**Draft 2 for Feedback (“Treating soils like dirt: the failure of the European Union to combat soil degradation in Europe”)**

**1. Introduction**

Under current agricultural practice, global food security will fail within the next thirty years (Baldos et al., 2014). With a projected global population exceeding 9 billion by 2050 (Lutz et al., 2010), there is an increasing deficit of available land required to meet consumption demands (Schmidhuber et al., 2007). Should I include a statistic here about the remaining available agricultural land left? Consumption per capita is estimated to surge over the coming decades as disposable incomes soar (Baldos et al., 2014). Growing disparity over the division of available agricultural land between crops for direct consumption, feedstuffs for livestock, and first generation bio-fuels adds further pressure to the land constraint issue (Fischer et al., 2009). Increasing the productivity of the current land already under cultivation by 70-100% would be enough to achieve global food security by 2050 (Connor et al., 2012), while maintaining ecosystem functioning (Bindraban et al,. 2012). Total factor productivity can be defined as the increase in aggregate output relative to an index of all inputs (Baldos et al., 2014); in other words, the closure of yield gaps between actual yield and potential yield under optimised management. Currently, agricultural systems in Western Europe are only producing at 80% of their potential yield (Bindraban et al,. 2012). It is estimated that in the last half a century, 85% of agricultural land has been left available for other uses due to increases in productivity and yield, mitigating the release of up to 590 Gt land-use-related CO2 (Valin et al., 2013). Yields are often maximised by the intensification of production via increased chemical inputs (Sparovek et al.,, 2001), and correlate to significant negative effects on surrounding ecosystems due to poor management strategies and trade-offs at the expense of the environment (Bindraban et al., 2012). Models predict that in order to double global agricultural productivity, fertilisation rates would have to triple, irrigation would double, and cropland would have to expand by almost 20% (Tilman, 1999).

Soil degradation is arguably the greatest limiting factor of global productivity growth (Bindraban et al.*,* 2012). Global trends have shown stable increases in the average yields of most major crops throughout the last 50 years (Connor et al., 2012), a notable example includes the supply of 35% of global wheat from Europe, one of the worlds most productive cropping systems (Senapati et al., 2020). Today, 2850 Mt of field crops are being produced on only 12% of the total global land area (Connor et al., 2012). However, there is increasing concern from farmers and policy makers alike as growing evidence suggests high-yielding production systems are headed towards a yield ceiling (Grassini et al., 2011). A consensus seems to have been reached within the literature that relative (%) gain is dropping due to biological limits of the current agronomical systems reaching their attainable maximum (references). Soil degradation refers to the loss of the present and/or future ability of the soil to provide functions and services due to a decline in soil quality (Lal, 2015). Global food security is significantly threatened by soil degradation as it suppresses land productivity and breeds undesirable, deteriorated physical surface conditions (Baumhardt et al., 2015). Socioeconomic pressures and exponential population growth correlate with anthropogenic activities that deplete the natural environment (Abrol et al., 1990) and exacerbate soil degradation, including deforestation, urbanisation, and intensive agriculture. These processes disrupt the natural fortification of the soil and its vegetative cover, leaving it vulnerable to climatic hostility (Oldeman, 1992). Ecological degradation is a culmination of physical, chemical and biological soil degradation (see Table 1 for definitions), and presents itself as the impairment of ecosystem processes, including nutrient and hydrological cycling, and a reduction in net biome productivity. Currently, there is insufficient acknowledgment of soil degradation as a serious threat to ecosystem services and global food security due to poor understanding of soil as a critical natural resource.

Agriculture is one of the leading causes of stress to the environment (Seybold et al., 1999), and its effects can be seen long after the period of cultivation has ended. Agriculture refers to any crop-plant production system, and has huge ramifications for soils and surrounding ecosystems (McLauchlan, 2006). Soil can be categorised as “resilient” or “resistant” based on how it responds to agricultural perturbation (Seybold et al., 1999). A soil that is particularly vulnerable to change under agricultural management practice but rapidly returns to its original state would be defined as “resilient”. In comparison, a “resistant” soil can withstand greater levels of management intensity without changing, but once changed may lose the ability to return to its initial condition. Biomass amendments, tillage, fertilisation, and altered hydrology are specific agricultural management practices that have the potential to drastically change the quality of the soil. The elimination of naturally competing plant species and introduction of annual crop plants significantly alters plant biomass. Annual harvest diminishes organic carbon returns to the soil (Imhoff et al., 2004); in an agricultural system SOC is a linear function of carbon loads from crop residues (McLauchlan, 2006). Inversion tillage creates belowground disturbance through the pulverisation of the topsoil that increases SOC decomposition rates (Reicosky et al., 1997; Collins et al., 2000; Six et al., 2000) and enables physical soil erosion due to the lack of vegetation cover. Research shows that after 100 years of maize cultivation, agricultural land contains less than half the amount of topsoil in comparison with perennial grasslands (Gantzer et al., 1991). The full extent of soil degradation caused by inversion tillage can only be realised decades after its use: subsoil begins to appear at the surface, tillage-related landforms such as tillage banks form, and tillage translocation (the movement of the cultivation layer) takes place (Van Oost et al., 2006). Nutrient inputs through fertilisation can cause changes to the microbiome, leading to shifts in soil characteristics (Lin et al., 2019), and subsequently, whole ecosystems (Wang et al., 2011). Irrigation exacerbates siltation, salinisation and sodicity, and can cause an anaerobic shift in the soil, leading to loss of soil hydroecological functioning (Assouline et al., 2015) and irreversible soil damage (Yin et al., 2021). Additional factors influencing soil degradation include continuous cropping/grazing and the use of heavy machinery. This is due to grazing interrupting the natural cycle of returning mineral-rich, dead plant matter to the soil, and machinery compacting the soil, which in turn prevents water infiltration and accelerates erosion. Soil degradation processes such as these call conventional management practices into question.

Three quarters of terrestrial ecosystems are impacted by soil degradation, with no intervention this number could rise to 90% by 2050 (Pereira et al., 2019). Soil degradation affects more than half of the global agricultural systems, adding a huge strain to approximately 54% of global ecosystem services (Nkonya et al., 2016). More recently, there has been significant interest from scientific researchers regarding the increasing interactions of the pedosphere, biosphere, and atmosphere as a result of soil degradation. The pedosphere is the second largest carbon reservoir on the planet (Stolte et al., 2016). An estimated 2400 Pg of soil organic carbon (SOC) is locked in the upper 2 metres of the soil (Kirschbaum, 2000). However, soil degradation processes such as soil erosion have the potential of unlocking this soil organic carbon and releasing it into the atmosphere. Currently, there is an active debate in the scientific community regarding whether soil degradation and associated ecosystem changes translates into a net C sink or source for atmospheric CO2. It is widely known that soil degradation has huge influence over the global carbon budget. Berhe (2007) estimated that over time the volume of CO2 unlocked from the soil and released into the atmosphere would be equal to three quarters of all fossil fuel carbon emissions. Within the last two centuries, 200 Pg C has entered the atmosphere as a direct result of land conversion and soil degradation (DeFries et al., 1999). The work of Lal (Lal 1995, 2001, 2003a, 2003b, 2003c, 2004) lends huge support to this debate, having found that soil degradation process, such as soil erosion, are a source term as opposed to sink, in the global carbon budget. They predict that annually 1.14Pg C is being released into the atmosphere from the soil due to aggregate breakdown. There is significant support for these finding in the literature (Schlesinger et al., 1995; Starr et al., 2000). A n evidence-supported conclusion of this debate would have huge implications for the future of soil science, ecology and environmental policy. Soil degradation also has important economical consequences. At the going rate, the cost of no intervention is significantly higher than the cost of intervention (Mirzabaev et al., 2015). Estimations of the global revenue lost as a result of soil degradation processes range from 300 billion US dollars (Nkonya et al., 2016; Pancheco et al., 2018) to 6.3 trillion (Sutton et al., 2016), on account of impaired ecosystem function. In Europe, it is thought that the European Union suffers losses of 1.25 billion Euros annually, due to a reduction in agricultural productivity, as a result of 12 million ha of European soils being degraded (Panagos et al., 2018). Similarly, the United Kingdom bears the brunt of a heavy 1.4 billion pound deficit as a consequence of the aforementioned soil degradation (Graves et al., 2015).

Land degradation must not only be reduced, but reversed to ensure long-term productivity. As intensive land management practices are largely responsible for soil degradation, it would be logical to conclude the implementation of less intensive practices is key to rehabilitating degraded land. Conservation Agriculture (CA) has the ability to reduce soil degradation and increase soil productivity (Pereira et al., 2019). Conservation Agriculture, also known as no-till farming, is based on three principles: i) no-till seed drilling; ii) continuous vegetation cover with organic mulch e.g. crop residue, green manure, cover crops; and iii) maintaining plant diversity by creating polycultures of crops. In contrast to conventional agricultural practices, CA wields the natural diversity of the ecosystem as a tool to propagate soil health and productive capacity (FAO, 2011). CA has the ability to encourage infiltration and storage of water in the soil, and indirectly improve interactions in the rhizosphere between plant roots and soil microbes, as well as the uptake of nutrients (Kassam et al., 2014). This builds resilience in the soil to environmental stressors. There is ample documentation surrounding the benefits of CA practices in long term studies, including: positive responses of soil biota to the removal of mineral fertilisers and pesticides (Henneron et al., 2015), increased soil fertility as a result of crop diversification (DiFalco et al., 2017), and boosted soil productivity under wide crop rotations and cover crops (Ranaivoson et al., 2017; Garcia-Gonzalez et al., 2018). There is enough evidence to imply that the implementation of CA, in conjunction with other defensive management practices to combat erosion and enhance SOC, has the potential to conserve our soils (Lal et al., 2013). For this reason, CA is at the forefront of approaches proposed in the FAO sustainable agricultural intensification strategy (FAO, 2011). Additionally, among the 17 sustainable development goals (SDG’s) proposed by the United Nations (UN) in their agenda for 2030, there is mention of achieving Land Degradation Neutrality (LDN), which cannot be achieved without CA (Kust et al., 2017; Pereira et al., 2019) as agriculture is a primary cause of degradation.

Ecosystem services are not just provided by the physical aspects of soil but also by the living things within. Despite bacteria and archaea being the smallest independently living organisms on the planet (Aislabie et al., 2013), they are, in addition to fungi and other soil biota, the drivers of critical ecosystem services such as those provided in Table 1. ‘Soil health’ is a novel attempt to define and quantify the agricultural sustainability potential of soils (Norris et al., 2020). Maintaining the supply of ecosystem services is a common prerequisite for attaining soil health, according to most definitions in the literature (Ferris al., 2015). What differentiates this soil measurement from previous metrics is the inclusion of soil biota and their subsequent soil processes (Doran et al., 2000). Recent developments in microbial community quantification methods has allowed for empirical measurements of soil biology, and consequently soil health. Specifically, DNA-based analyses are a significantly under-utilised metric in the attempt to comprehend soil health (Fierer et al., 2021). The inclusion of soil biota in soil health assessments provides many advantages. Where other soil parameters such as nitrogen levels or SOC may be temporally variable, microbial communities are generally stable from week to week (Lauber et al., 2013). Therefore, any analysis that finds changes in the microbiome can be more confident that something significant is happening, as opposed to natural variation from environmental conditions. Hermans et al., (2020) found that microbial diversity could be used to anticipate soil quality physico-chemical attributes by quantifying relative abundances of bacterial taxa. Similarly, correlations between functional microbial genes and common soil processes have been identified, such as nitrification and methane production (Fierer et al., 2021). These are processes often indicative of soil health, therefore microbial genetic material can be used to determine oscillations in soil properties, demonstrating the significance of DNA-based microbial analyses.

*Modern agriculture thus faces great challenges not only in terms of ensuring global food security by increasing yields, but also mitigating the environmental costs particularly in the context of a changing environment and growing competition for land, water, and energy (Chen et al., 2014). Therefore, there is an urgent need to find early indicators of soil health degradation in response to agricultural management (Grime, 1997; Cardoso et al., 2013). Changes in land use are altering both microbial community structure and diversity in terrestrial ecosystems (Rodrigues et al., 2013). Since soil bacterial communities drive many different ecosystem functions (e.g., Delgado-Baquerizo et al., 2016b), and their abundance, richness, and composition are sensitive to the changes in the land use and management (Singh et al., 2014), they have been considered as early indicators of change in the quality of soil ecosystems (Kennedy and Stubbs, 2006). In some instances, changes in microbial populations or activity can precede detectable changes in soil physical and chemical properties, thereby providing an early sign of soil improvement or an early warning of soil degradation (Nielsen et al., 2002). Herein, we conducted a meta-analysis to explore how soil properties (soil microbial diversity, community composition, and abundance) are affected in response to agricultural management across the main Europe. The aim of the meta-analysis was to identify the impact of agriculture practices on soil nutritional health and microbial communities. We also aimed to examine if the response of microbial community to agriculture is consistent across different management intensities. We collected data from 102 peer-reviewed publications as well as unpublished data to create a global dataset of soil bacterial diversity and composition evaluated with next generation sequencing techniques (mostly 454 Pyrosequencing). Our meta-analysis revealed foreseeable nature of the microbial community responses to vegetation types suggesting that the microbial indicators can be developed as tools for prediction for primary productivity and soil health.*

**2. Materials and methodology**

2.1. Data source and selection criteria

In order to quantify the long-term effects of conventional and organic agriculture on the bulk soil microbiome, 19 studies were amassed and their results analysed in a meta-analysis (Table 1). A comprehensive literature search was conducted from June 2022 to July 2022 in online databases Google Scholar and Web of Science using search terms produced by the R package “LitSearchR” (available at <https://github.com/elizagrames/litsearchr>). The particular search strategy outputted by the programme was ((bacteria OR microbes OR eukaryote) AND (abundance OR biodiversity OR incidence OR occurrence OR diversity OR \”community composition\”) AND (\”agricultural soil\” OR agriculture OR conservation OR \”crop rotation\” OR cultivation OR farming OR \”land use\” OR natural OR \”organic farming \” OR soil OR \”conventional farming\”)). The literature search was performed reverse chronologically in order to identify the most recent significant findings.

Additionally, papers were identified using manual searching of bibliographies of previously selected studies, if they were not already found using the search terms. Papers were initially identified and classed as relevant based off the title and abstract. Once a list of papers had been collected, the list was refined using the following criteria:

* Comparison of organic and conventional farming provided quantitative results on soil microbial diversity and/or abundance.
* Agricultural practices must have been applied for at least two consecutive years prior to sampling.
* *Farming system comparison needs to be pairwise, meaning the compared fields are subjected to the same pedo-climatic conditions (e.g. temperature, precipitation, soil type).*
* *Only data deriving from open field and greenhouses were considered. Experimental incubation studies in the lab or pot trials were not included since they do not reflect in situ soil conditions.*
* The study was related to European farming systems, in order to answer our secondary research question regarding EU agricultural policy.
* Used cereal crops, either as the primary crop or as cover crop, in order to control for differences in the soil microbiome due to species of crop.
* The document type was defined as ‘article’, language as ‘English’ and was published in a scientific, peer-reviewed journal.

Additional criteria were applied to studies in order to prepare them for data handling.

* If the study sites being compared did not specifically state ‘conventional’ or ‘organic’, or if a third intermediate/transition site using some aspects of organic/conventional management strategies were identified, then the sites were assigned into one of four categories: natural, organic, intermediate or conventional, using definitions provided by the author (Table 2).
* *If several fertilisation doses were applied, always ‘full fertilisation’ often referred to as ‘recommended’ was chosen. Furthermore, if several fertiliser treatments were compared within the same study, they were handled as individual comparisons.*
* *When different crop rotations from a certain site existed, they were handled as individual comparisons.*

The target variables extracted from each study are stated in Table 3. Additional study characteristics recorded as *covariates for further categorical analysis* are present in Table 4. In cases were data was only reported in graphical format, the R package “*metaDigitise*” (Pick et al., 2018) (<https://doi.org/10.1101/247775>) was used *for more standardised data extraction*.

2.2 Data analysis

*2.2.1 Meta-analysis*

A meta-analysis aims to summarise the current and existing literature by using previously determined search terms followed by subsequent evaluation and synthesis of primary research (Mikolajewicz et al., 2019). The statistical analysis itself computes the weighted average across multiple studies proposing the same research question, in order to conclude a quantitative estimate of the research topic. Multiple models have been constructed based on sample size, variance and other criteria. A Random Effects model assumes variation from study level differences, while a Fixed Effects model explains it as that of sampling error. Both Random Effect and Fixed Effect models were produced by employing the ‘*metafor*’ package *v.3.4-0* implemented in *RStudio v.4.2.1*.

*2.2.2 Calculating effect size and variance*

Effect sizes were calculated by taking individual observations from each agricultural management treatment in the same study and pairing them together (see Table 1). The ‘conventional’ treatment was used as a baseline in the analysis to compare all other treatments against, therefore in every pair there was a ‘conventional’ and ‘non-conventional’ treatment. One study could generate multiple effect sizes if there were multiple treatments, soil microbial properties or experimental sites. Effect sizes and variances were calculated by applying the *escalc()* function in the ‘*metafor*’ package, and specifying for the log transformed ratio of means (LRR, Ti). This can also be shown as:

*Equation 1:* *Ti = LRR = ln (ȲCC/ȲNC)*

where (ȲCC) is the mean of the response variable under a non-conventional treatment over the mean of the response variable under conventional treatment acting as a control (ȲNC).

*There was at least one pairwise comparison of H in each study. However, several studies reported data from more than one pairwise comparison, with multiple organic and/or conventional treatments investigated. In addition, several studies were conducted at the same research site. We fitted multi-level random- and mixed-effects (i.e. testing for the effects of moderator variables) models by restricted maximum likelihood.*

*To calculate the combined effect size and its corresponding variance, we used a multilevel meta-analysis model (mixed model) run with the ‘rma.mv’ function from the Metafor package. The model was fed calculated effect sizes and sampling variances (as described above) and fitted via restricted maximum-likelihood estimation. As we obtained multiple effect size values per study (e.g. several sampling dates), we used a mixed model with “study identification” (study ID) as a random factor to take into account the dependencies among estimates from the same study. To estimate the overall effect of agricultural practices on soil microbial variables under investigation, we first ran a random-effects model without any moderator variables. This model was run for: I) each non-conventional agricultural system contrasted against a conventional agricultural system; and ii) each subgroup of domain (as defined in Table 1). The responses of the soil microbial community parameters were statistically significant if the 95% confidence intervals of the agricultural effects did not overlap zero. The avergae response ratios were tested for heterogeneity using the QE statistic. When the QE was significant (P-value < 0.05), moderators have been added in the models to explain this heterogeneity. When the QE was not significant, we did not perform further models with moderators (corresponding to 15% of the models, Tables S2).*

*To further determine whether the collected metadata (moderators) explained the heterogeneity of effect sizes, we used a mixed model with moderator variables as a fixed factor and ‘study ID’ as a random factor. This second mixed-model was run separately for each moderator variable under study. Publication bias was first evaluated by conducting a visual inspection of the funnel plot (x-axis: observed effect sizes; y-axis: standard error) to identify asymmetrical distributions and heterogeneity.*

With the study effect sizes and variances calculated, we used R package ‘metafor’ and its function rma to calculate the global effect sizes, 95% confidence intervals (CI), and total between-study heterogeneity (I2). If the CI of a global effect size mean does not include zero, then the management intensity effect on a soil microbial parameter is statistically significant. Function ‘funnel’ was used to produce the funnel plots for each soil microbial parameters to visually check significant heterogeneity and publication bias.

**3. Results**

**4. Discussion**

*4.1 Significance of agricultural moderators*

Agricultural moderators hold influence over the response of microbial properties to intensity treatment. Soil type was found to have a significant role in the response of soil microbial abundance and diversity in all treatments except conservation, however upon inspection this was due to the data-set being too small for a moderator to be applied to the mixed effects model for this treatment. Interestingly, results showed consistent significant effects for groups in all treatments except the intermediate treatment. No significant difference between soil types were observed for total community in non-conventional systems, however, loess chernozem appeared to have the largest response ratio while stagnic luvisol had the smallest. This could partially be explained by stagnic luvisol and loess chernozem being the soil type in 54% and 27% of the studies in this meta-analysis, respectively. A small response ratio in stagnic luvisols may be attributed to luvisols being highly fertile and often used in agriculture, therefore moving to organic practices has smaller effects on already productive soils? Do some research here about the effect of different soil type and agriculture. In general, soil type does seem to have some effect on the response of soil microbial and abundance to different intensity cropping treatments, in particular loess chernozem. However, our results could be skewed due to the lack of data, in 69% of the studies in this meta-analysis did not provide soil type data. If more time was available an interesting moderator to investigate would be soil sampling time. In many of the studies included in this analysis, crop rotation and cover cropping were used in the non-conventional treatments. It is known that both of these processes has significant effects on soil microbial properties, therefore at which point in these processes that sampling occurred could be having a major effect on our results. For example, if sampling occurs during the cover crop phase, we might expect larger response ratios. Future research should take timing bias into account when assessing the effects of intensity treatment on the soil microbiome as they are time dependent.

*4.2 Data-set and limitations*

Ultimately, meta-analyses will fall victim to the methodological limitations of the primary studies that they are composed of. Microbial parameters such as abundance, that are optimised and quantified under laboratory conditions, risk overestimating true counts *in vivo*, this is specifically true of metrics such as colony forming units (CFU). A meta-analysis can only be as strong as its literature review and screening process. Small datasets have the ability of lending validity to incorrect conclusions, and as such, both significant and insignificant results from small datasets should be accepted with caution. Potential publication bias should be kept in mind when conducting a meta-analysis; the ‘file drawer effect’ introduces bias when non-significant findings are published less often in the literature. Funnel plot asymmetry is a common way of testing for publication bias, however ,in the case of small datasets, is significantly underpowered. The conservation soil microbial diversity and abundance datasets were hugely limited by their size, although funnel plots testing for publication bias detected no asymmetry. The notable absence of conservation agricultural system data may be explained by the lack of reduced or no-tillage farming practices in Europe, while this also highlights a potential area for future growth. All further parameters and treatments were tested for publication bias, and when none was found, included in further analysis. The risk of publication bias is significantly higher in meta-analyses comprised of older research. Therefore, as this review exclusively analysed studies from within the last decade, bias should be limited and concerns over underpowered tests for publication bias disregarded. Going forward, meta-analyses will gain strength as our knowledge and understanding of appropriate methodologies evolve, and our conclusions regarding the future of agricultural systems will be more robust.

*4.3 Consequences for agricultural sustainability*

Conventional systems are defined by their harmful, synthetic inputs, depletion of SOC, and evidently, significant losses in soil biodiversity. Additionally, they have the ability to transform the ecosystem from a carbon sink to a carbon source, emitting greenhouse gases such as CO2 and SO4, which is proven to accelerate global climate change. In order to ameliorate agricultural-mediated global warming, the conventional systems that global food security is so dependent on must be reduced or replaced with less intensive alternatives. Arguably, we need to move away from the idea that a total switch from intensive to no-input, no-tillage, conservation agriculture is the solution, as this is not realistic in terms of yield, and also not completely necessary. Agriculture is always evolving as new technologies and practices are being developed, some even say there is no such thing as conventional agriculture. Instead, specific non-conventional practices should be adopted and integrated into conventional systems to relieve the pressure on the environment while also attaining high enough yields to achieve global food security. Such practices could involve switching from synthetic nitrogen fertilisers to organic manures, slurries or residues. Encouragement of non-synthetic inputs would result in the restoration of SOC, and possibly the reversal of soil degradation, which would allow for agricultural intensification in a sustainable way. With the integration of a combination approach such as this, global food demand may be achieved while simultaneously reversing climate change, and reducing the insatiable exploitation of our planets finite resources.