UNIVERSIDAD POLITÉCNICA DE MADRID
Escuela Técnica Superior de Ingeniería Agronómica, Alimentaria y de Biosistemas

Agronomic Performance and Ecosystem Services of Co-composted Biochar-based Substrates for Rooftop Agriculture in Madrid

DOCTORAL THESIS

Submitted for the degree of Doctor by:

Giuseppe Picca
M.Sc. in Sustainable Management of the Mediterranean Countryside

Madrid, 2024

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Abstract

Unprecedented urbanization and growing awareness of environmental consequences have recently revived the historical relationship between cities and agriculture. Urban agriculture (UA) is increasingly recognized for its potential to transform urban ecosystems, enhance local food security, and reduce dependence on an energy-intensive food system. However, in-ground UA still faces significant constraints to its spread; the high profitability of building construction creates a competitive environment for space, often relegating urban farmers to marginal, potentially contaminated vacant lots, posing risks to food safety.

Among various forms of UA, Rooftop Agriculture (RA) offers a viable solution by retrofitting underutilized rooftops with soilless productive systems. The rise of RA underscores the need for a growth substrate that is fertile, physically stable, and lightweight to manage structural weight constraints. Due to its optimal physical and chemical properties, peat is currently the primary growing medium in soilless systems. However, the environmental concerns associated with using this non-renewable resource have driven research into alternative organic and inorganic materials. Composting the organic fraction of municipal solid waste enables the recycling of nutrient-rich by-products as RA substrates, aligning with the circular economy paradigms proposed by the European Union. However, organic substrates are prone to gradual decomposition, leading to diminished physical and chemical characteristics and failing to ensure favourable conditions for plant growth over time.

To address this gap, this thesis proposes using biochar, a carbon-rich material derived from biomass pyrolysis, as a structural agent in compost-based substrates. This innovative approach holds significant potential for enhancing the properties of substrates for RA.

Two exploratory studies were conducted before scaling up the experiment on the green roof. The first study tested the potential of coffee silverskin (CS) to produce high-quality compost. Four composting mixtures were prepared by mixing CS with pruning waste and biochar at different ratios, aiming to maximize the amount of compostable CS per batch. These mixtures were monitored for 60 days. The contents of macro-, micro-, and trace elements in the final composts met the strictest requirements of the Spanish national regulation on compost quality (Class A amendments), proving that CS composts are high-value amendments rich in nitrogen (N) and potassium (K). The four composts had a high water-holding

capacity (237-351 % dry weight), potentially promoting the persistence of plant-available water in the media.

Despite the highly phytotoxic effect of raw CS material, seed germination tests showed that all the mature composts exhibited phytostimulant properties, allowing their harmless application as potential substrates. The study revealed that composting all the CS produced in Europe would recover 2,420 to 3,481 tons of nitrogen and 1,873 tons of potassium, reducing dependency on mineral fertilizers and meeting the growing demand for sustainable and low-cost amendments.

The second study investigated the potential of two coffee ground-based composts, with and without biochar addition, as peat replacements. The composts were mixed with a peat-based control at different volumetric proportions: 100 % (pure compost), 50 %, 30 %, and 15 %. To assess the agronomic potential, the compost-based substrates were tested during three stages of tomato (*Solanum lycopersicum* L.) plant development: seed germination (0-6 days), seedling development (7-49 days), and growth up to fruit maturity (36-100 days). While the substrates had a stimulative effect on seed germination, they also induced stunted growth due to the elevated electrical conductivity of the growing media. Compost and biochar-blended compost mixed with peat at 50 % promoted higher fruit productivity than the peat control. The study highlighted that combining biochar with spent coffee ground compost represents a potential alternative to peat-based growing media, promoting a circular production model in the RA sector.

The encouraging results recorded in the preliminary studies enabled the expansion of the study to the rooftop of the Instituto de Ciencias Agrarias (ICA-CSIC) in Madrid. Six different substrates from organic wastes (spent coffee grounds, coffee silverskin, and seaweed) composted with and without biochar were mixed with peat to test their potential replacement rate in RA. The substrate fertility, crop production, and quality were evaluated in a three-year crop rotation of a tomato landrace of the Community of Madrid Madrid (*Solanum lycopersicum* L., cv. Moruno de Aranjuez) and a consociation of lettuce (*Lactuca sativa* L., cv. Romana) and chard (*Beta vulgaris* var. Cicla).

First-year results showed up to a 102 % and 9 % increase in tomato production compared to peat-based and open-air soil-based control systems. For lettuce and Swiss chard, compost-based substrates exhibited production up to 380 % and 217 % higher than the peat-based control, respectively. From the second year onwards, total production decreased for all the studied crops; despite this, the tested

substrates confirmed higher productivity than the two controls, with quality parameters and secondary metabolite levels comparable to open-field systems.

The study confirmed the hypothesis formulated at the beginning of this thesis. Urban waste co-composted with biochar is a viable alternative for creating sustainable substrates for RA, with the potential to achieve considerable productivity and reduced environmental impact.

Resumen

La urbanización sin precedentes que estamos viviendo y la mayor conciencia social de los problemas ambientales han conducido a un resurgimiento de la relación histórica entre las ciudades y la agricultura. La agricultura urbana (UA) está recibiendo cada vez más atención por su potencial para transformar los ecosistemas de las ciudades, mejorar la seguridad alimentaria en las mismas y reducir su dependencia de un sistema alimentario intensivo energéticamente. No obstante, la expansión de la UA sobre suelo se enfrenta a restricciones significativas; la alta rentabilidad de la construcción de edificios crea un entorno muy competitivo por el espacio, lo que a menudo relega a los agricultores urbanos a terrenos marginales potencialmente contaminados, suponiendo riesgos para la seguridad alimentaria.

Entre las diversas formas de UA, la agricultura en azoteas (RA) ofrece una solución viable, al convertir espacios infrautilizados en un sistema productivo sin suelo. Para el impulso de la RA se necesita un sustrato de cultivo fértil, físicamente estable y lo suficientemente ligero como para cumplir las limitaciones de peso estructural. Debido a sus propiedades físicas y químicas óptimas, la turba es actualmente el medio de cultivo principal en los sistemas sin suelo. Sin embargo, los problemas ambientales asociados al uso de este recurso no renovable han impulsado la búsqueda de materiales orgánicos e inorgánicos alternativos. El compostaje de la fracción orgánica de los residuos sólidos urbanos permite reciclar subproductos ricos en nutrientes como sustratos para la RA, en línea con los paradigmas de economía circular propuestos por la Unión Europea. Sin embargo, los sustratos orgánicos sufren una descomposición gradual, que conduce a un empeoramiento de sus propiedades físicas y químicas y no permite condiciones favorables para el crecimiento de las plantas a lo largo del tiempo.

Para abordar esta problemática, esta tesis propone el uso de biochar, un material rico en carbono derivado del pirólisis de biomasa, como agente estructurante de sustratos hechos de compost. Este enfoque innovador tiene un potencial significativo para mejorar las propiedades de los sustratos para la RA.

Se llevaron a cabo dos estudios exploratorios antes de ampliar la experimentación a una azotea verde. El primer estudio tuvo como objetivo comprobar el potencial de la cascarilla de café (CS) para producir compost de alta calidad. Se prepararon cuatro mezclas de CS con restos de poda y biochar en diferentes proporciones, con el fin de maximizar la cantidad de CS. Estas mezclas se monitorizaron durante 60 días. El contenido de macro y micronutrientes y elementos traza de los productos

finales cumplió con los requisitos más estrictos de la normativa española sobre calidad (enmiendas de Clase A), demostrando que los compost de CS son enmiendas de alto valor, ricas en nitrógeno (N) y potasio (K). Los cuatro compost mostraron una alta capacidad de retención hídrica (237-351 % sobre peso seco), que podría promover la persistencia de agua disponible para las plantas en el medio.

A pesar del efecto altamente fitotóxico del CS sin tratar, pruebas de germinación de semillas mostraron que todos los compost maduros presentan propiedades fitoestimulantes, lo que indica su seguridad como sustratos. El estudio reveló que compostar toda la CS producida en Europa permitiría recuperar de 2,420 a 3,481 toneladas de nitrógeno y 1,873 toneladas de potasio, reduciendo la dependencia de fertilizantes minerales y satisfaciendo la creciente demanda de enmiendas sostenibles y de bajo coste.

El segundo estudio tuvo como objetivo investigar el potencial de dos compost de posos de café, uno con y otro sin adición de biochar, como sustitutos de la turba. Los compost se mezclaron con turba en diferentes proporciones volumétricas: 100 % (compost puro), 50 %, 30 % y 15 %. El potencial agronómico de los sustratos de compost se evaluó durante tres etapas del desarrollo de la planta de tomate (*Solanum lycopersicum* L.): germinación de semillas (0-6 días), desarrollo de plántulas (7-49 días) y crecimiento hasta la madurez del fruto (36-178 100 días). Aunque los sustratos tuvieron un efecto estimulante en la germinación, también redujeron el crecimiento debido a la elevada conductividad eléctrica del medio de cultivo. El compost y el compost mezclado con biochar y turba al 50 % promovieron una mayor productividad de frutos en comparación con el control de turba. El estudio señaló que combinar biochar con compost de posos de café representa una alternativa potencial a la turba como medio de cultivo, promoviendo un modelo de producción circular en el sector de la RA.

Los resultados registrados en los ensayos preliminares alentaron la expansión del estudio a la azotea del Instituto de Ciencias Agrarias (ICA-CSIC), situado en Madrid. Seis sustratos diferentes provenientes de residuos orgánicos (posos de café, cascarilla de café y algas marinas) compostados con y sin biochar se mezclaron con turba para analizar hasta qué punto este material se puede reemplazar en RA. La fertilidad del sustrato y la producción y calidad de los cultivos se evaluaron en una rotación de tres años de una variedad de tomate de la Comunidad de Madrid (*Solanum lycopersicum* L., cv. Moruno de Aranjuez) y una asociación de lechugas (*Lactuca sativa* L., cv. Romana) y acelgas (*Beta vulgaris* var. Cicla).

Los resultados del primer año mostraron un aumento del 102 % y del 9 % en la producción de tomate en comparación con los controles de turba y de campo, respectivamente. Para lechugas y acelgas, los sustratos de compost exhibieron una producción hasta 380 % y 217 % mayor que la de la turba. A partir del segundo año, la producción total disminuyó; sin embargo, los sustratos siguieron mostrando una mayor productividad que los controles, con parámetros de calidad y niveles de metabolitos secundarios comparables a los obtenidos en campo.

El estudio confirmó la hipótesis formulada al inicio de esta tesis. Los residuos urbanos co-compostados con biochar son una alternativa viable como sustratos sostenibles para la RA, con potencial para lograr una producción considerable y un impacto ambiental reducido.

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Abbreviations and Acronyms

BC	Biochar
BHT	Butylated Hydroxytoluene
BD	Bulk Density
CAT	Calcium Chloride/DTPA
CG	Spent Coffee Grounds
CO_2e.	Carbon Dioxide Equivalent
CS	Coffee Silverskin
CRFS	City Region Food System
DTPA	Diethylenetriaminepentaacetic Acid
EC	Electrical Conductivity
EDTA	Ethylenediaminetetraacetic
EU	European Union
FRAP	Ferric Reducing Antioxidant Power
GW	Green Waste
HS	Commercial Horticultural Substrate
ICA	Instituto de Ciencias Agrarias de Madrid
IMIDRA	Instituto Madrileño de Investigación y Desarrollo Rural, Agrario y Alimentario
MUFPP	Milan Urban Food Policy Pact
NEDD	N-(1-Naphthyl)ethylenediamine dihydrochloride
UN	United Nations
RA	Rooftop Agriculture
SW	Seaweed
TA	Total Acidity
TAC	Total Antioxidant Capacity
TP	Total Phenols

TSS	Total Soluble Solids
TPTZ	2,4,6-Tris(2-pyridyl)-s-triazine
UA	Urban Agriculture
WHC	Water Holding Capacity

PART I

Introduction, Objectives and Methodology

Chapter 1

The Little Garden of Paradise by the Upper Rhenish Master (Städel Museum - Frankfurt, Germany).

Historical Background

1. Historical Background

The recent environmental and economic crises have renewed the interest in the role of agriculture in cities. The COVID-19 epidemic and the Ukraine-Russia conflict have highlighted the food system's vulnerability and the need to rethink the food supply chain. Beyond the geopolitical implications in primary sector supplies, the United Nations 2030 Agenda and particularly the goals focused on achieving food security and sustainability have proven essential in recent years (United Nations, 2015). As urbanization increases rapidly, urban agriculture (UA) is becoming central in the debate between academia, non-governmental organizations and public administration, contributing to food security, food system resilience and cities' sustainability (Caputo, 2022). The relationship between cities and agriculture is back to weld as it evolved into a mutually causal and symbiotic relationship.

The present chapter aims to guide the reader through the history of this relationship between cities and agriculture.

1.1. From the Neolithic Revolution to the Modern Era

As archaeologist Vere Gordon Childe theorized, the Neolithic and the Urban Revolutions marked the transition from a "savagery" to a "civilized" society. The Neolithic Revolution, also called Agricultural Revolution - about 10,000 years ago - described the path of small nomadic bands from hunting and gathering to larger sedentary settlements (Childe, 1950). The process relied on the gradual adoption of agriculture by domesticating the wild progenitors of cereals and legumes and animal breeding (Bellwood, 2005). The implementation of agriculture and the technological innovations that enhanced productivity, such as irrigation systems, generated a surplus in production that could be used for saving or trading. Guaranteed food security played a crucial role in stimulating the increase in the population of these settlements. The reorganization of these early farmers had a significant transformation of society and government institutions, enhancing the Urban Revolution - about 5,000 years ago. One of the hallmarks of the city's rise was the construction of defensive fortifications for possible threats of military hostilities. In this new urban organization, the agricultural plots within and around the urban boundaries provided the primary share of the city's subsistence, and constituted the ancient roots of modern urban agriculture (Barthel & Isendahl, 2013).

From the first traces in Mesopotamia, history offers proofs that agriculture was not antithetical to the city; on the contrary, it was a fully integrated urban activity, which not only guaranteed food and forage yield but was also relevant for producing fencing, building materials, ornamentals, and medicinal herbs (Elmqvist et al., 2013). UA flourished from ancient Egypt to China, from the walled gardens of Persia to Babylon's hanging gardens, reaching Greece, where in the classical cities, the sacred lands were committed to food production (Orsini et al., 2020) (Figure 1.1).

Figure 1.1: The Hanging Gardens of Babylon by Ferdinand Knab (Public Domain).

In the Roman society, the city was surrounded by large suburban farms (owned by the nobles) organized according to agroforestry principles, the production of which was traded in the city markets (Scazzosi, 2020). Fascinating models of UA were also noted in the pre-Columbian civilizations. The Aztecs created the "*chinampas*", artificial floating islands by alternating layers of mud and decaying vegetation for cultivating on the lakes which surged their cities (Torres-Lima et al., 1994).

At the same time, the Incas constructed terraced farms in Machu Picchu, enabling cultivation in the Peruvian Andes (Wright, 2021).

In the Medieval Age, UA flourished in the form of the "*hortus conclusus*" adopting the geometric principles of the Pythagorean school. These gardens were mainly associated with monasteries and convents, where food production and processing were established under the Rule of Saint Benedict. The *horti* were often placed between the defensive walls, completing the geometry of villages and towns and providing sustenance for the local community (Orsini et al., 2020). Marketplaces offered another testimony of UA's contribution to the city's cultural construction process. These locations, used for several functions, including commercializing local production, are still included in the cityscape of many Western historical city centres, retaining their original function (Barthel et al., 2015; Scazzosi, 2020).

Continuing the study of the iconography of paintings and historical maps of European cities from the Modern Ages, it is evident that farmlands within and beyond urban boundaries created a continuum between rural and urban areas, demonstrating the ability of UA to structure the city's territory and shape its landscape (Scazzosi, 2020). However, the relationship started to crack in the subsequent centuries as a consequence of socio-economic upheavals.

1.2. The Industrial Revolution

In the 18th and 19th centuries, the socio-economic and technological changes from the Industrial Revolution profoundly transformed the city's face. During the transition from feudal agrarianism to urban industrialism, city expansion rapidly shrank farms and orchards, replacing them with more profitable industrial and residential buildings. The new urban areas proliferated without proper planning or conditions for social welfare, being congested, polluted, and unsanitary (Bell et al., 2016). Massive economic migration from rural areas raised a significant social problem; these families were living under deplorable conditions in large slums and suffering from malnutrition (Iaquinta & Drescher, 2015). The social and environmental situation prompted the new idea of reshaping industrial cities. The common opinion was to allot - hence the term "allotment garden" - small plots established in open spaces provided by municipal authorities, churches, and employers to poor or working families to produce food, fibres and wood, in addition to dissuade the scourge of alcoholism (Bell et al., 2016). The British urbanist Ebenezer Howard shed light on the uppermost importance of public agricultural and green areas within cities in the manifesto of the garden city movement, Garden Cities of Tomorrow (1902), theorizing that the town had to be food-self-sufficient and recycle its food waste into green manure to be reused (Cabannes & Ross, 2018).

The allotment gardens were first introduced in the UK in 1795, soon spreading all over the Country, increasing from 100,000 in 1850 to 448,586 in 1890. The great success of the practice required first the enactment of The Allotment Extension Act (1887), allowing local authorities to acquire land to provide as allotment gardens, and the municipality to force private landowners to sell land for shared urban gardening purposes. After that, the Small Holdings and Allotments Act of 1908 made local councils responsible for providing allotments to the labouring class, meeting the constant demand for gardens (Mees, 2017a).

In Germany, the municipal *Armengärten*, or "garden of the poor", started to appear in 1814 in Kiel, Berlin, Leipzig and Frankfurt, among others, allowing citizens to grow their own vegetables. The popularity of allotment was prompted by the idea of Dr Daniel Gottlob Moritz Schreber (1808-1861), an orthopaedist who investigated the physical and moral benefits for the urban youth to do outdoor activities. In 1864, the pedagogue Dr Ernst Innozenz Hauschild created the first association in honour of his colleague in Leipzig, consisting of a play area for health relief and an orchard for education and healthy production; soon, families took over the areas to supplement their weak diets and the idea of *Schrebergarten* “family garden” (also called *Kleingartens* “small gardens”), proliferated in the rest of Germany (Cabral et al., 2017; Van Molle & Segers, 2008).

In France and Belgium, the *Jardin Ouvriers* originated from the initiative of the French priests Jules Auguste Lemire and Léon Gruel, and the Belgian publisher Joseph Goemaere, who founded in 1897 the *Ligue Française du Coin de Terre* and the *Ligue du Coin de Terre et du Foyer Insaisissables* in 1899. In both countries, the movement had a pronounced anti-revolutionary and religious character, considered a political instrument for social order. The founders emphasized that gardening activities would have strengthened family relationships and, by improving the financial benefits for workers, limited the insinuation of subversive ideas: the title of the Belgian Ligue manifesto was clear evidence: *Plus de socialistes*! (No more socialists!). In 1910, there were 17,000 urban orchards in France and 2,000 in Belgium (Van

Molle & Segers, 2008). In those years, a virtuous and interesting example of UA was established in Montreuil sous Bois, a suburb of Paris, France: 23 hectares of farming land, mainly consisting of *murs à pêches* - peach walls. The structures were composed by walls covered with white plaster to reflect solar radiation providing extra-heating and allowing the cultivation of peaches and other fruit tree in the cold Parisian climate. To date, it remains the centre of a conservation project aimed at expanding the area dedicated to UA activities in the French capital (Mancebo, 2018).

Urban gardening spread to the rest of Europe and North America. UA started to appear in the United States in response to the economic depression in the 1890s (Iaquinta & Drescher, 2015). The Mayor of Detroit, Haze S. Pingree, leased municipal land to families of unemployed workers. The initiative became known as Pingree Potato Patches and was replicated in 25 cities, among which Chicago, Philadelphia, New York, Omaha and Baltimore (Mees, 2017b; Van Molle & Segers, 2008). From the end of the 20th century, three different urban garden programmes in the USA were established: vacant-lot cultivation associations - imitated in Canada -, children's school gardens and civic garden campaigns (Bell et al., 2016; Van Molle & Segers, 2008). Even downsized and outcast to small unused or unusable areas for industrialization, UA presented a new function: provide poverty alleviation and food security for the disadvantaged (Barthel et al., 2015; Dobele & Zvirbule, 2020).

1.3. War Gardens and the Great Depression

In the first half of the 20th century, the global geopolitical framework was distraught by the World Wars and the economic depression. With farmers called to the war front and large areas devastated by fighting, many cities were cut off from rural sources of agricultural production (Iaquinta & Drescher, 2015). Britain, France, Italy, and Belgium could count on only 60 % of their average wheat production (Mok et al., 2014). Allotment gardens were crucial for alleviating food shortages. Their diffusion peaked through the introduction of national campaigns that authorized the conversion of vacant lands, urban lots, parks and other urban areas into gardens. The National War Garden Commission promoted the creation of shared urban gardens as "Freedom Gardens" or "Liberty Gardens", encouraging the "Soldiers of the Soil" to grow food and ensuring the export of significant commercial production for the Allied forces in Europe; in the biennium 1917-1918, 3.5 million war gardens produced 875 million US dollars' worth of food (Mees, 2017b; Mok et al., 2014) (Figure 1.2). The idea inspired the British government to promote the "Every Man a Gardener" campaign to face the mass starvation that the Country was experiencing due to the German U-boat operation of torpedoing merchant ships. By 1918, there were 1,500,000 allotment gardens in the UK, providing Britons with 2 million tons of fresh vegetables (Barthel et al., 2015).

Figure 1.2: US government War Garden propaganda from World War I. (Public Domain).

The policy of urban gardens was maintained during the post-war to cope with the hardships of the Great Depression (1929-1939). In Germany, the *Schrebergarten* number rose to 450,000 (Barthel et al., 2015). The economic situation of low-income residents was so disastrous that gardeners used their parcels for dwelling purposes and building shelter structures (Mees, 2017a). In the USA, the established Relief Gardens were distinguished as allotments or shared urban gardens. They provided food and income to thousands of unemployed people, alleviating the loss of morale and self-respect (Mok et al. 2014; Mees 2017b).

UA was also critical in Spain during the Civil War (1936-1939) when food supplies became scarce in the Republican zone. In Barcelona and Madrid, the local population started cultivating urban and peri-urban lands, converting small plots and flower beds, with women playing a pivotal role as food producers and providers (Calvet et al., 2021). In the Catalan city, the activities promoted by the anarcho-syndicalists CNT through the *Colectividad Agrícola de Barcelona y su Radio* involved 3,500 workers, between unemployed farmers and war refugees, managing 1,000 ha of urban and peri-urban land, providing 90 % of the vegetables sold in the city (Camps-Calvet et al., 2022). In 1938, the Catalan Government approved a decree to restrain the use of public land for its transformation into family gardens, providing a maximum of 150 m^2 for families and prohibiting the market of the obtained products. During the conflict, 10,000 plots were in town, with the *Agrupació d'Hortolans de la Muntanya de Montjuïc* representing the largest group, including about 3,000 plots (Camps-Calvet et al., 2022). Gardens were also started in schools or refugee shelters for the younger, where the provisioning function of UA merged with its educational aspect and had a propaganda implication. The occupation of Barcelona by Franco's troops caused the disbanding of the *Colectividad Agrícola*; however, the experience of urban gardens in town survived to the present day (Calvet et al., 2021).

During the Second World War (WWII), the same campaigns of WWI were adopted. In the US, the War Food Administration supported the National Victory Garden Program; First Lady Eleonore Roosevelt's "Victory Garden" at the White House inspired the cultivation of 20 million orchards that accounted for 40 % of the nation's fresh products. In the UK, during the "Dig for Victory" campaign, the Tower of London moat was adopted for cultivation to symbolize the importance of UA for the entire Country, raising the number of allotment gardens to 1,400,000, providing British citizens 1,300,000 tons of food (Figure 1.3).

Figure 1.3: US government Victory Garden (above) and UK government Dig for Victory (below) propaganda posters from World War II (Public Domain).

Meanwhile, the Commonwealth Department of Commerce and Agriculture founded the "Grow Your Own" campaign in Australia, increasing the vegetable production area to 18,378 and 3,529 ha in Victoria and Western Australia, respectively (Mok et al., 2014).

In the Soviet Union, during their advance, the Nazis devastated government food reserves throughout the Country, and took control of key grain-producing areas, such as Belarus. Struggling to secure minimum sustenance, the central government had to prioritize provision for the Red Army, rationing food for the civilians. In response to the massive starvation and malnutrition suffered by the home front, the Soviet state promoted the urban gardening movement, boosting individual and collective gardens (*ogorody*). The distinctive feature of the soviet movement was the mobilization of millions of women, leaning on their identification with the allotment, family and Mother Russia (*Rodina-mat'*) (Charon Cardona & Markwick, 2019). The initiatives launched by the Allies' forces, strongly binding gardening and the language of war, had analogue functions: improved food security, reduced demand for commercial production that could have been supplied to the armies, and boosted national morale.

In Germany, under the rule of the National Socialist Party, UA was reframed by the nazist blood and soil (*Blut- und Bodenideologie*) ideology, inviting the racially pure dwellers to reconnect with the soil from which their race was born (Van Molle & Segers, 2008). Despite the appropriation of food from occupied territories during the expansionist aggression, the state seized the allotment to achieve autarchy, restricting membership to Germans of Aryan origin. By the war's end, the number of gardens in the Country had grown to 800,000 (Barthel et al., 2015); with the Country torn apart by the advance of the allies, *Kleingartens* continued to be used also as accommodation for many families, becoming increasingly necessary for large parts of the urban population (Figure 1.4) (Mees 2017a).

Figure 1.4: Schrebergarten by the ruins of the Reichstag, 29 May 1946 (Fred Ramage; dpa/akg-images®, Frankfurt, Germany).

1.4. Decline and Rise of Urban Agriculture

The post-war was a period of relative peace and political stability, characterized by welfare policy, solid economic growth and full employment conditions that increased financial resources. In this new scenario, UA gradually lost the unanimous recognition it had covered in facing the global crisis, and instead was considered a symbol of inferior social and economic conditions and an element of landscape degradation.

A new consumerist lifestyle was introduced in contrast to poverty and the needs suffered during the war (Mok et al., 2014). In the agri-food sector, technological advances and supportive policies for farmers - such as the Common Agricultural Policy (CAP) in Europe - increased food production, prompting reduced commodity prices. A modernist approach to urban life separate from rural life stood out, decoupling cities from their food supply sources. The affirmation of this idea was strongly linked to innovations in transportation technology that made food transportation over great distances possible (Elmqvist et al., 2013).

The publishment of "Silent Spring" (1962) by Rachel Carson and "The Limits of Growth" by the Club of Rome (1972) created a profound environmental awareness about the over-exploitation of nature in the 1960s-1970s, highlighting the limits of the current economic model. The first articles on organic and ecological gardening were published to warn about the dangers of insecticides and pesticides and the excessive use of artificial fertilizers (Van Molle & Segers, 2008). The new consciousness prompted the formation of environmental and counter-culture movements that rekindled the interest in urban gardening, becoming increasingly connected to environmental and food justice activism. In the late 1960s, urban gardens re-emerged in North America as community open spaces called "Counter-Culture Gardens" (Mees, 2017b).The inflation and unemployment caused by the oil crisis in 1973 set up a process of degradation and abandonment of residential spaces in American cities, especially in low-income neighbourhoods. New York City residents built the first community gardens through green guerrilla actions. The resonance of these actions prompted the city council to open the GreenThumb Municipal Agency to manage the public land for community gardens and orchards (Smith & Kurtz, 2003).

However, with the Bruntland Report (Our Common Future) by the United Nations (Commission on Environment, 1987), UA took on a new central role in the political panorama. The document, introducing the word "sustainability" into the common vocabulary, noted the function of cities in guaranteeing future sustainable development. It depicted UA as one of the tools for reaching the goal, emphasizing its ecosystem services – the benefits that people obtain from ecosystem functioning (Millennium Ecosystem Assessment, 2005). This event marked the renaissance of UA, which took on a leading part in the modern environmental action agendas (Dobele & Zvirbule, 2020)

Chapter 2

Urban Garden in Nantes, France (Giuseppe Picca®)

Current Situation of Urban Agriculture

2. Current Situation of Urban Agriculture

The chapter addresses the global issues stemming from urbanization and introduces the theme of urban agriculture as a tool to ensure the sustainable development of our cities. Finally, it emphasizes the potential limitations of its diffusion.

2.1. Urbanization

In 2007, for the first time in human history, the world's population living in urban areas overtook the rural inhabitants (Figure 2.1) (United Nations, 2018b). With the rise of the *Homo Urbanus,* the present is described by West (2017) as the Urban Anthropocene or the Urbanocene.

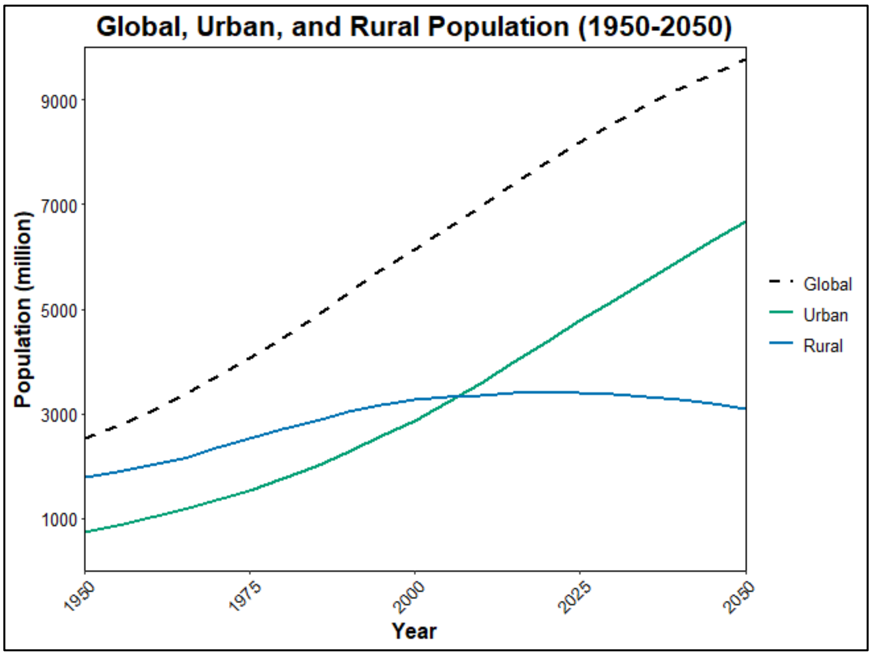


Figure 2.1: United Nations, Department of Economic and Social Affairs, Population Division (2018). World Urbanization Prospects: The 2018 Revision, custom data acquired via website.

The unprecedented urbanization - net rural to urban migration - is taking part with sensitive geographical differences, with growth rate variations even more significant among individual countries or municipalities (Wiskerke, 2015). The world's urban population has proliferated between 1950 and 2018, increasing from 751 million to 4.2 billion; to date, the Americas, Europe, and Oceania are the most urbanized areas. By 2050, urban dwellers are projected to grow by 2.5 billion, with 90 % of the increase happening in Asia and Africa (United Nations, 2018b). The demographic change is and will occur in medium-sized (1–5 million residents) and small (less than 1 million residents) urban settlements, merging multiple built-up areas into a continuous city network, leading to the constitution of mega-urban regions (D'Amour et al., 2017; United Nations, 2018b).

Global urbanization poses a serious impact on food security since it causes a direct loss of cropland (Barthel et al., 2019), and an associated loss of ecosystem services provided by soils such as carbon storage and food production (García-Nieto et al., 2018). Historically, towns have been compacted and rationally growing around the most fertile world regions due to the dependencies on local agricultural production (Langemeyer et al., 2021; Seto et al., 2013). Today, cities are expanding unevenly twice as fast as urban populations through peri- and ex-urban landscapes, displacing agriculture production to less suitable areas and requiring more resource and energy inputs. From 1970 to 2010, over 60 % of reported urban expansion (nearly 40,000 km^2) occurred on agricultural land (Güneralp et al., 2020), with this percentage increasing to about 70 % between 1992 and 2015 (Liu et al., 2020). By 2000, urban land accounted for 213 million hectares and by 2040, it is projected to increase to 621 million hectares, rising from 2.06 % to 4.72 % of the Earth's surface; at this rate, 65 million tons of crop production will be displaced by urbanization (van Vliet et al., 2017). Given that urban soils are nearly twice as productive as the global average (D'Amour et al., 2017), cultivating on less fertile land will require the exploitation of larger natural areas and a significant increase in the use of external resources (e.g., mineral fertilizers, water, energy), upon which half of the global food production already relies (Barthel et al., 2019; Langemeyer et al., 2021). Adopting unsustainable practices leads to the depletion of soil nutrients and subsequent soil erosion, exacerbating the already degraded state of global soil health, of which two-thirds are already affected (Andersson et al., 2014). As a vicious cycle, the perturbation of the ecosystems evolves into a series of cascading effects reinforcing climate change.

The geographic decoupling of cities from their food supply sources and the prevailing consumption habits in urban settings also challenge environmental sustainability. Modern urban lifestyles have shifted dietary habits towards higher consumption of meat-based products, fruits, and vegetables (Barthel et al., 2019; Langemeyer et al., 2021). With the global population projected to increase to 9.77 billion by 2050, food production will need to grow by 100–110 % (Langemeyer et al., 2021; United Nations, 2018a). However, even though current production is more than 1.5 times enough to provide food security for the predicted population peak, one-quarter to one-third of the food produced is lost or wasted, with associated emissions of approximately 2.5 Gt of carbon dioxide equivalent (CO_2e) (Caputo, 2022; Holt-Giménez et al., 2012). Without a paradigm shift, wealthy nations would increasingly depend on imports to build food security, relying on food trade or unfair practices (i.e., land-grabbing) with underdeveloped countries; these actions are not without side effects (Barthel et al., 2019). Li et al. (2022) assessed that food imports correspond to about 3.0 $GtCO_2e$ and account for nearly 20 % of total food-system emissions. The flow of fresh products aggravates cities' environmental footprint and the impact will be even more severe considering that by 2050, more than 80 % of food will be consumed in urban areas (Robertson-Fall, 2019).

The role of cities as contributors to global warming is well established; they are responsible for 75 % of global energy consumption, half of global waste production, and 70 % of global CO_2 emissions (International Energy Agency 2021). In this context, cities have been compared to parasite organisms because of their growing ecological footprint, exploiting natural ecosystems while polluting them (Wiskerke, 2015). As the world continues to urbanize, sustainable development depends increasingly on rethinking urban management and food sovereignty.

2.2. Definition of Urban Agriculture

UA is drawing attention to offering synergic products and functions with rural agriculture production, contributing to food security and food system resilience while reshaping the city's ecosystem.

UA is a broad term that includes various agricultural activities, including crop, fruit, herb growth, livestock, and aquaculture, from rooftop gardens to larger cultivated systems (Figure 2.2) (Thebo et al., 2014). In light of the strong interest aroused, the literature has proposed several definitions of UA, even though a universally accepted one does not exist yet.

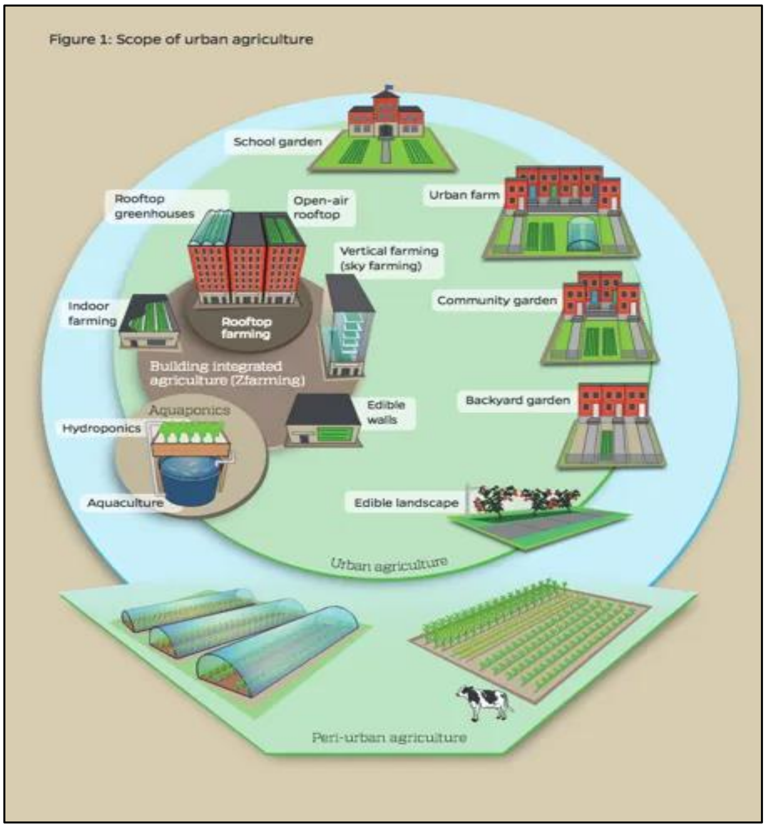


Figure 2.2: Different models of urban and peri-urban agriculture (Santo et al., 2016).

The challenge in defining urban agriculture stems from its extreme versatility. Probably the most used definition is the one proposed by Mougeot (2000), which is a revised version of the one proposed by Smith (1996):

"*Urban agriculture is an industry situated within (intraurban) or on the periphery (peri-urban) of a town, city, or metropolis, engaged in the cultivation, processing, and distribution of a variety of food and non-food products. It largely relies on human and material resources, products, and services available in and around the urban area while also supplying resources, products, and services predominantly to that area.*"

From this definition, three essential aspects of UA are highlighted: (i) the interaction between agricultural practices and the urban environment regarding resources and outcomes, (ii) its diversity of models, (iii) its multifunctionality. In the same work, Mougeot (2000) emphasizes that the crucial factor defining UA as "urban" is its seamless integration into the local urban environment's sociocultural, ecological, and economic fabric (Figure 2.3). Based on this principle, we must comprehend the pivotal role of UA in securing a sustainable future for cities (Figure 2.3).

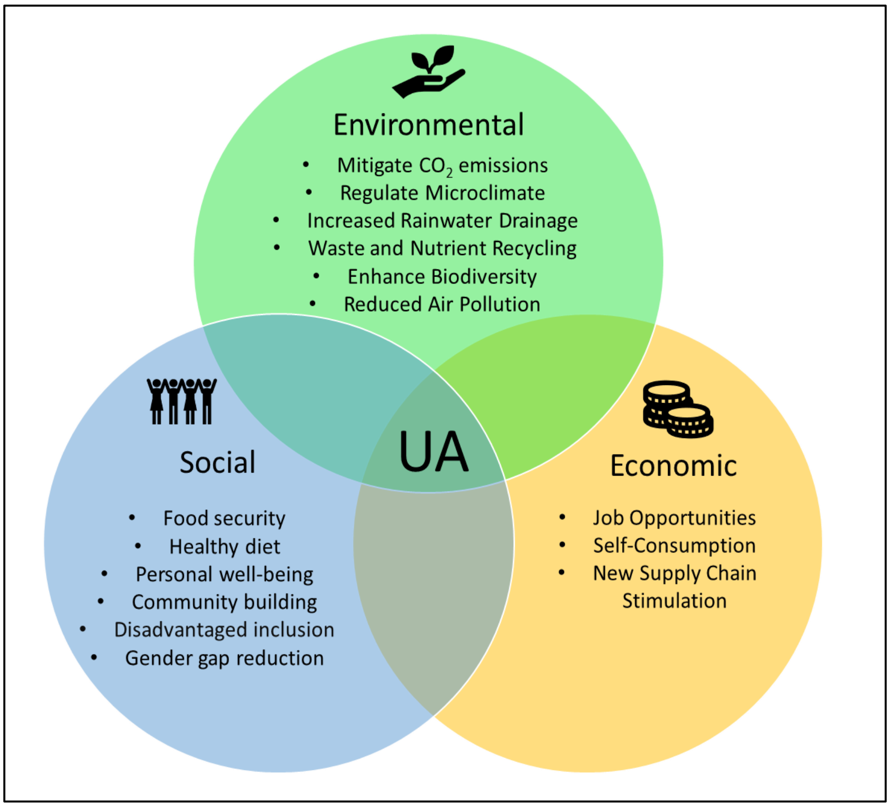


Figure 2.3: The impact of Urban Agriculture on the three pillars of sustainability.

2.2.1. Social Sustainability

The social sustainability pillar encompasses fulfilling essential human needs, with UA being crucial in ensuring food and nutrition security. Defined by the 1996 World Food Summit as "*…when all people, at all times, have physical and economic access to sufficient, safe and nutritious food that meets their dietary needs and food preferences for an active and healthy life*" (FAO, 2006), food security remains a pressing global issue.

In 2022, between 691 and 783 million people worldwide faced hunger, 122 million more than in 2019, and 26 % of adults in urban areas experienced moderate or severe food deprivation (FAO et al., 2023). Urban areas, where citizens are predominantly net food consumers dependent on imports, face significant challenges related to food insecurity, exacerbated by climate change and geopolitical tensions, affecting global food production and supplies.

UA emerges as a vital solution to address these challenges, potentially providing from 1.87% to 150% of vegetables and fruit demands in many cities (Weidner et al., 2019). However, a shift towards integrated policies that combine food production, health promotion, and community participation is required to achieve cities' pivotal role in the global food system. In this context, 148 cities signed the Milan Urban Food Policy Pact (MUFPP), an international protocol to develop food policy strategies based on sustainability and social justice principles (Filippini et al., 2019). The MUFPP complements the commitment of the Food and Agriculture Organization of the United Nations (FAO), which, since 1999, has integrated urban and peri-urban agriculture into agricultural production systems, recognizing its contributing role in feeding cities. The commitment was renewed in 2020 when FAO launched the Green Cities Initiative to enhance both urban dwellers' well-being and the environment, advocating for sustainable and resilient agri-food systems and green spaces in urban and peri-urban areas (FAO, Rikolto, and RUAF2022).

Initiatives have been established in developing countries to improve access to nutritious fresh food. For example, in Dakar, Senegal, the micro-gardening project promotes organic vegetable self-production using recycled materials. In Quito, Ecuador, where urban gardens have a long tradition, producer households consume 57 % of the 1,350 tons of food produced per year, and 43 % is sold through short supply chains, enhancing their livelihoods. Similarly, in Medellin, Colombia, the municipal program has implemented a food security policy targeting disadvantaged groups, particularly schools and the elderlies (FAO et al., 2022).

In developed countries, UA activities focus on providing a healthier diet to citizens, especially in low-income neighbourhoods transformed into "food deserts" by urbanization. These areas lack fresh produce retailers, leading to an increased consumption of sugary/high-fat diets available at fast-food retailers, that are associated with chronic diseases. While the prevalence of this phenomenon is growing, UA initiatives have successfully ensured food access for vulnerable populations by promoting a healthier vegetable-based diet (Horst et al., 2017; Rao et al., 2022). Although the problem is more pronounced in American cities, it is also widespread in Europe, where virtuous projects have been launched. One of them is the "*Huertos Saludables*" (Healthy Orchards) project in Madrid, Spain, which aimed to support community gardens and other civic activities in providing access to fresh healthy food (Filippini et al., 2019). In line with these policies, the municipality of Nantes, France, started "*Paysages Nourriciers*" (Nourishing Landscapes) initiatives, providing free food to 1,000 poor households based entirely on organic agricultural production (FAO et al., 2022).

Beyond its role in ensuring food security and nutrition, UA acts as a catalyst for social cohesion and civic participation, building stronger sense of community. In particular, urban farms and gardening spaces foster integration in multicultural settings, bringing together individuals of different genders and ages from various backgrounds (Santo et al., 2016). Notably, 65 % of urban farmers are women, often in managerial positions (Orsini et al., 2013). This sense of community and belonging is a powerful motivator for engagement and participation.

Urban food production also promotes therapeutic and health benefits to those involved through cultivation and harvesting activities and by fostering the connection to nature via the diffusion of green space in cities, boosting urban liveability (Nicholls et al., 2020). This improved physical and psychological health process, facilitated by the relationship with plants, has been defined as *Therapeutic Horticulture* (Shoemaker & Diehl, 2012). Indeed, urban farmers report greater self-esteem and mental well-being, experiencing less depression and fatigue (Ilieva et al., 2022). Furthermore, urban gardens serve as a paramount tool in therapy and rehabilitation programs for individuals experiencing mental ill-health, conducted with a therapist, as testified by Horticultural Therapy (Shoemaker & Diehl, 2012). Gardening spaces create a safe learning and occupational environment, encouraging the social inclusion of the people involved (Ilieva et al., 2022).

The benefits of urban agriculture extend to education (Santo et al., 2016). School gardens provide an excellent opportunity for interdisciplinary learning, contributing to a holistic approach to learning (Orsini et al., 2013).

Moreover, the garden helps connect students with food production, making them conscious of cultivation cycles and fostering environmental awareness among the younger generation.

2.2.2. Environmental Sustainability

Intense urbanization, characterized by the increasing densification of built-up areas and the loss of natural spaces, rapidly degrades the functionality and services of urban ecosystems, directly impacting biodiversity (Kabisch et al., 2017). Urban areas are suffering the effects of increasingly extreme natural events associated with climate change (e.g., heat waves, cold waves, floods, droughts, wildfires, and windstorms), making urban dwellers highly vulnerable (Kabisch et al., 2017; Kingsley et al., 2021). The prevalence of asphalt and roofing materials with low albedo and high thermal conductivity causes urban areas to absorb and retain more heat than outlying areas. This phenomenon, known as the urban heat island, results in temperatures reaching up to 10-15°C higher than rural surroundings, thereby increasing the risk of heat-related human illnesses and mortality (Mentaschi et al., 2022). Anthropogenic heat production (i.e., vehicular traffic, industries, heating, and air conditioning) exacerbates the effect to such an extent that temperatures have, on average, risen by 1 °C since 2003 in extreme urban heat island (Mentaschi et al. 2022; Mancebo 2018). Simultaneously, the rise of soil sealing and decline in greenery have altered urban hydrological systems, decreased groundwater infiltration and compromised flood management during heavy rainfall events (Senes et al., 2021).

The urgency of climate change, particularly in the context of urbanization, cannot be underestimated. Limiting global warming below 1.5 °C compared to pre-industrial levels is imperative, which necessitates a significant reduction in CO_2 emissions to net zero by 2050 (IPCC, 2022). The European Union (EU) has taken on this challenge by investing 600 billion euros as part of the European Green Deal (European Commission, 2019), committing to reduce CO_2 emissions by 55 % by 2030 and reach zero by 2050 (Fetting 2020). To create a concrete solution to this challenge, EU has established the priority EU Mission “Climate-Neutral and Smart Cities” to drive 100 cities in their path towards climate neutrality by 2030 (European Commission, 2020a).

Within this framework, UA possesses considerable potential to boost cities' resilience against extreme climatic events and shocks, advocating as an integral component of the shift towards more sustainable and liveable urban environments. In-ground and building-integrated non-conditioned UA offers various regulating ecological services. In-ground UA facilitates the revegetation of vacant lots or unused areas, promoting cooling through evapotranspiration—capable of reducing urban temperatures by 0.5–4.0 °C—and provides shade from short-wave solar radiation while prompting faster dissipation of long-wave radiation at night (Qiu et al. 2013; Kleerekoper, Van Esch, and Salcedo 2012). Additionally, enhanced soil permeability from agricultural activities enables water storage and infiltration, thereby decreasing stormwater runoff (Lucertini & Di Giustino, 2021). Building-integrated UA solutions, such as green roofs and walls, are critical in modulating the urban microclimate. Green roofs, for instance, have a cooling effect on ambient temperature ranging from 0.24–4.0 °C and reduce roof surface temperature by 0.8–60.0 °C (Qiu et al., 2013). Since buildings are responsible for 28 % of global energy-related CO_2 emissions (International Energy Agency, 2021), these models can significantly reduce heating and cooling requirements, thereby recycling the excess energy of the building in the production processes, improving thermal energy performance, ensuring energy savings, and reducing associated CO_2 emissions (Nadal et al., 2017). Furthermore, green roofs capture precipitation during rainfall, reducing runoff and returning about 45 % of water to the atmosphere through evapotranspiration (Walters and Midden 2018; Oberndorfer et al. 2007). Revegetating vacant spots, implementing green roofs and walls, filters atmospheric pollutants, and potentially reduces the morbidity and mortality associated with respiratory illnesses.

UA is also deemed an essential tool for achieving low-emission local food systems. Since food production contributes to one-third of global emissions, establishing a local food system would curtail emissions related to food miles while decreasing reliance on external inputs (i.e., petroleum-based energy, pesticides, and chemical fertilisers) (Kulak, Graves, and Chatterton 2013; Crippa et al. 2021). However, Hawes et al. (2024) revealed that the food carbon footprint of UA can be up to six times higher than conventional agriculture (420 gCO_2e versus 70 gCO_2e per serving), depending on the technology applied. In the same study, the authors suggested that the climate impacts of UA can be mitigated by cultivating crops typically grown in greenhouses (e.g., tomatoes) while establishing symbiotic relationships with cities. UA enables recycling organic fractions from municipal solid waste into food production. Organic waste compost can be used as fertiliser or as a substrate for soilless productive systems, simultaneously reducing CO_2 emissions from poor biomass management and increasing carbon sequestration (Grard et al., 2015; Nicholls et al., 2020). Additionally, subsidies for rainwater catchment infrastructure or guidelines for greywater reuse could reduce dependency on municipal potable water and cut emissions related to pumping, water treatment, and distribution—processes accounting for up to 83% of total emissions at one UA site (Hawes et al., 2024).

Finally, UA initiatives contribute to restoring plant and animal biodiversity. Urban farmers play a crucial role in conserving agro-biodiversity by preserving seeds and cultivating more traditional crop varieties and wild relatives (Santo et al., 2016). Establishing wildlife habitats in urban areas provides shelter and forage for pollinators and other beneficial insects, which are vital for food production, highlighting UA's significance in maintaining ecological balance (Nicholls et al., 2020).

The multifunctional benefits of UA should inspire further promotion and support for these initiatives, estimated to be worth approximately USD 33 billion annually (Clinton et al., 2018).

2.2.3. Economic Sustainability

Globally, 68 million hectares of urban land are under cultivation, involving 266 million urban households in developing countries (Thebo, Drechsel, and Lambin 2014). With a lack of employment in the industrial and services sectors, agriculture remains essential for addressing urban poverty and improving residents' wealth in emerging nations. This is achieved by strengthening the local economy through the creation of new supply chains and job opportunities, such as farm laborers, food transporters, and sellers (Rao et al. 2022; De Bon, Parrot, and Moustier 2010).

Moreover, UA effectively boosts household incomes of economically disadvantaged families by reducing dietary expenses, which account for up to 60–85 % of their earnings (Orsini et al., 2013). Indeed, UA serves as a source of subsistence, with self-consumption production ranging from 10 % to 90 %, depending on the country and the product (De Bon, Parrot, and Moustier 2010).

Additionally, local commercial production reduces price differentials between the farm and the final consumer by reorganisation of the supply chain and by establishing direct sales with transparent producer/consumer relationships (Skar et al., 2020). This practice mitigates the impact of macroeconomic fluctuations that lead to spikes in food prices. However, the concurrent shocks and crises between 2020 and 2022 have set back decades of sustained poverty reduction, pushing the number of people subsisting on less than 2.15 US dollars per day to almost 700 million (World Bank, 2024). Given the current rate of progress, achieving the zero-poverty goal established by the UN Agenda 2030 is under severe threat. (United Nations 2024).

In the Global North, the economic sustainability of newly established UA projects remains to be determined (Orsini et al., 2020). Despite needing further confirmation, potential economic strategies have been identified, ranging from focusing on high-value production, targeting specific product niches, and participating in the Alternative Food Network (Skar et al., 2020). Nevertheless, it is crucial to consider that the overall economic balance must account for the significant impact of carbon credits on the system's lower carbon footprint, as well as the invaluable ecosystem services provided as an “economic added value”.

2.2.4. Challenges in Urban Agriculture

The benefits of UA are undeniable; however, the challenges that urban gardeners face are significant. The uncertainty of future land access is a pressing issue. The availability of land, compounded by competing demands on urban space and spikes in land prices, poses a problematic obstacle to expanding in-ground UA (Harada & Whitlow, 2020). The profitability of building construction for landowners further intensifies this competition for space, complicating the justification of urban land use solely for agricultural purposes. Consequently, urban farmers are often relegated to marginal vacant lots in older and vulnerable neighbourhoods. These areas are generally affected by pollutants generated by human activities, such as previous industrial use of these sites, atmospheric deposition from industrial processes, and incinerators; additionally, located in the periphery, orchards are near main roads and exposed to traffic-related pollutants emissions (Jean-Soro et al., 2015).

Among the significant contaminants in urban soils, potentially toxic elements (As, Hg, Cd, Pb, Fe, Zn, and Mn) can accumulate in edible biomass by root absorption and enter the food chain. Nonetheless, as demonstrated in a recent study by Augustsson et al. (2023), the risk of contaminant exposure also lies in the adhering particles to leafy vegetables, even after regular washing, significantly contributing to the plant concentrations of As (9–20 %) and Pb (25–29 %). Trace elements' dietary assumptions seriously threaten food safety and can result in significant human health hazards (Izquierdo et al. 2015). These aspects highlight the importance of previous site-specific investigation for planning UA activities, as well as prioritising the education of urban gardeners in sustainable cultivation techniques for ensuring a safe environment (Bretzel et al., 2016; Rossini-Oliva and Nuñez, 2024). Indeed, the lack of training in agricultural practices represents a further threat, given the widespread amateur nature of UA (Pennisi et al. 2016; Nicholls et al. 2020). The reckless use of synthetic products (e.g., fertilisers, herbicides, and pesticides) may lead to accumulation in the urban ecosystem and jeopardise UA's ecosystem services since it negatively affects flora and fauna biodiversity. In this context, López et al. (2019) found evidence of Cu accumulation in an urban orchard soil in Sevilla, Spain, due to the improper use of Cu-based fungicides for over 25 years. In addition, cases of over-application of mineral fertilisers have been repeatedly reported in urban gardens, provoking environmental damage and a risk to human health, given the ability to accumulate high nitrate concentrations in edible tissues (Pennisi et al. 2016; Buscaroli et al. 2021).

The perceived risk of urban food contamination is a significant mental barrier to individual adoption of UA, underscoring the importance of local food production and the need for practical solutions (Ziss et al. 2021). Antisari et al. (2015) found that nearby pollution sources increase the risk of heavy metal accumulation by about 1.5 times when vegetables are grown within 10 meters of the road compared to those grown at 60 meters. Future planning and design must consider setting UA initiatives at a safe distance from pollution sources.

Upscaling UA initiatives is a smart solution for these challenges since cultivating in height decreases the accumulation rates of pollutants (B. Grard et al. 2018). Reassessing unexploited roofs could overcome the obstacles to UA development and transform cities into sustainable food hubs (Appolloni et al. 2021). This innovative approach holds immense potential to boost local food production significantly and offer a promising solution for designing resilient cities.

Chapter 3

Bird’s-Eye View of Brooklyn Grange-Future (Elizabeth Downer Riker®)

Green Roof Agriculture

3. Rooftop Agriculture

The current chapter reviews various models of rooftop agriculture, emphasizing their potential benefits and limitations. Finally, it addresses the challenge of identifying a suitable substrate for RA as an alternative to peat, suggesting the possibility of using growing media derived from municipal solid waste compost.

3.1. An Innovative Solution

Urban development and population growth dynamics have sparked significant environmental concerns. The sustainable evolution of urban communities increasingly relies on effectively managing cities expansion, minimizing ecosystem degradation, and maximizing benefits for the population (Calheiros and Stefanakis 2021). The pathway to sustainable urban development is outlined in The New Urban Agenda of the United Nations (Acioly et al. 2020): "Protect, conserve, restore, and promote ecosystems, water, natural habitats, and biodiversity, minimize environmental impact, and shift towards sustainable consumption and production patterns".

The EU recently launched several projects within the European Green Deal (European Commission 2019) framework to rethink urban settlements by promoting interdisciplinary collaboration among public and private stakeholders and the scientific community. Within the multifaceted interventions, the Driving Urban Transitions Towards a Sustainable Future (DUT) (Bylund et al. 2022) partnership has pinpointed transitions in the energy and mobility sectors, applying circular economy principles, and emphasizing nature-based solutions as paramount elements for urban transformation. In parallel, the New European Bauhaus (European Commission 2021b) initiative advocates the transition towards environmentally friendly and inclusive urban spaces, towns, and local communities. Lastly, the UE aims to create a better urban environment by reducing and progressively stopping soil sealing, thus enhancing city soil health within the framework of the Soil Mission (European Commission 2021a). On the path towards cities of tomorrow, urban food security and equity still need to be solved.

In the challenge of redefining open urban space, Bohn and Viljoen (2011) introduced the concept of a Continuous Productive Urban Landscape, envisioning the planning of interlinked UA actions through existing or emerging cities spaces to create a new sustainable urban infrastructure. The concept of "edible cities" does not stand alone. It is consistent with the City Region Food Systems (CRFS) proposal (FAO, Rikolto, and RUAF 2022), which aims to foster a food network within urban and peri-urban areas with rural surroundings and aligns with the EU Green Deal's Farm-to-Fork strategy (European Commission 2020c), which promotes a fairer, healthier, and more environmentally respectful food systems.

However, implementing in-ground UA in densely built-up areas still faces excessive policy and planning limitations, primarily due to inadequate space and competitive real estate markets (Harada and Whitlow 2020). In this perspective, retrofitting unused urban sites could offer massive opportunities for overpassing these problems, catalysing a paradigm shift in scope.

Rooftops represent primarily "wasted" urban surfaces, constituting up to 32 % of the city's area (Oberndorfer et al. 2007). For this reason, residential or commercial rooftops would provide ideal conditions for farming purposes, offering a viable alternative to the lack of vacant land in dense urban areas. Besides, rooftops are potentially free of pests, enabling nature-friendly production, and addressing the increasing demand for organic food driven by growing public concerns about chemical-free crop cultivation.

Rooftop Agriculture (RA) has emerged as an innovative solution that combines food production, design, and architecture. The benefits are manifold. Indeed, RA not only ensures food supply but also mitigates the negative impacts of urban areas by contributing to various ecosystem services (Figure 3.1). It regulates the urban microclimate (Oberndorfer et al. 2007; QIU et al. 2013), increases biodiversity (Coulibaly et al. 2023; Joimel et al. 2022; Bretzel et al. 2017), facilitates biowaste assimilation (Grard et al. 2015), enhances societal relations, improving human health and well-being (Pauleit et al. 2019), reduces greenhouse gas emissions, and purifies the air (Mihalakakou et al. 2023), as well as regulating water flows (Oberndorfer et al. 2007).

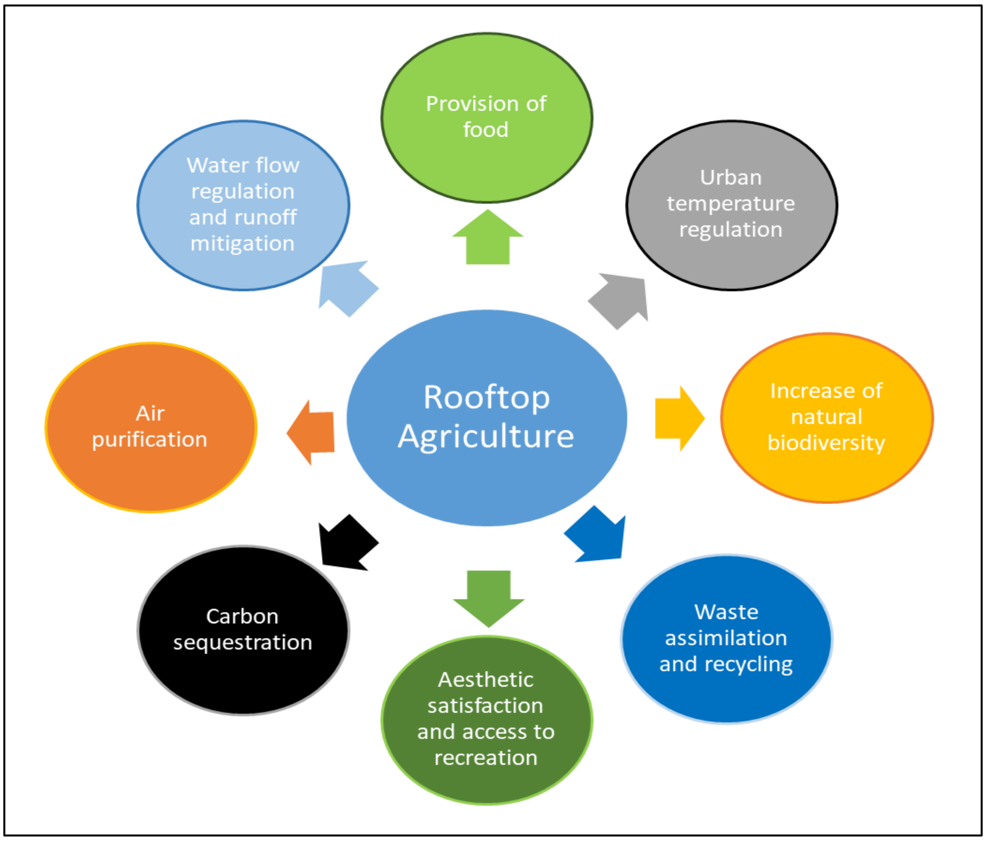


Figure 3.1: Ecosystem Services provided by Rooftop Agriculture

RA is a versatile concept, encompassing both protected (rooftop greenhouses) and non-protected (open-air rooftop systems) designs (Appolloni et al. 2021). It is defined as a form of building-integrated agriculture (Goldstein et al. 2016) or zero-acreage farming (Specht et al. 2014), demonstrating its adaptability to different urban contexts.

The RA project can be classified into four distinct categories, each with its unique focus and benefits:

- Commercial rooftop farms typically represent business-oriented initiatives focused on an economically viable farming business, usually operated by a start-up company;
- Social-educational and urban living quality RA projects are developed without profit aims, prioritizing the integration of minorities, youth education, and improving living conditions for urban dwellers by providing recreational and community spaces for personal food production;

- Image-oriented RA projects are frequently associated with hotels or restaurants, primarily utilizing rooftop cultivation for marketing and aesthetics;
- Innovative incubator RA projects focus on researching and developing new technology to enhance sustainable food production. They are primarily run-in research centres, universities, or start-ups. These projects contribute to the advancement of RA and have significant policy implications, highlighting the potential for scaling up this sustainable solution and integrating it into urban planning strategies.

Based on this, the choice of which RA design to adopt depends on the project's purpose, offering a range of approaches in terms of agricultural techniques, management practices, and growing media adopted, as described in the next sections.

3.2. Rooftop Greenhouses

Protected RA designs involve applying greenhouse techniques adapted for use atop buildings. These setups, primarily associated with commercial activities, employ soilless cultivation methods (i.e., hydroponic, aquaponic, and aeroponic), providing higher yield and optimizing farming inputs. Indeed, reported tomato harvests from rooftop greenhouses range from 16.2 to 19.6 kg m^{-2} year^{-1}, outperforming the results observed in RA open-air systems, which vary from 5.1 to 14.3 kg m^{-2} year^{-1} (Harada and Whitlow 2020; Nadal et al. 2017). The enhanced input efficiency is emphasized when the greenhouse is operated under closed-loop techniques, lowering water needs by 40 % and nutrient requirements by 35.54 % (Rufí-Salís et al. 2021).

The innovative aspect of this production model lies in its potential to establish symbiosis with the host building, achieved through direct exchanges of water, energy and CO_2, helping to decarbonize greenhouse-based production and promote efficient greenhouse heating. The greenhouse/building mutualism can potentially meet 80–90 % of the water demand by harvesting rainwater (Sanjuan-Delmás et al. 2018) or by reusing greywater, which represents approximately 95 % of the total volume of buildings' wastewater (D'ostuni et al. 2023). Additionally, by utilizing the exhaust air from the building for heating, the integrated method could facilitate the recycling of 341.93 kWh m^{2} year^{-1}, resulting in an annual CO_2 reduction of

113.8 kg CO_2 (eq) m^{-2} year^{-1} with a cost-saving of 19.63 € m^{-2} year^{-1} (Nadal et al. 2017).

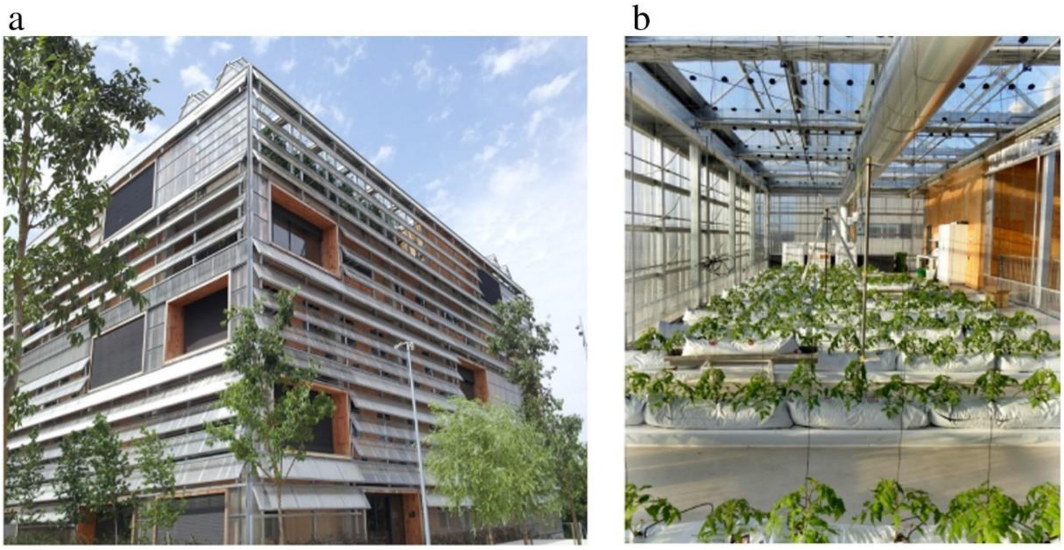


Figure 3.2: RA greenhouse project (b) on the Institut de Ciència i Tecnologia Ambientals (ICTA) of the Universitat Autonoma of Barcelona, Spain building (a) (Montero et al. 2017).

Despite the unquestioned capacity to deliver food production, it is important to note that the technology does encounter significant limitations. These include the substantial investment required in the early stages of development, the high maintenance and operational costs, and the need for highly specialized personnel (Specht et al. 2014). The necessary technology—structural components, mechanical and irrigation equipment (e.g., pumps, heaters, shading and lighting devices), sensors, and computers—entails an average installation cost of 880 € m^{-2}, a considerable economic barrier to the implementation of citizen's initiatives. Besides, operating expenses of 80 € m ^{2} year^{-1}, in front of an average net income of 26 € m^{-2} year^{-1} (Appolloni et al. 2021), lead to uncertainty regarding the economic viability of the investment. The economic disadvantage becomes even more apparent when compared to conventional greenhouses. Sanyé-Mengual et al. (2015) emphasized that rooftop greenhouse production costs are 2.8 times higher than conventional greenhouse, with an environmental impact of 17–75 % higher. Indeed, the environmental aspect is another major constraint, with the system showing limited capacity to provide ecosystem services compared to open-air designs. Even though recent studies have been conducted on wastewater recycling, the common usage of hydroponic techniques narrows the greenhouses' biowaste assimilation abilities (Goldstein et al. 2016). Furthermore, the compartmentalized productive environment makes their role in creating habitats for wildlife biodiversity low, if not negligible, and simultaneously limits the capacity for regulating water runoff (Goldstein et al. 2016; Harada and Whitlow 2020).

Finally, introducing greenhouses into the urban landscape may encounter architectural limitations due to their visual/aesthetic impact. Their implementation could be subject to planning consent, especially within historical centres, given the low harmony between the structure and the broader local built environment (Zambrano-Prado et al. 2021).

3.3. Open-Air Rooftop Systems

Open-air rooftop systems are the most prevalent form of RA, constituting 84 % of the global realities (Appolloni et al. 2021; Specht et al. 2014). By employing low-technology growing methods and often recycled materials, these designs represent a cost-effective solution for enhancing urban food access (Orsini, Dubbeling, and Gianquinto 2015). Open-air rooftop can be distinguished into "formal" and "informal" models (Caputo, Iglesias, and Rumble 2017).

Green roofs represent the formal model and are primarily designed for commercial or educational purposes. These farming systems incorporate multiple layers, including a root barrier to prevent damage to the underlying structure, a drainage layer to facilitate the removal of excess water, a filter fabric to prevent the drainage layer from clogging with media, the growing media, and the vegetation (Walters and Midden 2018). Typically occupying medium-to-large rooftops, they offer ample space for landscaping the entire roof surface, necessitating the know-how of professional figures (Caputo, Iglesias, and Rumble 2017). Green roofs vary in substrate depth, categorized as extensive (25–100 mm), semi-intensive (120–250 mm), or intensive (150–400 mm). Among these, intensive designs with a substrate depth of 200-400 mm are most recommended for rooftop farming due to their capacity to support diverse horticultural plants, shrubs, and small trees, albeit requiring frequent maintenance (Caputo, Iglesias, and Rumble 2017). However, the increased weight of more profound growing media may surpass the roof's load-bearing capacity, necessitating structural reinforcement and incurring higher financial costs (Walters and Midden 2018).

Container gardening is an 'informal' productive system, representing a means of community involvement rather than educational and social purposes. The design is based on planters positioned directly on the roof or some intermediate surface, such as wood pallets. It can be implemented by reusing wooden boxes, plastic or clay pots, and raised beds made of recycled timber allowing in-soil and soilless production models to be adopted (Orsini et al. 2014; Grard et al. 2018). This design requires few or no modifications to the existing roof structure, providing an adaptable solution for RA development, especially when financial availability is limited (Caputo, Iglesias, and Rumble 2017).

Figure 3.3: The Brooklyn Navy Yard in New York, USA (Brooklyn Grange®) (a). The Community Garden of Via Gandusio in Bologna, Italy (Francesco Orsini®) (b). The T4P project in Paris, France (Accueil - Lab Recherche Environnement).

Overall, open-air installation costs are estimated to range between 30 and 100 € m^2, being significantly lower than those of high-tech greenhouses (Harada and Whitlow 2020; Appolloni et al. 2021). The financial feasibility holds significant promise for advancing the development of UA actions.

Indeed, although presenting lower yields than high-tech systems, open-air roof farms proved to be potentially crucial in providing food security. In a study conducted in Bologna, Italy, the authors showed that, if extended to the city's 82 hectares of flat rooftops, open-air design could provide more than 12,000 tons of vegetables per year, satisfying 77 % of the inhabitants' requirements (Orsini et al. 2014). Moreover, upscaling open-air RA has the potential to form a dense web of green corridors spanning over 94 km with a density of up to 0.67 km^2 (Orsini et al. 2014). The newly formed green spaces restore habitat fragmentation triggered by urban expansion. The requalification contributes to mending the ecological network between rural and urban landscapes and increasing biodiversity in anthropized areas. The promoted ecological connections are supported by designing microhabitats for shelter and breeding using various structural elements, substrate types, and depths on open-air RA (Bretzel et al. 2017). In particular, the proper choice of growing media influences the formation of biodiversity hotspots, harbouring significant abundance and diversity of microorganisms and micro and mesofauna (Joimel et al. 2022). The system's achieved biological complexity would allow for counteracting the population of phytophagous organisms that naturally proliferate in an agricultural environment (e.g., aphids, spired mite, lepidopteran larvae) through the appearance of their predators, meeting agricultural production in balance with the environment as advocated by the EU (Bretzel et al. 2017; Bazzocchi and Maini 2017). Furthermore, growing local horticultural varieties in open-air RA represents a powerful tool for conserving the germplasm of these landraces severely threatened by genetic extinction, preserving local rural traditions that risk being permanently lost. Their introduction in the horticultural rotation is viewed favourably since they present genetic adaptation to the local environment, and their desynchronized ripening character allows the extension of the productivity calendar for the local growers (Bretzel et al. 2017). From a broader perspective, the valuable ecological contribution of open-air RA embraces the EU's efforts to reverse biodiversity loss and ecosystem degradation, as traced by the Ecosystem Restoration Law (European Commission 2022).

Among the other benefits, as observed for high-tech systems, open-air RA facilitates establishing a metabolic relationship between the rooftop and the hosting building, enabling recycling waste the tenants generate and collect. Specifically, organic waste can be composted on-site through a small-scale system, serving as substrate and a source of nutrients for horticultural production. Scaling down composting process would empower citizens to engage in these impactful practices while reducing CO_2 emissions associated with biomass displacement (Caputo, Iglesias, and Rumble 2017). Furthermore, since this carbon-rich amendment is widely utilized in RA, it contributes to the long-term storage of carbon in growing media (Walters and Midden 2018; Shafique, Xue, and Luo 2020). Together with vegetation capturing CO_2 from the atmosphere through photosynthesis, open-air RA can be transformed into a carbon sink system, potentially storing about 624 tons of CO_2 year in a city of the size of Bologna, Italy (Orsini et al. 2014; Shafique, Xue, and Luo 2020). It is crucial to emphasize that the compost used for green roofs must adhere to national quality standards, specifically showing trace metal levels below the allowable limits. Failure to meet these standards can pose a significant chemical hazard, as contaminants may enter into the food chain or disperse in the environment through water runoff following meteoric events. In this regard, greening grey rooftops has been proven to mitigate stormwater runoff through the joint action of plant absorption, evapotranspiration, and growing media retention, intercepting up to 74–84 % of the incoming rainfall, thus reducing the peak flow (Grard et al. 2018). However, particularly intense weather events can cause substrate waterlogging, resulting in excessive weight loading on the building, representing the principal constraint of open-air RA. This aspect is greatly influenced by building age, the scope of use (commercial or residential), and the country's regulatory framework (Caputo, Iglesias, and Rumble 2017). Most buildings are designed to support only the roof structure and a minimum live load to accommodate snow and occasional maintenance, with a typical bearing capacity between 120 and 150 kg/m^2 (Castleton et al. 2010). However, to date, more than half of the existing projects employ soil as the primary medium for plant cultivation; considering that one cubic meter of soil weighs up to 1.600 kg on the building's structure(Gorgolewski and Straka 2017), using this media can impose an unsustainable additional load on the building's structure, severely restricting the feasibility of this green technology. Furthermore, transporting rural soils to cities is an unsustainable practice, both from an environmental and economic point of view.

Developing soilless RA productive systems represents a promising solution to overcome these problems. Soilless culture is defined as any method of growing plants without using soil as a rooting medium (Barrett et al. 2016). This method encompasses various productive systems, generally using a porous rooting medium known as a ‘substrate’ or ‘growing medium’ (Barrett et al. 2016). Peat is the most commonly used medium because of its relative homogeneity and excellent physical and chemical and biological properties (Raviv 2014). However, the use of this non-renewable resource is questionable in the current climate change scenario, as illustrated in the following section. The scientific community should aspire to develop an eco-friendly substrate with adequate fertility and physical integrity that ensures optimal conditions for plant growth.

3.4. Substrates

Since the 1950s, peat has been the predominant component of growing medium in the horticultural sector (Gruda, Hirschler, and Stuart 2024) because of its excellent combination of physical and chemical properties such as low pH, high interchange cationic ability and optimal water-air storage (Carlile, Raviv, and Prasad 2019). Furthermore, peat initially provides a root environment free from pests and diseases.

Presently, approximately 90 million m^3 of peat is extracted annually, 40 million m^3 of which is used in horticulture (Leiber-Sauheitl, Bohne, and Böttcher 2021). However, the significant demand for peat-based growing substrates is leading to the degradation of peatlands. These ecosystems, formed by the accumulation of vegetative litter under permanent water saturation and anoxic conditions, play crucial roles in biodiversity conservation, regulation of local water quality, and flood protection (Kimmel and Mander 2010; Leifeld and Menichetti 2018). Despite covering 3 % of the terrestrial surface, these fragile ecosystems are long-term carbon sinks, storing between 21 % and 33 % of global organic soil carbon stock (Gruda 2019). However, human drainage and extraction practices have transformed peatlands into net greenhouse gas emitters, producing 1.2–1.9 Gt CO_2 $year^{-1}$ (Freeman et al. 2022), making peat the most ecologically impactful component of substrate mixtures (Gruda, Hirschler, and Stuart 2024).

Increasing environmental awareness has prompted the investigation of total or partial peat replacement in the last decades by recycling disposed organic fractions (i.e., urban waste, agricultural and industrial by-products) (Kern et al. 2017).

Compost presents a sustainable solution to valorise underutilised high-nutrient waste, providing a sound alternative to peat (Raviv 2014; Farrell and Jones 2010). Indeed, compost made from waste organic fraction proved to contain phytohormones that influence plant growth and development (i.e., auxins, cytokinins, gibberellins, brassinosteroids, abscisic acid, and salicylic acid) (Sienkiewicz et al. 2024). Another established advantage of compost is its ability to slow-release plant macronutrients and micronutrients in the rhizosphere over time, reducing the dependency on mineral fertiliser and advocating for a more sustainable horticultural sector (Gruda 2019; Atzori et al. 2021).

Composting involves the biodegradation of organic matter by various microbial communities in a four-stage process: an initial mesophilic stage (25–40°C), a thermophilic phase (35–65°C), a cooling stage (or second mesophilic stage), and a maturation phase. Decomposition mainly occurs during the thermophilic phase, during which the material is sterilised, maintaining an ideal temperature of 55°C for at least two weeks (Diaz, De Bertoldi, and Bidlingmaier 2007).

However, the high heterogeneity of the biomass due to the inefficient source separation may complicate the process standardisation, increasing its cost and potentially reducing the final product quality. Failure to complete the biodegradation process results in immature compost, whose application may result in undesirable effects on plant growth. Indeed, immature compost continues to decompose while in the planter, leading to oxygen depletion and often nitrogen immobilisation, potentially releasing phytotoxic compounds (i.e., short-chain aliphatic acids, ammonium ions, and polyphenols) (Raviv 2015); besides, the lack of sterilisation can potentially introduce pathogens, weed seeds, and insect larvae (Insam and de Bertoldi 2007). Another essential precaution to consider is the presence of heavy metals (Raviv 2015). Inadequate management of biowastes can lead to the accumulation of trace elements, posing risks to both human health and the environment. Strict quality control procedures are essential in preparing composts for use in growing media (Barrett et al. 2016). Efforts should involve tracking homogeneous raw materials to standardise the process.

Municipal organic waste represents a large flow of unexploited nutrients that can be recycled in the agricultural sector as compost. Globally, cities generate 884 million tonnes year^{-1} of food and green waste (approximately 115 kg per capita), representing the largest waste category (Kaza et al. 2018). The mismanagement of biowastes is already an environmental problem. Primarily disposed of through landfilling, they occupy large areas while generating concerning amounts of greenhouse gases, such as methane (CH_4) and nitrous oxide (N_2O) (Raviv 2015; Arcas-Pilz et al. 2023). Additionally, they may cause nutrients to leach into the environment, accumulating in soils or dispersing into waterways, triggering eutrophication (Arcas-Pilz et al. 2023). The EU aims to protect human health and the environment by limiting the share of municipal waste landfilled to 10 % by 2035 (European Commission 2018). Achieving this goal will require significant efforts, especially considering that in 2018, only 30 % of the European organic fraction was source-separated and recycled further (Möller et al. 2018).

Transitioning to a circular economy model, alternative processing of this organic waste fraction can help achieve the "recycling, recovering, and reusing" paradigm and overcome the linear "take-make-dispose" approach (Ghisellini, Cialani, and Ulgiati 2016), as promoted by the EU with the Green Deal objectives and the new Circular Economy Action Plan (European Commission 2020b).

Recently, among municipal organic wastes, the potential of composting coffee by-products to replace peat and synthetic fertilizers for potted plants has been highlighted (Ronga et al. 2016; Picca et al. 2023). As well as their application in an open-air RA produced interesting results (Grard et al. 2020; Grard et al. 2018).

Coffee is the most consumed beverage worldwide, and in 2022, 10.8 million tons of green coffee beans were produced (Faostat). Coffee production and consumption generate vast residuals, such as immature and defective beans, husks, pulp, silverskin (CS), and spent coffee grounds (CG). Among these by-products, CS and CG account for 50 % of the total waste generated by the coffee industry, mainly produced in consuming countries during the roasting process and final coffee consumption (Janissen and Huynh 2018).

Specifically, agricultural interest in these biowastes is aroused due to their high macro- and micro-nutrient content. For instance, dry CS presents values of 25.47 g kg^{-1} of nitrogen (N), 21.1 g kg^{-1} of potassium (K), 9.4 g kg^{-1} of calcium (Ca), 3.1 g kg^{-1} of magnesium (Mg), and 2.8 g kg^{-1} of sulfur (S) (Janissen and Huynh 2018; Ballesteros, Teixeira, and Mussatto 2014). On the other hand, CG shows values of 27.9 g kg^{-1} of N, 11.7 g kg^{-1} of K, 1.2 g kg^{-1} of Ca, 1.9 g kg^{-}

^{1} of Mg, and 1.6 g kg^{-1} of S (Janissen and Huynh 2018; Ballesteros, Teixeira, and Mussatto 2014). Moreover, both materials present a high-water holding capacity (WHC) of 748 % and 298 % (w:w), respectively. Despite their richness in macro and micronutrients and remarkable water retention capacity, phytotoxic compounds (i.e., caffeine, caffeine, tannins) have hindered the widespread use of these by-products in the agricultural sector. However, composting these biomasses has proved to be a straightforward and cost-effective opportunity to unlock their agronomic potential. Indeed, the biodegradation process offers a viable solution to mitigate their phytotoxicity while preserving the beneficial properties (Picca et al. 2022; 2023; Ronga et al. 2016).

Seaweed is another biomass that has attracted interest in reducing the use of peat, considering its physical and chemical characteristics (Gruda 2019). In recent years, there has been an unprecedented rise in seaweeds and algae bloom events, both autochthonous and invasive, registered in the Mediterranean Sea (Madejón et al. 2021). The excessive accumulation of seaweed biomass on the shorelines seriously affects the aquatic ecosystem and the local economies (e.g., tourism, commercial fisheries, waterfront property value, and waste management). As evidenced in several studies, composting seaweed is a sound and sustainable option for managing this biomass (Madejón et al. 2021) and using it as a peat substitute in tomato production (Castaldi and Melis 2004), lettuce transplants (Mininni et al. 2012), and melon and tomato seedlings (Berti et al. 2023; Mininni et al. 2013).

These biomasses can create optimal growing conditions in open-air RA substrates by optimising WHC, providing nutrients and structure, and minimising bulk weight without exceeding weight limit restrictions. Indeed, organoponics, a productive technique commonly used in Venezuela and Cuba, has been proposed as a sound agricultural model for RA. This eco-friendly agronomic technique involves crop growth in containers filled with compost or organic matter from various sources (Rodríguez-Delfín et al. 2017). However, this model may encounter potential disadvantages. Grard et al. (2018) tested organic substrates and reported mineral nitrogen leaching up to 345.3 kg ha^{-1} over two years due to compost biodegradation. The same study observed a steady carbon loss over time, mostly as dissolved organic carbon, leading to significant substrate shrinkage. In addition to possible environmental implications, it should be considered that higher organic matter levels, while improving water retention, could cause waterlogging and excessive weight risks (Caputo, Iglesias, and Rumble 2017).

Mixing compost-based substrates with light-weight and physically stable material presents an excellent way to improve the mixture's physical and chemical properties, obtaining a more stable and light-growing media.

Biochar (BC) is a lightweight, carbon-rich material produced by biomass and organic waste pyrolysis. During pyrolysis, the feedstock undergoes thermal decomposition in the absence of oxygen, losing most of its labile organic matter. The process produces a chemically stable and porous material with a high degree of condensed aromatic rings and cation exchange capacity (Goñi-Urtiaga et al. 2022).

Biochar has critical implications for developing a circular economy in agriculture. Indeed, pyrolysis can provide a sustainable strategy for managing pruning wastes, allowing their conversion into BC. Subsequently, the carbonaceous material can be integrated into fertilization strategies, mainly associated with organic amendments (Jindo et al. 2020). Due to its potential, BC has been widely studied as a peat substitute (Nocentini et al. 2021; Gascó et al. 2016; Kern et al. 2017) and recently suggested as an innovative substrate component for RA installations due to its high water-holding capacity and low bulk density (Lee and Kwon 2024). Depending on the application rate in RA, BC can improve media porosity by 5.3–9.3 %, lighten the substrate by lowering bulk density by up to 38 %, and improve cation exchange capacity by 38.1–75.9 % (Cao et al. 2014; Chen et al. 2021). BC has the capacity to reduce total substrate erosion by 39 % (Liao, Drake, and Thomas 2022). Moreover, it could increase plant-available water by 16 % while raising WHC by up to 74 %, extending crop options in dry climates and mitigating flooding effects in stormwater events (Cao et al. 2014). Lastly, BC can improve runoff water quality by reducing phosphorus, organic carbon, and organic nitrogen export in RA leachates (Lee and Kwon 2024).

While BC has demonstrated its effectiveness as an amendment in RA substrates, its use as a co-composted BC substrate has yet to be proven. Indeed, using BC as a compost bulking agent provides several beneficial effects by altering microbial activity and community composition, enhancing nitrogen retention, immobilising potentially toxic metals and organic pollutants, and reducing greenhouse gas emissions emitted during the process (Sanchez-Monedero et al. 2018; Godlewska et al. 2017). Moreover, the highest temperature reached during the process enhances compost disinfection, reducing the biological risk for agricultural use (Godlewska et al. 2017). Concurrently, composting induces changes in the surface composition of biochar, increasing the degree of oxygen functionalities (Goñi-Urtiaga et al. 2022) and its cation exchange capacity (CEC), thereby enhancing dissolved ion adsorption and, consequently, its plant-growth-promoting properties (Kammann et al. 2015).

To our knowledge, no studies have tested the potential of BC co-composted in open-air RA, and this thesis aims to find scientific answers to this question.

3.5. Objectives

The main objective of the present thesis was to assess the potential of co-composting wastes, with and without biochar, as potential substrate options in open-air RA aiming to lower peat concentration. To address this goal, the following scientific questions arose:

- ***Question 1*** - Are coffee-derived by-products a suitable feedstock for producing high-quality composts?
- ***Question 2*** - At which proportions do composts based-substrates ensure the best conditions for plant growth?
- ***Question 3*** - Are composts derived from organic waste suitable for rooftop agriculture, and what is their agronomic potential for plant production?

To answer these questions, the following comprehensive objectives were studied in detail:

Objective 1: Evaluate the quality parameters of coffee silverskin co-composted at different proportions with carbonaceous material (Chapter 5).

Objective 2: Assess the agronomical performance of two spent coffee ground-based composts with and without biochar in the different stages of tomato (*Solanum lycopersicum* L.) development (Chapter 6).

Objective 3: Study the total yield and food quality produced from six growing media made from organic waste-based composts with and without biochar used on a productive rooftop (Chapter 7).

Chapter 4

The making of compost (Cristina Gomez Ruano®)

Materials and Methods

4. Materials and Methods

The chapter outlines the materials and experimental methods used in this doctoral thesis. It describes the characteristics of the materials, their processing and treatment protocols, as well as the fundamentals of characterization techniques and equipment details.

4.1. Raw Materials

The choice of materials was driven by their availability, nutrient content, and the goal of exploring novel reuses for substances that may present environmental challenges.

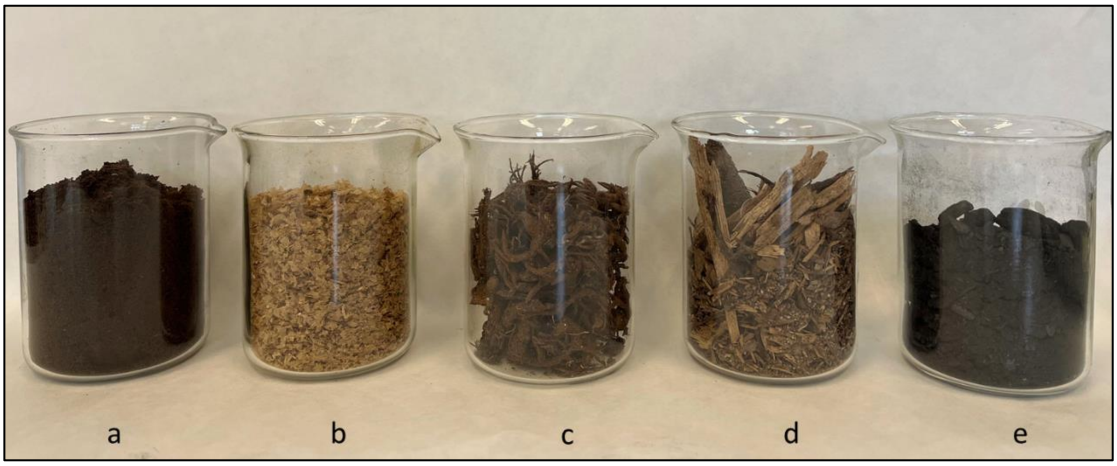


Figure 4.1: Raw Materials used in the study: (a) Spent coffee grounds, (b) Coffee Silverskin, (c) Seaweed, (d) Pruning waste, (e) Biochar.

The biowaste used for composting are as follows:

- *Spent coffee grounds* (CG) were collected from the coffee vending machines (CONSUM Vending SL) installed at the Institute of Agricultural Sciences, ICA-CSIC (Madrid, Spain) and from a local bakery (Café 31 Bakery & Food Madrid);
- Coffee Silverskin (CS) was provided by Café Candelas SA (Lugo, Spain). This by-product originated from a coffee roasting process with hot air ventilation and a temperature ramp up to 210 °C for 12 min to separate CS from coffee beans. Due to the high roasting temperature

before the pressing, CS has undergone a "sterilization-like" process, avoiding mould proliferation during storage;

- Seaweed (SW) were collected from the Atlantic beach of "Playa de la Costilla" in Rota (Cádiz, Spain) using an excavator shovel. The biomass collected was composed of native macroalgae of the western coast of Cádiz and *Rugulopteryx okamurae*, an invasive species originally from the western Pacific. This material was provided through the project "Bioresourcing sea-algal waste" led by Dr. Engracia Madejón at IRNAS-CSIC and funded by TAUW foundation (Germany).

Given the low bulk density and nutrient richness of the materials, two structuring agents were chosen to optimize the composting process. These agents were carefully selected to reach optimum C/N values (~25 for composts without biochar, ~35 for composts with biochar, since most of the biochar-derived C is considered inert), maintaining a neutral or basic pH, and exhibiting low electrical conductivity. Moreover, the material had to improve aeration within the mixture, mitigating the heap compaction risk and the consequent emergence of anaerobic conditions. The materials selected were:

- Green waste (GW) was provided by the company "Ana Rosa Gómez - podas" based in Brunete (Madrid). It consisted of a mix of tree trunks, branches and leaves of different species (mainly poplars and oaks);
- Biochar (BC) was purchased by the Carbón Vivo SL (Barcelona, Spain), produced from Aleppo pine wood trunks and branches, and pyrolyzed in a Kon Tiki Kiln oven at 650–750 °C for 3.5 h;

4.1.1. Physical, Chemical and Phytotoxicity Analyses of Raw Materials.

Samples were air-dried, and pH and Electrical Conductivity (EC) were determined. At the same time, samples were finely ground using a ball mill (MM400, Retsch Technology, Haan, Germany) for 10 minutes at a frequency of 25 s^{-1} to assess Total Carbon (TC) and Nitrogen (N). Analyses were conducted in triplicate.

The main physical and chemical characteristics of the raw materials used in the study are reported in Table 4.1.

Table 4.1: Main physical and chemical characteristics of the raw materials (mean ± standard deviation, n = 3)

Elements	**Spent Cofee Grounds**	**Coffee Silverskin**	**Seaweed** [a]	**Green Waste**	**Biochar**
pH	6 ± 0.1	5.0 ± 0.0	8.0 ± 0.2	7.9 ± 0.1	9.8 ± 0.1
EC (dS m^{-1})	1.0 ± 0.0	0.6 ± 0.0	1.8 ± 0.1	0.3 ± 0.1	0.6 ± 0.3
C (%)	54 ± 1.6	50 ± 1.2	16 ± 2.2	49 ± 0.5	63 ± 5.0
N (%)	2.6 ± 0.1	2.6 ± 0.4	0.7 ± 0.1	0.5 ± 0.0	1.2 ± 0.0
C/N	20 ± 1.1	20 ± 0.3	23	117 ± 6	39 ± 0.5
Bulk density (g cm^{-3})	0.7 ± 0.0	0.5 ± 0.0		0.1 ± 0.0	0.2 ± 0.0
WHC (% of DW)	298 ± 3	749 ± 83		186 ± 50	194 ± 1

[a] Data collected from Madejón et al. (2021)

pH and Electrical Conductivity (*EC*)

pH and EC were determined in a water-compost suspension and extract, respectively, at a 1:10 ratio (w/v) by weighing 2.5 g of the sample in a centrifuged tube and adding 25 ml of distilled water. Tubes were rotated for 1 h. pH was assessed using a micropH 2001 (Crison Instruments SA, Barcelona, Spain). Samples were then centrifuged for 10 minutes at 3000 rpm and filtered by Whatman® Grade 2 (Cytiva, Marlborough, Ma, USA) filtering paper with a pore size of 8 µm. EC was measured with a microCM 2201 (Crison Instruments SA, Barcelona, Spain).

Total Carbon (TC) and Nitrogen (N)

TC and N contents were determined by dry combustion and gas chromatography using a ThermoFlash 2000 NC Soil Analyzer (Thermo Fisher Scientific, Waltham, MA, USA). For this analysis, 15 mg of the sample was weighed within a tin capsule and a calibration curve of 5, 10, and 30 mg alfalfa standard (LECO Corporation, St. Joseph, MI, USA) with C and N content of 45.0 % and 3.66 %, respectively.

Bulk Density (BD)

Bulk density (BD) was determined by using a 1 L graduated cylinder filled with a known mass of the sample. The cylinder was manually tapped for 60 seconds to eliminate large void spaces within the sample. After tapping, the final volume occupied by the sample mass was measured as follow:

$$Bulkdensity = \frac{W}{V}$$

Where:

- W is the weight of sample;
- V is the volume occupied by the sample after 60' of tapping.

Water-Holding Capacity (WHC)

The water-holding capacity (WHC) of raw materials and final composts was determined by placing them in disposable plastic cups and oven-drying at 40 °C until (approximately one week) to calculate the dry weight. Samples were then saturated with water for 8 hours. Subsequently, the plastic cups were covered with parafilm at the top to avoid evaporation, and perforated at the bottom to allow water excess to leach for 24 hours. The material was then reweighed to measure the difference between the dry and wet material, determining the amount of water the material could retain, as follow:

$$WHC\ \% = \left(\frac{Ws - Wd}{Ws}\right) x100$$

Where:

- Ws is the weight of the sample after saturation and drainage;
- Wd is the weight of the dried sample.

4.2. Compost facilities

Compost production involving CG and CS occurred in the compost facilities established at the Institute of Agricultural Sciences - CSIC (Madrid, Spain). This facility encompassed four 200 L expanded polypropylene composters equipped with a passive aeration system (HOTBIN composting, Northampton, UK. Composts based on GW were prepared in windrow piles at the "Compost Ecológico" facility in Rota (Cádiz, Spain).

4.3. Coffee silverskin composts

To optimize the utilization of coffee silverskin (CS) and achieve ideal composting conditions for maximizing nutrient content in the final compost, four co-composting blends of the raw materials were meticulously prepared.

Table 4.2: Rates of raw materials used for the four composting mixtures.

Composting mixture	**Silverskin (v)**	**Biochar (v)**	**Green Waste (v)**
CS-1	1 (+1) [a]	0	1
CS-2	2	0	1
CS-3	3	0	1
BC-CS	1	1	1

[a] Mixture 1 started with a ratio 1:1 (v:v) of CS and GW; a second dose of CS was added halfway through the process (15 days) to a CS:GW ratio of 2:1 (v:v).

The compost pile was prepared with GW moistened to an optimal initial value of approximately 60 %. Moisture levels were maintained between 40 % and 60 % throughout the process by periodic watering. Mixtures were allowed to stand for three days, then manually stirred every three days to ensure adequate aeration and homogenization. Internal pile temperature was continuously monitored at a depth of 30 cm using a Decagon RT-1 probe connected to a Decagon Em50 datalogger (ICT International, Armidale, Australia).

Composite samples representing each pile were collected at the start and at 3, 9, 20, and 30-day intervals, stored at 4 °C for subsequent analysis.

After 30 days, compost mixtures were removed and left to mature in open air, enhancing compost quality for agronomic purposes.

4.3.1. Physical, Chemical and Phytotoxicity Analyses of Composts

Main physical and chemical analyses were conducted as described in Section 4.1.1. Further analyses were conducted as follows. Measurements were carried out in triplicate, except for macronutrients, micronutrients, and trace elements, for which a composite sample per treatment was prepared.

Moisture Content

The moisture content was assessed using a moisture balance equipped with infrared lamps (MA 110.R, Radwag, Radom, Poland) and calculated according to the formula:

$$Moisture\ \% = \left(\frac{Wm - Wd}{Wd}\right) x100$$

Where:

- Wm is the weight of the moisten sample;
- Wd is the weight of the dried sample.

Ammonium (NH_4^+) and Nitrate (NO_3^-)

NH_4^+ was extracted in KCl 1 M at a ratio of 1:10 (w/v). 2.5 g of sample was mixed with 25 ml of KCl, rotated per 1 h, centrifuged for 10 minutes at 3000 rpm and filtered by Whatman® Grade 2 filtering paper. NH_4^+ was quantified using a visible spectrophotometer HACH Lange DR2800 (HACH, Loveland, CO, USA) with pre-dosed testing cuvettes HACH LCK 303 and LCK 304, depending on the concentration range. NO_3^- was extracted in water at a ratio of 1:10 (w/v) following the same procedure and measured using HACH LCK 339 pre-dosed testing cuvettes with the same spectrophotometer.

Macro, Micronutrients and Trace Elements content

The contents of nutrients and trace elements were determined on a composite sample after digestion in nitric (HNO_3) and perchloric acid ($HClO_4$) in a sand bath at 200 °C, followed by analysis using an induction coupled plasma atomic emission spectrophotometer (ICP-AES, Perkin Elmer Optima 4300 DV, PerkinElmer Inc., Waltham, MA, USA). An interlaboratory sample MARS 2021-3.1 (Sample ID 281) Cow manure was used as a reference sample. The percentage recoveries of the standard sample ranged between 91.5% - 117.6%.

Phytotoxicity

The phytotoxicity of raw materials and matured composts was assessed according to Zucconi et al. (1981) with minor modifications. A mixture of 2.5 g dw of each sample in 25 mL of water was prepared, shaken for 2 h, centrifuged at 3000 rpm, and filtered by Whatman® Grade 2 filtering paper. Ten garden cress seeds (*Lepidium sativum* L.) were placed on filtering paper in a Petri dish, each containing 5 ml of water extract of the corresponding material. Control consisted of deionized water. Analysis was conducted in triplicate. After incubating the Petri dishes for 72 hours at 25 °C in darkness, the seed germination rate (G) and root length (L), considering roots longer than 2 mm, were measured. The germination index (GI) was then calculated using the following formula:

$$GI\ (\%) = \left(\frac{Gsample}{Gcontrol} x \frac{Lsample}{Lcontrol}\right) x100$$

Where:

- *Gsample* is the germination rate of seeds exposed to the aqueous extract of the sample.
- *Gcontrol* is the germination rate of seeds exposed to deionized water (control).
- *Lsample* is the root length of seeds exposed to the aqueous extract of the sample.
- *Lcontrol* is the root length of seeds exposed to deionized water (control).

The samples' degree of phytotoxicity was classified according to the GI values obtained, as described in Table 4.3.

Table 4.3: Phytotoxicity classification according to Zucconi et al. (1981).

GI (%)	Classification
<50	High Phytotoxic
50-80	Moderately Phytotoxic
80-100	Non-Phytotoxic
>100	Phytostimulant

4.4. Co-composted Growing Medias

4.4.1. Peat Replacement Using Compost: Preparation of the Mixtures

Two coffee-based composts were prepared to investigate the potential of co-composted-based substrates as peat replacements for horticultural production systems.

The mixture with biochar (BC-CG) consisted of one volumetric part of CG, one of BC, and one of GW (1:1:1 v/v/v), while the mixture without biochar (CG) was a blend of one volumetric part of CG and one of GW (1:1 v/v).

BC-CG and CG were mixed in varying volumetric proportions, ranging from 100 % (pure compost) to 50 %, 30 %, and 15 % of the total volume, with a commercial horticultural substrate (HS) (Jiffy GO PP7, Jiffygroup, Zwijndrecht, The Netherlands). HS was used as the control. This resulted in eight treatments, each with varying compost-to-substrate ratios, in addition to the control treatment, as described in Table 4.4.

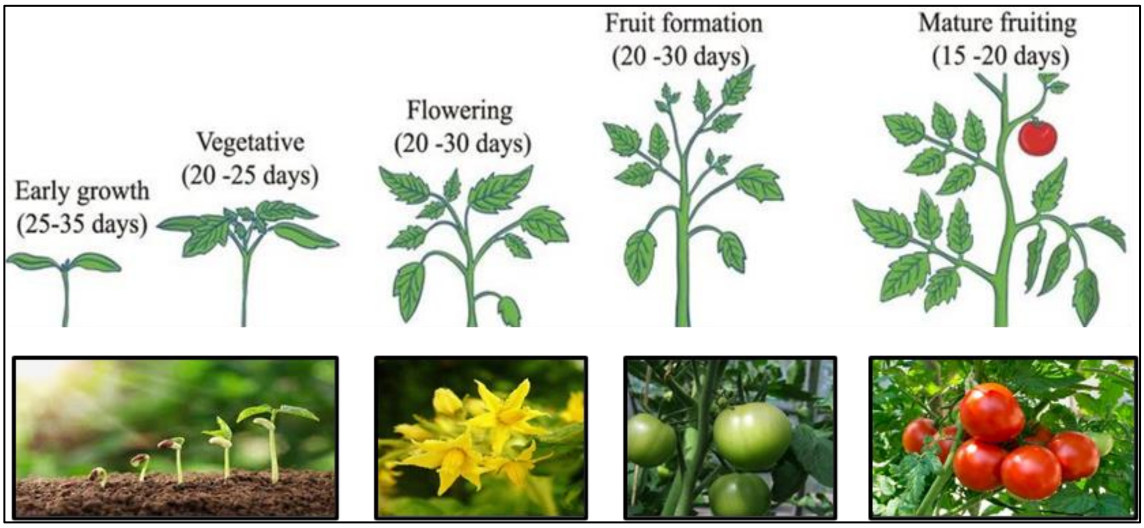
Table 4.4: Rates of substrates and number of replicates (n) used for the plantlet development and fruiting experiments.

Substrate	Peat (% v:v)	Compost (% v:v)	BC-Compost (% v:v)	n for seeds germination	n for plantlet development	n for fruiting plants
Control	100	0	0	5	5	5
BC-CG 100	0	0	100	5	5	6
BC-CG 50	50	0	50	5	5	6
BC-CG 30	70	0	30	5	5	6
BC-CG 15	85	0	15	5	5	5
CG 100	0	100	0	5	5	6
CG 50	50	50	0	5	5	6
CG 30	70	30	0	5	5	6
CG 15	85	15	0	5	5	5

4.4.2. Growing Tests

The agronomical performance of compost-based growing media was evaluated across three distinct stages of tomato (*Solanum lycopersicum* L., cv. Marmande) plant development: seed germination (0-6 days), seedling development (7-49 days), and plant-to-fruit maturity (36-100 days) (Figure 4.2).

Figure 4.2: Scheme of the tomato phenological stages studied in the growing media trials (Shamshiri et al., 2018).



Seed germination and germination index (GI %) were estimated by modifying the Zucconi's test indicated in section 4.1.3.; five tomato seeds placed on filtering paper in a Petri dish, with five replicate dishes for each growing substrate.

In the seedling development phase (Figure 4.3), tomato seeds were placed on filter paper atop a vermiculite bed within Petri dishes (25 seeds per dish). Subsequently, the dishes were placed in a growth chamber set at 20–22 °C with a 16:8 hours light-dark cycle for six days. Following germination, seedlings were transplanted into 7x7 cm pots filled with either BC-CG or CG composts at the various dilutions established (n=5) (Table 4.4) and cultivated for 35 days in a greenhouse maintained at 20–23 °C during the day and night, respectively, with a 12:12 hour light-dark cycle. Irrigation was performed manually and every three days by adding water to the pots' trays.

Figure 4.3: Seedling developing after 49 days.

For the plant production experiment, tomato seeds were germinated and then transplanted into 7x7 cm pots using the HS exclusively. After 36 days, the plantlets were transferred into 4.5 L pots filled with the respective growing substrates (n=5 or 6) (Table 4.4) and cultivated until full fruit ripeness in the greenhouse under the same environmental conditions as the seedling development trial (Figure 4.4).

Figure 4.4: Tomatoes reaching full maturity stage.

Plant height was recorded weekly throughout the experiments. Upon completion, an aliquot of aerial biomass and the total root biomass were collected to determine the fresh above- and below-ground biomass. Due to the occurrence of *Alternaria solani* (Sorauer) fungal pathogen, portions of the aerial biomass were trimmed, and the total aboveground biomass was not estimated.

4.4.3. Physical and Chemical Analysis of Growing Media and Plants

Replicates that did not complete the developmental cycle were excluded from the analysis. Substrate, biomass, and fruit samples were dried at 65 °C for three days and were finely ground in a Mixer Mill MM400 (Retsch GmbH, Haan, Germany).

Substrates were analysed at the beginning (t0) and end (t_{final}) of each experiment. At t0, each growing medium was analysed in triplicate. At t_{final}, an aliquot of 10 g was collected from each pot replicate.

The replicates from the seedling development experiment were pooled due to the scarce quantity of aboveground and belowground biomass produced, while the plant biomass samples were characterized separately.

Tomato average fresh fruit weight per treatment was calculated by dividing the total fresh weight by the total number of fruits harvested.

The substrate and biomass samples' physical and chemical characteristics were assessed in line with sections 4.1.1 and 4.1.3 of this chapter.

For the characterization of the macro, micronutrients, and trace elements concentration in plant biomasses by ICP-AES, an interlaboratory sample IPE 2022-1.2 (Sample ID 100) Ryegrass (*Lolium perenne* L.) was used. The percentage recoveries of the standard sample ranged between 90.6% - 105.3%.

4.5. MadreenRoof Project

4.5.1. Green Roof Construction

The green roof studied in this work was installed on the first floor of the Instituto de Ciencias Agrarias (ICA-CSIC) of Madrid (40° 26' 25.5" N 3° 41' 13.3" W; https://madreenroof.csic.es/).

Figure 4.5: The making of the MadreenRoof project.

It was designed with a modular structure made of wood pallets and metal supports, having multifunctional purposes. The upper section sustained shading netting during summer and anti-bird mesh in winter to protect the crops; additionally, it guaranteed trellising plants based on their growth characteristics. The structure was surrounded by concrete and high-density polypropylene slabs, protecting the rooftop insulation and ensuring safe passage for individuals involved in the experiment (Figure 4.5).

The growing media tested in the present study were obtained by blending CG, CS, and SW composts, with and without BC (Table 4.5), with the HS at a proportion of 50 % (v/v), which was the mixing percentage that gave best results in the experiment described in section 4.4. The composts were previously grounded with a garden shredder to reduce the granulometry of the material up to an optimum (~ <1.6 cm).

Table 4.5: Composting mixture composition used in this study.

Composting mixture	**Seaweed (v:v)**	**Biochar (v:v)**	**Coffee Silverskin (v:v)**	**Green Waste (v:v)**	**Spent Coffee Grounds (v:v)**
BC-CG	0	1	0	1	1
CG	0	0	0	1	2
BC-CS	0	1	1	1	0
CS	0	0	1 (+1)[a]	1	0
BC-SW	1	1	0	1	0
SW	1	0	0	1	0

[a] Compost CS started with a ratio 1:1 (v:v) of CS and GW; a second dose of CS was added halfway through the process (15 days) to a CS:GW ratio of 2:1 (v:v).

The substrates were prepared in 70 L plastic pots using two distinct architectures (Figure 4.6):

1. The Mixed model consisted of a bottom layer of BC (5 cm) mimicking clay balls, followed by a BC-compost/peat mixture (33 cm). This architecture was used for the BC-containing composts: BC-CG, BC-CS, and BC-SW;
2. The “Lasagna” model featured a structural bottom layer of BC (19 cm) topped with compost/peat mixture (19 cm). The pattern was followed for the non-BC-containing composts: CG, CS, and SW.

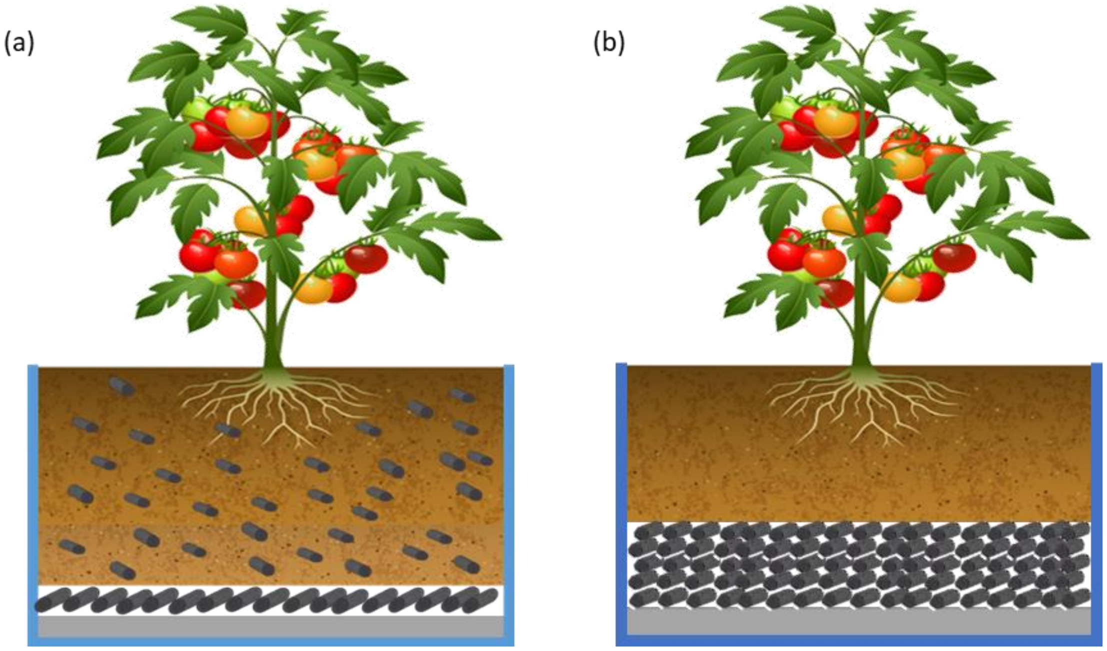


Figure 4.6: Growing media architectural models used: a) Mixed model, b) “Lasagna” model.

Additionally, a control growing medium was prepared using the same scheme of the mixed model. Each growing medium was replicated five times and arranged in a randomized block design (Figure 4.7). Flower pots were established to attract pollinating and beneficial insects.

After each cropping cycle, the total pot volume was refilled to its initial value to maintain consistent nutrient input conditions and provide sufficient volume for root anchorage. Watering was facilitated using a drip irrigation system equipped with self-compensating drippers with a flow rate of 2 L/h (Irritec® S.p.A.), adjusted according to the plants' phenological phases.

Under blocks A, C and D, a 10 L plastic tank was allocated for collecting leachates to evaluate nutrient loss and the quality of water discharged into the sewage system.

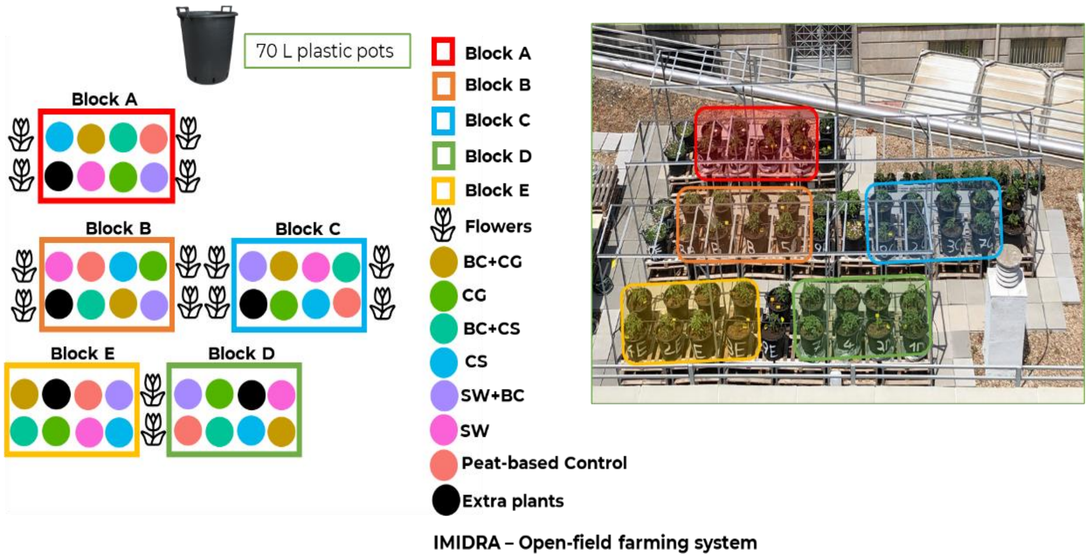


Figure 4.7: Randomly organized experimental design of the project Madreenroof (n=5).

Furthermore, an open-field on-soil farming system was used as a secondary control for the summer cycles, supervised by Drs Isabel Fernández Navarro and Maria Victoria Colombo Rodriguez. The experiment was conducted at the experimental farm of the *Instituto Madrileño de Investigación y Desarrollo Rural, Agrario y Alimentario* (IMIDRA), "Finca la Isla" in Arganda del Rey, set in the peri-urban area of Madrid, approximately 25 km from the rooftop experiment site. The study was designed in three randomly organized blocks containing ten plants each (n=10), with a planting layout of 0.5 m x 1.5 m.

4.5.2. Plant Development and Agronomical Performance

The agronomic performances of the growing media were tested through a three-year crop rotation cycle, consisting of a tomato landrace of the Community of Madrid (*Solanum lycopersicum* L., cv. Moruno) during spring/summer and a consociation of lettuce (*Lactuca sativa* L., cv. Romana) and swiss chard (*Beta vulgaris* var. cicla) during autumn/winter (Figure 4.8).

Figure 4.8: The three-year crop rotation established in the green roof experiment, from May 2021 to November 2023.

Tomato plantlets were sourced by the Instituto Madrileño de Investigación y Desarrollo Rural, Agrario y Alimentario (IMIDRA) and transplanted at 15 cm height. The tomato variety used in this study is a registered conservatory variety according to the Commission Directive 2008/62/EC of 20 June 2008 (European Commission, 2008), whose seeds are conserved at the Germplasm Bank of the Community of Madrid.

Prior to planting in the growing media, lettuce and chard seeds were germinated in a growing chamber under the condition described in section 4.4.2.

Tomatoes were progressively harvested once they reached a pink colour class (U.S.D.A., 1991), while lettuce and swiss chard were harvested after more than 90 days from transplantation (Figure 4.9). Total yield was calculated and expressed as fresh weight per plant.

Figure 4.9: Summer and winter crops at harvest time.

Data for tomato yield in the peat-based control for the third year are not shown due to plant death before harvest caused by the proliferation of *Alternaria solani* (Sorauer).

4.5.3. Physical and Chemical Analysis

4.5.3.1. Growing Media

Growing media were analysed at the beginning (t0, t12, t24) of each summer cycle. An aliquot of 100 g was collected per pot replicate and dried at 40 °C for at least three days. Samples' pH, EC, WHC, and bulk densit. Afterwards, samples were finely ground with a micro-whisk mill (Culatti AG, Steinerberg, Switzerland) equipped with a 2 mm diameter sieve to asses TC and N, as described in section 4.1.1., and Macro, Micronutrients and Trace Elements Content. A second aliquot of 20 g was collected and stored at 4 °C. Finally, samples were characterised as follow.

Phyto-available Nutrients

Phyto-available nutrients in growing media were assessed using the CAT-method (calcium chloride/DTPA) extraction method according to UNE-EN 13651.

A concentrated extraction solution was prepared with 0.1 M calcium chloride ($CaCl_2$) and 0.02 M DTPA ($C_{14}H_{23}N_3O_{10}$) in Milli-Q water at approximately 80 °C. After cooling, the concentrated solution was diluted with water at a ratio of 1:9 (v:v), resulting in the CAT extractant solution with a concentration of 0.01 mol/l $CaCl_2$ and 0.002 mol/l DTPA (pH=2.6). Subsequently, phyto-available nutrients were extracted in a 1:10 ratio (w:v) by mixing 4 g of the growing media sample with 40 ml of the CAT solution. Sample was rotated for 1 hour, centrifuged at 3000 rpm for 15 minutes, and then filtered using Whatman® Grade 2 filter paper. Finally, the extract was collected and stored at -20 °C. Sample were later analysed using an ICP-OES VARIAN 720-ES (Varian, Inc., Palo Alto, CA, United States), available at IRNAS-CSIC analytical service (Seville, Spain). The same interlaboratory sample, previously described in Section 4.3.1, was used as a reference sample.

Ammonium (NH_4^+) and Nitrate (NO_3^-)

NH_4^+ and NO_3^+ were extracted with KCl 1M at a 1:10 (w/v) ratio. Briefly, a 2.5 g fresh sample was combined with 25 ml of KCl, rotated for 1 hour, centrifuged for 10 minutes at 3500 rpm and filtered by Whatman® Grade 2 filtering paper.

NH_4^+ concentration was assessed according to the method described by Richard Mulvaney (1996), with minor modifications. Three reactive solutions were prepared as follows:

1) Ethylenediaminetetraacetic (EDTA) acid reagent was set by dissolving disodium salt of ethylenediaminetetraacetic (Na_2EDTA) 0.6 g in deionised water;

2) Sodium salicylate-sodium nitroprusside reagent was obtained with 1.5626 g sodium salicylate ($NaC_7H_5O_3$) and 0.025 g of sodium nitroprusside ($Na_2Fe(CN)_5NO\text{-}5H_20$) in deionised water. The reagent was left in the darkness;

3) Buffer hypochlorite reagent was made by mixing 0.296 g sodium hydroxide (NaOH) with 0.996 g sodium monohydrogen phosphate heptahydrate ($Na_2HPO_4 \cdot 7H_2O$) in deionised water, then 1 ml of sodium hypochlorite solution was added. Finally, pH was adjusted to 13 with NaOH.

Afterwards, 150 µL of the standards and the sample were pipetted into the multi-well transparent microplate, to which 15 µL of EDTA (1), 60 µL of Na-salicylate-nitroprusside reagent (2), and 30 µL of hypochlorite reagent (3) were added sequentially and mixed. After waiting in dark until the colour turned green (approx. 30 minutes), absorbance was read at 667 nm with a using a multimode microplate reader Synergy HTX (BioTek Instruments, Inc., Winooski, VE, United States). Results were referred to a 0-0.6 mg/l ammonium sulphate $(NH_4)_2SO_4$ calibration curve.

NO_3^- concentration was measured according the VCl3/Griess method described by Miranda, Espey, and Wink (2001). Three reactive solutions were prepared by:

1) Dissolving 0.16 g of Vanadium(III) chloride (VCl3) in an HCl 1M solution and vertexing until it turned in a clear blue colour;

2) Coupling reagent by blending N-(1-Naphthyl)ethylenediamine dihydrochloride (NEDD) 0.02 g in deionised water;

3) Diazotizing reagent by mixing 0.4 g of Sulphanilamide (C6H8N2O2S) in an HCl 5 % solution.

Reagent (1) and (3) were then filtered with a 0.45 µm nylon syringe. Hence, 100 µl of sample or standard were pipped in the microplate with 100 µL of VCl_3 (1). Then, 50 µl of NEDD (2) and Sulphanilamide (3) were added, and mixing was performed. After waiting in the dark until the colour turned pink (approx. 30 minutes), absorbance was read at 540 nm with the same microplate reader. Results were referred to a 0–10 mg l^{-1} potassium nitrate (KNO_3) calibration curve.

The sum of $N\text{-}NH_4^+$ and $N\text{-}NO_3^-$ was expressed as mineral nitrogen (N_{min}).

4.5.4. Tomato Quality Analysis

Quality analyses were conducted on tomatoes harvested at the red colour stage (U.S.D.A., 1991)during the second and third harvests, by selecting six tomatoes per treatment (n = 6).

They were carefully washed with distilled water, and green parts removal was ensured; subsequently, they were crushed using a blender and processed according to the quality parameter analysed.

Total Soluble Solids Content and Acidity

An aliquot of 10 ml of the sample was measured in a centrifuge tube, which was then centrifuged at 3000 rpm for 15 minutes. Finally, the supernatant (i.e., tomato juice) was collected.

Total soluble solids (TSS) were determined using 1 ml of tomato juice, with results expressed as Brix°. Simultaneously, 1 g of juice was diluted in water at a 1:50 (v:v) ratio, and 1 ml of the resulting tomato/water extract was employed to measure total acidity (TA), expressed as g citric acid l^{-1}. Total soluble solids content and acidity were assessed using a pocket Brix and acidity meter (PAL-BX/ACID3, Pal, Atago Co., Ltd., Tokyo, Japan). The ratio between the total soluble solids and total acidity content was expressed as TSS:TA.

Lycopene and β-carotene

Lycopene and β-carotene contents were quantified using the hexane/ethanol/acetone extraction method described by Anthon and Barrett (2007), with mild modifications.

A 2:1:1 (v:v:v) extraction solution was prepared by blending hexane, acetone and ethanol and adding 0.5 g l^{-1} of BHT (Butylated hydroxytoluene). Afterwards, 0.5 g of the frozen tomato sample, including exocarp and mesocarp, was combined with 10 ml of extracting solution and manually shaken. After resting for 30 min in darkness, the sample was centrifuged at 3000 rpm for about 10 min. Finally, 2 ml of supernatant was collected and analysed at 503 nm for lycopene and 444 nm for β-carotene, respectively.

The concentration of lycopene was calculated using the following formula:

$$Lycopene\left(\frac{mg}{kg}\right) = 1717xA503x\,V/W,$$

where 1.717 is a constant, A503 is the absorbance of the solution at 503 nm, V (ml) is the volume of mixed solvents added, and W (mg) is the weight of tomato analysed.

The following formula assessed the concentration of β-carotene:

$$\beta - carotene\left(\frac{mg}{kg}\right) = (9.38xA444 - 6.70xA503)x0.55x537x\,V/W,$$

where 9.38 and 6.70 are constants, A444 and A503 are the absorbance values at 444 nm and 503 nm, respectively, 0.55 is the ratio of the final hexane layer volume to the volume of mixed solvents added (hexane:acetone:ethanol in a 2:1:1 ratio), 537 is the molecular weight of β-carotene (g mol^{-1}), V (ml) is the volume of mixed solvents added, W (mg) is the weight of tomato juice analysed.

Total Antioxidant Capacity and Total Polyphenols

To evaluate total antioxidant capacity and total polyphenols, tomato samples underwent the extraction method outlined by Hartmann et al. (2008).

Four grams of fresh tissue were weighed in a plastic centrifuge tube and stored at -20 °C. The sample was then suspended in 8 mL of extraction mixture (60 % methanol, 30 % H_2O, 10 % acetone) and centrifuged at 5000 rpm for 15 min at 5 °C. The collected supernatant was placed in clean tubes.

The total antioxidant capacity (TAC) was assessed using the FRAP (Ferric Reducing Antioxidant Power) method developed by Benzie and Strain (1999), with slight modifications.

The FRAP reaction mixture was prepared by combining a 10:1:1 ratio (v:v:v) of 300 mM acetate buffer (pH 3.6) with 10 mM 2,4,6-Tris(2-pyridyl)-s-triazine (TPTZ) in 40 mM HCl and 20 mM $FeCl_3$ and incubating it in darkness at 37°C for 2 hours. Then, 2.4 ml of FRAP was added to 40 µl of samples and standards and incubated for 1 h at room temperature in darkness. Deionized water served as blank. Total antioxidant capacity was read at 593 nm and values were referred to a 0.625–10 mM $FeSO_4$ calibration curve and expressed as Fe^{2+} equivalents on a fresh weight basis.

Total polyphenols (TPF) were measured using the Folin-Ciocalteau colourimetric protocol described by Waterhouse (2003).

A 15:1 (v:v) reaction mixture was prepared by mixing H_2O and Folin-Ciocalteu reagent. Afterwards, 1.6 mL of the reaction mixture was added to 100 µl of samples and standards and incubated for 5 minutes at room temperature, so 0.3 ml of 20 % Na-carbonate solution was added. Deionized water represented the blank. Samples were incubated for an additional hour at room temperature. and the absorbance was read at 765 nm. Values were referred to a gallic acid ($C_7H_6O_5$) calibration curve at 0.625–1 mg/ml concentration range and expressed in gallic acid equivalents on a fresh weight basis.

Spectrophotometric analysis for lycopene and ß-carotene contents, total antioxidant capacity, and total polyphenols was performed using a UV-1800 UV-Vis spectrophotometer (Shimadzu Corporation, Kyoto, Japan).

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PART II

Composting and Compost-based Substrates

Chapter 5

Urban Garden in Barcelona, Spain (Giuseppe Picca®)

A Sustainable Solution for Coffee Waste Management

5. A Sustainable Solution for Coffee Waste Management

The current chapter describes the evolution of the studied parameters of the co-composting process of coffee silverskin (CS) with carbon-rich materials at different dilutions. Finally, evidence of the potential use of this biomass in the agricultural sector is provided.

5.1. Composting Stages

The composts followed the standard course of the composting process, adhering to a first mesophilic phase (18–45 °C), followed by a thermophilic phase (45–70 °C), and a second mesophilic stage, concluding with a maturation phase that occurred in the open air for thirty days (Figure 5.1a). The first mesophilic phase was notably more prolonged for CS-3 compared to the other composts. Following the second addition of silverskin, the temperature of CS-1 initially dropped but rebounded three days later, reaching 59.7 °C within five days.

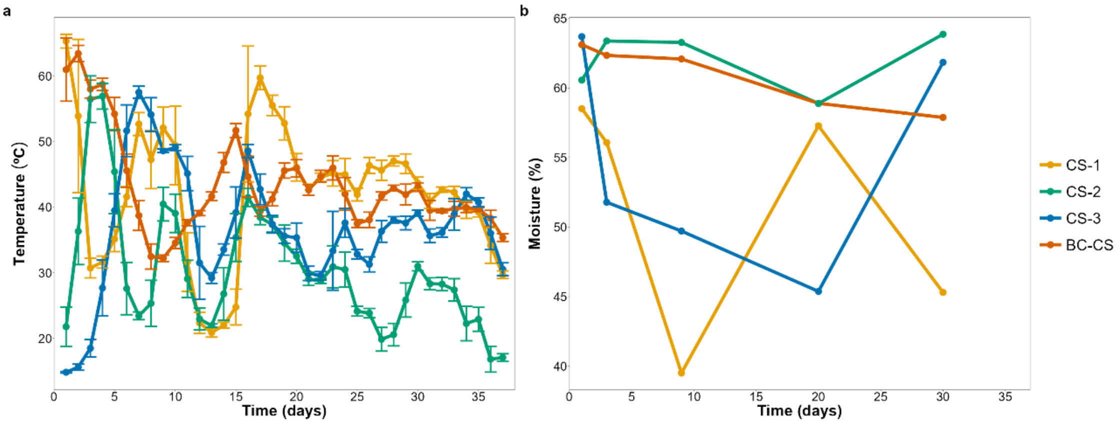


Figure 5.1: Temperature trend (a) and moisture content (b) of the four composts throughout the process. Mean values are expressed as solid lines, shadows indicate standard deviations.

Temperature increases are generally attributed to the metabolization of easily soluble compounds (e.g., sugars, starch, and simple proteins) and the insulation provided by the composting mass against heat loss (Diaz, De Bertoldi, and Bidlingmaier 2007). The thermophilic phase lasted 25 days for

CS-1, 24 days for BC-CS, 6 days for CS-2, and 13 days for CS-3; the respective peak temperatures recorded were 59.7, 63.4, 56.9, and 58.5 °C, respectively. An exposure time of at least two weeks at 55 °C is considered the optimum condition to effectively eliminate human and plant pathogens, weed seeds, and insect larvae (Diaz, De Bertoldi, and Bidlingmaier 2007). The shorter thermophilic phase of CS-2 may be attributed to the heap compaction observed during composting (Diaz, De Bertoldi, and Bidlingmaier 2007). In the case of BC-CS, the rapid temperature increase and overall temperature profile could be attributed to the presence of BC. Biochar may fill free spaces between particles of raw materials, reducing heat losses during composting (Zhang et al. 2016). Moreover, BC addition provides suitable conditions for the proliferation and activity of microorganisms through increased aeration, water and nutrient retention, and toxin adsorption (Sanchez-Monedero et al. 2018). Temperature drops registered after around 30 days indicated the slowdown of microbiological processes, signalling that all composts reached stability, even before the maturation phase lasted another 30 days.

The moisture content of the four composts was maintained between 40 and 60 % throughout the process to ensure optimal conditions for microbial communities (Figure 5.1b). However, the moisture content for CS-3 was purposely adjusted to a lower value to prevent compaction and ammonia emissions due to the high silverskin content. Conversely, the coarser texture of CS-1 resulted in greater water evaporation during the first ten days.

At the beginning of composting, CS-1, BC-CS, and CS-2 exhibited a slightly alkaline pH (7–8) (Figure 5.2a). Since silverskin had a pH of 5.5, CS-3, the mixture with the highest content of silverskin, showed an acidic pH (Figure 5.2a). As composting progressed, the pH of all mixtures increased to between 8 and 9 during the first week, stabilising after 30 days, except for CS-1, where a slight decrease in pH was observed during the maturation process. By the end of composting, the pH of CS-1 was slightly lower than those of BC-CS and CS-2 (8.5) and notably lower than that of CS-3 (9) (Figure 5.2a).

The pH increase observed during composting may be attributed to the release of ammonia associated with protein degradation. Conversely, the subsequent decrease during maturation may result from the volatilisation of ammoniacal nitrogen and proton release due to microbial nitrification (Gómez-Brandón, Lazcano, and Domínguez 2008). Monitoring pH is crucial for controlling N-losses through ammonia volatilisation, which can be particularly significant if the pH rises above 7.5. Furthermore, pH values within the range of 6.7–9.0 support good microbial activity during composting, and levels lower than 9 indicate that the four composts were safe for plant health and agricultural use (Madejón et al. 2021).

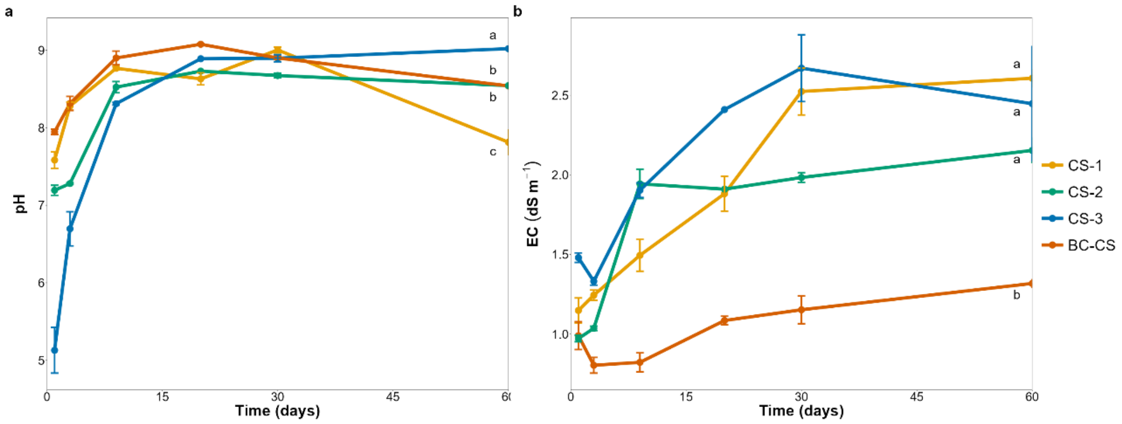


Figure 5.2: Mean (solid line) ± SD (shadow) of the pH (a) and the electrical conductivity (EC) values (b) of the four composts throughout the process. Different letters identify significant differences among final composts at a $p < 0.05$ level.

Composts' EC increased along the process, especially in CS-1, CS-2, and CS-3, registering final values above 2 dS m^{-1} (Figure 5.2b). As organic matter degrades, the concentration of cations increases, leading to a surge in EC of the material (Diaz, De Bertoldi, and Bidlingmaier 2007). The lower BC-CS electrical conductivity (below 2 dS m^{-1}) might be attributed to the presence of reactive surfaces within the high pore space of biochar, which can enhance the adsorption of cations. This adsorption mechanism is typically described as a result of the interactions of cations with the functional groups in the biochar skeleton (Kammann et al. 2015) or, with the π electrons of the condensed aromatic rings or the formation of colloids (Verheijen et al. 2014).

EC is commonly used as an indicator of the salinity of soils and growing substrates. High salinity can induce osmotic stress, which may adversely affect microbial populations involved in composting and can limit plant growth and productivity. Recommended EC limits are typically around 2 dS m^{-1}. However, species of the Solanaceae family, such as tomatoes and aubergines, can tolerate values up to 2.5 dS m^{-1} (Maas & Hoffman, 1977; Machado & Serralheiro, 2017).

All final composts generally achieved satisfactory pH and EC values. The presence of biochar enhances the thermophilic profile and reduces the electrical conductivity of the product, although it has minimal effect on their pH.

5.2. Carbon and nitrogen evolutions during composting

The carbon content (C) of CS-1, CS-2, and CS-3 slightly decreased throughout the composting process, a common trait of organic matter degradation (Figure 5.3a). However, BC-CS had an opposite trend due to the presence of biochar. At the end of composting, the C contents of CS-1, CS-2, and CS-3 were significantly lower than that of BC-CS (Figure 5.3a). The increase in the C content of BC-CS can be attributed to the high stability of biochar C, which is likely to be left barely intact during the process, and the relatively higher loss of other elements within composting. Indeed, its structure gives biochar higher stability against microbial degradation carbon, meaning that carbon pools from biochar have longer persistence times and a higher carbon sequestration potential than labile carbon sources from silverskin and green waste (Johannes Lehmann and Stephen Joseph 2012). In addition, functional groups on the biochar surface led to the chemisorption of the dissolved organic carbon (Sanchez-Monedero et al. 2018), rendering them stable and contributing to the increasing trend for C content observed for BC-CS. These results highlight the positive effects of BC, a co-composting agent, on reducing the amount of organic fraction degraded and GHG emissions (Khan et al. 2016).

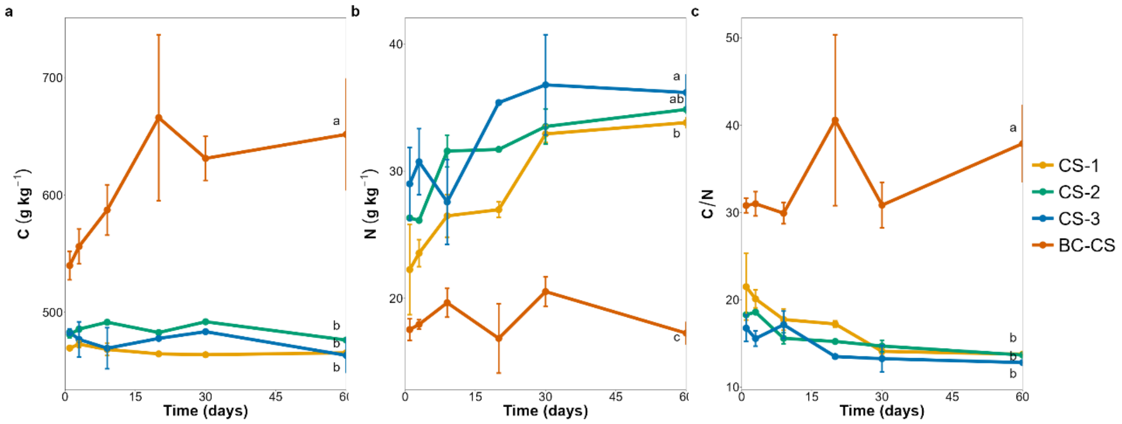


Figure 5.3: Mean (solid lines) ± SD (shadows) of the Total Carbon (a) and Nitrogen (b) contents, and the C/N ratio (c) of the four mixtures measured during the composting process and for final composts. Different letters identify significant differences between final composts at a $p < 0.05$ level.

The nitrogen (N) content of compost without biochar tended to increase during the process, while that of the mixture with BC remained constant (Figure 5.3b). Indeed, mature BC-CS had a significantly lower content of N than the other composts. In contrast, N content increased with the proportion of added CS, and CS-3 was significantly richer in N than CS-1 (Figure 5.3b). All the composts presented more than 1.5 % N content and thus can be classified as high-quality products following the classification by Diaz et al. (2007).

The C/N ratios measured ranged from 18 to 30 (Figure 5.3c). The C/N ratio is commonly assumed to decrease from the optimal value of 25 for the starting pile to 15 at the final stage (Diaz, De Bertoldi, and Bidlingmaier 2007). The use of low C/N ratio amendments poses a phytotoxicity risk, favouring the conversion of NH_4^+ into NH_3 and/or the release of organic acids due to the immaturity of compost. At the same time, when compost with a high C/N ratio is added to soil, the microbial population tends to compete with plants for soil N, leading to the so-called "nitrogen starvation" (Azim et al. 2018). The typical targets for the C/N ratio of composting mixtures could not be applied for blended biochar composts because of the high stability of most biochar C, meaning that some of the C pools are not involved in the microbial metabolism of the compost; Khan et al. (2016) suggested that in matured co-composted biochar the value of C/N can be higher than 21, as in the present study.

Namely, 85-90 % of the total N content in composting mixtures is organic (e.g., proteins, simple peptides, nucleic acids), and only 10-15 % is inorganic N forms immediately available to the plants (Diaz, De Bertoldi, and Bidlingmaier 2007). During composting, the organic N fraction undergoes microbial mineralisation, a two-stage process: the ammonification of complex organic nitrogen to inorganic ammonia leading to the accumulation of NH_4^+, followed by the nitrification or microbial oxidation of ammonia (NH_3) via nitrite (NO_2^-) to nitrate (NO_3^-) (Wong, Wang, and Selvam 2017).

The low ammonium concentration in CS-1 and BS-CS at the beginning of composting (Figure 5.4a) can be explained by the low starting concentration of CS and by the greater efficiency of the system in transforming NH_4^+ to NO_3^-, especially for BS-CS since it has been proved that the BC presence provides a favourable microenvironment for nitrifying bacteria (Kammann et al. 2015). CS-2 exhibited the lowest initial concentration of ammonium, a marked peak raising to 13.6 g kg^{-1} at around day 30, and a final drop after 30 days to 0.09 g kg^{-1} (see Figure 5.4a). The ammonification peak usually reached in the first stages of composting corresponds with the maximum biodegradation period. The high NH_4^+ peak registered on day 30 can be related to the 10 °C rise recorded as the outcome of the growth of microbial activity (Figure 5.1a). For CS-3, ammonia-like odours were noticed during the aeration procedures. This may be attributable to the physical and chemical characteristics of CS presented at high doses in this mixture. On the one hand, N contents increased with the proportion of added CS; on the other hand, CS has a great WHC and a compaction behaviour when humid, favouring the onset of anaerobic spots, provoking N-loss via ammonia, nitrogen oxides (NO_x), and nitrous oxide (N_2O) emissions (Wong, Wang, and Selvam 2017).

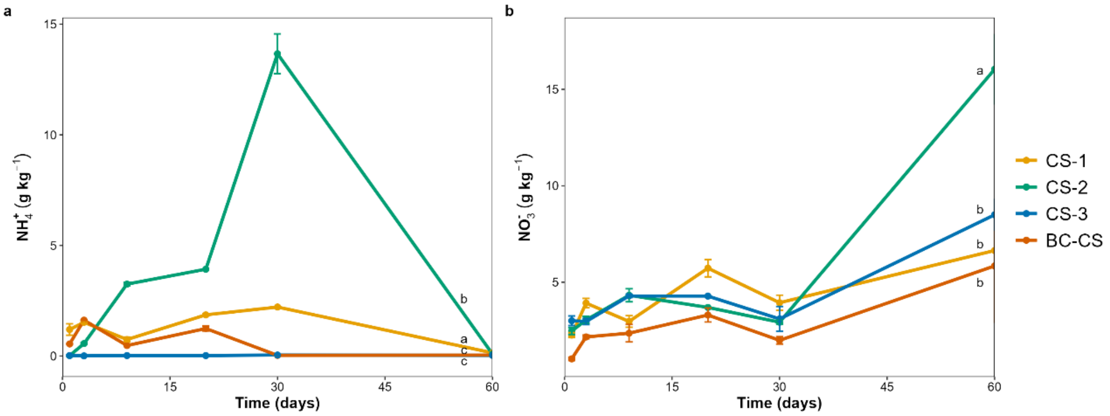


Figure 5.4: Mean (solid lines) ± SD (shadows) of the ammonium (a) and nitrate (b) contents of the four composts throughout the process. Different letters identify significant differences between final composts at a $p < 0.05$ level.

Nitrate concentration for all the composts showed an upward trend throughout the first phases of the composting process (see Figure 5.4b), reaching more than 6 g kg^{-1} NO_3^- for CS-1 and BC-CS, more than 9 g kg^{-1} for CS-3, and more than 16 g kg^{-1} for CS-2 in their final composts (see Figure 5.4b). The declining NH_4^+ and the proportional increment of NO_3^- were in line with the steady evolution of the curing stage; indeed, nitrification occurs predominantly at mesophilic temperatures (20-35 ° C) (Wong, Wang, and Selvam 2017). This trend, together with the NH_4^+/NO_3^- ratio < 0.16 registered for all the mixes, is an index of high-quality amendments and confirms that the compost reaches the maturation phase (Khan et al. 2016).

Several authors agreed that passive methods, such as adsorption by adding BC to compost, effectively reduce nitrogen losses (Khan et al. 2016; Kammann et al. 2015). Indeed, BC absorbs/adsorbs NH_3, NO_3^- and NH_4^+, and a wide range of organic N compounds, reducing greenhouse gas emissions and improving the fertiliser properties of compost (Sanchez-Monedero et al. 2018). Thus, using BC constitutes a solution for excessive compaction, anaerobiosis and N losses for those composting mixtures aiming for high N contents, even using a higher proportion of CS than those evaluated by the present work.

5.3. Agrochemical quality of the final compost

CS is a by-product of the food chain, and its trace element content is strictly regulated. Thus, the final composts' micro-, macro-, and trace element contents must align with Spanish national compost quality regulations (Royal Decree 506/2013) to qualify as Class A amendments (Boletín Oficial del Estado 2020) (Table 5.1).

Table 5.1: Final characterization of the four composts obtained. Final characterization of the four composts obtained. Mean values for different measured variables (N = 3).

Elements	**Units**	**CS-1**	**CS-2**	**CS-3**	**BC-CS**	**Limits established by the Spanish regulation [a]**
pH		7.8	8.5	9	8.5	
EC	(dS m^{-1})	2.6	2.15	2.4	1.3	
Bulk density	(g cm^{-3})	0.4	0.5	0.4	0.5	
WHC	(% of DW)	350	304	335	237	
C	g kg-1	465	476	463	651	
N	g kg-1	33.8	34.8	36.2	17.3	
C/N		13.8	13.7	12.8	37.9	
NH_4^+	g kg^{-1}	0.14	0.09	0.03	0.04	
NO_3^-	g kg^{-1}	6.65	16.0	8.50	5.85	
P	g kg^{-1}	1.32	1.62	1.76	1.78	
K	g kg^{-1}	3.88	4.50	5.30	1.87	
Ca	g kg^{-1}	1.72	2.35	1.91	2.14	
Fe	g kg^{-1}	0.78	1.00	1.03	1.75	
Mg	g kg^{-1}	4.19	5.21	5.27	3.73	
Al	g kg^{-1}	1.37	1.85	3.12	3.80	
Na	mg kg^{-1}	223	369	541	340	
Mn	mg kg^{-1}	79	101	89	73	
Cd	mg kg^{-1}	< 0.003	< 0.003	< 0.003	< 0.003	0.7
Cu	mg kg^{-1}	45	59	55	21	70
Ni	mg kg^{-1}	3.3	5.0	5.2	5.4	25
Pb	mg kg^{-1}	< 0.02	< 0.02	< 0.02	< 0.02	45
Zn	mg kg^{-1}	19	32	24	31	200
Cr (Total)	mg kg^{-1}	6	10	14	38	70

[a] Spanish Royal Decree 506/2013 on Fertiliser Products – Class A (Boletín Oficial del Estado 2020).

All four composts demonstrated optimal phosphorus (P) levels and notably high potassium (K) values, surpassing those typically found in composts derived from vegetal by-products. The elevated micronutrient concentrations, such as calcium (Ca), iron (Fe), and magnesium (Mg), were also observed. The nutrient range generally increased with higher proportions of CS in the initial mixture. However, exceeding the CS proportion beyond that of CS-3 could lead to copper (Cu) levels surpassing legal limits in compost. While Cu is crucial for plant micronutrition, excessive levels can induce plant chlorosis by interfering with redox and free radical reactions (Yruela 2005). Nevertheless, the composts' basic pH may restrict Cu mobility (Abd El-Azeem et al. 2013), mitigating potential contamination risks.

The registered composts' WHC was 351 %, 237 %, 305 %, and 335 %, respectively (Table 5.1). These high values are attributed to CS's superior WHC compared to other plant by-products like cereal husks and straws (Ballesteros, Teixeira, and Mussatto 2014). These amendments will likely enhance water retention, benefiting plant growth and substrate structure (Domínguez et al. 2020).

Germination tests (Table 5.2) confirmed the phytotoxic effect of CS, likely mediated by Chlorogenic acid (Al-Charchafchi and Al-Quadan 2010), which discourages its agricultural use as an amendment or mulching agent. However, the resulting composts (see Table 5.2) did not exhibit phytotoxicity and even demonstrated phytostimulant properties, especially BC-CS and CS-3. The reduction in phytotoxicity confirms that the compost achieved full maturity concerning the nitrogen cycle and decomposed potentially phytotoxic organic substances from CS, primarily phenolic compounds and Chlorogenic acid, during composting. These results support the safe application of the four composts (Gómez-Brandón, Lazcano, and Domínguez 2008).

Table 5.2: Germination test of the co-composted materials and the final composts according to Zucconi et al. (1981).

Zucconi Test	**Biochar**	**Pruning waste**	**Silverskin**	**CS-1**	**CS-2**	**CS-3**	**BC-CS**
Germination Index	94.89	111.25	0	124.97	101.08	109.50	121.84
Classification	No phytotoxic	Phyto-stimulant	Phytotoxic	Phyto-stimulant	Phyto-stimulant	Phyto-stimulant	Phyto-stimulant

5.4. The potential of recycling coffee silverskin in agriculture through composting

Our results affirm that composting presents a low-cost, viable, and efficient means to recycle CS, a by-product rich in nitrogen (N) and potassium (K). CS contains trace metals well below concerning levels, and composting diminishes its phytotoxicity.

CS, being a food-grade by-product, poses minimal safety concerns and is readily available throughout the year due to the consistency in roasting processes, which are relatively unaffected by seasonal variations. Once the proportions for composting piles are established, minimal adjustments are needed, simplifying handling and logistical aspects.

In 2020, Europe imported a significant volume of green coffee, producing approximately 167 thousand tons of CS during the roasting process. Wong et al. (2017) estimated N losses during composting at 18 to 41 % for materials akin to CS under similar conditions. Based on this and our study's findings, composting Europe's produced CS could recover 2,420 to 3,481 tons of N and 1,874 tons of K and other essential micro and macro-nutrients. Considering that composting CS accumulated annually could fulfil 0.01–0.02 % of total N fertiliser demand and 0.05 % of K_2O fertiliser demand in Europe (Faostat), it significantly reduces reliance on mineral fertilisers, thereby addressing the challenge of meeting a more sustainable food production system.

Promoting eco-friendly management of agricultural by-products is crucial in combating global warming. It aids in reducing greenhouse gas emissions, minimising reliance on mineral fertilisers, and aligning with the EU's target of reducing landfill by 10 % by 2030 (European Commission 2018).

Ronga et al. (2016) suggest that composts derived from coffee represent a feasible substitute for peat in the production of potted plants. Specifically, the combination of coffee by-products and biochar (BC) enhances water and air retention capacity, decreases bulk density, and increases cation exchange capacity, thereby reducing the risk of nitrogen (N) loss (Jindo et al. 2020). These properties make this material appealing for use in open-air RA systems.

Chapter 6

Ripe Tomatoes in the ICA greenhouse (Giuseppe Picca®)

Co-composted Growing Media

6. Co-composted Growing Media

The present chapter outlines the agronomical performances of two spent coffee ground-based composts with (BC+CG+GW) and without (CG+GW) biochar. Composts were tested at different dilutions to assess their potential as peat replacements. The study was conducted on three different stages of tomato (*Solanum lycopersicum* L.) development: seed germination (0-6 days), seedling development (7-49 days), and plant-to-fruit maturity (36-100 days).

6.1. General Characteristics of Coffee Grounds-based Composts

During the composting process, BC+CG+GW and CG+GW underwent the thermophilic phase for 22 and 14 days, respectively, achieving sanitation conditions. After composting, BC+CG+GW and CG+GW had pH values of 8.6 and 6.8, respectively (Table 6.1); the higher pH value recorded for BC-based compost may be attributed to the alkaline nature of biochar, which increases the pH of growing substrates (Huang and Gu 2019). Electrical conductivity (EC) ranged between 1.05 and 2.02 dS m^{-1}, with BC+CG+GW showing the lower value (Table 6.1). Biochar may help reduce salinity by retaining nutrients, allowing for larger compost in growing media (Sánchez-Monedero et al. 2021). In compost, the carbon-to-nitrogen ratio (C/N) is deemed as maturity index in compost, with a C/N ratio lower than 21 indicating adequate maturity (Leege 1998). Khan et al. (2014) suggested that for co-composted biochar, the maturity of the compost could be achieved at higher C/N ratios due to the high stability of BC, as observed in this study (Table 6.1.1).

Table 6.1.1: Main physical and chemical characteristic of the growing media (mean values ± standard deviations, N = 3).

	Units	Control	BC-CG 100	BC-CG 50	BC-CG 30	BC-CG 15	CG 100	CG 50	CG 30	CG 15
pH		5.4 ± 0.01 ^{h}	8.6 ± 0.04 ^{a}	6.9 ± 0.01 ^{b}	6.7 ± 0.02 ^{c}	6.1 ± 0.03 ^{f}	6.8 ± 0.01 ^{b}	6.4 ± 0.03 ^{d}	6.2 ± 0.02 ^{e}	**5.8 ± 0.01** ^{g}
EC	(dS m^{-1})	0.5 ± 0.00 ^{b}	1.1 ± 0.01 ^{b}	0.8 ± 0.01 ^{d}	0.6 ± 0.00 ^{e}	0.8 ± 0.00 ^{c}	2.0± 0.01 ^{a}	1.1 ± 0.00 ^{b}	0.8 ± 0.00 ^{d}	**0.8 ± 0.01** ^{d}
Bulk density	(g cm^{-3})	0.3 ± 0.01 ^{a}	0.2 ± 0.02 ^{b}	0.2 ± 0.01 ^{b}	0.2± 0.00 $^{c, d}$	0.2 ± 0.00 $^{b, c}$	0.2 ± 0.00 ^{b}	0.1 ± 0.01 $^{d, e}$	0.1 ± 0.01 $^{d, e}$	**0.1 ± 0.01** ^{e}
WHC	(% of DW)	323 ± 17 $^{a, b, c}$	269 ± 49 $^{b, c}$	278 ± 40 $^{b, c}$	304 ± 20 $^{a, b, c}$	346 ± 13 $^{a, b}$	358 ± 56 $^{a, b}$	256 ± 21 ^{c}	282 ± 15 $^{a, b, c}$	**371 ± 7** ^{a}
Ctot	g kg^{-1}	327 ± 1.0 ^{f}	612 ± 2.0 ^{a}	438 ± 7.0 ^{d}	515 ± 17 ^{b}	496 ± 1.0 $^{b, c}$	490 ± 2.0 ^{c}	437 ± 3.0 $^{d, e}$	419 ± 3.0 ^{e}	**496 ± 0.1** ^{c}
Ntot	g kg^{-1}	12 ± 0.1 ^{d}	23 ± 0.2 $^{b, c}$	17 ± 0.3 $^{c, d}$	21 ± 1.4 $^{b, c}$	20 ± 0.4 $^{b, c}$	37 ± 7.3 ^{a}	27 ± 0.7 ^{b}	19 ± 0.6 $^{c, d}$	**21 ± 0.9** $^{b, c}$
C/N		27 ± 0.1 ^{a}	27 ± 0.1 ^{a}	26 ± 0.3 ^{a}	25 ± 2.3 $^{a, b}$	25 ± 0.4 $^{a, b}$	14 ± 2.7 ^{c}	16 ± 0.4 ^{c}	22 ± 0.8 ^{b}	**24 ± 1.0** $^{a, b}$
NO_3^-	mg kg^{-1}	5.9 ± 0.7 ^{b}	1.4 ± 0.1 ^{c}	1.8 ± 0.1 ^{c}	1.3 ± 0.1 ^{c}	1.7 ± 0.0 ^{c}	12 ± 1.8 ^{a}	2.9 ± 0.4 ^{b}	2.1 ± 0.2 ^{c}	**1.3 ± 0.1** ^{c}
NH_4^+	mg kg^{-1}	0.01 ± 0.00 ^{b}	0.09 ± 0.02 ^{b}	0.02 ± 0.00 ^{b}	0.01 ± 0.00 ^{b}	0.01 ± 0.01 ^{b}	2.5 ± 0.10 ^{b}	0.01 ± 0.00 ^{b}	0.01 ± 0.01 ^{b}	**0.004 ± 0.00** ^{b}

* Safety limits established by the Spanish regulation on compost (Class A compost). † Safety limit of Cd concentration for "Class B" compost is 2 mg kg^{-1}. BC-CG: biochar-blended compost. CG: compost. The percentages of peat substitution (v:v) with compost materials were 100, 50, 30, and 15

Table 6.1.2: Main physical and chemical characteristic of the growing media (mean values ± standard deviations, N = 3).

	Units	Control	BC-CG 100	BC-CG 50	BC-CG 30	BC-CG 15	CG 100	CG 50	CG 30	CG 15
P	g kg^{-1}	0.4	1.5	1.2	1.1	0.7	3.6	2.0	1.2	**0.8**
K	g kg^{-1}	3.3	7.9	7.1	4.8	2.4	17.0	11.1	7.5	**2.3**
Ca	g kg^{-1}	10.7	15.6	17.6	24.0	25.5	13.2	15.6	15.8	**23.7**
Fe	g kg^{-1}	7.9	3.5	8.5	2.6	2.3	0.9	0.9	9.8	**1.9**
Mg	g kg^{-1}	3.1	3.5	4.4	4.2	4.1	3.7	4.6	4.1	**3.7**
Al	g kg^{-1}	11.3	2.5	7.7	1.7	1.3	0.8	10.3	10.5	**1.0**
Mn	mg kg^{-1}	59	96	131	58	37	62	118	98	**31**
Na	mg kg^{-1}	178	178	207	175	148	229	216	196	**145**
Cd	mg kg^{-1} (0.7) *	< 0.003	< 0.003	< 0.003	< 0.003	< 0.003	< 0.003	1.20 †	< 0.003	**0.7**
Cu	mg kg^{-1} (70) *	20	25	32	26	23	62	49	35	**26**
Ni	mg kg^{-1} (25) *	12.6	6.2	11.9	3.8	10.5	5.0	12.6	18.8	**4.9**
Pb	mg kg^{-1} (45) *	6.5	0.7	4.8	1.2	2.1	3.1	7.4	8.7	**1.2**
Zn	mg kg^{-1} (200) *	25	30	40	24	27	49	51	40	**20**
Cr (Total)	mg kg^{-1} (70) *	6.1	48.0	34.3	26.5	18.8	23.9	15.9	24.6	**17.4**

* Safety limits established by the Spanish regulation on compost (Class A compost). † Safety limit of Cd concentration for "Class B" compost is 2 mg kg^{-1}. BC-CG: biochar-blended compost. CG: compost. The percentages of peat substitution (v:v) with compost materials were 100, 50, 30, and 15

Mixing the two composts with peat altered the compost-based substrate characteristics. Globally, pH values ranged between 6.9 and 5.8, conducive to tomato production (Ronga et al. 2016). However, the EC values exceeded the recommended ≤ 0.5 dS m^{-1} level for optimal growing media (Sánchez-Monedero et al. 2021). The lower EC level observed in the BC-CG substrates could be caused by cation sorption on the reactive surfaces within the high BC pore space (Picca et al. 2022).

The heavy metal content was not a concern, as it remained below the threshold of potentially toxic elements set by the Spanish Royal Decree 506/2013 for 'Class A' compost (Boletín Oficial del Estado 2020). This finding provides reassurance about the safety of the compost, except for Cd in the BC-CG 50, which would comply with the Class B compost limits.

The bulk density values of all compost-based growing media fell within the optimal range (≤0.5 g cm^{-3}) required for an ideal substrate (Nieto et al. 2016). Regarding water holding capacity (WHC), no significant differences were noted among substrates compared to the control, except for CG 50, which decreased by 20.7 %, and CG 15, which increased by 15 %.

6.2. Germination and Seedling Development Phases

All the growing media induced a Germination Index (GI) greater than 80 %, indicating that the materials were free from phytotoxins and, in most cases, exhibited phyto-stimulant properties (Table 6.2). Ronga et al. (2016) reported a decline in germination rates of tomato and basil seeds when mixed with CG compost at percentages greater than 30 %. Conversely, Hachicha et al. (2012) found that co-composted CG with olive mill wastewater sludge and poultry manure had phyto-stimulant effects on lettuce and barley seeds. Adverse effects on plants have been attributed to high chlorogenic acid concentrations, inhibiting germination and reducing root development (Janissen and Huynh 2018). In a previous study, composting coffee by-products was found to facilitate the decomposition of phytotoxic compounds, phenolic substances, and chlorogenic acid, enabling the recycling of these valuable resources in agriculture (Picca et al. 2022).

Table 6.2: Germination index (±standard deviations) of the co-composted substrates.

	Control	***BC-CG 100***	***BC-CG 50***	***BC-CG 30***	***BC-CG 15***	***CG 100***	***CG 50***	***CG 30***	***CG 15***
Germination Index	115 ± 9	103 ± 10	110.5 ± 15	106.5 ± 11	331 ± 8.5	114 ± 5.5	86 ± 17	131 ± 15	271 ± 5
Classification	Phyto-stimulant	Phyto-stimulant	Phyto-stimulant	Phyto-stimulant	Phyto-stimulant	Phyto-stimulant	No phytotoxic	Phyto-stimulant	Phyto-stimulant

After 49 days, only 60 % of the plants grown in BC-CG 100 and none in CG 100 survived (Table 6.2). One Control seedling perished due to mechanical damage during watering, reducing the control survival rate to 80 %. The heights of the seedlings cultivated in BC-CG and CG were, on average, one-third of those in the Control (Figure 6.1). The reduction in seedling height was more pronounced as the compost concentration in the mixture increased, although no significant differences were found between the different peat-replacement treatments.

Table 6.3: Mean values ± standard deviations of survival rates (%), above- and below-ground biomass (g), and above/below-ground ratio (%) of seedlings grown in BC-CG and CG substrates. Different letters indicate significant differences among treatments at $p < 0.05$ level.

	Control	*BC-CG 100*	*BC-CG 50*	*BC-CG 30*	*BC-CG 15*	*CG 100*	*CG 50*	*CG 30*	*CG 15*
Survival rates (%)	80	60	100	100	100	0	100	100	100
Above-ground biomass (g)	7.5 ± 0.4 ^{a}	1.7 ± 1.3 ^{b}	1.2 ± 0.9 ^{b}	0.5 ± 0.2 ^{b}	1.7 ± 0.6 ^{b}	-	1.3 ± 0.2 ^{b}	1.0 ± 0.8 ^{b}	1.7 ± 0.6 ^{b}
Below-ground biomass (g)	3.4 ± 0.9 ^{a}	0.6 ± 0.5 ^{b}	0.7 ± 0.1 ^{b}	0.2 ± 0.1 ^{b}	0.7 ± 0.1 ^{b}	-	0.5 ± 0.2 ^{b}	0.4 ± 0.3 ^{b}	1.1 ± 0.6 ^{b}
Above/below-ground ratio	2.3 ± 0.6	3.5 ± 0.6	1.7 ± 1.3	2.3 ± 0.8	2.5 ± 0.6	-	2.7 ± 0.9	2.4 ± 0.5	1.8 ± 0.6

The heights of the seedlings cultivated in BC-CG and CG were, on average, one-third of those in the control (Figure 6.1). The decrease in seedling heights was more pronounced as the compost concentration in the mixture increased, although no significant differences were found between the different peat-replacement treatments.

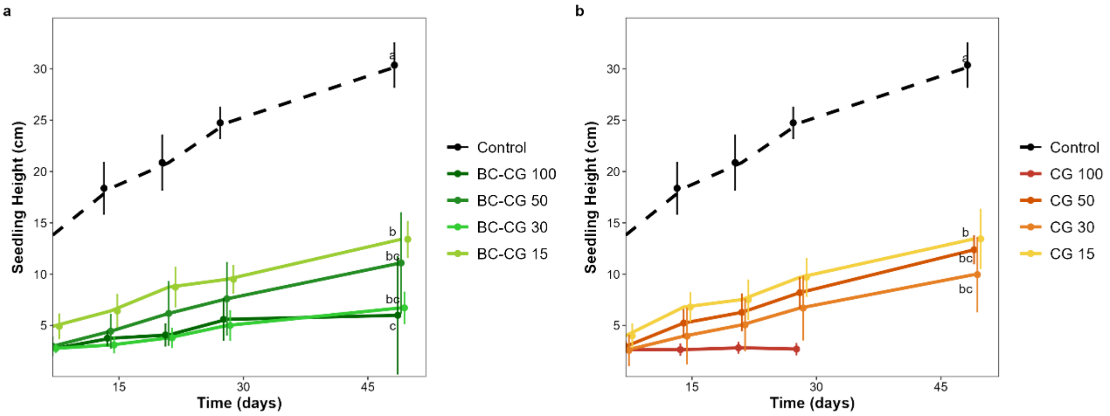


Figure 6.1: Mean (solid lines) ± standard deviations (shadows) of the height of the seedlings grown in BC-CG (a) and CG (b) over 49 days. Different letters indicate significant differences among treatments at $p < 0.05$ level.

High EC values have been shown to limit the inclusion of compost in nursery media (Nocentini et al. 2021; Nieto et al. 2016). Elevated salinity levels lead to osmotic stress, ion imbalance, oxidative stress, and metabolic abnormalities, ultimately resulting in stunted growth (Yang and Guo 2018).

Herrera et al. (2008) observed that excessive soluble salts in municipal solid waste compost, used at doses higher than 30 % in growing media, hindered the development of tomato seedlings. Similarly, Kumar et al. (2022) in their study of spent mushrooms digestate with slightly high EC values, noted a decrease in seed germination and seedling stem and root length exceeding the optimal dilution of 10 %. Huang et al. (2019) found that the high EC of biochar co-composted with chicken manure caused slow seed germination and suppressed the growth of various ornamental and agricultural species. However, Gascó et al. (2016) highlighted that biochar's phytotoxicity on seed germination is a multifaceted issue influenced by the type of biochar and seeds.

The biomass produced during the experiment (Table 6.4) was 3.4 % to 14 % lower for BC-CG and CG growing media compared to the control. Seedlings from all treatments produced approximately 1.5 % more above-ground biomass than below-ground biomass. Exceptions were observed for BC-CG 50 and CG 15, which resulted in lower above/below-ground mass ratios, and for BC-CG 100, which exhibited the highest quantity of above-ground biomass and above/below-ground ratios.

6.3. Plant Development

Despite the less-than-satisfactory results from the seedling development experiment, all compost-based substrates exhibited favourable agronomic performances when plants were transplanted into compost, and compost-biochar blended substrates after 36 days of growth on the peat-based substrate, ensuring an impressive overall survival rate of 100 % (Table 6.4). This high survival rate is a testament to the effectiveness of the compost and biochar substrates, providing reassurance about their agronomic potential. Plants develop salinity tolerance during ontogeny, which may explain the varied results observed in seedlings and plant development trials. Notably, tomato plants are more susceptible to salinity stress during germination and initial seedling growth stages (Sánchez-Monedero et al. 2004).

All treatments demonstrated superiority over the 63-day test period by surpassing the control regarding heights, except for CG 100. BC-CG 100 and BC-CG 50, in particular, recorded the highest peaks at 88.3 cm and 80.7 cm, respectively (Figure 6.2). These findings, which are in line with those of Kammann et al. (2015), who observed a three-fold increase in *Chenopodium* *quinoa* growth in co-composted BC growing media compared to the peat-based control, are truly impressive and underscore the positive effect of nutrient loading of biochar during composting.

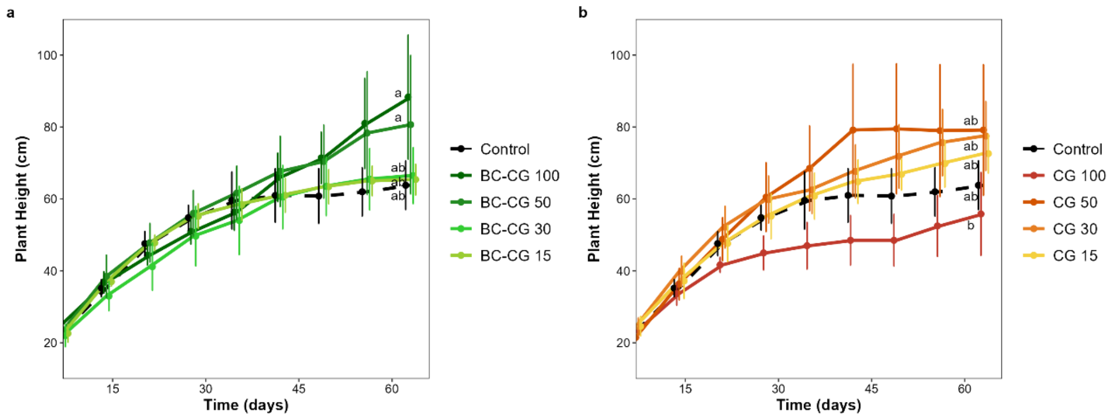


Figure 6.2: Mean (solid lines) ± standard deviations (shadows) of plants height in (a) BC-CG and (b) CG substrates. Different letters indicate significant differences among treatments at $p < 0.05$.

Both compost and biochar serve as slow-release sources of nutrients for plants during distinct phenological phases (Lazcano et al. 2009; Rombel, Krasucka, and Oleszczuk 2022), with a particular richness in potassium (K) (Atzori et al. 2021). The high concentration of K in soluble and exchangeable forms near the root zone enhances its efficiency of use by plants, positively impacting plant development by promoting a higher photosynthetic rate, leaf expansion, plant growth, and biomass accumulation (Atzori et al. 2021; Rombel, Krasucka, and Oleszczuk 2022). This multifaceted role of compost and biochar as slow-release nutrient sources and enhancers of plant development is crucial information for researchers and professionals in agriculture and plant development.

However, the factors influencing plant growth under compost and biochar application are synergistic and not attributable to a single mechanism. Improved plant growth in the compost-biochar blended substrate may be attributed to better physical properties (e.g., water holding capacity, texture) or microbiological properties (e.g., microbial biomass quantity, distribution, and diversity) (Jindo et al. 2020; Yang et al. 2021)

As reported in Table 6.4, the increased height observed in plants grown in co-composted biochar substrates did not correspond to an increase in belowground biomass. Plants grown in a highly nutrient-rich environment typically develop lower root mass but higher leaf and stem mass fractions, promoting light interception and photosynthesis (Yan et al. 2019). Moreover, Grafmüller et al. (2022) suggested that adding nutrient-enhanced biochar in the root area may act as a hotspot for the plant, requiring fewer fine roots for nutrient supply.

Table 6.4: Mean values ± standard deviations of survival rates (%), below-ground biomass (g), and dry/fresh below-ground ratio (%) of plants cultivated in BC-CG and CG substrates. Different letters indicate significant differences among treatments at $P < 0.05$.

	Control	*BC-CG 100*	*BC-CG 50*	*BC-CG 30*	*BC-CG 15*	*CG 100*	*CG 50*	*CG 30*	*CG 15*
Survival rates (%)	100	100	100	100	100	100	100	100	100
Below-ground biomass (g)	38 ± 5.0 ^{b}	26 ± 6.0 ^{b,c}	14 ± 5.3 ^{c,d}	38 ± 8.7 ^{b}	54 ± 9.7 ^{a}	12 ± 6.8 ^{d}	30 ± 5.9 ^{b}	38 ± 6.4 ^{b}	52 ± 2.8 ^{a}
Dry/Fresh below-ground ratio	12 ± 1.6 ^{b}	11 ± 2.3 ^{b}	22 ± 7.9 ^{a}	11 ± 1.7 ^{b}	10 ± 0.6 ^{b}	20 ± 4.3 ^{a}	13 ± 1.7 ^{b}	11 ± 1.6 ^{b}	12 ± 0.9 ^{b}

Below-ground biomass showed an inverse correlation with compost dilution. Lazcano et al. (2009) also observed the most significant increase in root volume with compost dilution varying between 10 % and 20 % in compartmentalized grown tomato plants. Incorporating compost into growing media at up to 30 % has been documented to enhance total pore space and organic matter content, thereby fostering root development (Zawadzińska et al. 2022). Furthermore, compost nutrients and phytohormones stimulate root tip proliferation, enhancing root volume (Kumar et al. 2022). Similar to the effects seen with compost addition, the utilization of biochar in growing media augments root length and the presence of root air, resulting in an expansion of the root surface area and, consequently, an enhanced capacity for water and nutrient absorption (Xiang et al. 2017). Additionally, biochar may trigger the synthesis of indole-3-acetic acid (IAA) (Farhangi-Abriz and Torabian 2018), the primary auxin in plants responsible for regulating the elongation of the primary root and the formation of lateral and adventitious roots (Guardiola 1996).

6.4. Macronutrient Contents in Plant Biomass and Growing Media

Tomatoes cultivated in compost and biochar-blended compost showed higher accumulation of nitrogen (N), phosphorus (P), and potassium (K) compared to those grown in control peat substrates, except for P content in the aboveground biomass of fruiting plants grown in BC-CG 50, and K contents in the below-ground biomass in most treatments of the seedlings experiment (Table 6.5). The biomass N content was directly proportional to the dilution rates for all treatments in both experiments. Seedlings exhibited higher N content in roots than in leaves, while plants had similar concentrations in above- and below-ground biomass. However, plants grown in biochar-blended substrates showed lower N content than those in corresponding compost dilutions. No N limitations were detected, as the initial and final N content remained consistent in all treatments (Table 6.6) although the final N available forms were not measured. P content was higher in above than below-ground biomass. However, this trend was less pronounced for seedlings/plants grown in biochar-blended substrates, which accumulated less P than compost ones (Table 6.4). In the plant development experiment, growing media P content decreased for all treatments, although the values were still higher than those for the control substrate (Table 6.6). Despite similar K content in seedlings' above- and below-ground biomass in the control media, all treatments in both experiments, including plants in the control substrate, showed higher K contents in the aboveground biomass (Table 6.5). The lower macronutrient values in seedlings and plants grown in biochar-blended treatments compared to compost may be attributed to higher cationic retention by the biochar matrix (Hagemann et al. 2017). The final substrate K content was higher than the control for all treatments except BC-CG 15 and CG 15, indicating that the growing media's macronutrient content did not limit plant development (Table 6.6). The lower seedling development in these substrates may be due to excess cations and/or salinity, as reported in other studies (Nocentini et al. 2021; Nieto et al. 2016), rather than macronutrient limitation

Table 6.5: Macronutrients contents in aboveground/belowground biomass. Different letters within the same experiment indicate significant differences among treatments at p <0.05.

Experiment		***N***		***P***		***K***	
		(g kg^{-1})		**(g kg^{-1})**		**(g kg^{-1})**	
	Biomass	**Above**	**Below**	**Above**	**Below**	**Above**	**Below**
Seedlings	**Control**	8.0 ± 0.1 ^{e}	10.0± 0.7 ^{e}	4.1	3.8	19.5	21.3
	BC-CG 100	18.8 ± 0.2 ^{a}	19.2 ± 1.02 $^{b, c}$	6.9	2.8	41.3	14.6
	CG- BC 50	13.4 ± 0.2 ^{c}	19.4 ± 0.4 $^{a, b, c}$	8.9	8.8	33.9	23.3
	BC-CG 30	14.9 ± 0.5 ^{b}	20.9± 0.6 ^{a}	13.6	7.7	18.4	30.5
	BC-CG 15	11.6 ± 0.0 ^{e}	17.4 ± 0.1 ^{d}	9.2	10.0	27.5	18.4
	CG 100	-	-	-	-	-	-
	CG 50	12.2 ± 0.3 $^{d, e}$	21.0 ± 0.4 ^{a}	9.4	7.7	35.4	25.7
	CG 30	12.4 ± 0.1 ^{d}	20.4 ± 0.3 $^{a, b}$	9.0	10.2	29.0	19.9
	CG 15	11.9 ± 0.0 $^{d, e}$	18.01 ± 0.4 $^{c, d}$	9.5	8.0	26.7	10.4
Plants	**Control**	11.4 ± 0.9 $^{c, d}$	11.0 ± 0.4 ^{c}	3.8 ± 0.5 $^{c, d}$	2.1 ± 0.2 ^{d}	12.4 ± 1.3 ^{f}	4.4 ± 0.7 ^{f}
	BC-CG 100	15.0 ± 1.7 ^{b}	14.8 ± 1.4 ^{b}	4.4 ± 0.6 $^{b, c}$	2.9 ± 0.4 $^{b, c}$	33.0 ± 3 $^{a, b}$	24.1 ± 8.1 $^{a, b}$
	CG- BC 50	12.8 ± 1.4 $^{b, c, d}$	11.7 ± 0.7 ^{c}	3.2 ± 0.2 ^{d}	2.2 ± 0.2 ^{d}	25.0 ± 3 $^{c, d}$	15.5 ± 1.3 $^{c, d}$
	BC-CG 30	11.5 ± 0.5 ^{d}	11.2 ± 1.3 ^{c}	4.5 ± 0.6 $^{b, c}$	3.4 ± 0.2 ^{b}	23.7 ± 2.3 $^{c, d}$	13.2 ± 2.7 $^{d, e}$
	BC-CG 15	11.0 ± 1.2 ^{d}	11.03 ± 0.7 ^{c}	4.5 ± 0.7 $^{b, c}$	3.1 ± 0.2 $^{b, c}$	18.5 ± 4.03 $^{d, e, f}$	8.8± 1.1 $^{d, e, f}$
	CG 100	22.3 ± 0.4 ^{a}	22.4 ± 1.8 ^{a}	8.3 ± 0.8 ^{a}	4.2 ± 0.6 ^{a}	36.4 ± 4.8 ^{a}	31.1 ± 6 ^{a}
	CG 50	14.4± 3.3 $^{b, c}$	14.2 ± 1.5 ^{b}	5.3 ± 0.8 ^{b}	2.5 ± 0.7 $^{c, d}$	26.8 ± 5.8 $^{b, c}$	20.8 ± 1 $^{b, c}$
	CG 30	12.8 ± 1.1 $^{b, c, d}$	12.5 ± 1.4 $^{b, c}$	5.0 ± 0.3 $^{b, c}$	2.9 ± 0.2 $^{b, c}$	23.3 ± 2.7 $^{c, d, e}$	10.7 ± 2.9 $^{d, e, f}$
	CG 15	11.0 ± 1.3 ^{d}	10.7 ± 0.7 ^{c}	4.5 ± 0.7 $^{b, c}$	3.3 ± 0.1 $^{b, c}$	16.3 ± 3 ^{f}	7.4 ± 0.7 $^{e, f}$

Table 6.6: Macronutrient contents in growing media at the beginning and end of seedling and fruiting experiments. Different letters within experiments indicate significant differences among treatments at $p < 0.05$.

Experiment		***N***		***P***		***K***	
		(g kg^{-1})		**(g kg^{-1})**		**(g kg^{-1})**	
	Sampling	**Initial**	**Final**	**Initial**	**Final**	**Initial**	**Final**
Seedlings	**Control**	12 ± 0.1^{d}	17.9 ± 0.5^{c}	0.4	0.3 ± 0.0^{e}	3.3	0.9 ± 0.3^{f}
	BC-CG 100	$22.5 \pm 0.2^{b, c}$	$24.3 \pm 3.7^{a, b}$	1.6	1.5 ± 0.0^{a}	7.2	6.7 ± 0.5^{a}
	CG- BC 50	$16.8 \pm 0.3^{c, d}$	19.5 ± 1.5^{c}	1.0	1.1 ± 0.1^{b}	8.4	$4.1 \pm 0.5^{b, c}$
	BC-CG 30	$20.9 \pm 1.4^{b, c}$	19.8 ± 0.8^{c}	0.9	0.8 ± 0.0^{c}	3.5	$3.5 \pm 0.3^{c, d}$
	BC-CG 15	$20.1 \pm 0.4^{b, c}$	18.7 ± 0.8^{c}	0.7	0.7 ± 0.1^{c}	2.1	2.5 ± 0.2^{e}
	CG 100	36.9 ± 7.3^{a}	-	2.7	-	11.0	-
	CG 50	27.3 ± 0.7^{b}	29.02 ± 3.5^{a}	1.7	1.0 ± 0.0^{b}	12.0	4.4 ± 0.3^{b}
	CG 30	19.0 ± 0.6^{c}	27.4 ± 2.5^{a}	1.2	0.8 ± 0.1^{c}	8.2	3.2 ± 0.3^{c}
	CG 15	$20.7 \pm 0.9^{b, c}$	28.3 ± 0.9^{a}	0.7	0.6 ± 0.0^{d}	2.1	2.0 ± 0.3^{d}
Plants	**Control**	12 ± 0.1^{d}	16.6 ± 0.5^{e}	0.4	0.3 ± 0.0^{f}	3.3	1.54 ± 0.6^{e}
	BC-CG 100	$22.5 \pm 0.2^{b, c}$	$21.5 \pm 4.1^{a, b}$	1.6	1.4 ± 0.2^{b}	7.2	$4.5 \pm 1^{b, c}$
	CG- BC 50	$16.8 \pm 0.3^{c, d}$	$15.0 \pm 0.8^{d, e}$	1.0	$0.7 \pm 0.1^{c, d}$	8.4	7.7 ± 0.8^{a}
	BC-CG 30	$20.9 \pm 1.4^{b, c}$	$20.1 \pm 1.1^{b, c}$	0.9	$0.5 \pm 0.05^{e, f}$	3.5	$2.8 \pm 1.0^{c, d, e}$
	BC-CG 15	$20.1 \pm 0.4^{b, c}$	18.8 ± 0.4^{e}	0.7	$0.4 \pm 0.0^{e, f}$	2.1	$1.07 \pm 0.1^{d, e}$
	CG 100	36.9 ± 7.3^{a}	34.0 ± 8.3^{a}	2.7	2.1 ± 0.4^{a}	11.0	5.5 ± 2.3^{b}
	CG 50	27.3 ± 0.7^{b}	24.0 ± 2.0^{b}	1.7	1.0 ± 0.1^{c}	12.0	$3.1 \pm 0.4^{c, d}$
	CG 30	19.0 ± 0.6^{c}	$19.8 \pm 2.6^{b, c, d}$	1.2	$0.7 \pm 0.1^{d, e}$	8.2	$2.8 \pm 0.7^{c, d, e}$
	CG 15	$20.7 \pm 0.9^{b, c}$	$20.8 \pm 1.1^{b, c, d}$	0.7	$0.4 \pm 0.0^{e, f}$	2.1	1.0 ± 0.4^{e}

6.5. Fruit Production

No significant differences were found in the number of tomatoes per plant among the examined substrates (Figure 6.3a). Regarding fresh fruit biomass, BC-CG 100 and CG 100 lowered production by 47.7 % and 39.3 %, respectively, compared to the control (Figure 6.3b). The decrease in fruit biomass may be justified by the imbalance between plant vegetative and reproductive growth under overfertilization conditions, favouring canopy growth over fruit production (Vaccari et al. 2015). Conversely, plants grown in 50 % and 15 % diluted substrates significantly increased tomato biomass. CG 50 resulted in the highest productivity with a 100.3 % increase compared to the control; BC-CG 50 and BC-CG 30 produced 60.8 % and 47.1 % more fresh fruit biomass than the control, followed by 32.24 % and 30.75 % increases in CG 30 and CG 15, respectively. Huang et al. (2019) demonstrated that tomato plants grown in BC-compost mixtures produced more fresh and dry fruit biomass, attributing this to a synergistic effect where compost increased nutrient availability while biochar provided high nutrient-retention capacity. Similarly, Zawadzińska et al. (2022) attributed higher fresh tomato weights in compost-growing media to greater nutrient availability and uptake. Although the effect of biochar on growing substrate properties and plant growth has been broadly studied (Jindo et al. 2020; Huang and Gu 2019), little is known about its effects on horticultural fruit production. Massa et al. (2019) reported that BC stimulated tomato plant growth but not fruit yield, suggesting that replacing peat with biochar-based growing media improves plant biomass rather than crop yield. However, our study highlights the positive results of combining biochar with compost, offering a promising strategy to enhance growth and production in a soilless agricultural system.

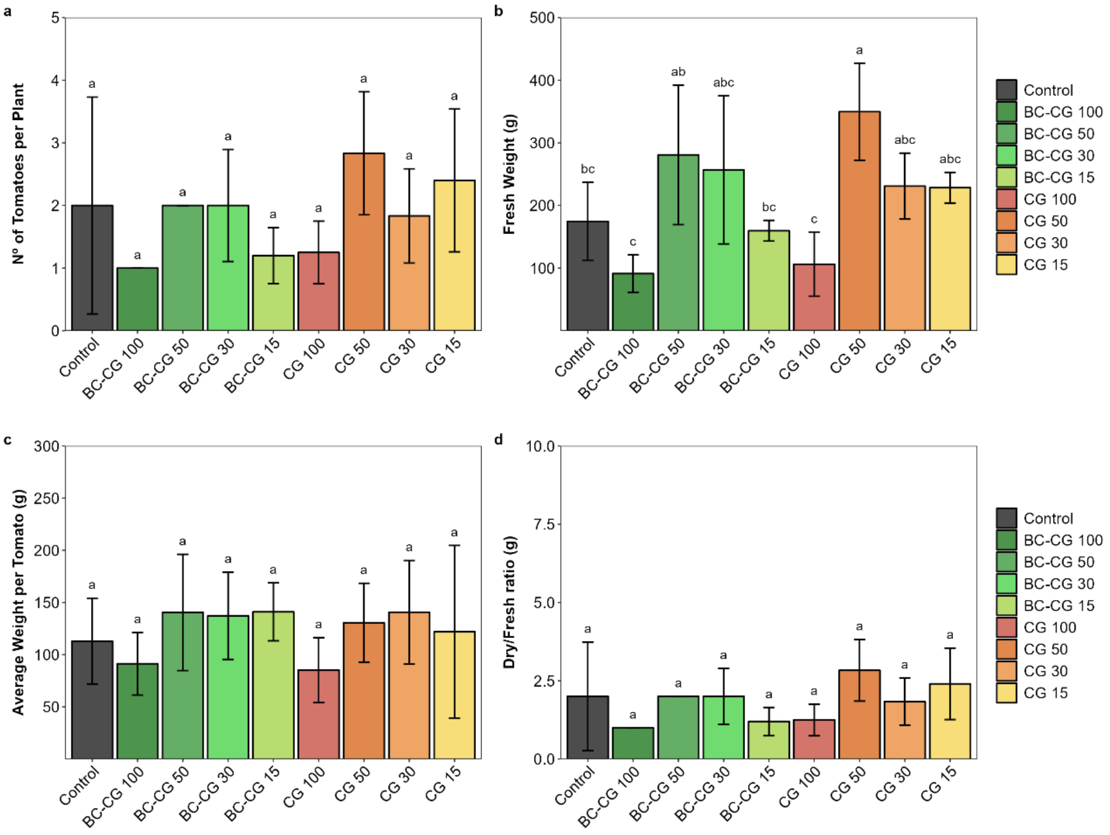


Figure 6.3: Mean values ± standard deviations of the:(a) Number of tomatoes per plant, (b) Fresh biomass production (g), (c) Average weight per tomato (g), (d) Dry/fresh weight ratio (%). Different letters indicate significant differences among treatments at $p < 0.05$

PART III

MadreenRoof a Productive Green Roof in the Centre of Madrid

Chapter 7

A view on MadreenRoof (Giuseppe Picca®)

MadreenRoof

7. MadreenRoof

The current chapter aimed to investigate the agronomic performance of six different substrates made from organic wastes (spent coffee grounds, coffee silverskin, and seaweeds) composted with and without biochar as a peat replacement for rooftop agriculture. The fertility, crop production, and yield quality of these substrates were evaluated over a three-year crop rotation involving a tomato landrace of the Community of Madrid (*Solanum lycopersicum* L., cv. Moruno de Aranjuez) and a consociation of lettuce (*Lactuca sativa* L., cv. Romana) and chard (*Beta vulgaris* var. Cicla).

7.1. Physical and chemical characteristics of the substrates

All tested substrates exhibited a slightly higher pH than the control, but remained within the recommended pH range of 5.5-7.5 for the cultivated species (Table 7.1.1) (Illera-Vives et al. 2015). Peat effectively neutralizes compost alkalinity, thereby increasing the potential bioavailability of phosphorus (P) and micronutrient (Carlile, Cattivello, and Zaccheo 2015). The slight increase observed in the subsequent cycles may be linked to the progressive mineralization of acidic compounds, such as carboxylic and phenolic groups, as well as the degradation of amino acids and peptides within the organic matter, resulting in the release of ammonium ions (Madejón et al. 2021).

Overall, electrical conductivity (EC) met the salinity tolerance for the tested plants (2.0 - 2.5 dS m^{-1}) (Machado and Serralheiro 2017). However, the general EC decrease over time could be related to the depletion of available nutrients from plant uptake or leaching (Figure 7.1). CS and SW showed the most considerable reductions (-1.55 and -1.21 dS m^{-1}, respectively), whereas the presence of biochar buffered the EC at the beginning of the experiment, probably due to the sorption of cations on the reactive surfaces within the pore network (Kammann et al. 2015). The EC range is a critical property for composted materials used as growing media, since elevated salinity in the rhizosphere, induced by fresh compost, can induce significant osmotic stress and lead to unbalanced ion adsorption, ultimately inhibiting vegetable growth and reducing yield (Machado and Serralheiro 2017).

Throughout the entire crop cycle, the composted media maintained a bulk density (BD) within the recommended level of less than 0.5 g cm^{-3} for horticultural substrates (Nieto et al. 2016), with the exception of SW at t24. However, the increase in BD observed throughout the experiment indicates potential compaction and decreased substrate porosity. The presence of biochar counteracted these effects, confirming its potential as a structuring agent and enhancing substrate aeration. The lowering effect of biochar on BD enables the installation of growing media with greater depth in RA and longer lifetime, without posing structural problems and expanding the range of cultivable plant species (Cao et al. 2014).

The addition of biochar during composting resulted in an average total C increase of up to 36 % compared to the compost-based substrates without biochar (Table 7.1.2). Furthermore, at the end of the experiment, BC-CG, BC-CS, and BC-SW treatments respectively exhibited 38 %, 44 % and 8 % higher C concentrations than the peat control. This behaviour was probably driven by the combination of the highly aromatic and persistent C-rich compounds constituting the biochar material and by the chemisorption of dissolved organic carbon on its surface functional groups. These biochar properties could enhance the stability of the substrate, reducing the biochemical mineralization of the organic matter (Sánchez-Monedero et al. 2021).

Similarly, substrates with biochar maintained more stable nitrogen (N_{tot}) levels than those without. At t24, SW showed the most significant decrease, losing 28% of its initial value. Notably, biochar diminishes N_{tot} losses in compost by the chemical-physical sorption of mineral and organic N compounds and by enhancing nitrification (Sánchez-Monedero et al. 2021). Likewise, Laird et al. (2010) found that the addition of biochar could be an effective tool to improve N use efficiency in agriculture, reducing N losses through leaching and denitrification.

The C/N ratio is considered a reliable index of the maturity and stability of organic matter. Throughout the cropping cycles, BC-composted substrates consistently maintained a C:N ratio within the recommended range (20-40) (Agarwal, Saha, and Hariprasad 2023), comparable to the control. Therefore, the values slightly below the optimal range exhibited by substrates without biochar could imply a more rapid organic matter mineralization with prompt releases of nitrates compared to biochar-blended substrates. This trend could favour the availability nitrogen for plants while simultaneously raising the risk of nitrogen losses through runoff due to rainfall or irrigation water.

Table 7.1.1: Main physical and chemical characteristic of the growing media at the begging of each crop cycle (mean values ± standard deviations, N = 5). Different letters indicate significant differences among treatments at a $p < 0.05$ level.

	Time	**Peat**	**BC-CG**	**CG**	**BC-CS**	**CS**	**BC-SW**	**SW**
pH	0	5.95 ± 0.14 ^{b}	6.89 ± 0.02 ^{a}	6.22 ± 0.15 ^{b}	6.86 ± 0.03 ^{a}	6.68 ± 0.56 ^{a}	7.10 ± 0.04 ^{a}	7.05 ± 0.10 ^{a}
	12	6.54 ± 0.22 ^{d}	7.04 ± 0.04 $^{b, c}$	6.41 ± 0.10 ^{d}	7.27 ± 0.13 ^{b}	6.92 ± 0.12 ^{c}	7.57 ± 0.07 ^{a}	7.68 ± 0.07 ^{a}
	24	6.55 ± 0.16 ^{d}	7.04 ± 0.15 ^{c}	6.64 ± 0.06 ^{d}	7.70 ± 0.15 ^{a}	7.38 ± 0.07 ^{b}	7.68 ± 0.05 ^{a}	7.85 ± 0.10 ^{a}
EC (dS m^{-1})	0	0.58 ± 0.34 ^{c}	0.57 ± 0.09 ^{c}	0.91 ± 0.50 $^{b, c}$	1.08 ± 0.12 $^{b, c}$	1.93 ± 0.35 ^{a}	0.96 ± 0.16 $^{b, c}$	1.44 ± 0.2 $^{a, b}$
	12	0.15 ± 0.04 ^{d}	0.25 ± 0.05 $^{c, d}$	0.42 ± 0.07 $^{a, b}$	0.28 ± 0.11 $^{c, d}$	0.55 ± 0.09 ^{a}	0.27 ± 0.03 $^{c, d}$	0.30 ± 0.06 $^{b, c}$
	24	0.08 ± 0.01 ^{d}	0.25 ± 0.04 $^{b, c}$	0.18 ± 0.03 ^{c}	0.28 ± 0.06 ^{b}	0.38 ± 0.07 ^{a}	0.21 ± 0.04 $^{b, c}$	0.23 ± 0.04 $^{b, c}$
Bulk Density (g cm^{-3})	0	0.24 ± 0.01 ^{b}	0.27 ± 0.01 ^{b}	0.26 ± 0.01 ^{b}	0.24 ± 0.01 ^{b}	0.26 ± 0.01 ^{b}	0.32 ± 0.003 ^{a}	0.34 ± 0.004 ^{a}
	12	0.29 ± 0.02 $^{c, d}$	0.31 ± 0.01 $^{c, d}$	0.33 ± 0.01 ^{c}	0.31 ± 0.02 $^{c, d}$	0.32 ± 0.02 ^{c}	0.36 ± 0.01 ^{b}	0.43 ± 0.02 ^{a}
	24	0.36 ± 0.01 $^{c, d}$	0.36 ± 0.02 $^{c, d}$	0.37 ± 0.02 ^{c}	0.33 ± 0.01 ^{d}	0.37 ± 0.01 ^{c}	0.42 ± 0.02 ^{b}	0.53 ± 0.02 ^{a}

Table 7.1.2: Main physical and chemical characteristic of the growing media at the begging of each crop cycle (mean values ± standard deviations, N = 5). Different letters indicate significant differences among treatments at a $p < 0.05$ level.

	Time	**Peat**	**BC-CG**	**CG**	**BC-CS**	**CS**	**BC-SW**	**SW**
C_{tot} (g kg^{-1})	0	359 ± 9.49 ^{c}	475 ± 13.7 ^{a}	409 ± 11.7 $^{b, c}$	504 ± 46.7 ^{a}	414 ± 21.9 ^{b}	370 ± 35.2 $^{b, c}$	272 ± 22.3 ^{d}
	12	374 ± 21.8 ^{c}	499 ± 16.8 ^{a}	396 ± 6.28 $^{b, c}$	513 ± 22.0 ^{a}	415 ± 5.89 ^{b}	404 ± 29.8 $^{b, c}$	292 ± 14.4 ^{d}
	24	333 ± 18.4 ^{b}	458 ± 14.2 ^{a}	374 ± 30.6 ^{b}	480 ± 32.2 ^{a}	384 ± 32.4 ^{b}	361 ± 47.0 ^{b}	214 ± 18.7 ^{c}
N_{tot} (g kg^{-1})	0	11.7 ± 0.41 ^{d}	17.5 ± 0.67 ^{b}	20.7 ± 1.97 ^{a}	14.1 ± 0.59 ^{c}	20.9 ± 1.57 ^{a}	10.8 ± 0.52 ^{d}	14.4 ± 0.96 ^{c}
	12	12.0 ± 0.37 ^{d}	17.5 ± 0.94 ^{b}	21.5 ± 0.96 ^{a}	14.1 ± 1.37 ^{c}	19.7 ± 1.32 ^{a}	12.1 ± 0.57 ^{d}	13.6 ± 0.52 $^{c, d}$
	24	11.5 ± 0.50 ^{d}	17.8 ± 1.44 ^{b}	20.6 ± 1.61 ^{a}	14.5 ± 0.81 ^{c}	19.9 ± 1.82 $^{a, b}$	9.9 ± 0.71 ^{d}	10.4 ± 0.91 ^{d}
C/N	0	30.6 ± 1.79 $^{b, c}$	27.2 ± 1.11 ^{c}	19.9 ± 1.40 ^{d}	35.9 ± 3.16 ^{a}	19.9 ± 0.84 ^{d}	34.4 ± 2.62 $^{a, b}$	18.9 ± 0.54 ^{d}
	12	31.2 ± 1.65 $^{b, c}$	28.6 ± 1.10 ^{c}	18.4 ± 0.92 ^{d}	36.7 ± 3.33 ^{a}	21.2 ± 1.73 ^{d}	33.5 ± 3.36 $^{a, b}$	21.5 ± 0.99 ^{d}
	24	28.9 ± 0.38 ^{b}	25.9 ± 1.83 ^{b}	18.2 ± 0.94 ^{c}	33.1 ± 1.71 ^{a}	19.3 ± 0.80 ^{c}	36.3 ± 3.43 ^{a}	20.5 ± 0.21 ^{c}

7.2. Phytoavailable Macro- and Micronutrients

Compost-based substrates consistently exhibited nutrient levels within or above the recommended range for tomato cultivation (Figure 7.1) (Sainju, Dris, and Singh 2003).

Notably, N_{min} levels showed significant variability across treatments over time (Figure 7.1a). At t0, BC-CG was the only growing media with N_{min} within or close to the optimal range (50–100 mg kg^{-1}). In contrast, the other substrates, including the control, exceeded this range, with SW having values more than ten times higher, posing a risk of N_{min} leaching.

Over the 24 months, N_{min} levels generally diminished across all substrates approaching the optimal range, with a steep reduction from t0 to t12, particularly for peat-based control. Notably, substrates based on CS and CG sustained high N_{min} levels throughout, confirming that composts from coffee biowastes provide a stable, long-lasting nitrogen source for horticultural use (Picca et al., 2022, 2023). Conversely, SW and the peat-based control and showed the greatest decline, with N_{min} levels dropping by 89 % and 88 % from t0 to t24 , respectively. The reduction indicates a trend toward substrate exhaustion, even if N_{min} was still within the optimal range, which could hamper plant growth and increase the risk of N_{min} leaching into drainage water. Generally, the correlation matrix (Figure A.1) highlighted a strong link between N_{min} concentration and crop productivity. This observation is expected, since adequate N_{min} supports plant growth, enhances foliage production, but also protects the fruit from intense sunlight stress (Sainju, Dris, and Singh 2003), which is a crucial for thriving in the challenging conditions of a green roof.

All substrates, except BC-CS, had available P levels within or above the optimal range for tomatoes at t0 (60–70 mg kg^{-1}) (Figure 7.1b). Similarly, compost-based substrates were generally rich in available potassium (K), with concentrations at t0 exceeding the optimal range for tomato cultivation (600–700 mg kg^{-1}), specifically up to four times higher for CS due to the particularly high K level of the feedstock (Figure 7.1c) (Picca et al. 2022). In contrast, the available K content of the peat-based control was below the recommended value, implying potential deficiencies for tomato growth. Over the 24 months, a general decline in available P and K levels across different treatments was observed, with values nearing or falling below the optimal range. Despite significant drops of -67.3 % and -64.5 % from t0 to t24, CG and CS were the only substrates maintaining optimal P levels, demonstrating their capacity to support P availability over time. Similarly, the CS-based substrates and CG recorded satisfactory available K values for tomato production throughout the cycle. However, the progressive nutrient exhaustion recorded in the other substrates suggests that supplemental P and K fertilisation may be necessary to prevent a drop in production when more than three growing cycles are forecasted. As evidenced by the correlation matrix (Figure A.1), there is a strong, positive, and significant correlation between available P and K and crop productivity, highlighting the effect of these two nutrients in stimulating early blossom and fruit set while improving tomato yield (Sainju, Dris, and Singh 2003).

Available Calcium (Ca) concentrations were notably higher than the optimal range (1000 mg kg^{-1}) at the beginning of the experiment, being 5 and 3.5 times higher in SW and CS (Figure 7.1d). Available Ca levels appear to increase for BC-CG, CG, CS, and BC-SW substrates from t0 to t12, probably due to a concentration effect provoked by the mineralisation of the substrate organic matter, followed by a slight decrease from t12 to t24. Despite this event, all the growing media maintained concentrations far exceeding the optimal level by t24. Likewise, available magnesium (Mg) levels in the substrates were within or above the optimal range, with peat control, CS, and SW registering the highest concentrations (Figure 7.1e). Available Mg concentrations followed the same pattern over time as available Ca concentrations, indicating no evidence of deficit for Ca and Mg over the three growing cycles, even though a possible effect of Mg retention within the biochar pore network, due to adsorption or entrapment of the ions could be inferred for the biochar-blended substrates.

Peat-based control presented B concentrations below the optimal range (1.5–2.5 mg kg^{-1}) throughout the cycles, with CG also presenting insufficient values at t24 (Figure 7.1f). Except at t0, CS-based substrates maintained B concentrations within the recommended range, ensuring sufficient B availability for tomato growth. Excessive B values observed in BC-SW and SW were due to the high concentration of this element in seaweed debris, which could cause inhibition of plant development (Mininni et al. 2012). However, tomato plants grown in seaweed-based composts were consistently the tallest among all treatments during the 3-year experiment (data not shown), which suggests that plant growth was no inhibited by a boron concentration excess .

The results indicated that the compost-based substrates were more resilient to exhaustion and maintained adequate macro- and micronutrient concentration longer than the peat-based control, justifying the feedstock selection for growing media in RA. The effectiveness of biochar in preventing fertility depletion varies depending on the specific nutrient and substrate. Its addition proved effective in reducing N_{min} and P loss among the substrates, allowing for sustainable management of these nutrients, although it was less effective in BC-CS for N_{min} reduction. Moreover, while its presence was effective only in BC-CS in limiting K loss, it was useful in preventing Mg reduction in BC-CG and BC-SW and helped prevent Ca loss in BC-SW and B loss in BC-CG. However, the progressive nutrient depletion reported in all the substrates suggests that setting a fertilisation plan over three years would be advisable to mitigate potential plant abiotic stress and ensure sustained growth and productivity.

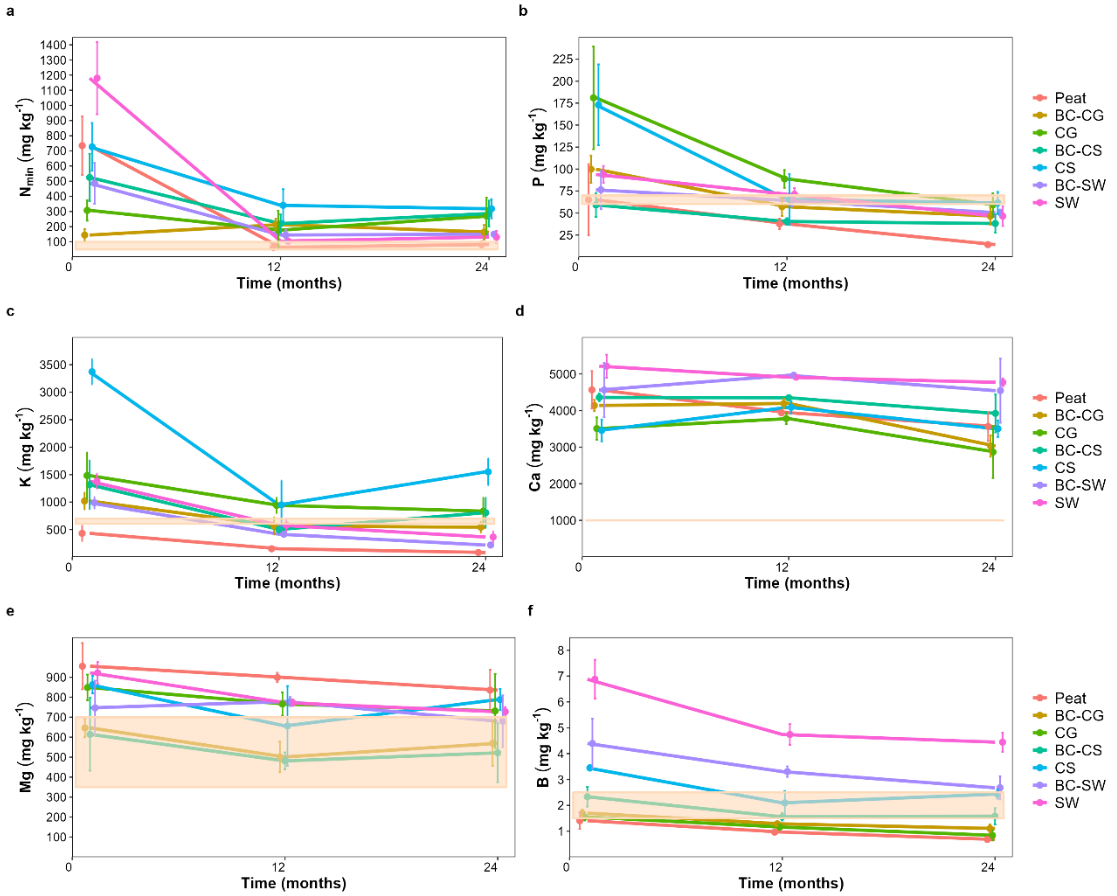


Figure 7.1: Phytoavailable main macro- and micronutrients of the growing media at the beginning of each summer cycle (mean values ± standard deviations, N = 5). The orange-coloured panel indicates the recommended nutrient ranges for tomato cultivation, as suggested by Sainju (2003).

7.3. Crop Yields

7.3.1. Tomatoes

Overall, tomato yields of compost-based substrates were comparable to or higher than those of peat-based and open-field (IMIDRA) controls. In the second and third cycles, a significant yield reduction was observed both in rooftop and in open-field systems as compared to the first cycle. In addition to the depletion of the nutrients observed in the growing media, this trend could be explained the heat waves that impacted European summers in 2022 and 2023. During these crop cycles, the average rooftop temperature rose by

3 °C in 2022 and 1 °C in 2023 compared to 2021. Additionally, the average maximum temperature reached 32 °C in 2022 and 34°C in 2023, increasing by 1°C and 3°C, respectively, compared to 2021. Temperatures above 30 °C cause fatal damage to the reproductive organs of tomato plants, definitively hindering physiological and biochemical activities above 35 °C. The induced heat stress diminishes flower pollination rates, decreases fruit setting, and ultimately lowers production (Alsamir et al. 2021). However, despite the yield setback, compost-based substrates maintained higher productivity than peat-based and IMIDRA controls, indicating greater resilience of the tested substrates to extreme temperatures. BC-CS emerged as the most productive growing media, yielding an average of 9.36 kg of tomatoes per plant during the 3-year study, overall producing 43 % more than open-field during the second year and 9-times more during the third year (Figure 7.2a). Although the cumulative value for the peat-based substrate was affected by plant mortality during the third year, BC-CS production doubled peat-based control yield during the first and second years, respectively.

Despite its lower overall yield, IMIDRA open air control produced the highest number of tomatoes per plant during the first year and the second-highest number in the second year. However, BC-CS presented the highest cumulative value due to the largest number of tomatoes produced during the third year (Figure 7.2b). Adding biochar tended to increase the number of tomatoes produced per plant, although the difference was not significant (Figure 7.2b).

Despite considerable variability, BC-SW showed the highest average weight per tomato during the first year, exceeding the peat-based and open-field controls by 84 % and 56 %, respectively (Figure 7.2c). Conversely, BC-CG exhibited the highest average weight per tomato over the three years, outperforming the peat control and IMIDRA treatment by 52 % and 42 %, respectively (Figure 7.2c).

The significant increase in tomato yield between peat-control and compost-based growing media supports previous research findings (Montesano et al. 2014; Picca et al. 2023). Notably, the use of compost in growing media improves nutrient availability and the substrate's physical-chemical characteristics for plant growth. In agreement with our findings, Di Bonito et al. (2018) reported that using compost alone or mixed with biochar as an amendment in rooftop agriculture improved tomato size, but contrary to our findings they did not find an increase of the total fresh weight. Additionally, Massa et al. (2019) noted that biochar stimulated tomato plant growth at the expense of fruit yield in soilless cultivation. Confirming the trends observed in this study, Gruda (2009) reported that soilless systems can produce higher yields and more uniform tomato weight and size compared to soil-based systems. Our results confirm that higher yields compared to controls are maintained at least during a 3-year period. This underscores the critical role green roofs based on organic substrates can play in bolstering urban food security alongside other production systems (Grard et al. 2020). However, the influence of compost feedstock and biochar addition may vary (Table 7.2) depending on annual variability and other factors such as the climatic conditions. Tomato yields were significantly and positively affected by feedstock type and biochar during the first year and in the cumulative value for the 3-year cropping period (Table 7.2). This implies that both biochar blending and the feedstock material used for the compost could improve tomato productivity, especially during the first year in a scenario of higher nutrient availability, such as high P and K for BC-CS substrate. However, the interaction between the two factors was significant only for the cumulative value (Table 7.2), suggesting that when nutrients are reduced due to plant uptake and/or leaching, the specific combination of biochar and substrate could help maintain adequate fertility levels (Huang et al. 2019).

Table 7.2: Results of ANOVA (F and p-value) showing the main significant factors (i.e. feedstock, biochar addition, and interactions) influencing crops yield along with cumulative values for all seasons

ANOVA results	**Time**	**Feedstock**		**Biochar**		**Feedstock*Biochar**	
		F	***p***	**F**	***p***	**F**	***P***
Tomato Yield	2021	9.94	***	17.0	***	0.25	
	2022	0.01		1.55		1.01	
	2023	1.65		2.40		0.21	
	Overall	136	***	9.30	**	24.8	***
Lettuce Yield	2021/2022	28.2	***	0.56		19.6	***
	2022/2023	2.97		2.27		2.13	
	Overall	8.60	**	0.18		2.13	
Swiss chard Yield	2021/2022	11.9	***	1.10		2.75	
	2023/2024	2.45		1.14		1.60	
	Overall	11.1	***	0.06		3.23	

Significance level: *$p<0.05$; **$p<0.01$, *** $p<0.001$

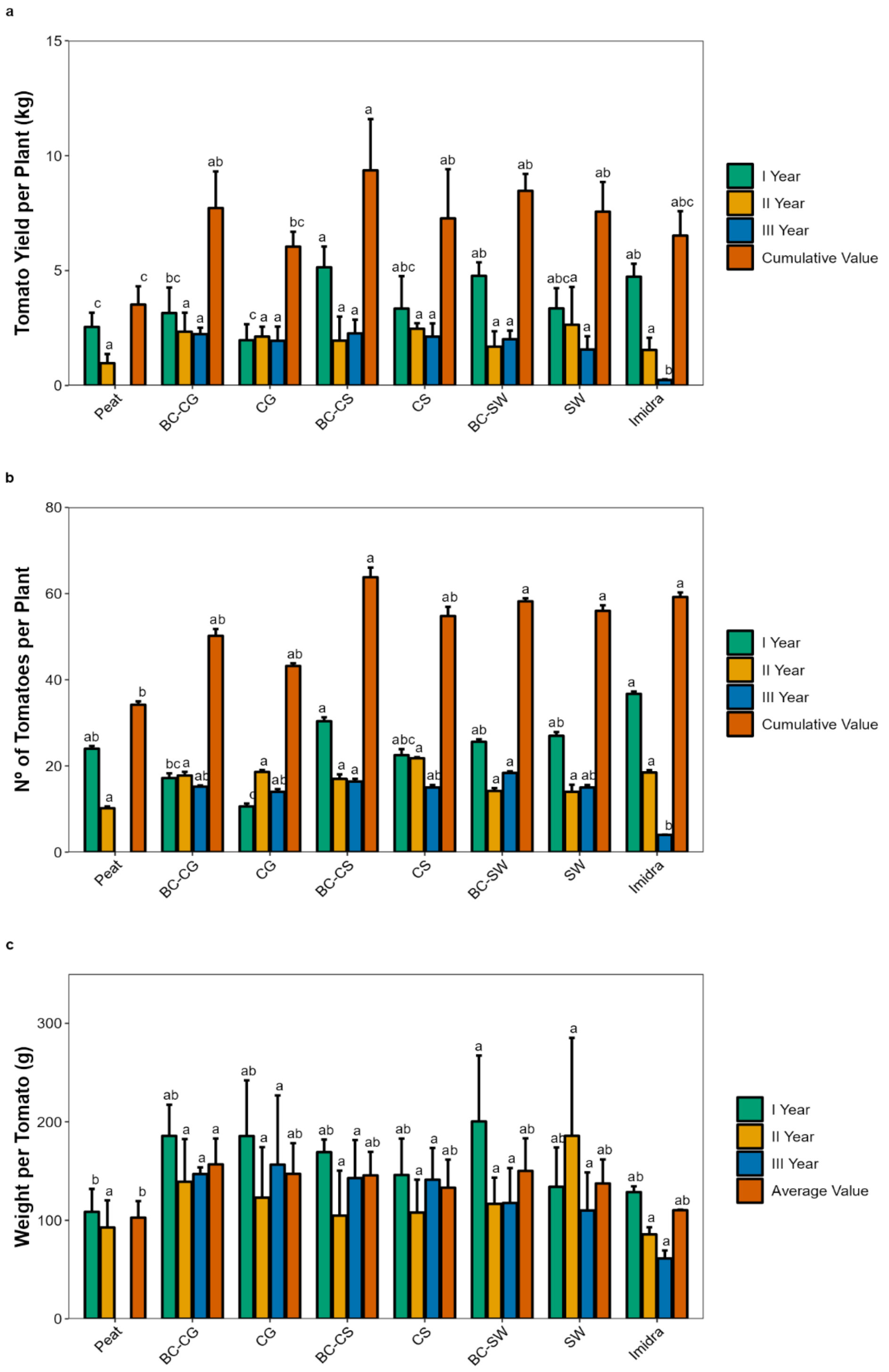


Figure 7.2: (a) Tomato yield expressed as kg of tomato per plant (mean ± SD) along with cumulative values for all seasons, (b) N° of tomatoes per plant (mean ± SD) along with cumulative values for all seasons, and (c) Weight per tomato (mean ± SD) along with average values for all seasons. Data for Peat in the third year are not shown due to pre-harvest plant mortality. Different letters indicate significant differences among treatments at a $p < 0.05$ level.

7.3.2. Winter crops

Similar to tomatoes yields, winter crops also showed higher production rates in the first year, with a decay during the second cycle. Silverskin-based composts exhibited the highest lettuce productivity (Figure 7.3a). CS surpassed the peat control production by 4.8 times in the first year and BC-CS by 1.5 times during the second cycle. Concerning Swiss chard, SW exhibited the highest production throughout the cycle, notably with a biomass two times higher than peat-based substrate in both cycles (Figure 7.3b).
The results obtained for leafy vegetables yields agreed with those reported by Schröder et al. (2021), who noted that using compost substrates in urban farming can lead to lettuce production between 161 and 244 g. Similarly, Marutani and Clemente (2021) indicated that substrates blended with food waste compost could support higher lettuce yield than commercial peat substrates. Mininni et al. (2012) suggested that mixing seaweed-based compost with peat at up to 50 % increases leafy vegetables growth due to the richness in plant hormone compounds, avoiding possible adverse effects related to high B content, normally found in seaweed.

Feedstock material was the main factor affecting lettuce and Swiss chard yields in the first year and for the cumulative value (Table 7.2). Regarding the role of biochar, it positively interacted with feedstock in the first year of lettuce production, potentially increasing nutrient content and improving nitrogen utilization in leafy vegetables, as noted by Pereira, Conz, and Six (2017). This interaction initially led to higher yields despite a subsequent decline in production due to fertility depletion among the substrates observed in the second cycle.

In contrast, biochar addition showed no effect on Swiss chard production. The winter crop results suggest that the effectiveness of biochar as a structural agent strongly depends on the type of substrate and crop. While biochar positively affected the CG-based compost substrate for both crops, it did not improve productivity for CS and SW substrates. The beneficial effects of co-composted biochar still need to be clarified, underscoring the demand for further research in this area. As stated by Nobile, Denier, and Houben (2020) and Méndez et al. (2015), the positive effect of biochar on leafy vegetable harvest is further linked to the increased plant water availability by modulating the hydrophysical properties of the growing media. Similarly, Wang and Akdeniz (2023) noted that even though co-composted biochar did not positively affect lettuce yield, it improved water-holding capacity. However, the beneficial effects of co-composted biochar still need to be clarified, underscoring the demand for further research in this area.

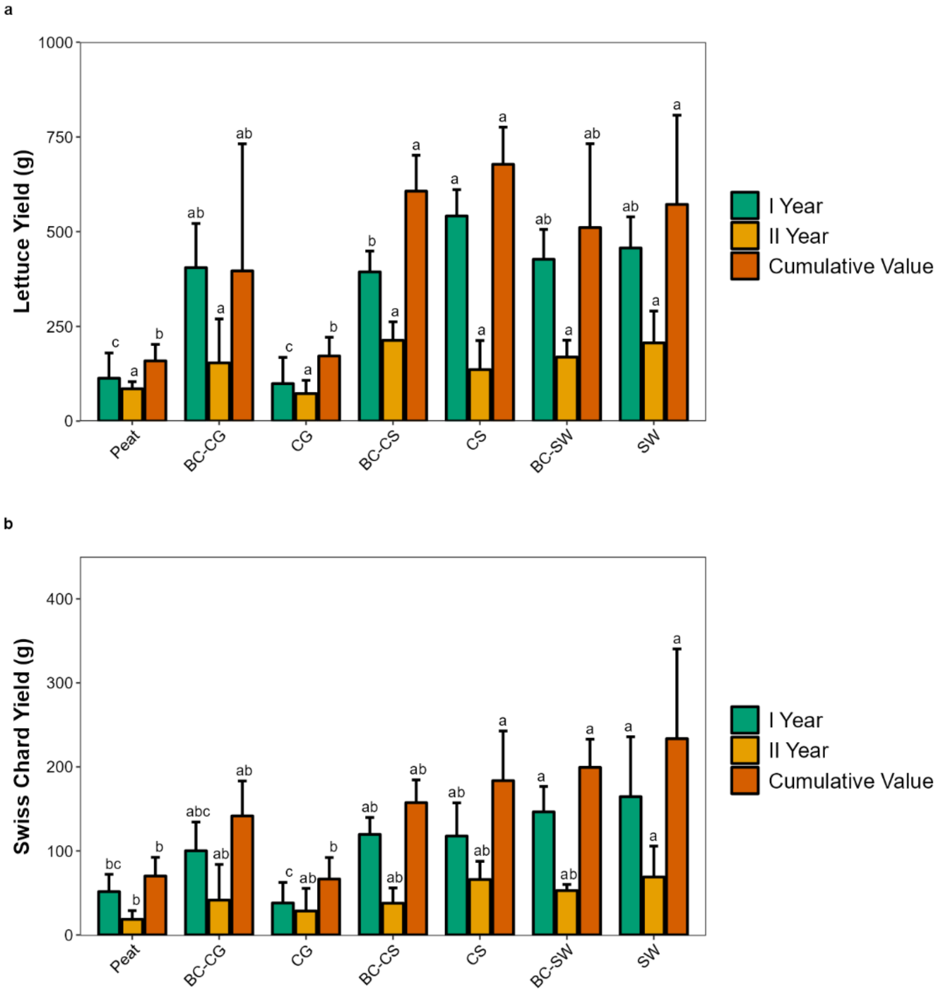


Figure 7.3: (a) Lettuce and (b) Swiss chard yield expressed as kg of fresh weight (mean ± SD), along with cumulative values for all seasons. Different letters indicate significant differences among treatments at a $p < 0.05$ level.

7.4. Tomato quality

Evaluating tomato quality parameters and secondary metabolites (Figure 4) is fundamental to understanding the potential of substrates to sustain the growth of nutrient-rich and marketable fruits. Previous studies have indicated no significant differences in tomato quality between soilless and conventional cultivation systems (Gruda 2009).

During the 2022 harvest, compost-based growing media outperformed the peat control for most variables, demonstrating values comparable to open-field production (Table 7.3). The tomatoes produced generally presented a pH lower than 4.4, displaying a safe environment for preservation from microbial degradation (Anthon, Lestrange, and Barrett 2011) (Tigist, Workneh, and Woldetsadik 2013), thereby extending shelf life and reducing food waste. Regarding total soluble solids (TSS), fruits had values within the optimal range of 3 to 5 Brix° for beefsteak tomatoes (Beckles 2012), with SW having TSS values 22.7 % higher than the peat-based control. These results align with those Montesano et al. (2014) reported, who found similar TSS values (4.9 Brix°) for tomatoes grown in seaweed-based substrates. The SW-based compost also had the highest total acidity (TA) and a sweetness and acidity profile (the TSS:TA ratio) comparable to the IMIDRA and second only to peat control.

Lycopene content, an anticarcinogenic compound and tomato colour determinant (Dorais, Ehret, and Papadopoulos 2008), peaked in BC-SW, exceeding the peat-based and open-field controls by 80 % and 48 %, respectively. Similarly, CG overreached peat-based and open-field controls by 36 % and 8 % in β-carotene concentration, a primary vitamin A precursor in the human diet, emphasizing the potential nutraceutical benefits associated with rooftop agriculture tomatoes(Dorais, Ehret, and Papadopoulos 2008). Moreover, even though for total antioxidant capacity (TAC) no evident difference was reported among the tested substrates and open-soil, BC-SW had the highest levels of total phenols (TP), exceeding the peat-based control by 48 % being second only to open-soil control.

During 2023, BC-CG and SW were the only substrates to approach the suitable level of 12.5 required to produce a savoury table tomato (Beckles 2012), the open-field control exhibited higher quality parameters and secondary metabolite values than compost-based substrates for most of the parameters considered (Table 7.4). However, the lower yield reported in the open-field system may have contributed to this difference, since the ratio of fruit yield to total plant mass plays a crucial role in regulating the metabolite content of the mature fruit pericarp (Schauer et al. 2006). However, the lower tomatoes quality between rooftop and open-field production could have been influenced by the progressive depletion of nutrients.

Both the composting feedstock and the presence of biochar positively influenced the majority of the quality parameters considered in the study for the 2022 season, but feedstocks have a larger effect on the same parameters for the season 2023 if compared to biochar (Table 7.5). In addition, the interaction of biochar and feedstocks influenced the quality parameters especially the secondary metabolites in 2022. This suggests that combining specific feedstocks and biochar can optimize tomato quality more effectively than either factor alone if nutrient availability is guaranteed. In 2023, while the effects of feedstock and biochar on tomato quality parameters were consistent with those observed in 2022, their interaction effects were less pronounced.

Previous studies have reported that the higher electrical conductivity (EC) mediated by compost could enhance the accumulation of major osmotic compounds in tomato fruit, contributing to the increase of TA and TSS (Beckles 2012). Additionally, high EC conditions induce plant stress, stimulating the accumulation of antioxidant compounds (Gruda 2009; Massa et al. 2019). Therefore, the difference between the two years of the study could be explained by the decrease in EC observed during the third year. The enriched nutrient availability in the root zone provided by compost improves the tomato characteristics. Notably, compost is a primary source of K, which positively influences TSS and TA concentration by increasing photosynthetic activity (Yang et al. 2023; Sainju, Dris, and Singh 2003) and the differentiation of chloroplasts into chromoplasts, thereby improving carotenoid accumulation (Massa et al. 2019). On the other hand, the influence of biochar on tomato quality has yet to be thoroughly characterized. Simiele et al. (2022) and Massa et al. (2019) attributed the positive influence of biochar on TSS and TA to increased nutritional availability in the root zone as an indirect consequence of improved physical properties of the substrates. Meanwhile, Petruccelli et al. (2015) reported neither positive nor negative effects on TSS and TA in tomatoes produced in substrates amended with biochar but they reported an increase in TAC driven by the hormesis effect: an exposure to low doses of chemicals derived from biochar, which can be phytotoxic or biocidal at high concentrations, which may stimulate phenolic synthesis.

The results from this study demonstrate that compost-based growing media, particularly when combined with biochar, can positively impact key quality parameters, demonstrating the potential to match field production. However, the substantial variation among the different compost-based media over the years suggests the need for further exploring these interactions to optimize co-composted substrates for RA use.

Table 7.3: Main quality parameters and secondary metabolites of the tomatoes harvest during the summers 2022 (mean values ± standard deviations, N = 3). Different letters indicate significant differences among treatments at a $p < 0.05$ level.

		pH	**Soluble Solid (TSS)**	**Acidity (TA)**	**TSS:TA**	**Lycopene**	**β-Carotene**	**Total Antioxidant Capacity (TAC)**	**Total Phenols (TP)**
			(Brix°)	(% Citric acid)		(mg 100 g^{-1} FW)	(mg 100 g^{-1} FW)	(mmol Fe_2^+ 100 g^{-1} FW)	(mg Gallic Acid 100 g^{-1} FW)
Peat	2022	3.82 ± 0.01 ^{d}	3.83 ± 0.06 ^{e}	0.36 ± 0.03 ^{d}	10.6 ± 0.59 ^{a}	3.16 ± 0.33 $^{c, d}$	1.18 ± 0.04 $^{b, c}$	0.22 ± 0.01 $^{a, b}$	18.0 ± 2.77 ^{b}
BC-CG	2022	4.15 ± 0.02 a	3.63 ± 0.06 f	0.42 ± 0.02 c, d	8.73 ± 0.24 b	2.88 ± 0.04 d	1.12 ± 0.03 c	0.20 ± 0.01 b	22.5 ± 1.35 b
CG	2022	4.07 ± 0.02 a	3.83 ± 0.06 e	0.47 ± 0.01 b, c	8.16 ± 0.09 b, c	4.80 ± 0.52 a, b	1.54 ± 0.17 a	0.24 ± 0.01 a	25.6 ± 0.98 a, b
BC-CS	2022	4.01 ± 0.04 ^{b}	4.07 ± 0.10 $^{c, d}$	0.50 ± 0.01 $^{a, b}$	8.14 ± 0.20 $^{b, c}$	4.51 ± 0.25 $^{a, b, c}$	1.20 ± 0.06 $^{b, c}$	0.22 ± 0.01 $^{a, b}$	23.2 ± 2.84 ^{b}
CS	2022	4.05 ± 0.005 ^{b}	3.97 ± 0.06 $^{d, e}$	0.53 ± 0.04 $^{a, b}$	7.51 ± 0.59 ^{c}	3.45 ± 0.12 $^{b, c, d}$	1.12 ± 0.04 $^{b, c}$	0.24 ± 0.01 ^{a}	23.0 ± 1.45 ^{b}
BC-SW	2022	4.07 ± 0.01^{a}	4.23 ± 0.00 ^{c}	0.52 ± 0.01$^{a, b}$	8.14 ± 0.08 $^{b, c}$	5.68 ± 1.13 ^{a}	1.50 ± 0.21 ^{a}	0.22 ± 0.01 $^{a, b}$	26.7 ± 1.95 $^{a, b}$
SW	2022	4.06 ± 0.02 ^{d}	5.07 ± 0.29 ^{a}	0.55 ± 0.04 ^{a}	9.24 ± 0.66 ^{b}	3.36 ± 0.30 $^{c, d}$	1.42 ± 0.10 $^{a, b}$	0.22 ± 0.01$^{a, b}$	35.6 ± 9.04 ^{a}
IMIDRA	2022	4.06 ± 0.02 ^{d}	5.07 ± 0.29 ^{a}	0.55 ± 0.04 ^{a}	9.24 ± 0.66 ^{b}	3.36 ± 0.30 $^{c, d}$	1.42 ± 0.10 $^{a, b}$	0.22 ± 0.01$^{a, b}$	35.6 ± 9.04 ^{a}

Table 7.4: Main quality parameters and secondary metabolites of the tomatoes harvest during the summers 2023 (mean values ± standard deviations, N = 3). Different letters indicate significant differences among treatments at a $p < 0.05$ level.

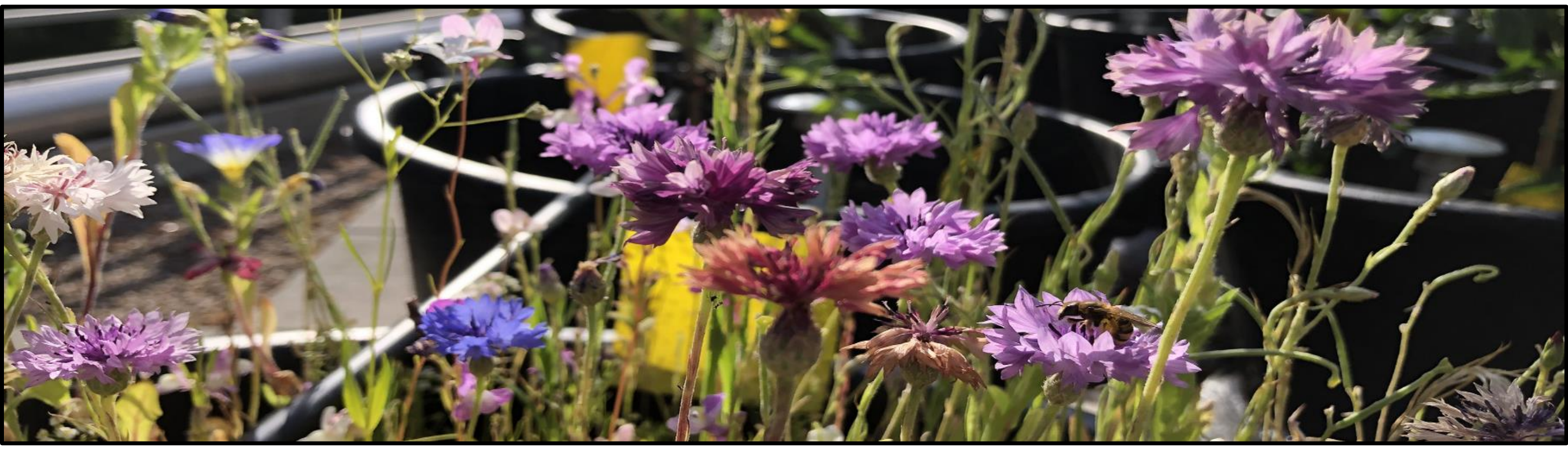
		pH	Soluble Solid (TSS)	Acidity (TA)	TSS:TA	Lycopene	β-Carotene	Total Antioxidant Capacity (TAC)	Total Phenols (TP)
			(Brix°)	(% Citric acid)		(mg 100 g^{-1} FW)	(mg 100 g^{-1} FW)	(mmol Fe_2^+ 100 g^{-1} FW)	(mg Gallic Acid 100 g^{-1} FW)
Peat	2023								
BC-CG	2023	3.99 ± 0.01 ^{a}	3.47 ± 0.06 $^{c, d}$	0.29 ± 0.01 ^{c}	12.1 ± 0.04 ^{b}	2.36 ± 0.13 $^{b, c}$	1.07 ± 0.11 ^{b}	0.19 ± 0.01 ^{c}	14.3 ± 0.50 ^{b}
CG	2023	3.93 ± 0.00 b	3.20 ± 0.06 d	0.29 ± 0.03 c	11.0 ± 0.84b	2.35 ± 0.18 b, c	0.95 ± 0.09 b	0.17 ± 0.01c	12.6 ± 1.23 b
BC-CS	2023	3.94 ± 0.01 ^{c}	3.63 ± 0.06 ^{c}	0.40 ± 0.01 ^{b}	9.09 ± 0.36 ^{c}	2.05 ± 0.32 ^{c}	0.89 ± 0.08 ^{b}	0.22 ± 0.04 $^{b, c}$	13.1 ± 0.87 ^{b}
CS	2023	3.99 ± 0.02 $^{b, c}$	4.30 ± 0.06 ^{b}	0.38 ± 0.01 ^{b}	11.2 ± 0.17 ^{b}	2.77 ± 0.44 $^{a, b, c}$	0.96 ± 0.16 ^{b}	0.22 ± 0.04 $^{b, c}$	14.5 ± 3.45 ^{b}
BC-SW	2023	3.89 ± 0.02 ^{b}	4.17 ± 0.06 ^{b}	0.29 ± 0.01^{c}	14.2 ± 0.25 ^{a}	3.16 ± 0.52 $^{a, b}$	0.99 ± 0.15 ^{b}	0.17 ± 0.03 ^{c}	16.4 ± 0.23 ^{b}
SW	2023	3.81 ± 0.00 $^{b, c}$	4.53 ± 0.10 ^{b}	0.37 ± 0.02 ^{b}	12.1 ± 0.20 ^{b}	2.85 ± 0.18 $^{a, b, c}$	0.78 ± 0.10 ^{b}	0.25 ± 0.01 ^{b}	16.6 ± 1.28 ^{b}
IMIDRA	2023	4.15 ± 0.00 $^{b, c}$	5.43 ± 0.06 ^{a}	0.62 ± 0.02 ^{a}	8.77 ± 0.49 ^{c}	3.31 ± 0.16^{a}	1.42 ± 0.05 ^{a}	0.35 ± 0.02^{a}	40.2 ± 4.86 ^{a}

Table 7.5:Results of ANOVA (F and p-value) showing the main significant factors (i.e. feedstock, biochar addition, and interactions) influencing quality parameters and secondary metabolites of the tomatoes harvest during the summers 2022 and 2023

ANOVA results	**Time**	**Feedstock**		**Biochar**		**Feedstock*Biochar**	
		F	**p**	**F**	**p**	**F**	**p**
pH	2022	33.4	***	19.3	***	24.1	***
	2023	29.8	***	7.14	*	18.4	***
Soluble Solid (TSS)	2022	185	***	36.1	***	27.1	***
	2023	91.8	***	17.1	**	19.8	***
Total Acidity (TA)	2022	38.9	***	19.1	***	0.57	
	2023	56.2	***	8.82	*	13.7	***
TSS:TA	2022	7.98	**	5.07	*	5.17	*
	2023	51.2	***	2.67		28.7	***
Lycopene	2022	14.9	***	0.05	**	14.5	***
	2023	7.21		0.71		3.87	
β-Carotene	2022	3.87		0.05		19.1	***
	2023	1.74		2.33		2.01	
Total Antioxidant Capacity (TAC)	2022	1.91		23.4	***	5.05	*
	2023	6.77	*	4.01		9.81	**
Total Phenols (TP)	2022	0.79	*	0.19		4.49	*
	2023	6.31		0.002		1.34	

Significance level: *p<0.05; **p<0.01, *** p<0.001

Chapter 8



Wild bees visiting MadreenRoof (Giuseppe Picca®)

Conclusions and Future Perspective

8. Conclusions and Future Perspective

This chapter outlines the main conclusions and contributions of this thesis based on the three scientific questions that were asked. Additionally, it suggests potential directions for future research on the covered topics.

8.1. Conclusions

The primary aim of this thesis was to evaluate the potential of co-composting wastes, both with and without biochar, to generate viable substrates for open-air Rooftop Agriculture capable of replacing peat. To achieve this objective, the following scientific questions were addressed:

Question 1 - *Are coffee-derived by-products a suitable feedstock for producing high-quality composts* (Chapter 5)?

The results confirmed that composting is a low-cost, eco-friendly method for recycling coffee-derived by-products, such as CS, promoting a zero-waste strategy. The study indicated that starting the composting process with a 1:1 ratio of CS to co-substrate and adding extra doses of CS as the process continues ensures the recommended exposure time of two weeks at 55 °C during the thermophilic phase, maximising the amount of compostable CS. Additionally, incorporating biochar (BC) into the compost blend reduces the risk of heap compaction, promotes airflow, and enhances conditions for aerobic digestion.

The macro and micronutrient contents make the resulting compost particularly valuable as an organic substrate, rich in nitrogen (N) and potassium (K). Notably, the study revealed that composting the CS produced in Europe could recover 2,420–3,481 tons of N and K, reducing reliance on mineral fertilisers and contributing to the challenge of meeting the growing demand for food security without compromising the environment.

The high water-holding capacity (237–351 % dw) suggests that these materials can potentially increase the quantity and persistence of plant-available water, a crucial feature in the challenging conditions of a green roof environment.

Lastly, the trace metal content in CS-derived composts was far below levels of concern, and the phytotoxicity caused by chlorogenic acid significantly decreased through composting. This indicates that CS-derived compost can be used safely, as demonstrated by its phytostimulant behaviour in germination tests.

Question 2 - *At which proportions do composts-based substrates ensure the best conditions for plant growth* (Chapter 6)?

Despite the substrates' efficiency being influenced by dilution rates and the plant development stage, the results proved that up to 50 % of peat can be successfully replaced by the co-composted growing media, both with and without biochar. The substrates had a stimulatory effect on seed germination but induced stunted growth due to elevated electrical conductivity. Conversely, for later stages of plant development, compost mixed with peat at 50 %, both with and without biochar, promoted an increase in tomato fruit production by 60.8 % and 100.3 %, respectively, compared to the peat-control substrate.

Question 3 - *Are composts derived from organic waste suitable for rooftop agriculture, and what is their agronomic potential for plant production* (Chapter 7)?

The three-year green roof study confirmed that compost-based substrates offer a sustainable alternative to peat-based ones for rooftop urban agriculture. Remarkably, adding biochar improved the physical and chemical stability of the substrates while positively affecting yield, as hypothesized in this thesis. However, its effectiveness varied among feedstocks. Specifically, biochar addition effectively reduced N_{min} and P loss, which allows for sustainable management of these nutrients, although its role was less evident for the other nutrients examined in this study. Even under unfavourable climatic conditions, the tested substrates resulted in higher productivity of tomatoes, up to 102 % more than peat-based controls and 43 % more than open-field cultivation systems. For lettuce, compost-based substrates, 32 with or without biochar, resulted in 149% and 380% higher yields, respectively, when compared 33 to peat-based controls. Moreover, the interaction between biochar and the selected feedstock proved crucial in optimising fruit quality, comparable to but lower than for the open-field control.

However, the reported progressive nutrient depletion suggests that setting a fertilisation plan over three years would be advisable to mitigate potential plant abiotic stress and ensure sustained growth and productivity. Despite these variations, the general trend confirms the robustness of biochar-blended compost-based substrates in supporting urban horticultural production. The results suggest that this practice could effectively reduce pressure on peatlands by adopting a circular production chain model in the urban horticultural sector, thereby contributing to a more resilient and sustainable food system.

8.2. Future Perspective

This thesis has made significant strides in bridging the knowledge gap regarding the use of organic substrates in RA and the application of biochar as a structural agent. However, scaling up these practices at an urban level requires deeper exploration into optimising this technique.

Future studies should explore additional feedstocks for RA growing media by optimising the proportions of compost and biochar for various crops and stages of plant development. The aim is to overcome the use of peat by implying additional by-products, thereby enhancing substrate sustainability. Identifying locally available materials year-round and collaborating with public and private producers to ensure a steady supply is essential. This strategy will improve the agronomic efficiency of compost-based substrates and strengthen local circular economy networks.

Evaluating the long-term impact of these substrates on plant growth, designing more efficient crop rotations, and ensuring optimal fertilisation will provide crucial information for adopting these practices and reducing the need for external inputs.

A critical aspect to consider is evaluating the spatial extension potential of RA on a city scale. Conducting a detailed study to determine the extent of rooftops that could be effectively utilised for urban agriculture projects would provide essential data for the large-scale implementation of these practices. This includes a comprehensive evaluation of nutrient dynamics, food security and safety impacts, and the potential ecosystem services that RA based on co-composted substrates can provide across the urban environment.

Lastly, conducting an economic feasibility study would help better understand the costs and benefits of adopting these practices on a large scale, providing a comprehensive framework for strategic decisions.

These new research perspectives could significantly contribute to enhancing the understanding and adoption of composting practices and the use of biochar in RA, promoting a more resilient and sustainable urban food system. Future exploration cannot be limited to the scientific community but requires the active engagement of agricultural professionals, urban planners, and policymakers interested in reshaping city sustainability.

References

Abd El-Azeem, S. A. M., Ahmad, M., Usman, A. R. A., Kim, K. R., Oh, S. E., Lee, S. S., & Ok, Y. S. (2013). Changes of biochemical properties and heavy metal bioavailability in soil treated with natural liming materials. *Environmental Earth Sciences*, *70*(7), 3411–3420. https://doi.org/10.1007/S12665-013-2410-3

Acioly, Claudio., Vignol, Raphaelle., Jonsson, Asa., & UN-Habitat. (2020). *The New Urban Agenda illustrated*. ONU-Habitat.

Agarwal, P., Saha, S., & Hariprasad, P. (2023). Agro-industrial-residues as potting media: physicochemical and biological characters and their influence on plant growth. *Biomass Conversion and Biorefinery*, *13*(11), 1. https://doi.org/10.1007/S13399-021-01998-6

Al-Charchafchi, F., & Al-Quadan, F. (2010). Effect of Chlorogenic Acid on Germination and Seedling Growth, and on the Enzymes Activity Extracted from Artemisia herba alba ASSO. Part I: Germination and Seedling Growth. In *Dirasat: Pure Sciences* (Vol. 33, Issue 2). https://doi.org/10.1016/j.chemosphere.2012.03.053

Alsamir, M., Mahmood, T., Trethowan, R., & Ahmad, N. (2021). An overview of heat stress in tomato (Solanum lycopersicum L.). *Saudi Journal of Biological Sciences*, *28*(3), 1654–1663. https://doi.org/10.1016/J.SJBS.2020.11.088

Andersson, E., Barthel, S., Borgström, S., Colding, J., Elmqvist, T., Folke, C., & Gren, Å. (2014). Reconnecting cities to the biosphere: Stewardship of green infrastructure and urban ecosystem services. *Ambio*, *43*(4), 445–453. https://doi.org/10.1007/S13280-014-0506-Y/FIGURES/3

Anthon, G., & Barrett, D. M. (2007). Standardization of a rapid spectrophotometric method for lycopene analysis. *Acta Horticulturae*, *758*, 111–128. https://doi.org/10.17660/ActaHortic.2007.758.12

Anthon, G. E., Lestrange, M., & Barrett, D. M. (2011). Changes in pH, acids, sugars and other quality parameters during extended vine holding of ripe processing tomatoes. *Journal of the Science of Food and Agriculture*, *91*(7), 1175–1181. https://doi.org/10.1002/jsfa.4312

Antisari, L. V., Orsini, F., Marchetti, L., Vianello, G., & Gianquinto, G. (2015). Heavy metal accumulation in vegetables grown in urban gardens. *Agronomy for Sustainable Development*, *35*(3), 1139–1147. https://doi.org/10.1007/S13593-015-0308-Z/FIGURES/3

Appolloni, E., Orsini, F., Specht, K., Thomaier, S., Sanyé-Mengual, E., Pennisi, G., & Gianquinto, G. (2021). The global rise of urban rooftop agriculture: A review of worldwide cases. *Journal of Cleaner Production*, *296*, 126556. https://doi.org/10.1016/J.JCLEPRO.2021.126556

Arcas-Pilz, V., Gabarrell, X., Orsini, F., & Villalba, G. (2023). Literature review on the potential of urban waste for the fertilization of urban agriculture: A closer look at the metropolitan area of Barcelona. In *Science of the Total Environment* (Vol. 905). https://doi.org/10.1016/j.scitotenv.2023.167193

Atzori, G., Pane, C., Zaccardelli, M., Cacini, S., & Massa, D. (2021). The role of peat-free organic substrates in the sustainable management of soilless cultivations. In *Agronomy* (Vol. 11, Issue 6). https://doi.org/10.3390/agronomy11061236

Augustsson, A., Lundgren, M., Qvarforth, A., Engström, E., Paulukat, C., Rodushkin, I., Moreno-Jiménez, E., Beesley, L., Trakal, L., & Hough, R. L. (2023). Urban vegetable contamination - The role of adhering particles and their significance for human exposure. *Science of The Total Environment*, *900*, 165633. https://doi.org/10.1016/J.SCITOTENV.2023.165633

Azim, K., Soudi, B., Boukhari, S., Perissol, C., Roussos, S., & Thami Alami, I. (2018). Composting parameters and compost quality: a literature review. In *Organic Agriculture* (Vol. 8, Issue 2, pp. 141–158). Springer Netherlands. https://doi.org/10.1007/s13165-017-0180-z

Ballesteros, L. F., Teixeira, J. A., & Mussatto, S. I. (2014). Chemical, Functional, and Structural Properties of Spent Coffee Grounds and Coffee Silverskin. *Food and Bioprocess Technology*, *7*(12), 3493–3503. https://doi.org/10.1007/S11947-014-1349-Z/FIGURES/4

Barrett, G. E., Alexander, P. D., Robinson, J. S., & Bragg, N. C. (2016). Achieving environmentally sustainable growing media for soilless plant cultivation systems – A review. In *Scientia Horticulturae* (Vol. 212, pp. 220–234). Elsevier B.V. https://doi.org/10.1016/j.scienta.2016.09.030

Barthel, S., & Isendahl, C. (2013). Urban gardens, agriculture, and water management: Sources of resilience for long-term food security in cities. *Ecological Economics*, *86*, 224–234. https://doi.org/10.1016/J.ECOLECON.2012.06.018

Barthel, S., Isendahl, C., Vis, B. N., Drescher, A., Evans, D. L., & van Timmeren, A. (2019). Global urbanization and food production in direct competition for land: Leverage places to mitigate impacts on SDG2 and on the Earth System. *Anthropocene Review*, *6*(1–2), 71–97. https://doi.org/10.1177/2053019619856672

Barthel, S., Parker, J., & Ernstson, H. (2015). Food and Green Space in Cities: A Resilience Lens on Gardens and Urban Environmental Movements. *Urban Studies*, *52*(7), 1321–1338. https://doi.org/10.1177/0042098012472744

Bazzocchi, G., & Maini, S. (2017). *Sustainable Pest Management*. 167–193. https://doi.org/10.1007/978-3-319-57720-3_11

Beckles, D. M. (2012). Factors affecting the postharvest soluble solids and sugar content of tomato (Solanum lycopersicum L.) fruit. *Postharvest Biology and Technology*, *63*(1), 129–140. https://doi.org/10.1016/J.POSTHARVBIO.2011.05.016

Bell, S. , Fox-Kämper R., Keshavarz, N. , Benson, M. , Caputo, S. , Noori, S. , & Voigt, A. (2016). Urban Allotment Gardens in Europe. *Urban Allotment Gardens in Europe*. https://doi.org/https://doi.org/10.4324/9781315686608

Benzie, I. F. F., & Strain, J. J. (1999). Ferric reducing/antioxidant power assay: Direct measure of total antioxidant activity of biological fluids and modified version for simultaneous measurement of total antioxidant power and ascorbic acid concentration. *Methods in Enzymology*, *299*(1995), 15–27. https://doi.org/10.1016/S0076-6879(99)99005-5

Berti, F., Salas-Sanjuán, M. del C., Hernández-López, F., Correa-Bustos, A., & Segura-Pérez, M. L. (2023). Use of Compost Based on Invasive Algae Rugulopteryx okamurae as a Peat Alternative in Nursery Growing Media. *Agronomy*, *13*(4). https://doi.org/10.3390/agronomy13040948

Bohn, K., & Viljoen, A. (2011). The edible city: envisioning the Continuous Productive Urban Landscape (CPUL). *Field Journal*, *4*(1), 149–161. http://www.field-journal.org/index.php?page=issue-4

Boletín Oficial del Estado. (2020). Real Decreto 506/2013, de 28 de junio, sobre productos fertilizantes. In *Boletín Oficial del Estado (BOE)* (Vol. 1855). https://www.boe.es/buscar/pdf/2013/BOE-A-2013-7540-consolidado.pdf

Bretzel, F., Calderisi, M., Scatena, M., & Pini, R. (2016). Soil quality is key for planning and managing urban allotments intended for the sustainable production of home-consumption vegetables. *Environmental Science and Pollution Research*, *23*(17), 17753–17760. https://doi.org/10.1007/S11356-016-6819-6/FIGURES/5

Bretzel, F., Vannucchi, F., Benvenuti, S., & Rumble, H. (2017). Biodiversity of Flora and Fauna. In *Rooftop Urban Agriculture* (pp. 235–252). Springer, Cham. https://doi.org/10.1007/978-3-319-57720-3_14

Buscaroli, E., Braschi, I., Cirillo, C., Fargue-Lelièvre, A., Modarelli, G. C., Pennisi, G., Righini, I., Specht, K., & Orsini, F. (2021). Reviewing chemical and biological risks in urban agriculture: A comprehensive framework for a food safety assessment of city region food systems. *Food Control*, *126*, 108085. https://doi.org/10.1016/J.FOODCONT.2021.108085

Bylund, J., Gollner, C., Jäger, M., Klaming, G., Noll, M., Riegler, J., Rodenstedt, A., & Wallsten, B. (2022). *European Partnership Driving Urban Transitions to a Sustainable Future Roadmap*.

Cabannes, Y., & Ross, P. (2018). *Food Planning in Garden Cities: The Letchworth Legacy Pioneering urban agriculture and food integration into urban planning and design*. www.ruaf.org

Cabral, I., Keim, J., Engelmann, R., Kraemer, R., Siebert, J., & Bonn, A. (2017). Ecosystem services of allotment and community gardens: A Leipzig, Germany case study. *Urban Forestry & Urban Greening*, *23*, 44–53. https://doi.org/10.1016/J.UFUG.2017.02.008

Calheiros, C. S. C., & Stefanakis, A. I. (2021). Green Roofs Towards Circular and Resilient Cities. *Circular Economy and Sustainability*, *1*(1), 395–411. https://doi.org/10.1007/S43615-021-00033-0/FIGURES/3

Calvet, M. C., Gorostiza, S., & Pujol, D. S. (2021). Growing food under fire: Urban and peri-urban agriculture in Barcelona during the Spanish civil 0, 1936-1939. In *Historia Agraria* (Issue 84, pp. 141–171). Universidad de Murcia. https://doi.org/10.26882/HISTAGRAR.084E08C

Camps-Calvet, M., Gorostiza, S., & Saurí, D. (2022). Feeding the City and Making the Revolution: Women and Urban Agriculture in Barcelona during the Spanish Civil War (1936–1939). *Antipode*, *54*(4), 1021–1042. https://doi.org/10.1111/anti.12819

Cao, C. T. N., Farrell, C., Kristiansen, P. E., & Rayner, J. P. (2014). Biochar makes green roof substrates lighter and improves water supply to plants. *Ecological Engineering*, *71*, 368–374. https://doi.org/10.1016/J.ECOLENG.2014.06.017

Caputo, S. (2022). Introduction. In *Small Scale Soil-less Urban Agriculture in Europe* (pp. 1–15). Springer, Cham. https://doi.org/10.1007/978-3-030-99962-9_1

Caputo, S., Iglesias, P., & Rumble, H. (2017). *Elements of Rooftop Agriculture Design*. 39–59. https://doi.org/10.1007/978-3-319-57720-3_4

Carlile, W. R., Cattivello, C., & Zaccheo, P. (2015). Organic Growing Media: Constituents and Properties. *Vadose Zone Journal*, *14*(6), 1–13. https://doi.org/10.2136/vzj2014.09.0125

Carlile, W. R., Raviv, M., & Prasad, M. (2019). Organic soilless media components. In *Soilless Culture: Theory and Practice Theory and Practice* (pp. 303–378). Elsevier. https://doi.org/10.1016/B978-0-444-63696-6.00008-6

Castaldi, P., & Melis, P. (2004). Growth and Yield Characteristics and Heavy Metal Content on Tomatoes Grown in Different Growing Media. *Communications in Soil Science and Plant Analysis*, *35*(1–2), 85–98. https://doi.org/10.1081/CSS-120027636

Castleton, H. F., Stovin, V., Beck, S. B. M., & Davison, J. B. (2010). Green roofs; Building energy savings and the potential for retrofit. In *Energy and Buildings* (Vol. 42, Issue 10, pp. 1582–1591). Elsevier Ltd. https://doi.org/10.1016/j.enbuild.2010.05.004

Charon Cardona, E., & Markwick, R. D. (2019). The kitchen garden movement on the Soviet home front, 1941–1945. *Journal of Historical Geography*, *64*, 47–59. https://doi.org/10.1016/J.JHG.2018.12.006

Chen, H., Du, X., Lai, M., Nazhafati, M., Li, C., & Qi, W. (2021). Biochar Improves Sustainability of Green Roofs via Regulate of Soil Microbial

Communities. *Agriculture 2021, Vol. 11, Page 620*, *11*(7), 620. https://doi.org/10.3390/AGRICULTURE11070620

Childe, V. G. (1950). The Urban Revolution. *The Town Planning Review*, *21*(1), 3–17. http://www.jstor.org/stable/40102108.

Clinton, N., Stuhlmacher, M., Miles, A., Uludere Aragon, N., Wagner, M., Georgescu, M., Herwig, C., & Gong, P. (2018). A Global Geospatial Ecosystem Services Estimate of Urban Agriculture. *Earth's Future*, *6*(1), 40–60. https://doi.org/10.1002/2017EF000536

Commission on Environment, W. (1987). *Report of the World Commission on Environment and Development: Our Common Future Towards Sustainable Development 2. Part II. Common Challenges Population and Human Resources 4.*

Coulibaly, S. F. M., Aubry, C., Provent, F., Rousset-Rouvière, S., & Joimel, S. (2023). The role of green roofs as urban habitats for biodiversity modulated by their design: a review. *Environmental Research Letters*, *18*(7), 073003. https://doi.org/10.1088/1748-9326/ACD801

Crippa, M., Solazzo, E., Guizzardi, D., Monforti-Ferrario, F., Tubiello, F. N., & Leip, A. (2021). Food systems are responsible for a third of global anthropogenic GHG emissions. *Nature Food*, *2*(3), 198–209. https://doi.org/10.1038/s43016-021-00225-9

D'Amour, C. B., Reitsma, F., Baiocchi, G., Barthel, S., Güneralp, B., Erb, K. H., Haberl, H., Creutzig, F., & Seto, K. C. (2017). Future urban land expansion and implications for global croplands. *Proceedings of the National Academy of Sciences of the United States of America*, *114*(34), 8939–8944. https://doi.org/10.1073/PNAS.1606036114/-/DCSUPPLEMENTAL/PNAS.201606036SI.PDF

De Bon, H., Parrot, L., & Moustier, P. (2010). Sustainable urban agriculture in developing countries. A review. *Agronomy for Sustainable Development*, *30*(1), 21–32. https://doi.org/10.1051/AGRO:2008062/METRICS

Di Bonito, R., Biagiotti, D., Giagnacovo, G., Viola, C., & Campiotti, C. A. (2018). Sustainable and energy saving urban horticulture on rooftop gardens in Mediterranean climatic conditions. *Acta Horticulturae*, *1215*, 383–388. https://doi.org/10.17660/ActaHortic.2018.1215.70

Diaz, L. F. , De Bertoldi, M. , & Bidlingmaier, W. (2007). *Compost Science and Technology*.

Dobele, M., & Zvirbule, A. (2020). The concept of urban agriculture – Historical development and tendencies. *Rural Sustainability Research*, *43*(388), 20–26. https://doi.org/10.2478/plua-2020-0003

Domínguez, M. T., Panettieri, M., Madejón, E., & Madejón, P. (2020). Thistle crops in marginal lands after compost addition: Plant biomass and effect on soil physical, chemical and biological properties. *Land Degradation & Development*, *31*(9), 1167–1175. https://doi.org/10.1002/LDR.3510

Dorais, M., Ehret, D. L., & Papadopoulos, A. P. (2008). Tomato (Solanum lycopersicum) health components: From the seed to the consumer. *Phytochemistry Reviews*, *7*(2), 231–250. https://doi.org/10.1007/s11101-007-9085-x

D'ostuni, M., Stanghellini, C., Boedijn, A., Zaffi, L., & Orsini, F. (2023). Evaluating the impacts of nutrients recovery from urine wastewater in Building-Integrated Agriculture. A test case study in Amsterdam. *Sustainable Cities and Society*, *91*, 104449. https://doi.org/10.1016/J.SCS.2023.104449

Elmqvist, T., Redman, C. L., Barthel, S., & Costanza, R. (2013). History of urbanization and the missing ecology. *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment*, 13–30. https://doi.org/10.1007/978-94-007-7088-1_2/FIGURES/2

European Commission. (2018). Directive (EU) 2018/850 of the European Parliament and of the Council of 30 May 2018 amending Directive 1999/31/EC on the landfill of waste. *Official Journal of the European Union*, *April*, 100–108. https://eur-lex.europa.eu/legal-content/EN/TXT/PDF/?uri=CELEX:32018L0850&from=EN

European Commission. (2019). *The European Green Deal*.

European Commission. (2020a). *100 climate-neutral cities by 2030 - by and for the citizens*. https://doi.org/10.2777/345941

European Commission. (2020b). Circular economy action plan. In *European Commission* (Issue March). https://doi.org/10.2775/855540

European Commission. (2020c). Farm to Fork Strategy. In *DG SANTE/Unit 'Food information and composition, food waste'"* (Issue DG SANTE/Unit 'Food information and composition, food waste'). https://ec.europa.eu/food/sites/food/files/safety/docs/f2f_action-plan_2020_strategy-info_en.pdf

European Commission. (2021a). *A Soil Deal for Europe*.

European Commission. (2021b). *New European Bauhaus*.

European Commission. (2022). *Proposal for a regulation on nature restoration*. https://doi.org/10.5281/zenodo.5657041

FAO. (2006). *Policy Brief Changing Policy Concepts of Food Security*. http://www.foodsecinfoaction.org/

FAO, IFAD, UNICEF, WFP, & WHO. (2023). The State of Food Security and Nutrition in the World 2023. In *The State of Food Security and Nutrition in the World 2023*. FAO; IFAD; UNICEF; WFP; WHO; https://doi.org/10.4060/cc3017en

FAO, Rikolto, & RUAF. (2022). *Urban and peri-urban agriculture sourcebook – From production to food systems*. FAO; Rikolto International s.o.n.; RUAF Global Partnership on Sustainable Urban Agriculture and Food Systems; https://doi.org/10.4060/cb9722en

FAOSTAT. (n.d.). Retrieved 27 April 2024, from https://www.fao.org/faostat/en/#data/QCL

Farhangi-Abriz, S., & Torabian, S. (2018). Biochar Increased Plant Growth-Promoting Hormones and Helped to Alleviates Salt Stress in Common Bean Seedlings. *Journal of Plant Growth Regulation*, *37*(2), 591–601. https://doi.org/10.1007/s00344-017-9756-9

Farrell, M., & Jones, D. L. (2010). Food waste composting: Its use as a peat replacement. *Waste Management*, *30*(8–9), 1495–1501. https://doi.org/10.1016/j.wasman.2010.01.032

Filippini, R., Mazzocchi, C., & Corsi, S. (2019). The contribution of Urban Food Policies toward food security in developing and developed countries: A network analysis approach. *Sustainable Cities and Society*, *47*, 101506. https://doi.org/10.1016/J.SCS.2019.101506

Freeman, B. W. J., Evans, C. D., Musarika, S., Morrison, R., Newman, T. R., Page, S. E., Wiggs, G. F. S., Bell, N. G. A., Styles, D., Wen, Y., Chadwick,

D. R., & Jones, D. L. (2022). Responsible agriculture must adapt to the wetland character of mid-latitude peatlands. In *Global Change Biology* (Vol. 28, Issue 12, pp. 3795–3811). John Wiley and Sons Inc. https://doi.org/10.1111/gcb.16152

García-Nieto, A. P., Geijzendorffer, I. R., Baró, F., Roche, P. K., Bondeau, A., & Cramer, W. (2018). Impacts of urbanization around Mediterranean cities: Changes in ecosystem service supply. *Ecological Indicators*, *91*, 589–606. https://doi.org/10.1016/J.ECOLIND.2018.03.082

Gascó, G., Cely, P., Paz-Ferreiro, J., Plaza, C., & Méndez, A. (2016). Relation between biochar properties and effects on seed germination and plant development. *Biological Agriculture and Horticulture*, *32*(4), 237–247. https://doi.org/10.1080/01448765.2016.1166348

Ghisellini, P., Cialani, C., & Ulgiati, S. (2016). A review on circular economy: The expected transition to a balanced interplay of environmental and economic systems. *Journal of Cleaner Production*, *114*, 11–32. https://doi.org/10.1016/j.jclepro.2015.09.007

Godlewska, P., Schmidt, H. P., Ok, Y. S., & Oleszczuk, P. (2017). Biochar for composting improvement and contaminants reduction. A review. In *Bioresource Technology* (Vol. 246, pp. 193–202). Elsevier. https://doi.org/10.1016/j.biortech.2017.07.095

Goldstein, B., Hauschild, M., Fernández, J., & Birkved, M. (2016). Urban versus conventional agriculture, taxonomy of resource profiles: a review. *Agronomy for Sustainable Development*, *36*(1), 1–19. https://doi.org/10.1007/S13593-015-0348-4/TABLES/3

Gómez-Brandón, M., Lazcano, C., & Domínguez, J. (2008). The evaluation of stability and maturity during the composting of cattle manure. *Chemosphere*, *70*(3), 436–444. https://doi.org/10.1016/J.CHEMOSPHERE.2007.06.065

Goñi-Urtiaga, A., Courtier-Murias, D., Picca, G., Valentín, J. L., Plaza, C., & Panettieri, M. (2022). Response of water-biochar interactions to physical and biochemical aging. *Chemosphere*, *307*, 136071. https://doi.org/10.1016/J.CHEMOSPHERE.2022.136071

Gorgolewski, M., & Straka, V. (2017). *Integrating Rooftop Agriculture into Urban Infrastructure*. 113–126. https://doi.org/10.1007/978-3-319-57720-3_8

Grafmüller, J., Schmidt, H. P., Kray, D., & Hagemann, N. (2022). Root-Zone Amendments of Biochar-Based Fertilizers: Yield Increases of White Cabbage in Temperate Climate. *Horticulturae*, *8*(4), 1–17. https://doi.org/10.3390/horticulturae8040307

Grard, B., Bel, N., Marchal, N., Madre, F., Castell, J. F., Cambier, P., Houot, S., Manouchehri, N., Besancon, S., Michel, J. C., Chenu, C., Frascaria-Lacoste, N., & Aubry, C. (2015). Recycling urban waste as possible use for rooftop vegetable garden. *Future of Food: Journal on Food, Agriculture and Society*, *3*(1), 21–34.

Grard, B., Chenu, C., Manouchehri, N., Houot, S., Frascaria-Lacoste, N., & Aubry, C. (2018). Rooftop farming on urban waste provides many ecosystem services. *Agronomy for Sustainable Development*, *38*(1). https://doi.org/10.1007/S13593-017-0474-2

Grard, B. J. P., Manouchehri, N., Aubry, C., Frascaria-Lacoste, N., & Chenu, C. (2020). Potential of technosols created with urban by-products for rooftop edible production. *International Journal of Environmental Research and Public Health*, *17*(9), 1–21. https://doi.org/10.3390/ijerph17093210

Gruda, N. (2009). Do soilless culture systems have an influence on product quality of vegetables? *Journal of Applied Botany and Food Quality*, *82*, 141–147.

Gruda, N. (2019). Increasing sustainability of growing media constituents and stand-alone substrates in soilless culture systems. *Agronomy*, *9*(6), 298. https://doi.org/10.3390/agronomy9060298

Gruda, N. S., Hirschler, O., & Stuart, J. (2024). Peat reduction in horticulture - an overview of Europe. *Acta Horticulturae*, *1391*, 545–559. https://doi.org/10.17660/ActaHortic.2024.1391.75

Guardiola, J. L. (1996). Plant hormones. Physiology, biochemistry and molecular biology [Book]. *Scientia Horticulturae*, *66*(3–4), 267–270. https://doi.org/10.1016/s0304-4238(96)00922-3

Güneralp, B., Reba, M., Hales, B. U., Wentz, E. A., & Seto, K. C. (2020). Trends in urban land expansion, density, and land transitions from 1970 to 2010: a global synthesis. *Environmental Research Letters*, *15*(4), 044015. https://doi.org/10.1088/1748-9326/AB6669

Hachicha, R., Rekik, O., Hachicha, S., Ferchichi, M., Woodward, S., Moncef, N., Cegarra, J., & Mechichi, T. (2012). Co-composting of spent coffee ground with olive mill wastewater sludge and poultry manure and effect of Trametes versicolor inoculation on the compost maturity. *Chemosphere*, *88*(6), 677–682. https://doi.org/10.1016/j.chemosphere.2012.03.053

Hagemann, N., Joseph, S., Schmidt, H. P., Kammann, C. I., Harter, J., Borch, T., Young, R. B., Varga, K., Taherymoosavi, S., Elliott, K. W., McKenna, A., Albu, M., Mayrhofer, C., Obst, M., Conte, P., Dieguez-Alonso, A., Orsetti, S., Subdiaga, E., Behrens, S., & Kappler, A. (2017). Organic coating on biochar explains its nutrient retention and stimulation of soil fertility. *Nature Communications*, *8*(1). https://doi.org/10.1038/s41467-017-01123-0

Harada, Y., & Whitlow, T. H. (2020). Urban Rooftop Agriculture: Challenges to Science and Practice. *Frontiers in Sustainable Food Systems*, *4*, 511117. https://doi.org/10.3389/FSUFS.2020.00076/BIBTEX

Hartmann, A., Patz, C. D., Andlauer, W., Dietrich, H., & Ludwig, M. (2008). Influence of processing on quality parameters of strawberries. *Journal of Agricultural and Food Chemistry*, *56*(20), 9484–9489. https://doi.org/10.1021/jf801555q

Hawes, J. K., Goldstein, B. P., Newell, J. P., Dorr, E., Caputo, S., Fox-Kämper, R., Grard, B., Ilieva, R. T., Fargue-Lelièvre, A., Poniży, L., Schoen, V., Specht, K., & Cohen, N. (2024). Comparing the carbon footprints of urban and conventional agriculture. *Nature Cities 2024 1:2*, *1*(2), 164–173. https://doi.org/10.1038/s44284-023-00023-3

Herrera, F., Castillo, J. E., Chica, A. F., & López Bellido, L. (2008). Use of municipal solid waste compost (MSWC) as a growing medium in the nursery production of tomato plants. *Bioresource Technology*, *99*(2), 287–296. https://doi.org/10.1016/j.biortech.2006.12.042

Holt-Giménez, E., Shattuck, A., Altieri, M., Herren, H., & Gliessman, S. (2012). We Already Grow Enough Food for 10 Billion People... and Still Can't End Hunger. *Journal of Sustainable Agriculture*, *36*(6), 595–598. https://doi.org/10.1080/10440046.2012.695331

Horst, M., Mcclintock, N., & Hoey, L. (2017). The Intersection of Planning, Urban Agriculture, and Food Justice: A Review of the Literature. *Journal* *of the American Planning Association*, *83*(3), 277–295. https://doi.org/10.1080/01944363.2017.1322914

Huang, L., & Gu, M. (2019). Effects of biochar on container substrate properties and growth of plants—a review. In *Horticulturae* (Vol. 5, Issue 1, p. 14). Multidisciplinary Digital Publishing Institute. https://doi.org/10.3390/horticulturae5010014

Huang, L., Niu, G., Feagley, S. E., & Gu, M. (2019). Evaluation of a hardwood biochar and two composts mixes as replacements for a peat-based commercial substrate. *Industrial Crops and Products*, *129*, 549–560. https://doi.org/10.1016/j.indcrop.2018.12.044

Iaquinta, D. L., & Drescher, A. W. (2015). Urban agriculture: A comparative review of allotment and community gardens. In *Urban Ecosystem Ecology* (pp. 199–226). wiley. https://doi.org/10.2134/agronmonogr55.c10

Ilieva, R. T., Cohen, N., Israel, M., Specht, K., Fox-Kämper, R., Fargue-Lelièvre, A., Poniży, L., Schoen, V., Caputo, S., Kirby, C. K., Goldstein, B., Newell, J. P., & Blythe, C. (2022). The Socio-Cultural Benefits of Urban Agriculture: A Review of the Literature. In *Land* (Vol. 11, Issue 5). MDPI. https://doi.org/10.3390/land11050622

Illera-Vives, M., Seoane Labandeira, S., Brito, L. M., López-Fabal, A., & López-Mosquera, M. E. (2015). Evaluation of compost from seaweed and fish waste as a fertilizer for horticultural use. *Scientia Horticulturae*, *186*, 101–107. https://doi.org/10.1016/J.SCIENTA.2015.02.008

Insam, H., & de Bertoldi, M. (2007). Chapter 3 Microbiology of the composting process. In *Waste Management Series* (Vol. 8, pp. 25–48). Elsevier. https://doi.org/10.1016/S1478-7482(07)80006-6

International Energy Agency, I. (2021). *Empowering Cities for a Net Zero Future: Unlocking resilient, smart, sustainable urban energy systems*.

IPCC. (2022). Framing and Context. *Global Warming of 1.5°C*, 49–92. https://doi.org/10.1017/9781009157940.003

Izquierdo, M., De Miguel, E., Ortega, M. F., & Mingot, J. (2015). Bioaccessibility of metals and human health risk assessment in community urban gardens. *Chemosphere*, *135*, 312–318. https://doi.org/10.1016/J.CHEMOSPHERE.2015.04.079

Janissen, B., & Huynh, T. (2018). Chemical composition and value-adding applications of coffee industry by-products: A review. *Resources, Conservation and Recycling*, *128*, 110–117. https://doi.org/10.1016/j.resconrec.2017.10.001

Jean-Soro, L., Le Guern, C., Bechet, B., Lebeau, T., & Ringeard, M. F. (2015). Origin of trace elements in an urban garden in Nantes, France. *Journal of Soils and Sediments*, *15*(8), 1802–1812. https://doi.org/10.1007/S11368-014-0952-Y/METRICS

Jindo, K., Sánchez-Monedero, M. A., Mastrolonardo, G., Audette, Y., Higashikawa, F. S., Silva, C. A., Akashi, K., & Mondini, C. (2020). Role of biochar in promoting circular economy in the agriculture sector. Part 2: A review of the biochar roles in growing media, composting and as soil amendment. In *Chemical and Biological Technologies in Agriculture* (Vol. 7, Issue 1, p. 16). https://doi.org/10.1186/s40538-020-00179-3

Johannes Lehmann and Stephen Joseph. (2012). Biochar for Environmental Management. In *Biochar for Environmental Management*. https://doi.org/10.4324/9781849770552

Joimel, S., Grard, B., Chenu, C., Cheval, P., Mondy, S., Lelièvre, M., Auclerc, A., & Vieublé Gonod, L. (2022). One green roof type, one Technosol, one ecological community. *Ecological Engineering*, *175*. https://doi.org/10.1016/j.ecoleng.2021.106475

Kabisch, N., Korn, H., Stadler, J., Bonn, A., Kabisch, N., Korn, H., Stadler, • J, & Bonn, A. (2017). Nature-Based Solutions to Climate Change Adaptation in Urban Areas—Linkages Between Science, Policy and Practice. *Theory and Practice of Urban Sustainability Transitions*, 1–11. https://doi.org/10.1007/978-3-319-56091-5_1

Kammann, C. I., Schmidt, H. P., Messerschmidt, N., Linsel, S., Steffens, D., Müller, C., Koyro, H. W., Conte, P., & Stephen, J. (2015). Plant growth improvement mediated by nitrate capture in co-composted biochar. *Scientific Reports*, *5*, 11080. https://doi.org/10.1038/srep11080

Kaza, S., Yao, L. C., Bhada-Tata, P., & Van Woerden, F. (2018). What a Waste 2.0. *What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050*. https://doi.org/10.1596/978-1-4648-1329-0

Kern, J., Tammeorg, P., Shanskiy, M., Sakrabani, R., Knicker, H., Kammann, C., Tuhkanen, E. M., Smidt, G., Prasad, M., Tiilikkala, K., Sohi, S., Gascó, G., Steiner, C., & Glaser, B. (2017). Synergistic use of peat and charred material in growing media–an option to reduce the pressure on peatlands? In *Journal of Environmental Engineering and Landscape Management* (Vol. 25, Issue 2, pp. 160–174). https://doi.org/10.3846/16486897.2017.1284665

Khan, N., Clark, I., Sánchez-Monedero, M. A., Shea, S., Meier, S., & Bolan, N. (2014). Maturity indices in co-composting of chicken manure and sawdust with biochar. *Bioresource Technology*, *168*, 245–251. https://doi.org/10.1016/j.biortech.2014.02.123

Khan, N., Clark, I., Sánchez-Monedero, M. A., Shea, S., Meier, S., Qi, F., Kookana, R. S., & Bolan, N. (2016). Physical and chemical properties of biochars co-composted with biowastes and incubated with a chicken litter compost. *Chemosphere*, *142*, 14–23. https://doi.org/10.1016/j.chemosphere.2015.05.065

Kimmel, K., & Mander, Ü. (2010). Ecosystem services of peatlands: Implications for restoration. *Progress in Physical Geography*, *34*(4), 491–514. https://doi.org/10.1177/0309133310365595

Kingsley, J., Egerer, M., Nuttman, S., Keniger, L., Pettitt, P., Frantzeskaki, N., Gray, T., Ossola, A., Lin, B., Bailey, A., Tracey, D., Barron, S., & Marsh, P. (2021). Urban agriculture as a nature-based solution to address socio-ecological challenges in Australian cities. *Urban Forestry & Urban Greening*, *60*, 127059. https://doi.org/10.1016/J.UFUG.2021.127059

Kleerekoper, L., Van Esch, M., & Salcedo, T. B. (2012). How to make a city climate-proof, addressing the urban heat island effect. *Resources, Conservation and Recycling*, *64*, 30–38. https://doi.org/10.1016/J.RESCONREC.2011.06.004

Kulak, M., Graves, A., & Chatterton, J. (2013). Reducing greenhouse gas emissions with urban agriculture: A Life Cycle Assessment perspective. *Landscape and Urban Planning*, *111*(1), 68–78. https://doi.org/10.1016/J.LANDURBPLAN.2012.11.007

Kumar, P., Eid, E. M., Taher, M. A., El-morsy, M. H. E., Osman, H. E. M., Al-bakre, D. A., Adelodun, B., Abou Fayssal, S., Goala, M., Mioč, B., Držaić, V., Ajibade, F. O., Choi, K. S., Kumar, V., & Širić, I. (2022). Biotransforming the Spent Substrate of Shiitake Mushroom (Lentinula edodes Berk.): A Synergistic Approach to Biogas Production and Tomato (Solanum lycopersicum L.) Fertilization. *Horticulturae 2022, Vol. 8, Page 479*, *8*(6), 479. https://doi.org/10.3390/HORTICULTURAE8060479

Laird, D. A., Fleming, P., Davis, D. D., Horton, R., Wang, B., & Karlen, D. L. (2010). Impact of biochar amendments on the quality of a typical Midwestern agricultural soil. *Geoderma*, *158*(3–4), 443–449. https://doi.org/10.1016/J.GEODERMA.2010.05.013

Langemeyer, J., Madrid-Lopez, C., Mendoza Beltran, A., & Villalba Mendez, G. (2021). Urban agriculture — A necessary pathway towards urban resilience and global sustainability? *Landscape and Urban Planning*, *210*, 104055. https://doi.org/10.1016/J.LANDURBPLAN.2021.104055

Lazcano, C., Arnold, J., Tato, A., Zaller, J. G., & Domínguez, J. (2009). Compost y vermicompost como componentes de sustratos artificiales de cultivo en viveros: Efectos en el crecimiento y morfología del tomate. *Spanish Journal of Agricultural Research*, *7*(4), 944–951. https://doi.org/10.5424/sjar/2009074-1107

Lee, J., & Kwon, E. E. (2024). Biochar in green roofs. *Journal of Building Engineering*, *89*, 109272. https://doi.org/10.1016/J.JOBE.2024.109272

Leege, P. B. (1998). Introduction of Test Methods for the Examination of Composting and Compost. In *Beneficial Co-Utilization of Agricultural, Municipal and Industrial by-Products* (pp. 269–282). Springer, Dordrecht. https://doi.org/10.1007/978-94-011-5068-2_22

Leiber-Sauheitl, K., Bohne, H., & Böttcher, J. (2021). First steps toward a test procedure to identify peat substitutes for growing media by means of chemical, physical and biological material characteristics. *Horticulturae*, *7*(7). https://doi.org/10.3390/horticulturae7070164

Leifeld, J., & Menichetti, L. (2018). The underappreciated potential of peatlands in global climate change mitigation strategies. *Nature Communications*, *9*(1), 1–7. https://doi.org/10.1038/s41467-018-03406-6

Li, M., Jia, N., Lenzen, M., Malik, A., Wei, L., Jin, Y., & Raubenheimer, D. (2022). Global food-miles account for nearly 20% of total food-systems emissions. *Nature Food 2022 3:6*, *3*(6), 445–453. https://doi.org/10.1038/s43016-022-00531-w

Liao, W., Drake, J., & Thomas, S. C. (2022). Biochar granulation enhances plant performance on a green roof substrate. *Science of The Total Environment*, *813*, 152638. https://doi.org/10.1016/J.SCITOTENV.2021.152638

Liu, X., Huang, Y., Xu, X., Li, X., Li, X., Ciais, P., Lin, P., Gong, K., Ziegler, A. D., Chen, A., Gong, P., Chen, J., Hu, G., Chen, Y., Wang, S., Wu, Q., Huang, K., Estes, L., & Zeng, Z. (2020). High-spatiotemporal-resolution mapping of global urban change from 1985 to 2015. *Nature Sustainability 2020 3:7*, *3*(7), 564–570. https://doi.org/10.1038/s41893-020-0521-x

López, R., Hallat, J., Castro, A., Miras, A., & Burgos, P. (2019). Heavy metal pollution in soils and urban-grown organic vegetables in the province of Sevilla, Spain. *Biological Agriculture & Horticulture*, *35*(4), 219–237. https://doi.org/10.1080/01448765.2019.1590234

Lucertini, G., & Di Giustino, G. (2021). Urban and peri-urban agriculture as a tool for food security and climate change mitigation and adaptation: The case of mestre. *Sustainability (Switzerland)*, *13*(11). https://doi.org/10.3390/su13115999

Maas, E. V., & Hoffman, G. J. (1977). Crop Salt Tolerance—Current Assessment. *Journal of the Irrigation and Drainage Division*, *103*(2), 115–134. https://doi.org/10.1061/JRCEA4.0001137

Machado, R. M. A., & Serralheiro, R. P. (2017). Soil salinity: Effect on vegetable crop growth. Management practices to prevent and mitigate soil salinization. In *Horticulturae* (Vol. 3, Issue 2). https://doi.org/10.3390/horticulturae3020030

Madejón, E., Panettieri, M., Madejón, P., & Pérez-de-Mora, A. (2021). Composting as Sustainable Managing Option for Seaweed Blooms on Recreational Beaches. *Waste and Biomass Valorization*, *1*, 1–13. https://doi.org/10.1007/s12649-021-01548-1

Mancebo, F. (2018). Gardening the city: Addressing sustainability and adapting to global warming through urban agriculture. *Environments - MDPI*, *5*(3), 1–11. https://doi.org/10.3390/environments5030038

Marutani, M., & Clemente, S. (2021). Compost-based growing media improved yield of leafy lettuce in pot culture. *Agronomy*, *11*(9). https://doi.org/10.3390/agronomy11091762

Massa, D., Bonetti, A., Cacini, S., Faraloni, C., Prisa, D., Tuccio, L., & Petruccelli, R. (2019a). Soilless tomato grown under nutritional stress increases green biomass but not yield or quality in presence of biochar as growing medium. *Horticulture Environment and Biotechnology*, *60*(6), 871–881. https://doi.org/10.1007/S13580-019-00169-X/TABLES/6

Massa, D., Bonetti, A., Cacini, S., Faraloni, C., Prisa, D., Tuccio, L., & Petruccelli, R. (2019b). Soilless tomato grown under nutritional stress increases green biomass but not yield or quality in presence of biochar as growing medium. *Horticulture Environment and Biotechnology*, *60*(6), 871–881. https://doi.org/10.1007/s13580-019-00169-x

Mees, C. (2017a). *The Emergence of Shared Gardens and Self-Built Structures in European Cities*. 7–56. https://doi.org/10.1007/978-3-319-75514-4_2

Mees, C. (2017b). *The Emergence of Shared Gardens and Self-Built Structures in North American Cities*. 57–73. https://doi.org/10.1007/978-3-319-75514-4_3

Méndez, A., Paz-Ferreiro, J., Gil, E., & Gascó, G. (2015). The effect of paper sludge and biochar addition on brown peat and coir based growing media properties. *Scientia Horticulturae*, *193*, 225–230. https://doi.org/10.1016/J.SCIENTA.2015.07.032

Mentaschi, L., Duveiller, G., Zulian, G., Corbane, C., Pesaresi, M., Maes, J., Stocchino, A., & Feyen, L. (2022). Global long-term mapping of surface temperature shows intensified intra-city urban heat island extremes. *Global Environmental Change*, *72*, 102441. https://doi.org/10.1016/J.GLOENVCHA.2021.102441

Mihalakakou, G., Souliotis, M., Papadaki, M., Menounou, P., Dimopoulos, P., Kolokotsa, D., Paravantis, J. A., Tsangrassoulis, A., Panaras, G., Giannakopoulos, E., & Papaefthimiou, S. (2023). Green roofs as a nature-based solution for improving urban sustainability: Progress and perspectives. *Renewable and Sustainable Energy Reviews*, *180*, 113306. https://doi.org/10.1016/J.RSER.2023.113306

Millennium Ecosystem Assessment. (2005). *Ecosystems and human well-being: synthesis*. Island Press.

Mininni, C., Bustamante, M. A., Medina, E., Montesano, F., Paredes, C., Pérez-Espinosa, A., Moral, R., & Santamaria, P. (2013). Evaluation of posidonia seaweed-based compost as a substrate for melon and tomato seedling production. *The Journal of Horticultural Science and Biotechnology*, *88*(3), 345–351. https://doi.org/10.1080/14620316.2013.11512975

Mininni, C., Santamaria, P., Abdelrahman, H. M., Cocozza, C., Miano, T., Montesano, F., & Parente, A. (2012). Posidonia-based Compost as a Peat Substitute for Lettuce Transplant Production. *HortScience*, *47*(10), 1438–1444. https://doi.org/10.21273/HORTSCI.47.10.1438

Miranda, K. M., Espey, M. G., & Wink, D. A. (2001). A rapid, simple spectrophotometric method for simultaneous detection of nitrate and nitrite. *Nitric Oxide : Biology and Chemistry*, *5*(1), 62–71. https://doi.org/10.1006/NIOX.2000.0319

Mok, H. F., Williamson, V. G., Grove, J. R., Burry, K., Barker, S. F., & Hamilton, A. J. (2014). Strawberry fields forever? Urban agriculture in developed countries: A review. *Agronomy for Sustainable Development*, *34*(1), 21–43. https://doi.org/10.1007/S13593-013-0156-7/FIGURES/5

Möller, K., Oberson, A., Bünemann, E. K., Cooper, J., Friedel, J. K., Glæsner, N., Hörtenhuber, S., Løes, A. K., Mäder, P., Meyer, G., Müller, T., Symanczik, S., Weissengruber, L., Wollmann, I., & Magid, J. (2018). Improved Phosphorus Recycling in Organic Farming: Navigating Between Constraints. *Advances in Agronomy*, *147*, 159–237. https://doi.org/10.1016/BS.AGRON.2017.10.004

Montesano, F. F., Parente, A., Italy, B., Grassi, F., & Santamaria, P. (2014). Posidonia-Based Compost as a Growing Medium for the Soilless Cultivation of Tomato. *Acta Horticulturae*, *1034*, 277–282. https://doi.org/10.17660/ActaHortic.2014.1034.34

Mougeot, L. J. A. (2000). Urban Agriculture: Definition, Presence, Potentials and Risks, and Policy Challenges Cities Feeding People Series. In *Cities feeding people series*. http://www.idrc.ca/cfp

Nadal, A., Llorach-Massana, P., Cuerva, E., López-Capel, E., Montero, J. I., Josa, A., Rieradevall, J., & Royapoor, M. (2017). Building-integrated rooftop greenhouses: An energy and environmental assessment in the mediterranean context. *Applied Energy*, *187*, 338–351. https://doi.org/10.1016/j.apenergy.2016.11.051

Nicholls, E., Ely, A., Birkin, L., Basu, P., & Goulson, D. (2020). The contribution of small-scale food production in urban areas to the sustainable development goals: a review and case study. *Sustainability Science*, *15*(6), 1585–1599. https://doi.org/10.1007/S11625-020-00792-Z/FIGURES/1

Nieto, A., Gascó, G., Paz-Ferreiro, J., Fernández, J. M., Plaza, C., & Méndez, A. (2016). The effect of pruning waste and biochar addition on brown peat based growing media properties. *Scientia Horticulturae*, *199*, 142–148. https://doi.org/10.1016/j.scienta.2015.12.012

Nobile, C., Denier, J., & Houben, D. (2020). Linking biochar properties to biomass of basil, lettuce and pansy cultivated in growing media. *Scientia Horticulturae*, *261*, 109001. https://doi.org/10.1016/j.scienta.2019.109001

Nocentini, M., Panettieri, M., García de Castro Barragán, J. M., Mastrolonardo, G., & Knicker, H. (2021). Recycling pyrolyzed organic waste from plant nurseries, rice production and shrimp industry as peat substitute in potting substrates. *Journal of Environmental Management*, *277*, 111436. https://doi.org/10.1016/j.jenvman.2020.111436

Oberndorfer, E., Lundholm, J., Bass, B., Coffman, R. R., Doshi, H., Dunnett, N., Gaffin, S., Köhler, M., Liu, K. K. Y., & Rowe, B. (2007). Green Roofs as Urban Ecosystems: Ecological Structures, Functions, and Services. *BioScience*, *57*(10), 823–833. https://doi.org/10.1641/B571005

Orsini, F., Dubbeling, M., & Gianquinto, G. (2015). Multifunctional rooftop horticulture: a promising strategy for intensifying horticulture production in cities. *Chronica Horticulturae*, *55*(4), 12–17.

Orsini, F., Gasperi, D., Marchetti, L., Piovene, C., Draghetti, S., Ramazzotti, S., Bazzocchi, G., & Gianquinto, G. (2014). Exploring the production capacity of rooftop gardens (RTGs) in urban agriculture: the potential impact on food and nutrition security, biodiversity and other ecosystem services in the city of Bologna. *Food Security*, *6*(6), 781–792. https://doi.org/10.1007/s12571-014-0389-6

Orsini, F., Kahane, R., Nono-Womdim, R., & Gianquinto, G. (2013). Urban agriculture in the developing world: A review. *Agronomy for Sustainable Development*, *33*(4), 695–720. https://doi.org/10.1007/S13593-013-0143-Z/FIGURES/5

Orsini, F., Pennisi, G., Michelon, N., Minelli, A., Bazzocchi, G., Sanyé-Mengual, E., & Gianquinto, G. (2020). Features and Functions of Multifunctional Urban Agriculture in the Global North: A Review. *Frontiers in Sustainable Food Systems*, *4*, 562513. https://doi.org/10.3389/FSUFS.2020.562513/BIBTEX

Pauleit, S., Andersson, E., Anton, B., Buijs, A., Haase, D., Hansen, R., Kowarik, I., Stahl Olafsson, A., & Van der Jagt, S. (2019). Urban green infrastructure – connecting people and nature for sustainable cities. *Urban Forestry & Urban Greening*, *40*, 1–3. https://doi.org/10.1016/J.UFUG.2019.04.007

Pennisi, G., Orsini, F., Gasperi, D., Mancarella, S., Sanoubar, R., Vittori Antisari, L., Vianello, G., & Gianquinto, G. (2016). Soilless system on peat reduce trace metals in urban-grown food: unexpected evidence for a soil origin of plant contamination. *Agronomy for Sustainable Development*, *36*(4), 1–11. https://doi.org/10.1007/S13593-016-0391-9/TABLES/2

Pereira, E. I. P., Conz, R. F., & Six, J. (2017). Nitrogen utilization and environmental losses in organic greenhouse lettuce amended with two distinct biochars. *Science of The Total Environment*, *598*, 1169–1176. https://doi.org/10.1016/J.SCITOTENV.2017.04.062

Petruccelli, R., Bonetti, A., Traversi, M. L., Faraloni, C., Valagussa, M., & Pozzi, A. (2015). Influence of biochar application on nutritional quality of tomato (Lycopersicon esculentum). *Crop and Pasture Science*, *66*(7), 747–755. https://doi.org/10.1071/CP14247

Picca, G., Goñi-Urtiaga, A., Gomez-Ruano, C., Plaza, C., & Panettieri, M. (2023). Suitability of Co-Composted Biochar with Spent Coffee Grounds Substrate for Tomato (Solanum lycopersicum) Fruiting Stage. *Horticulturae*, *9*(1), 89. https://doi.org/10.3390/horticulturae9010089

Picca, G., Plaza, C., Madejón, E., & Panettieri, M. (2022). Compositing of Coffee Silverskin with Carbon Rich Materials Leads to High Quality Soil Amendments. *Waste and Biomass Valorization*, *0123456789*. https://doi.org/10.1007/s12649-022-01879-7

Qiu, G. yu, Li, H. yong, Zhang, Q. tao, Chen, W., Liang, X. jian, & Li, X. ze. (2013). Effects of Evapotranspiration on Mitigation of Urban Temperature by Vegetation and Urban Agriculture. *Journal of*

Integrative Agriculture, *12*(8), 1307–1315. https://doi.org/10.1016/S2095-3119(13)60543-2

Rao, N., Patil, S., Singh, C., Roy, P., Pryor, C., Poonacha, P., & Genes, M. (2022). Cultivating sustainable and healthy cities: A systematic literature review of the outcomes of urban and peri-urban agriculture. In *Sustainable Cities and Society* (Vol. 85, p. 104063). Elsevier. https://doi.org/10.1016/j.scs.2022.104063

Raviv, M. (2014). Composts in growing media: Feedstocks, composting methods and potential applications. *Acta Horticulturae*, *1018*, 513–524. https://doi.org/10.17660/actahortic.2014.1018.56

Raviv, M. (2015). Can the use of composts and other organic amendments in horticulture help to mitigate climate change? *Acta Horticulturae*, *1076*, 19–28. https://doi.org/10.17660/ActaHortic.2015.1076.1

Richard Mulvaney. (1996). Nitrogen-Inorganic Forms. In *Methods of Soil Analysis. Part 3. Chemical Method* (pp. 1123–1184). Wiley.

Robertson-Fall, T. (2019). Cities and Circular Economy for Food. *Ellen MacArthur Foundation*, 23–44. https://www.ellenmacarthurfoundation.org/publications/cities-and-circular-economy-for-food

Rodríguez-Delfín, A., Gruda, N., Eigenbrod, C., Orsini, F., & Gianquinto, G. (2017). *Soil Based and Simplified Hydroponics Rooftop Gardens*. 61–81. https://doi.org/10.1007/978-3-319-57720-3_5

Rombel, A., Krasucka, P., & Oleszczuk, P. (2022). Sustainable biochar-based soil fertilizers and amendments as a new trend in biochar research. In *Science of the Total Environment* (Vol. 816, p. 151588). Elsevier. https://doi.org/10.1016/j.scitotenv.2021.151588

Ronga, D., Pane, C., Zaccardelli, M., & Pecchioni, N. (2016). Use of Spent Coffee Ground Compost in Peat-Based Growing Media for the Production of Basil and Tomato Potting Plants. *Communications in Soil Science and Plant Analysis*, *47*(3), 356–368. https://doi.org/10.1080/00103624.2015.1122803

Rossini-Oliva, S., & Nuñez, R. L. (2024). Is it healthy urban agriculture? Human exposure to potentially toxic elements in urban gardens from Andalusia, Spain. *Environmental Science and Pollution Research*, *31*(25), 36626–36642. https://doi.org/10.1007/S11356-024-33500-W/FIGURES/5

Rufí-Salís, M., Petit-Boix, A., Villalba, G., Gabarrell, X., & Leipold, S. (2021). Combining LCA and circularity assessments in complex production systems: the case of urban agriculture. *Resources, Conservation and Recycling*, *166*, 105359. https://doi.org/10.1016/J.RESCONREC.2020.105359

Sainju, U. M., Dris, R., & Singh, B. (2003). Mineral nutrition of tomato. *Journal of Food, Agriculture & Environment*, *1*(2), 176–183. www.world-food.net.

Sanchez-Monedero, M. A., Cayuela, M. L., Roig, A., Jindo, K., Mondini, C., & Bolan, N. (2018). Role of biochar as an additive in organic waste composting. *Bioresource Technology*, *247*, 1155–1164. https://doi.org/10.1016/J.BIORTECH.2017.09.193

Sánchez-Monedero, M. A., Roig, A., Cegarra, J., Bernal, M. P., Noguera, P., Abad, M., & Antón, A. (2004). Composts as media constituents for vegetable transplant production. *Compost Science and Utilization*, *12*(2), 161–168. https://doi.org/10.1080/1065657X.2004.10702175

Sánchez-Monedero, M. A., Sánchez-Garcıá, M., Castejón-Del Pino, R., Fornes, F., Belda, R. M., Lidón, A., & Cayuela, M. L. (2021). Biochar as an additive in composting: impact on process performance and on the agronomical quality of the end product. *Acta Horticulturae*, *1317*, 175–187. https://doi.org/10.17660/ActaHortic.2021.1317.21

Sanjuan-Delmás, D., Llorach-Massana, P., Nadal, A., Sanyé-Mengual, E., Petit-Boix, A., Ercilla-Montserrat, M., Cuerva, E., Rovira, M. R., Josa, A., Muñoz, P., Montero, J. I., Gabarrell, X., Rieradevall, J., & Pons, O. (2018). *Improving the Metabolism and Sustainability of Buildings and Cities Through Integrated Rooftop Greenhouses (i-RTG)* (pp. 53–72). https://doi.org/10.1007/978-3-319-67017-1_3

Santo, R., Palmer, A., Kim, B., Banks, K., Burns, C., Clancy, K., Havers, R., Milbourne, P., Nachman, K., & Winne, M. (2016). *A Review of the Benefits and Limitations of Urban Agriculture*.

Sanyé-Mengual, E., Oliver-Solà, J., Montero, J. I., & Rieradevall, J. (2015). An environmental and economic life cycle assessment of rooftop greenhouse (RTG) implementation in Barcelona, Spain. Assessing new forms of urban agriculture from the greenhouse structure to the final product level. *International Journal of Life Cycle Assessment*, *20*(3), 350–366. https://doi.org/10.1007/s11367-014-0836-9

Scazzosi, L. (2020). *Urban Agriculture as Heritage: Methodological Issues and Perspectives*. 17–44. https://doi.org/10.1007/978-3-030-49012-6_2

Schauer, N., Semel, Y., Roessner, U., Gur, A., Balbo, I., Carrari, F., Pleban, T., Perez-Melis, A., Bruedigam, C., Kopka, J., Willmitzer, L., Zamir, D., & Fernie, A. R. (2006). Comprehensive metabolic profiling and phenotyping of interspecific introgression lines for tomato improvement. *Nature Biotechnology*, *24*(4), 447–454. https://doi.org/10.1038/nbt1192

Schröder, C., Häfner, F., Larsen, O. C., & Krause, A. (2021). Urban organic waste for urban farming: Growing lettuce using vermicompost and thermophilic compost. *Agronomy*, *11*(6), 1175. https://doi.org/10.3390/agronomy11061175

Senes, G., Ferrario, P. S., Cirone, G., Fumagalli, N., Frattini, P., Sacchi, G., & Valè, G. (2021). Nature-based solutions for storm water management—creation of a green infrastructure suitability map as a tool for land-use planning at the municipal level in the province of Monza-Brianza (Italy). *Sustainability (Switzerland)*, *13*(11). https://doi.org/10.3390/su13116124

Seto, K. C., Parnell, S., & Elmqvist, T. (2013). A global outlook on urbanization. *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment*, 1–12. https://doi.org/10.1007/978-94-007-7088-1_1/FIGURES/2

Shafique, M., Xue, X., & Luo, X. (2020). An overview of carbon sequestration of green roofs in urban areas. *Urban Forestry & Urban Greening*, *47*, 126515. https://doi.org/10.1016/J.UFUG.2019.126515

Shamshiri, R. R., Jones, J. W., Thorp, K. R., Ahmad, D., Man, H. C., & Taheri, S. (2018). Review of optimum temperature, humidity, and vapour pressure deficit for microclimate evaluation and control in greenhouse cultivation of tomato: A review. In *International Agrophysics* (Vol. 32, Issue 2, pp. 287–302). Walter de Gruyter GmbH. https://doi.org/10.1515/intag-2017-0005

Shoemaker, C. A., & Diehl, E. R. M. (2012). The practice and profession of horticultural therapy in the United States. *Acta Horticulturae*, *954*, 161–164. https://doi.org/10.17660/ACTAHORTIC.2012.954.20

Sienkiewicz, A., Krasowska, M., Kowczyk-Sadowy, M., Obidziński, S., Piotrowska-Niczyporuk, A., & Bajguz, A. (2024). Occurrence of plant hormones in composts made from organic fraction of agri-food industry waste. *Scientific Reports*, *14*(1). https://doi.org/10.1038/s41598-024-57524-x

Simiele, M., Argentino, O., Baronti, S., Scippa, G. S., Chiatante, D., Terzaghi, M., & Montagnoli, A. (2022). Biochar Enhances Plant Growth, Fruit Yield, and Antioxidant Content of Cherry Tomato (Solanum lycopersicum L.) in a Soilless Substrate. *Agriculture (Switzerland)*, *12*(8). https://doi.org/10.3390/agriculture12081135

Skar, S. L. G., Pineda-Martos, R., Timpe, A., Pölling, B., Bohn, K., Külvik, M., Delgado, C., Pedras, C. M. G., Paço, T. A., Ćujic, M., Tzortzakis, N., Chrysargyris, A., Peticila, A., Alencikiene, G., Monsees, H., & Junge, R. (2020). Urban agriculture as a keystone contribution towards securing sustainable and healthy development for cities in the future. *Blue-Green Systems*, *2*(1), 1–27. https://doi.org/10.2166/BGS.2019.931

Smith, C. M., & Kurtz, H. E. (2003). Community Gardens and Politics of Scale in New York City. *Geographical Review*, *93*(2), 193–212. https://doi.org/10.1111/J.1931-0846.2003.TB00029.X

Specht, K., Siebert, R., Hartmann, I., Freisinger, U. B., Sawicka, M., Werner, A., Thomaier, S., Henckel, D., Walk, H., & Dierich, A. (2014a). Urban agriculture of the future: An overview of sustainability aspects of food production in and on buildings. *Agriculture and Human Values*, *31*(1), 33–51. https://doi.org/10.1007/s10460-013-9448-4

Specht, K., Siebert, R., Hartmann, I., Freisinger, U. B., Sawicka, M., Werner, A., Thomaier, S., Henckel, D., Walk, H., & Dierich, A. (2014b). Urban agriculture of the future: An overview of sustainability aspects of food production in and on buildings. *Agriculture and Human Values*, *31*(1), 33–51. https://doi.org/10.1007/S10460-013-9448-4/FIGURES/7

Thebo, A. L., Drechsel, P., & Lambin, E. F. (2014). Global assessment of urban and peri-urban agriculture: irrigated and rainfed croplands. *Environmental Research Letters*, *9*(11), 114002. https://doi.org/10.1088/1748-9326/9/11/114002

Tigist, M., Workneh, T. S., & Woldetsadik, K. (2013). Effects of variety on the quality of tomato stored under ambient conditions. *Journal of Food*

Science and Technology, *50*(3), 477. https://doi.org/10.1007/S13197-011-0378-0

Torres-Lima, P., Canabal-Cristiani, B., & Burela-Rueda, G. (1994). Urban sustainable agriculture: The paradox of the chinampa system in Mexico City. *Agriculture and Human Values*, *11*(1), 37–46. https://doi.org/10.1007/BF01534447/METRICS

United Nations. (2015). *Transforming our world: the 2030 Agenda for Sustainable Development.*

United Nations. (2018a). *United Nations Department of Economic and Social Affairs Population Division. The World's Cities in 2018 - Data Booklet.*

United Nations. (2018b). *World Urbanization Prospects.*

United Nations. (2024). *Ending Poverty | United Nations*. United Nations. https://www.un.org/en/global-issues/ending-poverty

U.S.D.A. (1991). *United States Standards for Grades of Fresh Tomatoes.*

Vaccari, F. P., Maienza, A., Miglietta, F., Baronti, S., Di Lonardo, S., Giagnoni, L., Lagomarsino, A., Pozzi, A., Pusceddu, E., Ranieri, R., Valboa, G., & Genesio, L. (2015). Biochar stimulates plant growth but not fruit yield of processing tomato in a fertile soil. *Agriculture, Ecosystems & Environment*, *207*, 163–170. https://doi.org/10.1016/J.AGEE.2015.04.015

Van Molle, L., & Segers, Y. (2008). Micro-farming on other men's land Allotments from the 19th to the 21st century: Belgian history in a global perspective. *Working Papers*, *23*. https://ideas.repec.org/p/hub/wpecon/200823.html

van Vliet, J., Eitelberg, D. A., & Verburg, P. H. (2017). A global analysis of land take in cropland areas and production displacement from urbanization. *Global Environmental Change*, *43*, 107–115. https://doi.org/10.1016/J.GLOENVCHA.2017.02.001

Verheijen, F. G. A., Graber, E. R., Ameloot, N., Bastos, A. C., Sohi, S., & Knicker, H. (2014). Biochars in soils: New insights and emerging research needs. In *European Journal of Soil Science* (Vol. 65, Issue 1, pp. 22–27). John Wiley & Sons, Ltd. https://doi.org/10.1111/ejss.12127

Walters, S. A., & Midden, K. S. (2018). Sustainability of Urban Agriculture: Vegetable Production on Green Roofs. *Agriculture 2018, Vol. 8, Page 168*, *8*(11), 168. https://doi.org/10.3390/AGRICULTURE8110168

Wang, Y., & Akdeniz, N. (2023). Utilizing co-composted biochar as a growing medium for buttercrunch lettuce. *Environmental Challenges*, *13*, 100748. https://doi.org/10.1016/J.ENVC.2023.100748

Waterhouse, A. L. (2003). Determination of Total Phenolics. In *Current Protocols in Food Analytical Chemistry*. https://doi.org/10.1002/0471142913.fai0101s06

Weidner, T., Yang, A., & Hamm, M. W. (2019). Consolidating the current knowledge on urban agriculture in productive urban food systems: Learnings, gaps and outlook. *Journal of Cleaner Production*, *209*, 1637–1655. https://doi.org/10.1016/J.JCLEPRO.2018.11.004

West, G. B. (2017). *Scale: the universal laws of life and death in organisms, cities and companies*. https://books.google.com/books/about/Scale.html?hl=es&id=PZjCoAEACAAJ

Wiskerke, J. S. C. (2015). Urban food systems. *Cities and Agriculture: Developing Resilient Urban Food Systems*, 1–25. https://doi.org/10.4324/9781315716312-7/URBAN-FOOD-SYSTEMS-JOHANNES-WISKERKE

Wong, J. W. C., Wang, X., & Selvam, A. (2017). Improving Compost Quality by Controlling Nitrogen Loss During Composting. In *Current Developments in Biotechnology and Bioengineering: Solid Waste Management* (pp. 59–82). Elsevier. https://doi.org/10.1016/B978-0-444-63664-5.00004-6

World Bank. (2024). *Poverty Overview: Development news, research, data*. https://www.worldbank.org/en/topic/poverty/overview#2

Wright, K. R. (2021). The masterful water engineers of machu picchu. *Water (Switzerland)*, *13*(21). https://doi.org/10.3390/w13213049

Xiang, Y., Deng, Q., Duan, H., & Guo, Y. (2017). Effects of biochar application on root traits: a meta-analysis. *GCB Bioenergy*, *9*(10), 1563–1572. https://doi.org/10.1111/gcbb.12449

Yan, Z., Eziz, A., Tian, D., Li, X., Hou, X., Peng, H., Han, W., Guo, Y., & Fang, J. (2019). Biomass allocation in response to nitrogen and phosphorus availability: Insight from experimental manipulations of Arabidopsis thaliana. *Frontiers in Plant Science*, *10*. https://doi.org/10.3389/FPLS.2019.00598/FULL

Yang, J., Mattoo, A. K., Liu, Y., Zvomuya, F., & He, H. (2023). Trade-offs of organic and organic-inorganic fertilizer combinations in tomato quality and yield: A global meta-analysis (1992–2021). *European Journal of Agronomy*, *151*, 126985. https://doi.org/10.1016/J.EJA.2023.126985

Yang, Y., & Guo, Y. (2018). Unraveling salt stress signaling in plants. *Journal of Integrative Plant Biology*, *60*(9), 796–804. https://doi.org/10.1111/JIPB.12689

Yang, Z., Muhayodin, F., Larsen, O. C., Miao, H., Xue, B., & Rotter, V. S. (2021). A review of composting process models of organic solid waste with a focus on the fates of C, N, P, and K. *Processes*, *9*(3). https://doi.org/10.3390/PR9030473

Yruela, I. (2005). Copper in plants. In *Brazilian Journal of Plant Physiology* (Vol. 17, Issue 1, pp. 145–156). Brazilian Journal of Plant Physiology. https://doi.org/10.1590/s1677-04202005000100012

Zambrano-Prado, P., Orsini, F., Rieradevall, J., Josa, A., & Gabarrell, X. (2021). Potential Key Factors, Policies, and Barriers for Rooftop Agriculture in EU Cities: Barcelona, Berlin, Bologna, and Paris. *Frontiers in Sustainable Food Systems*, *5*, 733040. https://doi.org/10.3389/FSUFS.2021.733040/BIBTEX

Zawadzińska, A., Salachna, P., Nowak, J. S., Kowalczyk, W., Piechocki, R., Łopusiewicz, Ł., & Pietrak, A. (2022). Compost based on pulp and paper mill sludge, fruit-vegetable waste, mushroom spent substrate and rye straw improves yield and nutritional value of tomato. *Agronomy*, *12*(1). https://doi.org/10.3390/agronomy12010013

Zhang, J., Chen, G., Sun, H., Zhou, S., & Zou, G. (2016). Straw biochar hastens organic matter degradation and produces nutrient-rich compost. *Bioresource Technology*, *200*, 876–883. https://doi.org/10.1016/j.biortech.2015.11.016

Ziss, E., Friesl-Hanl, W., Götzinger, S., Noller, C., Puschenreiter, M., Watzinger, A., & Hood-Nowotny, R. (2021). Exploring the potential risk of heavy metal pollution of edible cultivated plants in urban gardening contexts using a citizen science approach in the project "heavy metal city-zen". *Sustainability (Switzerland)*, *13*(15). https://doi.org/10.3390/su13158626

Zucconi, F., Forte, M., Monaco, A., & De Bertoldi, M. (1981). Biological evaluation of compost maturity. *BioCycle*, *22*(4), 27–29.

Annexes

Table A.1: Agronomical problems that occurred during the three years of the experiment and the corresponding interventions.

Problem	**Intervention**
Year I	
Onset of Blossom end Rot (BER).	Foliar spray of calcium deficiency corrector and adjustments to irrigation supply.
Spread of spider mite (*Tetranychus urticae*).	Application of the predatory mite *Phytoseuilus persimilis,* followed by successful treatment with paraffin oil.
Spread of green aphids (*Mizus Persicae*).	Application of larvae of the syrphid *Sphaerophoria rueppellii.*
Year II	
Spread of spider mite (*Tetranychus urticae*).	Foliar application of the acaricide abamectin.
Proliferation of early blight (*Alternaria Solani*).	Foliar application of the fungicide difenoconazole.
Year III	
Spread of whiteflies (*Trialeurodes vaporariorum*) and green aphids (*Mizus Persicae*).	Foliar application of neem oil and potassium soap.
Spread of spider mite (*Tetranychus urticae*).	Foliar application of the acaricide abamectin.
Proliferation of early blight (*Alternaria Solani*).	Foliar application of the fungicide azoxistrobin.

Table A.2: Pot weight installed on the MadreenRoof rooftop at the beginning of the experiment

Composting mixture	**Pot Weight (Kg)**
BC-CG	16.5
CG	15.4
BC-CS	16.6
CS	15.1
BC-SW	19.5
SW	17.9
Control	15.3
Control (saturated with water)	60.0

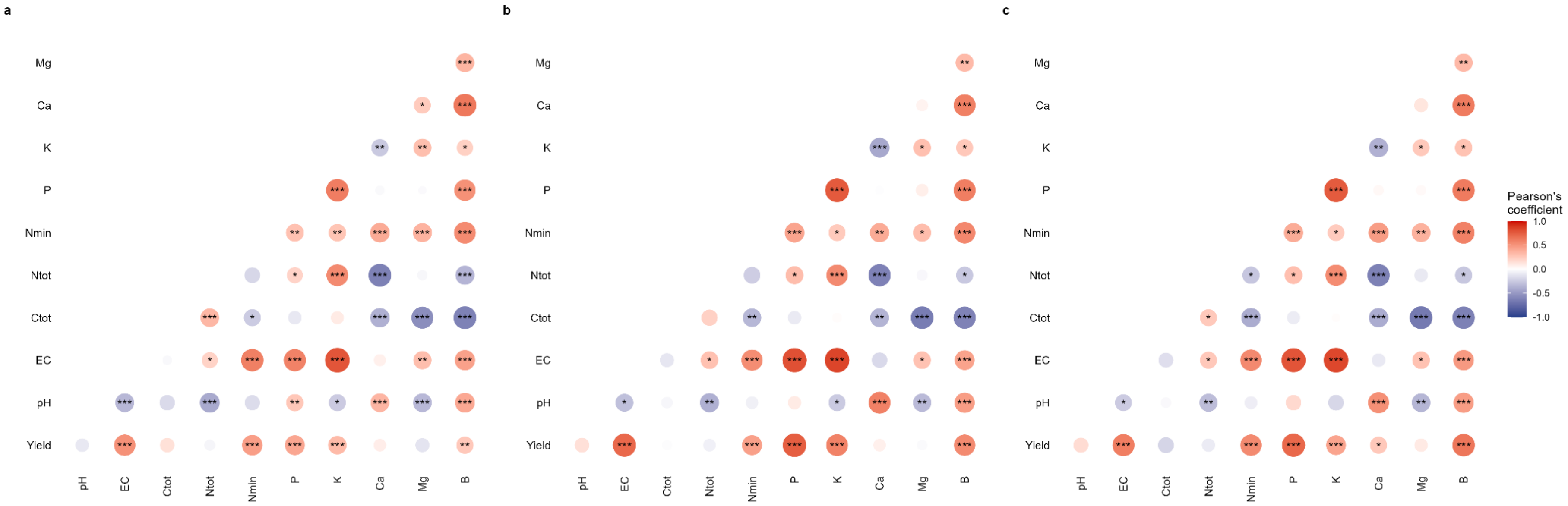
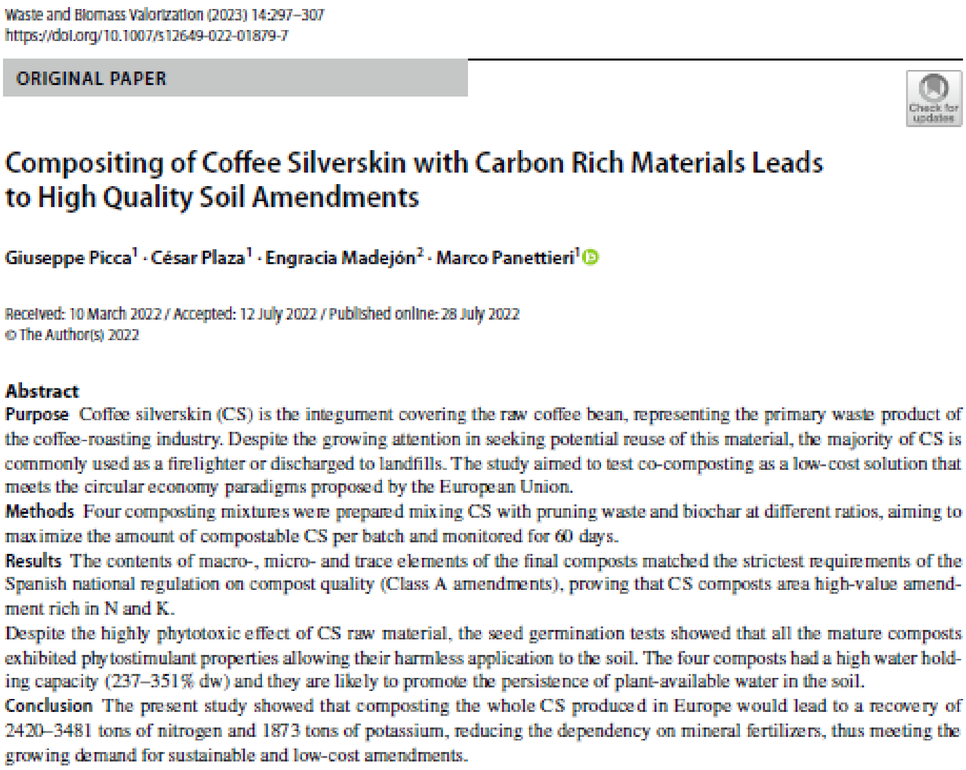


Figure A.1: Pearson rank correlation between substrate chemical properties and (a) Tomato yield, (b) Lettuce yield, and (c) Swiss Chard yield. Asterisks denote the significance of each pairwise correlation (*$p < 0.05$, **$p < 0.01$, ***$p < 0.001$



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ORIGINAL PAPER

Compositing of Coffee Silverskin with Carbon Rich Materials Leads to High Quality Soil Amendments

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Abstract

Purpose Coffee silverskin (CS) is the integument covering the raw coffee bean, representing the primary waste product of the coffee-roasting industry. Despite the growing attention in seeking potential reuse of this material, the majority of CS is commonly used as a firelighter or discharged to landfills. The study aimed to test co-composting as a low-cost solution that meets the circular economy paradigms proposed by the European Union.

Methods Four composting mixtures were prepared mixing CS with pruning waste and biochar at different ratios, aiming to maximize the amount of compostable CS per batch and monitored for 60 days.

Results The contents of macro-, micro- and trace elements of the final composts matched the strictest requirements of the Spanish national regulation on compost quality (Class A amendments), proving that CS composts area high-value amendment rich in N and K.

Despite the highly phytotoxic effect of CS raw material, the seed germination tests showed that all the mature composts exhibited phytostimulant properties allowing their harmless application to the soil. The four composts had a high water holding capacity (237–351% dw) and they are likely to promote the persistence of plant-available water in the soil.

Conclusion The present study showed that composting the whole CS produced in Europe would lead to a recovery of 2420–3481 tons of nitrogen and 1873 tons of potassium, reducing the dependency on mineral fertilizers, thus meeting the growing demand for sustainable and low-cost amendments.

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Article

Suitability of Co-Composted Biochar with Spent Coffee Grounds Substrate for Tomato (*Solanum lycopersicum*) Fruiting Stage

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Abstract: Peat is the predominant component of growing media in soilless horticultural systems. However, peat extraction from peatlands destroys these fragile ecosystems and emits greenhouse gas emissions (GHG). Peat replacement by other growing media is, thus, paramount to ensure a more sustainable horticultural sector. This study investigated the agronomical performances of two spent coffee ground-based composts with and without biochar, during three different stages of tomato (*Solanum lycopersicum* L.) development: seeds germination (0–6 days), seedling development (7–49 days), and plant-to-fruit maturity (36–100 days). The two composts were used as peat replacement and mixed with peat at four different volumetric proportions: 100% (pure compost), 50%, 30%, and 15%. The substrates had a stimulant effect on seed germination but induced stunted growth due to the elevated electrical conductivity. For the latest stages of plant development, compost with and without biochar mixed with peat at 50% promoted an increase in fruit production of 60.8% and 100.3%, compared to the control substrate. The present study provides evidence that combining biochar with spent coffee ground compost represents a potential alternative for peat-based growing media promoting a circular production model in the horticultural sector, but the results are dilution- and plant development stage-dependent.

Keywords: biochar; spent coffee grounds; compost; circular economy; peat replacement

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1. Introduction