Ballaré, 2010; Austin, Méndez and Ballaré, 2016), however further research needs to be conducted to assess this relationship i.e. is the effect is linear and is it present in mesic systems? Further, the lignin content of leaf litter is highly variable between different plant functional types; it is, therefore, likely that the different plant functional types will exhibit differences in the effect of UV-B exposure on decomposition

There is an increasing body of evidence to suggest that climatic factors, such as solar radiation and specifically exposure to the UV-B fraction, could play a significant role in microbial decomposition in terrestrial ecosystems (Austin and Vivanco, 2006; Brandt, King and Milchunas, 2007). Specifically, UV-B facilitates microbial decomposition through the photodegradation of biologically recalcitrant material, which in turn makes other fractions of easily decomposable litter more accessible to microbes (Austin, Méndez and Ballaré, 2016). The effects of UV-B on microbial decomposition processes have mainly been studied in arid and semi-arid systems, whilst little is known about the impacts in more mesic systems (Song *et al.*, 2013). Within mesic systems, microbial degradation dominates decomposition due to relatively high soil moisture contents and ambient temperatures (Grace and Rayment, 2000; Janzen, 2004). However, photodegradation is a physical process and even at lower levels of UV and in mesic environments, UV has the potential to result in the breakdown of leaf litter, although its role is probably masked by the dominant microbial decomposition processes. Currently, not all of decomposition in mesic systems can be explained by conventional climate and biological variables (Smith, Gao and Steltzer, 2009).

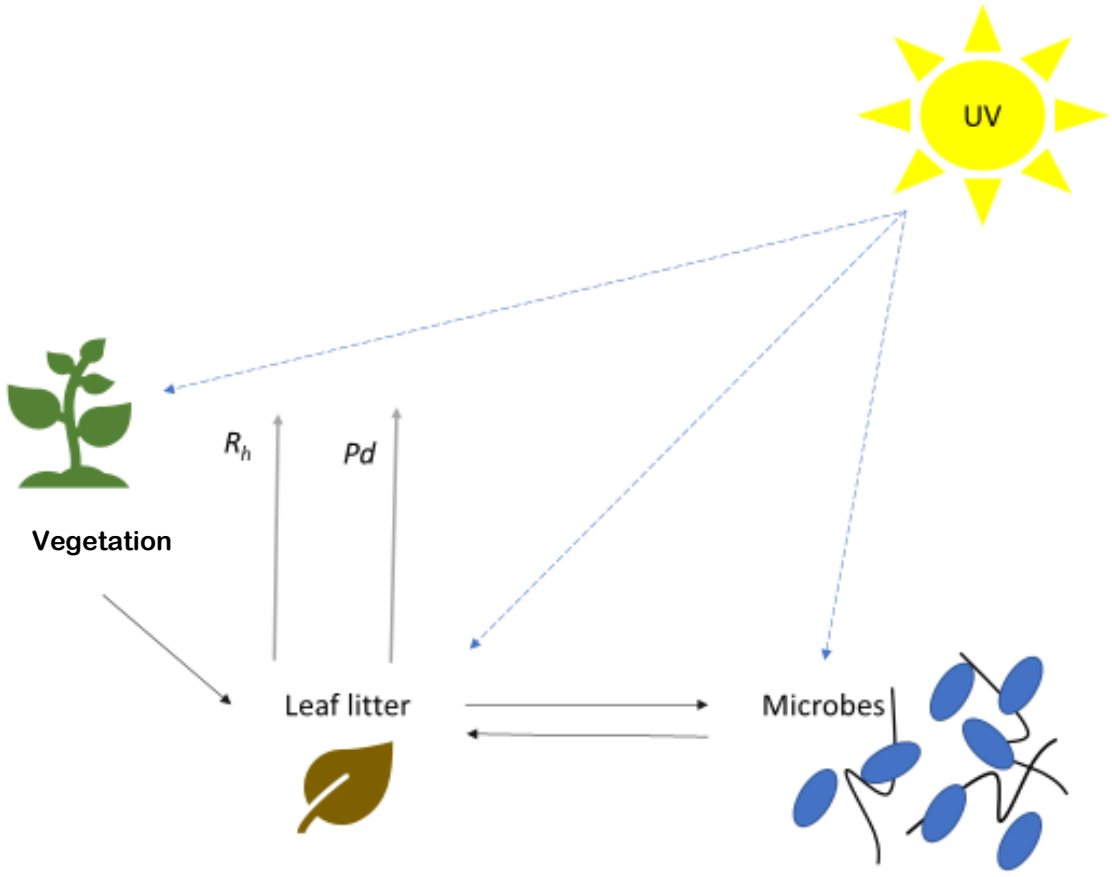


Figure 13. The effect of UV on leaf litter decomposition. Directly UV affects litter decomposition through photodegradation of leaf litter. Indirectly UV affects litter decomposition through changes in vegetation which affects litter quantity and quality. In addition, UV affects microbial populations and the microbial decomposition of leaf litter, however, changes in litter quality also have feedback effects on microbes. R_h- *heterotrophic respiration,* Pd- *photodegradation.*

With the projected shifts in climate, changes to the ozone layer, and land-use changes such as the continued expansion of PV arrays on grasslands (Williamson *et al.*, 2014; Armstrong *et al.*, 2014; Foereid *et al.*, 2011), solar radiation effects on decomposition in mesic systems may play an burgeoning role in C cycling (Foereid *et al.*, 2011), however, our understanding is currently very limited. The overall aim of this research was to investigate whether UV-B pre-exposure affects litter decomposition in different plant functional types from mesic systems. We hypothesised that: (1) rates of leaf litter decomposition will be greater for leaf litter which has been pre-exposed to UV-B, and (2) the effect of UV-B pre-treatment on decomposition will be dependent on leaf litter

type. These hypotheses were tested using two experiments, which assessed how UV-B exposure prior to microbial decomposition could affect the decomposition of three litter types with varying litter chemistry characteristics, taking measurements of mass loss and litter chemistry.

4.3 Methods

4.3.1 Litter collection

Three recently senesced leaf litters were collected in October 2014. Grass litter (*Agrostis capillaris*), conifer litter (*Picea abies*) and broadleaf litter (*Acer pseudoplanatus), were collected from* Hazelrigg Research Station (SD 492 579), Beacon Fell Country Park (SD 568 429) and Lancaster University, Bailrigg Campus (SD 482 573), respectively. Each of the collection sites was located in Lancashire, UK. The three litter types were dried at 60 $^{\circ}$C, until a constant weight was reached. The broadleaf and grass litter were cut up to the same length as the pine needles, approximately 1.5 cm, and the broadleaf litter was then cut into strips the same width as the grass, approximately 0.5 cm. The leaf litter was autoclaved (121 $^{\circ}$C for 15 minutes under 1.05 kg/cm^2 pressure) (Brandt, Bohnet and King, 2009) in order to sterilise and prevent microbial decomposition during UV-B/ no UV-B treatments.

4.3.2 UV-B treatment

Half of the litter from each litter type was exposed to a high UV-B treatment for 6 months. Meanwhile, the remaining litter was kept in an airtight container, in the dark, at the same temperature (20 ^{o}C). The UV-B treatment was applied using four Philips TL 40W/01 UV-B Narrowband Phototherapy fluorescent tubes, which were housed in a wooden chamber (UV-B chamber). The UV-B chamber provided a UV-B exposure of 30 Wm^{-2} and had a spectral footprint between 290 nm to 315 nm. Litter was then placed in the UV-B chamber in a thin layer, for 6 months, and turned over at weekly intervals.

4.3.3 Decomposition

Mass loss was used to determine decomposition rates in both experiment A and experiment B. Soil was collected from Hazelrigg Research Station, Lancaster University, UK and passed through a 10-mm sieve (removing stones and woody debris) and sealed in air tight bags and stored at 4 ^{o}C until use. 20 g of the fresh soil was placed in a petri dish to act as the inoculum. On top of the soil a piece of mesh (1 mm mesh

size PVC coated fibre glass) was placed, and on top of the mesh, 1 g of leaf litter for experiment A and 0.5 g of leaf litter for experiment B was placed. The litter was wetted with 20 ml of deionised water and the petri dish was then sealed 75 % shut to allow for gaseous exchange, whilst minimising water loss. For the duration of the decomposition periods, the petri dish set up containing the litter was stored in the dark at a constant temperature (20 ^{o}C/ 15 ^{o}C dependent on the experimental stage). In both experiments, mass loss was determined by weighing the litter (dried at 60 ^{o}C to a constant weight) at the end of each incubation period. In addition, decomposition rates were calculated for each treatment combination at the end of the decomposition period using mass loss data. Specifically, replicates from the separate time points were randomly assigned a number 1-5 and grouped accordingly. Using the grouped mass loss data points, mass loss was first changed to mass remaining, then a single-pool negative exponential model was then fitted to this data using a non-linear regression and decomposition rates (*k*) were calculated (Adair, Hobbie and Hobbie, 2010):-

$$X(t) = e^{-kt}$$

Where X(t) is the proportion of the original litter remaining and t is the time in days since the microcosms were set up. Higher k values represent greater levels of leaf litter mass loss and subsequently higher decomposition rates.

Experiment A

Experiment A assessed the effects of UV-B pre-exposure on the decomposition of three litter types (grass, conifer and broadleaf). The factorial design combined two levels of radiation treatment (UV-B pre-exposed and unexposed) and three litter types (grass broadleaf, conifer) with five replicates of each treatment combination destructively harvested at four time points (120 microcosms in total). All litter used in experiment A was initially decomposed for 2 months at 20 $^{\circ}$C[1]. After the initial decomposition period, mass loss was determined on one set of replicates, for the remaining sampling sets, the soil in each petri dish was replaced with fresh soil before

the secondary decomposition period, at 15 °C. In the secondary decomposition period mass loss was measured over 6 months, at two monthly intervals[2].

Experiment B

Experiment B used only the UV-B pre-exposed and unexposed grass litter. This experiment assessed the impact of UV-B pre-exposure on short-term decomposition, as grass litter was found to have a large mass loss in the 0-2 month decomposition period in experiment A. The same decomposition set up was used as previously described, however this time 0.5 g of litter was used and mass loss at 15 °C was measured after 1, 2, 4 and 10 weeks. For each time point, there were five replicates.

4.3.4 Litter characteristics

Leaf litter (untreated and UV-B pre-treated) (from two randomly selected replicates), was analysed for lignin, cellulose and hemicellulose prior to and after the UV-B treatment, and after 2 and 8 months of decomposition to assess which fractions of the leaf litter were affected by the UV-B treatment and decomposition. Lignin content was determined by the solubilisation of cellulose in sulphuric acid (acid detergent lignin), using an Ankom 220 Analyser (Gomes *et al.*, 2011). Cellulose and hemicellulose were obtained through determination of sulphuric acid detergent fibre (Ankom 220 Analyser), acid detergent lignin (Ankom 220 Analyser) and neutral detergent fibre (enzymatic gravity) (Hall *et al.*, 1999), where:-

Hemicellulose content = Neutral Detergent Fibre – Acid Detergent Fibre

Cellulose content = Acid Detergent Fibre - Acid Detergent Lignin

[2] This unconventional approach was undertaken as the initial temperature of 20 °C had caused the soil to dry out. The water could not be reapplied as it was unknown as to how long ago the samples had dried out and decomposition subsequently halted, essential information to calculate true decomposition rates.Therefore, the soil was replaced and the petri dishes were then incubated at 15 °C. After this a decision was made to investigate short term decomposition. Ideally, all three litter types would have been used in this analysis, however, there was insufficient quantities of broadleaf and conifer litters. Consequently, experiment B only includes the grass leaf litter.

In addition, we determined the N content of the freshly collected leaf litter. To determine leaf total N, a 30 mg subsample of oven dried (60 °C) and ground (using a ball mill for 5 minutes) material, was analysed using a LECO Truspec CN Analyser (LECO, USA) (Carter, 2007).

4.3.5 Statistical analysis

All data were checked for normality and p values < 0.05 were deemed significant. All analysis was conducted in R Studio (Rstudio Team 2015).

To test hypothesis 1, a two-way ANOVA was run using the nlme package, to test for the effects and interactions of UV-B treatment and litter type on k. For hypothesis 1, we focused on the results for effect of UV-B treatment on k. Experiment B was used to further test hypothesis 1. Here a one-way ANOVA was used to assess the effect of UV-B treatment on k for the whole decomposition period.

For hypothesis 2, we focused on the results for effect of litter chemistry on k or % mass loss and the interaction between UV-B treatment and litter chemistry. Further, a multiple linear regression was conducted for the effects and interactions of starting lignin/cellulose/hemicellulose content and UV-B pre-treatment on litter k. Subsequently, Tukey's, posthoc analysis was used to test whether the individual treatment combinations differed in k or % mass loss. Finally, a multiple regression analysis was conducted using k as the criterion variable, with predictor variables of UV-B, litter characteristics from the freshly collected litter (N, lignin, hemicellulose and cellulose) and all two-way interactions, created using centred and standardised data. The regression coefficients from this analysis were used to determine relative effects size.

4.4 Results

In the following section, the variation in leaf litter decomposition between the UV-B treated and untreated litter, for the three litter types is outlined and explained using an analysis of leaf litter characteristics.

4.4.1 Hypothesis 1

Litter type controlled the effect of UV-B on k, with differences in k only found between UV-B treated and untreated litter in the broadleaf and conifer litters ($df = 2$, $F = 14$, $p < 0.05$) (Table 13, Table 14, Figure 14). In grass litter, UV-B pre-treatment did not affect

k, however, UV-B pre-treatment increased k by 23 % in broadleaf litter and by 40 % in conifer litter. Over the total 8-month decomposition period (initial 2 months at 21 °C + 2nd stage 6 months at 15 °C), mean litter k decreased in the order grass > broadleaf > conifer, with all three litter species differing from each other (df = 2, F = 1499, p < 0.001). K in grass litter was 203 % greater than in the conifer litter, and k in the broadleaf litter was 138 % greater than in the conifer litter.

The results of the experiment B showed that there was no difference in k between the UV-B pre-treated (k = - 0.07021) and untreated grass leaf litter (k = -0.06983) (df = 1, F = 9.4, p > 0.05) (Figure 15).

Table 13. Average k values for the UV-B treated litter and untreated litter by litter type.

Litter type	UV-B	Untreated
Broadleaf	-0.01427	-0.0159
Conifer	-0.01765	-0.01815
Grass	-0.0122	-0.01306

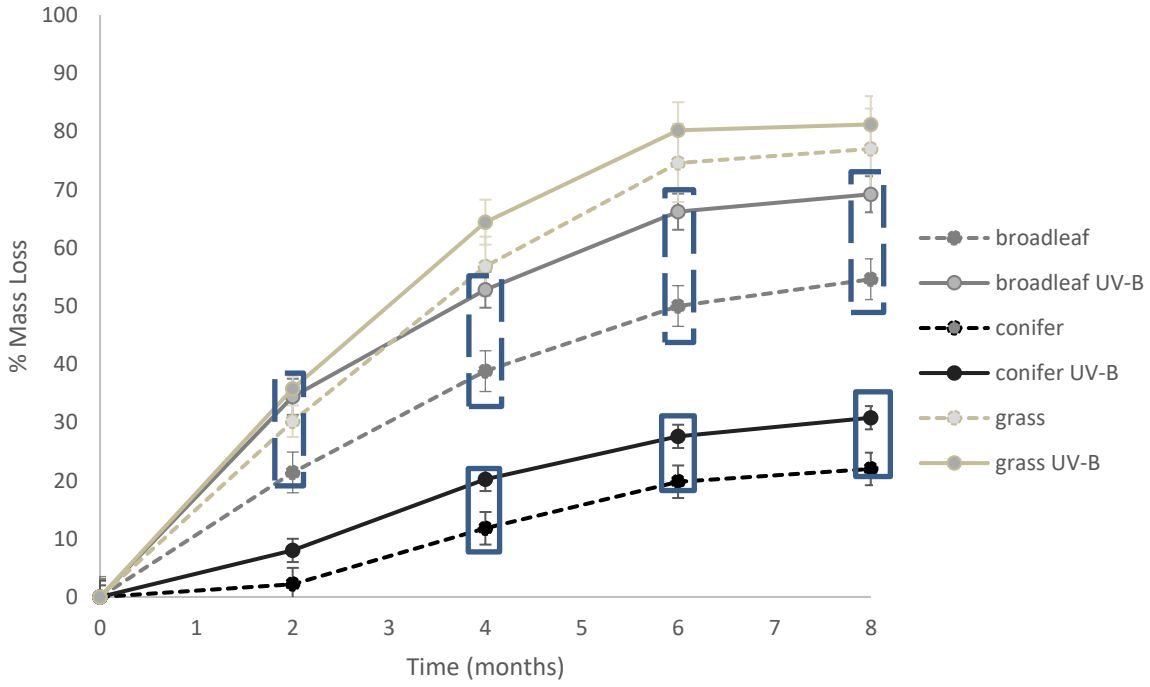


Figure 14. Differences in mass loss between litter type and UV-B pre-treatment. Error bars represent standard error. N = 5. Data points in boxes show where the posthoc analysis revealed significant differences between the UV-B treated and untreated litter for the different litter types (solid boxes= differences in conifer decomposition and dashed boxes = differences in broadleaf litter decomposition) at the sampling points.

Table 14. ANOVA results for the effects and interactions of litter type and UV-B pre-treatment on k.

Time	**Df**	**F**	**P**
K			
Litter type	2	1499	<0.001
UV	1	131	<0.01
UV*Litter type	2	14	<0.05

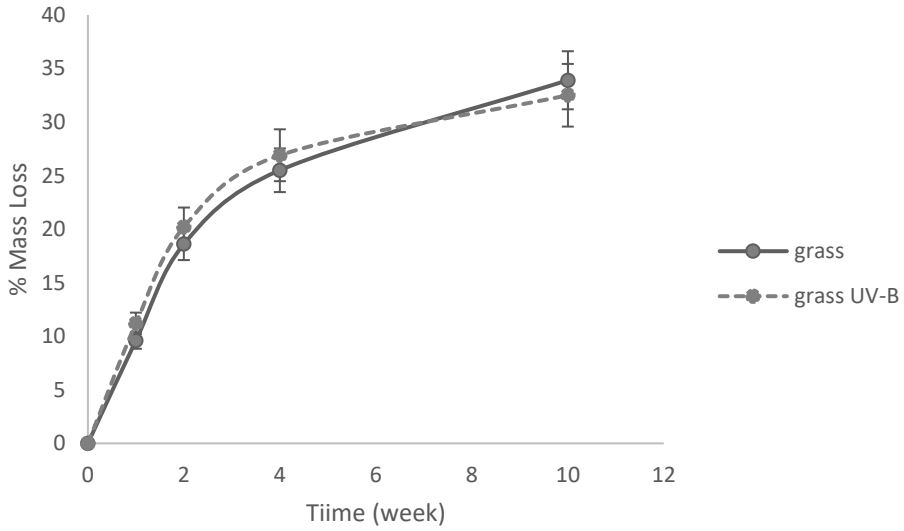


Figure 15. Differences in the % mass loss between UV-B treated and untreated grass litter. N = 5.

4.4.2 Hypothesis 2

The three freshly collected litter types had very different chemical compositions (Figure 16) (Table 16). The grass litter had intermediate levels of cellulose and hemicellulose contents, the lowest lignin content and highest N content. Broadleaf litter had the highest cellulose and hemicellulose contents and intermediate lignin and N levels. The conifer litter was found to have the lowest cellulose, hemicellulose and N levels but the highest lignin content. UV-B pre-treatment was not found to affect litter chemical composition, or result in mass loss prior to the decomposition phase of the experiment for any of the leaf litters (Figure 16) (Table 18).

The litter chemistry of freshly collected leaf litter was found to influence k (Table 15). When lignin % was high, k was lower ($df = 1$, $F = 2940$, $p < 0.001$). Lignin was highest in the conifer leaf litter, which had the lowest k. Whereas, lignin content was lowest in the grass litter and corresponded with the highest k. Lignin content of the leaf litter interacted with the UV-B pre-treatment to affect k in broadleaf and conifer litters ($df = 1$, $F = 4.2$, $p < 0.05$). In the grass litter where the lignin content was very low, the k was unaffected by UV-B pre-treatment. After 8 months of decomposition, k in the UV-B pre-treated broadleaf litter was 27 % greater than the untreated litter, and for the conifer litter this difference was 40 %.

The hemicellulose and cellulose content of the freshly collected litter was also found to affect k and the effect of UV-B pre-treatment on k (Table 15). As hemicellulose and cellulose content increases, k also increases (hemicellulose- $df = 1$, $F = 1440$, $p < 0.001$, cellulose- $df = 1$, $F = 1308$, $p < 0.001$). Hemicellulose and cellulose content was also found to interact with UV-B pre-treatment (hemicellulose*UV-B- $df = 1$, $F = 7.5$, $p < 0.05$, cellulose*UV-B- $df = 1$, $F = 8.7$, $p < 0.05$). When hemicellulose and cellulose contents were greatest i.e. in the grass litter, UV-B pre-treatment did not affect k. Whereas, in the broadleaf and conifer litters where hemicellulose and cellulose contents were lower, UV-B pre-treated was found to affect k. This relationship is directly in contrast to the positive relationship between lignin content and the effect of UV-B pre-treatment on k.

The N content of the starting leaf litter correlated positively with k, with high N contents corresponding to greater k ($df = 1$, $F = 1670$, $p < 0.001$) (Table 15). However, the effect

of the UV-B treatment on k was greatest in the litter with the lowest N content, and lowest in the litter with the highest N content ($df = 1$, $F = 6.2$, $p < 0.05$).

The relative importance of initial leaf litter chemistry (N, lignin, hemicellulose and cellulose) and UV-B treatment on k using the correlation coefficients from a multiple regression analysis (Table 17). In this analysis litter lignin and N contents an UV-B treatment were found to affect k. Lignin content had the greatest relative effect on k followed by UV-B, then N content. This revealed that of the tested leaf chemistry traits lignin content was most closely associated with changes in K, as it had the greatest effects size. The regression coefficient for leaf N was significant however, hemicellulose and cellulose were not significant. Leaf litter lignin had a greater effects size than the UV-B treatment, and the positive interaction between UV-B and lignin was a stronger driver than the effect of UV-B alone. Furthermore, there was a negative interaction between UV-B and leaf N, however, this had a lower effect on k than the interaction between UV-B and lignin. The interaction between UV-B and N was negative, this indicates that as leaf litter N content increases the effect of UV-B decreases. Whereas the positive interaction between UV-B and lignin indicates that as lignin content increases the effect of UV-B increases.

In addition, the litter chemistry was analysed after 2 and 8 months of decomposition (Figure 14) (Table 18). This revealed that the lignin content of the broadleaf and conifer litters was affected by the UV-B treatment, with lignin content lower in the UV-B treated litters after microbial decomposition. Whereas, the hemicellulose and cellulose content was unaffected by the UV-B treatment for all leaf litter types. Specifically, after 2 months of decomposition % lignin of the UV-B treated broadleaf litter was 8 % lower than the untreated litter ($df = 1$, $F = 18$, $p < 0.05$), and for the conifer litter this value was 7 % ($df = 1$, $F = 40$, $p < 0.05$). After 8 months, the lignin content of the UV-B treated litter was 14 % lower for broadleaf ($df = 1$, $F = 27$, $p < 0.05$) and 11 % lower for conifer ($df = 1$, $F = 76$, $p < 0.01$). UV-B treatment did not affect grass litter chemistry at the two time points.

Table 15. Two-way ANOVA for the effect of starting lignin/cellulose/hemicellulose content and UV-B pre-treatment on litter k.

	Df	*F*	*P*

Lignin	1	2940	< 0.001
Cellulose	1	1308	< 0.001
Hemicellulose	1	1440	< 0.001
Nitrogen	1	1670	< 0.001
UV-B	1	131	< 0.01
*Lignin *UV-B*	1	4.2	< 0.05
*Cellulose*UV-B*	1	8.7	< 0.05
*Hemicellulose*UV-B*	1	7.5	< 0.05
*Nitrogen*UV-B*	1	6.2	< 0.05

Table 16. Average initial leaf litter fibre chemistry of broadleaf, conifer and grass litter. Lignin, hemicellulose and cellulose N = 2. Nitrogen N = 5.

Leaf Litter	***Leaf N g kg^{-1}***	***Lignin %***	***Hemicellulose %***	***Cellulose %***
Broadleaf	27.4	16.9	21.4	14.2
Conifer	11.2	33.4	18.2	13.7
Grass	34.7	6.8	33.2	31.6

Table 17. Regression coefficients from a multiple regression analysis of initial leaf litter chemistry and interactions with UV-B

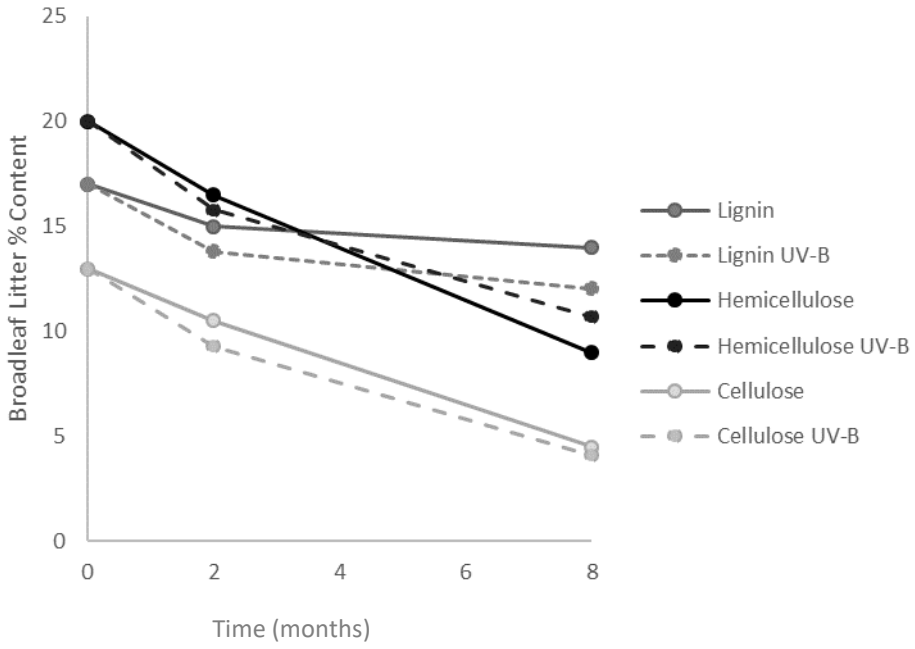
Regression *Coefficients*

Lignin	-20.4
Hemicellulose	NS

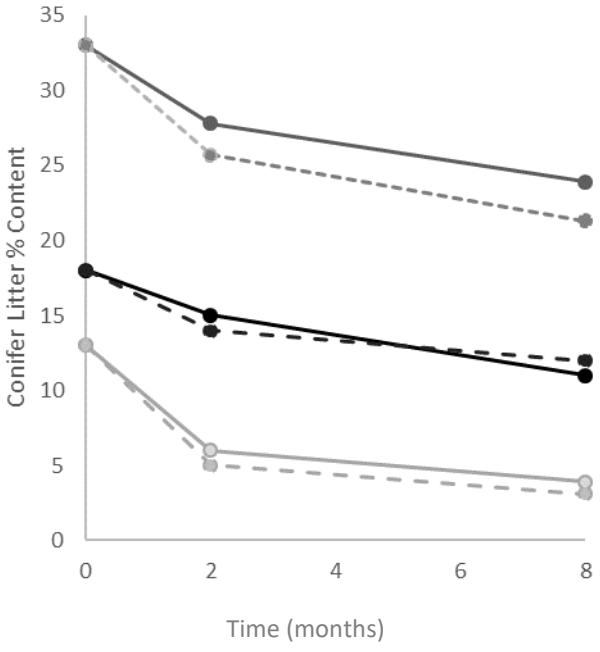
Cellulose	NS
N	3.9
UV-B	4.5
*Lignin*UV-B*	4.8
*Hemicellulose*UV-B*	NS
*Cellulose*UV-B*	NS
*N*UV-B*	-2.9

Table 18. One way ANOVA analysis for the effect of UV-B on leaf litter chemistry of broadleaf, conifer and grass leaf litters after the initial UV-B exposure period and after 2 and 8 months of microbial decomposition. N = 2.

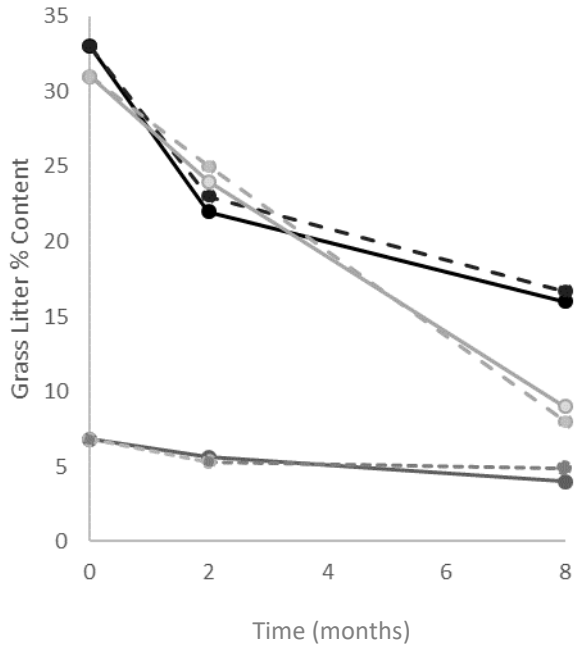
	Initial			2 months			8 months		
	Df	*F*	*P*	*df*	*F*	*P*	*df*	*F*	*P*
Broadleaf									
Lignin	1	4.6	> 0.05	1	18	< 0.05	1	27	< 0.05
Hemicellulose	1	0.5	> 0.05	1	3.9	> 0.05	1	2.5	> 0.05
Cellulose	1	1.2	> 0.05	1	2.2	> 0.05	1	0.01	> 0.05
Conifer									
Lignin	1	2.7	> 0.05	1	40	< 0.05	1	76	< 0.01
Hemicellulose	1	1.8	> 0.05	1	3.1	> 0.05	1	3.6	> 0.05
Cellulose	1	0.5	> 0.05	1	4.7	> 0.05	1	3.2	> 0.05
Grass									
Lignin	1	0.3	> 0.05	1	3.8	> 0.05	1	0.2	> 0.05
Hemicellulose	1	1.5	> 0.05	1	0.8	> 0.05	1	0.1	> 0.05
Cellulose	1	1.1	> 0.05	1	2.3	> 0.05	1	0.9	> 0.05


25
20
15
10
5
0
Broadleaf Litter % Content
0
2
4
6
8
Time (months)
Lignin
Lignin UV-B
Hemicellulose
Hemicellulose UV-B
Cellulose
Cellulose UV-B

A


35
30
25
20
15
10
5
0
Conifer Litter % Content
0
2
4
6
8
Time (months)

B



C

Figure 16. Differences in the % composition of lignin, cellulose and hemicellulose, after 2 and 8 months of decomposition between untreated and UV-B treated leaf litters of (A) Broadleaf, (B) Conifer, and (C) Grass. N = 2.

4.5 Discussion

Given the current and predicted changes in climate and land-use, and the potential implications for UV-B receipts, there is clearly a need to better understand how UV-B receipts affect decomposition processes. The aim of our study was to examine the relative effect of UV-B pre-exposure and its interactive effects with litter type on decomposition. We hypothesised that: (1) rates of leaf litter decomposition will be greater for leaf litter which has been pre-exposed to UV-B, and (2) the effect of UV-B pre-treatment on decomposition will be dependent on leaf litter type.

Hypothesis (1): rates of leaf litter decomposition will be greater for leaf litter which has been pre-exposed to UV-B

We found that UV-B pre-exposure increased the rate of leaf litter decomposition, in broadleaf and conifer litters, although not the grass litter, partially supporting our hypothesis (Lin, Scarlett and King, 2015). Our results are supported by previous studies which have also found differences in decomposition due to solar radiation exposure (Day, Zhang and Ruhland, 2007; Foereid *et al.*, 2010; Lin, Scarlett and King, 2015).

We explored the effect of UV-B on the microbial facilitation of leaf litter decomposition. High levels of UV-B radiation was found to change the physiochemical properties of leaf litter: subsequently, UV-B exposed leaf litter had greater microbial decomposition rates, as labile C is more available. This "photo-priming" effect may be due to the increased decomposability of the litter by the direct breakdown of lignin which surrounds some cell wall components i.e. cellulose and hemicellulose (Austin, Méndez and Ballaré, 2016). Under drought laboratory conditions, UV-B pre-exposure has been found to facilitate microbial decomposition, by making leaf litter more degradable (Foereid *et al.*, 2010). However, our study showed that this mechanism could occur in wetter, cooler conditions: environments where microbial decomposition dominates. Other laboratory studies have found no effect of UV-B pre-exposure on subsequent microbial decomposition (Brandt, Bohnet and King, 2009). This is likely to be as a result of the short exposure period used in the experiment, three weeks compared to our six months, and higher radiation levels (30 Wm^{-2} vs 14 Wm^{-2}). This indicates that there is potentially a minimum exposure of UV-B required for detectable effects on leaf litter decomposition.

Unlike (Foereid *et al.*, 2010), where the exposure of leaf litter to high-intensity UV-B, resulted in a small direct effect of radiation on mass loss ~ 2 %, we found no detectable change in mass during the UV-B pre-exposure stage. This may be due to the temperature differences between the studies (20 ^{o}C in this study and 30 ^{o}C in (Foereid *et al.*, 2010)): at higher temperatures, a lower radiation dosage may be required for the photo-oxidation of leaf litter (Zepp *et al.*, 2011; Porcal, Dillon and Molot, 2015). The findings of our study taken with results from the other studies (Foereid *et al.*, 2010; Brandt, Bohnet and King, 2009) who observed a small mass loss/ CO_2 during the exposure period with no microbial decomposition evolution, indicates that a large radiation dosage is required, however, the dosage required is temperature dependent for direct mass loss of leaf litter through exposure to UV-B radiation

Antithetical to our findings, some studies in the past have found a negative relationship between UV-B radiation and leaf litter mass loss (Pancotto *et al.*, 2003). This decrease in decomposition, however, is likely due to the direct effects of UV-B on the microbial community. The high energy wavelengths of UV-B can directly damage soil microbes, and subsequently inhibit decomposition (Rohwer and Azam, 2000). UV-B exposure was found to affect the colonisation of leaf litter by microbes prior to decomposition,

with greater colonisation by pigmented bacteria and fungi (Pancotto *et al.*, 2003), whose absorbing pigments may provide protection from UV-B radiation. This means that the first stages of decomposition in the UV-B exposed litter could differ, due to the dominance of pigmented bacteria and fungi. Yet, the studies which found the negative effects of UV-B on decomposition were conducted in arid environments with low-density vegetation cover. Vegetation intercepts UV-B receipts, therefore, we can infer that as vegetation cover increases negative effects on microbial decomposition will be reduced, to the point where the ratio of vegetation cover to UV-B dosage reaches a value where UV-B exposure will enhance leaf litter decomposition. However, in mesic environments, vegetation cover is generally high and little soil is directly exposed to sunlight, which potentially means that UV-B will not have negative effects on microbial communities and decomposition will be enhanced by UV-B due to changes in the physiochemical properties of leaf litter. To explore this theory, a fully factorial experiment should be conducted with soil and leaf litter exposed to different doses of UV-B radiation prior to microbial decomposition. It should be noted that the soil used as the inoculum in this experiment was not UV-B treated, and therefore the effects we observe on microbial decomposition are due to the pre-exposure of leaf litter to high levels of UV-B radiation. A meta-analysis of the direct and indirect effects of UV-B on leaf litter decomposition has found that generally, UV-B induced lignin photodegradation accelerates litter decomposition, although negative effects on microbial communities limits the effect on mass loss (Song *et al.*, 2013).

A wide range of studies have demonstrated the interactive effects between UV-B exposure and factors such as precipitation, temperature, and vegetation communities, on decomposition. (Brandt, King and Milchunas, 2007) demonstrated that the effects of UV-B radiation on decomposition are highly dependent on precipitation, with an inverse relationship between precipitation and the effect of UV-B. However, this is likely due to associated changes in vegetation as discussed above and moisture limited microbial populations. The growing body of evidence, although inconclusive as to the effects of UV-B exposure on decomposition, reveals an under-appreciated role of UV-B in decomposition. Our results which indicate large differences in mass loss due to UV-B treatments which persist over time, may result in substantial implications for the processing of carbon, nutrient mineralisation, and carbon storage in the ecosystem (Swift, 1979)

There are various scenarios where the UV radiation receipts may/have changed through land-use change e.g. deforestation and land based renewables, or environmental change e.g. ozone depletion, atmospheric aerosol presence, and cloud cover (Mercado *et al.*, 2009a; Zepp *et al.*, 2011; United Nations Environment Programme, 2016). On solar farms where UV-B exposure will be lowered or even excluded by the effect of shading caused by the presence of PV arrays (Armstrong *et al.*, 2014). This may have consequences for C cycling in these systems, potentially reducing k of leaf litters in areas directly under or are shaded by the PV arrays where solar radiation receipts are reduced. Although this experiment analysed the effects of high-intensity UV-B exposure on leaf litter decomposition, other studies have found consequences of ambient UV-B radiation on decomposition (Gehrke *et al.*, 1995; Pancotto *et al.*, 2003; Paoletti, 2005). This work is particularly relevant to solar farms arid ecosystems where photodegradative effects have been found to be greatest (Bosco, Bertiller and Carrera, 2016; Gallo, Sinsabaugh and Cabaniss, 2006; Gaxiola and Armesto, 2015). In mesic ecosystems, the role of photodegradation in decomposition was assumed to be minimal, however, our previous work has revealed how changes in solar radiation receipts can affect ecosystem respiration (a measure which includes soil microbial activity I.e. decomposition). Temperature and precipitation are known to interact with UV-B exposure to affect decomposition. Further work needs to be conducted to assess whether changes in temperature and radiation induced by solar farms in mesic systems with lower ambient UV-B radiation levels, could affect decomposition. It is, therefore necessary, for any analysis of changes in decomposition on solar farms to include assessments of how these factors may interact with UV-B radiation. However, it is likely that if UV-B is a factor in decomposition in the area where the solar farm is located, the presence of PV arrays could reduce k and lead to an increase in ecosystem carbon storage. An improved understanding in essential to better predict the full consequences of changes in UV-B, on carbon cycling.

Hypothesis (2): the effect of UV-B pre-treatment on decomposition will be dependent on leaf litter type

As anticipated, in our second hypothesis, litter decomposition and the effect of UV-B pre-treatment was strongly influenced by the leaf litter type. The greatest effect of UV-B pre-treatment was observed in the conifer litter, where decomposition was 36 % greater in the UV-B pre-treated litter, in comparison to the untreated litter. For the

broadleaf litter, this difference was 26 %, and there was no statistically significant difference in decomposition between the UV-B pre-treated and untreated grass litter. These differences are most likely the result of differences in litter quality. Over time, for all litter types, the difference in decomposition between the UV-B pre-treated and untreated litters decreased. The greatest difference was after the initial 2 month decomposition period for the conifer litter, where decomposition of the UV-B pre-treated litter was 300 % greater than in the untreated litter. At the same stage, the difference between the broadleaf and grass litter decomposition was 61 % and 16 % respectively, although the difference in grass litter decomposition was no statistically significant at this stage. A key difference in the leaf litters used in this experiment was the lignin content. The starting lignin content of the leaf litter negatively correlated with decomposition, however, was found to positively correlate to the effect of UV-B pre-treatment. The grass litter had the lowest lignin content and was found not to be affected by UV-B pre-treatment, whilst the conifer litter with the highest lignin content had the greatest difference in decomposition between the UV-B pre-treated and untreated litter. These findings are supported by other studies which have found that lignin may absorb UV-B wavelengths, changing the structure and/ or chemistry of the leaf litter which then facilitates decomposition by microbes (Austin and Ballaré, 2010; Brandt, King and Milchunas, 2007; Day, Zhang and Ruhland, 2007).

As a recalcitrant material, lignin is resistant to microbial decomposition, with only specialised biota, predominantly fungi, able to synthesise extracellular enzymes that breakdown these structures into bioavailable forms (Austin and Ballaré, 2010). Lignin is one of three important cell wall constituents, the others being hemicellulose and cellulose. Lignin is difficult to biodegrade and reduces the lability of other cell wall constituents. Therefore, the percentage composition of these three constituents for different leaf litter types is a key factor controlling rates of decomposition. High lignin content leaf litters generally inhibit microbial decomposition, enhancing carbon stores. Our results support this inhibitory role; however, our research also reveals an interaction between the lignin content of leaf litter and exposure to UV-B radiation: where high lignin content leaf litters can facilitate microbial decomposition under high levels of UV-B exposure. Many studies have observed this dual role of lignin in decomposition, due to the spectral absorption of lignin in the UV-B spectrum (Austin and Ballaré, 2010; Austin, Méndez and Ballaré, 2016; Brandt, King and Milchunas, 2007; Day, Zhang and

Ruhland, 2007). Our analysis of litter chemistry after 2 and 8 months of decomposition provides further evidence that it is the lignin fraction of the litter which is affected by UV-B exposure, with UV-B exposed litters having lower lignin contents. In addition, the loss of lignin from the litters may increase the lability of the hemicellulose and cellulose fractions of the leaf litter, as lignin can act as a barrier inhibiting the microbial breakdown of these cell constituents.

We found that k increased with decreasing N content. N content is known to be an important indicator of litter quality and a known control on litter decomposition. This study also revealed an interaction between initial leaf litter N and the effect of UV-B pre-treatment on decomposition. UV-B exposure was found to increase decomposition in litter with lower N contents i.e. grass had the highest N but the UV-B exposure had the lowest effect, whilst the conifer litter had the lowest N content, but the effect of UV-B exposure was greatest. This is in contrast to other studies which have found the opposite effect (Pan *et al.*, 2015): the contribution of solar radiation was found to correlate positively with the N content of leaf litter, as the microbes were not N limited and therefore decompose UV-B exposed leaf litters more efficiently leading to greater k. However, (Pan *et al.*, 2015) did not analyse the lignin content of the leaf litters used, unlike our study which has data for leaf litter N and lignin contents. A key difference in methodologies is the type of soil used during the decomposition period: we utilised a loamy and clayey soils whilst (Pan *et al.*, 2015) used a sandy soil. We can only speculate, however, sandy soils generally have lower N contents than loamy and clayey soils, and therefore the correlation observed between the high N and UV-B could be due to the substrate limiting microbial decomposition. It is, therefore, fair to assume that when microbes are not N limited it is the lignin content and not the N content of the leaf litter, which is the controlling variable in the effect of UV-B exposure on leaf litter decomposition.

The differential effects of UV-B on the decomposition of the three leaf litters has potential implications for decomposition processes dependent on ecosystem type. In ecosystems where decomposition rates are high due to labile leaf litters such as grasslands, the effect of changes in UV-B is likely to be minimal. However, in high diversity grasslands which are more likely contain a variety of plant functional types, there may be a shift to a more recalcitrant leaf litter type which is more sensitive in terms of decomposition to changes in UV-B. Therefore, if a high diversity grassland

was to experience a decrease in UV-B, through shading i.e. PV arrays on solar farms, decomposition rates in the shade may be suppressed, potentially leading to beneficial effects on ecosystem carbon storage. However, in woodland or forest ecosystems, where leaf litter inputs are recalcitrant due to high lignin and low N contents, changes in UV-B are likely to influence decomposition processes; with increasing UV-B receipts increasing decomposition rates and potentially decreasing carbon storage.

The aim of the experiment was to assess whether changes in UV-B could affect decomposition processes in mesic ecosystems. Ultimately, we achieved this and found that UV-B pre-exposure increased microbial decomposition for broadleaf and conifer leaf litter, although, no effects were present for grass litter. However, interpretation of the results presented in this paper into ecosystem processes should be done with an awareness that field conditions will be very different to the controlled laboratory conditions used in this experiment..

4.6 Conclusion

This study highlights the positive effect of UV-B exposure on litter decomposition. We suggest that this relationship should be accounted for in carbon cycle modelling as UV-B receipts are changing globally. With an increasing knowledge of UV-B controls on decomposition strategies may be developed which could reduce UV-B exposure subsequently enhancing soil carbon storage. On solar farms reductions in UV-B radiation receipts in the areas directly under the PV arrays may result in reduced k, which in turn could lead to greater soil carbon storage and a potential strategy to mitigate the impact of atmospheric carbon loading. In addition, our results reveal that variations in the effect of UV-B exposure are likely to be as a result of leaf litter chemistry, specifically lignin content. With predicted changes in intraspecific leaf chemistry and vegetation community composition under climate change, it is crucial to develop our knowledge of how decomposition processes may alter due to abiotic and biotic changes and assess the implications of this on the global carbon balance.

4.7 References

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5. Solar farm effects on productivity and vegetation properties

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Data referred to in this chapter can be viewed in full in the appendices presented at the end of this thesis.

5.1 Abstract

Land use change for solar energy is accelerating globally, with potential impacts on valuable ecosystem services. Solar farms are increasingly being installed on UK grasslands which under the pressures of global environmental change are increasingly important carbon stores. The known microclimatic changes under photovoltaic (PV) arrays are of a magnitude which could affect productivity and vegetation properties. These factors are important controls on grassland ecosystem functions and the services these functions provide. To investigate the effects of solar farms on vegetation, we measured above and below-ground biomass, species composition, vegetation height and leaf C:N in grasses, under PV arrays, in gaps between PV arrays and in control areas, at 17 sites across England and Wales. Our results show that under the PV arrays, above-ground biomass was lower and there was a change in carbon allocation, reflected by a decrease in the above:below-ground biomass ratio. Further, there were differences in species composition under the PV arrays, with fewer forbs and legumes and a graminoid dominated community in high diversity grasslands. PV arrays promoted in differences in leaf and root C:N, with the ratio of C:N increasing under the PV arrays for root C:N, and decreasing under the PV arrays for leaf C:N. In the gap areas, vegetation height was greatest and vegetation height under the PV arrays did not differ from the control areas. These results demonstrate the effect of solar farms on grassland vegetation, and our findings could be used to inform solar farm management strategies to maximise ecosystem service provision. We anticipate our analysis of the effects of PV arrays on

vegetation to be a starting point for further studies of solar farm effects on ecosystem services.

5.2 Introduction

Land use change is a major driver of global environmental change, which is predominantly caused by rapid human population growth, industrial development and the increasing demand for food, fibre, and energy (Machell *et al.*, 2015). Solar farms represent a significant land-use change as the number and size of solar farms continue to grow (DECC, 2012; DECC, 2014); however, solar farms have the potential to provide multiple ecosystem services i.e. energy, food, biodiversity and C storage (Hernandez *et al.*, 2013; Alexander *et al.*, 2015). Solar farms may have consequences for biodiversity, productivity and decomposition, influencing ecosystem service provision (Armstrong, Ostle and Whitaker, 2016; Montag, Parker and Clarkson, 2016). On solar farms, the move away from traditional agricultural practices in addition to the implementation of site appropriate biodiversity action plans has been found to have a positive impact on biodiversity (Hernandez *et al.*, 2013; Montag, Parker and Clarkson, 2016; Natural England, 2011). Many solar farm in the UK are grazed by sheep, providing another dual land use scenario (BRE, 2014). As part of the Solar Trade Associations "10 commitments of good practice" (Solar Trade Association, 2013), solar farm developers actively encourage multi-purpose land-use, through the implementation of land management plans which aim to support multiple ecosystem service production (biodiversity-food-energy) to maximise the benefits of this technology through careful land management strategies (BRE, 2014).

The presence of PV arrays has been shown to alter the microclimatic conditions both above and below ground (Armstrong, Ostle and Whitaker, 2016; Marrou *et al.*, 2013). The presence of PV arrays has been found to reduce the total photosynthetically active radiation reaching the grassland surface by 92 % and alter the proportion of radiation which is diffuse from 79 % in the control areas to 90 % under the PV arrays (Armstrong, Ostle and Whitaker, 2016). The temperature response to PV arrays varies within the location of the solar farm and seasonally. In the summer the presence of PV arrays has been found to suppress temperatures under the PV arrays by as much a 5.2 ^{o}C in comparison to the control and gap areas (Armstrong, Ostle and Whitaker, 2016). Whereas in the winter, temperatures are lowest in the gap areas, and warmest under the PV arrays (Armstrong, Ostle and Whitaker, 2016). The impact of solar farms on soil

moisture is unclear(Armstrong *et al.*, 2014; Armstrong, Ostle and Whitaker, 2016). The design of many solar arrays often causes the water to be funnelled through the gaps in the panels, potentially creating a mosaic of varying soil moisture (from high to low) under the panels. However, away from the areas where water is funnelled, the areas under the PV arrays in summer have much lower soil moisture content than the control and gap areas (Stott *et al.*, 2017). Whereas, in the winter the effect was reversed with the areas under the PV arrays having greater soil moisture content than the control and gap areas.

Through changes to radiation receipts, temperature and soil moisture, solar farms may alter primary productivity and C allocation. Total biomass and the ratio of above:below ground biomass are good indicators of these processes respectively. Primary production is primarily controlled by climatic factors. Primary production tends to increase with radiation receipts, although thresholds exist, where increases in radiation recipients suppress photosynthetic processes (Mercado *et al.*, 2009b). However, recent studies have demonstrated that photosynthesis can be more efficient under diffuse light conditions, as found under the PV arrays, due to a more homogenous distribution of light in the canopy (Mercado *et al.*, 2009b; Kanniah *et al.*, 2012). Further, temperature is a major control on productivity processes, with increasing temperatures generally increasing the rate at which biotic processes occur (Bradford, 2013; Davidson and Janssens, 2006; Day, Ruhland and Xiong, 2008). However, the effects of changes in temperature due to the presence of PV arrays, on grassland functions are uncertain, due to contrasting seasonal effects (Armstrong, Ostle and Whitaker, 2016). Primary productivity tends to increase with soil moisture, to a threshold, where soils become waterlogged and root exchange processes are impeded (Klumpp *et al.*, 2011). However, the extent of variation in soil moisture may be affected by factors such as root infiltration, soil type, and soil structure (Schoonover and Crim, 2015). It is uncertain how changes in soil moisture caused by solar farms, and interactions with temperature and radiation receipts may impact primary productivity and evapotranspiration. In addition to effects on biomass production, changes in microclimate may affect plant physiology, specifically growth height. Shading is known to affect the height or vegetation, which can affect the bushiness

In addition to changes in primary productivity, the allocation of photo-assimilates may also be affected by the microclimatic changes of solar farms (Armstrong *et al.*, 2014).

Carbon allocation strongly influences plant and soil processes, including above and below ground biomass production, which in turn could affect ecosystem services (Bahn *et al.*, 2013). Carbon allocation dynamics in ecosystems and their responses to environmental changes are still relatively poorly understood (Cao and Woodward, 1998; Poorter and Nagel, 2000). However, studies have shown that a variety of factors influence the allocation of carbon: a process which has been suggested to be strongly sink driven, with photosynthates being preferentially transferred to tissues with the highest demand i.e. in shaded conditions above ground plant growth would be preferential to maximise leaf area (Pugh *et al.*, 2016; Sevanto, Dickman and Way, 2015; Lambers, 1998). Whereas, under reduced nutrient and/or water supply, they invest more C to the root system (Schmitt, Pausch and Kuzyakov, 2013; Poorter and Nagel, 2000; Kobe, Iyer and Walters, 2010). Sustained shading (as would be present in static solar farms) has been found to result in decreased carbohydrate pools above- but not below-ground, and reduced leaf respiration more strongly than root respiration (Bahn *et al.*, 2013), which could be observable in the above: below ground biomass.

The effects of changes in microclimate on above ground biomass may be dependent on plant functional type. Some plant functional types are more resistant to drought, temperature change and reduced radiation receipts, than others. Specifically, many legumes and forbs are known to be intolerant to shading, high temperatures and drought, as under these conditions graminoids have an interspecific advantage (Boeck *et al.*, 2008; Fry *et al.*, 2013). This could potentially lead to changes in species composition overtime, with the loss of plant functional types which are key to ecosystem processes i.e. the loss of legumes may affect N cycling (Steinbeiss *et al.*, 2008; Conti and Díaz, 2013; Wood, Cavaleri and Reed, 2012). In addition to changes in the vegetation community, N cycling may be affected directly by the microclimatic changes. Leaf N has been found to increase under shaded conditions, due to the translocation of N to photosynthetic tissue (Ma *et al.*, 2015). Changes to N and C cycling could affect the C:N ratio, a key factor affecting microbial nutrient cycling and subsequently productivity processes (Nie *et al.*, 2015)

This study aims to assess how solar farms in the UK affect vegetation. Surveys were conducted and samples taken from areas under the PV arrays, the gap areas in between two rows of PV arrays, in addition to field control samples.

With this study we investigate the following hypotheses on UK solar farms:

1. Under the PV arrays, above ground biomass will be reduced, with greater allocation to root biomass. However, the extent of the differences will be governed by site specific factors such as climate, soil type and management practices.
2. Under the PV arrays, the presence of forbs and legumes will be reduced, with the vegetation community dominated by grasses.
3. Under the PV arrays, there will be an increase in vegetation height.
4. C:N ratios in the leaf, root and soil will be affected by the presence of PV arrays.

5.3 Methods

We tested our hypotheses by surveying and sampling at 17 solar farms across England and Wales in June 2016. All solar farms were at least 6 months old and had established vegetation. Information regarding local climate, soil type and previous and current land management of each individual solar farm was collated (using data from the Met Office, Soilscapes and land owners respectively) (Table 19).

Table 19. Location, soil type, climate, land management, age and size of solar farms used in this study. Soil type is based on Soilscapes classification (CSAI, 2016), and mean annual temperature and precipitation acquired from the Met Office 1981-2010 averages (MetOffice, 2016).

Solar farm	County	Soil type	Mean annual precipitation (mm year^{-1})	Mean annual temperature (^{o}C)	Previous land management	Current land management	Year established	Size
1	Gloucestershire	Lime-rich loamy and clayey soils with impeded drainage	644	9.4	Sheep grazed grassland	Partial grazing, high diversity vegetation	2014	3.8 MW
2	Oxfordshire	Shallow lime-rich soils over chalk or limestone	658	9.3	Arable	Partial grazing, high diversity vegetation	2011	5 MW
3	Wiltshire	Slowly permeable seasonally wet slightly acid	663	9.4	Sheep grazed grassland	Partial grazing, high diversity vegetation	2015	4.7 MW

		but base-rich loamy and clayey soils						
4	Wiltshire	Slowly permeable seasonally wet slightly acid but base-rich loamy and clayey soils	741	9.3	Sheep grazed grassland	Partial grazing, high diversity vegetation	2015	4.9 MW
5	Devon	Slowly permeable seasonally wet acid loamy and clayey soils	1120	10.0	Cow grazed and summer grass crop production	High diversity vegetation	2011	250 kW
6	Cornwall	Freely draining slightly acid loamy soils	932	10.7	Sheep grazed grassland	Sheep grazed grassland	2014	1.8 MW

7	Cornwall	Freely draining acid loamy soils over rock	976	10.7	Cow grazed and summer grass crop production	Sheep grazed grassland	2013	5 MW
8	Cornwall	Freely draining acid loamy soils over rock	1049	10.1	Sheep grazed grassland	Partial grazing, high diversity vegetation	*2015*	*8.4 MW*
9	Cornwall	Freely draining acid loamy soils over rock	1049	10.1	Sheep grazed grassland	Partial grazing, high diversity vegetation	*2015*	*8.4 MW*
10	Cornwall	Freely draining slightly acid loamy soils	1023	10.4	Cow grazed and summer grass crop production	Sheep grazed grassland	2011	12 MW
11	Dorset	Naturally wet very acid	792	10.0	Sheep grazed grassland, with extensive ditches to	Partial grazing, high diversity vegetation	2014	5 MW

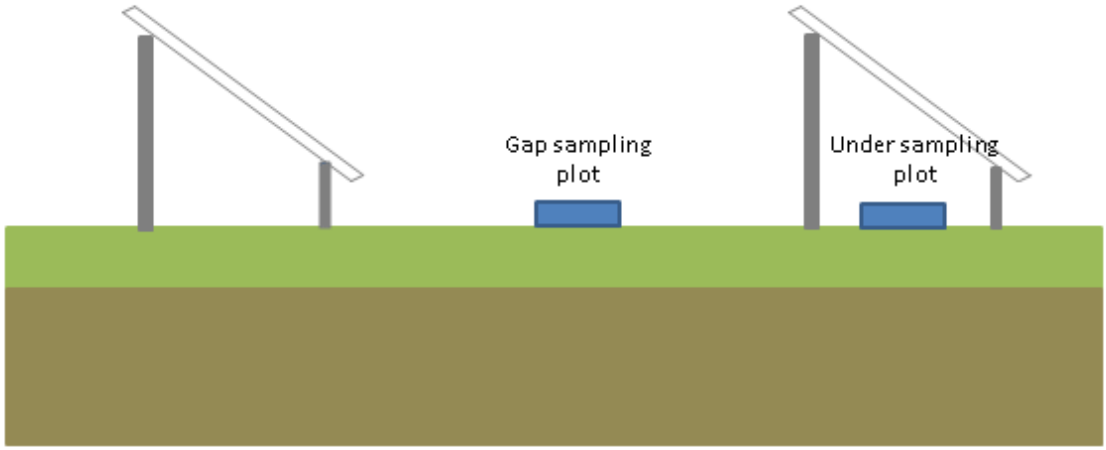
		sandy and loamy soils			improve drainage of naturally very wet land			
12	Dorset	Naturally wet very acid sandy and loamy soils	792	10.0	Mixture of forestry, grassland and race track	Partial grazing, high diversity vegetation	2015	5.8 MW
13	Hampshire	Loamy soils with naturally high groundwater	785	10.3	Grassland restored from landfill	Partial grazing, high diversity vegetation	2015	2.4 MW
14	Hampshire	Freely draining very acid sandy and loamy soils	785	9.8	Cow grazed and summer grass crop production-manure and artificial fertiliser inputs	Partial grazing, high diversity vegetation	2015	5 MW
15	Anglesey	Slowly permeable seasonally wet	925	9.7	Sheep grazed grassland	Sheep grazed grassland	2014	12.6 MW

		acid loamy and clayey soils						
16	Cheshire	Freely draining slightly acid sandy soils	754	9.2	Cow grazed and summer grass crop production-manure and artificial fertiliser inputs	Sheep grazed grassland	2015	14 MW
17	Shropshire	Freely draining slightly acid loamy soils	707	8.8	Cow grazed and summer grass crop production-manure and artificial fertiliser inputs	Sheep grazed grassland	2014	5 MW

5.3.1 Experimental design

At all of the 17 solar farms, five plots (1 m x 1 m) were randomly selected within each of the three designated treatment areas: under the PV arrays, in the gaps between the PV arrays and in the control area, giving a total of 15 plots per site. Control measurements were taken away from the PV arrays, whilst still remaining in the enclosed area of the solar farm to ensure management strategies were similar. Gap measurements were taken directly in the centre of the area between two rows of arrays (Figure 17). Under the PV arrays, measurements were taken in the centre of the panel away from areas where water may be channelled through gaps between individual panels. At all of the sampling locations, we identified all of the distinct species present, using a 1 x 1 m open quadrat (Klimek *et al.*, 2007) and collected soil cores for below ground biomass analysis, and soil and grass samples for C:N analysis (leaf, root and soil). In addition, at sites which had not been grazed in the three months prior to sampling we also measured total above ground biomass, above ground biomass by plant functional type and vegetation height.

Figure 17. Sampling location under the PV arrays (panel edges) and in the gap area between two rows of PV arrays.



5.3.2 Biomass- above and below and by plant functional type

Above-ground biomass samples were collected from the 11 solar farms that had been not been grazed (Table 19). A 10 x 10 cm area from inside the designated plot areas,

was randomly selected and all the vegetation in this area was cut to the level of the soil. The cut vegetation was field dried in paper bags and stored in cool boxes. On return to the laboratory, the samples were oven dried at 60 ^{o}C to a constant weight. The samples were weighed to give values for total above-ground biomass. In addition, the biomass samples were separated into plant functional types, and above-ground biomass values were attained for grasses, legumes and forbs. After the above-ground vegetation had been removed, a 10 x 10 x 10 cm soil core was extracted from the same area. The soil samples were stored in cool boxes with ice packs until laboratory processing began. The intact soil cores were first soaked in a water bath of 1 % TWEEN 20 (Sigma-Aldrich) solution for 10 minutes and then sonicated in the TWEEN water bath at 70 % for 5 minutes. These stages removed the majority of the soil from around the roots. Any remaining soil was removed through hand root washing. The roots were then oven dried at 60 ^{o}C to a constant weight to give values for below-ground plant biomass. In addition, we calculated the above to below-ground plant biomass ratio(Ma *et al.*, 2015).

5.3.3 Vegetation Height

Within the designated plot area, a 50 x 50 cm open quadrat was randomly placed. In order to assess vegetation height, the height of the vegetation at each corner of the quadrat and at the centre point of the quadrat was measured.

5.3.4 Leaf, root and soil C:N

To determine leaf, root and soil total C and N, a 30 mg subsample (from the additional soil cores and vegetation samples- soil was sieved and roots were hand washed) of oven dried (60 °C) and ball mill ground (three minutes) material, was analysed using a LECO Truspec CN Analyser (LECO, USA) and C:N was calculated (Carter, 2007). For the leaf and root C:N analysis samples were taken of *festuca rubra* at 15 sites, and *poa annua* at the 2 sites where *festuca rubra* was absent.

5.3.5 Statistical analysis

All data were checked for normality and p values < 0.05 were deemed significant. All analysis was conducted in R Studio (Rstudio Team 2015). To test hypothesis 1, the influence of treatment (control/gap/under) and site on above, below and above:below ground biomass was tested using linear-mixed effects models with lme4 and lmerTest (to derive p values) packages in the R statistical program. If site was a significant factor affecting above, below or above:below ground biomass, additional analysis was

conducted where linear mixed effects models were developed which assessed the variables against treatment and site specific factors i.e. mean annual precipitation, mean annual temperature, soil type (categorised as either acidic or alkaline) and year established. For hypothesis 2, the effects and interactions between treatment and site on the above ground biomass % by plant functional type were tested using linear-mixed effects models. Hypothesis 3 used linear mixed effects model analysis to determine the effects and interactions of treatment and site on vegetation height. Site was found to be a significant factor, so site specific factors i.e. soil type, mean annual temperature, precipitation, and year the solar farm was established were tested using linear mixed effects models against treatment on vegetation height. Finally, to test hypothesis 4, we used linear mixed effects model analysis for the effects and interactions of treatment and site on leaf C:N, root C:N and soil C:N. For all the linear mixed effects models R values were used to gauge effects size of variables such as treatment (under/gap/control) and site (Fairchild *et al.*, 2009; Trafimow, 2015), where < 0.10: trivial, 0.10 - 0.30: small to medium, 0.30 -0 .50: medium to large, > 0.50: large to very large (Cohen, 1977). Model assumptions were scrutinised using fitted values versus residuals plots and QQ plots, and post hoc comparisons were made using Tukey's (Zuur, 2010). Coefficients of fixed effects for the model were used to determine effects size.

5.4 Results

The following sections outline how biomass, vegetation community composition, vegetation height and plant and soil C:N ratios are affected by solar farms.

5.4.1 Biomass

Above ground biomass was on average 27 % lower under the PV arrays in comparison to the control and gap areas, however, the extent of the differences varied by site (Table 20). Treatment was the most influential factor, with an R^2 value of 0.24. Precipitation, soil type and the age of the solar farm explained the variability in the differences in above ground biomass between the treatments (Table 21). At the driest sites (Table 19), there was no difference in above ground biomass in control, gap and under plots. However, when precipitation exceeded 660 mm per year, above ground biomass under the panels was lower than in the control and gap area (Table 21). In acidic soils, above ground biomass decreased by a greater extent between control/gap and under areas, in comparison to the decrease in alkaline soils (Table 21). Solar farms established in the last year had the largest differences in above ground biomass (Table 21). As the age of

the solar farm increases, the difference between above-ground biomass in control, gap and under areas decreased (Table 21).

We found that below ground biomass was only affected by the presence of PV arrays at some sites. Specifically, in alkaline soils, under the PV arrays there was a decrease in below ground biomass, however, in acidic soils, below ground biomass did not significantly change between the control, gap and under areas. At sites with the lowest precipitation levels, below ground biomass was significantly affected by the presence of PV arrays, with below ground biomass under the PV arrays greater than in the control and gap areas. However, when precipitation levels exceeded 660 mm per year (Table 19), there was no difference between the under, gap and control areas. The age of the solar farm, was found to influence the effect of the presence of PV arrays on below ground biomass. At the newest sites, under the PV arrays below ground biomass was significantly lower than in the control and gap areas. Whilst at sites which were 2 or 3 years old, below ground biomass was found not to be affected by the presence of PV arrays. At the oldest sites (5 years old), the relationship had inversed, and below ground biomass was found to be higher under the PV arrays than in the control and gap areas.

Treatment (control/gap/under) was found to affect the ratio of above to below-ground biomass in solar energy farms (table 20). Under the PV arrays, there was a greater proportion of below-ground biomass in comparison to above-ground biomass levels. The ratio of above: below ground biomass did not differ between sites, and there was no interaction between site and treatment for the ratio of above to below-ground biomass (Table 20).

Table 20. Linear mixed effects model analysis for the effects and interactions of treatment and site on above-ground biomass, below-ground biomass and above:below-ground biomass. R is used here to gauge effects size, where r = 0.1 = small; r = 0.3 = medium; r = 0.5 = large

	Df	*F*	*P*	*R*
Above-ground Biomass				
Treatment	2	20.8	< 0.05	0.48

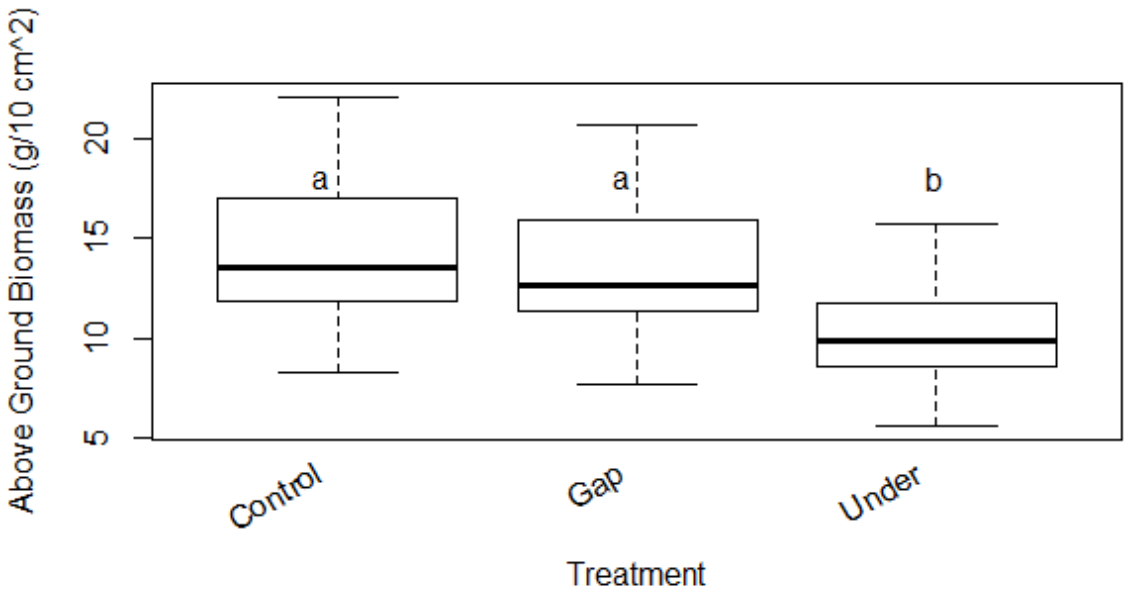
Site	1	10.8	< 0.05	0.24
Treatment*Site	2	3.4	< 0.05	0.58
Below-ground Biomass				
Treatment	2	0.3	> 0.05	0.10
Site	1	5.3	< 0.05	0.20
Treatment*Site	2	3.5	< 0.05	0.32
Above:Below-ground Biomass				
Treatment	2	53.0	< 0.05	0.69
Site	1	0.3	> 0.05	0.10
Treatment*Site	2	0.2	> 0.05	0.69

Table 21. Linear mixed effects model analysis for the effects and interactions of treatment, year of solar farm establishment, precipitation, mean annual temperature and soil type on above and below-ground biomass.

	Df	*F*	*P*
Above-ground biomass			
Treatment	2	66.3	< 0.05
Soil type	1	3.6	> 0.05
Temperature	1	27.1	< 0.05
Precipitation	1	58.8	< 0.05
Year	1	19.2	< 0.05
Treatment* Soil type	2	6.7	< 0.05
Treatment* Temperature	2	2.5	> 0.05
Treatment* Precipitation	2	4.8	< 0.05
Treatment* Year	2	30.1	< 0.05
Below-ground Biomass			

Treatment	2	0.7	> 0.05
Soil type	1	9.5	< 0.05
Temperature	1	29.6	< 0.05
Precipitation	1	13.2	< 0.05
Year	1	6.1	< 0.05
Treatment* Soil type	2	30.4	< 0.05
Treatment* Temperature	2	0.7	> 0.05
Treatment* Precipitation	2	6.0	< 0.05
Treatment* Year	2	18.8	< 0.05

A



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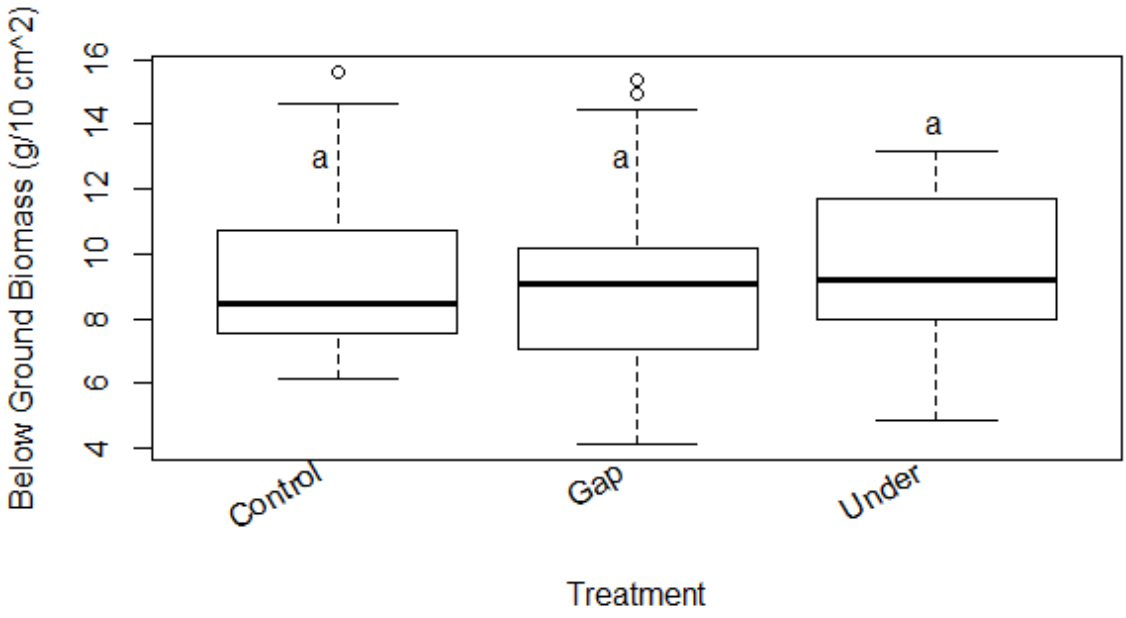

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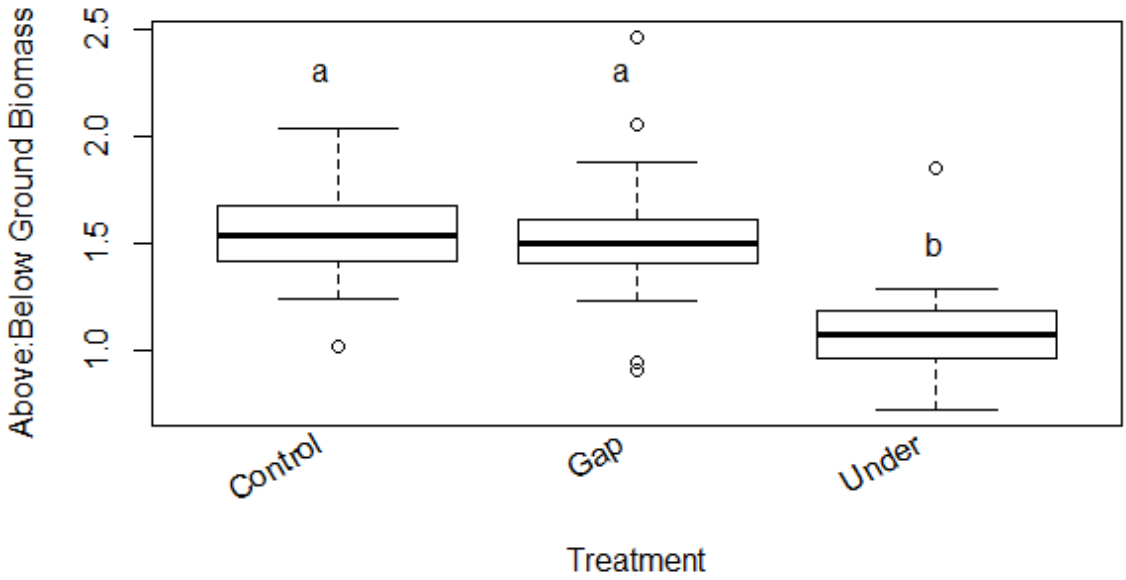
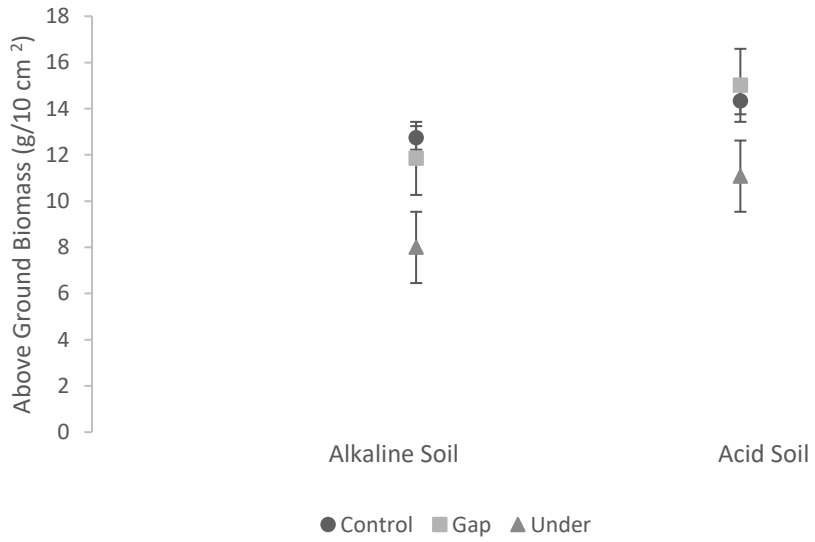


Figure 18. Differences in (A) above-ground biomass, (B) below-ground biomass and (C) above:below-ground biomass between the control areas, gap areas and under the PV arrays. The top and bottom lines of the rectangle are the 25th and 75th quartiles and

*the midline represents the median. The error bars represent the 25th and 75th quartile -/+ 1.5*interquartile range, and circles out-lying data. The box plots with different letters are significantly different from one another, n = 165 for above and above:below ground biomass (samples only included from ungrazed sites), n = 255 for below ground biomass (samples included from both grazed and ungrazed sites).*



A

B

C

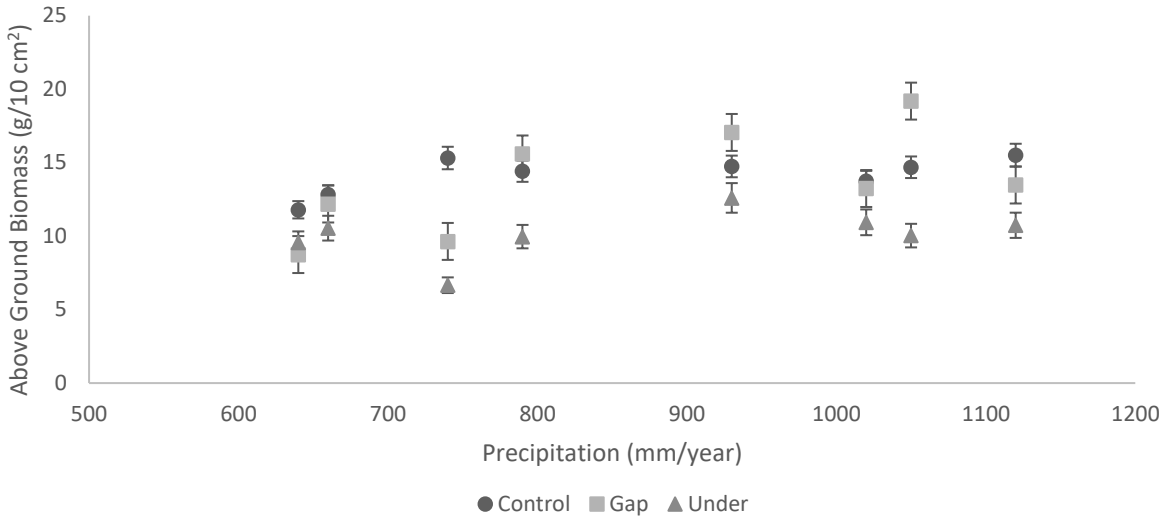


Figure 19. Interaction plots between treatment and (a) soil type (categorised as either acidic or alkaline), (b) year the solar farm was established, and (c) annual precipitation (to the nearest 10 mm), on above ground biomass data.

5.4.2 Vegetation community composition

There was a difference observed in species presence under the PV arrays in comparison to the control and gap areas (Table 22). Under the PV arrays there was a notable absence of legumes and forbs, with the exceptions of *Achillea millefolium* and *Onopordum acanthium* whose presence under the PV arrays was greater than in the control and gap areas, however, the presence of grass species was unaffected by the presence of PV arrays.

Table 22. Species presence survey in the control, gap and under the PV arrays at 17 solar farms in England and wales.

Species	Presence at number of sites		
Grasses	Control	Gap	Under
Brachypodium sylvaticum	3	2	1
Agrostis capiliaris	8	8	8

Anthoxanthum oderatum	6	6	6
Festuca rubra	15	15	15
Festuca ovina	5	5	5
Poa annua	8	8	8
Poa trivialis	13	13	13
Phleum pratense	9	6	7
Legumes			
Trifolium repens	13	11	1
Lotus corniculatus	5	7	0
Trifolium pratense	5	3	0
Onobrychis viciifolia	5	4	0
Forbs			
Achillea millefolium	2	3	6
Leucanthemum vulgare	4	3	0
Plantago lanceolate	5	6	2
Ranunculus acris	9	11	4
Ranunculus repens	14	13	5
Onopordum acanthium	3	3	6
Taraxacum officinale	3	4	2
Anethum graveolens	2	2	0

Campanula glomerate	2	1	0
Rubus fruiticosus	0	0	1
Primula veris	3	3	0
Rumex acetosella	4	2	0
Rumex acetosa	7	7	2

Biomass by plant functional type (graminoid, legume and forb) was found to individually vary by treatment (Table 22) (Table 23). Graminoid biomass was greatest under the PV arrays, accounting for 87 % of the total biomass (Figure 20). In the control and gap areas, graminoids accounted for 52 % and 59 % of total biomass respectively. Legume biomass was lowest under the PV arrays, accounting for 2 % of total biomass, whereas in the control and gap areas legumes accounted for 11 % and 9 % respectively. Forbs on average made up 36 % and 31 % of total biomass in the control and gap areas. Under the PV arrays, this fell by 72 % to make up only 10 % of above-ground biomass.

Table 23. Linear mixed effects model analysis for the effects and interactions of treatment and site above ground biomass (%) by plant functional type. R is used here to gauge effects size, where r = 0.1 = small; r = 0.3 = medium; r = 0.5 = large

	Df	*F*	*P*	*R*
% Graminoids				
Treatment	2	22.5	< 0.05	0.73
Site	1	2.1	> 0.05	0.14
Treatment*Site	2	1.3	> 0.05	0.77
% Legumes				
Treatment	2	6.0	< 0.05	0.51
Site	1	0.0	> 0.05	0.10

Treatment*Site	2	0.1	> 0.05	0.52
% Forbs				
Treatment	2	13.3	< 0.05	0.65
Site	1	2.1	> 0.05	0.17
Treatment*Site	2	1.1	> 0.05	0.69

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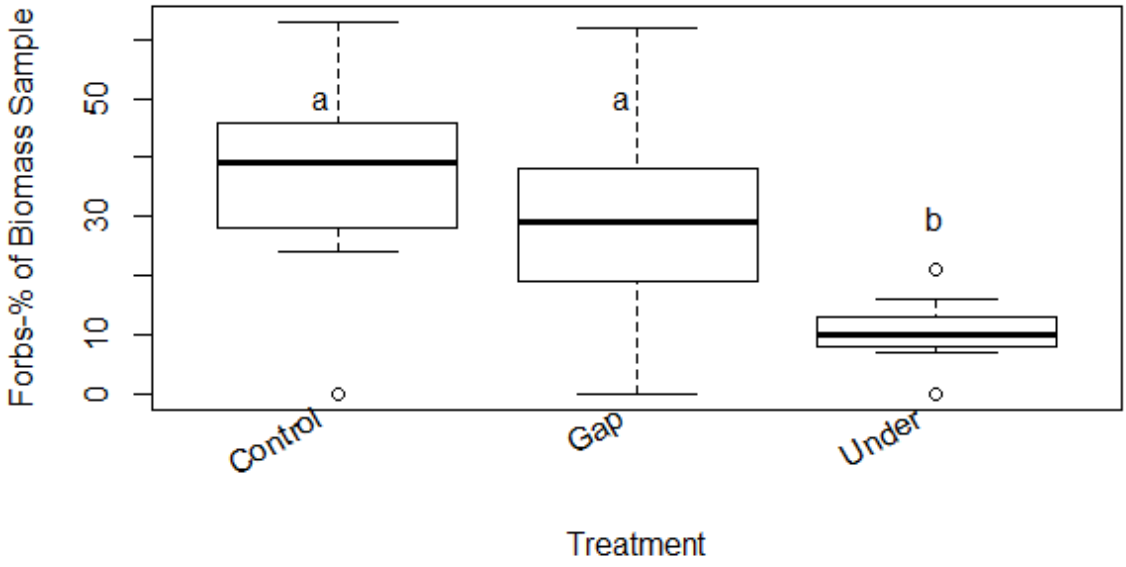

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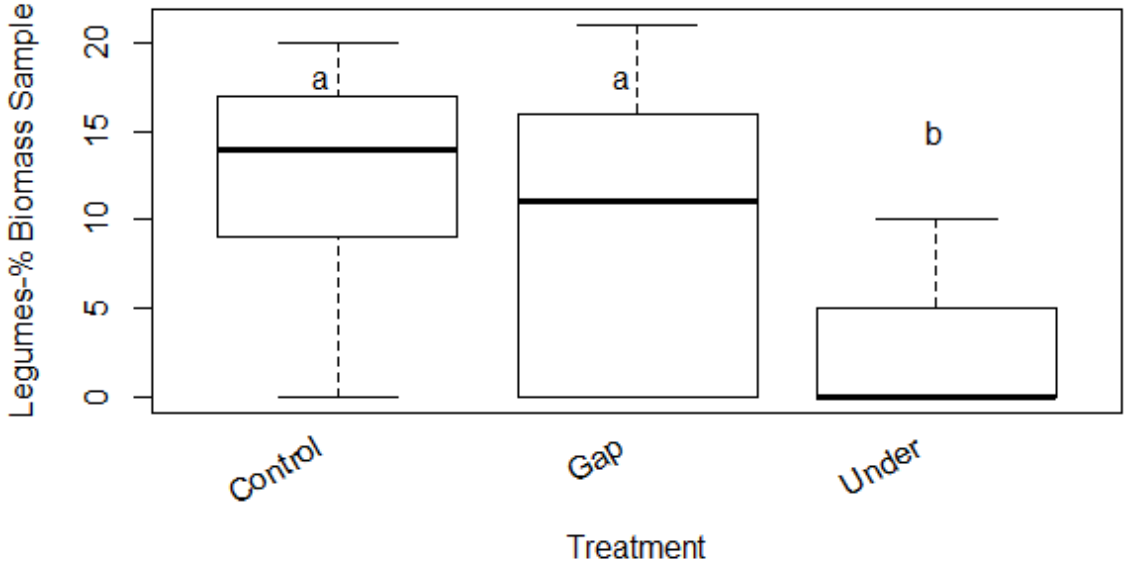

Legumes-% Biomass Sample
0
5
10
15
20
a
a
b
Control
Gap
Under
Treatment

C

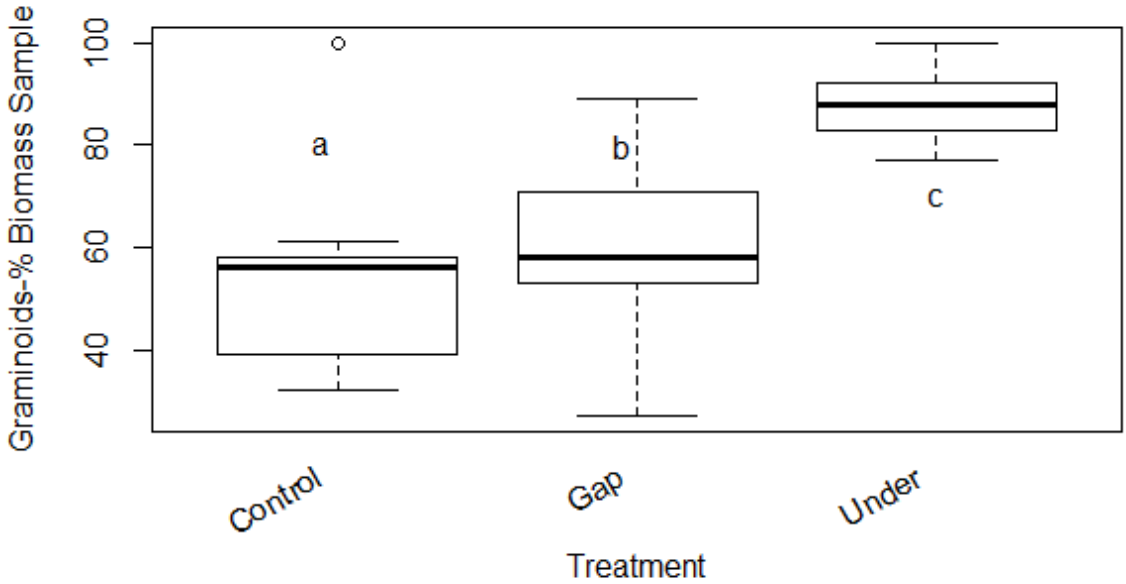

Graminoids-% Biomass Sample
40
60
80
100
a
b
c
Control
Gap
Under
Treatment

*Figure 20. Differences in (A) forb, (B) legume and (C) graminoid vegetation community composition between the control areas, gap areas and under the PV arrays. The top and bottom lines of the rectangle are the 25th and 75th quartiles and the midline represents the median. The error bars represent the 25th and 75th quartile -/+ 1.5*interquartile range, and circles out-lying data. The box plots with different letters are significantly different from one another, n = 165.*

5.4.3 Vegetation Height

Vegetation height varied by treatment and site (Table 24) (Figure 21). Of these two variables treatment had the largest effects size with an R value of 0.47, whereas site had an R value of 0.17. In the gap areas, vegetation height was greatest and lowest under the PV arrays (Figure 19). Between the control and gap areas there was a 17 % increase in vegetation height, and between the control and under areas there was a 21 % decrease in vegetation height. Soil type, and precipitation, explained the variability in the differences in vegetation height between the treatments (Table 25). In alkaline soils, the presence of PV arrays resulted in a greater decrease in vegetation height, than in acidic soils. Additionally, the increase in gap vegetation height was greater at sites with alkaline soils than at sites with acidic soils. When annual precipitation levels exceeded 1000 mm, the decrease in vegetation height under the PV arrays was much smaller than on sites where annual precipitation was < 1000 mm (Table 19)(Table 25).

Table 24. Linear mixed effects model analysis for the effects and interactions of treatment and site on vegetation height. R is used here to gauge effects size, where r = 0.1 = small; r = 0.3 = medium; r = 0.5 = large

	Df	F	P	R
Treatment	2	20.6	< 0.05	0.47
Site	1	6.0	< 0.05	0.17
Treatment*Site	2	3.5	< 0.05	0.51

Table 25. Linear mixed effects model analysis for the effects and interactions of treatment, soil type, mean annual temperature, precipitation, and year the solar farm was established on vegetation height.

	Df	F	P
Treatment	2	32.7	< 0.05
Soil type	1	0.3	> 0.05
Temperature	1	11.2	< 0.05
Precipitation	1	9.0	< 0.05
Year	1	0.2	> 0.05
Treatment* Soil type	2	5.5	< 0.05
Treatment* Temperature	2	0.9	> 0.05
Treatment* Precipitation	2	5.2	< 0.05
Treatment* Year	2	0.3	> 0.05

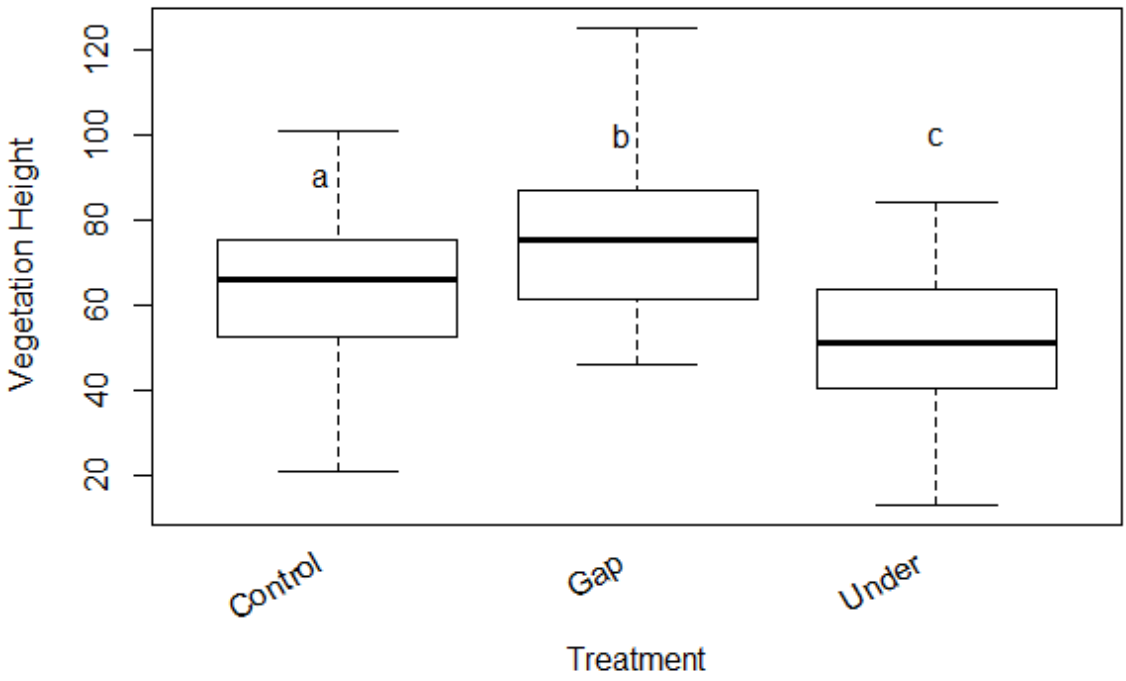
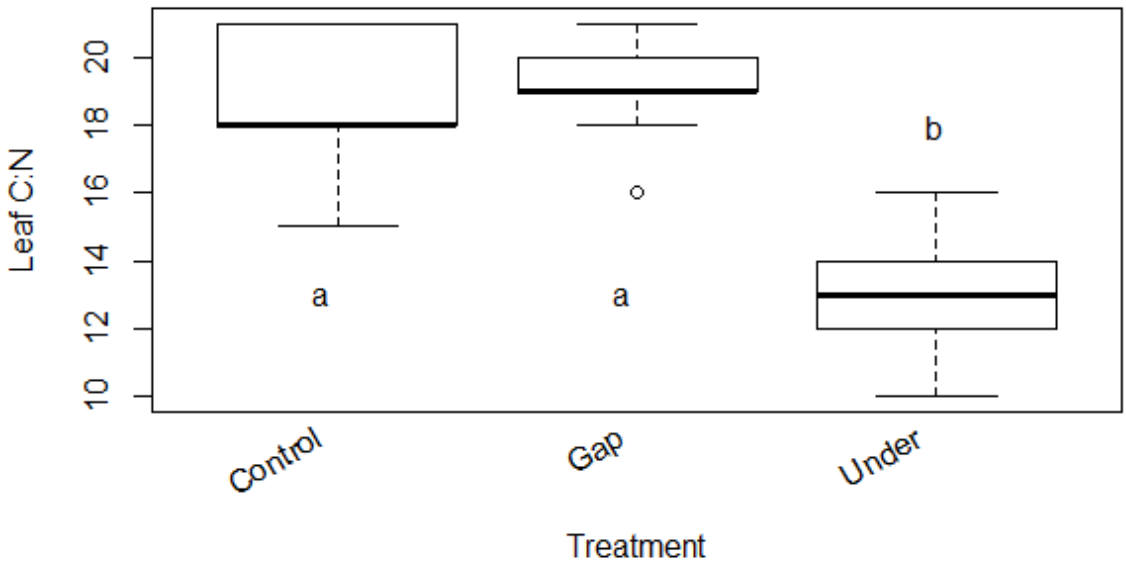


*Figure 21. Differences in vegetation height between the control areas, gap areas and under the PV arrays. The top and bottom lines of the rectangle are the 25th and 75th quartiles and the midline represents the median. The error bars represent the 25th and 75th quartile -/+ 1.5*interquartile range, and circles out-lying data. The box plots with different letters are significantly different from one another, n = 165.*

5.4.4 C:N

There was no significant difference between the C:N ratios of the two grass species selected, therefore further statistical analysis was conducted on the whole dataset. Grass leaf C:N varied significantly by treatment and site (Table 26)(Figure 22). Leaf C:N ratios for the dominant grass species was lower under the PV arrays, compared to the gap and control areas. On average the leaf C:N ratio was 32 % lower under the PV arrays, than in the control and gap areas. Root C:N varied by treatment and site (Table 26). Root C:N ratio was 19 % greater under the PV arrays in comparison to the control and gap areas. Soil C:N did not vary in response to the presence of PV arrays.


Leaf C:N
10
12
14
16
18
20
a
a
b
Control
Gap
Under
Treatment

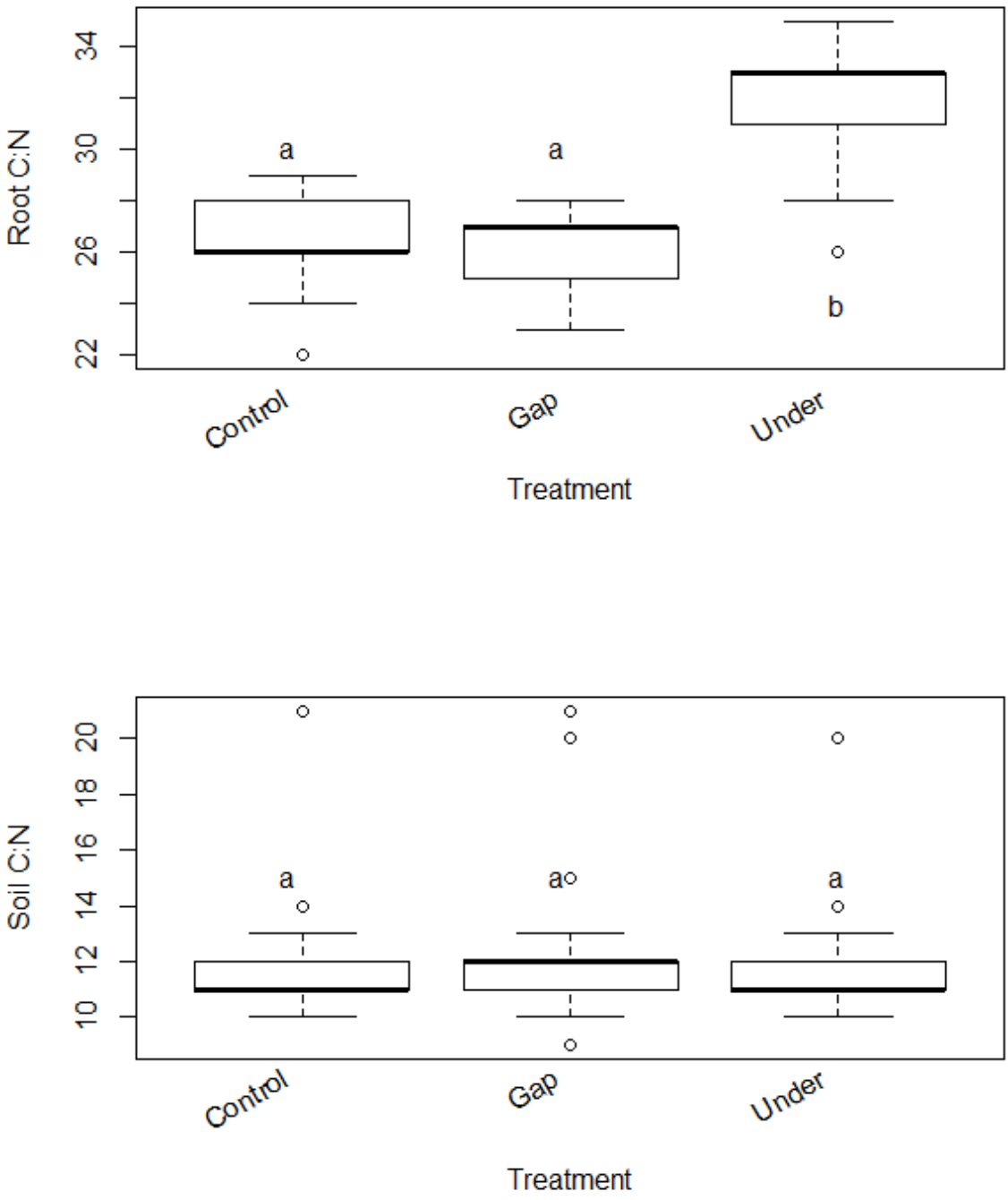


*Figure 22. Soil C:N and leaf and root C:N for grasses between the control areas, gap areas and under the PV arrays. The top and bottom lines of the rectangle are the 25th and 75th quartiles and the midline represents the median. The error bars represent the 25th and 75th quartile -/+ 1.5*interquartile range, and circles out-lying data. The box plots with different letters are significantly different from one another, n = 255.*

Table 26. Linear mixed effects model analysis for the effects and interactions of treatment and site on leaf C:N, root C:N and soil C:N. R is used here to gauge effects size, where r = 0.1 = small; r = 0.3 = medium; r = 0.5 = large

	Df	F	P	R
Leaf C:N				
Treatment	2	73.9	< 0.05	0.85
Site	1	4.2	< 0.05	0.14
Treatment*Site	2	0.1	> 0.05	0.86
Root C:N				
Treatment	2	45.2	< 0.05	0.78
Site	1	5.8	< 0.05	0.20
Treatment*Site	2	0.6	> 0.05	0.81
Soil C:N				
Treatment	2	0.1	> 0.05	0.20
Site	1	1.9	> 0.05	0.20
Treatment*Site	2	0.1	> 0.05	0.20

5.5 Discussion

In this paper, we present the first multi-site study of UK solar farm impacts on ecosystem functions. Given the potential for solar farms in the UK and limited data, there is clearly a need to better understand the impacts associated with establishing solar farms on biomass production and allocation, vegetation communities, vegetation height and C and N cycling. To address this we hypothesised:-

1. Under the PV arrays, above ground biomass will be reduced, with greater allocation to root biomass, due to shading and reduced growing season temperatures. However, the extent of the differences will be governed by site specific factors such as climate, soil type and management practices.
2. Under the PV arrays, the presence of forbs and legumes will be reduced, with the vegetation community dominated by grasses, due to shading.
3. Under the PV arrays, there will be an increase in vegetation height, due to the effect of shading. The magnitude of the effect of shading will interact with site specific factors.
4. C:N ratios in the leaf, root and soil will be affected by the presence of PV arrays.

5.5.1 Solar farm effects on above and below-ground biomass

Our first hypothesis was partially supported. We found that above-ground biomass, and above: below ground biomass was affected by the presence of PV arrays, however, below ground biomass did not vary in solar farms. Above ground biomass was lowest under the PV arrays and that above-ground biomass in the gap areas did not differ from the control areas. The reduction in above-ground biomass under the panels reflects the decrease in growing season temperature and radiation receipts which under the PV arrays are of a magnitude known to affect plant productivity.

The presence of PV arrays interacted with soil type, precipitation and solar farm age to affect above ground biomass. Above ground biomass in alkaline soils was more resistant to the microclimatic changes imposed by PV arrays than above ground biomass at the sites with acid soils. This difference may be attributed to the different plant communities which inhabit these soil types. Our site species identification survey, identified differences in communities by soil type, with alkaline soils generally supporting more species diverse and forb rich communities. High diversity grasslands, which are most commonly associated with alkaline soils in the UK (Janssens *et al.*, 1998), are generally more resilient to climate change than low diversity grasslands (Joseph *et al.,* 2012). High diversity in grasslands means that there is a greater the chance of having species ready to take over critical functions if there is a change in climate (Craine et al. 2012). Aside from vegetation communities, soil pH also affects a number of other factors which are known to influence productivity. In alkaline soils macronutrient availability increases, which can have positive impacts on plant growth. Subsequently, difference in nutrient availability and toxicity between different soil

types may be a factor controlling the response of plant growth to changes in microclimate on solar farms. At the sites with the lowest levels of precipitation, above ground biomass was unaffected by the presence of PV arrays. When annual precipitation exceeded 660 mm per annum, above ground biomass in the areas under the PV arrays was lower than in the control and gap areas. It is possible that the vegetation community on the solar farms with the lowest precipitation levels are already adapted to drought conditions and that the drought effect of PV arrays is not great enough to cause differences in productivity on these sites. Our data from the site species survey support this: at the sites with the lowest precipitation levels, drought tolerant species were found, such as *festuca rubra* and *brachypodium sylvaticum (Grime, 1988)*. The age of the solar farm also affected the response of above ground biomass to the presence of PV arrays. At the oldest solar farms we surveyed, we found that the presence of PV arrays did not affect above ground biomass. In fact, there was a linear relationship between the age of the solar farm and the effect of the PV arrays on above ground biomass, with the largest differences observed at the most recently established sites. This may be as a result of vegetation community adaptation to the microclimatic changes imposed by PV arrays over time. Drought tolerant plants may become more established in the areas under the PV arrays over time, and their increased growth may compensate for the reduced growth of non-drought tolerant species which are more dominant earlier on due to faster growth cycles.

The presence of PV arrays on solar farms was found not to affect total below-ground biomass. However, interestingly our data showed that the ratio of above:below-ground biomass was lower under the PV arrays, and that above:below-ground biomass in the gap areas did not differ from the control areas. These results reveal that there is a change in the allocation of photo-assimilates to promote growth below-ground under the PV arrays. The root biomass under the panels often contained large tap roots in addition to an increase in grass roots. These results are supported by studies which have found that sustained shading decreased carbohydrate pools above- but not below-ground, and reduced leaf respiration more strongly than root respiration (Bahn *et al.,* 2013). The change in root biomass may also be due to the change in vegetation type, with grasses in general having large root systems than forbs, and the forbs that are present under the PV arrays such as *achillea millefolium* and *onopordum acanthium* are known to produce large tap roots (Grime, 1988). PV arrays have been found to decrease PAR receipts by

92 % in UK solar farms (Armstrong, Ostle and Whitaker, 2016). Previously, it was assumed that carbon allocation sinks were demand driven (Lambers, 1998). Thus under reduced light conditions, carbon would be preferentially allocated above-ground to maximise light capture (Poorter and Nagel, 2000). However, our findings negate this, with a greater proportion of biomass allocated below ground, in the highly shaded areas under the PV arrays. These results also add to a growing body of evidence that contradicts this demand driven theory (Chapin, Schulze and Mooney, 1990; **Farrar** and **Jones**, 2000; Poorter *et al.*, 2012), indicating that there are a variety of factors which influence C allocation. The reduced temperatures under the PV arrays may also have been a factor which lead to an increase in below-ground biomass: temperature has a positive relationship with above:below-ground biomass ratios, indicating that in lower temperature conditions there will be an increase in the below-ground C sink (Kang *et al.*, 2013). Carbon storage has been shown to be positively related to changes in below ground biomass (Yue *et al.*, 2016). Therefore, with more C allocated to below-ground biomass, through changes in temperature and shading, there is potential for this to result in an increase or decrease in C storage under PV arrays.

5.5.2 Solar farm effects on vegetation community composition

We assessed above-ground biomass by plant functional type, on the solar farms which had not yet been grazed in the 2015 growing season. In addition to this, species presence surveys were conducted at all the solar farms used in this study. We found that under the PV arrays, graminoids dominated the vegetation community with relatively few legumes and forbs. The forbs that were present under the PV arrays consisted of shade tolerant species such as *achillea millefolium* and *onopordum acanthium* (Grime, 1988). In the gap areas, species composition did not vary from the control areas, with a large proportion of the species composition made up of forbs and legumes. Legumes are shade intolerant, which explains the dramatic decline in legumes in the biomass by plant functional type and species presence data (Grime, 1988). The loss of legumes will result in lower levels of nitrogen fixation under the PV arrays(Bergersen, 1982). Nitrogen dynamics in grasslands are important controls on carbon cycling particularly growth and decomposition (Janzen, 2004; Joshuaschimel and Seanmichaelschaeffer, 2012). Lower levels on N fixation may result in suppressed levels of productivity (Bauer *et al.*, 2012), which corresponds with the total biomass data that revealed productivity under the PV arrays is lower than in the control and gap areas.

Forb biomass was lower under the PV arrays than in the control and gap areas. This finding is further support by the data from the species presence surveys which found a lower number of forb species under the PV arrays. Many forbs in the UK are shade intolerant, which may explain their decline under the PV arrays where PAR has been found to be 92 % lower than in control and gap areas. However, *achillea millefolium* and *onopordum acanthium w*ere found on a number of the solar farms surveyed under the PV arrays. These forbs are more tolerant of shade than the majority of other forb species (Grime, 1988). Although forb richness declines under the PV arrays solar farms are still a viable management options for promoting biodiversity. The low system disturbance levels (grazing, cutting, ploughing etc.) on solar farms means that the grasslands are able to support a wide variety of forbs which in turn is beneficial for fauna diversity (Ribeiro, Fernandes and Espírito-Santo, 2014; Dover *et al.*, 2011). Plant diversity is also a key component to C cycling in grasslands (De Deyn *et al.*, 2009). Therefore, solar farms managed for biodiversity have the potential to help preserve grasslands important ecosystem functions which are currently under threat from global environmental change.

5.5.3 Solar farm effects on vegetation height

In the gap areas, vegetation height was found to be the greatest, and under the PV arrays vegetation height was lower than in the control areas. This is likely to be the result of changes in microclimate, as vegetation community composition did not change between the gap and control treatments. Gap soil and air temperatures on solar farms have found to be cooler in the autumn and winter than control and under areas. In addition, wind speed in gap areas is notably slower. These microclimatic factors may have resulted in an increase in vegetation height in the gap areas. High wind speeds have been found to negatively affect vegetation height, through anatomical changes, specifically structural (Whitehead, 1963). Therefore, it is likely, that the reduction in wind speed may have led to the increase in vegetation height. Vegetation height is an important plant physiological feature and an indication of growth conditions.

5.5.4 Solar farm effects on root and leaf C:N ratios

Under the PV arrays the C:N ratio of grass leaf decreased and root C:N ratios increased, partially supporting our fourth hypothesis that C:N ratios in the leaf, root and soil will be affected by the presence of PV arrays. Soil C:N ratios were unaffected by the presence of PV arrays.

It is likely that this decrease in leaf C:N under the PV arrays is the result of changes in plant C and N allocation under the microclimatic changes imposed by solar PV arrays. Other studies have found that deep and sustained shading also resulted in a decrease in leaf C:N ratio (Ma *et al.*, 2015). Shading strongly reduces total C in shots but not roots (Bahn *et al.*, 2013). This decrease in C in shade is attributed to a decrease in photosynthetic capacity, and also results in a decrease in the C:N ratio. In addition to changes in C, leaf N has been found to increase under shaded conditions, due to the translocation of N to photosynthetic tissue (Ma *et al.*, 2015). This would again result in a decrease in leaf C:N, and potentially increase root C:N ratios, which is consistent with the results of this study.

C:N ratios negatively correlate with microbial decomposition (Lamb, Kennedy and Siciliano, 2011; Thornton and Rosenbloom, 2005). Litter decomposition plays an important role in carbon cycling in terrestrial ecosystems (Aerts, 1997). Litter decomposition is highly dependent on litter quality and climatic controls (Bardgett, De Deyn and Ostle, 2009). One of the most influential litter quality controls is the C:N ratio, which is a key factor controlling lability (Ping *et al.*, 2016). With regards to climatic factors, temperature and moisture are important controls (Zhu, Yang and He, 2013). The decrease in above-ground biomass C:N ratios for grasses (the dominant plant functional type in these areas) may not, in fact, lead to an increase in plant decomposition, due to reduced growing season temperatures under the PV arrays. However, due to the variability in precipitation under the PV arrays, along with other abiotic factors such as radiation receipts, further experimental work is needed in order to draw conclusions regarding changing C:N ratios and interactions with climate as controls on leaf litter decomposition.

Increased litter N in grasses may be beneficial for farmers in terms of livestock productions. Although, above-ground biomass production is lower under the PV arrays the increase in grass N may mean that this food source is now more efficient at in terms of livestock growth.

5.6 Conclusion

Solar energy farms represent a significant and increasingly common land-use change in the UK. The microclimatic and land management changes have the potential to affect vegetation communities and primary production affecting ecosystem service provision

and the sustainability of this renewable energy technology; however, understanding of solar farms effects on these factors is limited. This study has shown that the establishment of solar farms on UK grasslands is likely to affect vegetation community composition, vegetation height, above and above:below ground biomass and plant C:N. Some of these metrics are controlled by site specific factors such as annual site precipitation, soil type and the age of the solar farm. As the first solar farms in the UK were stablished in 2011, it is important to assess how these effects may change throughout its operational life.

5.7 References

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Chapter 6: Thesis Discussion

6.1 Overview of key findings

Solar farms represent a significant land use change as the number and size installed on UK grasslands continues to grow (Turney and Fthenakis, 2011; BRE, 2014). This thesis was devised in response to the need to better understand how the microclimatic changes imposed by PV arrays might affect grassland ecosystem functions (Armstrong, Ostle and Whitaker, 2016; Armstrong *et al.*, 2014). The overarching aim of this thesis was to assess how solar farms affect UK grassland carbon cycling. This was addressed, firstly, in the form of a literature review (Chapter 1) and secondly via four experimental research chapters (Chapters 2-5), assessing how diversity and the microclimatic changes in solar radiation, temperature and precipitation caused by the presence of solar farms affect carbon cycling properties and processes. Specifically two research objectives were devised:-

1. How does diversity and solar radiation affect decomposition processes? (Chapter 2, Chapter 3 and Chapter 4)
2. How does diversity and solar radiation affect productivity processes? (Chapter 2 and Chapter 5)

This research has focused on impacts in UK grassland ecosystems, due to the rapid expansion of this renewable energy technology and the importance of grassland C stores, but limited understanding of the effects of this technology on ecosystem functions (Armstrong *et al.*, 2014).

This thesis has demonstrated for the first time how shading can reduce warming effects on grassland decomposition processes. In addition, UV-B exposure was found to facilitate, microbial decomposition processes but this effect was determined by leaf litter type. Moreover, decomposition processes in high diversity grasslands were more resistant to changes in solar radiation and temperature. Further, this effect of increased resistance in high diversity grasslands is mirrored in ecosystem productivity processes. Ecosystem productivity is reduced by shading induced by PV arrays, however, under the PV arrays, there is a proportional increase in below ground biomass and a shift in leaf and root C:N.

6.1.1 How does diversity and solar radiation affect decompsosition processes?

It is well established that changes in climatic variables can affect decomposition, with decomposition generally increasing in warmer and wetter environments, however thresholds exist (Cao and Woodward, 1998; Briones *et al.*, 2014; Bradford, 2013). Solar radiation is a key driver in arid ecosystems, however, understanding of effects in more mesic ecosystems is unclear (Zhang and Wang, 2015; Gallo *et al.*, 2009). Further, vegetation communities are known to be important controls on decomposition processes in ecosystems, through effects on leaf litter and rhizodeposit inputs (Cornwell *et al.*, 2008; Bardgett, De Deyn and Ostle, 2009), and have the potential to mediate decomposition process responses to changes in climate (Waldrop and Firestone, 2006; Classen *et al.*, 2015). The research in Chapter 2 demonstrated how shading and warming have differential effects on decomposition, as indicated by ecosystem respiration CO_2 fluxes, in high and low diversity grassland ecosystems. Ultimately, high diversity grasslands released less CO_2 decreases in solar radiation and increases temperature than low diversity grasslands, this indicates that high diversity grasslands may be more resistant to warming and shading than low diversity grasslands. This research is in line with other research which suggests that diversity is a principal component of ecological stability: the greater the diversity of a community the more likely the community will be able to adapt to perturbations (Holling, 1973).

In chapter 3, further evidence is provided of the effect of solar farms on soil temperature, soil moisture, and PAR receipts. In addition, I establish how these controls partially explain the observed differences in leaf litter decomposition and soil microbial community composition between the grassland and the grassland directly under the PV arrays. Specifically, lower decomposition rates under the PV arrays, are associated with lower growing season soil temperature and soil moisture, reduced PAR receipts, and changes to the microbial community composition, specifically affecting the fungal component. These results further support the findings of chapter 2, where the ecosystem respiration CO_2 measurements indicate that decomposition is lower due to reductions in temperature and solar radiation.

Soil microbial communities were affected by the presence of PV arrays: with lower total fungal PLFAs in soils under the PV arrays. This change in the fungal community also affected total soil PLFAs and the ratio of fungi: bacteria. The effect of PV arrays on soil microbial communities is likely due to direct changes in microclimate and indirectly through changes in vegetation communities (Bradford, 2013; Zhang *et al.*, 2013;

Waldrop and Firestone, 2006; De Deyn *et al.*, 2009). Fungal abundance decreases under the PV arrays most likely due to the lower summer temperatures. Further, multiple studies have found that fungal abundance positively correlates to temperature (Crowther and Bradford, 2013; Briones *et al.*, 2009; Castro *et al.*, 2010).

Warming treatments in chapter 3 were used to explain how decomposition processes on solar farms might be affected as climate change progresses and temperatures rise further. Interestingly, the analysis revealed that the temperature sensitivity of leaf litter decomposition is reduced in the areas directly under the PV arrays. This is attributed to changes in solar radiation receipts which are known to have direct and indirect consequences for decomposition processes (Brandt *et al.*, 2010; Zepp *et al.*, 2007b). This finding has consequences for our understanding of carbon cycling under climate change, as in addition to rising temperatures, solar radiation receipts are likely to be affected by changes in cloud cover, atmospheric particle loading, and land-use change. The presence of PV arrays dampened the effect of warming on decomposition processes. This may be due to lower soil moisture under the panels during the peak decomposition period, the effect of shading, and the change in soil microbial community amongst other factors such as the vegetation community composition. The temperature sensitivity of the decomposition of soil organic matter in grasslands has been found to positively correlate with soil moisture (Bradley-Cook *et al.*, 2016). This means that the lower soil moisture contents under the PV arrays during the peak decomposition period could have mitigated the effect of warming on the decomposition of leaf litter in our study.

Further, the variation in litter decomposition is accounted for to a greater extent, by the change in abiotic variables as opposed to the biotic variables. The change in temperature under the PV arrays is the main factor in determining decomposition rates, followed by changes in PAR receipts and soil fungal community.

Considering the findings of Chapter 3, which suggest that the variation in solar radiation receipts on a solar farm in a mesic ecosystem can partly explain the reduced rates of decomposition, follow-up experiments were conducted. Chapter 4 demonstrates the positive effect of UV-B pre-exposure on the microbial decomposition of leaf litter. On solar farms reductions in UV-B radiation receipts in the areas directly under the PV arrays may result in reduced k, which in turn could lead to greater soil carbon storage

and a potential strategy to mitigate the impact of atmospheric carbon loading. In addition, the variations in the effect of UV-B pre-exposure on microbial decomposition were more associated with changes in lignin content than cellulose, hemicellulose or C:N. Our results considered with the predicted changed in leaf litter chemistry as a result of changes in climate, specifically, solar radiation, could be used to better predict how changes to different leaf litter chemistry components will affect the rate of decompsoiton under different climate change scenarios (Aerts, 1997).

Existing research largely overlooks the role of solar radiation receipts in mesic ecosystems, although, there is extensive evidence of its effects in arid ecosystems. However, together chapter 2, chapter 3 and chapter 4, indicate that decomposition processes in mesic systems have the potential to be affected by changes in solar radiation receipts, and therefore should be incorpotated in to ecosystem C cycling models.

6.1.2 How does diversity and solar radiation affect productivity processes?

In chapters 2 and 5, I demonstrate how changes in solar radiation receipts, can affect productivity through changes in biomass, C allocation, and vegetation community composition. As shading increased, rates of photosynthesis decreased, although CO_2 fluxes were generally more sensitive to changes in PAR receipts in low diversity grasslands than in high diversity grasslands. Shading resulted in a decrease in C uptake through photosynthesis, although this was also associated with a decrease in C release through ecosystem respiration. However, net ecosystem exchange, which is representative of the balance between ecosystem exchange and photosynthesis, reveals that at intermediate levels of shading (74 % reduction in PAR), high diversity grasslands have a greater overall level of C sequestration than low diversity grasslands. However, when subjected to full shade (90 % reduction in PAR), both high and low diversity grasslands, were at times found to be net sources of CO_2. This response of high and low diversity grasslands to the intermediate level of shading is likely due to the fact that high diversity grasslands could be more resistant to change, due to the more complex community possessing a higher number of functional traits than our low diversity community (Holling, 1973). Light availability has been found to control the strength of diversity effects on primary productivity, and how this response depends on plant functional types (Siebenkäs, Schumacher and Roscher, 2016). Specifically, when legumes were present the effect of shading on above ground biomass was less than when

legumes were absent (Siebenkäs, Schumacher and Roscher, 2016). This further backs up this research which shows that high diversity grasslands, which contained more legumes, shading effects on above ground biomass were less than in low diversity grasslands which contained fewer legumes. Therefore, high diversity grasslands could maintain ecosystem functioning in the 74 % shading level better than the low diversity species. However, a 90 % reduction in shading for both high and low diversity communities meant that the grassland even at the height of the growing season was often a net source of CO_2.

The findings of chapter 5 are in line with those of chapter 2, where productivity was found to be affected by the presence of PV arrays at 17 solar farms across England and Wales. Specifically, our results show that under the PV arrays, above-ground biomass was lower and there was a change in carbon allocation, reflected by a decrease in the above:below-ground biomass ratio. Changes in carbon allocation strongly influences plant-soil processes, and our findings are supported by existing research which has found that sustained shading reduces photosynthates in shoots but not root, which as indicated by results in chapter 5 could change the above:below ground biomass ratio (Bahn *et al.*, 2013). Chapter 2 also shows how shading affected leaf C:N ratios in grasses, chapter 5 reflects the same trend showing that C:N ratios in plant shoots decreased in the shaded conditions under the PV arrays, chapter 5 furthers the research by analysing root C:N and finding that this increased under the PV arrays. This further supports the above:below ground biomass results showing shading may be a factor which determines the allocation of C and N in grasses. In addition to changes in above ground biomass, chapter 5, also addressed how shading may affect below ground biomass. Chapter 2 and chapter 5 also demonstrated how shading can affect vegetation community composition. Shading caused grasses to become dominant, with legumes and forbs lost from either the shading treatments (chapter 2) or areas under the PV arrays (chapter 5).

Diversity interacted with temperature to affect vegetation productivity as indicated by photosynthesis CO_2 fluxes. Photosynthesis exhibited contrasting directional effects in low and high diversity grasslands: in low diversity grasslands warming increased photosynthesis, whereas in high diversity grasslands warming decreased photosynthesis. This photosynthesis data does not support the theory that high diversity grasslands will be more resistant to warming than low diversity grasslands. Although

rates of biological reactions tend to increase with temperature, thresholds exist where increases in temperature can have negative effects on biological processes such as photosynthesis. Indeed, midday stress is recognised as a key process which can result in decreased photosynthetic capacity, when certain climatic thresholds are exceeded (Mathur, Agrawal and Jajoo, 2014b). Supporting our findings, warming has been found to decrease photosynthectic capacity in high diversity grasslands, and this finding was attributed to midday stress (De Boeck *et al.*, 2007). This may be because forbs and legumes are more susceptible to midday stress than graminoids, in addition to the negative effects of interspecific competition under environmental stress (De Boeck *et al.*, 2007). However, the above ground biomass data contradicts the photosynthesis results, showing that warming increases above ground biomass in low and high diversity grasslands by 13 % and 10 % respectively. This indicates that overall photosynthesis in high diversity grasslands increased with temperature, however, due to the timing of the CO_2 flux measurements (around midday), the high diversity grasslands were probably experiencing midday stress which temporarily decreases rates of photosynthesis.

6.1.3 Ecosystem carbon balance

The asymmetry of productivity and decomposition processes determines whether the ecosystem is a source or a sink of carbon. This thesis demonstrates the sensitivity of these processes to microclimatic changes and shows how vegetation communities can mediate the effects of these changes on productivity and decomposition processes. Although this research did not find evidence of changes to soil carbon a key indicator of changes to ecosystem C stores, chapter 2 shows how net ecosystem exchange was affected by changes in PAR and temperature. With warming and shading both decreasing net ecosystem exchange, however, shading decreased net ecosystem exchange in low diversity grasslands to a greater extent than in high diversity grasslands.

6.2 Remaining Knowledge Gaps

This research has demonstrated the critical need to improve our understanding of the effects of solar farms on ecosystem function, however, the improved understanding developed in this thesis raises more questions.

A ^{13}C and ^{15}N pulse labelling experiment could be used to track how microclimatic changes alter the cycling of C and N in the plant soil system in high and low diversity

grasslands. Specifically, using the experimental set up of chapter 2, ^{13}C and ^{15}N could be assessed in ecosystem CO_2 fluxes, soil microbial community, plant roots and shoots.

The research in chapters 2,4 and 5 contributed to a growing body of evidence that shows how vegetation communities can mediate responses to changes in climate. It would be beneficial to identify plant functional types or interspecific interactions which cause the differential effects of the response of high and low diversity grasslands to microclimatic changes.

The results of chapter 3 could be strengthened by repeating the study using leaf litter from different species and mixtures of species over a longer period. Given the results of the microbial community analysis which shows a decrease in fungal community abundance in areas directly under the PV arrays, an interesting area of research could assess how this change affects higher trophic groups in the soil food web.

6.3 Overall Conclusion

Under the pressures of climate change and energy demands, solar farms are increasingly being deployed on UK grasslands. However, the microclimatic effects of the presence of PV arrays, have potential consequences for the grassland carbon cycle and the potential to affect the carbon balance of this technology. Further, due to the interplay between climate and biotic factors on the carbon cycle, the effect of the microclimatic changes may be partly governed by the vegetation and soil microbial communities on the solar farms. This research found that high diversity vegetation communities could be used to mitigate the effects of warming and shading on grassland CO_2 emissions. Moreover, the presence of shading may reduce the temperature sensitivity of CO_2 emissions. Further, changes in temperature, solar radiation, and soil moisture can result in changes in decomposition processes on solar farms in the UK. The effect of changes in solar radiation on decomposition processes has largely been overlooked in mesic ecosystems. However, this research demonstrates how changing UV-B receipts have the potential to effect decomposition, in environments where microbial decomposition dominates. Additionally, we show how vegetation on solar farms may be affected by microclimatic changes and vegetation communities. This research indicates that solar farms affect above-ground biomass, carbon allocation, species composition and vegetation C:N ratios, all which have potential consequences for grassland carbon cycling. Taken together this research provides some of the first understanding of the

effects of solar farms on grassland carbon cycling. This research adds to a growing body of evidence which demonstrates how solar farms can be managed to provide multiple benefits of solar farms on grasslands, which could deliver energy, food and other ecosystem services including carbon sequestration, biodiversity. Finally, further research is needed to draw definitive conclusions on the size and direction of effect of solar farms on grassland carbon cycling over their operational lifespan. Specifically, will solar farms, over time affect soil carbon storage and could management strategies be utilised to maximise the benefits of this renewable energy technology.

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