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Effects of biodegradable mulch on soil quality

C. Li [a], J. Moore-Kucera [a,*], J. Lee [b], A. Corbin [c], M. Brodhagen [d], C. Miles [e], D. Inglis [e]

[a] *Department of Plant and Soil Science, Texas Tech University, United States*
[b] *Department of Biosystems Engineering & Soil Science, University of Tennessee, United States*
[c] *Washington State University Extension, Snohomish County Extension, United States*
[d] *Biology Department, Western Washington University, United States*
[e] *Department of Horticulture and Plant Pathology, respectively, Washington State University, Northwestern Washington Research & Extension Center, United States*

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ABSTRACT

Biodegradable plastic films are desirable alternatives to traditional black polyethylene plastic for use as mulches in agroecosystems. Efforts are ongoing to engineer biodegradable plastic mulches that could be incorporated into the soil at the end of the crop season, and decomposed by microorganisms, ultimately to CO_2, H_2O, and biomass. Whether changes in soil quality occur during or following biodegradation is unknown. An 18-month study evaluated the effects on soil quality following burial of four potentially biodegradable mulches and a no mulch control in high tunnel and open field tomato production systems across three geographically distinct locations (Knoxville, TN; Lubbock, TX; Mount Vernon, WA). The mulch treatments included: two starch-based mulches (BioAgri® Ag-Film and BioTelo Agri); one experimental 100% polylactic acid mulch (Spunbond-PLA-10); one cellulose-based mulch (WeedGuardPlus; positive control); and a negative control (no mulch). The soil management assessment framework (SMAF) was used to calculate a soil quality index (SQI) according to five dynamic soil properties: microbial biomass carbon, β-glucosidase, electrical conductivity, total organic carbon (TOC), and pH. Within the 18-month evaluation period, the effects of the biodegradable mulches on the SQI were minor, and dependent upon production system and time of incubation at all locations. In general, the SQI was higher in the high tunnel systems for some of the mulch treatments at Knoxville and Lubbock but the opposite was true at Mount Vernon. By the final sampling at 18 months, the SQI was lowest for WeedGuardPlus at Lubbock and Mount Vernon but at Knoxville, the WeedGuardPlus SQI was not significantly different from the no mulch control. Of the five SMAF indicators evaluated, soil microbial biomass and β-glucosidase activity were the most responsive to mulch and production systems, supporting the use of these variables as soil quality indicators for short-term changes due to this agricultural management practice.

1. Introduction

From a crop production perspective, plastic mulches can increase yields, extend the growing season, reduce weed pressure, increase fertilizer use efficiency, conserve soil moisture, and increase soil temperature (Lalitha et al., 2010; Lament, 1993; Lamont, 2005; Riggle, 1998). For these reasons, polyethylene mulches have been used in agriculture for over half a century (Lamont, 2005). A major limitation of polyethylene mulch involves disposal of mulch material following use. Current disposal options as reviewed by Hayes et al. (2012) and Ren (2003) include burning, incineration, recycling, composting and using landfills; each have major economic or environmental disadvantages (Kyrikou and Briassoulis, 2007; Lamont, 2005). Material that is not recycled or properly disposed of can fragment, and cause environmental degradation of land and water resources. For example, crop yields were decreased when residual plastic film left in the soil was 58.5 kg ha^{-1} (Ma et al., 2008). Further, plastic fragments have been found to adsorb toxins that persist in the environment and disrupt terrestrial and aquatic ecosystems (Derraik, 2002; Shimao, 2001). Research advances continue to improve the degradability of polyethylene materials (Esmaeili et al., 2013; Kasirajan and Ngouajio, 2012), but without modification, the high molecular weight, 3-dimensional structure and hydrophobic properties prevent biodegradation from occurring (Klemchuk, 1990). Moreover, polyethylene plastics are non-renewable petroleum-based products, and are produced at an annual increase of 9% (Hayes et al., 2012), warranting the development of alternative mulch products

* Corresponding author. Tel.: +1 806 834 5485.
E-mail address: Jennifer.moore-kucera@ttu.edu (J. Moore-Kucera).

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that better align agronomic demand with long-term ecological sustainability. In the U.S.A., approximately 162,000 ha are covered with plastic mulch (Miles et al., 2012) with more than 130,000 metric tons of agricultural plastics used for vegetable production alone (Shogren and Hochmuth, 2004). Recently, biodegradable mulch films have been viewed as a more sustainable ecological alternative to plastic polyethylene mulch (see review by Briassoulis and Dejean, 2010). Biodegradable plastic mulch first was synthesized in the mid-1970s (Albregts and Howard, 1972; Otey et al., 1974). However, early biodegradable mulches broke down only partially (Narayan, 2010). Although some commercially available mulches are compostable, reliable in-soil degradation of plastic mulches has not yet been achieved. Biodegradable mulch films theoretically can save significant labor and disposal costs through incorporation via soil tillage operations rather than disposal in landfills (Kasirajan and Ngouajio, 2012). By definition by the American Society for Testing and Materials (ASTM), biodegradable plastics are broken down by naturally occurring microorganisms (ASTM, 2012a), and ultimately converted to carbon dioxide and water under aerobic conditions (Narayan, 2010). To be part of sustainable farming systems, *in situ* biodegradation of mulches should occur (i) within a reasonable timeframe (within 2 years as proposed by the new ASTM in-soil plastic biodegradation standard D5988-12 and WK29802) (ASTM, 2012b), and (ii) the soil quality should not be impacted negatively (Corbin et al., 2013a, 2013b; Goldberger et al., 2013; Miles et al., 2009). Currently, use of biodegradable plastic mulch in U.S. certified organic production remains prohibited, however, these materials may be allowable in the near future and the ASTM in-soil plastic degradation standards (D5988-12 and WK29802) may be used to assess product suitability for use in certified organic production. Sufficient data will be needed to verify that each different biodegradable mulch product is truly biodegradable in an agricultural system (Corbin et al., 2013b).

The majority of studies on the effects of polyethylene and biodegradable films on crop and soil properties have been conducted during the growing season (Briassoulis et al., 2013; Briassoulis and Dejean, 2010; Kasirajan and Ngouajio, 2012). In contrast, few studies have evaluated the impacts on soil biological and chemical properties during decomposition of biodegradable mulches (Bailes et al., 2013; Cowan et al., 2013; Kapanen et al., 2008; Moreno and Moreno, 2008). During *in situ* decomposition of biodegradable films that are C-rich but nutrient poor, microbial community composition and functionality may be altered. Since the decomposition of C-rich residues is associated with N immobilization, subsequent plant growth may be affected, especially if partially degraded mulch fragments exist and are decomposed at the next cropping cycle. Additionally, a 'priming effect' may occur in which native soil organic matter is mineralized at accelerated rates in response to the pulse of added C (Kuzyakov, 2010) or as a result of increased soil temperatures associated with mulches at the surface (Li et al., 2004). Soil microbial biomass, as measured by ATP concentration, C mineralization rates, and the ratio of CO_2 production to ATP concentration were lowest following incorporation of polyethylene films compared to biodegradable films or bare soil in a tomato cropping study in Italy (Moreno and Moreno, 2008). The authors attributed these reductions in soil microbial properties to high soil temperatures under the mulches and as a reflection of progressive depletion of soil organic matter and overall microbial activity.

Soil quality can vary due to both inherent and dynamic attributes of a soil. Inherent attributes (e.g., texture, climate, slope) are factors that change little, if at all, with land use or management practices, and reflect the basic five soil-forming factors (climate, organisms, relief/topography, parent material, and time) proposed by Jenny (1941). On the other hand, dynamic properties of soil quality (e.g., organic matter, microbial biomass C and bulk density) are more responsive to changes in land use or management practices but the magnitude and rate of change are constrained by inherent soil properties (Larson and Pierce, 1991). The soil management assessment framework (SMAF) provides a single soil quality index (SQI) value, and takes into account multiple soil physical, chemical, and biological indicators that represent different soil functions (Karlen and Stott, 1994). The SMAF also allows for site-specific interpretation because indicators can be chosen and the model can be modified based on specific sets of parameters and management goals. The scores for each indicator are based on inherent soil properties, climate and crops, which then translate into unitless values that then are combined into an overall value ranging from 1 (lowest) to 100 (highest) (Andrews et al., 2004). Currently, SMAF includes 13 indicators with scoring curves, of which five were used in this study: pH, electrical conductivity, total organic carbon (TOC), microbial biomass C, and β-glucosidase activity (Stott et al., 2010). These indicators were chosen because of their functional importance in agricultural production. For example, soil pH influences nutrient availability and microbial activity, and thus affects nutrient cycling. Soil electrical conductivity is an indicator of salinity, which can negatively impact plant growth and is responsive to fertilizer applications. Soil organic matter, (measured as TOC for the SMAF), acts as a reservoir of nutrients and water, reduces bulk density, and provides habitat and nutrients to microorganisms. Because TOC changes relatively slowly, measurements of soil microbial biomass C provide early indications of future changes in TOC, and thus, may facilitate early detection of changes resulting from shifts in field management practices (Powlson et al., 1987). Soil enzymes also are sensitive indicators of management changes and are responsible for plant residue decomposition and nutrient cycling (Dick et al., 1996). Soil β-glucosidase catalyzes the final step of cellulose degradation, releasing glucose monomers, which universally fuel cellular metabolism (Deng and Popova, 2011). Soil β-glucosidase activity is a measure of metabolic capacity, and has been used as a soil quality indicator for over two decades (Acosta-Martinez et al., 2007; Dick et al., 1996).

The objectives of this study were to determine: (1) whether SQI, using the SMAF model, was affected by buried mulch treatments over an 18-month *in situ* incubation, (2) the effect of two tomato (*Lycopersicon esculentum*) production systems (high tunnel and open field) on SQI, and (3) the relationship between SQI and the extent of mulch degradation as measured previously (Li et al., 2014). In order to maximize potential differences in environmental and soil conditions in this study, the experimental design was established at three contrasting locations (Knoxville, TN, Lubbock, TX, and Mount Vernon, WA).

2. Materials and methods

2.1. Experimental locations and agricultural production systems

The locations and agricultural systems used in this study are described in detail in Li et al. (2014) and Miles et al. (2012). Briefly, experimental field plots were established at three geographically distinct locations which differed primarily by climate and soil type: (1) the University of Tennessee, East Tennessee Research & Education Center at Knoxville, which receives an average of 1355 mm precipitation year^{-1}, and has Dewey silt loams (fine, kaolinitic, thermic Typic Paleudults) (Soil Survey Staff, 2011); (2) the Texas A&M AgriLife Research & Extension Center at Lubbock, which receives 475 mm precipitation year^{-1}, and has Acuff loams (fine-loamy, mixed, superactive, thermic Aridic Paleustolls) and Olton clay loams (fine, mixed, superactive, thermic Aridic Paleustolls); and (3) the Washington State University Northwestern Washington Research & Extension Center at Mount Vernon,