Woodhead Publishing
Advances in Pollution Research

Recent Trends in Solid Waste Management

Edited by Balasubramani Ravindran, Sanjay Kumar Gupta,
Sartaj Ahmad Bhat, Puneet Singh Chauhan, and Neha Tyagi

ELSEVIER

Recent Trends in Solid Waste Management

Woodhead Publishing Advances in
Pollution Research

Recent Trends in Solid Waste Management

Edited by

Balasubramani Ravindran
Department of Environmental Energy and Engineering, Kyonggi
University Yeongtong-Gu, Suwon, Republic of Korea

Sanjay Kumar Gupta
Department of Civil Engineering, Indian Institute of Technology Delhi,
India

Sartaj Ahmad Bhat
River Basin Research Center, Gifu University, Gifu, Japan

Puneet Singh Chauhan
Microbial Technology Division, Council of Scientific and Industrial
Research-National Botanical Research Institute (CSIR-NBRI), Lucknow,
Uttar Pradesh, India

Neha Tyagi
Lead Consultant Rebound Envirotech Pvt Ltd, Dharamshala,
Himachal Pradesh, India

ELSEVIER
WP
WOODHEAD
PUBLISHING
An imprint of Elsevier

Woodhead Publishing is an imprint of Elsevier
50 Hampshire Street, 5th Floor, Cambridge, MA 02139, United States
The Boulevard, Langford Lane, Kidlington, OX5 1GB, United Kingdom

Copyright © 2023 Elsevier Inc. All rights reserved.

No part of this publication may be reproduced or transmitted in any form or by any means, electronic or mechanical, including photocopying, recording, or any information storage and retrieval system, without permission in writing from the publisher. Details on how to seek permission, further information about the Publisher's permissions policies and our arrangements with organizations such as the Copyright Clearance Center and the Copyright Licensing Agency, can be found at our website: www.elsevier.com/permissions.

This book and the individual contributions contained in it are protected under copyright by the Publisher (other than as may be noted herein).

Notices

Knowledge and best practice in this field are constantly changing. As new research and experience broaden our understanding, changes in research methods, professional practices, or medical treatment may become necessary.

Practitioners and researchers must always rely on their own experience and knowledge in evaluating and using any information, methods, compounds, or experiments described herein. In using such information or methods they should be mindful of their own safety and the safety of others, including parties for whom they have a professional responsibility.

To the fullest extent of the law, neither the Publisher nor the authors, contributors, or editors, assume any liability for any injury and/or damage to persons or property as a matter of products liability, negligence or otherwise, or from any use or operation of any methods, products, instructions, or ideas contained in the material herein.

ISBN: 978-0-443-15206-1

For Information on all Elsevier publications
visit our website at https://www.elsevier.com/books-and-journals

Publisher: Candice G. Janco
Acquisitions Editor: Jessica Mack
Editorial Project Manager: Ali Afzal-Khan
Production Project Manager: Rashmi Manoharan
Cover Designer: Mathew Limbert

Typeset by MPS Limited, Chennai, India

ELSEVIER
Book Aid International
Working together
to grow libraries in
developing countries
www.elsevier.com • www.bookaid.org

Contents

13. Biowaste valorisation in a circular economy 245

Wentao Chen, Yong Li, Qianlong Tan, Lingli Xie, Yuanyuan Hou, Wenxi Jiang, Dan Peipei, Haimei Wu, Hong Liu, Zekai Chen, Can Mao and Taimoor Hassan Farooq

14. Biogas production as a sustainable waste valorisation technology: perspectives from Namibia 259

Andreas S. Namwoonde, Pineas Tomas and Hupenyu A. Mupambwa

15. Environmental waste regarding soft foreign body (plastic) – a threat to livestock health 275

Taimoor Hassan Farooq, Vinod Bhateshwar, Awais Shakoor, Uttam Kumar, Norela C.T. Gonzalez, Dinesh Chandra Rai, V.P. Aparnna and Yong Li

16. Temple floral waste for various bio-products in India 293

Sachi Gupta, Shri Krishna Tewari and Sanjay Pathak

List of contributors

V.P. Aparnna Department of Dairy Science and Food Technology, Institute of Agricultural Sciences, Banaras Hindu University, Varanasi, Uttar Pradesh, India

Kolli Balakrishna Department of Chemistry, GIS, GITAM (Deemed to be University), Visakhapatnam, Andhra Pradesh, India

Balasubramani Ravindran Department of Environmental Energy and Engineering, Kyonggi University Yeongtong-Gu, Suwon, Republic of Korea

Sartaj Ahmad Bhat River Basin Research Center, Gifu University, Gifu, Japan

Vinod Bhateshwar Department of Dairy Science and Food Technology, Institute of Agricultural Sciences, Banaras Hindu University, Varanasi, Uttar Pradesh, India

María Gómez Brandón Grupo de Ecología Animal (GEA), University of Vigo, Vigo, Spain

Susanta Kumar Chakraborty Department of Zoology, Vidyasagar University, Midnapore, West Bengal, India

Puneet Singh Chauhan Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India; Academy of Scientific and Innovative Research (AcSIR), Ghaziabad, Uttar Pradesh, India

Wentao Chen College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

Zekai Chen College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

Subhaprada Dash Department of Agronomy, Institute of Agricultural Sciences, Siksha “O” Anusandhan (Deemed to be University), Bhubaneswar, Odisha, India

Randhi Uma Devi Department of Chemistry, School of Science, GITAM (Deemed to be University), Hyderabad, Telangana, India

Arabinda Dhal Department of Plant Pathology, College of Agriculture, Odisha University of Agriculture and Technology, Bhubaneswar, Odisha, India

Vijaykant Dixit Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India

Taimoor Hassan Farooq College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China

Andi Febrisiantosa Research Centre for Food Technology and Processing, National Research and Innovation Agency, Yogyakarta, Indonesia

Mishel Pulikondan Francis Research Scholar, Department of Botany, Bharathidasan University, Palkalaiperur, Tiruchirappalli, Tamil Nadu, India

Steffi Pulikondan Francis Department of Microbiology, Cauvery College for Women (Autonomous), Tiruchirappalli, Tamil Nadu, India

Gedda Gangaraju Department of Chemistry, Presidency University, Bangalore, Karnataka, India

Ram Kumar Ganguly Department of Zoology, Vidyasagar University, Midnapore, West Bengal, India

Norela C.T. Gonzalez College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China

Babett Greff Department of Food Science, Albert Casimir Faculty at Mosonmagyaróvár, Széchenyi István University, Mosonmagyaróvár, Hungary

Sachi Gupta Botanic Garden, PCAT, CSIR-National Botanical Research Institute, Lucknow, Uttar Pradesh, India

Yuanyuan Hou College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

Heribert Insam Department of Microbiology, University of Innsbruck, Innsbruck, Austria

Ishmael Festus Jaja Department of Livestock and Pasture Science, University of Fort Hare, Alice, South Africa; Department of Agriculture and Animal Health, University of South Africa, Johannesburg, South Africa

Wenxi Jiang College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

Marina Fernández-Delgado Juárez Norwegian University of Science and Technology (NTNU), Trondheim, Norway

Srishti Kar Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India

Jagdeep Kaur Department of Botanical and Environmental Sciences, Guru Nanak Dev University, Amritsar, Punjab, India

Pardeep Kaur Department of Botany, Khalsa College, Amritsar, Punjab, India

Shri Krishna Tewari Botanic Garden, PCAT, CSIR-National Botanical Research Institute, Lucknow, Uttar Pradesh, India

Kolli Suresh Kumar Department of Environmental Science, School of Science, GITAM (Deemed to be University), Visakhapatnam, Andhra Pradesh, India

Uttam Kumar College of Plant Protection, Fujain Agriculture and Forestry University, Fujian, China

Konathala Kusumavathi Department of Agronomy, Bidhan Chandra Krishi Viswavidyalaya, Mohanpur, West Bengal, India

Erika Lakatos Department of Food Science, Albert Casimir Faculty at Mosonmagyaróvár, Széchenyi István University, Mosonmagyaróvár, Hungary

Yong Li College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China

Hong Liu College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

Basudev Majhi Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India; Academy of Scientific and Innovative Research (AcSIR), Ghaziabad, Uttar Pradesh, India

Satyabrata Mangaraj Krishi Vigyan Kendra, Odisha University of Agriculture and Technology, Ganjam-I, Odisha, India

Thenmozhi Manivel Askoscen Probionics (R&D), Trichy, Tamil Nadu, India

Can Mao College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

P.F. Mishel Department of Botany, Bharathidasan University, Trichy, Tamil Nadu, India

Shashank Kumar Mishra Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India; Academy of Scientific and Innovative Research (AcSIR), Ghaziabad, Uttar Pradesh, India

Sankalp Misra Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India; Academy of Scientific and Innovative Research (AcSIR), Ghaziabad, Uttar Pradesh, India

Hupenyu A. Mupambwa Sam Nujoma Marine and Coastal Resources Research Centre, Sam Nujoma Campus, University of Namibia, Henties Bay, Namibia

Karnan Muthusamy Grassland and Forage Division, National Institute of Animal Science, Rural Development Administration, Chungnam-do, South Korea

Ágnes Nagy Department of Food Science, Albert Casimir Faculty at Mosonmagyaróvár, Széchenyi István University, Mosonmagyaróvár, Hungary

Andreas S. Namwoonde Sam Nujoma Marine and Coastal Resources Research Centre, Sam Nujoma Campus, University of Namibia, Henties Bay, Namibia

Adornis D. Nciizah Agricultural Research Council—Natural Resources & Engineering, Pretoria, South Africa; Department of Agriculture and Animal Health, University of South Africa, Roodepoort, South Africa

Patrick Nyambo Risk and Vulnerability Science Centre, Faculty of Science and Agriculture, University of Fort Hare, Alice, South Africa

James Wabwire Oguttu Department of Agriculture and Animal Health, University of South Africa, Johannesburg, South Africa

Kshitipati Padhan Department of Soil Science and Agricultural Chemistry, College of Agriculture, Odisha University of Agriculture and Technology, Bhubaneswar, Odisha, India

Narayan Panda Department of Soil Science and Agricultural Chemistry, College of Agriculture, Odisha University of Agriculture and Technology, Bhubaneswar, Odisha, India

Deen Dayal Pandey Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India

Sanjay Pathak College of Horticulture and Forestry, Acharya Narendra Deva University of Agriculture and Technology, Kumarganj, Uttar Pradesh, India

Sushanata Kumar Pattanayak Department of Soil Science and Agricultural Chemistry, College of Agriculture, Odisha University of Agriculture and Technology, Bhubaneswar, Odisha, India

Dan Peipei College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

Miklós Posgay Wittmann Antal Multidisciplinary Doctoral School in Plant, Animal, and Food Sciences, Széchenyi István University, Mosonmagyaróvár, Hungary

K. Prasanna Department of Civil Engineering, College of Engineering and Technology, SRM Institute of Science and Technology, Kattankulathur, Tamil Nadu, India

Maraike Probst Department of Microbiology, University of Innsbruck, Innsbruck, Austria

Dinesh Chandra Rai Department of Dairy Science and Food Technology, Institute of Agricultural Sciences, Banaras Hindu University, Varanasi, Uttar Pradesh, India

Robin Agilent Technologies India Pvt. Ltd. & Regional Water Testing Laboratory, Department of Water Supply and Sanitation, Government of Punjab, Amritsar, Punjab, India

András Sáhó Wittmann Antal Multidisciplinary Doctoral School in Plant, Animal, and Food Sciences, Széchenyi István University, Mosonmagyaróvár, Hungary; Kisalföldi Agricultural Ltd, Nagyszentjános, Hungary

Tapas Ranjan Sahoo Krishi Vigyan Kendra, Odisha University of Agriculture and Technology, Kendrapara, Odisha, India

V.M.V. Sai Krishna Department of Civil Engineering, College of Engineering and Technology, SRM Institute of Science and Technology, Kattankulathur, Tamil Nadu, India

Smritikana Sarkar Department of Agronomy, Bidhan Chandra Krishi Viswavidyalaya, Mohanpur, West Bengal, India

Vijayalakshmi Selvakumar Food Science and Biotechnology, School of Agriculture and Lifesciences, Kangwon National University, Chuncheon, Gangwon-do, Republic of Korea

Pradeep Semwal Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India

Debadatta Sethi Department of Soil Science and Agricultural Chemistry, College of Agriculture, Odisha University of Agriculture and Technology, Bhubaneswar, Odisha, India; Regional Research and Technology Transfer Station, Odisha University of Agriculture and Technology, G. Udayagiri, Kandhamal, Odisha, India

Awais Shakoor Environment, Soils and Land Use Department, Teagasc, Johnstown Castle, Wexford, Ireland

Huijuan Shao College of Resources and Environment, Shandong Agricultural University, Tai'an, P.R. China

R. Shiam Babu Department of Civil Engineering, College of Engineering and Technology, SRM Institute of Science and Technology, Kattankulathur, Tamil Nadu, India

Tanya Singh Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India; Academy of Scientific and Innovative Research (AcSIR), Ghaziabad, Uttar Pradesh, India

Mhlangabezi Slayi Risk and Vulnerability Science Centre, University of Fort Hare, Alice, South Africa

P.F. Steffi Department of Microbiology, Cauvery College for Women (Autonomous), Trichy, Tamil Nadu, India

J.S. Sudarsan School of Energy and Environment, NICMAR University, Pune, Maharashtra, India

Susanta Kumar Swain Directorate of Research, Odisha University of Agriculture and Technology, Bhubaneswar, Odisha, India

Qianlong Tan College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

Pineas Tomas Multidisciplinary Research; Center for Research Services, Main Campus, University of Namibia, Windhoek, Namibia

Haimei Wu College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

Lingli Xie College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China; National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

Leocadia Zhou Risk and Vulnerability Science Centre, Faculty of Science and Agriculture, University of Fort Hare, Alice, South Africa

Chapter 1

Nutrient recycling of fly ashes from fast pyrolysis as an innovative treatment for organic waste

María Gómez Brandón[1], Maraike Probst[2], Heribert Insam[2] and Marina Fernández-Delgado Juárez[3]
[1]Grupo de Ecología Animal (GEA), University of Vigo, Vigo, Spain, [2]Department of Microbiology, University of Innsbruck, Innsbruck, Austria, [3]Norwegian University of Science and Technology (NTNU), Trondheim, Norway

1.1 Introduction

A growing interest has recently arisen in the use of biomass for heat and energy production in response to the energy targets established by the European Union, aiming to replace fossil fuels with renewable energy sources (AEBIOM, 2017). Besides combustion and gasification, there exists a more recent process known as fast pyrolysis in which biomass, such as wood, straw and energy crops, can be used for energy production (Bridgewater, 2012; Krutof and Hawboldt, 2018). Fast pyrolysis comprises rapid heating of the input material to 400 °C−600 °C under anaerobic conditions (Fig. 1.1). This temperature enables the breakdown of the biomass structure devoid of melting of the inorganic elements (Leijenhorst et al., 2016). The subsequent vapours produced during this process are cooled and condensed into a brown liquid called ‘Fast Pyrolysis Bio-Oil (FPBO)’ that can be used for heating, power generation and as a substitute in conventional diesel engines (Van de Beld et al., 2013; Lehto et al., 2014). Typically, 50−75wt.% of the dry biomass can be converted into FPBO. Other streams include low calorific gases and char (Fig. 1.1), which can be processed for electricity and heat generation (Lohri et al., 2016), with the resulting production of biomass fly ashes (FAs, Fig. 1.1). To assure the sustainability of FPBO technology it is of utmost importance to properly manage the FPBO-FAs so as

Recent Trends in Solid Waste Management. **DOI:** https://doi.org/10.1016/B978-0-443-15206-1.00015-3

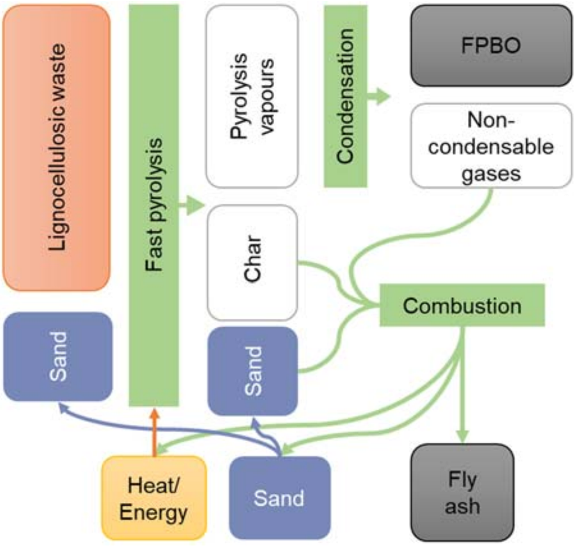


FIGURE 1.1 Conceptual scheme of the fast pyrolysis process.

to avoid potential negative environmental impacts (Zhong et al., 2010; Fernández-Delgado Juárez et al., 2020; Kurzemann et al., 2021).

The disposal of ashes in landfills has been used as a common practice, but it implies considerable costs for biomass plant operators and negates the recycling potential of ashes. This raises the necessity to search for more profitable and sustainable options to mark a shift in the mindset towards biomass FAs as an unwanted burden to a valued resource, promoting waste prevention and new recycling goals. In this regard, the use of FAs as a nutrient supplement in agriculture and/or the forestry sector seems to be an option for their recycling (Knapp and Insam, 2011; Fernández-Delgado Juárez et al., 2015; Kurzemann et al., 2021), helping to both counteract acidification and correct nutrient deficiency in the soil.

On the one hand, the buffering capacity of FAs is an important asset in overcoming potential negative effects on soils and plants. The pH rise induced by ash addition may help to reduce the solubility of heavy metals (HMs) present in the biomass FAs when applied to soil (Dimitriou et al., 2006; Fernández-Delgado Juárez et al., 2020). On the other hand, biomass FAs are rich in macronutrients, such as Ca, Mg, K and P, and micronutrients like Fe and Mn that are essential for plant development (Knapp and Insam, 2011; Bougnom et al., 2012; Fernández-Delgado Juárez et al., 2013, 2020). Their deficit in C and N, which are mostly volatilised during the combustion of the biomass, can be alleviated by combining the ashes with organic materials, including compost or manure (Bougnom et al., 2010; Fernández-Delgado Juárez et al., 2013; Ibeto et al., 2020), or with mineral fertilizers

(Saarsalmi et al., 2006; Knapp and Insam, 2011). The potential benefits and drawbacks of FAs on soil properties will be inextricably linked to the original biomass source and the development of the entire process chain from the waste to the ash production (Kurzemann et al., 2021). All in all, it will determine the FAs' properties in terms of both nutrients and toxic elements, and also their behaviour once applied to soil (Li et al., 2012; Chen et al., 2016; Lanzerstorfer, 2017, 2019). This latter aspect may also vary depending on the dosage and the form of application of FAs, that is if they are incorporated or applied to the soil surface (Ou et al., 2020).

Following this rationale, the purpose of this chapter is to provide a comprehensive characterisation of FPBO-FAs derived from different waste streams and an overview of their potential effects on soil properties and plant growth. Nitrogen deficiency is often a limiting factor for microbial and plant growth, and in particular, in natural ecosystems, the availability of ammonia and nitrate is still considered a bottleneck for the activity of most of the organisms present, ranging from microorganisms to plants (Lehtovirta-Morley, 2018). Bearing this in mind, we will present a case study focused on the impact of FAs recovered from the fast pyrolysis on nitrification and N-mineralisation rates, and on key microbial groups involved in the N cycle. Finally, we also address legal aspects related to the use of ashes, and in particular FPBO-FAs, in the fields of waste management and fertilizer production.

1.2 Characterisation of FPBO-ashes from different waste streams

Combustion typically generates three types of biomass ashes: bottom ash, fly ash and a combination of both, known as mixed ash. The properties of each fraction vary largely, especially in terms of their particle size and their potential pollutant content (Ingerslev et al., 2011). In the FPBO production process, only FAs are produced (Fig. 1.1; Leijenhorst et al., 2016), and their properties might not necessarily match with those of FAs derived from other processes of biomass combustion. Besides the initial feedstock, the properties of the biomass ashes are directly related to the combustion temperature and boiler size (Dahl et al., 2009; Vassilev et al., 2013a, b; Pugliese et al., 2014).

For soil application, not only are the pH and the macronutrient content of the FPBO-FAs of relevance, but also their potential phytotoxic effect on plants and their content in HMs must be considered. In fact, the presence of HMs could promote the absorption of polycyclic aromatic hydrocarbons (PAHs) and catalyse their formation in FAs (Wey et al., 1998).

1.2.1 Physico-chemical properties of FPBO-FAs

In Table 1.1 we give an overview of the FPBO-FAs derived from five different waste streams, including crumbled clean pine wood, bark, *Miscanthus* sp.,

TABLE 1.1 **Elemental composition of fly ashes derived from the fast pyrolysis process (FPBO-FAs) of crumbled clean pine wood, bark, *Miscanthus* sp., wheat straw and forest residues.**

Properties	Unit	FPBO-FAs					FAs – literature[a]
		Pine wood	Bark	*Miscanthus* sp.	Wheat straw	Forest residues	
Dry matter	%	99.7 ± 0.27	100.0 ± 0.07	100.8 ± 0.28	100.4 ± 0.00	100.2 ± 0.10	99.8–40
pH		12.5 ± 0.03	12.0 ± 0.01	11.5 ± 0.06	12.2 ± 0.06	12.8 ± 0.04	
Electrical conductivity	mS cm^{-1}	19.1 ± 0.20	0.6 ± 0.02	1.4 ± 0.06	3.7 ± 0.23	4.8 ± 0.25	
Total organic carbon	(%)	n.m	2.02 ± 0.14	1.39 ± 0.17	17.9 ± 0.03	8.73 ± 0.53	1.6–32.8
Ca	g kg^{-1}	122.2 ±1.86	43 ± 1.4	20 ± 3.07	156.8 ± 0.07	240 ± 11.5	104–280
K	g kg^{-1}	51.1 ± 0.94	212.0 ± 0.77	49 ± 7.4	81 ± 1.2	61 ± 3.4	40–160
Mg	g kg^{-1}	32.8 ± 1.14	3.14 ± 0.063	3.77 ± 0.084	16.2 ± 0.13	17.2 ± 0.37	19.3–50
P	g kg^{-1}	9.1 ±0.09	1.80 ± 0.064	2.68 ± 0.082	24.7 ± 0.77	19.2 ± 0.66	4.3–22.9
Zn	g kg^{-1}	2.0 ± 0.07	0.242 ± 0.0034	0.137 ± 0.0028	1.22 ± 0.018	1.17 ± 0.028	0.37–40
Mn	mg kg^{-1}	n.m.	1464 ± 23	290 ± 8.7	7244 ± 45	11885 ± 326	1300–30300
As	mg kg^{-1}	5.5 ± 0.98	< 1.2	< 1.2	< 1.2	< 1.2	1.5–24
Ni	mg kg^{-1}	54.0 ± 3.3	320 ± 11	80 ± 0.4	690 ± 2.0	385 ± 6.6	19–74
Cd	mg kg^{-1}	9.4 ± 0.27	2.5 ± 0.16	1.8 ± 0.09	2.1 ± 0.01	3.2 ± 0.10	5.1–34
Cr	mg kg^{-1}	170 ± 7.2	570 ± 29	150 ± 10	1247 ± 1.6	730 ± 22	26.5–290

Cu	mg kg^{-1}	550 ± 10.5	34 ± 2.8	29.0 ± 0.05	102 ± 1.9	55.0 ± 0.94	106–1100
Pb	mg kg^{-1}	230 ± 3.9	19 ± 0.6	15 ± 1.4	16 ± 0.1	25 ± 1.3	10.7–470
V	mg kg^{-1}	n.m	16 ± 0.2	7 ± 0.1	18 ± 0.1	17 ± 0.8	5.1–43
Co	mg kg^{-1}	n.m	5.83 ± 0.23	<1.66	7.94 ± 0.10	7.45 ± 0.42	5.79–13
Mo	mg kg^{-1}	n.m	9 ± 0.4	2 ± 0.1	19 ± 0.3	13 ± 0.2	1.46–16
Hg	mg kg^{-1}	n.m	6 ± 0.8	0.6 ± 0.1	29 ± 0.8	46 ± 0.4	1.7

n.m, Not measured. Values are expressed on a dry mass basis ($n = 3$, average ± standard deviation).

[a]*Comparison with reference values of FAs found in literature (Maresca et al., 2017).*

wheat straw and forest residues. All FPBO-FAs were characterised by an alkaline pH (around 12), showing their potential as lime replacement in acidic soils. The alkalinity of FAs is associated with the CaO content and/or the CaO to SO_4 ratio (Ram and Masto, 2014), and their liming effect is well documented (Schönegger et al., 2018). Fly ashes are also rich in soluble salts, which could explain the high electrical conductivity (EC) levels of the FPBO-FAs obtained from clean crumbled pine wood (19.1 mS cm^{-1}; Table 1.1). The other four FPBO-FAs were characterised by a more reduced salt concentration as indicated by their lower EC values, ranging from 0.5 to 5 mS cm^{-1} (bark < *Miscanthus* sp. < wheat straw < forest residues; Table 1.1).

Furthermore, biomass-derived FAs may act as a direct source of elements such as P, Ca, Mg and K (Maresca et al., 2017) and micronutrients (Pan and Eberhardt, 2011). In agreement with previously characterised FAs, the FPBO-FAs studied here presented high amounts of macro-nutrients and micronutrients and their respective content varied with the biomass feedstock (Table 1.1).

1.2.2 Heavy metal and polycyclic aromatic hydrocarbons in FPBO-FAs

Certain HMs recognised as being pollutants (e.g. Pb, Zn, Cu, Cr, Cd, Ni and As) were present in all of the studied FPBO-FAs (Table 1.1), and their content mostly fell within the range of other biomass combustion-derived FAs (Table 1.1; Maresca et al., 2017). From a legal perspective, we observed that with regards to the FPBO-FAs derived from crumbled clean pine wood the contents of Cd, Cu, Pb and Zn slightly exceeded the threshold values established by the Austrian Compost Ordinance (BMLFUW, 2001) and the Guidelines for the use of biomass ash in Austria (BMLFUW, 2011). The same occurred for the elements Ni and Cr, the levels of which were above the limit values stated by legislation in the other four FPBO-FAs, with the exception of those obtained from *Miscanthus* sp. (Tables 1.1 and 1.2). The mobility and potential bioavailability of these two HMs were also assessed in the resulting leachates. We found that the content of Ni was below the detection limit in the leachates from the different biomass feedstocks, except for wheat straw (0.7 ± 0.03 mg kg^{-1}). In the case of Cr, the respective values were generally higher than Ni (*Miscanthus* sp.: 0.7 ± 0.05 mg kg^{-1}; bark: 4.2 ± 0.5 mg kg^{-1}; wheat straw: 10.7 ± 0.2 mg kg^{-1}; forest residues: 51.1 ± 2.1 mg kg^{-1}), but not proportional to the Cr content of the initial ashes.

HMs can also be present in specific chemical forms that largely influence their mobility and bioavailability in soil (Pan and Eberhardt, 2011). This is the case for the hexavalent form of chromium, known as Cr (VI), which is highly toxic and has been classified as a human carcinogen by several regulatory and non-regulatory agencies (Tchounwou et al., 2012). In this regard, we found that Cr (VI) was present in the studied FPBO-FAs, reaching higher

TABLE 1.2 Maximum heavy metal concentrations for ash utilisation according to the national legislation in Austria, Finland, Germany, and the Netherlands.

Limits and concentrations [mg kg^{-1}]	Austria plant ash		Austria compost	Finnish ash fertilizer		German	Dutch fertilizer			FPBO-FA examples			
Element	A	B		Horticulture and agriculture	Forest	Ash evaluation	P_2O_5	N	K_2O	Bark	*Miscanthus sp.*	Wheat straw	Forest residues
Zn	1,200	1,500	1,500	1,500	4,500		7,500	6,000	4,000	242 ± 3.4	137 ± 2.8	1221 ± 18	1168 ± 28
Cu	200	250	250	600	700		1,875	1,500	1,000	34 ± 2.81	29 ± 0.05	102 ± 1.92	55 ± 0.94
Cr	150	250	250	300	300		1,875	1,500	1,000	574 ± 29	151 ± 10	1247 ± 1.6	734 ± 22
CrIV						2				5.2 ± 0.57	1.2 ± 14	11.5 ± 0.71	41.0 ± 5.6
Pb	100	200	100	100	150	150	2,500	2,00	1,333	19 ± 0.6	15 ± 1.4	16 ± 0.1	25 ± 1.3
Ni	150	200	100	100	150	80	750	600	400	317 ± 11	80 ± 0.4	688 ± 2	385 ± 6.6

(*Continued*)

TABLE 1.2 (Continued)

Limits and concentrations [mg kg^{-1}]	Austria plant ash		Austria compost	Finnish ash fertilizer		German	Dutch fertilizer			FPBO-FA examples			
Element	A	B		Horticulture and agriculture	Forest	Ash evaluation	P_2O_5	N	K_2O	Bark	*Miscanthus sp.*	Wheat straw	Forest residues
Cd	5	8	8	1.5	25	1.5	31.3	25	16.7	2.45 ± 0.16	1.77 ± 0.09	2.10 ± 0.01	3.20 ± 0.10
As	20	20	20	25	40	40	375	300	200	b.d.l.	b.d.l.	b.d.l.	b.d.l.
V			100			1				16 ± 0.2	7 ± 0.1	18 ± 0.1	17 ± 0.8
Co			100							5.83 ± 0.23	<1.66	7.94 ± 0.10	7.45 ± 0.42
Mo			20							9 ± 0.4	2 ± 0.1	19 ± 0.3	13 ± 0.2
Hg				1	1	1	18.8	15	10	6 ± 0.8	0.6 ± 0.1	29 ± 0.8	46 ± 0.4

b.d.l., Below detection limit.

values in those FAs obtained from the fast pyrolysis of forest residues (Table 1.3).

PAHs constitute another group of critical pollutants that might be present in biomass ashes. Of all known PAHs, 16 are listed as priority pollutants by USEPA (2013), and can be found in high levels in FAs due to poor combustion conditions (Masto et al., 2015; Rey-Salgueiro et al., 2016). Benzene, toluene, ethylbenzene, the *ortho*, *para* and *meta* xylenes and styrene are considered petroleum-derived volatile organic compounds, and the knowledge about their presence and concentration in FPBO-FAs is of high environmental relevance.

In general, PAHs are formed when hydrocarbon compounds undergo incomplete combustion, and their quantity is mainly affected by the physical (i.e. particle size, adsorptive surface area) and chemical (i.e. carbon and metal compounds) properties of the FA (Wey et al., 1998). The thinner the particles in the FAs, the higher the specific surface area, and in turn, more condensed PAHs can be absorbed in the ash (Rey-Salgueiro et al., 2016). Nonetheless, all of the tested FPBO-FAs showed very low PAH levels (Table 1.2) according to both WHO (BMFLUW, 2004) and USEPA (2013). This is an indication of a good combustion process independent of the biomass feedstock used in the fast pyrolysis process.

1.2.3 Phytotoxicity

Standard ecotoxicological tests on the ashes' leachates are often performed to assess the potential hazardous effects of combustion ashes. The evaluation of the toxic effects on small aquatic organisms, such as *Daphnia magna* has been considered for such purpose (Lapa et al., 2007; Tsiridis et al., 2012). Nevertheless, Römbke et al. (2009) indicated that the application of ecotoxicological tests based on terrestrial rather than aquatic organisms is more sensitive and reliable for measuring the potential toxicological effects of biomass ashes. Plant tests with *Lepidium sativum* as a reference organism have been widely used in this context (Kuba et al., 2008; Oleszczuk et al., 2012; Fernández-Delgado Juárez et al., 2018).

In this regard, we observed that the application of the non-diluted extracts derived from the FPBO-FAs of crumbled pine wood and forest residues had a negative effect on seed germination (Fig. 1.2A) when compared to the control with distilled water and the other biomass feedstocks. This phytotoxic effect was substantially reduced by dilution (Fig. 1.2A), thereby reaching germination rates higher than the threshold value of 40% defined by Zucconi et al. (1981).

When considering the root development to evaluate ash phytotoxicity, all of the tested FPBO-FAs showed a high toxicity index on the concentrated extracts (Fig. 1.2B). As occurred with seed germination, the toxicity index diminished with increasing ash dilution rates (Fig. 1.2B).

TABLE 1.3 Content of polycyclic aromatic hydrocarbons (PAHs) in fly ashes derived from the fast pyrolysis process (FPBO-FAs) of different biomass feedstocks.

FPBO-FAs	Acenaphthene	Acenaphthylene	Anthracene	Benzo [a] anthracene	Benzo [b + k] fluoranthene*	Benzo [g,h,i] perylene*	Benzo [a] pyrene*	Chrysene	Dibenzo [a,h] anthracene	Naphthalene	Fluoranthene*	Fluorene	Indeno [1,2,3-cd] pyrene*	Phenanthrene	Pyrene	Σ PAHs (16 n. EPA)	Σ PAHs (6 n. WHO)
Bark	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	0.025	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	0.025	0
Miscanthus sp.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	0.066	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	0.066	0
Wheat straw	b.d.l.	b.d.l.	0.057	b.d.l.	0.041	0.029	b.d.l.	b.d.l.	b.d.l.	0.032	0.056	b.d.l.	0.022	b.d.l.	0.021	0.25	0.148
Forest residues	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	b.d.l.	0.13	b.d.l.	b.d.l.	b.d.l.	0.048	b.d.l.	0.18	0

b.d.l., Below detection limit (0.020 mg kg^{-1} for all the PAHs and 0.040 mg kg^{-1} for Benzo [b + k] fluoranthene). Values are expressed as mg kg^{-1} on a dry mass basis.
**PAHs considered by the World Health Organization (WHO) according to BMLFUW (2004).*

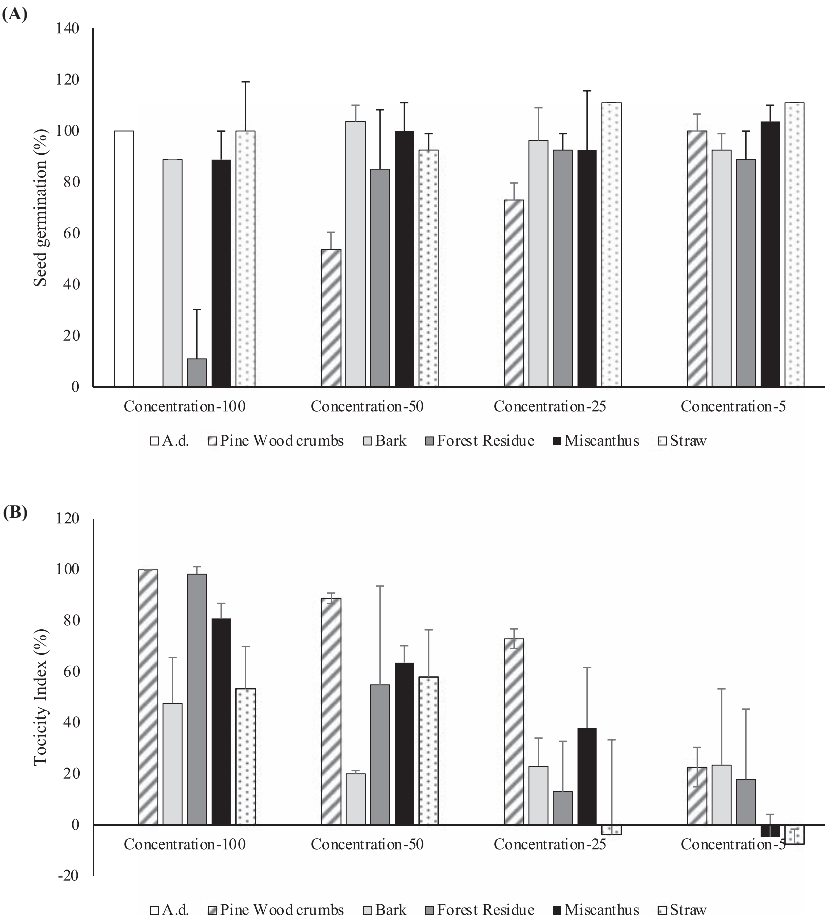


FIGURE 1.2 Germination percentage of *Lepidum sativum* seeds (A) and toxicity index (B) in the ash extracts at different concentrations (100%, 50%, 25% and 5%). Values are means ± standard deviation ($n = 3$). Note: for bark, forest residues, *Miscanthus* sp., and wheat straw 'concentration 5' represents 6.25% of the extract.

1.3 Effect of FPBO-ashes on soil properties and plant yield: a case study

The effect of FPBO-FAs on soil chemical and biological properties has been researched in several studies conducted by the Department of Microbiology at the University of Innsbruck. Their aim was to shed light on the potential usefulness of FPBO-ashes as a soil amendment (Schönegger et al., 2018; Fernández-Delgado Juárez et al., 2020; Kurzemann et al., 2021). Nonetheless, information is still scarce and based on the existing knowledge it cannot be concluded if FPBO-FA is a safe, environmentally friendly and a beneficial soil amendment. In this section, we will present the core findings of a soil mesocosm experiment, with special emphasis on the N-cycle, in which we tested the effects of FAs derived from the fast pyrolysis of ‘crumbled’ clean pine wood ($pH_{ash} = 12.5 \pm 0.03$) on an acidic grassland soil ($pH_{soil} = 6.2 \pm 0.05$). FPBO-FAs were mixed with the soil columns by turning at a rate equivalent to 2% (w/w, fresh weight). This amount is equivalent to 100 kg of ash per ha and year, which is the dose recommended for agricultural soils according to the guidelines for the use of biomass ash in Austria (BMLFUW, 2011). A control treatment in the absence of ashes was also included. Ten seeds of a traditional wheat variety (Tiroler Früher Dinkel; *Triticum aestivum* subsp. *spelta*) were spread in half of the columns with and without ashes in order to study the effect of the ashes on seed germination and plant growth. All soil columns were arranged in triplicate in a randomised block design under greenhouse conditions and destructively sampled at the beginning of the trial and after 60 and 100 days.

1.3.1 Effect of FPBO-FAs on phosphorus and nitrogen cycles

In the work by Schönegger et al. (2018), we found that, at the mesocosm scale, FPBO-FAs represent a viable alternative to mineral phosphorus fertilizers by leading to higher soil P-pools (total, inorganic and plant-available P) after 60 days; however, from a microbiological perspective, neither the abundance of phosphatase harbouring bacterial communities nor the respective enzymatic activities were influenced by the FPBO-FAs application. These findings provide evidence that the FAs derived from FPBO production improve soil nutrient status by helping to sustain the phosphorus levels in the mid-term without causing an effect on soil microbial communities involved in the P cycle.

The addition of biomass ashes may also induce soil mineralisation processes (Odlare and Pell, 2009), and result in increases in organic and inorganic N forms (Saarsalmi et 2012; Fernández-Delgado Juárez et al., 2015). At the mesocosm scale, we, however, did not observe significant changes in N mineralisation following the FPBO-FAs addition, irrespective of the sampling time and the presence of plants (Fig. 1.3B). As earlier reported by Ring et al. (2006), N mineralisation appears not to be affected by ash addition in soils with low N availability. Contrarily, amending soil with the FPBO-FAs led to a pronounced increase in the potential nitrification rate in the presence and the absence of plants (Fig. 1.3A), particularly after 60 and 100 days of incubation.

Ammonia-oxidising bacteria and archaea (AOB and AOA) are responsible for the first and limiting step of nitrification by converting ammonia to nitrite (Lehtovirta-Morley, 2018). In agreement with the increased nitrification rate, the addition of FPBO-FAs was accompanied by a higher AOB abundance (gene copy number; real-time PCR-based) on days 60 and 100,

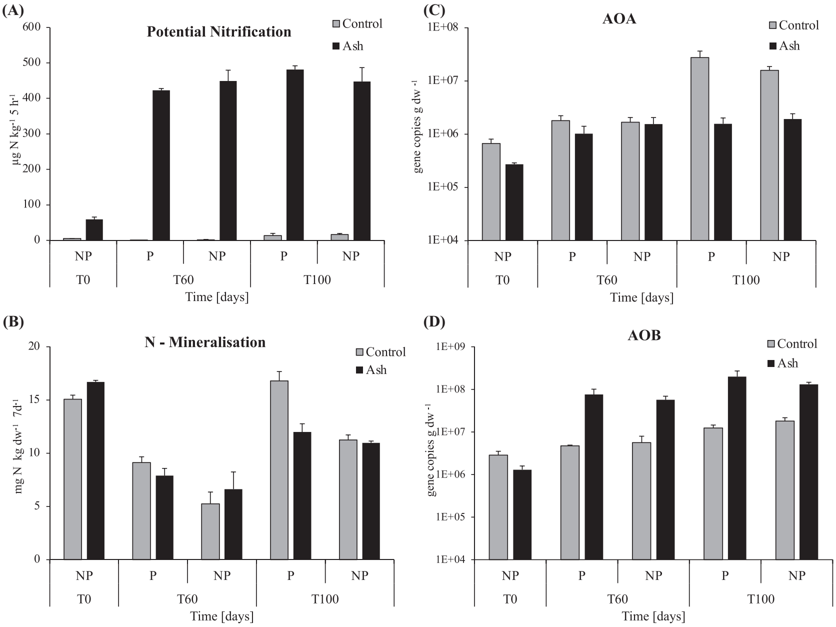


FIGURE 1.3 Potential nitrification (A), N-mineralisation (B), and abundance of ammonia-oxidising archaea (AOA; C) and bacteria (AOB; D) in the control and the ash-treated soils at the three incubation times (0, 60 and 100 days) in the presence and the absence of plants (P and NP, respectively). Values are means ($n = 3$) with the standard deviation and are expressed on a dry weight basis.

regardless of plant presence (Fig. 1.3D); however, the abundance of AOA either did not change or was reduced when FPBO-FAs were applied into the soil (Fig. 1.3C). Some authors have suggested a niche differentiation between both microbial groups (Di et al., 2010; Schleper and Nicol, 2010), with one or the other being more competitive under a given set of conditions, as they belong to separate phylogenetic domains with different cell biochemical and metabolic processes. In particular, nutrient-rich environments characterised by a pH close to neutrality favour AOB rather than AOA (Verhamme et al., 2011). Indeed, in line with the higher AOB abundance, the use of FPBO-FAs increased the soil pH by two units, reaching an average value of 7.2 in the presence and absence of plants after 60 and 100 days of incubation.

1.3.2 Effect of FPBO-ashes on microbial biomass and activity

Microbial properties are sensitive indicators when evaluating soil disturbance and impact, as they respond promptly and sensibly to changes in soil management (Bünemann et al., 2018; Schloter et al., 2018). In this regard, at the

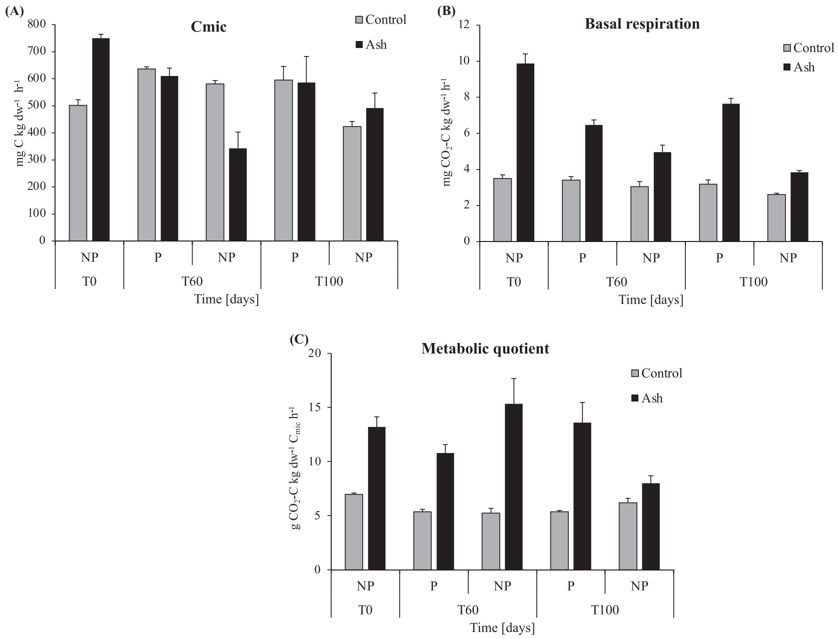


FIGURE 1.4 Microbial biomass carbon (Cmic, A), basal respiration (B), and metabolic quotient (C) in the control and the ash-treated soils at the three incubation times (0, 60 and 100 days) in the presence and the absence of plants (P and NP, respectively). Values are means ($n = 3$) with the standard deviation and are expressed on a dry weight basis.

mesocosm-scale, the application of FPBO-FAs led to higher levels of basal respiration used as a proxy of microbial activity for all the time points and regardless of plant presence (Fig. 1.4B). Nonetheless, an increase in microbial biomass carbon (C_{mic}) relative to the control without ashes was only observed in the ash-amended soils at the beginning of the trial and in the absence of plants (Fig. 1.4A). No further changes in Cmic were recorded until the end of the mesocosm trial. Other authors did not detect any changes in soil C_{mic} (Nayak et al., 2015) and in microbial biomass (García-Sánchez et al., 2015) following FA addition.

The positive effects on microbial activity could be related to the increased nutrient availability in the presence of FPBO-FAs. Amending the soil with the FPBO-FAs from 'crumbled' pine wood was accompanied by a twofold increase in the dissolved organic carbon (DOC) content compared to the control soil, with mean values of 158 and 172 mg C kg^{-1} dry mass in the ash-treated soils on days 60 and 100 irrespective of plant presence. Sharma and Kalra (2006) reported that FAs might also have adverse effects on soil respiration, microbial biomass and enzymatic activities; such effects are proportional to the amount of ash. This negative impact could be attributable to the accumulation of HMs up to toxic levels in the FAs (Sharma and Kalra, 2006).

The FPBO-FAs used in our mesocosm trial were, however, characterised by low contents in HMs (Table 1.1).

The metabolic quotient or specific activity of the microbial biomass (microbial respiration per unit biomass; qCO_2) can be used as a measure of microbial efficiency and environmental stress (Anderson and Domsch, 1993; Wardle and Ghani, 1995). Despite the overall stimulation of microbial activity, the metabolic quotient was higher in the ash-treated soils than in the control soil, independent of the sampling time and plant presence (Fig. 1.4C). This suggests that although the use of FPBO-FAs provided other nutrients that allowed for a boost of microbial activity, their application likely induced changes in the environmental conditions of the soil matrix, forcing the microbial communities to adapt.

1.3.3 Effect of FPBO-ashes on plant yield

On the mesocosm scale, amending the soil with FPBO-FAs resulted in a fourfold increase in the aboveground biomass of wheat relative to the control without ashes after 60 and 100 days of incubation (Schönegger et al., 2018). Accordingly, the application of FPBO-FAs also had a positive effect on the root biomass and the grain yield regardless of the incubation period (Schönegger et al., 2018). This is in agreement with the improved soil nutrient status and the boost in microbial activity and nitrification rate following the FPBO-FA addition (Figs. 1.3A, 1.4B). The positive effects of FAs on plant growth and crop yields have already been documented in previous studies (Katiyar et al., 2012; Li et al., 2016; Masu et al., 2016).

In discordance with the previous mesocosm trial, Kurzemann et al. (2021) reported a lack of effects in plant yield and nutritional properties in a field-scale study following the application of FPBO-FAs. These latter authors used FPBO-FAs from three different biomass sources (bark, forest residues and *Miscanthus* sp.), while in Schönegger et al. (2018), the FPBO-FAs were obtained from crumbled pine wood. Besides this distinction, of note is also that in Kurzemann et al. (2021) the initial soil pH was neutral (7.6 ± 0.04) prior to ash application. For acidic soils, however, FPBO-FAs application proved to be beneficial (Schönegger et al., 2018).

1.4 Legal aspects related to FPBO-ashes

1.4.1 European waste law

The European Directive 2008/98 EC protects the environment against the harmful effects caused by waste. The term ‘waste’ means any substance or object, which the holder discards or intends or is required to discard (article 3§1), and it must be interpreted broadly (Bergthaler and Wolfslehner, 2004) in order to ensure a high level of protection (Treaty on the functioning of the

European Union, chapter 191, section 2). Against some attempts for application shortcuts, ashes do not classify as by-products (Directive 2008/98 EC, chapter 1, article 5, section 1, 32): (1) their further use is uncertain; (2) they need processing, such as stabilisation prior to application; and (3) generally, they are potentially harmful to the environment. In short: According to European law, ashes are waste, and consequently, their application to soils — regardless of any environmental benefit (Bumberger and Hinterwirth, 2013) — is against the law.

While across the EU, regulations and decisions are legally binding for all Member States (Schröder, 2015), directives, such as the European Directive 2008/98 EC, need to be translated into national law for them to apply. Consequently, waste law is national law. Although national regulations need to be 'compatible with all treaties' and 'notified to the Commission' (Treaty of the functioning of the European Union chapter 193, section 1), they can be stricter compared to EU regulations. In addition, the national waste law has to agree with other (EU and national) protection laws, such as soil and water protection laws.

Ashes might lose their waste status after undergoing recovery operations, which might enable the recycling of FPBO-FAs as a soil amendment. Options for losing the waste status depend on the category of the waste. The European Commission differentiates four types of ash (European Commission 2017), both hazardous and non-hazardous, depending on the original biomass source. The fast pyrolysis process generates charcoal, which is not coal, and it originates from exclusively untreated, woody biomass. Therefore, the FPBO-FAs discussed here best fit the category 10 01 03 'fly ash from peat and untreated wood', which is a non-hazardous waste category. This categorisation, however, needs confirmation by the authorities before their soil application. According to European law, the best treatment option for biomass ashes is R10 'land treatment resulting in benefits to agriculture or ecological improvements', thereby generally offering the recycling of (FPBO) FA nutrients back into the environment. Following the legally binding hierarchy of waste treatment according to the Directive 2008/98 EC (article 4), after waste prevention, options, such as recycling and recovery, are superior to disposal. Nonetheless, this recycling option is prohibited if maximum pollutant concentrations are exceeded, in which case the ash has to be disposed of. Although the European Directive provides guidelines for such thresholds, national waste law frequently installs more stringent limitations (Table 1.3). Only if the waste cannot be recovered, it needs to be disposed of (article 3, section 15). Disposal options, according to Annex I of the Directive 2008/98 EC, include landfill (D1), land treatment (D2), release into a water body except seas/oceans (D6), and permanent storage (mining) (D12).

The high nutrient concentration of FPBO-FAs also allows their consideration as a fertilizer, which is a product. In order for fertilizers to be traded and applied all over the European Union, they have to be classified as EC fertilizers. Currently, there are no ashes of any kind listed in the European list of fertilizers or fertilizer additives. Consequently, FPBO-FA cannot be labelled as an EC fertilizer. As fertilizers, the use of FPBO-FAs is possible only at a national level, and this depends on national regulations. Moreover, the application of national fertilizers across borders depends on mutual recognition between Member States and is, therefore, usually difficult due to diverging regulations. Considering that the application of FPBO-FAs to soil underlies national law, we will further review some EU Member States' laws in this regard.

1.4.2 Waste law in selected EU Member States

In general, there is a difference between the legal option of applying FPBO-FAs to soil and their commercial application. Products made out of waste need to obtain their end-of-waste status. In most countries (e.g. Austria, France, United Kingdom and The Netherlands), individual companies can apply for an end-of-waste status for their products in a standardized procedure. In some countries (e.g. Germany, Italy), this is not an option. In those cases, a change in laws is required for the commercial application of FPBO-FAs to the soil.

1.4.2.1 Austria

Within the Austrian waste classification, FPBO-FAs are not explicitly listed. It best fits wood ash (31306) and relevant subcategories therein (bottom ash (31306 70), fly ash (31306 72), and fine fly ash (31306 74)) (Abfallwirtschaftsgesetz 2002, article 17, section 1), which are all non-hazardous waste categories. In Austria, the application of wood ash to soil is legal; however, it cannot be termed fertilizer. The best option for wood ash recycling is its addition to compost. In any case, the legal limits of HMs apply (Table 1.3).

Compost: According to the Austrian *Kompostverordnung* (Compost ordinance), the addition of ash to compost is allowed with several restrictions.

- Ash addition improves the composting process.
- Maximum ash percentages (m/m) in compost: $<$ 15%, 'ash from biomass combustion' (material 303) $<$ 5% (Kompostverordnung Table 1.3), aggregated ash $<$ 2%, no fine fly ash ('Feinstflugasche')[1].
- Polychlorinated dibenzodioxins and polychlorinated dibenzofurans (PCDD/PCDF) concentrations in the ash $<$ 100 mg toxicity equivalents (TE) / kg dry matter.

---

[1] Caution with the terminology of fine fly ash (German 'Feinstflugasche') vs. fly ash (German 'Flugasche').

- Maximum HM concentrations apply, both for the input ash and the final product compost (Kompostverordnung, annex 2, part 2).

Following its addition to compost, ash loses its waste status and becomes part of a product (Kompostverordnung, articles 1 and 11). Depending on the properties of the compost, it is classified A + , A or B; the class determines application options: for organic farming, only A + ; for agriculture, only A + or A can be used.

Soil protection: The Austrian soil protection law does not regulate ash application. Relevant legislation for ash is part of the *Düngemittelgesetz*, *Kompostverordnung*, *Forstgesetz* and *Wasserrechtsgesetz* (Holzer and Reischauer, 2015).

Forest protection: According to the *Forstgesetz*, the application of ash to forest soil is not considered devastation, given it is undertaken by authorised personnel and within legal measures (a maximum amount of 2000 kg can be applied over a period of 20 years).

Water protection: The application of fertilising agents, including composts and ashes, of N amounts > 175 kg h^{-1} y^{-1} requires a water usage permit in accordance with the *Wasserrechtsgesetz* (article 32). Below this limit, there is no need for such a special permit.

Fertilizers: Residues from combustion, including ashes, cannot be termed fertilizers (Düngemittelgesetz, article 4, section 4). Ash application to agricultural and forest soil is, however, allowed. The authorities even recommend the use of plant ashes for nutrient recycling if minimum nutrient and maximum pollutant concentrations are met (Federal Ministry of Agriculture, Forestry, Environment and Water 2011). Based on their nutrient and pollutant properties, especially concerning Cd, ashes are classified and their classes determine the application options. In particular, the maximum ash dosages within a 20 y period are 1000, 500 and 2000 kg h^{-1} y^{-1} for agriculture, grasslands and forests, respectively.

1.4.2.2 Finland

The Finnish waste law, the *Jätelaki*, categorises ash from peat and untreated wood (10 01 03) following the European waste key. It does not explicitly mention FPBO-FAs. The *Jätelaki* also installs proper treatment options for wastes, such as 'land treatment resulting in benefit to agriculture or in an ecological improvement' (Jäteasetus, annex I, R10). In any case of recovering ash in land treatment, maximum concentrations for HMs and other pollutants apply (Table 1.3).

The Finnish *Ympäristönsuojelulaki* protects the environment, including soil and water, from pollution. It requires a permit for waste treatment on a professional base or at a plant. Nevertheless, the application of ash into the soil is a common practice in Finland; it is legal to recover or dispose 'natural, non-hazardous waste of vegetal origin deriving from the industry's own operations in agriculture and forestry' (Ympäristönsuojelulaki, section 4). In Finland, ash is also listed as a fertilizer.

Soil protection: The *Ympäristönsuojelulaki* prohibits the deposition of waste onto the ground and into the soil (article 42, section 1).

Forest protection: The *Metsälaki* permits the application of ash into forests for fertilisation purposes, in line with the aim of managing forests 'in such a manner that the general conditions for the preservation of those habitats, which are important for the biological diversity of forests, are safeguarded' (section 10§1).

Water protection: The *Ympäristönsuojelulaki* prohibits any action posing a potential threat to (ground)water (section 8). There are no explicit entries on the ash application.

Fertilizers: Both the fertilizer product regulation, the *Lannoitevalmistelaki*, and the fertilizer decree, the *Maa-ja metsätalousministeriön asetus lannoitevalmisteista*, regulate fertilisation. In accordance with these laws, fertilizers may not contain 'harmful substances, products or organisms in such quantities that their use in accordance with the instructions may cause any danger to human or animal health or safety, plant health or the environment' (Lannoitevalmistelaki, article 5). In addition, any fertilizer has to promote plant growth [Maa-ja metsätalousministeriön asetus lannoitevalmisteista, chapter 1, A7, annex I]. Ashes from wood, peat or agricultural biomass are explicitly listed as fertilizer for agricultural, forest and horticultural applications. Minimum nutrient and maximum pollutant, including HM concentrations, apply (Haglund et al., 2008).

1.4.2.3 Germany

Similar to the European Directive, the German waste classification system, the *Abfallverzeichnisverordnung*, does not explicitly mention FPBO- FAs but offers the non-hazardous class of 'filter ashes of peat and untreated wood combustion' (10 01 03). The waste can be recovered from its waste status by operations, such as R10 'Land treatment resulting in benefit to agriculture or ecological improvement' (Kreislaufwirtschaftsgesetz, annex 2). Concerning ashes, their application to soil is legal as long as they do not exceed the maximum values of pollutants set for fertilizers (Table 1.3). Concerning fertilizers, FPBO-FAs do not contain sufficient nutrients to classify as fertilizer; an option, however, is their use as an additive to fertilizers. Otherwise, they would need to be landfilled (Bundesgütegemeinschaft Holzasche eV 19).

Soil protection: The *Bundesbodenschutzgesetz* installs precautions preventing harmful changes in soil and their surrounding area (§7). These obligations are met if the harmful effects are avoided or reduced. Both agricultural and horticultural soils are potential recipients of FPBO-FA fertilisation, and good agricultural practice has to be considered (Bundesbodenschutzgesetz, articles 17 and 7). The responsibility is on the property owner, the occupant of a site and the party who carries out/has carried out actions by others.

Forest protection: The *Bundeswaldgesetz* allows the application of fertilizers, including those containing wood ash, into forests for the purposes of soil conservation and restoration.

Water protection: The *Wasserhaushaltsgesetz* aims at protecting water as an ecosystem, habitat and recreational area. It does not specifically mention ash or request permits as long as the *Bodenschutzgesetz* and good agricultural practices apply.

Fertilizers: The *Düngegesetz* classifies fertilizers into categories based on their nutrients and strictly regulates their primary and secondary constituents (article 5, section 3). The law does not prohibit the use of ash but installs maximum concentrations of pollutants and minimum concentrations of nutrients. FPBO-FAs contain a range of different nutrients that usually do not come in the concentrations required to be classified as a fertilizer. While this prevents FPBO-FAs from becoming fertilizers on their own, they can be used as additives for fertilizers. For the final fertilizer, the ash addition needs to be indicated (Düngemittelverordnung, chapter 6), and organic fertilizers with added ash are considered organic-mineral fertilizers.

1.4.2.4 The Netherlands

Considering the legislation of all European Member States reviewed in this chapter, the national legislation on ash application to soil is stricter in the Netherlands. The Dutch *Wet milieubeheer*, in line with the European classification system, offers the waste category of 'Fly ash from peat or untreated wood' (10 01 03), which does not explicitly mention FPBO-FAs. A possible, sound treatment option for ashes in this category is R10 'Land treatment resulting in benefit to agriculture or ecological improvement'. In the Netherlands, the legal use of FPBO-FAs as a fertilizer requires an approved application by the Fertilization Committee. Maximum pollutant and minimum nutrient concentrations apply (Table 1.3). As a blend with other fertilising agents, FPBO-FAs might have a high potential for successful fertilizer application. Nevertheless, this will require more research for proper validation.

Soil protection: Since the 1970s, the Netherlands has applied systematic and strict soil protection policies, partly through radical programmes and measures in terms of prevention, research and restoration (Lee and Bückmann, 2007). The legislation for soil protection is contained in both the *Wet milieubeheer* and especially the *Wet bodembescherming*. They both strictly prohibit any deposition of waste or waste products into the soil.

Forest protection: The use of fertilizers on forest land is not common in the Netherlands. The *Wet natuurbescherming* does not mention the application of ashes in forests.

Water protection: The *Waterwet* protects and improves the (ground) water and prevents pollution or impairment by any reasonable action. It does not mention the application of ash.

Fertilizers: The *Meststoffenwet* prohibits the use of wastes for fertilisation purposes (article 4, section2) and lists the national fertilizers allowed in addition to EC fertilizers (annex AA, sections I–IV). In order to make the list, the applicant fertilizer has to pass the approval of the Fertilization Committee. For approval, applicant fertilizers have to prove their benefit for the environment, and environmental safety and they have to come with a high nutrient content, based on which they are grouped into a fertilizer category (Uitvoeringsregelingen Meststoffenwet, article 5, section 2). Ashes rarely make the list as their nutrient to pollutant ratio is relatively low.

1.5 Conclusions and final remarks

The European Directive 2008/98 EC clearly defines that ashes, including those derived from the fast pyrolysis process, need to be treated as waste. Consequently, their application to soil – regardless of a potential benefit – is against the law. Ashes can exit their waste status by undergoing a pretreatment, which could eventually involve the recycling of their nutrients into the soil. The tested FPBO-FAs derived from five different biomass sources exhibited a high nutrient status, and their potential inhibitory effect on seed germination disappeared after dilution. Following the application doses recommended in legislation or when used in acidic soils that require lime amendments, the beneficial effects of the addition of FPBO-FAs are evident. Thus, the reincorporation of the nutrients extracted with biomass harvesting may warrant a circular use. Nonetheless, long-term studies dealing with different crops and soils are still needed to shed light on the full potential of FPBO-FAs as soil amendments. Within this context, it is essential to determine the maximum rate at which FPBO ashes can be safely applied to soil in terms of the bioavailability of the potential pollutants present in the ashes. Altogether, it would contribute to increasing the environmental sustainability of the overall FPBO production process.

Acknowledgements

The Residue2Heat project has received funding from the European Union's Horizon 2020 Research and Innovation Programme under Grant Agreement #No. 654650. María Gómez-Brandón acknowledges support by the Programa Ramón y Cajal (RYC-2016-21231; Ministerio de Economía y Competitividad). We are indebted to the master students Hannes Hauser, Deborah Schönegger, Ginevra Fabiani and Thomas Mazzier for their great help with the sample collection and legal aspects. The authors also thank Paul Fraiz for his valuable help in English editing.

Declarations of interest

None.

References

AEBIOM, 2017. 2017. European Biomass Association. European Bioenergy Outlook Annual statistical report. European Biomass Association, Brussels, Belgium, p. 264.

Anderson, T.H., Domsch, K.H., 1993. The metabolic quotient for CO_2 (qCO_2) as a specific activity parameter to assess the effects of environmental conditions, such as pH, on the microbial biomass of forest soils. Soil. Biol. Biochem. 25, 393−395.

Bergthaler W., Wolfslehner E., 2004. Das Recht der Abfallwirtschaft 11. Wien.

BMLFUW, 2001. Bundesminister für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft. Verordnung über Qualitätsanforderungen an Komposte aus Abfällen, vol. BGBl II, Nr. 292/2001. p. 60. (Austrian Compost Ordinance).

BMLFUW, 2011. Bundesministerium für Land- und Forstwirtschaft, Umwelt und Wasserwirtschaft. Richtlinie für den sachgerechten Einsatz von Pflanzenaschen zur Verwertung auf Land- und Forstwirtschaftlich genutzten Flächen. pp. 74. https://www.bmlfuw.gv.at/service/publikationen/umwelt/richtlinie_fuer_den_sachgerechten_einsatz_-von_pflanzenaschen_zur_verwertung_auf_land-und_forstwirtschaftlich_genutzten_flaechen.html (accessed 21.02.22.)

Bougnom, B.P., Knapp, B.A., Elhottová, D., Koubová, A., Etoa, F.X., Insam, H., 2010. Designer compost with biomass ashes for ameliorating acid tropical soils: effects on the soil microbiota. Appl. Soil. Ecol. 45, 319−324.

Bougnom, B.P., Niederkofler, C., Knapp, B.A., Stimpfl, E., Insam, H., 2012. Residues from renewable energy production: their value for fertilizing pastures. Biomass Bioenergy 39, 290−295.

Bridgewater, A.V., 2012. Review of fast pyrolysis of biomass and product upgrading. Biomass Bioenerg 38, 68−94.

Bumberger, L., Hinterwirth, D.W. W.R.G.; Kommentar2 (Wien 2013). Bundesgütegemeinschaft Holzasche eV https://www.holzaschen.de/faq.html (access 25.02.22)

Bünemann, E.K., Bongiorno, G., et al., 2018. Soil quality − a critical review. Soil. Biol. Biochem. 120, 105−125.

Chen, Z., Lu, S., Mao, Q., Buekens, A., Chang, W., Wang, X., et al., 2016. Suppressing heavy metal leaching through ball Milling of fly ash. Energies 9, 524.

Dahl, O., Nurmesniemi, H., Pöykiö, R., Watkins, G., 2009. Comparison of the characteristics of bottom ash and fly ash from a medium-size (32 MW) municipal district heating plant incinerating forest residues and peat in a fluidized-bed boiler. Fuel Process. Technol. 90, 871−878.

Di, H.J., Cameron, K.C., Shen, J.-P., Winefield, C.S., O'Callaghan, M., Bowatte, S., et al., 2010. Ammonia-oxidizing bacteria and archaea grow under contrasting soil nitrogen conditions. FEMS Microbiol. Ecol. 72, 386−394.

Dimitriou, I., Eriksson, J., Adler, A., Aronsson, P., Verwijst, T., 2006. Fate of heavy metals after application of sewage sludge and wood−ash mixtures to short-rotation willow coppice. Environ. Pollut. 142, 160−169.

Fernández-Delgado Juárez, M., Waldhuber, S., Knapp, A., Partl, C., Gómez-Brandón, M., Insam, H., 2013. Wood ash effects on chemical and microbiological properties of digestate- and manure-amended soils. Biol. Fertil. Soils 49, 575−585.

Fernández-Delgado Juárez, M., Gómez-Brandón, M., Knapp, A., Stöhr, D., Insam, H., 2015. Chemical and microbiological properties of alpine forest soils: effects of pelletized ashes in a short-term trial. For. Ecol. Manage. 357, 42–49.

Fernández-Delgado Juárez, M., Mostbauer, P., Knapp, A., Müller, W., Tertsch, S., Bockreis, A., et al., 2018. Biogas purification with biomass ash. Waste Manag 71, 224–232.

Fernández-Delgado Juárez, M., Fabiani, G., Mazzier, T., Schönegger, D., Pietramellara, G., Gómez-Brandón, M., et al., 2020. Reclamation of acid soils with biomass ashes from pyrolytic wood liquefaction. Waste Biomass Valor 11, 5067–5078.

García-Sánchez, M., Siles, J.A., Cajthaml, T., García-Romera, I., Tlustoš, P., Száková, J., 2015. Effect of digestate and fly ash applications on soil functional properties and microbial communities. Eur. J. Soil. Biol 71, 1–12.

Haglund N., 2008. Guideline for classification of ash from solid biofuels and peat utilised for recycling and fertilizing in forestry and agriculture.

Holzer, G., Reischauer, E., 2015.Agrarumweltrecht. Landwirtschaft – Umwelt – Recht; Handbuch. Wien: Verlag Österreich.

Ibeto, C.N., Lag-Brotons, A.J., Marshall, R., Semple, K.T., 2020. The nutritional effects of digested and undigested organic wastes combined with wood ash amendments on carrot plants. Soil. Sci. Plant. Nutr 20, 460–472.

Ingerslev, M., Skov, S., Sevel, L., Pedersen, L.B., 2011. Element budgets of forest biomass combustion and ash fertilisation – a Danish case-study. Biomass Bioenerg 35, 2697–2704.

Katiyar, D., Singh, A., Malaviya, P., Pant, D., Singh, P., Abraham, G., et al., 2012. Impact of fly-ash-amended soil on growth and yield of crop plants. Int. J. Environ. Waste Manag 10, 150–162.

Knapp, B.A., Insam, H., 2011. Recycling of biomass ashes: current technologies and future Research needs. In: Insam, H., Knapp, B.A. (Eds.), Recycling of Biomass Ashes. Springer, Heidelberg, pp. 1–16.

Krutof, A., Hawboldt, K.A., 2018. Upgrading of biomass sourced pyrolysis oil review: focus on co-pyrolysis and vapour upgrading during pyrolysis. Biomass Convers. Biorefin. 8, 775–787.

Kuba, T., Tschöll, A., Partl, C., Meyer, K., Insam, H., 2008. Wood ash admixture to organic wastes improves compost and its performance. Agric. Ecosyst. Environ. 127, 43–49.

Kurzemann, F.R., Juárez, Fernández-Delgado, Probst, M., Gómez-Brandón, M., Partl, M., Insam, H., C., 2021. Effect of biomass fly ashes from fast pyrolysis bio-oil production on soil properties and plant yield. J. Environ. Manage. 298, 113479.

Lanzerstorfer, C., 2017. Chemical composition and properties of ashes from combustion plants using *Miscanthus* as fuel. J. Environ. Sci 54, 178–183.

Lanzerstorfer, C., 2019. Combustion of *Miscanthus*: composition of the ash by particle size. Energies 12, 178.

Lapa, N., Barbosa, R., Lopes, M.H., Mendes, B., Abelha, P., Gulyurtlu, I., et al., 2007. Chemical and ecotoxicological characterization of ashes obtained from sewage sludge combustion in a fluidised-bed reactor. J. Hazard. Mater 147, 175–183.

Lehto, J., Oasmaa, A., Solantausta, Y., Kytö, M., Chiaramonti, D., 2014. Review of fuel oil quality and combustion of fast pyrolysis bio-oils from lignocellulosic biomass. Appl. Energy 116, 178–190.

Lehtovirta-Morley, L.E., 2018. Ammonia-oxidation. Ecology, physiology, biochemistry and why they must all come together. FEMS Microbiol. Lett. 365, fny058.

Leijenhorst, E.J., Wolters, W., van de Beld, L., Prins, W., 2016. Inorganic element transfer from biomass to fast pyrolysis oil: review and experiments. Fuel Process. Technol. 149, 96–111.

Li, L., Yu, C., Bai, J., Wang, Q., Luo, Z., 2012. Heavy metal characterization of circulating fluidized bed derived biomass ash. J. Hazard. Mater 233–234, 41–47.

Li, X., Rubæk, G.H., Sørensen, P., 2016. High plant availability of phosphorus and low availability of cadmium in four biomass combustion ashes. Sci. Total. Environ. 557, 851–860.

Lohri, C.R., Rajabu, H.M., Sweeney, D.J., Zurbrügg, C., 2016. Char fuel production in developing countries – a review of urban biowaste carbonization. Renew. Sust. Energ. Rev. 59, 1514–1530.

Maresca, A., Hyks, J., Astrup, T.F., 2017. Recirculation of biomass ashes onto forest soils: ash composition, mineralogy and leaching properties. Waste Manag 70, 127–138.

Masu, S., Jurj, N.L., Lixandru, B., Burtica, G., 2016. Application of fly ash as amendment in agricultural field of *Avena sativa*. Environ. Eng. Manag. J 15, 2681–2688.

Nayak, A.K., Raja, R., Rao, K.S., Shukla, A.K., Mohanty, S., Shahid, M., et al., 2015. Effect of fly ash application on soil microbial response and heavy metal accumulation in soil and rice plant. Ecotoxicol. Environ. Safe 114, 257–262.

Odlare, M., Pell, M., 2009. Effect of wood fly ash and compost on nitrification and denitrification in agricultural soil. Appl. Energ. 86, 74–80.

Oleszczuk, P., Rycaj, M., Lehmann, J., Cornelissen, G., 2012. Influence of activated carbon and biochar on phytotoxicity of air-dried sewage sludges to *Lepidium sativum*. Ecotoxicol. Environ. Safe 80, 321–326.

Ou, Y., Ma, S., Zhou, X., Wang, X., Shi, J., Zhang, Y., 2020. The effect of a fly ash-based soil conditioner on corn and wheat yield and risk analysis of heavy metal contamination. Sustainability 12, 7281.

Pan, H., Eberhardt, T.L., 2011. Characterization of fly ash from the gasification of wood and assessment for tis application as a soil amendment. Bioresources 6, 3987–4004.

Pugliese, S., Jones, T., Preston, M.D., Hazlett, P., Tran, H., Basiliko, N., 2014. Wood ash as a forest soil amendment: The role of boiler and soil type on soil property response. Can. J. Soil. Sci. 94, 621–634.

Ram, L.C., Masto, R.E., 2014. Fly ash for soil amelioration: a review on the influence of ash blending with inorganic and organic amendments. Earth-Sci. Rev. 128, 52–74.

Rey-Salgueiro, L., Omil, B., Merino, A., Martínez-Carballo, E., Simal-Gándara, J., 2016. Organic pollutants profiling wood ashes from biomass power plants linked to ash characteristics. Sci. Total. Environ. 544, 535–543.

Ring, E., Jacobson, S., Nohrstedt, H.-Ö., 2006. Soil-solution chemistry in a coniferous stand after adding wood ash and nitrogen, Can. J. For. Res, 36. pp. 153–163.

Römbke, J., Moser, T.H., Moser, H., 2009. Ecotoxicological characterisation of 12 incineration ashes using 6 laboratory tests. Waste Manag 29, 2475–2482.

Saarsalmi, A., Kukkola, M., Moilanen, M., Arola, M., 2006. Long-term effects of ash and N fertilization on stand growth, tree nutrient status and soil chemistry in a Scots pine stand. For. Ecol. Manage. 235, 116–128.

Schleper, C., Nicol, G.W., 2010. Ammonia-oxidising archaea—physiology, ecology and evolution. Adv. Microb. Physiol. 57, 1–41.

Schloter, M., Nannipieri, P., Sørensen, S.J., van Elsas, J.D., 2018. Microbial indicators for soil quality. Biol. Fertil. Soils. 54, 1–10.

Schönegger, D., Gómez-Brandón, M., Mazzier, T., Insam, H., Hermanns, R., Leijenhorst, E., et al., 2018. Phosphorous fertilizing potential of fly ash and effects on soil microbiota and crop. Resour. Conserv. Recycl. 134, 262–270.

Schröder W., 2015. Grundkurs Europarecht München.

Sharma, S.K., Kalra, N., 2006. Effect of fly ash incorporation on soil properties and productivity of crops: a review. J. Sci. Ind. Res. 65, 383–390.

Tchounwou, P.B., Yedjou, C.G., Patlolla, A.K., Sutton, D.J., 2012. Heavy metals toxicity and the environment. Mol. Clin. Environ. Toxicol. EXS 101, 133–164.

Tsiridis, V., Petala, M., Samaras, P., Kungolos, A., Sakellaropoulos, G.P., 2012. Environmental hazard assessment of coal fly ashes using leaching and ecotoxicity tests. Ecotoxicol. Environ. Saf. 84, 212–220.

USEPA, 2013. Code of Federal Regulations (CFR), 40 CFR—Chapter I—Part 423: Appendix A to Part 423. 2013. https://www3.epa.gov/region1/npdes/permits/generic/prioritypollutants.pdf (access 21.02.22).

Van de Beld, B., Holle, E., Florijn, J., 2013. The use of pyrolysis oil and pyrolysis oil derived fuels in diesel engines for CHP applications. Appl. Energy 102, 190–197.

Vassilev, S.V., Baxter, D., Andersen, L.K., Vassileva, C.G., 2013a. An overview of the composition and application of biomass ash. Part 1. Phase–mineral and chemical composition and classification. Fuel 105, 40–76.

Vassilev, S.V., Baxter, D., Vassileva, C.G., 2013b. An overview of the behaviour of biomass during combustion: Part I. Phase-mineral transformations of organic and inorganic matter. Fuel 112, 391–449.

Verhamme, D.T., Prosser, J.I., Nicol, G.W., 2011. Ammonia concentra-tion determines differential growth of ammonia-oxidising archaea and bacteria in soil microcosms. ISME J 5, 1067–1071.

Wardle, D.A., Ghani, A., 1995. A critique of the microbial metabolic quotient (qCO2) as a bioindicator of disturbance and ecosystem development. Soil. Biol. Biochem. 27, 1601–1610.

Wey, M.-Y., Chao, C.-Y., Chen, J.-C., Yu, L.-J., 1998. The relationship between the quantity of heavy metal and PAHs in fly ash. J. Air Waste Manag 48, 750–756.

Zhong, Z.W., Song, B., Zaki, M.B.M., 2010. Life-cycle assessment of flash pyrolysis of wood waste. J. Clean. Prod 18, 1177–1183.

Zucconi, F., Forte, M., Monac, A., de Bertoldi, M., 1981. Biological evaluation of compost maturity. Biocycle 22, 27–29.

Chapter 2

Bioconversion of organic wastes into wealth by vermitechnology: a review

Debadatta Sethi[1,2,*], Konathala Kusumavathi[3,*], Balasubramani Ravindran[4], Narayan Panda[1], Kshitipati Padhan[1], Subhaprada Dash[5], Tapas Ranjan Sahoo[6], Satyabrata Mangaraj[7], Arabinda Dhal[8], Susanta Kumar Swain[9], Smritikana Sarkar[3], Sushanata Kumar Pattanayak[1,*] and Andi Febrisiantosa[10]

[1]Department of Soil Science and Agricultural Chemistry, College of Agriculture, Odisha University of Agriculture and Technology, Bhubaneswar, Odisha, India, [2]Regional Research and Technology Transfer Station, Odisha University of Agriculture and Technology, G. Udayagiri, Kandhamal, Odisha, India, [3]Department of Agronomy, Bidhan Chandra Krishi Viswavidyalaya, Mohanpur, West Bengal, India, [4]Department of Environmental Energy and Engineering, Kyonggi University Yeongtong-Gu, Suwon, Republic of Korea, [5]Department of Agronomy, Institute of Agricultural Sciences, Siksha "O" Anusandhan (Deemed to be University), Bhubaneswar, Odisha, India, [6]Krishi Vigyan Kendra, Odisha University of Agriculture and Technology, Kendrapara, Odisha, India, [7]Krishi Vigyan Kendra, Odisha University of Agriculture and Technology, Ganjam-I, Odisha, India, [8]Department of Plant Pathology, College of Agriculture, Odisha University of Agriculture and Technology, Bhubaneswar, Odisha, India, [9]Directorate of Research, Odisha University of Agriculture and Technology, Bhubaneswar, Odisha, India, [10]Research Centre for Food Technology and Processing, National Research and Innovation Agency, Yogyakarta, Indonesia

2.1 Introduction

The escalating human population, industrialisation, urbanisation, and agricultural production have resulted in the accumulation of huge quantities of solid wastes, which have had a hazardous effect on the environment. About 50% of solid waste generated was organic (Kumar and Agrawal, 2020). According to the central pollution control board of India, the waste generation rate has increased exponentially from 0.26 kg per capita^{-1} day^{-1} to 0.35 kg per capita^{-1} day^{-1} (Central Pollution Control Board CPCB India, 2018).

---

* Equal contribution.

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00009-8

The agriculture-based waste products are 100% organic and lignocellulosic in nature. The lignocellulosic biomass could be used in the production of biofuel, vermicomposting and glycerol (Sharma et al., 2018; Pandit et al., 2020). It was estimated that 80%−90% of municipal wastes are disposed off in landfilling without any proper management. Landfilling of waste materials makes the vicinity unpleasant by producing fouls smell, creating a breeding place for the insects. Landfilling with wastes also produces several greenhouse gases. Nowadays, the availability land for waste dumping is a serious challenge (Annepu, 2012). For disposing off wastes safely, they need to be converted effectively. This may achieve by segregating the undecomposable from decomposable ones and vermicomposting these wastes.

Decomposition involves the degradation of larger organic molecules into smaller and simpler components. During decomposition of organic wastes, CO_2, H_2O, energy and microbial biomass and mineralisable nutrient will be produced (Fig. 2.1). When organic tissues are added to aerobic soil, generally, three reactions take place. Before such reactions occur in the soil, the physical shredding of organic tissues by soil fauna is a prerequisite.

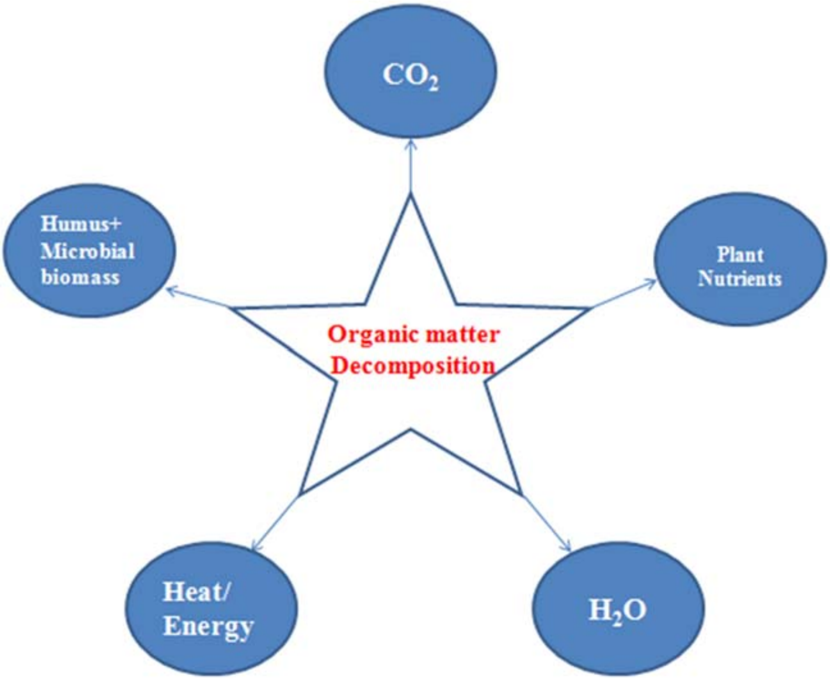


FIGURE 2.1 Schematic diagram of different products produced from organic matter decomposition.

1. Carbon dioxide, energy, water and decomposed biomass produces from the enzymatic oxidation of "C" compounds

$$R\text{-}(C,\ 4\mathrm{H}) + 2\mathrm{O}_2\mathrm{CO}_2 + \mathrm{H}_2\mathrm{O} + 478\ \mathrm{kJ\ mol}^{-1}\mathrm{C}$$

2. Mineralisation/immobilisation of essential plant nutrients by a series of specific reactions that are unique for each element
3. Formation of resistant compounds for microbial action

The biomolecules like cellulose and starch are long-chain polysaccharides of sugar molecules. Enzymatic degradation proceeds to a stepwise breaking of C—O—C linkage, ultimately converting to CO_2 and energy. The plant proteins also subjected to microbial decay, yielding not only CO_2 and H_2O but also amino acids, glycine (CH_2NH_2COOH) and cysteine ($CH_2HSCHNH_2COOH$). In turn, these N and S compounds are further broken down, yielding NH_4^+, NO_3^- and SO_4^{2-}, which are available to plants. The CO_2 is lost to the atmosphere. The C content decreases, whereas N and S increase during this process. The lignin molecules are very large and complex and require specific microorganisms for further break down (white rot fungi). These materials help granulation and aggregate formation in soil. Several scientists have reported the change in physicochemical biological properties during vermicomposting and also characterised the quality of vermicompost produced from different organic wastes (Pandit et al., 2020; Pottipati et al., 2021). The two-stage combined system of composting + vermicomposting was applied to treat two different mixtures of organic wastes: Eucalyptus sp. sawdust, rice hulls and Poultry Litter (Vicentin et al., 2021). A combination of composting and vermicomposting has been successfully used for removing polycyclic aromatic hydrocarbons and heavy metals (Grasserová et al., 2020).

Combined composting and vermicomposting strategy reduce the organic matter and nutrient content (Vicentin et al., 2021). Pre-composting facilitates its breakdown, suppresses pathogens and decomposes toxic compounds, which could harm the earthworms, thus is an important step when decomposing organic waste by vermicomposting (Grasserová et al., 2020).

2.2 Vermiculture

Vermiculture refers to a scientific method of cultivating of earthworms by keeping the intention of increasing growth and population. The production of higher no of earthworm, now a day's becoming a challenge as it is essential to produce more nutrient rich compost for the meeting crop nutrient demand as organic source. The earthworms after vermiculturing can be used in feeding the domestic or commercial birds, such as poultry, fishery and duck farming.

Earthworm is a terrestrial invertebrate belongs to phylum annelid and are able to convert garbage to "gold." Charles Darwin described as "Unheralded

Soilders of Mankind" as earthworm could digest various types of organic matter and Aristotle called as "Intestine of Earth." Earthworm influences soil physical properties, microflora, micro fauna; therefore, it is called as "drilosphere" (Pathma and Sakthivel, 2012) also called as "farmer's friend" as it enhances soil fertility and "natures ploughmen" or "ecosystem engineer." Earthworms are sources of high-quality animal protein, essential amino acids, nutrients, fats, vitamins and minerals (Ding et al., 2019; Byambas et al., 2019). Earthworms have high tendency of bioaccumulation of toxic organic compounds like pesticides, herbicides, antibodies and heavy metals (Byambas et al., 2019; Katagi and Ose, 2015; Sadia et al., 2020). Earthworms are classified as three types according to their ecotype, such as epigeics (those are living in upper surface of soil), endogeics (those are living in subsoil) and anecics (those are living in deep soil). Although all the three types are responsible for increasing soil fertility but only epigeics can be used in composting. The epigeics earthworms like *Eisenia fetida* (Boruah et al., 2019; Dohaish, 2020; Pottipati et al., 2021), *Eudrilus eugeniae* (Pandit et al., 2020; Rini et al., 2020; Pottipati et al., 2021), *Perionyx excavatus* (Rini et al., 2020; Pottipati et al., 2021*), Perionyx sansibaricus* (Suthar and Singh 2008), *Perionyx ceylanensis* (John Paul et al., 2011) and *Eisenia anderi* have been used in decomposing the organic matters. The epigeic earthworms are needed to be cultured for producing nutrient rich vermicompost.

2.3 Suitable earthworms

The suitability of earthworm for vermicomposting has been measured by considering certain criteria like, (1) easy to culture, (2) high affinity towards substrates for eating and (3) high rate of vermi cast production (Gajalakshmi and Abbasi, 2004). Generally, epigeic earthworms are advocated for vermicomposting as they are voracious feeders of organic matter. Epigeics are generally surface dweller in habitat and has greater potential than anecics. Because of ravenous feeding pattern, wider tolerability of organic waste, and high fecundity (ability to produce offspring's) epigeic earthworms are considered as most suitable earthworm for vermicomposting (Rini et al., 2020). The common epigeic earthworms used for vermicomposting are *E. fetida* (Pottipati et al., 2021), *E. eugeniae* (Pandit et al., 2020; Pottipati et al., 2021), *P. excavatus* (Rini et al., 2020; Pottipati et al., 2021). All the above earthworms are heavy feeders, and a possess high growth rate. The *E. fetida* is tolerant to wider temperature variation than *E. eugeniae* and *Perionyx excavates* (Gajalakshmi and Abbasi, 2004). The IVRI (Indian Veterinary Research Institute) has developed a new earthworm *Perionyx ceylanesis* named as Jai Gopal after mating *E. fetida and E. eugeniae*, which can tolerate heat up to 43°C (Singh et al., 2020). Tripathi and Bhardwaj (2004) reported that *E. fetida* has short maturity cycle, higher adaptability to several organic matters and high fecundity.

2.4 Substrates

Different types of substrates and substrate combinations have been used by many researchers for vermiculture and vermicomposting. The biodegradable substrates are generally used as a feeding material for earthworm. The substrates are generally of three types based on the production, such as (1) industrial wastes, (2) agricultural/farm waste and (3) municipal solid waste. Generally, municipality solid waste and industrial waste are to be segregated into biodegradable and non-biodegradable materials before vermicomposting. The industrial waste, such as tea waste combined with paper mill sludge and cow dung (Badhwar et al., 2020), coir industry waste with *Sesbania sesban* (Karmegam et al., 2021), silk and cotton processing sludge (Paul et al., 2022), brewers' spent grain (Saba et al., 2019), press mud amended with green manure crop (Balachandar et al., 2020), citronella bagasse with paper mill sludge (Boruah et al., 2019), paper industry sludge (Karmegam et al., 2019), bakery industry sludge (Yadav and Garg, 2019) and grape pomace waste with brewery waste yeast *Saccharomyces cerevisiae* (Rubio et al., 2020) has been used for producing nutrient rich vermicompost. Several agricultural/farm wastes like field crop residues, such as sugar cane trass (Biruntha et al., 2020), rice straw and maize stover (Pandit et al., 2020), wheat straw, millet straw, pulse bran (Suthar, 2009a, b); horticultural crop residues like cabbage, lettuce, carrot, potato and banana peel (Li et al., 2020; Das and Deka, 2021), java citronella (Deka et al. 2011a), weeds like, seaweeds, duckweed (Gusain and Suthar, 2020b), *Ageratum conyzoids* (Gusain and Suthar, 2020a), *Parthenium hysterophorus* (Yadav and Garg, 2011), *Lantana camara* (Devi and Khwairakpam 2020), *Eichhornia crassipes* (Gajalakshmi et al., 2002), Pistachio waste (Esmaeili et al., 2020), Pineapple waste (Zziwa et al., 2021) has been used for vermicomposting. Likewise, municipality soil wastes have been used as substrates in vermicomposting (Soobhany et al., 2017; John Paul et al., 2011). The vermicomposts have also been produced from household kitchen waste by means of biotechnological methods using *E. fetida* and *D. veneta* and constitute wholesome organic fertilizer of good microelements composition (Kostecka et al., 2018).

2.5 Growth and fecundity

Earthworm production is an important aspect of vermicompost production (Deka et al., 2011b; Ananthavalli et al., 2019). The growth of earthworms depends on the availability of food material and environmental factors. The earthworm's growth rate is dependent upon the initial feed mixtures, represented by the C/N ratio. The ideal C/N ratio is responsible for enhancing the growth and fecundity of the earthworms (Biruntha et al., 2020). The use of suitable food materials influences the growth and reproductive potential of earthworms (Sadia et al., 2020). The substrates' have non-assimilated carbohydrates and easily metabolised organic matter enhances the growth and fecundity of earthworms (Vodounnou et al., 2016; Balachandar et al., 2020). Belmeskine et al. (2020) found an increase (11.2%–44.2%) in the earthworm biomass after vermicomposting. The feed type and population density influence biomass (Neuhauser et al., 1980). Growth is also influenced by variations in the palatability and quality of the substrate (Gajalakshmi et al., 2005; Yadav and Garg, 2011; Balachandar et al., 2020). As poultry manure contains a higher amount of nitrogen and mineral, it is not recommended (Sherman, 2003). The feed having low nitrogen should be supplemented with proteinaceous material for easy degradation. the higher reproductive growth of the earthworms reported in the pure cattle dung (Yadav and Garg, 2011; Sharma and Garg, 2017). The mortality of earthworms was observed in vegetable waste due to the presence of sand in the substrate (Vodounnou et al., 2016). Co-composting substrates with cattle dung increase feed acceptability; hence, it is consider to be the best suitable mixture for highest growth and fecundity and low mortality (Bhat et al., 2016; Balachandar et al., 2020). Biochemical characteristics of feeding materials influence the rate of the cocoon (Yadav and Garg, 2011). The fungus in substrates acts as additional food for earthworms, which contributes higher biomass. Earthworm stocking density is also important and influences the feeding rate during vermicomposting. Optimum earthworm stocking density correlated to positive macro-nutrient mineralisation and reduction in toxic heavy metals of vermicompost. 10–12 g of worms per kg substrate are reported as the optimum earthworm stocking density (Mupambwa and Mnkeni, 2018).

2.6 Vermicomposting

The process of vermicomposting is "bio-oxidation and stabilisation of organic materials by the collaborative action of earthworm and microorganisms." It is the combination of both biological and chemical processes for recycling nutrients with the help of earthworms and microorganisms (Misra et al., 2003; Domínguez, 2004; Aira et al., 2007). Therefore, vermicompost is called a "highly bio-activated nutrient" source with various microbial communities (Pathma and Sakthivel, 2013). During vermicomposting, the organic wastes (Organic matter and mineral fraction bound with organic matter) undergo hydrolytic enzyme action and release free ions, organometal complex by humification and mineralisation process and a certain amount absorbed by earthworm tissue (Fig. 2.2). According to Garg et al. (2006), vermicomposting is a sustainable, cost-effective and eco-friendly approach for recycling hazardous organic wastes into valuable products. The organic matter passed through the earthworm's gut is quite different from the parent material and is a peat-like material favouring

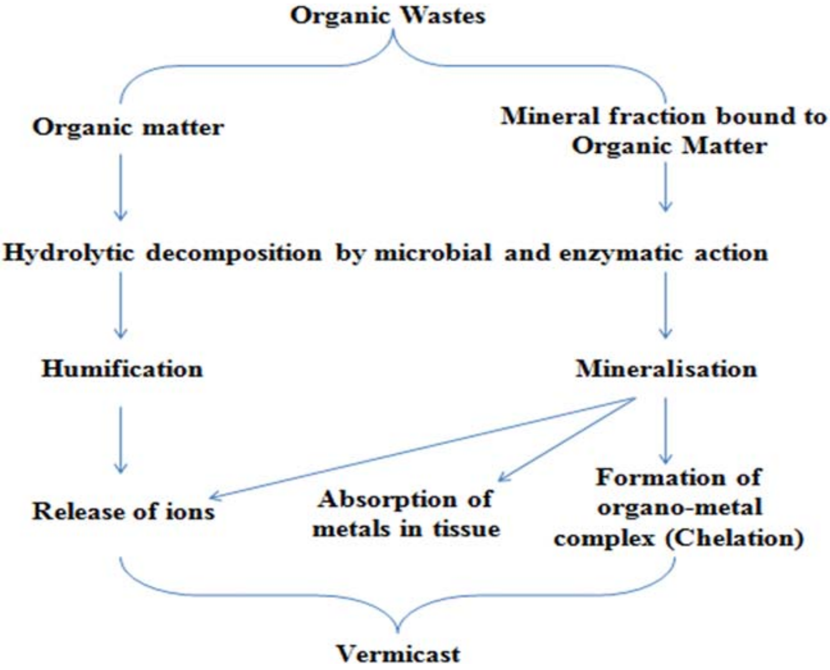


FIGURE 2.2 Role of earthworm during vermicomposting of organic wastes.

better drainage and water-holding capacity (Edwards and Burrows, 1988). The presence of earthworms in the composting process will stimulate bacterial and actinomycetes growth. Earthworms play the role of a conditioner of the substrate. Earthworm activity may accelerate the bio-oxidation of the substrate by increasing surface area for microbial activity (Domínguez, 2004). Earthworms eat and move in the substrate; this activity keeps turning the substrate frequently. By the earthworm treatment in vermicomposting, labour requirement could be reduced for turning the pile frequently; however, labour is needed for separating casting and earthworm when harvesting at the end of the process.

According to Ansari and Jaikishun (2011), vermicompost improves the soil's physicochemical and biological properties and also contributes to the organic enrichment of the soil. Bhole (1992) pointed out that vermitechnology can be used for city garbage recycling, management of effluents from livestock houses and decomposition of organic wastes from industries.

2.7 Change in physical properties of vermicompost

Generally, the physical appearance of vermicompost is fragmented, structurally homogeneous and crispy and dark brown/black in colour. The earthworm and enzyme activity on organic matter made them into fine humus-like substances (Karmegam et al., 2021). The ash content of vermicompost indicates the mineralisation of organic matter (Khwairakpam and Bhargava, 2009). The bulk density of vermicompost is generally decreased due to the decomposition of organic matter (Goswami et al., 2014; Karmegam et al., 2021) and particle size reduction (Hanc and Dreslova, 2016) during vermicomposting. The vermicompost or vermicast having high water-holding capacity and porosity (Pathma and Sakthivel, 2012) is desirable for agricultural use (Lim et al., 2015). The higher specific surface area provides high microsites for microbes to decay the living matter (Ganguly and Chakraborty, 2020). Moisture is an important factor for vermicomposting. Optimum moisture is a prerequisite for both, as well as earthworm activity. The moisture of 65%–90% (optimum 85%) is essential for maximum biological activity (Cincin and Agdag, 2019), and leachate circulation enhances solid waste degradation (Sponza and Ağdağ, 2004; Cincin and Agdag, 2019). A moisture content below 50% may cause dehydration of microorganisms (Liang et al., 2003). The particle size of vermicompost varies according to the nature of the substrates (Hanc and Pliva, 2013). A suitable particle size and shape of vermicompost are important for the soil (Wang and Ai 2016; Hanc and Dreslova, 2016). The larger particle size has less specific area than small size particles; as a result, the larger size particles are less accessible to microbes (Verma and Marschner, 2013; Hanc and Dreslova, 2016). Therefore, the release of minerals is more from finer particles than from the larger particles (Hanc and Dreslova, 2016).

2.8 Change in chemical properties

The pH refers to H^+ activity of a solution. The pH of vermicompost decreases in comparison to substrates. The organic acids and CO_2 production during the breakdown of complex organic compounds resulted in decreasing pH (Dominguez and Edwards, 2011; Pandit et al., 2020). The pH directly affects the microbial activity and mineralisation (Gusain and Suthar, 2020a). During initial stage of vermicomposting pH increases due to production of nitrogenous substance from degradation protein in organic matter (Yu et al., 2020) and decreases at the end due to nitrification and organic acid formations (Cáceres et al., 2016), volatilisation of ammonia (Yu et al., 2020), moisture content and earthworm activity (Sáez et al., 2021; Karmegam et al., 2021). The optimum pH for earthworm functioning is 7.0 (Ansari and Rajpersaud, 2012; Karwal and Kaushik, 2020). The trend of change of pH during vermicomposting is not constant, while pH decreased during vermicomposting of water hyacinth and grass waste (Ansari and Rajpersaud, 2012; Cai et al., 2018), lignocellulosic agro-wastes (Pandit et al., 2020), kitchen waste (Karwal and Kaushik, 2020) whereas increase in sewage sludge vermicomposting (Nayak, Varma, and Kalamdhad, 2013). The pH is an important factor for nutrient availability in soil. In low-pH soil, macronutrient availability will be less, whereas micronutrients availability will be more, and the reverse in high-pH soil. Thus, a near-neutral or neutral pH vermicompost production is desirable for making the availability of plant nutrients. Vermicompost of mixed cattle dung and fly ash using *E. fetida* showed lower electric conductivity (EC) and drop in pH value and lower C/N ratio as compared to feed mixtures without earthworms (Sohal et al., 2021).

The nontoxic level of electrical conductivity to plants is >4 Ms m^{-1} (Wong et al., 2001). The electrical conductivity (EC) increases during vermicomposting due to nitrification; acidifications transform ammoniacal N and other minerals to the labile form of cations (Gusain and Suthar, 2020a). The higher concentration of soluble salts resulted ina higher EC of vermicompost (Yadav and Garg, 2011; Pandit et al., 2020). The reduction of electrical conductivity was reported during vermicomposting of fruit and vegetable waste (Li et al., 2020), and it was due to the loss of dissolved ions during the decomposition of organic matter (Li et al., 2020). The vermicompost that has an EC of < 4.0 dSm^{-1} is the minimum standard for recommending an organic compost quality standard of EC (Central Public Health and Environmental Engineering Organization CPHEEO, 2016; Balachandar et al., 2021). Organic matters are rich in carbon. Several researchers have reported that there was a reduction of total organic carbon (TOC) during vermicomposting. The reduction of TOC reflects the richness of humic substances and the stability and maturity of vermicompost (Pandit et al., 2020; Srivastava et al., 2020). The reduction of TOC is due to the consumption of organic matter by earthworms as an energy source by the microbes and loss to the atmosphere as carbon dioxide through respiration (Alidadi et al., 2016; Sharma and Garg, 2017; Balachandar et al., 2021; Rai et al., 2021). According to Gusain and Suthar (2020a) microbial respiration is only one key for C reduction during vermicomposting. A higher concentration of cellulose in the substrate produces hydrosoluble carbon (Yu et al., 2020), which is easily available to microbes; hence their population increased (Gusain and Suthar, 2020b). The digestion of different polysaccharide by vermiworms during vermicomposting may reduce the carbon content and some part may convert into earthworm biomass through assimilation (Suthar, 2010). The reduction of TOC depends on the substrate quality; for food and vegetable wastes, TOC reduced ranging from 23.5% to 48.3% (Sharma and Garg, 2017), 21.7% in municipal solid wastes (Srivastava et al., 2020), 34.0%–51.0% in different agro-wastes (Pandit et al., 2020).

The cation exchange capacity (CEC) of substrates during vermicomposting increased in comparison to control and normal compost and varied according to the substrate quality (Pandit et al., 2020) and earthworm species (Karmegam et al., 2021). The enhancement of CEC during vermicomposting was because of the accumulation of negatively charged compounds, carboxylic and phenolic groups during the humification process (Fornes et al., 2012). According to Goswami et al. (2014), the higher CEC in vermicompost is due to organic matter fraction and associate chemicals changes by earthworms and microflora. Higher CEC indicates a higher nutrient-supplying capacity.

2.9 Change in nutritional properties

The plant nutrient content increases during vermicomposting due to the volume reduction of substrates and degradation of complex organics. Several researchers have reported that the total N, P, K, Ca, Mg and S concentrations increases during vermicomposting of different substrates (Table 2.1). During the partial decomposition of substrates, the extent of increase in nutrient concentration is less compared to the final vermicompost product (Pandit et al., 2020). Total N concentration increases due to mineralisation, polysaccharides, production of nitrogenous excretory products, mucus, hormones produced by earthworms and reduction in volume (Srivastava et al., 2020). The addition of green manure crops also increased the total N content of vermicompost (Karmegam et al., 2021). Vermicompost of mixed cattle dung and fly ash using *E. fetida* showed higher nutrient content (Total kjeldahl nitrogen (TKN), total available phosphorus (TAP), Total potassium (TK), and Total Natrium (TNa)) (Sohal et al., 2021). Bioconversion of organic waste is an effective method of converting fixed phosphorus into available (Bhat et al., 2016). The total P increases due to phosphatase activity and action of P-solubilizing bacteria on the mineralisation of P from organic matter (Ghosh et al., 2018; Karmegam et al., 2019). An increase in potassium (K) concentration is due to the decomposition of organic matter and volume reduction of substrates during vermicomposting (Thomas et al., 2018). The concentration of Ca and Mg increased in vermicompost than the substrate due to organic matter degradation and volume reduction of feedstock during the vermicomposting process (Karmegam et al., 2021). During vermicomposting, the mineralisation of organically bound micronutrients transforms into free form by earthworms.

Heavy metals are metals of higher density and atomic weight. The higher concentration of these elements leads to toxicity in plant tissue. Therefore, it should be examined the concentration of any agro-product before use in the field. The concentration of heavy metals in vermicompost should be tested before use. During vermicomposting, due to mineralisation and humification by several enzyme actions, the short-chain organic acid binds with metal ions and produces a stable complex (Hafeez et al., 2013; He et al., 2016; Karwal and Kaushik, 2020). The decreased concentration of heavy metals in vermicompost than substrate is due to bioaccumulation in tissues of earthworms (Wang et al., 2018) during ingestion and digestion. The absorbed heavy metals are bound with low molecular weight (6–7 kDa) (Babula et al., 2012) cysteine-rich metal binding proteins (metallothioneins) (Yuvaraj et al., 2020) as indicated in Fig. 2.3. Wang et al. (2013) have reported that the

TABLE 2.1 Change in nutrient content during vermicomposting of different organic waste.

Substrates	Nitrogen (%)		Phosphorus (%)		Potassium (%)		Calcium (%)		Magnesium (%)		Sulphur (%)	
	Initial	Final	Initial	Final	Initial	Final	Initial	Final	Initial	Final	Initial	Final
Paddy straw	0.72^{d*}	1.99^{c}	0.64^{c}	1.22^{b}	1.06^{bc}	1.49^{b}	0.40^{a}	0.58^{b}	0.28^{a}	0.49^{a}	0.17^{a}	0.21^{a}
Maize stover	0.44^{c}	1.49^{ab}	0.49^{ab}	0.92^{a}	0.82^{b}	1.20^{ab}	0.53^{b}	0.58^{b}	0.35^{a}	0.45^{a}	0.14^{a}	0.19^{a}
Leaf litter	0.76^{e}	1.99^{c}	0.55^{bc}	0.88^{a}	0.53^{a}	1.10^{a}	0.97^{c}	1.09^{c}	0.63^{b}	0.74^{b}	0.89^{c}	0.94^{c}
Vegetables waste	0.39^{b}	1.43^{a}	0.99^{d}	1.25^{b}	0.87^{b}	1.45^{b}	0.42^{ab}	0.54^{b}	0.26^{a}	0.90^{c}	1.00^{d}	1.16^{d}
Temple waste flowers	0.35^{a}	1.56^{b}	0.41^{a}	0.86^{a}	1.29^{c}	1.80^{c}	0.32^{a}	0.45^{a}	0.28^{a}	0.41^{a}	0.43^{b}	0.62^{b}

*Different superscripted letters is statistically different (ANOVA; Duncan's multiple test range, $P \leq .05$).
Source: Pandit, L., Sethi D., Pattanayak, S.K., Nayak, Y., 2020. Bioconversion of lignocellulosic organic wastes into nutrient rich vermicompost by Eudrilus eugeniae. Bioresour. Technol. Rep. 12,100580.

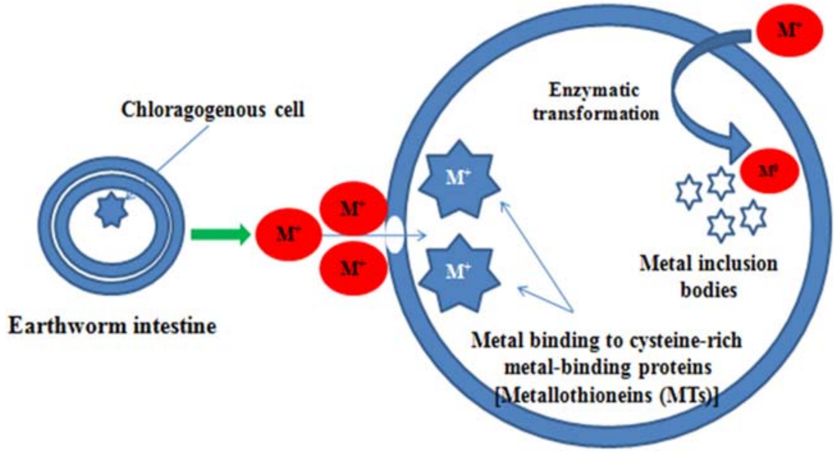


FIGURE 2.3 Schematic representation of metal bioaccumulation during vermicomposting.

cutaneous tissues of earthworm can easily absorbable heavy metals. The Pb inhibits plant growth, chlorophyll formation and seed germination by inhibiting essential enzyme action and electron transport chain (Pourrut et al., 2011). Cadmium (Cd) and lead (Pb) are not essential elements of plants, but they inhibit the metal-sensitive enzyme action, which leads to plant death (Karwal and Kaushik, 2020). The Pb concentration in the substrate during vermicomposting has been reported by Karwal and Kaushik (2020) during vermicomposting of lawn and kitchen waste and during vermicomposting of sludge (Liu et al., 2012). The Pb concentration was reduced during vermicomposting of sludge, and the Pd concentration accumulated more in earthworm tissue (Liu et al., 2012). The As concentration of substrate was reduced from 36% to 49% during vermicomposting (Karwal and Kaushik, 2020). It is necessary to decrease the concentration of As in the agro-inputs as it is easily assimilated in plants. It has a high affinity towards plant thiol-containing enzymes and compounds. Elevated affinity towards the sulfhydryl group inhibits the vegetative growth of plants. Farooq et al. (2016) reported that there was a reduction of As content vermicomposting of waste sludge. Cadmium (Cd) inhibits chlorophyll synthesis and Chromium (Cr) inhibits seed germination, plant growth synthesis and oxidative stress (Andresen and Küpper, 2013), so their concentration in vermicompost needs to be tested before using for agricultural purposes. According to the fertilizer control order of the Government of India (FCO-1985), the permissible limit of Cd, Cr and Pb in the vermicompost should be $\leq 5\ \mathrm{mg\ kg^{-1}}$, $\leq\ 50\ \mathrm{mg\ kg^{-1}}$ and $\leq 100\ \mathrm{mg\ kg^{-1}}$, respectively.

Several researchers have also reported that the higher metal concentration in the earthworm growing environment leads to high mortality (Spurgeon et al., 1994, 2000), cocoon production, cocoon viability (Spurgeon and Hopkin, 1996) and loss of biomass (Lemtiria et al., 2016). The potentially toxic heavy metal (PTHM) content of mixed fly ash-cattle dung observed decline in post-vermicompost mixtures due to earthworm's activity (Sohal et al., 2021).

2.10 Change in biological properties

Microorganisms are the major architecture of the biochemical conversion of waste to bioactive enriched compost, where earthworms indirectly stimulate microbes for their growth and regeneration through degradation of organic matter. The earthworm regulates microbial populations by digestion, stimulation, and spreading in casts. The population of microbes and their activity are directly proportional to the enzyme activity. The microbial population is also influenced by substrate composition.

In the nutrient and energy flow relationship, earthworms play a very important role (Goswami et al., 2018). Earthworm activity decreases the growth rate of bacteria but does not affect fungal growth rates. The fungi population is relatively high in vermicompost than in normal compost (Goswami et al., 2018). Ananthavalli et al. (2019) reported that vermicomposting enhances the microflora in organic matter, which is vital for decomposition. Several beneficial microbes are being protected by earthworm's intestines coming through the cast (Hussain et al., 2016; Goswami et al., 2018). Animal manures are rich in microbes, where bacteria constitute the largest fraction followed by fungi. The first stages of decomposition of organic wastes are dominated by bacteria because of the availability of water and easily decomposable substrates. Carbon availability is a limiting factor for earthworm growth. According to Tiunov and Scheu (2004) both earthworms and microorganisms compete with each other for carbon resources. The enzymes secreted in the earthworm's gut provide a better environment and nutrients for microbes. The earthworm casts are the combination of material ingested by the earthworms, which can be a resource of readily assimilable compounds (Brown and Doube 2004; Gómez-Brandón et al., 2011). The microbial diversity depends on substrate composition (Chen et al., 2018). Bacterial and fungal community composition changed significantly after gut-associated processes and during cast-associated processes of sewage sludge vermicomposting. Vermicomposting of sewage resulted in a stable and rich microbial community with potential biostimulant properties that may aid plant growth (Domínguez et al., 2021).

Generally, earthworms ingest PGPR (*Rhizobium, Pseudomonas*, *Bacillus, Azosprillium*, *Azotobacter*, etc.) along with soil and are activated under the favourable microenvironment of the gut and increase the population (Sinha et al., 2010; Pathma and Sakthivel 2012). These bacteria stimulate plant growth by increasing essential nutrient's bioavailability (Ayyadurai et al., 2007; Naik et al., 2008), 1-aminocyclopropane-1-carboxylate deaminase, growth hormone production (Correa et al., 2004), N_2 fixation (Han et al., 2005), and indirectly by suppressing fungal pathogens. Earthworms increase microorganism population fivefold in soil (Edwards and Lofty, 1977; Pathma and Sakthivel, 2012); total bacteria, proteolytic bacteria and actinomycetes population increases by passing through earthworms' gut (Devliegher and Verstraete, 1995). Large changes in bacterial community composition were found after the transit of the sewage sludge through the gut of the earthworms (GAP), with significant decreases in the abundance of Campilobacterota, Firmicutes and Bacteroidota and significant increases in the abundance of Verrucomicrobiota, Proteobacteria and Bacteroidota (Domínguez et al., 2021).

The enzyme activity of vermicompost is directly proportional to the microbial activity. Various extracellular enzymes came out with vermicast from the elementary canal of earthworms. The hydrolytic enzyme produced either from the microbes of earthworm gut or environmental microbes acted upon the substrate for degradation and mineralisation of nutrients (Fig. 2.4). Earthworm helps in the colonisation of microbes in substrates mixture (Sudkolai and Nourbakhsh, 2017; Karwal and Kaushik, 2020). The organic matter decomposition facilitated by different hydrolytic enzymes (proteases, dehydrogenases (DHA), β-galactosidase and phosphatases) during vermicomposting (Wyszkowska and Wyszkowski, 2006; Zhang and Sun, 2018).

Organic matter oxidation is a dehydrogenation process, which is mediated by the dehydrogenase enzyme and its activity declines with increasing organic matter degradation (Karmegam et al., 2021). The dehydrogenase

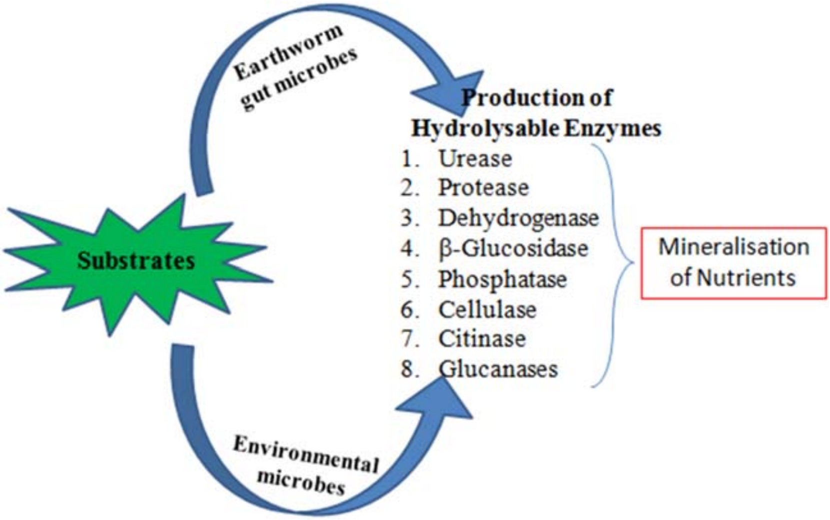


FIGURE 2.4 Different types of hydrolytic enzymes action on the substrate for nutrient mineralisation.

activity indicates the increasing activity of hydrolytic enzymes (Fernández-Gómez et al., 2011). During the oxidation of organic matter, ATP synthesis is catalysed by the dehydrogenase enzyme (Gusain and Suthar, 2020a). Earthworms have a strong effect on the dehydrogenase activity during the first 60 days of their activity. Till 30 days it increased two times during vermicomposting of lawn waste and kitchen waste (Karwal and Kaushik, 2020). The enhancement of DHA activity is due to the higher availability of labile carbon during lignocelluloses degradations (Voberková et al., 2017; Hrebeckova, Wiesnerova, and Hanc, 2019; Yu et al., 2020). The urease activities involve in mineralisation of organic ureic compounds. The ®-glucosidase and ureases are involved in organic matter degradation in vermicomposting (Benítez et al., 2005). During vermicomposting the urease activity initially increases (Ke et al., 2010) and declines after 30 days till the harvest (Sudkolai and Nourbakhsh, 2017). Urease transforms complex N-compounds into simpler N-forms (Swati and Hait, 2017). The initial amplification of urease enzyme activity was due to a high level of NH_4^+ in the substrates (Karwal and Kaushik, 2020). The β-glucosidase is mainly involved in the degradation of cellulose into glucose and plays a crucial role in C-cycle (Zhang and Sun, 2018). The ®-glucosidase is synthesised and released during the vermicomposting process, and it indicates ahigher rate of organic matter decomposition. The β-glucosidase is sustained for a prolonged period during the vermicomposting due to the slow rate of cellulose degradation (Gusain and Suthar, 2020a). Earthworm-mediated nutrient mineralisation supports microbial growth and enzyme syntheses, which are responsible for C-cycle (Rai and Suthar, 2020). The phosphatase enzyme is considered an enzyme of agronomic importance (Karwal and Kaushik, 2020). Organic phosphate is transformed into a labile form of phosphorus through extracellular enzymes, such as acid phosphatase and alkaline phosphatase (Hrebeckova et al., 2019). The phosphatase enzyme activity in the substrate mixture indicates the presence of organic phosphate matter and high microbial activity (Raut et al., 2008). The phosphatase activity increases till 30 days and after that, declines till the harvest; a similar trend was reported during vermicomposting of lawn waste (Karwal and Kaushik, 2020), olive oil mill wastewater (Macci et al., 2010), filter cake (Busato et al., 2016). The rapid increase in phosphatase was recorded between the period 5–10 days; thereafter, it declined until the end of the vermicomposting process (Gustain and Suthar, 2020). The phosphatase enzyme activity transforms organic P into inorganic/easily available P. Generally, phosphatase activity shows a declining trend towards the end of the vermicomposting process. The enzyme activity of microbes in a sample can be estimated by using a DFA hydrolysis assay. The fungal growth suppresses due to the production of antibiotics, fluorescent pigments, siderophores and fungal cell-wall degrading enzymes, namely chitinases and glucanases by bacteria (Pathma et al., 2010; Pathma et al., 2011a,b). Cellulase enzyme helps in the degradation of cellulose in the substrate, and its activity is reduced with the completion of organic matter degradation (Karmegam et al., 2021). The oligopeptides and polypeptides are responsible for elevated protease activities (Voberková et al., 2017). Initial proteolytic activities in worm gut produces worm cast and further hydrolysis of proteolytic hydrolates is performed by microbial communities inhabiting in deposited casts (Fernández-Gómez et al., 2011). Protease activity prolongs for more periods during vermicomposting; it is due to protein depolymerisation and further hydrolysis of polypeptides (Hrebeckova et al., 2019). Proteases increase rapidly at day 10 of *A. conyzoids* vermicomposting; thereafter, they decrease continuously until the end of the process (Gusain and Suthar, 2020a,b). The organic matter aromaticity increased until the 45th day, subsequently decreasing.

2.11 Maturity indices

Compost quality is closely related to its stability and maturity, which cannot be established by a single parameter (Bernal et al., 2009). Several parameters, such as temperature, organic matter loss, C/N ratio, cation exchange capacity, humification index and seed germination index, have been proposed for evaluating compost stability. A high degree of stability and maturity is the principal requirement of a compost to be safely used in soil. It implies a stable OM content and the absence of phytotoxic compounds and plants or animal pathogens (Bernal et al., 2009).

The C:N, C:P and C:S ratios are very important parameters for vermicompost. These parameters indicate the maturity of vermicompost and the acceptability of compost for agricultural use. The narrowing of C:N, C:P and C:S ratios in vermicomposting was due to microbial respiration and escalating N concentration due to mineralisation and volume reduction (Swarnam et al., 2016). Several researchers agreed that the C:N ratio $<$ 20:1, indicating an advanced degree of organic matter stabilisation and organic wastes maturity, whereas a ratio $<$ 15:1 is preferred as composts in crop production (Soobhany et al., 2017; Pandit et al., 2020). According to the chemical, physical, and biological properties analysis, the vermicomposting process can be divided into initial ($<$45 days) and final (45–120 days) phases. Furthermore, the initial phase was characterised by high microbial activity and the final by high physical–chemical transformation of the vermicompost and an increase in earthworm density.

Higher microbial biomass carbon (MBC) indicates the higher activity of microbes. Microbial quotient (MQ) is defined as the ratio of MBC and TOC. The MQ is one of the most dependable maturity indices (Goswami et al., 2018). The MQ value between lies 1–4 is considered as healthy for the biological system (Bhattacharyya et al., 2008). Less MQ indicates the microbial stress condition (Tripathy et al., 2014), and higher MQ (MBC/TOC) indicates higher substrate usage efficiency by microbes.

Instrumental analysis of vermicompost is a modern technique that provides necessary information on maturity (Bhat et al., 2017). Several instruments are used for analysing vermicomposts, such as Fourier-transform infrared (FT-IR), spectroscopy, ultraviolet-visible (UV-vis) spectroscopy, thermogravimetry (TG) and scanning electron microscopy (SEM) techniques.

The SEM analysis provides the surface morphology. Higher roughness and pores over the surface of vermicompost indicate superior decomposition of organic matter (Pandit et al., 2020). The final vermicompost revealed a higher fragmented texture than the initial waste mixture (Bhat et al., 2017). The FT-IR analysis indicates different functional groups present in vermicompost, which indicates organic matter decomposition. The presence of functional groups in vermicompost indicates mineralisation. The TG method is used to characterise organic waste mineralisation. The TG graph provides information about the loss of mass, mineralisation and decomposition of organic waste under different temperatures. UV-vis spectroscopy gives information on the degree of humification. The sharp fall in the humification index during vermicomposting indicates a high level of organic material humification. A long length of 30 days vermicomposting issufficient to obtain a high-quality vermicompost for organic fertilizer; however, 120 days are necessary for producing matrices.

2.12 Conclusion

The bioconversion of organic waste through vermitechnology is an important eco-friendly approach. Efficient earthworm selection is very important for the verniculture process. Earthworms having a higher growth, fecundity and conversion ability are suitable for vermicomposting. The physicochemical and biological properties of substrates influence the final product. The carbon content reduces during vermicomposting, and other nutrients concentration increases. The vermicomposts having C:N below 15 are suitable for agronomical use.

Contribution

Conceptualisation and planning - Konathala Kusumavathi, Debadatta Sethi, Kshitipati Pradhan, Narayan Panda, Sushanata Kumar Pattanayak; Contributed the manuscript writing - Debadatta Sethi, Konathala Kusumavathi, Subhaprada Dash, Tapas Ranjan Sahoo, Satyabrata Mangaraj, Andi Febrisiantosa; Review and Editing - Sushanata Kumar Pattanayak, Arabinda Dhal, Sushanata Kumar Swain, Andi Febrisiantosa. All authors have read and agreed to the published version of the manuscript.

Funding

This research received no external funding.

Institutional review board statement

Not applicable.

Informed consent statement

Not applicable.

Data availability statement

Not applicable.

Acknowledgements

Not applicable

Conflicts of interest

The authors declare no conflict of interest.

References

Aira, M., Monroy, F., Dominguez, J., 2007. Microbial biomass governs enzymes activity decay during aging of worm-worked substarates through vermicomposting. J. Environ. Qual. 36, 448–452.

Alidadi, H., Hosseinzadeh, A., Najafpoor, A.A., Esmaili, H., Zanganeh, J., Dolatabadi, T.M., et al., 2016. Waste recycling by vermicomposting: maturity and quality assessment via dehydrogenase enzyme activity, lignin, water soluble carbon, nitrogen, phosphorous and other indicators. J. Environ. Manage. 182, 134–140.

Ananthavalli, R., Ramadas, V., Arockia, J., Paul, J., 2019. Seaweeds as bioresources for vermicompost production using the earthworm, *Perionyx excavatus* (Perrier). Bioresour. Technol. 275, 394. –01.

Andresen, E., Küpper, H., 2013. Cadmium toxicity in plants, in Cadmium: from toxicity to essentiality. Springer 1, 395–413. Available from: https://doi.org/10.1007/978-94-007-5179-8.

Annepu, R.K., 2012. Sustainable solid waste management in India. MS Dissertation 1–189. Available from: https://doi.org/10.1007/978-981-4451-73-4.

Ansari, A.A., Jaikishun, S., 2011. Vermicomposting of sugarcane bagasse and rice straw and its impact on the cultivation of Phaseolus vulgaris L. in Guyana, South America. J. Agric. Technol. 7 (2), 225–234.

Ansari, A.A., Rajpersaud, J., 2012. Physicochemical changes during vermicomposting of water hyacinth (*Eichhornia crassipes*) and grass clippings. ISRN Soil. Sc. 2, 1261–1262. Available from: https://doi.org/10.5402/2012/984783.

Ayyadurai, N., Naik, P.R., Sakthivel, N., 2007. Functional characterization of antagonistic fluorescent pseudomonads associated with rhizospheric soil of rice (*Oryza sativa* L.). J. Microbiol. Biotechnol. 17, 919−927.

Babula, P., Masarik, M., Adam, V., Eckschlager, T., Stiborova, M., Trnkova, L., et al., 2012. Mammalian metallothioneins: properties and functions. Metallomics 4, 739. Available from: https://doi.org/10.1039/c2mt20.

Badhwar, V.K., Singh, S., Singh, B., 2020. Biotransformation of paper mill sludge and tea waste with cow dung using vermicomposting. Bioresur. Technol. 318, 124097. Available from: https://doi.org/10.1016/j.biortech.2020.124097.

Balachandar, R., Baskaran, L., Yuvaraj, A., Thangaraj, R., Subbaiya, R., Ravindran, B., et al., 2020. Enriched pressmud vermicompost production with green manure plants using *Eudrilus eugeniae*. Bioresour. Technol. Available from: https://doi.org/10.1016/j.biortech.2019.122578.

Balachandar, R., Biruntha, M., Yuvaraj, A., Thangaraj, R., Subbaiya, R., Govarthanan, M., et al., 2021. Earthworm intervened nutrient recovery and greener production of vermicompost from Ipomoea staphylina − an invasive weed with emerging environmental challenges. Chemosphere. 263, 128080. Available from: https://doi.org/10.1016/j.chemosphere.2020.128080.

Belmeskine, H., Ouameur, W.A., Dilmi, N., Aouabed, A., 2020. The vermicomposting for agricultural valorization of sludge from Algerian wastewater treatment plant: impact on growth of snap bean Phaseolus vulgaris L. Heliyon 6, 04679.

Benítez, E., Sainz, H., Nogales, R., 2005. Hydrolytic enzyme activities of extracted humic substances during the vermicomposting of a lignocellulosic olive waste. Bioresour. Technol. 96, 785−790.

Bernal, M.P., Alburquerque, Ja, Moral, R., 2009. Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresour. Technol. 100 (22), 5444−5453. Available from: https://doi.org/10.1016/j.biortech.2008.11.027.

Bhattacharyya, P., Tripathy, S., Chakraborty, K., Chakraborty, A., Banik, P., 2008. Fractionation and bioavailability of metals and their impacts on microbial properties in sewage irrigated soil. Chemosphere. 72, 543−550.

Bhat, S.A., Singh, J., Vig, A.P., 2017. Instrumental characterization of organic wastes for evaluation of vermicompost maturity. J. Anal. Sc. Tech. 8, 2. Available from: https://doi.org/10.1186/s40543-017-0112-2.

Bhat, S.A., Singh, J., Vig, A.P., 2016. Effect on growth of earthworm and chemical parameters during vermicomposting of pressmud sludge mixed with cattle dung mixture. Procedia Environ. Sci. 35, 425−434.

Bhole, J.R., 1992. Vermiculture biotechnology-Basic, scope for applications and development. Natl Semin. Org. Farming 50−51.

Biruntha, M., Karmegam, N., Archana, J., Selvi, B.K., Paul, J.A.J., Balamuralikrishnan, B., et al., 2020. Vermiconversion of biowastes with low-to-high C/N ratio into value added vermicompost. Bioresour. Technol. 297, 122398. Available from: https://doi.org/10.1016/j.biortech.2019.122398.

Boruah, T., Barman, A., Kalita, P., Lahkar, J., Deka, H., 2019. Vermicomposting of citronella bagasse and paper mill sludge mixture employing *Eisenia fetida*. Bioresour. Technol. 294, 122147. Available from: https://doi.org/10.1016/j.biortech.2019.122147.

Brown, G.G., Doube, B.M., 2004. Functional interactions between earthworms, microorganisms, organic matter, and plants. In: Edwards, C.A. (Ed.), Earthworm Ecology, 2nd ed. CRC Press, Boca Raton, pp. 213−224.

Busato, J.G., Papa, G., Canellas, L.P., Adani, F., de Oliveira, A.L., Leão, T.P., 2016. Phosphatase activity and its relationship with physical and chemical parameters during vermicomposting of filter cake and cattle manure. J. Sci. Food. Agric. 96, 1223–1230. Available from: https://doi.org/10.1002/jsfa.7210.

Byambas, P., Hornick, J.L., Marlier, D., Francis, F., 2019. Vermiculture in animal farming: a review on the biological and nonbiological risks related to earthworms in animal feed. Cogent. Environ. Sci. 5 (1), 1591328. Available from: https://doi.org/10.1080/23311843.2019.1591328.

Cai, L., Gong, X., Sun, X., Li, S., Yu, X., 2018. Comparison of chemical and microbiological changes during the aerobic composting and vermicomposting of green waste. PLoS ONE. 13, e0207494. Available from: https://doi.org/10.1371/journal.pone.0207494.

Central Pollution Control Board (CPCB) India, 2018. MSW- Generation & Composition CPCB 2018. https://cpcb.nic.in/uploads/MSW/Waste_generation_Composition.pdf.

Central Public Health and Environmental Engineering Organization (CPHEEO), 2016. Swachh Bharat Mission- Municipal Solid Waste Management Manual. Ministry of Urban Development . Available from: http://moud.gov.in/pdf/57f1e55834489Book03.pdf.

Chen, Y., Chang, S.K.C., Chen, J., Zhang, Q., Yu, H., 2018. Characterization of microbial community succession during vermicomposting of medicinal herbal residues. Bioresour. Technol. 249, 542–549.

Cincin, R.G.Y., Agdag, O.N., 2019. Co-vermicomposting of wastewater treatment plant sludge and yard waste: investigation of operation parameters. Waste Bio. Valor . Available from: https://doi.org/10.1007/s12649-019-00900-w.

Correa, J.D., Barrios, M.L., Galdona, R.P., 2004. Screening for plant growth promoting rhizobacteria in *Chamaecytisus proliferus* (tagasaste), a forage tree-shrub legume endemic to the Canary Islands. Plant. Soil. 266, 75–84.

Cáceres, R., Coromina, N., Malińska, K., Martínez-Farré, F.X., López, M., Soliva, M., et al., 2016. Nitrification during extended co-composting of extreme mixtures of green waste and solid fraction of cattle slurry to obtain growing media. Waste Manag. 58, 118–125.

Das, D., Deka, H., 2021. Vermicomposting of harvested waste biomass of potato crop employing *Eisenia fetida*: changes in nutrient profile and assessment of the maturity of the end products. Environ. Sci. Pollut. Res. 28, 35717–35727. Available from: https://doi.org/10.1007/s11356-021-13214-z.

Deka, H., Deka, S., Baruah, C.K., Das, J., Hoque, S., Sarma, H., et al., 2011a. Vermicomposting potentiality of *Perionyx excavatus* for recycling of waste biomass of java citronella - An aromatic oil yielding plant. Bioresour. Technol. 102 (24), 11212–11217. Available from: https://doi.org/10.1016/j.biortech.2011.09.102.

Deka, H., Deka, S., Baruah, C.K., Das, J., Hoque, S., Sarma, H., et al., 2011b. Vermicomposting of distillation waste of citronella plant (*Cymbopogon winterianus* Jowitt) employing *Eudrilus eugeniae*. Bioresour. Technol. 102, 6944–6950.

Devi, C., Khwairakpam, M., 2020. Bioconversion of *Lantana camara* by vermicomposting with two different earthworm species in monoculture. Bioresour. Technol. 296, 122308. Available from: https://doi.org/10.1016/j.biortech.2019.122308.

Devliegher, W., Verstraete, W., 1995. *Lumbricus terrestris* in a soil core experiment: nutrient-enrichment processes (NEP) and gut-associated processes (GAP) and their effect on microbial biomass and microbial activity. Soil. Biol. Biochem. 27, 1573–1580.

Ding, S., Lin, X., He, S., 2019. Earthworms: a source of protein. J. Food Sci. Eng. 9, 159–170. Available from: https://doi.org/10.17265/2159-5828/2019.05.001.

Dohaish, E.J.A.B., 2020. Vermicomposting of organic waste with Eisenia fetida increases the content of exchangeable nutrients in soil. Pak. J. Biol. Sc. 23 (4), 501–509. Available from: https://doi.org/10.3923/pjbs.2020.501.509.

Dominguez, J., Edwards, C.A., 2011. Biology and Ecology of Earthworm Species Used for Vermicomposting Vermiculture Technology: Earthworms, Organic Wastes, and Environmental Management. CRC Press, USA.

Domínguez, J., Aira, M., Crandall, K.A., et al., 2021. Earthworms drastically change fungal and bacterial communities during vermicomposting of sewage sludge. Sci. Rep. 11, 15556. Available from: https://doi.org/10.1038/s41598-021-95099-z.

Domínguez, J., 2004. State of the art and new perspectives on vermicomposting research. In: Edwards, C.A. (Ed.), Earthworm Ecology, second ed. CRC Press, Boca Raton, FL, pp. 401–424.

Edwards, C.A., Burrows, I., 1988. The potential of earthworm composts as plant growth media. In: Edwords, C.A., Neuhauser, E. (Eds.), Earthworms in Waste and Environmental Management. The Hauge, The, Netherlands, pp. 21–32.

Edwards, C.A., Lofty, R., 1977. The Biology of Earthworms. Chapmann and Hall, London.

Esmaeili, A., Khoram, M.R., Gholami, M., Eslami, H., 2020. Pistachio waste management using combined composting-vermicomposting technique: physico-chemical changes and worm growth analysis. J. Clean. Prod. 242, 118523. Available from: https://doi.org/10.1016/j.jclepro.2019.118523.

Farooq, M.A., Islam, F., Ali, B., Najeeb, U., Mao, B., Gill, R.A., et al., 2016. Arsenic toxicity in plants: cellular and molecular mechanisms of its transport and metabolism. Environ. Exp. 132, 42–52. Available from: https://doi.org/10.1016/j.envexpbot.2016.08.004.

Fernández-Gómez, M.J., Nogales, R., Insam, H., Romero, E., Goberna, M., 2011. Role of vermicompost chemical composition, microbial functional diversity, and fungal community structure in their microbial respiratory response to three pesticides. Bioresour. Technol. 102, 9638–9645.

Fornes, F., Mendoza-Hernandez, D., Garcia-de-la-Fuente, R., Abad, M., Belda, R.M., 2012. Composting versus vermicomposting: a comparative study of organic matter evolution through straight and combined processes. Bioresour. Technol. 118, 296–305.

Gajalakshmi, S., Abbasi, S.A., 2004. Earthworms and vermicomposting. Ind. J. Biotech. 3, 486–494.

Gajalakshmi, S., Ramasamy, E.V., Abbasi, S.A., 2002. High-rate composting–vermicomposting of water hyacinth (*Eichhornia crassipes*, Mart. Solms). Bioresour. Technolo. 83 (3), 235–239.

Gajalakshmi, S., Ramasamy, E.V., Abbasi, S.A., 2005. Composting– vermicomposting of leaf litter ensuing from the trees of mango (*Mangifera indica*). Bioresour. Technol. 96, 1057–1061.

Ganguly, R.K., Chakraborty,S.K., 2020. Eco-management of industrial organic wastes through the modified innovative vermicomposting process: a sustainable approach in tropical countries (Chapter 10), Earthworm Assisted Remediation of Effluents and Wastes, Edited by Bhat SA, Vig AP, Li F, Ravindran B. Springer Nature Singapore, doi.org/10.1007/978-981-15-4522-1.

Garg, P., Gupta, A., Satya, S., 2006. Vermicomposting of different types of waste using Eiseniafetides: a comparative study. Bioresour. Technol. 97, 391–395.

Ghosh, S., Goswami, A.J., Ghosh, G.K., Pramanik, P., 2018. Quantifying the relative role of phytase and phosphatase enzymes in phosphorus mineralization during vermicomposting of fibrous tea factory waste. Ecol. Eng. 116, 97–103. Available from: https://doi.org/10.1016/j.ecoleng.2018.03.001.

Goswami, L., Mukhopadhyay, R., Bhattacharya, S.S., Das, P., Goswami, R., 2018. Detoxification of chromium-rich tannery industry sludge by *Eudrillus eugeniae*: insight on compost quality fortification and microbial enrichment. Bioresour. Technol. . Available from: https://doi.org/10.1016/j.biortech.2018.07.001.

Goswami, L., Sarkar, S., Mukherjee, S., Das, S., Barman, S., Raul, P., et al., 2014. Vermicomposting of Tea Factory Coal Ash: metal accumulation and metallothionein response in *Eisenia fetida* (Savigny) and *Lampito mauritii* (Kinberg). Bioresour. Technol. 166, 96−102. Available from: https://doi.org/10.1016/j.biortech.2014.05.032.

Grasserová, A., Hanč, A., Innemanová, P., Cajthaml, T., 2020. Composting and vermicomposting used to break down and remove pollutants from organic waste: a mini review. Eur. J. Environ. Sci. 10 (1), 9−14. Available from: https://doi.org/10.14712/23361964.2020.2.

Gusain, R., Suthar, S., 2020b. Vermicomposting of invasive weed *Ageratum conyzoids*: assessment of nutrient mineralization, enzymatic activities, and microbial properties. Bioresour. Technol. 312, 123537.

Gusain, R., Suthar, S., 2020a. Vermicomposting of duckweed (*Spirodela polyrhiza*) by employing *Eisenia fetida*: changes in nutrient contents, microbial enzyme activities and earthworm biodynamics. Bioresour. Technol. 311, 123585.

Gómez-Brandón, M., Aira, M., Lores, M., Domínguez, J., 2011. Changes in microbial community structure and function during vermicomposting of pig slurry. Bioresour. Technol. 102, 4171−4178.

Hafeez, B., Khanif, Y., Saleem, M., 2013. Role of zinc in plant nutrition- a review. Am. J. Exp. Agric. 3, 374.

Hanc, A., Dreslova, M., 2016. Effect of composting and vermicomposting on properties of particle size Fractions. Bioresour. Technol. Available from: https://doi.org/10.1016/j.biortech.2016.02.058.

Hanc, A., Pliva, P., 2013. Vermicomposting technology as a tool for nutrient recovery from kitchen bio-waste. J. Mater. Cycles Waste Manage. 15, 431−439.

Han, J., Sun, L., Dong, X., Cai, Z., Yang, H., Wang, Y., et al., 2005. Characterization of a novel plant growth-promoting bacteria strain Delftiatsuruhatensis HR4 both as a diazotroph and a potential biocontrol agent against various pathogens. Syst. Appl. Microbiol. 28, 66−76.

He, X., Zhang, Y., Shen, M., Zeng, G., Zhou, M., Li, M., 2016. Effect of vermicomposting on concentration and speciation of heavy metals in sewage sludge with ad ditive materials. Bioresour. Technol. 218, 867−873. Available from: https://doi.org/10.1016/j.biortech.2016.07.045.

Hrebeckova, T., Wiesnerova, L., Hanc, N., 2019. Changes of enzymatic activity during a large-scale vermicomposting process with continuous feeding. J. Clean. Product. 239, 118127.

Hussain, N., Singh, A., Saha, S., Kumar, M.V.S., Bhattacharyya, P., Bhattacharya, S.S., 2016. Excellent N-fixing and P-solubilizing traits in earthworm gut-isolated bacteria: a vermicompost based assessment with vegetable market waste and rice straw feed mixtures. Bioresour. Technol. 222, 165−174.

John Paul, J.A., Karmegam, N., Daniel, T., 2011. Municipal solid waste (MSW) vermicomposting with an epigeic earthworm, *Perionyx ceylanensis* Mich. Bioresour. Technol. 102 (12), 6769−6773.

Karmegam, N., Jayakumar, M., Govarthanan, M., et al., 2021. Precomposting and green manure amendment for effective vermitransformation of hazardous coir industrial waste into enriched vermicompost. Bioresour. Technol. 319, 124136. Available from: https://doi.org/10.1016/j.biortech.2020.124136.

Karmegam, N., Vijayan, P., Prakash, M., John Paul, J.A., 2019. Vermicomposting of paper industry sludge with cowdung and green manure plants using Eisenia fetida: a viable option for cleaner and enriched vermicompost production. J. Clean. Prod. 228, 718−728. Available from: https://doi.org/10.1016/j.jclepro.2019.04.313.

Karwal, M., Kaushik, A., 2020. Bioconversion of lawn waste amended with kitchen waste and buffalo dung in to value-added vermicompost using *Eisenia fetida* to alleviate landfill burden. J. Mat. Cycl. Waste Manag. . Available from: https://doi.org/10.1007/s10163-020-01101-7.

Katagi, T., Ose, K., 2015. Toxicity, bioaccumulation and metabolism of pesticides in the earthworm. J. Pestic. Sci. 40 (3), 69−81. Available from: https://doi.org/10.1584/jpestics.D15-003.

Ke, G.R., Lai, C.M., Liu, Y.Y., Yang, S.S., 2010. Inoculation of food waste with the thermo-tolerant lipolytic actinomycete Thermoactinomyces vulgaris A31 and maturity evaluation of the compost. Bioresour. Technol. 101, 7424−7431. Available from: https://doi.org/10.1016/j.biortech.2010.04.051.

Khwairakpam, M., Bhargava, R., 2009. Bioconversion of filter mud using vermicomposting employing two exotic and one local earthworm species. Bioresour. Technol. 100, 5846−5852.

Kostecka, J., Garczyńska, M., Podolak, A., Pączka, G., Kaniuczak, J., 2018. Kitchen organic waste as material for vermiculture and source of nutrients for plants. J. Ecol. Eng. 19 (6), 267−274. Available from: https://doi.org/10.12911/22998993/99691.

Kumar, A., Agrawal, A., 2020. Recent trends in solid waste management status, challenges, and potential for the future Indian cities − a review. Cur. Res. Environ. Sus. 2, 100011. Available from: https://doi.org/10.1016/j.crsust.2020.100011.

Lemtiria, A., Liénarda, A., Alabib, T., Brostauxc, Y., Cluzeaud, D., Francisb, F., et al., 2016. Earthworms Eisenia fetida affect the uptake of heavy metals by plants *Vicia faba* and *Zea mays* in metal-contaminated soils. Appl. Soil. Ecol. 104, 67−78. Available from: https://doi.org/10.1016/j.apsoil.2015.11.021.

Liang, C., Das, K.C., McClendon, R.W., 2003. The influence of temperature and moisture contents regimes on the aerobic microbial activity of a biosolids composting blend. Bioresour. Technol. 86 (2), 131−137.

Lim, S.L., Wu, T.Y., Lim, P.N., Shak, K.P.Y., 2015. The use of vermicompost in organic farming: overview, effects on soil and economics. J. Sci. Food. Agric. 95 (6), 1143−1156.

Liu, F., Zhu, P., Xue, J., 2012. Comparative study on physical and chemical characteristics of sludge vermicomposted by *Eisenia fetida*. Procedia Environ. Sci. 16, 418−423.

Li, W., Bhat, S.A., Li, J., Cui, G., Wei, Y., Yamada, T., et al., 2020. Effect of excess activated sludge on vermicomposting of fruit and vegetable waste by using novel vermireactor. Bioresour. Technol. 302, 122816. Available from: https://doi.org/10.1016/j.biortech.2020.122816.

Macci, C., Masciandaro, G., Ceccanti, B., 2010. Vermicomposting of olive oil mill wastewaters. Waste Manag. Res. 28, 738−747.

Misra, R.V., Roy, R.N., Hiraoka, H., 2003. On Farm Composting Method. FAO, Rome.

Mupambwa, H.A., Mnkeni, P.N.S., 2018. Optimizing the vermicomposting of organic wastes amended with inorganic materials for production of nutrient-rich organic fertilizers: a review. Environ. Sci. Pollut. Res. 25, 10577−10595. Available from: https://doi.org/10.1007/s11356-018-1328-4.

Naik, P.R., Raman, G., Badri Narayanan, B.K., Sakthivel, N., 2008. Assessment of genetic and functional diversity of phosphate solubilizing fluorescent *pseudomonads* isolated from rhizospheric soil. BMC Microbiol. 8, 230. Available from: https://doi.org/10.1186/1471-2180-8-230.

Nayak, A.K., Varma, V.S., Kalamdhad, A.S., 2013. Effects of various C/N ratios during vermicomposting of sewage sludge using *Eisenia fetida*. J. Environ. Sci. Technol. 6, 63−78.

Neuhauser, E.F., Hartenstein, R., Kaplan, D.L., 1980. Growth of the earthworm Eisenia fetida in relation to population density and food rationing. Oikos. 35, 93−98.

Pandit, L., Sethi, D., Pattanayak, S.K., Nayak, Y., 2020. Bioconversion of lignocellulosic organic wastes into nutrient rich vermicompost by *Eudrilus eugeniae*. Bioresour. Technol. Rep. 12, 100580.

Pathma, J., Ayyadurai, N., Sakthivel, N., 2010. Assessment of genetic and functional relationship of antagonistic fluorescent Pseudomonads of rice Rhizosphere by repetitive sequence, protein coding sequence and functional gene analyses. J. Microbiol. 48, 715−727.

Pathma, J., Kamaraj, K.R., Sakthivel, N., 2011a. Mechanisms of fluorescent Pseudomonads that mediate biological control of phytopathogens and plant growth promotion of crop plants. In: Maheswari, D.K. (Ed.), Bacteria in Agrobiology: Plant Growth Responses. Springer Verlag, Berlin, pp. 77−105.

Pathma, J., Rahul, G.R., Kamaraj, K.R., Subashri, R., Sakthivel, N., 2011b. Secondary metabolite production by bacterial antagonists. J. Biol. Control. 25, 165−181.

Pathma, J., Sakthivel, N., 2012. Microbial diversity of vermicompost bacteria that exhibit useful agricultural traits and waste management potential. Springer Plus 1, 26. Available from: https://doi.org/10.1186/2193-1801-1-26.

Pathma, J., Sakthivel, N., 2013. Molecular and functional characterization of bacteria isolated from straw and goat manure based vermicompost. Appl. Soil. Ecol. 70, 30−47.

Paul, S., Goswami, L., Pegu, R., Chatterjee, S.K., Bhattacharya, S.S., 2022. Epigenetic regulations enhance adaptability and valorization efficiency in *Eisenia fetida* and *Eudrilus eugeniae* during vermicomposting of textile sludge: Insights on repair mechanisms of metal-induced genetic damage and oxidative stress. Bioresour. Technol. 345, 126493. Available from: https://doi.org/10.1016/j.biortech.2021.126493.

Pottipati, S., Kundu, A., Kalamdhad, A.S., 2021. Process optimization by combining in-vessel composting and vermicomposting of vegetable waste. Bioresour. Technol. 16, 126357. Available from: https://doi.org/10.1016/j.biortech.2021.126357.

Pourrut, B., Shahid, M., Dumat, C., Winterton, P., Pinelli, E., 2011. Lead uptake, toxicity, and detoxification in plants. Rev. Environ. Contam. T. 213, 113−136. Available from: https://doi.org/10.1007/978-1-4419-9860-6_4.

Rai, R., Singh, R.K., Suthar, S., 2021. Production of compost with biopesticide property from toxic weed Lantana: quantification of alkaloids in compost and bacterial pathogen suppression. J. Hazard. Mater. 401, 123332. Available from: https://doi.org/10.1016/j.jhazmat.2020.123332.

Rai, R., Suthar, S., 2020. Composting of toxic weed Parthenium hysterophorus: nutrient changes, the fate of faecal coliforms, and biopesticide property assessment. Bioresour. Technol. 311, 123523.

Raut, M., William, S.P.M.P., Bhattacharyya, J.K., Chakrabarti, T., Devotta, S., 2008. Microbial dynamics and enzyme activities during rapid composting of municipal solid waste—a compost maturity analysis perspective. Bioresour. Technol. 99, 6512−6519. Available from: https://doi.org/10.1016/j.biortech.2007.11.030.

Rini, J., Deepthi, M.P., Saminathan, K., Narendhirakannan, R.T., Karmegam, N., Kathireswari, P., 2020. Nutrient recovery and vermicompost production from livestock solid wastes with epigeic earthworms. Bioresour. Technolo. 313, 123690. Available from: https://doi.org/10.1016/j.biortech.2020.123690.

Rubio, F.T.V., Haminiuk, C.W.I., Martelli-Tosi, M., da Silvaa, M.P., Makimoric, G.Y.F., Favaro-Trindadea, C.S., 2020. Utilization of grape pomaces and brewery waste

Saccharomyces cerevisiae for the production of bio-based microencapsulated pigments. Food Res. Int. 136, 109470. Available from: https://doi.org/10.1016/j.foodres.2020.109470.

Saba, S., Zara, G., Bianco, A., Garau, M., Bononi, M., Deroma, M., et al., 2019. Comparative analysis of vermicompost quality produced from brewers' spent grain and cow manure by the red earthworm *Eisenia fetida*. Bioresour. Technolo. Available from: https://doi.org/10.1016/j.biortech.2019.122019.

Sadia, M.A., Hossain, M.A., Islam, M.R., Akter, T., Shaha, D.C., 2020. Growth and reproduction performances of earthworm (*Perionyx excavatus*) fed with different organic waste materials. J. Adv. Vet. Anim. Res. 7 (2), 331−337. Available from: https://doi.org/10.5455/javar.2020.g426.

Sharma, K., Garg, V.K., 2017. Management of food and vegetable processing waste spiked with buffalo waste using earthworms (*Eisenia fetida*). Environ. Sci. Pollut. Res. 24, 7829−7836.

Sharma, D., Yadav, K.D., Kumar, S., 2018. Role of sawdust and cow dung on compost maturity during rotary drum composting of flower waste. Bioresour. Technol. Available from: https://doi.org/10.1016/j.biortech.2018.05.091.

Sherman, R., 2003. Raising earthworms successfully. North Carolina Cooperative Extension Service, Raleigh.

Singh, S.I., Angmo,D., Dutta,R., 2020. Vermitechnology: a sustainable approach in the management of solid and liquid waste (Chapter 6), Earthworm assisted remediation of effluents and wastes, Edited by Bhat SA, Vig AP, Li F, Ravindran B. Springer Nature Singapore, Available from: https://doi.org/10.1007/978-981-15-4522-1.

Sinha, R.K., Agarwal, S., Chauhan, K., Valani, D., 2010. The wonders of earthworms and its vermicompost in farm production: Charles Darwin's 'friends of farmers', with potential to replace destructive chemical fertilizers from agriculture. Agric. Sci. 1, 76−94.

Sohal, B., Singh, S., Singh, S.I.K., et al., 2021. Comparing the nutrient changes, heavy metals, and genotoxicity assessment before and after vermicomposting of thermal fly ash using *Eisenia fetida*. Environ. Sci. Pollut. Res. 28, 48154−48170. Available from: https://doi.org/10.1007/s11356-021-13726-8.

Soobhany, N., Gunasee, S., Rago, Y.P., Joyram, H., Raghoo, P., Mohee, R., et al., 2017. Spectroscopic, thermogravimetric and structural characterization analyses for comparing Municipal Solid Waste composts and vermicomposts stability and maturity. Bioresour. Technol. 236, 11−19. Available from: https://doi.org/10.1016/j.biortech.2017.03.161.

Sponza, D.T., Ağdağ, O.N., 2004. Impact of leachate recirculation and recirculation volume on stabilization of municipal solid wastes in simulated anaerobic bioreactors. Process. Biochem. 39, 2157−2165.

Spurgeon, D.J., Hopkin, S.P., Jones, D.T., 1994. Effects of cadmium, copper, lead and zinc on growth, reproduction and survival of the earthworm *Eisenia fetida* (Savigny): assessing the environmental impact of point-source metal contamination in terrestrial ecosystems. Environ. Pollut. 84, 123−130.

Spurgeon, D.J., Hopkin, S.P., 1996. Effects of metal-contaminated soils on the growth, sexual development, and early cocoon production of the earthworm Eisenia fetida, with particular reference to zinc. Ecotoxicol. Environ. Saf. 35, 86−95.

Spurgeon, D.J., Svendsen, C., Rimmer, V.R., Hopkin, S.P., Weeks, J.M., 2000. Relative sensitivity of life-cycle and biomarker responses in four earthworm species exposed to zinc. Environ. Toxicol. Chem. 19, 1800−1808.

Srivastava, V., Goel, G., Thakur, V.K., et al., 2020. Analysis and advanced characterization of municipal solid waste vermicompost maturity for a green environment. J. Environ. Manag. 255, 109914.

Sudkolai, S.T., Nourbakhsh, F., 2017. Urease activity as an index for assessing the maturity of cow manure and wheat residue vermicomposts. Waste Manag. 64, 63−66. Available from: https://doi.org/10.1016/j.wasman.2017.03.011.

Suthar, S., 2009a. Vermicomposting of vegetable-market solid waste using *Eisenia foetida*: impact of bulking material on earthworm growth and decomposition rate. Ecol. Eng. 35, 914−920.

Suthar, S., 2010. Pilot scale vermireactors for sewage sludge stabilization and metal remediation process: comparison with small-scale vermireactors. Ecol. Eng. 36, 703−712.

Suthar, S., Singh, S., 2008. Vermicomposting of domestic waste by using two epigeic earthworms (*Perionyx excavatus* and *Perionyx sansibaricus*). Int. J. Evniron. Sci. Technol. 5, 99−106.

Suthar, S., 2009b. Bioremediation of agricultural wastes through vermicomposting. Bioremed. J. 13 (1), 21−28. Available from: https://doi.org/10.1080/10889860802690513.

Swarnam, T.P., Velmurugan, A., Pandey, S.K., Roy, S.D., 2016. Enhancing nutrient recovery and compost maturity of coconut husk by vermicomposting technology. Bioresour. Technol. 207, 76−84.

Swati, A., Hait, S., 2017. Fate and bioavailability of heavy metals during vermicomposting of various organic wastes-a review. Process. Saf. Environ. 109, 30−45. Available from: https://doi.org/10.1016/j.psep.2017.03.031.

Sáez, J.A., Pérez-Murcia, M.D., Vico, A., Martínez-Gallardo, M.R., Andreu-Rodríguez, F.J., López, M.J., et al., 2021. Olive mill wastewater-evaporation ponds long term stored: integrated assessment of in situ bioremediation strategies based on composting and vermicomposting. J. Hazard. Mater. 402, 123481. Available from: https://doi.org/10.1016/j.jhazmat.2020.123481.

Thomas, G.V., Mathew, A.E., Baby, G., Mukundan, M.K., 2018. Bioconversion of residue biomass from a tropical homestead agro-ecosystem to value added vermicompost by *Eudrilus* species of earthworm. Waste Biomass Valorzi . Available from: https://doi.org/10.1007/s12649-018-0203-3.

Tiunov, A.V., Scheu, S., 2004. Carbon availability controls the growth of detritivores (Lumbricidae) and their effect on nitrogen mineralization. Oecologia 138, 83−90.

Tripathy, S., Bhattacharyya, P., Mohapatra, R., Som, A., Chowdhury, D., 2014. Influence of different fractions of heavy metals on microbial eco-physiological indicators and enzyme activities in century old municipal solid waste amended soil. Ecol. Eng. 70, 25−34.

Verma, S.L., Marschner, P., 2013. Compost effects on microbial biomass and soil P pools as affected by particle size and soil properties. J. Soil. Sci. Plant. Nutr. 13, 313−328.

Vicentin, R., Masín, C.E., Lescano, M.R., Zalazar, C.S., 2021. Poultry litter stabilization by two-stage composting-vermicomposting process: environmental, energetic and economic performance. Chemosphere 281, 130872. Available from: https://doi.org/10.1016/j.chemosphere.2021.130872.

Voberková, S., Vaverková, M.D., Burešová, A., Adamcová, D., Vršanská, M., Kynicky, J., et al., 2017. Effect of inoculation with white-rot fungi and fungal consortium on the composting efficiency of municipal solid waste. Waste Manag. 61, 157−164.

Vodounnou, D.S.J.V., Kpogue, D.N.S., Tossavi, C.E., Mennsah, G.A., Fiogbe, E.D., 2016. Effect of animal waste and vegetable compost on production and growth of earthworm (*Eisenia fetida*) during vermiculture. Int. J. Recycl. Org. Waste. Agricult. . Available from: https://doi.org/10.1007/s40093-016-0119-5.

Wang, Y., Ai, P., 2016. Integrating particle physical geometry into composting degradation kinetics. Bioresour. Technol. 200, 514−520.

Wang, K., Qiao, Y., Zhang, H., Yue, S., Li, H., Ji, X., et al., 2018. Bioaccumulation of heavy metals in earthworms from field contaminated soil in a subtropical area of China. Ecotoxicol. Environ. Saf. 148, 876–883. Available from: https://doi.org/10.1016/J.ECOENV.2017.11.058.

Wang, L., Zhang, Y., Lian, J., Chao, J., Gao, Y., Yang, F., et al., 2013. Impact of fly ash and phophatic rock on metal stabilization and bioavailability during sewage sludge vermicomposting. Bioresour. Technol. 136, 281–287. Available from: https://doi.org/10.1016/j.biortech.2013.03.039.

Wong, J.W.C., Mak, K.F., Chan, N.W., Lam, A., Fang, M., Zhou, L.X., et al., 2001. Co-composting of soybean residues and leaves in Hong Kong. Bioresour. Technol. 76 (2), 99–106.

Wyszkowska, J., Wyszkowski, M., 2006. Role of compost, bentonite and lime in recovering the biochemical equilibrium of diesel oil contaminated soil. Plant. Soil. Env. 52, 505–514.

Yadav, A., Garg, V.K., 2011. Vermicomposting – an effective tool for the management of invasive weed Parthenium hysterophorus. Bioresour. Technol. 102 (10), 5891–5895. Available from: https://doi.org/10.1016/j.biortech.2011.02.062.

Yadav, A., Garg, V.K., 2019. Biotransformation of bakery industry sludge into valuable product using vermicomposting. Bioresour. Technol. 274, 512–517. Available from: https://doi.org/10.1016/j.biortech.2018.12.023. Volume.

Yuvaraj, A., Karmegam, N., Tripathi, S., Kannan, S., Thangaraj, R., 2020. Environment-friendly management of textile mill wastewater sludge using epigeic earthworms: Bioaccumulation of heavy metals and metallothionein production. J. Environ. Mgmt. 254, 109813. Available from: https://doi.org/10.1016/j.jenvman.2019.109813.

Yu, K., Li, S., Sun, X., Kang, Y., 2020. Maintaining the ratio of hydrosoluble carbon and hydrosoluble nitrogen within the optimal range to accelerate green waste composting. Waste Manag. 105, 405–413. Available from: https://doi.org/10.1016/j.wasman.2020.02.023.

Zhang, L., Sun, X., 2018. Influence of sugar beet pulp and paper waste as bulking agents on physical, chemical, and microbial properties during green waste composting. Bioresour. Technol. 267, 182–191.

Zziwa, A., Jjagwe, J., Kizito, S., Kabenge, I., Komakech, A.J., Kayondo, H., 2021. Nutrient recovery from pineapple waste through controlled batch and continuous vermicomposting systems. J. Environ. Manag. 279, 111784. Available from: https://doi.org/10.1016/j.jenvman.2020.111784.

Further reading

Ravindran, B., Dinesh, S.L., John, L., et al., 2008. Vermicomposting of solid waste generated from leather industries using epigeic earthworm *Eisenia fetida*. Appl. Biochem. Biotechnol. 151, 480–488. Available from: https://doi.org/10.1007/s12010-008-8222-3.

Suthar, S., 2007. Vermicomposting potential of *Purionx sansibaricus* in different waste materials. Bioresour. Technol. 98 (6), 1231–1237.

This page intentionally left blank

Chapter 3

A sustainable approach for an integrated municipal solid waste management

Randhi Uma Devi[1], **Gedda Gangaraju**[2], **Kolli Suresh Kumar**[3] and **Kolli Balakrishna**[4]

[1]Department of Chemistry, School of Science, GITAM (Deemed to be University), Hyderabad, Telangana, India, [2]Department of Chemistry, Presidency University, Bangalore, Karnataka, India, [3]Department of Environmental Science, School of Science, GITAM (Deemed to be University), Visakhapatnam, Andhra Pradesh, India, [4]Department of Chemistry, GIS, GITAM (Deemed to be University), Visakhapatnam, Andhra Pradesh, India

3.1 Introduction

Waste is defined by the European Directive 2008/98/EC as 'any material or object that the holder discards, wants to discard, or is obligated to trash'. The physical condition (solid, liquid, gaseous), source (commercial, domestic, agricultural, urban) and chemical composition (hazardous/nonhazardous) of waste can all be categorised into different divisions. Rapid urbanisation and industrialisation have resulted in improved socioeconomic conditions and consumption patterns among city dwellers around the world in recent decades (Chalkley, 1997; Ngoc and Schnitzer, 2009; Rana et al., 2017). The worldwide urban population is rising at a rate of 1.5% faster than the global population (Das et al., 2019). As a result, global municipal solid waste (MSW) production is expected to exceed 2200 million tonnes per year by 2025 (Tyagi et al., 2018). The urban community encompasses the highly heterogeneous mixture of squander from urban areas along with the homogeneous mixture of domestic, industrial and commercial wastes. Improper handling of society's by-products in the form of commercial, domestic and industrial wastes can pose unswerving intimidation to the health of both public and environmental resources. Hence, sustainable solid waste management (SSWM) has to implement to improve urban environmental quality, stimulate economic development and raise awareness of health and hygiene hazards caused by inappropriate garbage management (Abdel-Shafy and Mansour, 2018; Varjani et al., 2020). Many developed and developing

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00012-8

countries have faced massive population and economic growth in recent decades. Such tremendous growth is also accompanied by a massive increase in solid waste (Ahangar et al., 2021; Browning et al., 2021; Fei et al., 2016; Patwa et al., 2021). Subsequently, solid waste management (SWM) is generating major problems, causing a downgrading of air, land and water quality with negative consequences for natural ecosystems and social health (Siddiqi et al., 2020). The practical administration of waste with the general objective of limiting its effect on the earth in a financially and socially satisfactory way is a big challenge for the coming decades. Extensive waste requires a legitimate accumulation, transfer and transportation framework. It is essential to know what the squanders comprise and how they should be gathered and arranged. The inadequate MSW management system poses a problem for the city and is a critical concern for public health, safety and the environment, as well as having an impact on economic development and quality of life; hence, it is critical to ensure the safe and proper disposal of created solid waste in order to maintain the city's tranquillity and efficiency. It is argued that long-term initiatives to reduce solid waste can result in large reductions in the amount of garbage generated (Li et al., 2020; Yu et al., 2021).

Waste management has been a significant concern in recent years, necessitating the development of a sustainable and cost-effective system (Varjani et al., 2020; Pujara et al., 2019; Harris-Lovett et al., 2019; Yaashikaa et al., 2020). The waste administration framework should be designed with specific goals and objectives and a careful evaluation of the quantity produced, collection and transportation of the waste. Solid waste disposal, on the other hand, is a serious issue since, if burned, it can result in an increase in air pollution. If thrown in the open, it can contaminate the land and water in the nearby areas. Because they are preoccupied with growing industrial and economic growth, many emerging countries have failed to pay the necessary attention to controlling their created solid waste (Ike et al., 2018). The six stages of SWM can be split into six categories: generation, collection, storage, processing/recovery, transportation and finally, disposal (Ugwu et al., 2020).

SSWM is doubtless a new reliable waste treatment solution that improves operational quality while also meeting the aims of reduction, reuse and recycling. In the integrated solid waste management hierarchy, processing and recovery refer to the many treatment and recovery procedures that aid in the use, reduction and recycling (energy recovery) operations. In recognising waste as a valuable resource, it is stated that materials produced through reuse and recycling provide an effective answer to waste management issues (Tsai et al., 2020). To achieve sustainable development goals, Bui et al. (2020) suggested that trash should be kept as a resource to increase resource efficiency, reduce carbon emissions and advocate cleaner and greener production activities. This chapter intends to analyse and identify the gaps and obstacles in implementing existing SWM to develop a sustainable waste management system.

3.2 Functional elements and hierarchy of municipal solid waste management

Based on the currently available technology, superior methodologies to manage waste focus on the widely accepted 'Hierarchy of waste management', is the precedence of alternate waste management methods (Fig. 3.1). The grading provides an effective generic strategy on the comparative appeal of different administration choices. Right from generation to the final disposal of MSW, the administration and management of MSWs and their activities are categorised into six elements, such as (1) waste handling, (2) waste generation, storage, segregation and processing at the source, (3) waste sorting, (4) waste collection, processing and recycling, (5) waste transport and transfer, (6) waste disposal.

The highest in the municipal solid waste management (MSWM) hierarchy is waste minimisation or decrease at source, that is, bringing down the quantity of the waste produced. The decline of the waste at the source stands first in the chain of command since it is the bulk operations process to decrease the amount of waste, the price connected to it, and its environmental impacts. Next is recycling, which includes sorting and separating waste items for further reuse or processing. Recycling is a significant factor that helps in decreasing the load on handling and disposal. The third step is waste processing, which is done to recover conversion products as well as energy. A range of biological and thermal techniques can be used to convert the organic part of MSW. Aerobic composting is the most widely used

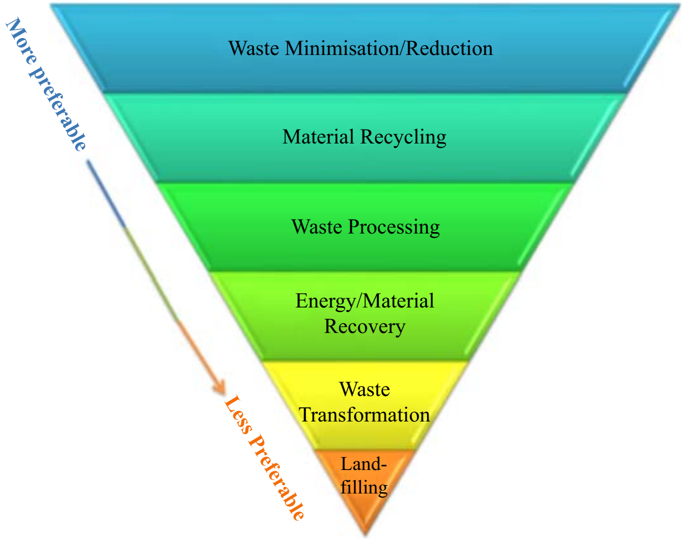


FIGURE 3.1 Hierarchy of solid waste management.

biological transformation method. Fourth in the order, change of waste, without recuperation of items or vitality, may be attempted to decrease squanders volume or to lesser toxicity. Eventually, some actions need to be taken for the waste that cannot be recycled. This is basically a residual matter post-recovery of energy. The conversion of nonrecyclable waste materials into useable heat, power, or fuel by a variety of processes, such as combustion, gasification, pyrolysis, anaerobic digestion and landfill gas recovery, is known as energy recovery from waste. Waste to energy is a term used to describe this process. Finally, the process of landfilling is at the bottom of MSWM order since it is the minimum attractive method for managing society's wastes. It is a major procedure for the eventual disposal of residual wastes.

3.3 Global scenario of municipal waste

Urban regions sustain 56% of the world's population (Allam and Sharifi, 2022), accounting for 80% of global production, 75% of resource consumption and the majority of solid waste production (Kalmykova et al., 2016; Arbolino et al., 2018). MSW created in urban development is linked to per capita disposable income in cities (Han et al., 2020). Global MSW production was 2.02 billion Mt in 2016, and it is expected to increase by 70% (3.4 billion Mt) by 2050 (Varjani et al., 2022). The amount of waste generated per capita varies significantly by region, such as the United States of America producing 2.21 kg/individual more than Europe and Central Asia (1.18), followed by Latin America and the Caribbean (0.99), the Middle East and North Africa (0.81), East Asia and Pacific (0.56), South Asia (0.52) and Sub-Saharan Africa (0.46) (Kaur et al., 2021). Managing such a massive amount of MSW in an environmentally acceptable manner is a severe concern in metropolitan areas, especially in emerging countries' fast-rising cities (Manaf et al., 2009; Rana et al., 2014; Rao et al., 2017).

The rapid pace of economic growth and advancement has resulted in urbanisation, posing a serious threat to MSW management (Yao et al., 2019). The garbage produced in wealthy and underdeveloped countries differs significantly, ranging from 0.11 to 4.54 kg per person. Asian and East Pacific countries, such as China, India and Indonesia, have experienced large increases in MSW among developing countries. China (300 MMT) is the world's greatest MSW producer, followed by the United States (292.4 MMT), India (226.57 MMT), Brazil (62.73), Indonesia (59.10) and Germany (50.52) (Allam and Sharifi, 2022). China and India are the two most populous developing countries in the world, and they also produce the most MSW. Ding et al. estimate that 50% of MSW produced in China ends up in landfills, 45% is burnt and 3% is composted (Ding et al., 2021). In India, 75%–80% of MSW is collected, 22%–28% is processed, and the rest is disposed at dump yards (Chattopadhyay et al., 2009). Brazil also produces a huge quantity of MSW every year, making it one of the fastest growing economies (Rodrigues et al., 2022), the ninth-largest economy by GDP and the fifth-largest and most populated country. According to ABRELPE (Alfaia et al., 2017), 57.6%–58.7% of MSW was disposed of in sanitary landfills, 24.3%–24.1% was disposed of in regulated landfills and 18.1%–17.2% was disposed of in open dump sites.

Around 70% of all MSW produced worldwide ends up in landfills, 19% is recycled and 11% is used to generate electricity. In 2018 the United States created 292.4 million tonnes of MSW, of which 50% ended up in landfills, 23.6% was recycled, 8.5% was composted, 11.8% was used for energy generation and 6.1% was used for other food management (Tiseo, 2021). In the European Union, 225 million tonnes of municipal garbage were produced, with 24% of MSW ending up in landfills, 48% being recycled/composted and 27% being used for energy generation. When compared to prior years, there has been a minor increase in the amount of MSW used for recycling and incineration, as well as a decrease in the proportion of MSW ending up in landfills. The management of MSW in industrialised and developing countries differs significantly. Developed countries have more resources, educated personnel and a stable economy to help them grow, plan and implement long-term waste management strategies. The move from wasteless to wasteful cultures has occurred in underdeveloped countries, while the industrialised world is shifting to ‘wasting less’ societies.

3.4 Solid waste and its impacts

The effects of urbanisation and development have had their share of disadvantages, mainly in the form of environmental pollution (Gangaraju et al., 2022). As the population of the world is increasing, the demand for food and other essentials also increases, which is directly proportional to growth in the waste generated daily from an individual household. This waste is eventually tossed into city waste accumulation areas. Incorrect MSW disposal pollutes streets, water bodies and other sites, exacerbating the current urban pollution problem (Chand Malav et al., 2020). MSW is produced as a result of the unsustainable use of natural resources, which results in resource depletion and environmental degradation (Mohammadi et al., 2019). Nonbiodegradable trash, such as plastic, metals, rubber and e-waste, are included, as well as biodegradable wastes, such as paper, food, vegetables and textile waste (Bhat et al., 2014; Mishra et al., 2020). Economic growth, urbanisation (Mamun et al., 2016) and high living standards all influence the complexity and quantity of municipal solid garbage generated (Bhat et al., 2018).

Littering and greenhouse gas (GHG) emissions from landfills create health issues, as well as environmental damage. Rural populations have been flocking to metropolitan regions for work, better facilities and convenience in developing economies around the world. It has resulted in a huge increase

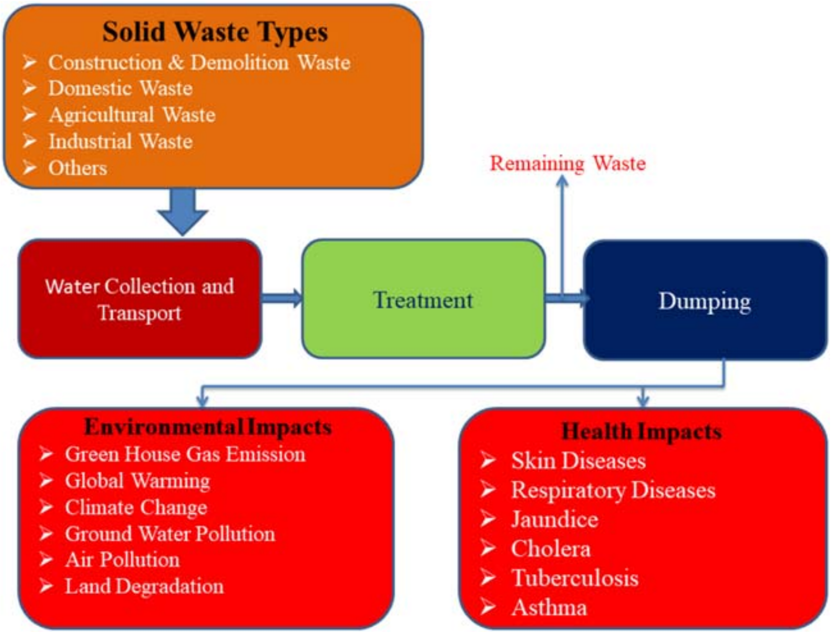


FIGURE 3.2 Environmental and health impacts of solid waste.

in municipal solid trash generation. Pollution of the soil, water and air occurs as a result of insufficient MSW segregation and removal. Furthermore, improper MSW dumping pollutes surface and groundwater, whereas unprofessional MSW cleanup has a negative impact on the ecosystem (Fig. 3.2). The majority of urban solid wastes are dumped into bodies of water and soil without or with inadequate treatment, resulting in significant environmental contamination (Varjani et al., 2021).

3.5 Sustainable solid waste management

SSWM relates to municipal advances in which resources are sufficient to meet daily consumer demand while ensuring ecosystem sustainability through trash collection, handling, reuse, recycling and resource conservation (Chang and Pires, 2015). The SSWM concept is a multidimensional management approach that considers social, economic and environmental factors (Florio et al., 2019). According to Tsai et al. (2021), SSWM is critical for all phases of the management process, from design to planning, operation and discharge (Tsai et al., 2021). According to Aid et al. (2017), SSWM not only facilitates resource conversion but also creates more occupational and economic prospects by introducing a new method for resource usage (Aid et al., 2017).

Implementing SSWM is one of the most important steps in municipal development. Different SSWM techniques can be used to address environmental preservation, societal resolutions and economic structures. When it comes to implementing SSWM in practise, however, weak reactions and environmental implications remain obstacles. According to Um et al. (2018), establishing an SSWM system is difficult due to complex and time-consuming government planning approval processes. Inadequate facilities and infrastructure, poor planning strategies, legislative shortcomings, a lack of vocational talents, knowledge, and informative communication systems and insufficient funding and sponsorship are all impediments to SSWM (Ikhlayel, 2018). According to Aid et al. (2017), the ecological impacts of discharged solid waste are putting pressure on local governments to establish appropriate instruments and regulations to address the problem. These data show that SWM is still a long way from meeting its sustainability goals. To control created waste, bring economic advantages and alleviate the collective problematic status, defining crucial indicators for an SSWM method is critical. The key parameters to achieve SSWM are proper characterisation, segregation, transportation and disposal of MSW (Aid et al., 2017).

3.6 Characterisation of urban solid waste

Urban solid waste comprises various categories depending on the characteristics of solid waste, such as industrial rejects, road cleaning dust, commercial waste, construction, demolition waste, institutional, food and sanitary waste (Table 3.1). MSW mainly consists of recyclables (including glass, plastics, paper, metals, etc.), toxic materials (paint items, used items, expired medicines and pesticides) and biological items (sanitary napkins, body fluid stained cotton, disposables syringes, etc.).

The characteristics and constitution of urban solid wastes differ across the globe, even within the country, city to city. There are various factors for this variation, such as commercial activities, climate, logistics, population, distribution of wealth, types of industries in the region, class of people living, education level, customs, floating population rate, etc.

Ecological squanders allude for the most part to substances comprising of natural materials, for example, food remains, vegetables and organic product peels, paper materials, wood and so forth, created from different family units and mechanical exercises on account of the activity of smaller scale life forms; these squander are debased from complex to fewer complex mixes (Table 3.2). Nonbiodegradable waste takes time to break down into simple biological forms from the complex structure, and they are made of inorganic materials summed up with recyclable items, that is, plastic, metal cans, etc.

TABLE 3.1 Origin of municipal waste generation.

Domestic waste	Household waste−garbage, decayed fruits, house cleaning dust, newspapers, packaging materials, remnants of vegetables, plastic waste, paper, food waste, etc.
Trading and business activities	Squander generation from business establishments, paper, plastic, stationary related, packaging materials, construction debris, broken items, shops, offices, vegetable markets, grocery stores, etc.
Institutions	Educational institutes, wastes from temples, churches, masjids, etc.
Roads cleaning	Dry leaves from trees, dust & soil on the road, paper & plastic waste, fruits, vegetables, carcass, etc.
Waste generated from industries	Packaging materials, metal pieces, wood, plastic materials, organic and inorganic waste, hazardous chemicals, chemical containers, oil rags, waste cloth, etc.
Waste from construction related activities	Soil and dust from the road excavation, construction of public and private buildings, debris, cement, sand and other waste
Carcass	Stray animals, carcasses, remnants of meat and bones from slaughterhouses.

TABLE 3.2 Degeneration duration for waste categories.

Category of waste	Particulars	Approximate degeneration time
Perishable	Garbage and food waste, vegetable and fruit peels	1−2 week
	Waste paper	10−30 days
	Clothes made of cotton	03−06 days
	Things made of wool	365 days
	Items made of wood	10−15 years
Imperishable	Metal items	100−500 years
	Plastic covers	1,000,000 years
	Glass items	Undetermined

3.7 Optimisation approaches on solid waste collection and transportation

Storage, collection, transportation, treatment, recycling and final disposal are all processes in the solid waste disposal process. Furthermore, solid waste collection and transportation is an important part of waste management. The transfer of solid waste from generation and collection to treatment or landfill disposal is referred to as waste collection and transportation. It includes the door-to-door or kerbside collection of municipal, private, informal or other waste collectors. MSWM relies heavily on the collection of segregated municipal garbage. Inefficient rubbish collection services have a negative influence on public health and aesthetics in cities. Recycling is maximised when wet, dry and home hazardous garbage are collected separately. It also improves the cost-effectiveness of treating such wastes in order to meet the minimum quality criteria set for various goods, such as composting organic waste or producing refuse-derived fuel (RDF).

Solid waste collecting systems come in a variety of shapes and sizes. Waste can be collected from sources of generation, communal collection locations, or transfer stations to the final disposal site using a variety of transport services, such as big motorised vehicles or garbage trucks. The availability of collection services, the manner of operation, and the type of waste materials collected are all used to classify solid waste collection systems. The following collection system divisions may assist the solid waste manager in more efficiently optimising the design and process of collection services, particularly if a diverse variety of collection vehicles is necessary to collect waste from various collection points.

- Primary collection is the first step of a solid waste collection system that begins at or near the site of generation. It is the process of collecting solid waste at the point of generation and transporting it to communal collection bins or points, as well as a processing or transfer station. Although this service is not common in poorer sections of the developing globe, a growing number of small enterprises and/or community-based groups are springing up in affluent areas to fill the need.
- Secondary collection refers to the collecting of solid trash from communal bins, storage locations, or transfer stations and transportation to a final disposal site.

3.8 Types of solid waste collection system

3.8.1 Communal systems

Communal solid waste collection systems are widespread in low-income nations where cost cutting takes precedence over service supply, as this approach minimises the number of collection stations. People are responsible for taking their trash to one or more communal collection sites or containers under this arrangement. The main downside of this approach is that the containers/collection stations are in public spaces, which leads to indiscriminate waste disposal outside the container in many cases. The success of this system is contingent on public cooperation. As a result, it is critical to focus more on improving the design, operation and maintenance of community systems in order to boost public approval and maximise efficiency. For a small selection of waste items, communal arrangements are typical in many industrialised countries. Bulky products and garden garbage, for example, are required to be brought to central collection locations, materials banks and drop-off centres by waste generators. The usage of portable containers increases worker and vehicle productivity. In the solid waste collection system, this is the least efficient.

3.8.2 Block collections

It is the responsibility of the individual to carry their trash to the collection vehicle. Generally, the vehicle comes to a halt at street junctions or designated collecting spots. This technique requires little to no effort and few vehicles, and it helps to keep rubbish off the streets. A consistent and well-organised collection service is required so that residents know when to bring their trash out.

3.8.3 Kerbside collections

In industrialised countries and certain developing countries' affluent towns, this is the most prevalent and successful solid waste collection system. On a set day, households and businesses leave waste in containers, commonly referred to as wheelie bins, on the kerbside for trucks to collect and transport the rubbish to local recycling centres. Typically, there are three bins: one for recycling paper, cardboard, or glass, one for green waste and one for general garbage.

3.8.4 Door-to-door collections

In industrialised countries, this type of solid waste collection system is more widespread. Smaller enterprises and community-based groups in affluent areas of emerging countries are still adopting it. On a certain day, people deposit waste containers at the rear gate or in the immediate neighbourhood of their home for collection in the door-to-door collection technique (Ahangar et al., 2021; Pujara et al., 2019). After emptying waste into collection vehicles, the collection staff enter each home, remove the containers or bags and if necessary, replace them. When opposed to the kerbside/alley collection approach, which requires accessing all properties, this system has a higher cost. Each member of the collection crew starts the day with a clean standard trashcan, which is taken to be left at the first property in the dustbin exchange system. The full container is carried out and emptied into the collection vehicle, after which the empty bin is transferred to the next location.

3.8.5 Transportation of solid waste

Trash transportation from temporary waste storage depots/sites might be organised according to the frequency with which containers fill up. The sites where the containers are put are divided into four categories:

- Containers that must be cleared multiple times per day
- Containers that must be cleaned once a day.
- Containers that must be cleared every other day.
- Containers that take longer to fill and require clearance twice a week. Vegetable, fruit, meat and fish market waste should be removed at least once a day.

3.8.6 Vehicle scheduling

The route for lifting containers may be devised based on the number of containers to be cleared each day, avoiding zigzagging as much as feasible. This will save a significant amount of gasoline and time. As the WET waste is collected everyday/most days of the week, a larger fleet of dedicated vehicles and equipment would be required. Dry waste can also be collected regularly if two-compartment systems are available in vehicles/waste collection tricycles. If the DRY waste collection system is once or twice a week may require a smaller fleet per volume, distance and time modelling. The DRY waste has the problem of excessive volume; hence larger capacities by volume of collection system vehicles may be required

3.8.7 Vehicles are used in two shifts

All vehicles might be used in two shifts to lift containers eliminating the need for additional vehicles, and ensuring maximum usage of the fleet of vehicles. Waste transportation may be done at night in regions where there is heavy traffic during the day, causing SWM activities to be hampered. Working at night will boost productivity while also lowering the cost of providing such a service.

3.8.8 Type of vehicles to be used

To avoid multiple trash handling, vehicles that can synchronise well with containers placed at temporary waste storage depots should be used for transportation. The quantity of garbage to be transported, the distances to be driven, the road widths, road conditions, workshop facilities and other factors should all be considered while selecting vehicles.

3.9 Call-based collection and transportation

A call-based collection system for waste is a premium service on a chargeable basis for those citizens and customers who want to give away their dry waste at their convenience. This premium service may also offer buy-in of WET and DRY wastes, classified waste, such as paper, plastic, glass metal, etc., by the collection vehicle itself. The call-based collection system is particularly suited for characteristic wastes such as flowers, animal dung, hotel/restaurant/chai waste/waste from functions/gatherings, etc., from nonhousehold areas/ activities.

3.10 Bulky waste collection and transportation

Bulky waste items, such as mattresses, furniture, plastic items, etc., should not be allowed to become part of ordinary or general MSW.

3.11 User charges

The sustainability of scientific SWM, including collection and transportation mechanisms, depends entirely on the ready availability of funds to meet fixed and variable costs. User charges are essential because scientific SWM involves many local operations compared to the old legacy model of street sweeping only.

3.12 Sustainable solid waste reuse and recycling approaches

Recycling is the process of converting waste resources into new materials, substances, or things. Reusing trash, on the other hand, comprises taking any items or product pieces and repurposing or inventively reusing them for the same or a new purpose (Villalba, 2020). Material reuse and recyclability are dependent on reverting to the original state. Material reuse and resource recycling have existed since the dawn of time to provide several benefits. Reuse and recycling are helpful since they reduce mineral and energy consumption, GHG emissions, pollution and solid waste disposal problems. According to Martin et al. (2017), these methods replace the usage of raw materials and conserve natural resources, resulting in a more sustainable environment. As a result, potentially usable trash is reused, and new material consumption is reduced, resulting in energy savings and pollution reduction from incineration and landfilling. Solid waste creation can be viewed as a potential source of renewable energy, a source of new jobs and economic benefits, as well as a way to raise community awareness about environmental issues (Ferronato and Torretta, 2019; Kheybari et al., 2019).

The most significant obstruction to recycling and recovery is that the majority of efforts are focused downstream of the waste management process (Siddiqi et al., 2020). Yu et al. (2021) proposed or complete understanding of the 4R development process (reduce, reuse, recycle and recover), as well as a reduction in total waste while repurposing or recycling any remaining rubbishes. Kihl and Aid (2016) suggested that the legislation regulation for sorting recyclable waste prevents the costly, time-consuming and convoluted government permission procedures (Batista et al., 2021; Naldi et al., 2021).

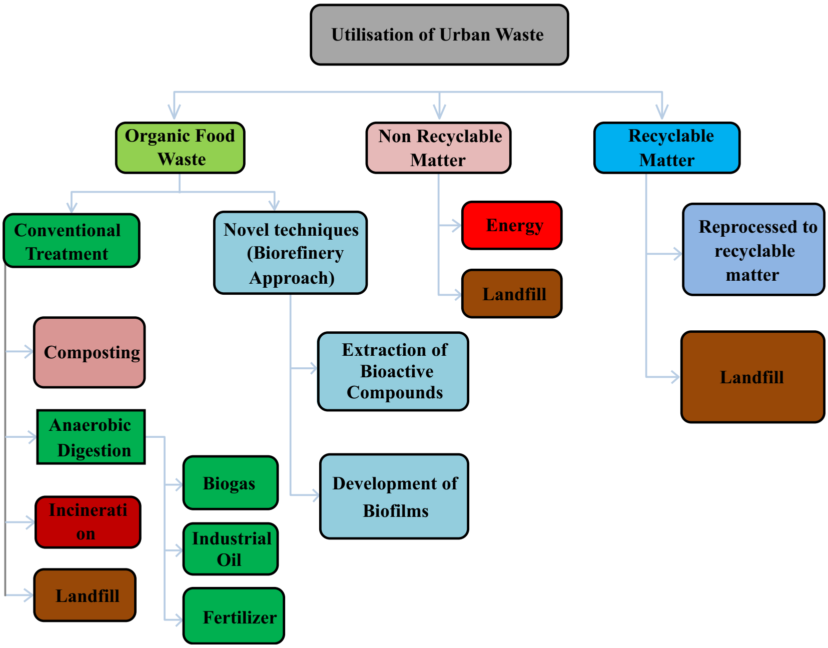


FIGURE 3.3 Various municipal solid waste usage approaches.

Solid waste and its reused items have similar standards, which could lead to a social distrust of recycled goods (Fig. 3.3). In addition, improper waste sorting makes recycling more difficult and imported technology is useless (Bui et al., 2021). There are still some gaps in defining how to refill resources while avoiding negative repercussions for long-term solid waste reuse and recycling, necessitating further research and implementation.

Composting, incineration, anaerobic digestion and landfills have all used MSW in the past. Recent research investigations, on the other hand, have been more focused on the biorefinery strategy, which entails generating a wide range of products with low energy requirements and waste output (Fig. 3.4). MSW has been used for a variety of applications, including fillers in polymer composites (Cabrera, 2021), energy generation (Abdallah et al., 2020) and the synthesis of biopolymers (Moretto et al., 2020). The use of MSW in the development of biopolymers, biofilms and other related products could also provide economic opportunities.

3.13 Usage of MSW for the development of bioproducts

The economic analysis for the MSWM found that urban waste output increased over the last decade. The traditional options, such as integrated waste conversion

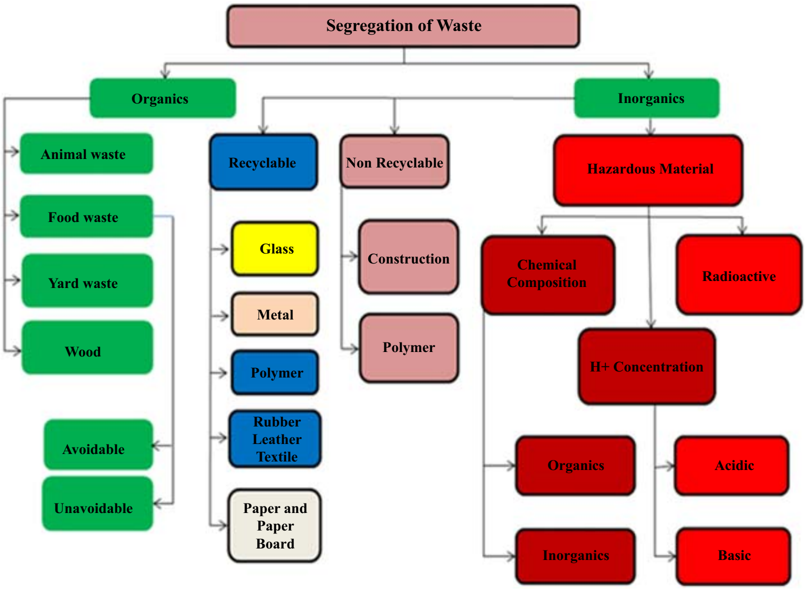


FIGURE 3.4 Schematic representation of municipal solid waste segregation.

and landfilling, became prohibitively outrageous due to limited land resources. As a result, using MSW to make bio-based materials, including biomaterials, bioplastics and biofertilizers offers a long-term, cost-effective and environmentally beneficial option (Ebrahimian et al., 2020). Farah et al. (2011) transformed the domestic organic part of MSW to PHB (poly-hydroxybutyrate) using Cupriavidus nectar CCGUG 52238, yielding 38.14 g/L with 52.79% PHB. PHB is appraised to cost roughly US$ 2.05/kg, making it an excellent economic value for MSW. Another study used MSW to make poly 3-hydroxybutyrate by enzymatic hydrolysis of the solid waste, which resulted in a lower yield of PHB (Izaguirre et al., 2021). In another study, basal medium components such a fruit by-products and glucose were supplemented with MSW, which resulted in a high fermentation yield via solvent extraction (Sindhu et al., 2019).

The economic studies reported that these industrial-scale conversion technologies for MSW-based biomaterials manufacturing are not economically viable. So, it is essential to create and develop new technologies to segregate waste for maximum usage in the production of biobased goods. Biobased goods, on the other hand, such as PHA, biofertilizers, PHB, biomaterials and biosurfactants, are biodegradable, and their fermentation method leaves no leftovers that are harmful to the environment (Fig. 3.5). As a result, the organic component of MSW, which has a high concentration of nutrients, can be used to produce bioproducts that contribute to the sustainable bioeconomy (Yadav et al., 2020).

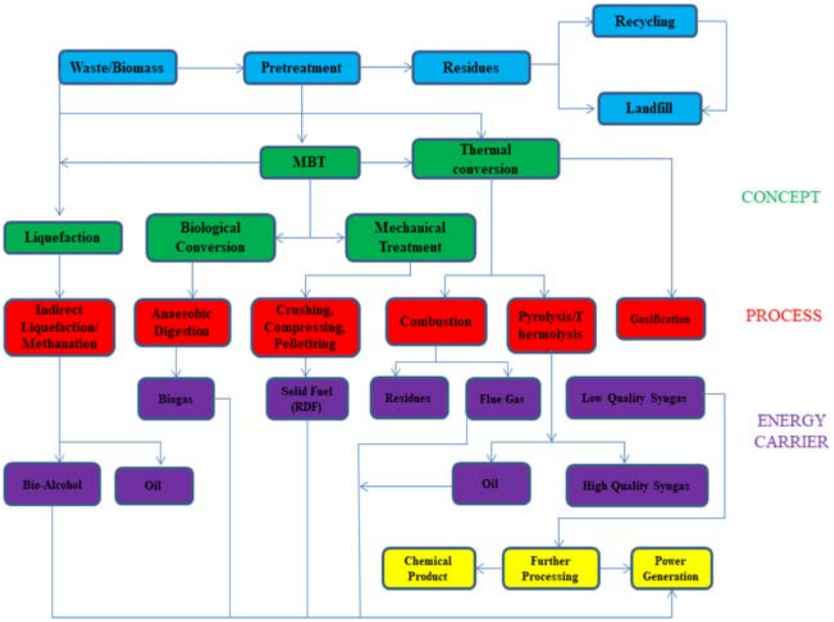


FIGURE 3.5 Different waste-to-energy by-products.

3.14 Treatment and disposal methods of solid waste

Segregation of waste at the origin of generation is significant for the technical and scientific disposal process, as the recyclable materials can be directly sent to recycling units (Moh and Abd Manaf, 2014). The municipalities can also make money out of it. The lethargic material can be sent to landfills after proper compaction and reduction, which might also prolong the life of the landfill sites. The local bodies have adopted the mechanism of open dumping, landfill, sanitary landfills and composting and incineration for the final disposal of solid waste (Jebaranjithama et al., 2022).

In contrast to an open dump that imposes severe environmental pollution, a landfill (Siddiqi et al., 2022) is a pit excavated in the earth where MSW is kept and covered with sand. A layer of earth is deposited on top of the pit at the end of each day, and the pit is closed to prevent the breeding of flies and rodents, which has now formed a cell. As a result, rubbish is dumped every day and transformed into a cell. When the landfill is full, the area is covered in a thick layer of mud, which allows the property to be developed as a parking lot or park. Landfills have numerous issues since all forms of trash are thrown in them, and the leachate becomes contaminated, polluting the surrounding soil and water supplies. Hence, sanitary landfill is an alternative to landfills that will alleviate the problem of leaching. Sanitary landfills are constructed over impermeable soil and are lined with impermeable materials, such as plastics and clay. Incineration is an important part of the hazardous waste and clinical waste treatment process. Medical waste must frequently be incinerated at high temperatures to remove pathogens and harmful contaminants. Incineration is a cost-effective approach to minimise waste volume and landfill demand. Incineration plants can be built close to the waste generation centre, lowering waste transportation costs. Using MSW incinerator ash for environmentally friendly buildings not only provides a low-cost aggregate but it also minimises the requirement for landfill space.

3.15 Conclusion

Urban regions currently support 56% of the global population and generate a large volume of municipal solid trash. Of this waste, 70% ends up in landfills, while 19% is recycled and 11% is used to generate electricity. The decline of the waste at the source is essential to decrease the amount of waste, the price connected to it and its environmental impacts. Minimising waste and boosting recovery are critical for economic and ecological sustainability. It is essential to develop SSWM methods to improve the urban environmental quality and stimulate economic development. This process helps to meet daily consumer demand while ensuring ecosystem sustainability through trash collection, handling, reuse, recycling and resource conservation.

References

Abdallah, M., Arab, M., Shabib, A., El-Sherbiny, R., El-Sheltawy, S., 2020. Characterization and sustainable management strategies of municipal solid waste in Egypt. Clean Technol. Environ. Pol. 22 (6), 1371−1383.

Abdel-Shafy, H.I., Mansour, M.S.M., 2018. Solid waste issue: sources, composition, disposal, recycling, and valorization. Egypt. J. Pet. 27 (4), 1275−1290.

Ahangar, S.S., Sadati, A., Rabbani, M., 2021. Sustainable design of a municipal solid waste management system in an integrated closed-loop supply chain network using a fuzzy approach: a case study. J. Ind. Prod. Eng. 38 (5), 323−340.

Aid, G., Eklund, M., Anderberg, S., Baas, L., 2017. Expanding roles for the Swedish waste management sector in inter-organizational resource management. Resour. Conserv. Recycl. 124, 85−97.

Alfaia, R.G., de, S.M., Costa, A.M., Campos, J.C., 2017. Municipal solid waste in Brazil: a review. Waste Manag. Res. 35 (12), 1195−1209.

Allam, Z., Sharifi, A., 2022. Research structure and trends of smart urban mobility. Smart Cities 5 (2), 539−561.

Arbolino, R., Carlucci, F., De Simone, L., Ioppolo, G., Yigitcanlar, T., 2018. The policy diffusion of environmental performance in the European countries. Ecol. Indicat. 89, 130−138.

Batista, M., Caiado, R.G.G., Quelhas, O.L.G., Lima, G.B.A., Leal Filho, W., Yparraguirre, I.T.R., 2021. A framework for sustainable and integrated municipal solid waste management: barriers and critical factors to developing countries. J. Clean. Prod. 127516.

Bhat, R.A., Nazir, R., Ashraf, S., Ali, M., Bandh, S.A., Kamili, A.N., 2014. Municipal solid waste generation rates and its management at Yusmarg forest ecosystem, a tourist resort in Kashmir. Waste Manag. Res. 32 (2), 165−169.

Bhat, R., Ahmad Dar, S., Ahmad Dar, D., Hamid Dar, G., 2018. Municipal solid waste generation and current scenario of its management in India. Int. J. Adv. Sci. Res. 7 (2), 419−431.

Browning, S., Beymer-Farris, B., Seay, J.R., 2021. Addressing the challenges associated with plastic waste disposal and management in developing countries. Curr. Opin. Chem. Eng. 32, 100682.

Bui, T.D., Tsai, F.M., Tseng, M.L., Ali, M.H., 2020. Identifying sustainable solid waste management barriers in practice using the fuzzy Delphi method. Resour. Conservat. Recycl. 154, 104625.

Bui, T.D., Tsai, F.M., Tseng, M.L., Tan, R.R., Yu, K.D.S., Lim, M.K., 2021. Sustainable supply chain management towards disruption and organizational ambidexterity: a data driven analysis. Sustain. Prod. Consum. 26, 371−410.

Cabrera, F.C., 2021. Eco-friendly polymer composites: a review of suitable methods for waste management. Polym. Compos. 1−25.

Chalkley, K., 1997. Population growth and consumption. Popul. Today 25 (4), 4−5.

Chand Malav, L., Yadav, K.K., Gupta, N., Kumar, S., Sharma, G.K., Krishnan, S., et al., 2020. A review on municipal solid waste as a renewable source for waste-to-energy project in India: current practices, challenges, and future opportunities. J. Clean. Prod. 277, 123227.

Chang, N.B., Pires, A., 2015. Sustainable Solid Waste Management: A Systems Engineering Approach. IEEE Press, New Jersey.

Chattopadhyay, S., Dutta, A., Ray, S., 2009. Municipal solid waste management in Kolkata, India—a review. Waste Manag. 29 (4), 1449−1458.

Das, S., Lee, S.H., Kumar, P., Kim, K.H., Lee, S.S., Bhattacharya, S.S., 2019. Solid waste management: scope and the challenge of sustainability. J. Clean. Prod. 228, 658−678.

Ding, Y., Zhao, J., Liu, J., Zhou, J., Cheng, L., Zhao, J., et al., 2021. A review of China's municipal solid waste (MSW) and comparison with international regions: management and technologies in treatment and resource utilization. J. Clean. Prod. 293, 126144.

Ebrahimian, F., Karimi, K., Kumar, R., 2020. Sustainable biofuels and bioplastic production from the organic fraction of municipal solid waste. Waste Manag. 116, 40−48.

Farah, N.O., NorrsquoAini, A.R., Halimatun, S.H., Tabassum, M., Phang, L.Y., Mohd, A.H., 2011. Utilization of kitchen waste for the production of green thermoplastic polyhydroxybutyrate (PHB) by *Cupriavidus necator* CCGUG 52238. Afr. J. Microbiol. Res. 5 (19), 2873−2879.

Fei, F., Qu, L., Wen, Z., Xue, Y., Zhang, H., 2016. How to integrate the informal recycling system into municipal solid waste management in developing countries: based on a China's case in Suzhou urban area. Resour. Conserv. Recycl. 110, 74−86.

Ferronato, N., Torretta, V., 2019. Waste mismanagement in developing countries: a review of global issues. Int. J. Environ. Res. Public Health 16 (6), 1060.

Florio, C., Fiorentino, G., Corcelli, F., Ulgiati, S., Dumontet, S., Güsewell, J., et al., 2019. A life cycle assessment of biomethane production from waste feedstock through different upgrading technologies. Energies 12 (4), 718.

Gangaraju, G., Ganesh, G., Umadevi, R., Balakrishna, K., 2022. Plant Material Assisted Magnetic nanomaterials (MNPs) for the Separation of Inorganic Pollutants. Springer, pp. 183−199.

Han, X., Hu, C., Lin, L., 2020. A study on the impact of China's urbanization on the quantity of municipal solid waste produced. Waste Manag. Res. 38 (2), 184−192.

Harris-Lovett, S., Lienert, J., Sedlak, D., 2019. A mixed-methods approach to strategic planning for multi-benefit regional water infrastructure. J. Environ. Manage. 233, 218–237.

Ike, C.C., Ezeibe, C.C., Anijiofor, S.C., Nik Daud, N.N., 2018. Solid waste management in nigeria: problems, prospects, and policies. J. Solid Waste Technol. Manag. 44, 163–172.

Ikhlayel, M., 2018. Indicators for establishing and assessing waste management systems in developing countries: a holistic approach to sustainability and business opportunities. Bus. Strategy Dev. 1 (1), 31–42.

Izaguirre, J.K., Baranano, L., Castanon, S., Santos, J.A.L., Cesario, M.T., daFonseca, M.M.R., et al., 2021. Economic and environmental assessment of bacterial poly(3- hydroxybutyrate) production from the organic fraction of municipal solid waste. Biores. Bioprocess. 8 (39), 1–12.

Jebaranjithama, J.N., Christyraj, J.D.S., Prasannan, A., Rajagopalan, K., Chelladurai, K.S., Gnanarajad, J.K.J.S., 2022. Current scenario of solid waste management techniques and challenges in Covid-19—a review. Heliyon 8 (7), e09855.

Kalmykova, Y., Rosado, L., Patrício, J., 2016. Resource consumption drivers and pathways to reduction: economy, policy and lifestyle impact on material flows at the national and urban scale. J. Clean. Prod. 132, 70–80.

Kaur, P., Kaur, G.J., Routray, W., Rahimi, J., Nair, G.P., Singh, A., 2021. Recent advances in utilization of municipal solid waste for production of bioproducts: a bibliometric analysis. Case Stud. Chem. Environ. Eng. 4, 100164.

Kheybari, S., Kazemi, M., Rezaei, J., 2019. Bioethanol facility location selection using best-worst method. Appl. Energy 242, 612–623.

Kihl, A., Aid, G., 2016. Driving forces and inhibitors of secondary stock extraction. Open Waste Manage. J. 9 (1), 11–18.

Li, D., Wang, M.Q., Lee, C., 2020. The waste treatment and recycling efficiency of industrial waste processing based on two-stage data envelopment analysis with undesirable inputs. J. Clean. Prod. 242, 118279.

Mamun, M.A.A., Hannan, M.A., Hussain, A., Basri, H., 2016. Theoretical model and implementation of a real time intelligent bin status monitoring system using rule based decision algorithms. Expert Syst. Appl. 48, 76–88.

Manaf, L.A., Samah, M.A.A., Zukki, N.I.M., 2009. Municipal solid waste management in Malaysia: practices and challenges. Waste Manag. 29 (11), 2902–2906.

Martin, G., Savaget, P., Bocken, N.M., Jan Hultink, E., 2017. The circular economy—a new sustainability paradigm. J. Clean. Prod. 423, 757–768.

Mishra, B., Varjani, S., Agarwal, D.C., Mandal, S.K., Ngo, H.H., Taherzadeh, M.J., et al., 2020. Engineering biocatalytic material for the remediation of pollutants: a comprehensive review. Environ. Technol. Innov. 20, 101063.

Moh, Y.C., Abd Manaf, L., 2014. Overview of household solid waste recycling policy status and challenges in Malaysia. Resour. Conserv. Recycl. 82, 50–56.

Mohammadi, M., Jounela, S.L.J., Harjunkoski, I., 2019. Optimal planning of municipal solid waste management systems in an integrated supply chain network. Comput. Chem. Eng. 123, 155–169.

Moretto, G., et al., 2020. Biopolymers from urban organic waste: influence of the solid retention time to cycle length ratio in the enrichment of a mixed microbial culture (MMC). ACS Sustain. Chem. Eng. 8 (38).

Naldi, A., Herdiansyah, H., Putri, L.S., 2021. Good governance role for a sustainable solid waste management in rural community, IOP Conference Series: Earth and Environmental Science, 819. IOP Publishing, p. 012033.

Ngoc, U.N., Schnitzer, H., 2009. Sustainable solutions for solid waste management in Southeast Asian countries. Waste Manag. 29 (6), 1982–1995.

Patwa, N., Sivarajah, U., Seetharaman, A., Sarkar, S., Maiti, K., Hingorani, K., 2021. Towards a circular economy: an emerging economies context. J. Bus. Res. 122, 725–735.

Pujara, Y., Pathak, P., Sharma, A., Govani, J., 2019. Review on Indian municipal solid waste management practices for reduction of environmental impacts to achieve sustainable development goals. J. Environ. Manage. 248, 109238.

Rana, R., Ganguly, R., Gupta, A., 2014. Solid waste management in Chandigarh—a case study. J. Civil Eng. Environ. Tech. 1 (4), 67–70.

Rana, R., Ganguly, R., Gupta, A., 2017. Evaluation of solid waste management in satellite towns of Mohali and Panchkula. J. Solid Waste Technol. Manag. 43, 280–294.

Rao, M.N., Sultana, R., Kota, S.H., 2017. Municipal solid waste (Chapter 2). Management Science and Engineering. pp. 3–120.

Rodrigues, L.F., Santos, I.F.S., Santos, T.I.S., Barros, R.M., Filho, G.L.T., 2022. Energy and economic evaluation of MSW incineration and gasification in Brazil. Renew. Energy 188, 933–944.

Siddiqi, A., Haraguchi, M., Narayanamurti, V., 2020. Urban waste to energy recovery assessment simulations for developing countries. World Dev. 131, 104949.

Siddiqi, S.A., Al-Mamun, A., Baawain, M.S., Sana, A., 2022. A critical review of the recently developed laboratory-scale municipal solid waste landfill leachate treatment technologies. Sust. Ener. Tech. Ass. 52, 102011.

Sindhu, R., Gnansounou, E., Rebello, S., Binod, P., Varjani, S., Thakur, I.S., et al., 2019. Conversion of food and kitchen waste to value-added products. J. Environ. Manag. 241, 619–630.

Tiseo, I., 2021. Generation and discards of municipal solid waste in the U.S. 1960–2018, Statista.

Tsai, F.M., Bui, T.D., Tseng, M.L., Wu, K.J., 2020. A causal municipal solid waste management model for sustainable cities in Vietnam under uncertainty: a comparison. Resour. Conserv. Recycl. 154, 104599.

Tsai, F.M., Bui, T.D., Tseng, M.L., Lim, M.K., Tan, R.R., 2021. Sustainable solid-waste management in coastal and marine tourism cities in Vietnam: a hierarchical-level approach. Resour. Conserv. Recycl. 168, 105266.

Tyagi, V.K., Fdez-Güelfo, L.A., Zhou, Y., Álvarez-Gallego, C.J., Garcia, L.I.R., Ng, W.J., 2018. Anaerobic co-digestion of organic fraction of municipal solid waste (OFMSW): progress and challenges. Renewable Sustainable Energy Rev. 93 (2017), 380–399.

Ugwu, C.O., Ozoegwu, C.G., Ozor, P.A., 2020. Solid waste quanti fi cation and characterization in university of Nigeria, Nsukka campus and recommendations for sustainable management. Heliyon 6, E04255.

Um, N., Kang, Y.Y., Kim, K.H., Shin, S.K., Lee, Y., 2018. Strategic environmental assessment for effective waste management in Korea: a review of the new policy framework. Waste Manag. 82, 129–138.

Varjani, S., Upasani, V.N., Pandey, A., 2020. Bioremediation of oily sludge polluted soil employing a novel strain of *Pseudomonas aeruginosa* and phytotoxicity of petroleum hydrocarbons for seed germination. Sci. Total Environ. 737, 139766.

Varjani, S., Rakholiya, P., Shindhal, T., Shah, A.V., Ngo, H.H., 2021. Trends in dye industry effluent treatment and recovery of value added products. J. Water Process. Eng. 39, 101734.

Varjani, S., Shahbeig, H., Popat, K., Patel, Z., Vyas, S., Shah, A.V., et al., 2022. Sustainable management of municipal solid waste through waste-to-energy technologies. Biores. Technol. 355, 127247.

Villalba, L., 2020. Material flow analysis (MFA) and waste characterizations for formal and informal performance indicators in Tandil, Argentina: decision-making implications, J. Environ. Manage., 264. p. 110453.

Yaashikaa, P.R., Kumar, P.S., Saravanan, A., Varjani, S., Ramamurthy, R., 2020. Bioconversion of municipal solid waste into bio-based products: a review on valorisation and sustainable approach for circular bioeconomy. Sci. Total Environ. 748, 141312.

Yadav, B., Pandey, A., Kumar, L.R., Tyagi, R.D., 2020. Bioconversion of waste (water)/residues to bioplastics—a circular bioeconomy approach. Bioresour. Technol. 298, 122584.

Yao, X.L., Guo, Z., Liu, Y., Li, J., Feng, W., Lei, H., et al., 2019. Reduction potential of GHG emissions from municipal solid waste incineration for power generation in Beijing. J. Clean. Prod. 241, 118283.

Yu, K.H., Zhang, Y., Li, D., Montenegro-Marin, C.E., Kumar, P.M., 2021. Environmental planning based on reduce, reuse, recycle and recover using artificial intelligence. Environ. Impact Assess. Rev. 86, 106492.

Chapter 4

Covid-19 pandemic and solid waste generation management strategies, challenges and approaches

P.F. Steffi[1], **P.F. Mishel**[2], **Vijayalakshmi Selvakumar**[3], **Thenmozhi Manivel**[4] and **Karnan Muthusamy**[5]

[1]Department of Microbiology, Cauvery College for Women (Autonomous), Trichy, Tamil Nadu, India, [2]Department of Botany, Bharathidasan University, Trichy, Tamil Nadu, India, [3]Food Science and Biotechnology, School of Agriculture and Lifesciences, Kangwon National University, Chuncheon, Gangwon-do, Republic of Korea, [4]Askoscen Probionics (R&D), Trichy, Tamil Nadu, India, [5]Grassland and Forage Division, National Institute of Animal Science, Rural Development Administration, Chungnam-do, South Korea

4.1 Introduction

In urban systems, solid waste is generated on a daily basis and must be handled on a daily basis. Local governments rely on solid waste management to provide hygienic conditions in residential areas. In early December 2019, a significant surge in pneumonia cases was reported in Wuhan, Hubei Province, China. It was reported to the World Health Organization (WHO) Headquarters in China on 31 December 2019. On 11 March 2020, the WHO proclaimed COVID-19 a pandemic. This novel coronavirus has been called SARS-CoV-2 because its symptoms are similar to those of severe acute respiratory syndrome (SARS). It is a beta coronavirus that originated in animals (particularly bats), and it has been classified as an HG3 (Hazardous Group 3) illness, similar to HIV and tuberculosis (TB). COVID-19 is the twenty-first century's third zoonotic pandemic, following SARS (2002–03) and Middle East Respiratory Syndrome (MERS) (2012). This epidemic, in particular, has had a global impact on several businesses, resulting in economic collapse. The ill person's respiratory droplets are the primary mode of human-to-human transmission, although other routes include indirect contact, such as viral transmission through an inert metallic surface touched by an

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00004-9

infected person. SARS-CoV-2 has infected a vast population and killed many people throughout the world (Ab Rahman et al., 2020).

Throughout the outbreak, the type of rubbish produced changed considerably. PPE, masks and hand sanitisers are becoming more widespread, and the rubbish they generate adds a huge load to waste treatment facilities. The lockdown, as well as the majority of people working from home to limit illness spread, have an impact on waste generation. The capacity of current treatment facilities has been exceeded since the generation of hazardous medical waste has outstripped the capacity of existing treatment facilities. Because the availability of working employees is insufficient to fulfil safety requirements, the use of single-use plastic, Personal protective equipment (PPE), and other products have placed an unprecedented strain on waste treatment facilities. As a result, it is apparent that waste management practises on a global scale will depart from the usual operations of garbage collection and recycling. Because of an unanticipated increase in positive instances against present medical facilities, people who are just moderately unwell or asymptomatic are urged to remain at home. Used tissues, body fluids, PPEs and so on must be disposed of properly during home quarantine so that they do not become a vector of infection for other family members or garbage collectors. Poor solid waste management might have a significant role in disease spread (Almulhim et al., 2021).

This article investigates the worldwide scenario of solid waste generation and disposal in connections to houses, quarantine facilities, hospitals, infected patients and other entities affected by the COVID-19 pandemic. In addition, the study discusses how rubbish might be processed or reused to avoid disease spread through solid waste. It also gives insight into the strategies adopted by several nations to reduce solid waste during the COVID-19 outbreak (Baker et al., 2020).

4.2 Healthcare waste composition

Healthcare debris is frequently characterised as waste generated by healthcare facilities, research institutes and laboratories in connection with medical activities. Approximately 75%–90% of medical, solid waste is comparable to household rubbish and so categorised as ‘non-hazardous’ or ‘generic healthcare’ waste. In reality, this waste is generated by the administrative, culinary and cleaning obligations of medical and healthcare establishments. The remaining 10%–25% of rubbish is classed as ‘hazardous materials’, posing serious environmental and health concerns (Brisolara et al., 2021). With the exception of a vast number of plastics and microplastics, the composition of healthcare solid waste during the COVID-19 pandemic is nearly equal to that produced under normal conditions; however, the outbreak has resulted in a significant increase in rubbish production. Under normal circumstances, the composition of healthcare solid waste is significant because it dictates its ability to be recycled and managed sustainably, which is critical during the current pandemic.

4.3 Hazardous waste from healthcare

4.3.1 Additives waste

Chemicals are all around us in healthcare. Because these institutions employ a lot of chemicals, the chemical waste they create might be harmful to their health and the environment. This type of rubbish accounts for around 3% of total waste created by healthcare activities. Laboratory chemicals, film developing reagents, expired/unused disinfectants, solvents and rubbish containing heavy metals are examples of chemical healthcare waste (batteries, damaged thermometers, blood-pressure gauges and so on). Due to severe health risks, an expanding number of hospitals have replaced safer alternatives for some of their most harmful pharmaceuticals and created careful management methods (Cai et al., 2021). Nonetheless, many organisations in both developing and industrialised countries continue to use these hazardous substances and have insufficient chemical waste disposal techniques.

4.3.2 Infectious waste

Viral healthcare waste is defined as trash containing germs that cause illness onset and progression; this includes things contaminated with blood and body fluids, human excreta, laboratory cultures and microbiological goods (Capoor and Parida, 2021). PPE, which includes boots, long-sleeved gowns, heavy-duty gloves, masks, goggles and face shields, is also classified as infectious waste, and garbage generated from these products has increased dramatically during the COVID-19 outbreak. As a result, regulating this type of garbage during a pandemic is extremely challenging.

4.3.3 Pathological waste

Pathological waste is defined as a smaller chunk, portion, or slice of any tissue, organ or bodily component taken from surgical or microbiological specimens from animal or human bodies. When tissues or tissue samples are inspected and/or assessed in a laboratory to detect or study abnormalities or sick tissues, this type of waste is produced (D'Alessandro et al., 2020). In essence, this junk is the same as contagious waste, and it must be treated with caution during the current pandemic. Because infective viral particles are present in the tissue samples, it has the potential to spread illness in the same manner that infectious garbage does.

4.3.4 Radioactive waste

Numerous nuclear technologies used in healthcare institutions, such as nuclear medicine, radiation and research reagents, generate radioactive waste as a by-product. This waste contains radioactive compounds, such as radiotherapy or laboratory research liquids that were no longer needed. Radioactive glassware, packages/absorbent paper, urine and excreta from people who have been treated or tested with unsealed radionuclides are likewise classified as radioactive waste (Dehghani et al., 2022). If radioactive materials are not properly managed, they can cause serious health problems, as well as environmental damage. The progress of the COVID-19 plague has jeopardised radioactive waste containment, and further controls must be put in place to minimise this hazardous waste's exposure to humans and the environment.

4.3.5 Sharps waste

Sharps garbage is a type of healthcare solid waste that consists of used or unused hypodermic, intravenous, or other needles, auto-disable syringes, syringes with linked needles, infusion sets, scalpels, pipettes, knives, blades and broken glasses. Sharps healthcare waste should be handled carefully and maintained effectively during the COVID-19 pandemic since SARS-CoV-2 has been observed to remain on numerous surfaces for a limited period of time (Elsaid et al., 2021). Sharps infected with the virus might easily infect garbage workers, spreading the infection across the neighbourhood.

4.3.6 Pharmaceutical waste

Pharmaceutical waste may be generated by a range of activities and locations in healthcare facilities, including pharmacies, distribution centres and hospitals. Expired and tainted pharmaceuticals are examples of medical waste. Pharmaceutical waste also contains contaminated pharmaceuticals, such as vaccines, as well as spent biological products for therapy and transdermal patches (Fadaei, 2021). The volume of pharmaceutical waste has increased dramatically as a result of the rising number of hospital admissions during the COVID-19 outbreak. Waste collectors from pharmacies, distribution centres, and hospitals are at risk of becoming infected with SARS-CoV-2 if they come into touch with COVID-19 patients or virally contaminated pharmaceutical waste while collecting it from approved treatment units.

4.4 Non-hazardous healthcare waste

Non-hazardous healthcare solid waste includes old plastic water bottles, office paper, magazines, newspapers, food trash and food packaging that is not combined with hazardous rubbish. Non-hazardous rubbish is the same as home waste and can be recycled to help with long-term waste management (Govindan et al., 2021). During their daily activities at healthcare institutions, both asymptomatic and symptomatic COVID-19 patients are expected to generate a high amount of non-hazardous SARS-CoV-2 contaminated healthcare waste, posing a significant risk of community transmission.

4.4.1 Other waste

This other type of healthcare waste that has been derived in significant quantities during the COVID-19 outbreak is test kits and waste generated from various COVID-19 diagnostic methods, as global transmission and prevalence have necessitated the detection of infections to aid with appropriate social distancing and quarantine measures. Because each kit is only used once, using fast test kits to detect an infected person contributes to waste (Gwenzi and Rzymski, 2021). If not properly managed, there is always the risk that this waste will get contaminated with SARS-CoV-2 and contribute to the disease's spread.

4.5 Waste management strategies during the COVID-19 pandemic

Different countries have used different techniques to deal with the massive and infectious volume of garbage generated during the COVID-19 outbreak. In regard to the outbreak, health waste management plans should incorporate a number of additional steps to guarantee proper containment and infection prevention. Based on their capacity, resources and dedication, several countries have implemented the finest management practises imaginable. Various organisations have provided suggestions for handling healthcare waste in a way that is both safe and sustainable (Haghnazar et al., 2022). Healthcare solid waste created during the COVID-19 pandemic is deemed infectious waste, according to criteria set by several European Union (EU) member states, and the ability to manage this garbage should be improved. If there are any concerns with incineration or disposal capacity, sufficient facilities for interim waste storage should be available. The garbage should be kept in sealed containers in secure places where only authorised workers are permitted to enter. Disinfectants should be applied on both the outside and interior surfaces to prevent viral spread. All employees in the region must adhere to adequate safety precautions.

A hospital waste management plan is a programme that a facility has in place to manage produced trash for disposal. It typically addresses the following topics: (1) regulatory compliance; (2) staff member duties; (3) definitions/classification of healthcare waste; (4) specialised processes for handling healthcare waste; and (5) training of associated professionals.

Infected healthcare solid trash was divided and packaged by waste management staff in hospitals during the COVID-19 pandemic in Hubei, China. They clean the trash with a 0.5% chlorine solution and double bag it before putting it in temporary healthcare storage within hospitals. The manner of disposing of healthcare waste is determined by the hospital and its waste management facilities. Before disposal of the piece of garbage in a permitted landfill, autoclave or irradiation sterilisation was used. Incineration on-site or in a specific isolated place has been used to dispose of healthcare waste in some hospitals (Ikiz et al., 2021). To help with the excess healthcare waste created during the epidemic, mobile incineration or autoclave equipment has been deployed. Alternative facilities for disposing of healthcare waste have been investigated, including cement kilns and other industrial furnaces. Additional healthcare waste has been temporarily kept in secure and separate facilities. Only designated vehicles were used to carry healthcare solid waste, and data were appropriately documented. Load locations have been closed, disinfected and segregated from drivers to reduce the danger of illness. In the United States, healthcare waste created by COVID-19 patients is treated the same as trash generated by other patients, according to Centres for Disease Control and Prevention (CDC) guidelines. As a result, the trash has been treated as standard healthcare waste, and no further special treatment is required.

During the COVID-19 epidemic, the Philippines created a specific provision to handle hospital waste. On the island of Luzon, there are particular registered transporters and treatment, storage and disposal (TSD) facilities for processing and disposing of healthcare waste. For smooth management, specific permission is in place to collect pathological and infectious hospital waste. Each registered transporter must go through a special checkpoint and submit the following online documents: an official request letter; a transporter and TSD registration certificate; a transport management plan; a transportation route; a schedule; and an agreement between the healthcare waste generator, the transporter and the treatment workers. Each vehicle used to transport garbage bears the following unique markings: transport name and ID; placard; waste class; and waste number. These are visible from a distance of 15 m away from the car (Gibbons et al., 2021). The registered transporter must produce a transportation compliance and completion report, which must be attested by a representative from the healthcare and TSD departments.

In Jordanian, healthcare waste management has been carried out in accordance with three basic principles: reduction of superfluous healthcare waste, separation of routine waste from hazardous waste, and proper treatment to limit dangers to health professionals and society. Personnel handling COVID-19-contaminated healthcare waste must wear PPE, which includes ultra-filtered masks (Nano) and fluid protective long-sleeved garments, as well as a cap, shoes, elastic leather gloves, protective glasses and a full-face protector (Islam and Chowdhury, 2021). The work is rigorously monitored by the individual authorities to ensure that local norms and regulations, such as new criteria for cleaning and disinfection surrounding COVID-19, are strictly followed. On a daily basis, waste from COVID-19 patients is disposed of quickly. Containers, temporary and permanent storage areas and hospital waste bags are cleaned to avoid the transmission of the infection.

Pharmaceutical waste must be managed on a regular basis during the pandemic, according to all accepted procedures. The trash must be collected, segregated, and labelled with particular identifying labels before being handled, transported and disposed of correctly. Personal protection, disinfection, and training should be regarded as essential for appropriate healthcare waste management. Waste from confirmed COVID-19 patients, such as infectious waste, sharps and pathological waste, should be collected safely and stored in specific bags labelled appropriately. PPE, such as boots, long-sleeved gowns, heavy-duty gloves, masks, goggles and face shields, should be worn by healthcare waste collectors, and they should wash their hands with sanitiser or disinfectant after disposing of the trash (Kalantary et al., 2021).

Reviewing the preceding examples, it is evident that techniques differ among countries. Infected persons have need medical assistance in their homes as the COVID-19 pandemic has spread and hospitals have been inundated with COVID-19 patients. There are no management plans in place for healthcare waste created by homes. Although a country's economic environment affects its management of healthcare solid waste, a specific management approach is critical in this case (Pourakbar et al., 2022). Nonetheless, given the current circumstances, the establishment of appropriate management is critical. Recycling and landfill reduction should also be addressed to help in the long-term management of healthcare solid waste during and after the COVID-19 pandemic.

4.6 COVID-19 virus persistence and probable transmission via hospital solid waste

COVID-19 virus spreads by sneezing, coughing, contact with touched things, and personal contact. The life duration of the COVID-19 virus on various substrates is critical for developing suitable management methods and procedures for dealing with healthcare solid waste. SARS-CoV-2 has a life span ranging from a few hours to a few days, depending on the substrate type and climatic circumstances. The COVID-19 virus survives 3 hours, 4 hours, 24 hours, and 2−3 days after aerosolisation on copper, cardboard, plastic and stainless steel, respectively (Kasim et al., 2021). Additional researchers have documented that the virus may persist for 9 days on inanimate surfaces such as metal, glass or plastic. It can survive for two days in dechlorinated tap water and hospital wastewater at 20°C. Because of the prolonged living span of this unique COVID-19 virus, there is a greater danger of community

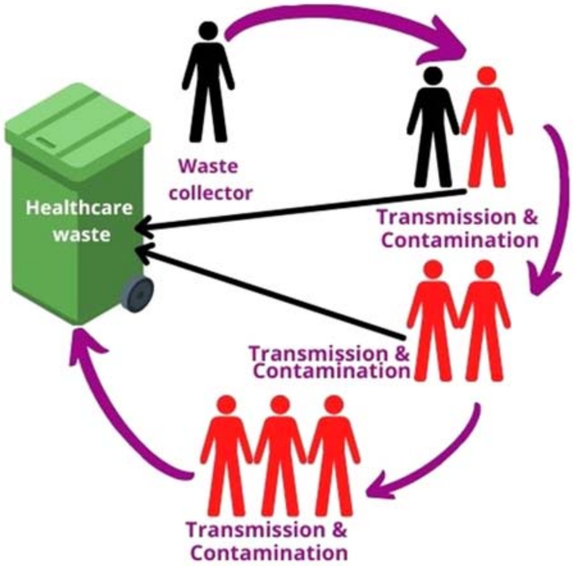


FIGURE 4.1 Depicts the transmission of the COVID-19 virus via hospital waste.

transmission. Inadequate handling of solid waste created by healthcare facilities may contribute to the spread of the COVID-19 virus. Rubbish pickers are directly exposed to hazardous waste and are, therefore, prone to illness. As a result, people may unknowingly spread the infection throughout their group. Fig. 4.1 depicts the new virus's transmission mechanism.

4.7 The impact of the COVID-19 pandemic on the quantity and composition of municipal solid waste

The fast shift in the quantity and character of urban solid waste, as shown in the COVID-19 pandemic, is an important result of disease epidemics. This change is caused by two important factors: first, in a pandemic situation, people's lifestyles will change to accommodate the new conditions; second, the community's health needs will increase production and consumption rates in certain areas, which will play a role in changing the composition and quantity of municipal solid waste (Kaya et al., 2022).

The COVID-19 pandemic's impacts on different waste components were not the same, and hence the pandemic lowered trash output in some places while disrupting plastic consumption reduction initiatives in others. Plastic garbage created since the COVID-19 epidemic is estimated to be 1.6 million tonnes per day globally. Everyday, about 3.4 billion single-use face masks and shields were discarded. Meanwhile, the COVID-19 PCRs were responsible for producing 15,000 tonnes of plastic, of which 97% were burnt.

For example, the extensive use of face masks, disposable gloves and other personal protective equipment (PPE) during the COVID-19 epidemic has resulted in a considerable increase in plastic and fibre-based municipal solid waste products (Lazuka et al., 2021).

Furthermore, lifestyle choices, particularly in the provision of daily essentials, such as the growing use of home delivery services, have increased paper and plastic waste from packing. These developments, as well as an increase in the ratio of using home food to ready meals in the community, have resulted in a rise in municipal solid waste in various societies. Increased plastic consumption is obvious as a result of its high capability to create masks and disposable gloves, as well as the packaging of health items and commodities delivered to homes and consuming locations (Rakib et al., 2021). In certain circumstances, particularly during the early COVID-19 outbreaks, an increase in internet buying has resulted in an increase in throwaway items due to decreased daily activities. Moreover, during the COVID-19 epidemic, food delivery services grew, leading to an increase in plastic packaging and food waste (household waste). Because of worries about virus transmission through ready-to-eat meals, increased in-home cooking has resulted in increased household trash creation in several nations. During the epidemic, younger people were more concerned about food waste (Loh et al., 2021).

In these settings, the ratio of food waste creation to municipal solid garbage increased by 12% in Spain. Nonetheless, as observed in the first two weeks of the COVID-19 in Tunisia, preserving food and minimising food waste might minimise waste output owing to worries about the pandemic's economic repercussions on the family component. Food waste output has declined in Italy and the United States owing to a rising desire to make more home-cooked meals and to live away from stores (Maher et al., 2021). Furthermore, the coronavirus pandemic has had a direct impact on municipal solid waste composition by increasing the volume of non-recyclable garbage and an indirect impact on organic waste creation owing to trade disruptions caused by lower agricultural and fisheries exports. Unlike home trash, industrial waste creation may, however, drop as a result of the pandemic's interruption of industrial activity and commodities commerce. Because of geographical location, lifestyle and the amount of the disease's influence on everyday activities, these circumstances will not be the same everywhere.

Especially in rare situations, in pandemic conditions, it may lower the pace of municipal solid trash output. According to a study relating to the coronavirus pandemic (in China), garbage creation in major and medium-sized cities in China has fallen by up to 30%. In contrast, medical waste has increased many times in these cities, with Wuhan reporting an increase of up to six times (Mostafa et al., 2021). Epidemiology and health policy in other countries, such as Italy, however, suggest using single-use disposable plates in restaurants and cafés, resulting in increased municipal solid waste creation. As a result, changes in the waste amount and component ratios have a dramatic impact on effective waste management systems. It is required to adapt programmes to these actual conditions in order to identify the quantity and composition of municipal solid trash based on changes in society's behaviour and to re-define municipal solid waste management (Park et al., 2021). Changes in management may need the development of new procedures for some municipal solid waste, as demonstrated by the South Korean model for the separate administration of masks and disposable gloves during the coronavirus epidemic. The investigation demonstrated the need for short-term, intermediate and long-term waste management measures in a pandemic event.

4.8 The impact of the coronavirus pandemic on the waste management process

Extremely contagious disease outbreaks and concerns about their transmission through municipal solid waste prompted a change in waste management based on previous or gained experiences in other countries, so some previous processes were discontinued or resumed with significant differences in defined requirements. The most essential reason for such a behavioural shift is to minimise the spread of pollution from contaminated (or suspected infected) trash, as well as to reduce the possibility of disease transmission. Because of worries about the virus spreading through the air, a compactor garbage collection vehicle for home waste may be kept aside for additional precaution (Penteado and Castro, 2021). Trucks, human resources and other fees will be required in this situation for garbage handling. The most essential aspect of waste management influenced by these changes is waste recycling, and hence municipal solid waste recycling is projected to decline dramatically in a pandemic situation. In the COVID-19 outbreak, there is evidence of a reduction in the waste-to-material business when compared to the prior era. Bans or restrictions on municipal solid waste recycling facilities in the United States and Europe during the coronavirus pandemic, for example, were imposed owing to worries about the virus's spread. Separation of domestic garbage has also been restricted in countries such as Italy, where a suspected or diseased individual is isolated or cared for at home, reducing the amount of recyclable material entering the reuse cycle. These circumstances have raised worries about decreasing garbage recycling and the environmental repercussions of the epidemic (Qiu et al., 2022).

In poor nations, the recycling business is based on separation at the disposal stage and garbage pickers working in landfills. Changing the situation in this part is extremely difficult and impossible. As a result, the risk of disease transmission through inadequate waste management is projected to be greater in underdeveloped nations than in industrialised countries. To cope with COVID-19, slums and refugee camps must pay particular attention to garbage collection and management initiatives (Ragazzi et al., 2020). As a result, municipal waste management systems must activate or increase the quality of services in these regions to avoid disease transmission in the community, which necessitates more money, as well as the use of equipment and human resources. On the other hand, disease outbreaks and quarantine regulations may cause citizens to relocate from their primary residence to secondary settlements, putting strain on the capacity of rural waste management, so the capacity of the waste management system in these areas must be increased in terms of equipment and personnel.

Because of the health guidelines, this circumstance has resulted in an improvement in the waste management system. Because of the likelihood of combining urban garbage with medical waste, the operating temperature of municipal waste incineration plants has been elevated to sufficiently wipe off viruses. As a result, it is critical to pay more attention to the waste management pandemic conditions recommendations in order to mitigate the damage caused by improper waste management (Ranjbari et al., 2022).

One of the most significant consequences of the epidemic is the handling of medical waste. Concerns have been expressed about the transmission of the coronavirus through medical waste, which has resulted in the establishment of different paths for storing, collecting and transferring potentially infected waste. It is proposed, for example, that medical waste and home garbage from clinics, isolated sections, special examination rooms and medical laboratories, particularly nucleic acid testing laboratories, be treated and managed as COVID-19-related medical waste. According to a Chinese hospital study, the sanitary dumping approach was replaced with garbage burning to better control the pandemic and reduce the danger of viral transmission. In addition, the storage of coronavirus-related medical waste was reduced from 48 to 24 hours, and all infectious and household garbage in hospitals should be cremated as soon as possible (Sharma et al., 2020).

4.9 Impact of the coronavirus pandemic on littered waste

Litter refers to waste materials that were incorrectly disposed away by humans in improper areas. Numerous studies on the number and nature of litter in diverse areas, such as beaches, have revealed that plastic compounds and cigarette butts are commonly littered objects. An important worry, however, is the disposal of possibly hazardous trash in public spaces and the environment. One of the most important health recommendations in the coronavirus pandemic is the usage of personal protective equipment (PPE) such as face masks and gloves (Silva et al., 2021). Nonetheless, the improper disposal of this technology by residents is regarded as a major worry in many nations. The studies conducted in the United States revealed that the widespread use of masks and gloves contributes significantly to the increase in litter; additionally, there is evidence in Spain and Asia of the transfer of used masks to water areas and an increase in the number of disposable masks and gloves discarded on beaches. Discarded masks on the South American shore have been identified as a source of worry. Long-term effects of textile fibres impregnated with Ag^+/Cu^{2+} nanoparticles on the aquatic environment According to one research, face masks are the most prevalent personal protection equipment seen on beaches during the epidemic, which has been increasing over time. The total number of wasted face masks in African nations is estimated to be 12 billion every month (around 105,000 tonnes of infected waste per month). These problems are not confined to beaches; during the COVID-19 epidemic, riverine detritus has surged in Jakarta. Mask was the most prevalent type of plastic, accounting for approximately 16% of total riverine trash. According to the findings of a research conducted in Toronto, Canada, disposable gloves and face masks accounted for around 44% and 31% of debris connected to personal protection equipment, respectively, which is recognised as a new source of plastic pollution (Tomasino et al., 2021). Another Kenyan research indicated that up to 16.5% of overall litter seen on the streets is connected with COVID-19 items, but litter associated with COVID-19 was not identified on recreational beaches.

The collapse of the travel and tourist business has, however, had a good impact on trash reduction in recreational areas such as beaches. Because of the danger of contamination, the increased usage of masks and gloves has resulted in an increase in urban trash, which requires adequate control (Vu et al., 2021). Littering pandemic-related garbage is a huge environmental hazard, increasing the possibility of disease transmission through municipal solid waste. Because most coronavirus masks and protective gloves are constructed of plastic-based materials, the presence of these materials in the environment might be regarded as a microplastic source. Because they are constructed of oil-derived polymers, discarded face masks may be a significant source of pollution for marine habitats, releasing substantial amounts of microplastics (Wang et al., 2021). The findings demonstrated that single-use face masks might serve as dye carriers in aquatic environments (methylene blue, crystal violet and malachite green). Although the pandemic outbreak may have a positive impact on urban litter reduction due to lower population density in public areas, an increase in the proportion of infectious or suspected litter may pose a higher risk factor for disease transmission and public health threats via municipal solid waste (Table 4.3).

As a result, a solution for waste management linked with COVID-19, particularly face masks, is required. Degradation and crushing of discarded masks were required for pavement applications and were viewed as a method for controlling the high volume of trash. In addition, using different materials to make face masks can help to mitigate the environmental harm caused by the COVID-19 epidemic (Warmadewanthi et al., 2021).

4.10 Primary waste disposal strategies: benefits and downsides comparing

Solid Waste Treatment and Disposal Tables 4.1 and 4.2 compare the benefits and drawbacks of the three primary garbage disposal systems. Infectious trash was typically incinerated or sterilised before disposal in hospitals, whereas non-infectious garbage was disposed of in a sanitary landfill. Of the hospitals that responded, 70% had an on-site incinerator for disposing of infectious trash. A time-temperature profile of less than or equal to 30 minutes at 121°C was employed by about half of the hospitals that used a steam steriliser for this purpose. Only 8% of hospitals used alternative sterilising technologies, such as gas and/or hot air, to render infected trash non-infectious before dumping it (Xu et al., 2021). The responding hospitals did not employ other accessible technologies (radiation, chemicals) to render infectious material non-infectious before disposal. 98% of the 96 responding hospitals disposed of their non-infectious solid waste in Class A landfills, whereas 2% disposed of their garbage in a Class B landfill. Some hospitals

TABLE 4.1 Benefits of primary waste disposal strategies.

Benefits	Incineration	Sanitary landfill	Grinders
1.	Destroys the majority of potentially pathogenic organisms	Cost-effective	Waste can be removed from the environment as soon as possible
2.	It is possible to create usable energy (e.g., steam)	Class A landfills (those that are covered with earth on a daily basis) should not pose a risk to public health	There is no need for storage or transportation
3.	Weight loss of up to 80%	–	Workforce savings as a result of reduced solid trash collection
4.	If you are on-site: Lowers transportation costs	–	Odour reduction
5.	Reduces the amount of rubbish that must be held before being disposed of at a landfill	–	–

TABLE 4.2 Downsides of primary waste disposal strategies.

Downsides	Incineration	Sanitary landfill	Grinders
1.	Expensive-both in terms of original investment and ongoing maintenance costs	Because of the perceived risk connected with hospital trash, several landfills refuse to accept hospital solid waste	It is sometimes forbidden due to increased organic load on the sewage system
2.	Must adhere to federal and local air pollution regulations.	The risk of fire and water contamination (both ground and surface) if poorly situated and operated	The majority of the time, it is used in kitchens at healthcare institutions
3.	There is a need to dispose of residue or ash.	Possibility of a storage issue prior to disposal	Acoustics, tremor, grinder clogging and drain line blockage
4.	–	Some infectious garbage may require sterilisation before disposal in a landfill	Microbial aerosols that may be produced during usage
5.	–	–	Certain combustible and non-combustible trash have limited application

(64%) ground up solid waste and drained it into the sanitary sewage system. The grinder was commonly used to dispose of rubbish (87%), but it was also used to discard waste resulting from direct patient care activities (13%) (Ye et al., 2022).

4.11 Challenges

Biomedical garbage is harmful to both human health and the environment. Doctors and nurses are at a higher risk of contracting an illness. Water is polluted near landfills that hold medical waste. The COVID-19 epidemic has put a significant strain on current waste management systems, as seen by increased garbage buildup. A COVID-19 patient can produce around 3.4 kg of healthcare waste everyday. As a result, the amount of healthcare waste has progressively grown during the epidemic. Healthcare waste in Hubei has surged 600% from 40 to 240 tonnes, overwhelming the current transport and disposal facilities. Other nations are experiencing comparable difficulties in dealing with massive amounts of trash; this tendency has been noticed in France, Italy and the Netherlands. Healthcare solid waste in France and the Netherlands has grown from 40% to 50% and from 45% to 50%, respectively (Yousefi et al., 2021). The same pattern has been noticed in India and Iran. During the COVID-19 epidemic, the number of used PPE and general trash, such as food and meal boxes and infusion bottles and bags used by nurses, has increased dramatically. Governments in Europe are confronting issues in keeping waste management employees, ensuring a safe working environment, handling domestic garbage generated by patients at home, and making room for extra waste generated by the epidemic. Water can be contaminated by landfills harbouring healthcare solid waste created during the epidemic. During the epidemic, the cost of healthcare has also risen.

4.12 Potential for healthcare solid waste management strategies

A good healthcare waste management system in a healthcare facility necessitates an assessment of the waste stream and existing environmental practises, an evaluation of waste management options, the development of waste management plans and the implementation of institutional policies and guidelines that clearly define personnel roles and responsibilities. The establishment of a waste management organisation, the allocation of human and financial resources, the implementation of plans, periodic training, monitoring, assessment and continuous improvement are all critical for managing healthcare waste in a sustainable manner (Yuwen et al., 2022). Effective waste management is totally based on a well-organised healthcare waste management organisation and methods. For establishing and implementing a waste management strategy, a waste management team or committee should be constituted. In low-income communities, an infection control committee should be formed, with one person at charge of healthcare waste management in healthcare institutions. It is critical to assess the plan on a regular basis, and all staff members involved in healthcare waste should be fully informed of the procedures and any modifications. Overall, optimising healthcare resources has the potential to minimise waste creation. Fig. 4.2 depicts an appropriate management approach for hospital solid waste during the COVID-19 epidemic.

Although healthcare solid waste categories vary by country, most countries prefer the WHO guidelines. Segregation is critical for effective healthcare waste management. It addresses the separation of various forms of trash based on categorisation at the place of origin. As a result, efficient segregation of recyclable garbage from other non-hazardous waste will greatly reduce waste. Rubbish segregation entails sorting waste into appropriate containers. To separate infectious trash, carefully labelled containers that indicate the kind and weight of the garbage are used. Following the removal of sharps and fluids,

FIGURE 4.2 Waste management categorization.

infectious waste is often stored in plastic bags, plastic-lined cardboard boxes, or other leak-proof containers that fulfil particular performance specifications. Colour coding is used to conveniently distinguish different forms of garbage (Zand and Heir, 2020). Red or yellow bags are routinely used to confine infectious garbage in most nations. General healthcare waste is collected in black or clear bags. Infectious waste containers are labelled in a contrasting colour with the international biohazard emblem. Sharps disposal containers are typically sturdy, leak-proof, break-resistant and puncture-resistant. Secondary leak-proof containers are chosen to avoid leakage from primary containers during shipment. Adequate methods should be used to identify proper container placement and labelling for increasing segregation efficiency and reducing improper container usage. In regions where both types of garbage are created, it is common practise to install normal trash containers next to infectious waste containers; this results in more effective and better segregation. It is also a good idea to use an adequate quantity of garbage bins. Posters depicting effective waste segregation systems are occasionally posted to walls in regions with many containers; these might act as reminders to health professionals about the benefits of using certain containers for distinct trash.

Handling the increasing volume of healthcare solid waste and the increased spread of infection, on the other hand, is the single most serious challenge originating from the new COVID-19 virus. Temporary healthcare waste treatment facilities and temporary transportation facilities can assist in properly managing waste and avoiding transmission during the COVID-19 pandemic. Waste collected from hospitals and other healthcare facilities can be moved directly or via temporary transit hubs to temporary or existing treatment units. Following that, the garbage can be sent to waste disposal facilities. As the number of illnesses fluctuates, waste creation is uncertain. As a result, temporary waste treatment and transportation may aid in the proper management of healthcare solid waste. The construction of more healthcare waste treatment capacity, as well as alternative technologies, may aid in the correct handling of waste. These alternate technologies, like autoclaves and high-temperature burn incinerators, may aid in waste management during the pandemic (Zhao et al., 2021a). The use of SF-CO_2 sterilising technology can assist in lowering the risk of infection from infectious healthcare waste. Sterilwave, an ultra-compact technology, can also aid in the treatment of healthcare waste since it successfully kills the COVID-19 virus on-site, preventing community spread during the processing of healthcare waste. This method can minimise waste weight, and processed garbage can even be managed as standard municipal waste. Furthermore, a transportable treatment system may assist in alleviating the additional strain of handling healthcare waste during the pandemic.

Boots, a long-sleeved gown, heavy-duty gloves, a mask and goggles or a face shield are all required PPE when managing hospital waste. After eliminating garbage, it is critical to maintain proper hand hygiene. PPE should be removed carefully after waste disposal, and sanitiser should be used to disinfect hands following waste disposal. Soiled personal protective equipment (PPE) must be placed in a sealed bag for safe cleaning, either off-site or on-site. The cleaning agent is a 10% lime slurry. Table 4.3 provides an overview of information on the measures used to manage healthcare waste during the COVID-19 pandemic.

Proper healthcare waste management may aid in increasing the percentage of recyclable trash. Autoclaving can enhance the amount of recyclable material, which may assist in minimising the volume of landfill garbage during the COVID-19 epidemic. An autoclave works on the basic principle of steam sterilisation by exposing infectious materials, such as viruses and bacteria, to direct steam at the requisite temperature and pressure for the specified period. Thus, autoclaving of healthcare solid waste can enhance recycling efficiency since the technique efficiently disinfects the contaminated waste. This recycling technique has the potential to minimise the cost of critical safety items for healthcare during a pandemic. Disinfectants such as sodium hypochlorite and alcohol, on the other hand, can inactivate the COVID-19 virus, which can live for up to 9 days (Zhao et al., 2022a). As a result, using disinfectants and storing healthcare waste for 9 days in a suitable containment facility can help to minimise the transmission of the virus among healthcare waste handling staff. Alternatively, healthcare waste might be used to generate electricity (Zhao et al., 2021b). During the

TABLE 4.3 An overview of quantifiable healthcare waste treatment techniques.

Measures taken	Relevance	Effect
Stockpiling	9-Day transient storage	Aids in the management of excess waste and the prevention of the virus's spread, which can last up to 9 days
Safety	Workers' PPE and hand sanitiser	Enables staff from being infected
Mechanism of treatment	Sterilwave (a transportable treatment device that is ultra-compact), autoclave, burn incinerator, SF-CO_2 sterilisation, microwave or radio-wave treatment	It eliminates the virus
Disinfectant	Prior to sorting, garbage must be disinfected	Restriction from being infected with the virus and transmitting it to others
The treatment of place	Treatment on-site	Suitable for dealing with excess garbage
Dealing	Sorting garbage and placing it in various bags according to the kind of waste	Simpler to use and dispose of
Social estrangement	1.5 m among personnel and the general public during the handling of waste	Prevents the transmission of infection
Education	Using personal protective equipment and sanitising and other methods	The use of these will aid prevention against infection
Publication of guidelines	Social media and workplace display	People will be more aware of health-related issues

pandemic, pyrolysis and combustion can be employed to create value-added products from hospital waste. Some prominent instances include the usage of incinerator ash from hospital waste in Portland cement. Furthermore, organic waste may be used to create compost and other valuable products, such as bioenergy. Considering all of these possibilities, employing healthcare waste for productive uses and a monetisation process during the pandemic can bring significant value to the circular economy (Zhao et al., 2022b).

4.13 Conclusions

Because of the unique COVID-19 virus's high infection rate, the volume of healthcare waste is rapidly growing. Workers in the waste management sector may become infected by virus-contaminated healthcare waste as a result of direct exposure to trash and inadequate safety precautions. As a result, the virus's transmission may be sluggish. During the epidemic, the WHO has issued clear recommendations for controlling healthcare waste. Different countries have taken various steps to appropriately manage healthcare waste. Effective safety measures and working techniques may enable effective healthcare waste management while preventing the infection from spreading to others. Disinfecting trash, followed by correct segregation and on-site treatment, can also result in better and healthier healthcare waste management. To accommodate excess healthcare waste, mobile treatment and temporary storage techniques may assist in the long-term management of healthcare waste while preventing the virus from spreading further. Proper healthcare waste management may also aid in the recycling of trash or the conversion of waste into valuable items, such as energy. As a result, good healthcare waste management may provide value to national economies and contribute to long-term growth. Furthermore, it will aid in the reduction of the COVID-19 virus's propagation.

References

Ab Rahman, M.F., Rusli, A., Misman, M.A., Rashid, A.A., 2020. Biodegradable gloves for waste management post-COVID-19 outbreak: a shelf-life prediction. ACS Omega. 5, 30329–30335. Available from: https://doi.org/10.1021/acsomega.0c04964.

Almulhim, A.I., Ahmad, I., Sarkar, S., Chavali, M., 2021. Consequences of COVID-19 pandemic on solid waste management: scenarios pertaining to developing countries. Remediation (N. Y.) . Available from: https://doi.org/10.1002/rem.21692, REM21692 [pii].

Baker, N., Bromley-Dulfano, R., Chan, J., Gupta, A., Herman, L., Jain, N., et al., 2020. COVID-19 solutions are climate solutions: lessons from reusable gowns. Front. Public. Health 8, 590275. Available from: https://doi.org/10.3389/fpubh.2020.590275.

Brisolara, K.F., Maal-Bared, R., Sobsey, M.D., Reimers, R.S., Rubin, A., Bastian, R.K., et al., 2021. Assessing and managing SARS-CoV-2 occupational health risk to workers handling residuals and biosolids. Sci. Total. Environ. 774, 145732. Available from: https://doi.org/10.1016/j.scitotenv.2021.145732.

Cai, M., Guy, C., Heroux, M., Lichtfouse, E., An, C., 2021. The impact of successive COVID-19 lockdowns on people mobility, lockdown efficiency, and municipal solid waste. Environ. Chem. Lett. 19, 3959–3965. Available from: https://doi.org/10.1007/s10311-021-01290-z; 1290 [pii].

Capoor, M.R., Parida, A., 2021. Current perspectives of biomedical waste management in context of COVID-19. Indian. J. Med. Microbiol. 39, 171–178. Available from: https://doi.org/10.1016/j.ijmmb.2021.03.003.

D'Alessandro, D., Gola, M., Appolloni, L., Dettori, M., Fara, G.M., Rebecchi, A., et al., 2020. COVID-19 and living space challenge. Well-being and public health recommendations for a healthy, safe, and sustainable housing. Acta Biomed. 91, 61–75. Available from: https://doi.org/10.23750/abm.v91i9-S.10115.

Dehghani, M.H., Roy, S., Karri, R.R., 2022. Novel coronavirus (COVID-19) in environmental engineering perspective. Environ. Sci. Pollut. Res. Int. Available from: https://doi.org/10.1007/s11356-022-18572-w, [pii].

Elsaid, K., Olabi, V., Sayed, E.T., Wilberforce, T., Abdelkareem, M.A., 2021. Effects of COVID-19 on the environment: an overview on air, water, wastewater, and solid waste. J. Environ. Manage. 292, 112694. Available from: https://doi.org/10.1016/j.jenvman.2021.112694.

Fadaei, A., 2021. Study of solid waste (municipal and medical) management during the COVID-19 pandemic: a review study. Rev. Environ. Health. Available from: https://doi.org/10.1515/reveh-2021-0092.

Gibbons, D.W., Sandbrook, C., Sutherland, W.J., Akter, R., Bradbury, R., Broad, S., et al., 2021. The relative importance of COVID-19 pandemic impacts on biodiversity conservation globally. Conserv. Biol. Available from: https://doi.org/10.1111/cobi.13781.

Govindan, K., Nasr, A.K., Mostafazadeh, P., Mina, H., 2021. Medical waste management during coronavirus disease 2019 (COVID-19) outbreak: a mathematical programming model. Comput. Ind. Eng. 162, 107668. Available from: https://doi.org/10.1016/j.cie.2021.107668; S0360-8352(21)00572-6 [pii].

Gwenzi, W., Rzymski, P., 2021. When silence goes viral, Africa sneezes! A perspective on Africa's subdued research response to COVID-19 and a call for local scientific evidence. Environ. Res. 194, 110637. Available from: https://doi.org/10.1016/j.envres.2020.110637.

Haghnazar, H., Cunningham, J.A., Kumar, V., Aghayani, E., Mehraein, M., 2022. COVID-19 and urban rivers: effects of lockdown period on surface water pollution and quality- a case study of the Zarjoub River, north of Iran. Environ. Sci. Pollut. Res. Int. Available from: https://doi.org/10.1007/s11356-021-18286-5, [pii].

Ikiz, E., Maclaren, V.W., Alfred, E., Sivanesan, S., 2021. Impact of COVID-19 on household waste flows, diversion and reuse: the case of multi-residential buildings in Toronto, Canada. Resour. Conserv. Recycl. 164, 105111. Available from: https://doi.org/10.1016/j.resconrec.2020.105111; S0921-3449(20)30428-6 [pii];105111 [pii].

Islam, M.S., Chowdhury, T.A., 2021. Effect of COVID-19 pandemic-induced lockdown (general holiday) on air quality of Dhaka City. Environ. Monit. Assess. 193, 343. Available from: https://doi.org/10.1007/s10661-021-09120-z, [pii].

Kalantary, R.R., Jamshidi, A., Mofrad, M.M.G., Jafari, A.J., Heidari, N., Fallahizadeh, S., et al., 2021. Effect of COVID-19 pandemic on medical waste management: a case study. J. Environ. Health Sci. Eng. 19, 831–836. Available from: https://doi.org/10.1007/s40201-021-00650-9; 650 [pii].

Kasim, O.F., Oyedotun, T.D.T., Famewo, A., Oyedotun, T.D., Moonsammy, S., Ally, N., et al., 2021. Household waste generation, change in waste composition and the exposure to COVID-19 in Guyana and Nigeria. Sci. Afr. 14, e01060. Available from: https://doi.org/10.1016/j.sciaf.2021.e01060; S2468-2276(21)00361-6 [pii].

Kaya, D., Niemeier, D., Ahmed, W., Kjellerup, B.V., 2022. Evaluation of multiple analytical methods for SARS-CoV-2 surveillance in wastewater samples. Sci. Total. Environ. 808, 152033. Available from: https://doi.org/10.1016/j.scitotenv.2021.152033.

Lazuka, A., Arnal, C., Soyeux, E., Sampson, M., Lepeuple, A.S., Deleuze, Y., et al., 2021. COVID-19 wastewater based epidemiology: long-term monitoring of 10 WWTP in France reveals the importance of the sampling context. Water Sci. Technol. 84, 1997–2013. Available from: https://doi.org/10.2166/wst.2021.418.

Loh, H.C., Looi, I., Ch'ng, A.S.H., Goh, K.W., Ming, L.C., Ang, K.H., 2021. Positive global environmental impacts of the COVID-19 pandemic lockdown: a review. GeoJournal. 1−13. Available from: https://doi.org/10.1007/s10708-021-10475-6; 10475 [pii].

Maher, O.A., Kamal, S.A., Newir, A., Persson, K.M., 2021. Utilization of greenhouse effect for the treatment of COVID-19 contaminated disposable waste - a simple technology for developing countries. Int. J. Hyg. Environ. Health 232, 113690. Available from: https://doi.org/10.1016/j.ijheh.2021.113690.

Mostafa, M.K., Gamal, G., Wafiq, A., 2021. The impact of COVID 19 on air pollution levels and other environmental indicators - a case study of Egypt. J. Environ. Manage 277, 111496. Available from: https://doi.org/10.1016/j.jenvman.2020.111496.

Park, C., Choi, H., Andrew Lin, K.Y., Kwon, E.E., Lee, J., 2021. COVID-19 mask waste to energy via thermochemical pathway: effect of co-feeding food waste. Energy (Oxf.) 230, 120876. Available from: https://doi.org/10.1016/j.energy.2021.120876; S0360-5442(21)01124-5 [pii].

Penteado, C.S.G., Castro, M.A.S., 2021. Covid-19 effects on municipal solid waste management: what can effectively be done in the Brazilian scenario? Resour. Conserv. Recycl. 164, 105152. Available from: https://doi.org/10.1016/j.resconrec.2020.105152; S0921-3449(20)30469-9 [pii]; 105152 [pii].

Pourakbar, M., Abdolahnejad, A., Raeghi, S., Ghayourdoost, F., Yousefi, R., Behnami, A., 2022. Comprehensive investigation of SARS-CoV-2 fate in wastewater and finding the virus transfer and destruction route through conventional activated sludge and sequencing batch reactor. Sci. Total. Environ. 806, 151391. Available from: https://doi.org/10.1016/j.scitotenv.2021.151391.

Qiu, Y., Yu, J., Pabbaraju, K., Lee, B.E., Gao, T., Ashbolt, N.J., et al., 2022. Validating and optimizing the method for molecular detection and quantification of SARS-CoV-2 in wastewater. Sci. Total. Environ. 812, 151434. Available from: https://doi.org/10.1016/j.scitotenv.2021.151434.

Ragazzi, M., Rada, E.C., Schiavon, M., 2020. Municipal solid waste management during the SARS-COV-2 outbreak and lockdown ease: lessons from Italy. Sci. Total. Environ. 745, 141159. Available from: https://doi.org/10.1016/j.scitotenv.2020.141159.

Rakib, M.R.J., De-la-Torre, G.E., Pizarro-Ortega, C.I., Dioses-Salinas, D.C., Al-Nahian, S., 2021. Personal protective equipment (PPE) pollution driven by the COVID-19 pandemic in Cox's Bazar, the longest natural beach in the world. Mar. Pollut. Bull. 169, 112497. Available from: https://doi.org/10.1016/j.marpolbul.2021.112497.

Ranjbari, M., Shams, E.Z., Gautam, S., Ferraris, A., Scagnelli, S.D., 2022. Waste management beyond the COVID-19 pandemic: bibliometric and text mining analyses. Gondwana. Res. Available from: https://doi.org/10.1016/j.gr.2021.12.015; S1342-937X(22)00027-2 [pii].

Sharma, H.B., Vanapalli, K.R., Cheela, V.S., Ranjan, V.P., Jaglan, A.K., Dubey, B., et al., 2020. Challenges, opportunities, and innovations for effective solid waste management during and post COVID-19 pandemic. Resour. Conserv. Recycl. 162, 105052. Available from: https://doi.org/10.1016/j.resconrec.2020.105052; S0921-3449(20)30369-4 [pii]; 105052 [pii].

Silva, D.A.L., Giusti, G., Rampasso, I.S., Junior, A.C.F., Marins, M.A.S., Anholon, R., 2021. The environmental impacts of face-to-face and remote university classes during the COVID-19 pandemic. Sustain. Prod. Consum. 27, 1975−1988. Available from: https://doi.org/10.1016/j.spc.2021.05.002; S2352-5509(21)00149-4 [pii].

Tomasino, M.P., Semedo, M., Moreira, V.E., Ferraz, E., Rocha, A., Carvalho, M.F., et al., 2021. SARS-CoV-2 RNA detected in urban wastewater from Porto, Portugal: method optimization and continuous 25-week monitoring. Sci. Total. Environ. 792, 148467. Available from: https://doi.org/10.1016/j.scitotenv.2021.148467.

Vu, H.L., Ng, K.T.W., Richter, A., Karimi, N., Kabir, G., 2021. Modeling of municipal waste disposal rates during COVID-19 using separated waste fraction models. Sci. Total. Environ. 789, 148024. Available from: https://doi.org/10.1016/j.scitotenv.2021.148024.

Wang, J., Chen, Z., Lang, X., Wang, S., Yang, L., Wu, X., et al., 2021. Quantitative evaluation of infectious health care wastes from numbers of confirmed, suspected and out-patients during COVID-19 pandemic: a case study of Wuhan. Waste Manag. 126, 323−330. Available from: https://doi.org/10.1016/j.wasman.2021.03.026.

Warmadewanthi, I., Wulandari, D., Cahyadi, M.N., Pandebesie, E.S., Anityasari, M., Dwipayanti, N.M.U., et al., 2021. Socio-economic impacts of the COVID-19 pandemic on waste bank closed-loop system in Surabaya, Indonesia. Waste Manag. Res. 39, 1039−1047. Available from: https://doi.org/10.1177/0734242X211017986.

Xu, G., Jiang, H., Stapelberg, M., Zhou, J., Liu, M., Li, Q.J., et al., 2021. Self-perpetuating carbon foam microwave plasma conversion of hydrocarbon wastes into useful fuels and chemicals. Environ. Sci. Technol. 55, 6239−6247. Available from: https://doi.org/10.1021/acs.est.0c06977.

Ye, Q., Asmi, F., Anwar, M.A., Zhou, R., Siddiquei, A.N., 2022. Health concerns among waste collectors during pandemic crisis. Environ. Sci. Pollut. Res. Int. 29, 6463−6478. Available from: https://doi.org/10.1007/s11356-021-16071-y, [pii].

Yousefi, M., Oskoei, V., Jonidi, J.A., Farzadkia, M., Hasham, F.M., Abdollahinejad, B., et al., 2021. Municipal solid waste management during COVID-19 pandemic: effects and repercussions. Environ. Sci. Pollut. Res. Int. Available from: https://doi.org/10.1007/s11356-021-14214-9, [pii].

Yuwen, C., Liu, B., Rong, Q., Zhang, L., Guo, S., 2022. Porous carbon materials derived from discarded COVID-19 masks via microwave solvothermal method for lithiumsulfur batteries. Sci. Total. Environ. 817, 152995. Available from: https://doi.org/10.1016/j.scitotenv.2022.152995.

Zand, A.D., Heir, A.V., 2020. Emerging challenges in urban waste management in Tehran, Iran during the COVID-19 pandemic. Resour. Conserv. Recycl. 162, 105051. Available from: https://doi.org/10.1016/j.resconrec.2020.105051; S0921-3449(20)30368-2 [pii]; 105051 [pii].

Zhao, H., Liu, H., Wei, G., Wang, H., Zhu, Y., Zhang, R., et al., 2021a. Comparative life cycle assessment of emergency disposal scenarios for medical waste during the COVID-19 pandemic in China. Waste Manag. 126, 388−399. Available from: https://doi.org/10.1016/j.wasman.2021.03.034.

Zhao, H.L., Wang, L., Liu, F., Liu, H.Q., Zhang, N., Zhu, Y.W., 2021b. Energy, environment and economy assessment of medical waste disposal technologies in China. Sci. Total. Environ. 796, 148964. Available from: https://doi.org/10.1016/j.scitotenv.2021.148964.

Zhao, H., Liu, H., Wei, G., Zhang, N., Qiao, H., Gong, Y., et al., 2022a. A review on emergency disposal and management of medical waste during the COVID-19 pandemic in China. Sci. Total. Environ. 810, 152302. Available from: https://doi.org/10.1016/j.scitotenv.2021.152302.

Zhao, X., Klemes, J.J., Fengqi, Y., 2022b. Energy and environmental sustainability of waste personal protective equipment (PPE) treatment under COVID-19. Renew. Sustain. Energy Rev. 153, 111786. Available from: https://doi.org/10.1016/j.rser.2021.111786; S1364-0321(21)01055-8 [pii].

Chapter 5

Conventional and modern waste treatment approaches – bioremediation of rubber waste

Mishel Pulikondan Francis[1], **Steffi Pulikondan Francis**[2] and **Vijayalakshmi Selvakumar**[3]

[1]Research Scholar, Department of Botany, Bharathidasan University, Palkalaiperur, Tiruchirappalli, Tamil Nadu, India, [2]Department of Microbiology, Cauvery College for Women (Autonomous), Tiruchirappalli, Tamil Nadu, India, [3]Food Science and Biotechnology, School of Agriculture and Lifesciences, Kangwon National University, Chuncheon, Gangwon-do, Republic of Korea

5.1 Introduction

Natural rubber is an elastic hydrocarbon polymer made from *Hevea brasiliensis* latex, which is a milky colloidal solution. Chemical polyisoprene is a refined version of rubber that may also be synthesised. Natural rubber is widely used in a variety of goods and applications. The release of effluent from the rubber processing sector to the environment without sufficient treatment may have catastrophic and long-term repercussions. As a result, appropriate methods must be applied to remediate this rubber waste (Arora et al., 2010).

Rubber accounts for more than 12% of all solid trash; however, it is difficult to decompose or recycle due to its size, shape and physiochemical makeup. Discarded tyres lose precious rubber, and hoarded tyres may easily catch fire and spew noxious gases, in addition to being a major source of worldwide environmental pollution (Chatterjee and Mohan, 2022). The yearly volume of waste rubber generated is rising and already exceeds 17 million tonnes, with automobile tyre rubber accounting for around 65%.

Rubber's natural deterioration is gradual because of the crosslinks between its polymer chains, as well as the numerous additives and stabilisers in its structure, making it very resistant to biodegradation. However, cremation may not be the ideal technique for maximising the rubber's economic potential, and recycling may be a superior option. Scrap rubber is converted

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00007-4

into finely ground particles called ‘rubber crumbs’ in most rubber recycling operations. Rubber crumbs can be mixed with new virgin rubber and rebounded with latex adhesives to create new rubber. However, some of the physical features of the original rubber are lost during this process, potentially affecting crucial performance attributes of the recycled rubber. Waste rubber has a variety of industrial uses in addition to direct recycling (Swarna Smitha et al., 2012).

More than half of all waste rubber is broken down and used as a component in the creation of road surface cement or burned as a source of power. Ground waste rubber may be used in asphalt mixtures to boost road surface durability at a low cost. The issues of biodegrading scrap tyre rubber will be discussed here. The methods that might be employed for rubber bioremediation will be discussed. Rubber recycling and bioremediation will be coupled to produce the greatest grade rubber for reuse, and suggestions will be offered (Stevenson et al., 2008).

Different polymers are used to make rubber. The product’s unique qualities are due to the polymer basis. These polymers frequently clash with one another. As a result, it is critical to understand rubber composition before reusing it. Rubber compounds may be made up of a variety of polymers, each of which may comprise 10–20 distinct chemical compounds. Metal or textiles are frequently used to strengthen rubber goods. Separating the reinforcing material from the rubber is required for treating tyres, conveyor belts or lip seals. It is technically challenging to accomplish since the aim during the development stage is to establish the strongest possible interaction between the rubber and the reinforcing material. As a result, collecting and classifying rubber waste is essential for effective rubber waste processing.

As a result, efficient strategies for resolving these difficulties must be used to treat rubber wastewater. Several procedures for treating rubber waste have been developed, which will be explored in depth in this article (Stevenson et al., 2008).

5.2 Rubber in everyday life

Rubber goods may be found all throughout the world, not just in affluent countries but also in developing ones, yet few people are aware of all of the uses for rubber. Since 1920, the car sector has been the largest user of rubber products, driving demand for rubber manufacture (Islam et al., 2021). Rubber is found in radios, televisions and telephones. Rubber insulation protects electrical cables. Rubber is used in a variety of kitchen mechanical gadgets. It helps to keep draughts at bay while also insulating against noise. Foam rubber cushions may be found on sofas and chairs, and natural rubber pillows and mattresses can be found on beds. Rubber can be found in clothing and footwear, such as elasticized threads in underwear or shoe soles (Bode et al., 2001). Rubber is used in some or all of the pieces of most sports equipment, practically all balls and many mechanical toys. Other uses have emerged as a result of the unique qualities of particular types of synthetic rubber, and there are currently over 100,000 different items that use rubber as a basic material.

Not only in industrialised countries but also in less developed nations, rubber products are everywhere to be found, although few people recognise rubber in all of its applications. Since 1920, demand for rubber manufacturing has been largely dependent on the automobile industry, the biggest consumer of rubber products. Rubber is used in radio and T.V sets and in telephones. Electric wires are made safe by rubber insulation. Rubber forms a part of many mechanical devices in the kitchen. It helps to exclude draughts and to insulate against noise. Sofas and chairs may be upholstered with foam rubber cushions, and beds may have natural rubber pillows and mattresses. Clothing and footwear may contain rubber: e.g., elasticized threads in undergarments or shoe soles. Most sports equipment, virtually all balls and many mechanical toys contain rubber in some or all of their parts.

Still other applications have been developed due to the special properties of certain types of synthetic rubber, and there are now more than 100,000 types of articles in which rubber is used as a raw material.

5.3 Rubber production and structure

Many plants and certain fungi generate NR, and the composition of NR produced by various species varies. Despite the widespread presence of NR, *H. brasiliensis* supplies nearly all commercially used rubber. NR is a biopolymer made nearly completely of poly (cis-1,4-isoprene) units, each with the formula C_5H_8 with a single cis double bond. In the terminal regions of the NR polymers, there are, however, two trans-isoprene units. Although the natural poly (cis-1,4-isoprene) version is harder to manufacture, it has a good tear and abrasion resistance. During vulcanisation, sulphur bridges are formed between the poly (cis-1,4-isoprene) chains, reducing the number of accessible sites for rubber-degrading bacteria to cleave the polymer chains (Stevenson et al., 2008).

As a result, the rubber's biodegradability is reduced. Antioxidants also limit microbial colonisation of NR, and the hydrophobicity of polyisoprene chains in NR improves the material's resistance to microbial destruction (Sharma and Garg, 2019).

5.4 Vulcanisation

Vulcanisation is a thermochemical process that includes a series of complicated chemical reactions involving rubber polymers, sulphur and auxiliary (supplementary) chemicals, which was pioneered by Charles Goodyear in 1839 (Stevenson et al., 2008).

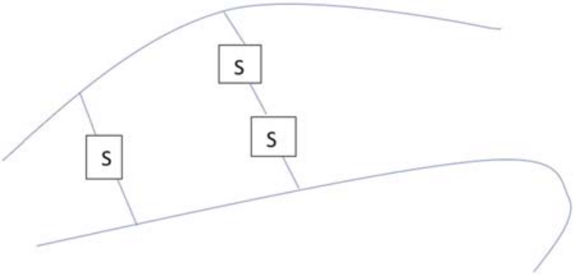


FIGURE 5.1 Rubber contains monosulphidic and disulphidic acids, among other things.

Rubber that has been vulcanised is less flexible than rubber that has not been vulcanised. On the other hand, it has more flexibility and is far more durable. During vulcanisation, sulphide bridges connect polyisoprene chains in rubber, ranging from monosulphidic and disulphidic bridges to polysulphidic bridges with as many as five sulphur atoms (see Fig. 5.1). The kind and number of crosslinks produced between rubber polymers determine the majority of a rubber material's qualities (Braaz et al., 2005).

Rubber materials can also be vulcanised using radiation. Greater doses of radiation were associated with rubber with higher tensile strength and thermal stability when NR latex was vulcanised with radiation in the presence of styrene butadiene rubber (SBR) latex (Chaudhari et al., 2005).

5.5 The presence of a three-dimensional network produced during vulcanisation

- The range of rubber compound compositions; and the complicated structure of rubber products make waste rubber extremely difficult to recycle.
- During vulcanisation, the polymer chains in rubber are cured. Chemical bonds with high energy are created.
- Sulphur is the most prevalent crosslinking agent. Sulphur crosslinks are equivalent to polymer chain bond energy.
- The most common crosslinking agent is sulphur. Sulphur crosslinks are given below (Table 5.1).

The rubber's three-dimensional network allows for large reversible deformations. This makes grinding rubber extremely tough. Rubber swells only to a limited amount in some solvents due to the three-dimensional network that prevents it from dissolving (Sharma and Garg, 2019). Chemically destroying the crosslinks in the bulk of rubber is impossible due to the intricacy of grinding and insolubility.

Vulcanised rubber with steel or fabric belts and reinforcing textile strands are other common components of rubber tyres. Tyre rubber is made up of a mix of natural and synthetic rubbers (most notably SBR) with carbon black

TABLE 5.1 Energy of bonds in the sulphur-cured rubber.

Bond type	Bond energy (Kj/mol)
$-C-S_X-C-$	<270
$-C-S-S-C-$	270
$-C-S-C-$	286
$-C-C-$	353

TABLE 5.2 A vulcanisation mixture's typical composition.

Components of mixture	Quantity of matter used	Content
Natural or synthetic polyolefin	100	>57
Carbon black	50	>39
Sulphur	0.5–4	<2.9
Zinc oxide	2–4	<3
Stearic acid	1–4	<3
Organic stimulators	0.5–2	<2
Antioxidants	Less than 0.5	<0.5
Retarders	Less than 0.5	<0.5
Other materials	Less than 0.5	<0.5

added for strength and abrasion resistance. Tread compounds based on NR or mixes of NR and synthetic polymers, as well as high-grade carbon black, are used in truck tyres. The specific composition of tyre rubber is determined by a variety of variables, and different tyre manufacturers use different chemicals in their tyres. A natural or synthetic polyolefin (polyalkane) typically makes up the majority of a vulcanisation mixture, with carbon black accounting for a sizable fraction. Other additives, such as elemental sulphur, zinc oxide, stearic acid and organic accelerators, as well as minor amounts of antioxidants, retarders and other compounds (see Table 5.2), are also included in smaller proportions. Although zinc oxide is widely recognised as the finest sulphur vulcanisation accelerator, additional chemicals like as stearic acid and sulphur-containing organic compounds are occasionally employed in conjunction with zinc compounds.

The Integrated Waste Management Board (IWMB) is a non-profit organisation that, In recent years, there has been pressure on the rubber industry to minimise or even eliminate zinc oxide from rubber products because of environmental concerns about the harmful effects of zinc compounds on aquatic environments (Braaz, et al., 2004).

5.6 Waste treatment using microbes

Microbial biotechnology in waste treatment is the use of contemporary scientific tools and procedures for a wide range of microorganisms in a controlled environment without causing ecological disruption. Composting, biodegradation, bioremediation and biotransformation are the most prevalent and efficient Waste Treatment procedures used at various levels. *Bacillus* sp., *Corynebacterium* sp., *Staphylococcus* sp., *Streptococcus* sp., *Scenedesmus platydiscus*, *S. quadricauda*, *Selenastrum capricornutum*, *Chlorella vulgaris* and others have all been shown to be effective for waste treatment.

5.7 Biological rubber degradation

Rubber may be detoxified, devulcanised and degraded by a variety of fungal and bacterial species (Tsuchii and Tokiwa, 2006). Rubber is a synthetic waste that is classified as solid waste. Rubber may be disposed of in an unscientific manner by burning it. Rubber burning produces dangerous components, such as carbon monoxide, which has a terrible effect on the environment. It cannot readily decompose or recycle due to its physical nature. Rubber degrades slowly in the natural world because it contains zinc oxides, which prevent the growth of naturally developed bacteria and sulphur-oxidising bacteria. To solve this problem, *Recinicium bicolor* (fungi) is used to remove all of the rubber's toxic and ecologically unfavourable ingredients, after which the rubber can be devulcanised using sulphur-reducing or sulphur-oxidising bacteria, such as *Pyrococcus furiosus* and *Thiobacillus ferrooxidans*. Rubbers may be easily recycled after these processes. Rubber waster may therefore be effectively managed by controlled kindling.

R. bicolor, Rhodococcus rhodochrous, Corynebateria spp., *Pseudomonas* spp. and *Escherichia coli* have all been used to successfully detoxify vulcanised rubber. By oxidising disulphide bonds in rubber, sulphur-oxidising bacteria, particularly thiobacillus spp., can be used to devulcanise it. A sulphur-reducing archeon, *P. furiosus*, has also been used to successfully devulcanise rubber. The poly(cis-1,4-isoprene) chains in rubber have been shown to be degraded by both clear zone—forming bacteria and adhesively growing bacteria (Swarna Smitha et al., 2012). The latex-clearing protein (lcp) and rubber oxygenase (roxa), both of which are thought to be involved in poly(cis-1,4-isoprene) cleavage, as well as the oxidoreductase complex, which is thought to be involved in subsequent metabolic processes, have all been identified as being involved in rubber metabolism.

5.8 Biological detoxification

Many substances introduced during vulcanisation, such as zinc oxide and zinc salts, restrict microbial development, and they must be eliminated before successful microbial devulcanisation and metabolism can occur. Detoxification by several fungal species has been effectively employed to detoxify rubber material by degrading certain aromatic chemicals contained in the substance (Sharma and Garg, 2019). Many white rot fungus species contain effective lignin-degrading enzymes and are already employed in soil bioremediation methods. White rot fungus has been shown to break down xenobiotics with structures that are remarkably similar to aromatic chemicals added to rubber. This implies that white rot fungus might be employed to detoxify rubber as well.

Different white rot and brown rot fungal species and strains have been evaluated to see which ones can break down aromatic chemicals and hence might be effective in rubber detoxification. *Pleu rotus sajor-caju, Trametes versicolor* and *R. bicolor* (Tsuchii et al., 1985), three white rot fungus species, were discovered to biodegrade Poly-R478 in the presence of cryo-ground tyre rubber (CGTR). *R. bicolor*, the most efficient fungus, was then used to treat pure CGTR. *Thiobacillus ferroxidans*, a desulphurising bacterium, grew considerably faster on rubber that had been prepared with the fungus *R. bicolor* than on untreated rubber. This suggests that the fungus *R. bicolor* detoxified the CGTR, making it easier for other bacteria to thrive on the rubber (Stevenson et al., 2008). These experiments demonstrate the feasibility of a multistep bioremediation procedure in which rubber is detoxified first and subsequently devulcanised. Different white rot and brown rot fungus strains have been investigated to discover which ones can break down aromatic chemicals and hence might be effective in rubber detoxification (see Table 5.3).

TABLE 5.3 Species of fungi that have been tested and investigated for their ability to break down rubber materials.

Fungal species	Strains	Fungal type
Coniophora puteana	MC 53	Brown rot
Serpul lacrimans	V 750829-01	Brown rot
Crucibulum laeve	LFB 9509-4	White rot
Pleurotus sajor-caju	MUCL 29527	White rot
Trametes versicolor	PRL 572	White rot
Recinicium bicolor	V 72284	White rot
Hypholoma fasciculare	LFV 9409-2	White rot
Antrodia vaillantii	V 72086-2	Brown rot
Collybia maculata	LFV 9509-2	White rot

TABLE 5.4 Overview of waste rubber recycling approaches, crumb rubber properties and associated environmental risks.

Approaches	Regarding surface activity	Size of the particle	Separation of materials	Pollution risk
Devulcanization	Highly chemical	30–100	Preliminary separation	Dust (Hazardous agents)
Ozone cutting	Low surface materials due to oxidation	Upto 140	Highly effective	Local ozone Leakage
Reclaiming	Rubber-like substances	n/a	Preliminary separation	Solvents

5.9 Bacterial detoxification

One of the toxic additives in rubber is 2-mercaptobenzothiazole (MBT), a vulcanisation accelerator; it is one of the very poisonous chemicals in rubber. The bacteria *R. rhodochrous* has been found to be able to break down or biotransform MBT. MBT is a common contact allergen that can cause severe dermatoses (Stevenson et al., 2008). Ingestion of MBT is especially dangerous, as it has been related to bladder cancer in people. Ben zothiazolylsulphonate and 2-methlthiobenzothiazole (MTBT) are less hazardous metabolites formed and discovered during MBT biodegradation (Chatterjee and Mohan, 2022). As a result, using *Rhodococcus sp*. to metabolise MBT might considerably improve tyre disposal safety and make rubber more biodegradable. Members of the *Corynebateria* and *Pseudomonas* genera, as well as *E. coli*, are among the bacteria that may metabolise MBT. Because toxic benzothiazole compounds have been discovered as an environmental pollutant in road dust, and it may be safer to employ an alternate accelerator to MBT in tyre manufacturing (Table 5.4).

5.10 Devulcanisation biologically

Rubber must be devulcanised before it can be recycled (at least to some extent). During devulcanisation, sulphur oxidation in the outer few micrometres of rubber particles produces an active surface, converting the outer layer of ground rubber from elastic to viscous. Many types of microbial strains are involved in Devulcanisation Biologically (Fig. 5.2). The viscous rubber can then be revulcanised by mixing it with fresh rubber. The sulphur-oxidising activity of enzymes generated by sulphur-using microorganisms can break down the sulphur crosslinks in vulcanised rubber. The hydrocarbon chains are

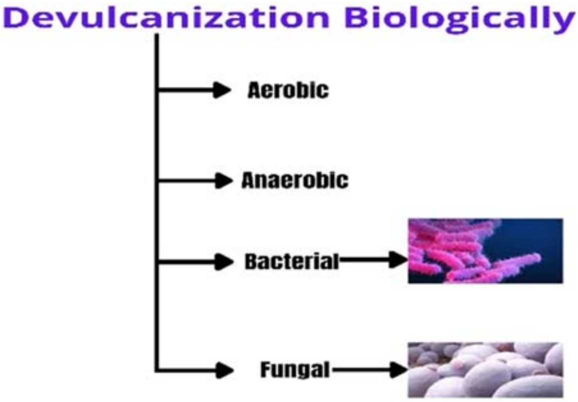


FIGURE 5.2 Devulcanisation biologically.

not broken during the sulphur oxidisation process (Stevenson et al., 2008). Microbes, on the other hand, can only degrade the surfaces of rubber particles. Rubber may be desulphurised by both anaerobic sulphur-reducing archaeons and aerobic sulphur-oxidising bacteria. Anaerobic sulphur reduction, on the other hand, inhibits hydrocarbon chain oxidation and generates higher-quality rubber than the aerobic sulphur-oxidising bacteria procedure.

5.11 Sulphur-oxidising bacteria

Different sulphur-oxidising bacteria were cultivated on rubber material in a variety of tests to examine bacterial devulcanisation. The rate of disulphide bond oxidation may be evaluated by measuring the amount of sulphate generated by a bacterial species growing on a plate containing rubber material (Navarro et al., 2007). According to research, recognised sulphur-oxidising bacteria have the ability to degrade rubber.

Three types of *Thiobacillus* bacteria were used to break down rubber in these experiments: *Thiobacillus ferrooxidans, Thiobacillus thioparus and Thiobacillus thiooxidans. T. ferrooxidans* was shown to be the most effective sulphur-oxidising species in terms of released sulphate. Microorganisms capable of creating desulphurised, surface-modified, ground tyre rubber have been identified via several trials (GTR) (Watanabe, 2001). *Thiobacillus, Rhodococcus* and *Sulfolobus* bacterial strains were evaluated for their ability to desulphurise sulphur-containing substances using DBT degradation as an indication of desulphurising activity. Strains that degraded DBT were studied further to determine whether their ability to desulphurise DBT might be used to devulcanise other rubber compounds. Ion chromatography (IC), Fourier-transform infrared spectroscopy (FT-IR), and X-ray examination of near-edge surfaces were all employed to chemically analyse the results of GTR desulphurisation (XANES). IC was used to track the amount of sulphate in the solution as the biodesulphurisation process progressed. Because untreated GTR samples had a sulphur level of around 1.8%, the change in sulphur concentration was determined from the data. The final results were displayed as the cumulative percentage of bound sulphur transformed to sulphate as a function of time to identify whether microbes were more bioreactive. The FT-IR and XANES analysis of GTR's surface chemistry confirmed that bound sulphur had been oxidised. The results of the Romine and Romine tests suggest that a GTR degenerative process involving sulphur oxidation and sulphur removal from the rubber surface exists. Bioreactivity was stronger in a mixed *Thiobacillus* culture than in individual Thiobacillus strains.

Experiments with *Sulfolobus acidocal darius*, on the other hand, revealed the greatest conservation of bonded sulphur to sulphate. Microorganisms were observed to permeate the rubber material to a depth of only 2 microns, showing that GTR desulphurisation is a surface event. For chemical analysis, scanning electron microscopy (SEM), energy-dispersive x-ray spectrometry (EDS) and electron spectroscopy can be used to assess the desulphurisation of rubber (ESCA) (Watanabe, 2001). Cryo ground tyre rubber (CGTR) with a blend of different rubber kinds was employed in *Thiobacillus* tests. The biggest drawback of the experiment was that the rubber's specific composition was unknown. This makes determining whether certain chemicals suppressed bacterial growth or hampered the desulphurisation process challenging (Zabaniotou and Stavropoulos, 2003). Wax, on the other hand, was thought to hinder desulphurisation and was removed from the rubber surface using an n-heptane solution prior to microbiological treatment. The concentration of sulphur around the borders of the rubber material was found to be lower when it was treated with several *Thiobacillus* sulphur-degrading species compared to the sulphur concentration near the middle of the material. Another issue with these studies was that the techniques for monitoring desulphurisation only included the discharge of completely oxidised sulphur from the rubber (Stevenson et al., 2008). As a result, sulphur bridges that the bacteria partially oxidised will not be identified. In vulcanised rubber, some sulphur is dissolved in the rubber and is not engaged in crosslinking. Some sulphur is adsorbed onto the surface of carbon black particles, but it has little effect on vulcanisation.

5.12 Archaeon enzyme eliminates sulphur

The sulphur-reducing anaerobic archaeon *P. furiosus* has been used to successfully devulcanise rubber. *P. furiosus* is a thermophilicheterotroph, and the hydrogen it creates inhibits its growth. However, by adding sulphur to the growth media, which reacts with the hydrogen to generate hydrogen sulphide, this inhibition may be avoided. Some rubber compounds, like as zinc salts and zinc oxide, have been demonstrated to be poisonous to *P. furiosus* because they contain heavy metal ions. *P. furiosus* is similarly poisonous to the curing agent *N*-oxydiethene-2-benzothiazylsulphenamide. Stabilisers added to the elastomers in CGTR may further contribute to the material's toxicity.

Rubber may be leached with solvents like ethanol to eliminate some of these harmful compounds, allowing microorganisms to flourish normally. For ten days, ethanol-leached CGTR was cultured with *P. furiosus*. The rubber was then vulcanised again with new rubber. Tensile testing, stress relaxation tests and swelling measurements were used to assess the mechanical properties of the rubber material formed after treatment. It was discovered that *P. furiosus* treated rubber was able to revulcanise more successfully than untreated rubber. This indicates that *P. furiosus* was able to destroy sulphide bridges between polyisoprene chains, allowing for the formation of new crosslinks during revulcanisation. Furthermore, *P. furiosus* thrived in the presence of NR, latex and ethanol-leached CGTR. Fresh tyre rubber, untreated CGTR and NR-containing accelerators and fillers, on the other hand, did not support *P. furiosus* growth.

Rubber desulphurisation may be observed in three ways: an increase in sulphur produced in the medium after treatment, a decrease in sulphur on the surfaces of rubber particles and changes in the rubber's mechanical characteristics. *P. furiosus* may metabolise sulphur in rubber, as evidenced by increased sulphur production when cultured on vulcanised NR. *P. furiosus* has a larger cell mass (Stephen et al., 2006), indicating that it develops better on rubber once additives are eliminated. The mechanical characteristics of rubber are changed by microbial treatment, as evidenced by tensile tests, stress relaxation tests and swelling measures (Watanabe, 2001). Uninoculated control samples, on the other hand, revealed a change in characteristics following treatment, demonstrating that high temperatures and ethanol-leaching processes modify rubber structure as well. Material treated with *P. furiosus* for 10 days had worse relaxation quality and was less flexible than virgin rubber. Hydrocarbon chains on the surface of a rubber material might be degraded using a combination of microbiological and heat-treatment methods, allowing new crosslink synthesis during revulcanisation. The stress-at-break values observed for impure crumbs in rubber material were lower than those found for virgin rubber material. *T. ferrooxidans*-treated rubber showed a stress-at-break value equivalent to *P. furiosus*-treated rubber. This shows that anaerobic therapy might be just as effective as an oxidative therapy.

5.13 Decomposition of rubber

Rubber degradation includes detoxification and devulcanisation, but it also requires the break down of polyisoprene monomers inside the rubber matrix. The diverse techniques used by rubber-degrading microbes to metabolise polyisoprene polymers identify two primary types of microorganisms. On latex agar plates (Lapkovskis et al., 2020), one group (clear zone–forming bacteria) may develop clear zones where latex is the only carbon and energy source. Members of the genera *Streptomyces, Xanthomonas, Micromonospora, Thermomonospora* and *Actinomyces* generate clear zones. The bacteria that grow adhesively on latex and require direct contact with a rubber substance to break down the polyisoprene chains in its structure are the other category. On latex agar plates, this group of bacteria does not grow or produce distinct zones. Members of the genera *Gordonia, Corynebacterium, Mycobacterium* and *Nocardia* adhere to one other and metabolise poly quicker than clear zone–forming bacteria. Some *Pseudomonas* bacteria have also been discovered to be capable of degrading both crosslinked and uncrosslinked NR.

5.14 Clear zone–forming bacteria

Both natural and manufactured poly (cis-1,4-isoprene) rubber have been found to be degraded by *Streptomyces* sp. strain K30. Even in vulcanised natural rubber, *Streptomyces coelicolor* has been demonstrated to break down high-molecular-weight poly (cis-1,4-isoprene) chains. A colony of *S. coelicolor* 1 A cultivated on vulcanised rubber produced degradation products, including aldehyde groups (Roy et al., 2006). Successful oxidation processes are thought to degrade poly (cis-1,4-isoprene) to methyl-branched diketones. The oxidative breakage of double bonds, driven by a dioxygenase or peroxidase enzyme, is thought to be the catalyst for the disintegration of poly (cis-1,4-isoprene) polymers. The resultant polyisoprene fragments are expected to be oxidised further to their respective carboxylic acids. Coenzyme A (CoA-SH) is thought to have a role in the break down of carboxylic acids through a series of -oxidation processes.

5.15 Adhesively growing bacteria

Gordonia bacteria have been discovered to be especially good at degrading rubber. *Gordonia polyisoprenivorans and Gordonia westfalica* may solubilise and mineralise rubber substrates. Although the hydrophobicity of *Gordonia* cell surfaces, which is controlled by the presence of mycolic acids, is assumed to be involved in the bacteria's rubber breakdown, the chemical basis for this is unclear (Watanabe, 2001).

The production of mycolic acids and biosurfactants is expected to enhance biofilm growth. Biofilms allow bacteria to come into intimate contact with cis-1,4-polyisoprene, which is required for these strains to break down rubber. *Nocardia* grows well on unvulcanised natural rubber and synthetic isoprene rubber, but not on other types of synthetic rubber. Carbon and energy may be obtained by *Nocardia* sp. from a range of unvulcanised and vulcanised NR products. The tread compound used in truck tyres can be partially damaged by a mutant *Nocardia* strain (835 A). This strain has the ability to mineralise or dissolve NR polymers in tread compounds, leaving microscopic black particles of organic fillers and leftover rubber behind. *Nocardia farinica S3* was isolated from Egyptian sandy soil and demonstrated a high polyisoprene breakdown rate.

5.16 Reclamation of rubber products

The challenge of trash disposal management is one of the many issues that humanity faces as it enters the 21st century. Waste polymers are a severe environmental hazard since polymeric materials do not break down readily (Ibrahim et al., 2006). Tyres for aircraft, lorries, cars, two-wheelers and other vehicles consume a lot of rubber. When discarded vulcanised scrap rubber is treated to generate a plastic substance that may be easily processed, compounded and vulcanised with or without the inclusion of natural or synthetic rubbers, the result is reclaimed rubber. Regeneration can take place via breaking existing crosslinks in the vulcanised polymer, encouraging the scission of the polymer's main chain, or a combination of both (Lapkovskis et al., 2020). The process of reclaiming leftover rubber might be complicated. However, there are several reasons why waste rubber should be salvaged or recovered.

- Natural or synthetic rubber can be half the price of recovered rubber.
- Some qualities of recovered rubber are superior to those of virgin rubber.
- Rubber made from reclaimed materials uses less energy in the manufacturing process than rubber made from virgin materials.
- It is a great method to get rid of undesirable rubber items, which might be tough to do.
- It reduces the usage of non-renewable petroleum products in the manufacture of synthetic rubbers.
- Recycling initiatives can help underdeveloped countries create jobs.
- Recycled tyres and other rubber materials may be used to make a variety of useful items.
- Tyres may be burnt to recover embodied energy and produce significant amounts of usable power. Some cement plants in Australia use discarded tyres as a fuel source.

As a result, reclaiming scrap rubber is the most ideal solution to the disposal problem. The vulcanised structure of vulcanised rubber granules is broken down using heat, chemicals and mechanical processes to generate reclaim (Holst et al., 1998). Reclaim possesses the flexibility of a fresh unvulcanised rubber compound, but because the molecular weight is lower, the physical qualities of reclaim compounds are inferior to new rubber.

The major reasons for its usage are cost and better rubber compound processing. The following are the primary processing benefits claimed:

1. quicker processing on extruders and calenders
2. shorter mixing periods

3. lower energy usage
4. lower heat development
5. the unvulcanised compound has a lower die swell
6. the compounds cure quicker

In many rubber products, 5%–10% reclaimed rubber may be added to the fresh rubber component without affecting the physical qualities significantly. In items like automobile mats, far larger amounts (20%–40%) are used (Haroune et al., 2004). However, due to processing benefits, compounds used in the fabrication of tyre carcasses have traditionally been the primary outlet for reclaim. Despite this, the percentage of radial tyres with reclamation is only about 2%–5%.

Reclaiming scrap rubber products, such as used automobile tyres and tubes, hoses and conveyor belts, is the process of converting a three-dimensionally interlinked, insoluble and infusible strong thermoset polymer into a two-dimensional, soft, plastic, more tacky, low modulus, processable and vulcanisable essentially thermoplastic product that mimics many of the properties of virgin rubber.

Rubber recovery and recycling from old and scrap rubber goods can therefore help preserve critical petroleum resources while also addressing scrap/waste rubber disposal issues.

5.17 Conclusion

It is now recognised that vulcanised tyre rubber may be detoxified by bacteria and fungus. Detoxification improves the biodegradability of rubber while also lowering the environmental risks connected with its disposal (Watanabe, 2001). The sulphur-removing capacity of sulphur-oxidising and sulphur-reducing microorganisms has been proven to devulcanise rubber products in studies. Many rubber-degrading bacterial species have been discovered as a result of extensive studies on the subject of rubber metabolism. Although our understanding of the processes involved in the enzymatic break down of rubber polymers is still restricted, we have made significant progress in recent years.

Rubber biodegradation products have shown plausible routes for rubber metabolism, whereas gene cloning has identified two nonhomologous enzymes involved in the early stages of rubber breakdown. The genes that produce these enzymes have now been found in *Streptomyces* and *Xanthomonas* (Bredberg et al., 2001). Microbial breakdown of waste rubber might be improved by changing the composition of tyres to make them more susceptible to microbial degradation, assuming that this could be done without affecting critical wheel performance. Unfortunately, some microbial-inhibiting compounds in tyre rubber may not be able to be reduced without affecting tyre performance and safety.

Synthetic rubbers and additives were first put into tyre rubber to increase physical qualities and even prevent microbial deterioration; however, in light of growing environmental concerns, it may be preferable to change tyre composition to improve rubber biodegradability. A multistage biodegradation approach involving multiple distinct microorganisms and biochemical processes will be efficient. For eliminating chemicals that limit microbial development, the initial step might involve detoxifying rubber using the fungus *R. bicolor* (Diepgen et al., 2006). After detoxification, the rubber might be devulcanised using a sulphur-oxidising bacterial species such *Thiobacillus* ferroxidans or the sulphur-reducing archaeon *P. furiosus*. After that, the devulcanised rubber may be recycled. Devulcanised rubber, on the other hand, might be broken down further by microbial rubber metabolism if deterioration is desired (Watanabe, 2001).

Rubber-metabolising microorganisms, such as adhesively growing *Gordonia* species and clear zone−forming Streptomyces species, are more sensitive to enzymatic destruction by devulcanised rubber (Alexander, 1994). Before successful recycling and bioremediation systems can be created, more molecular and biochemical research on rubber-degrading enzymes is required. The mechanisms involved in rubber biodegradation might be better understood if the molecular structures of these enzymes were more precise. More study is needed to see how rubber production processes might be changed to produce rubber that is simpler to desulphurise and biodegrade (Swarna Smitha et al., 2012).

Unfortunately, one universal or optimal rubber processing technique that may be deemed the best for all cases cannot be distinguished from the listed rubber processing methods (Roy et al., 2006). It would be beneficial if the best approaches for finding an optimal solution to a certain technical challenge were known and if some methods could be rejected to minimise confusion. It would also be beneficial to understand the options from which the best approach might be selected based on cost−benefit, environmental effect and feasibility evaluations. With the fast advancement of manufacturing technologies, the ever-increasing demands for product technological qualities, as well as the tightening of processing and reuse criteria, Certain technical procedures have been rejected in favour of integrating them into two or more processing techniques.

References

Alexander, M., 1994. Biodegradation and Bioremediation. Academic Press, San Diego, CA.

Arora, P.K., Srivastava, A., Singh, V.P., 2010. Application of Monooxygenases in dehalogenation, desulphurization, denitrification and hydroxylation of aromatic compounds. J. Bioremed. Biodegr. 1, 1−8.

Bode, H., Kerkhoff, K., Jendrossek, D., 2001. Bacterial degradation of natural and synthetic rubber. Biomacromolecules 2, 295−303.

Braaz, R., Fischer, P., Jendrossek, D., 2004. Novel type of hemedependent oxygenase catalyzes oxidative cleavage of rubber (polycis-1,4-Isoprene). App. Environ. Microbiol. 70, 7388−7395.

Braaz, R., Armbruster, W., Jendrossek, D., 2005. Heme-dependent rubber oxygenase RoxA of Xanthomonas sp. Cleaves the carbon backbone of poly(cis-1,4-isoprene) by a dioxygenase mechanism. Appl. Environ. Microbiol. 71, 2473−2478.

Bredberg, K., Persson, J., Christiansson, M., Stenberg, B., Holst, O., 2001. Anaerobic desulfurisation of ground rubber with the thermophilic archeaon *Pyrococcus furiosus*—a new method for rubber recycling. Appl. Microbiol. Biotechnol. 55, 42−48.

Chatterjee, S., Mohan, S.V., 2022. Fungal biorefinery for sustainable resource recovery from waste. Bioresour. Technol. 345, 126443.

Chaudhari, C.V., Bhardwaj, Y.K., Patil, N.D., Dubey, K.A., Kumar, V., Sabbarwal, S., 2005. Radiation induced vulcanization of natural rubber latex in the presence of styrene-butadiene rubber latex. Radiat. Phys. Chem. 72, 613−618.

Diepgen, T.L., Bruynzeel, D.P., Andersen, K.E., Brandao, F.M., Bruze, M., Goncalo, M., et al., 2006. Mercaptobenzothiazole or the mercapto-mix: which should be in the standard series? Contact Dermat. 55, 36−38.

Haroune, N., Combourieu, B., Besse, P., Sancelme, M., et al., 2004. Metabolism of 2-mercaptobenzothiazole by *Rhodococcus rhodochrous*. Appl. Environ. Microbiol. 70 (10), 6315−6319.

Holst, O., Stenberg, B., Christiansson, M., 1998. Biotechnological possibilities of waste rubber treatment. Biodegradation 9, 301−310.

Ibrahim, E.M.A., Arenskötter, M., Luftmann, H., Steinbüchel, A., 2006. Identification of poly (cis-1,4-isoprene) degradation intermediates during growth of moderately thermophilic actinomycetes on rubber and cloning of a functional lcp homologue from Nocardia farcinica strain E1. Appl. Environ. Microbiol. 71, 3375−3382.

Islam, A., Swaraz, A.M., Teo, S.H., Taufiq-Yap, Y.H., Vo, D.-V.N., Ibrahim, M.L., et al., 2021. Advances in physiochemical and biotechnological approaches for sustainable metal recovery from e-waste: a critical review. J. Clean. Prod. 323, 129015.

Lapkovskis, V., Mironovs, V., Kasperovich, A., Myadelets, V., Goljandin, D., 2020. Crumb Rubber as a secondary raw material from waste rubber: A short review of end-of-life mechanical processing methods. Recycling 5, 32.

Navarro, F.J., Partal, P., Martínez-Boza, F.J., Gallegos, C., 2007. Influence of processing conditions on the rheological behavior of crumb tire rubber-modified bitumen. J. Appl. Polym. Sci. 104, 1683−1691.

Roy, R.V., Das, M., Banerjee, R., Bhowmich, A.K., 2006. Comparative studies on cross linked and uncross linked natural rubber biodegradation by *Pseudomonas* sp. Bioresour. Technol. 97, 2485−2488.

Sharma, K., Garg, V.K., 2019. Chapter 10—Vermicomposting of Waste: A Zero-Waste Approach for Waste Management (pp. 133−164).

Stephen, R., Jose, S., Joseph, K., Thomas, S., Oommen, Z., 2006. Thermal stability and ageing properties of sulphur and gamma radiation vulcanized natural rubber (NR) and carboxylated styrene butadiene rubber (XSBR) lattices and their blends. Polym. Degrad. Stab. 91, 1717−1725.

Stevenson, K., Stallwood, B., Hart, A.G., 2008. Tire rubber recycling and bioremediation: a review. Bioremediation J. 15 (24), 37−41.

Swarna Smitha, S., Raghavendra, M.P., Shruthi, S., Girish, K., 2012. Bioremediation of rubber processing industry effluent by *Arthrobacter* sp. Int. J. Res. Environ. Sci. Technol. 2 (2), 31−34.

Tsuchii, A., Tokiwa, Y., 2006. Microbial degradation of the natural rubber in tire tread compound by a strain of Nocardia. J. Polym. Environ. 14, 403–409.

Tsuchii, A., Suzuki, T., Takeda, K., 1985. Microbial degradation of natural rubber vulcanizates. Appl. Environ. Microbiol. 50, 965–970.

Watanabe, K., 2001. Microorganisms relevant to bioremediation. Curr. Opin. Biotechnol. 12, 237–241.

Zabaniotou, A.A., Stavropoulos, G., 2003. Pyrolysis of used automobile tires and residual char utilization. J. Anal. Appl. Pyrolysis 70, 711–722.

This page intentionally left blank

Chapter 6

Composting of medicinal and aromatic plant waste: challenges and opportunities

Babett Greff[1], **Ágnes Nagy**[1], **Miklós Posgay**[2], **András Sáhó**[2,3] and **Erika Lakatos**[1]

[1]Department of Food Science, Albert Casimir Faculty at Mosonmagyaróvár, Széchenyi István University, Mosonmagyaróvár, Hungary, [2]Wittmann Antal Multidisciplinary Doctoral School in Plant, Animal, and Food Sciences, Széchenyi István University, Mosonmagyaróvár, Hungary, [3]Kisalföldi Agricultural Ltd, Nagyszentjános, Hungary

6.1 Introduction

Medicinal and aromatic plants have been commonly used since ancient times to improve the well-being of humans (Barata et al., 2016; Gonçalves et al., 2020). In the last few decades, a drastic increase has been observed on the market for these herbs and their metabolites because of their role in traditional, alternative, and complementary medicine (Arcury et al., 2007; Cadar et al., 2021; Fierascu et al., 2020; Pandey et al., 2019). The spread of COVID-19 has also had a substantial impact on public health, whereby traditional medicines and herbal remedies have drawn more attention (Zhang et al., 2021b). Based on these changes, the global market of herbal products and raw materials is projected to grow 5%−15% annually (Nilashi et al., 2020).

Along with the prosperous development of the herbal industry, significant amounts of solid biowaste have been generated, mostly derived from the extraction of active ingredients (Kong et al., 2019). Although the most common treatments of these herbal residues are incineration and landfilling (Tao et al., 2021), many efforts have been dedicated to the improvement of clean production strategies that can reduce waste issues (e.g. disposal and emission) and save valuable resources (Bejarano et al., 2020). Since the distillation waste of aromatic plants may retain its nutritional value, it can be recycled as compost/vermicompost to supplement crops with macronutrients and micronutrients (Verma et al., 2014). Nonetheless, the biodegradation of herbal biomass faces several disadvantages, including longer completion

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00003-7

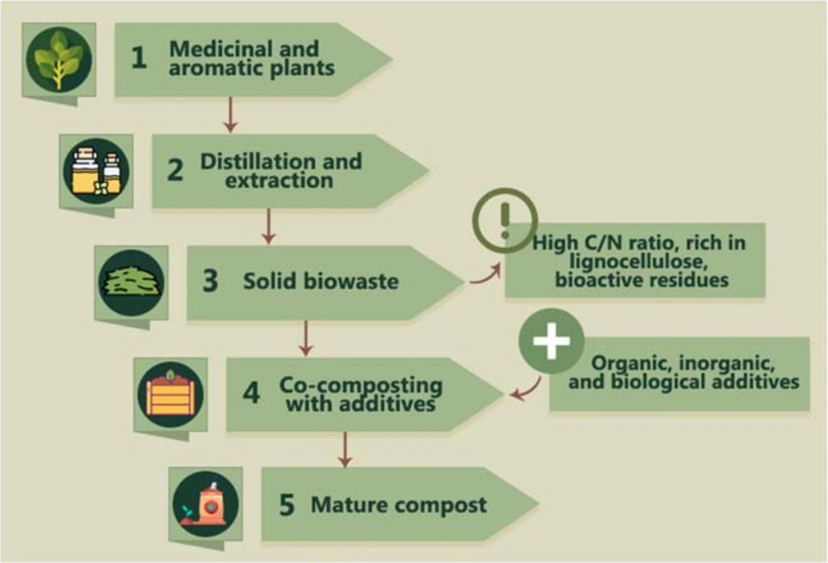


FIGURE 6.1 General methodology of co-composting medicinal and aromatic plant residues based on Greff et al. (2021a). The icons were made by Flat icons, Freepik, Kerismaker, Kliwir art, and Pixel Buddha (www.flaticon.com).

time and immature end product due to the lower nitrogen content and the presence of bioactive compounds (Fig. 6.1) (Lesage−Meessen et al., 2018; Teigiserova et al., 2021).

Nowadays, various biological methods are available (Huang et al., 2021) that can effectively expedite the degradation process of herbal waste and facilitate the maturation of the feedstock material. This chapter offers some important insight into the effective composting/vermicomposting methods of medicinal and aromatic plant residues and summarises the current literature on this topic.

6.2 Medicinal and aromatic plants: production, market, and important secondary metabolites

Medicinal and aromatic plants belong to a special class of plants that are primarily used for their aromatic and medicinal properties (Lubbe and Verpoorte, 2011; Saha and Basak, 2020). These herbs can synthesise an abundance of secondary metabolites (Cadar et al., 2021; Thakur et al., 2019) and thereby, they are widely used in the pharmaceutical, food, cosmetic, and pesticide industries (Celano et al., 2017; Elallem et al., 2021; Sánchez-Vioque et al., 2013) to produce, for instance, functional foods, herbal teas, new and traditional medicines, cosmetics, perfumes, cleaning products, antioxidants, preservatives, flavouring and colouring agents, herbicides, and insecticides

(Celano et al., 2017; Christaki et al., 2020; Fierascu et al., 2020; Fornari et al., 2012; Mahajan et al., 2020).

As a result of the continuously increasing demand for herbal products, the cultivation of these plants has become one of the fast-growing sectors of agriculture (Kozlowska et al., 2018) that has supplied almost two-thirds of the world's population, especially in the developing regions where the use of medicinal herbs is still a fundamental element of the traditional health system (Cheminal et al., 2020; Mahajan et al., 2020). According to the International Union for Conservation of Nature (IUCN) and the World and Wildlife Fund (WWF), over 50,000 plant species have been used for medicinal purposes, of which 1200–1300 medicinal and aromatic herbs are native to the European environment. Of these plants, 90% (20,000–30,000 tons) are collected from wild resources (Barata et al., 2016; Chen et al., 2016), while the remaining percentage of herbs is cultivated mainly in France, Poland, Spain, Bulgaria, and Germany (Table 6.1) (EIP-AGRI Focus Group, 2020).

Since these plant materials are usually traded as processed products or commodities on the markets (Bernard-Mongin et al., 2021), harvested medicinal and aromatic plants – except certain culinary herbs – need further industrial processing before their final use (Eurostat, 2020a). Specific attention is dedicated to the extraction of certain biochemical substances endowed with antimicrobial and antioxidant potential (Christaki et al., 2012). Although these phytochemicals have no significant role in the basic life processes of herbs, they can promote the adaptation of the plants to the environment and act as a defence system to protect them from phytopathogens, insects, pests, and herbivores (Thakur et al., 2019).

TABLE 6.1 Main cultivators of aromatic, medicinal, and culinary plants in the European Union (Eurostat, 2020b).

Country	Area (1000 ha)	Country	Area (1000 ha)
Poland	84.52	Czechia	5.66
France	61.52	Lithuana	4.17
Bulgaria	53.14	Austria	4.08
Spain	23.06	Hungary	3.96
Finland	15.00	Estonia	2.86
Greece	13.85	Italy	2.46
Germany	8.30	Romania	1.76
Croatia	7.24	Latvia	1.00
Slovakia	5.77	Portugal	1.00

Based on their biosynthetic pathways, secondary metabolites are divided into three major classes: phenolic compounds, terpenes, and nitrogen-containing compounds (Li et al., 2020). Essential oils are composed of a complex mixture of low molecular weight compounds. They can be applied for different purposes due to their aromatic, flavouring, and antimicrobial properties, but 90% of the global essential oil production is used by fragrance and flavour manufacturers. Currently, circa 3000 different essential oils are available worldwide, but only 300 of them are commonly traded (Lubbe and Verpoorte, 2011; Perricone et al., 2015). Nonetheless, the global market of plant products and essential oils is expected to reach more than 15.6 billion dollars by 2026 (Raveau et al., 2020).

Although several essential oil extraction methods have been developed previously and applied successfully, steam distillation and hydrodistillation are the most common techniques for purification (Lubbe and Verpoorte, 2011; Perricone et al., 2015; Rezaie-Tavirani et al., 2013). The quantity, quality, and chemical composition of volatiles derived from a single plant species may differ significantly (Radaelli et al., 2016) due to various factors, including climatic and growing conditions, genetic variation, phenological stages, and post-harvest processes (Farhadi et al., 2020; Raut and Karuppayil, 2014). Likewise, the annual production of essential oils varies considerably: certain extracted/distilled essential oils may exceed 32,000 tons (i.e. orange or corn mint oil), while others reach only a few kilograms (Schmidt, 2020).

In addition, aromatic plants are rich sources of other biologically active phytochemicals, such as phenolic compounds (Kumar et al., 2019; Mabrouki et al., 2018; Proestos and Varzakas, 2017). Polyphenols are a large group of 8000−10,000 individual molecules that have at least one aromatic ring with one or more hydroxyl groups attached (Mojzer et al., 2016; Pereira et al., 2016). In plants, they offer protection against various stresses, including solar radiation, plant pathogens, predators, parasites, oxidative damage, and cuts (Mojzer et al., 2016; Trivellini et al., 2016; Tzima et al., 2018). Along with terpenoids, they can be extracted from various plant parts (e.g., leaves, fruits, rhizomes, roots, seeds, flowers, etc.) (Kumar et al., 2019; Samarth et al., 2017). The global polyphenol consumption was 16,380 tons in 2015, but it is expected to reach 33,880 tons by the end of 2024 due to the continuously increasing usage of phenolic extracts in pharmaceuticals, foods, beverages, and cosmetics (Adebooye et al., 2018).

6.3 Medicinal and aromatic plant waste

Because of the growing public consciousness, strict environmental regulations, and shortage of landfills, solid waste disposal has become a serious problem (Kumari et al., 2011; Singh et al., 2013). During harvesting, processing, extraction, and distillation, enormous amounts of solid by-products (e.g. pruning materials, leftovers, defatted biomass, etc.) are generated (Deka et al., 2011a; Singh and Suthar, 2012a; Zaccardelli et al., 2021) that usually have no real industrial value (Turrini et al., 2021). For instance, essential oil extraction has resulted in 3 million tons of solid biomass in India only (Basak et al., 2021). Meanwhile, the amount of herbal waste in China has reached 35 million tons per annum (Tao et al., 2021). Furthermore, in European countries like Poland, some modern herbal companies may generate about 30 tons of green residues every year (Sienkiewicz et al., 2020).

Since there are no widespread techniques for the usage of herbal waste (Liu et al., 2021), the solid biomass has been discharged and treated by traditional methods like incineration, burying, stacking, used as fuel for distillation, or organic amendment for agricultural purposes that can contribute to the contamination of water, air, or soil and induce both ecological (leaching of nitrate and phytotoxic compounds, emission of greenhouse gases, eutrophication of surface water bodies, nutrient imbalances, odour pollution) and health concerns (dissemination of potential pathogens, leaching of toxic elements) (Basak et al., 2021; Chand et al., 2004; Lee et al., 2020; Lu and Li, 2021; Onursal and Ekinci, 2015; Rajan et al., 2019; Singh and Suthar, 2012b). Nevertheless, the exploitation of such by-products can promote the maintenance of a circular economy (Politi et al., 2020). Consequently, there is a need to find sustainable alternatives (Gupta et al., 2021; Slavov et al., 2017) that can decrease energy consumption, mitigate greenhouse gas emissions, and curb climate change (Evcan and Tari, 2015).

Nutrient recycling through the reuse of crop wastes is a simple but promising option, as it requires little investment or infrastructure (Chattopadhyay et al., 1993; Siles et al., 2016). Although herbal residues are often applied as organic amendments directly to the soil, the high biological oxygen demand, chemical oxygen demand, and the presence of polyphenols, volatile organic compounds, and heavy metals can create an environmental hazard (Lee et al., 2021; Liu et al., 2021; Sánchez-Vioque et al., 2013). Nonetheless, field and process residues, as a source of nutritional components, can be converted into industrially important products, such as organic fertilisers, adsorbents, biogas, animal feed, biofuel, and other bio-based chemicals (Das and Kumar, 2018; Kumla et al., 2020; Sadh et al., 2018; Tao et al., 2021). Among various technologies, composting is a cost-effective and environmentally friendly approach for the valorisation of this type of biowastes (Greff et al.; 2021a; Onursal and Ekinci, 2015).

6.4 Composting medicinal and aromatic plant waste: difficulties and feasible solutions

Aerobic composting is a controlled, thermophilic bio-decomposition process during which organic matter (OM) is degraded and transformed by microbes into a more stable humus-like product (Pergola et al., 2018; Sánchez et al., 2017; Wu et al., 2017b; Ye et al., 2019). The process itself can be divided into two stages: (1) a bio-oxidative stage where the feedstock material undergoes a rapid decomposition and (2) a maturation stage that is characterised by the stabilisation and humification of OM (Bernal et al., 2009). Although composting is dominated by aerobic microbes (Youngquist et al., 2016), their activity is highly dependent on certain factors, including moisture content, temperature, carbon-to-nitrogen (C/N) ratio, and the nature of the feedstock material (Azim et al., 2018). On the other hand, the traditional methods of vermicomposting operate under significantly different conditions (Singh and Suthar, 2012b). In a similar way to aerobic composting, vermitechnology is also controlled biodegradation, but this non-thermophilic process converts organic wastes into a stable product through the mutual interactions between microorganisms and earthworms (Chen et al., 2018; Deka et al., 2011a). In addition, the summarised findings of previous studies indicate that vermicomposting is often more effective, as earthworms may support the colonisation of aerobic microbes and the decomposition of the organic material by conditioning the substrate and converting it into casts (Romero et al., 2007; Singh and Suthar, 2012b; Verma et al., 2014).

In the past decades, several attempts were made to produce compost by using aromatic plant residues as sole feedstock (Table 6.2). For example, Dimitrijević et al. (2017) successfully composted mixed medicinal herbal waste containing Equiseti Herba, Valerianae Radix, Primulae Radix, Gentianae Radix, and the residues from herbal tea production and extraction process. Zaccardelli et al. (2020) prepared compost samples from raw or de-oiled residues of *Ocimum basilicum* L., *Rosmarinus officinalis* L., and *Salvia officinalis* L. (100%) to isolate potential biocontrol bacteria that can inhibit the growth of phytopathogenic fungi. In a study by Carrión-Paladines et al. (2016), the distilled residue of *Bursera graveolens* was used as feedstock. After 13 weeks, the finished vermicompost was free from phytotoxic compounds [germination index (GI): 103.6%]. Singh et al. (2012) produced a potent organic fertiliser from the distillation waste of lemongrass (*Cymbopogon flexuosus*) and citronella (*Cymbopogon winterianus*) using *Eisenia fetida*. Similarly, Deka et al. (2011b) vermicomposted citronella residues with *Eudrilus eugeniae*. By the end of their experiment, the resultant product had a low C/N ratio (7.3), while its composted counterpart was immature (final C/N ratio: 27.6). Meanwhile, Singh and Suthar (2012b) also confirmed that mixed herbal biomass is a suitable raw material for vermicomposting.

Nonetheless, the completion time and final compost quality depend not only on the activity of microbes and earthworms, but also on the composition of the feedstock. Single-source substrates are usually nutritionally imbalanced, and therefore, the produced organic fertilisers may not provide all essential properties for agricultural use (Carrión-Paladines et al., 2016; Isibika et al., 2021). Furthermore, the extraction efficiency of the bioactive components is relatively low (Chen et al., 2018) and the herbal material may contain

TABLE 6.2 Valorisation of medicinal and aromatic plants through composting and vermicomposting.

Plant biomass	Method	Source
Mint-residue from steam distillation	Composting	Chattopadhyay et al. (1993)
Mixed medicinal plant waste (Equiseti Herba, waste of Valerianae Radix, Primulae Radix, and Gentianae Radix, waste from herbal tea production, and mixed herb waste from extraction)	Composting	Dimitrijević et al. (2017)
Oil-free residues of *Ocimum basilicum* L., *Rosmarinus officinalis* L., or *Salvia officinalis* L./Raw residues of *O. basilicum* L., *R. officinalis* L., or *S. officinalis* L.	Composting	Zaccardelli et al. (2020)
Palo Santo (*Bursera graveolens*) distillation waste	Vermicomposting (*Eisenia fetida*)	Carrión-Paladines et al. (2016)
Java citronella (*Cymbopogon winterianus* Jowitt.) distillation waste	Vermicomposting (*Eudrilus eugeniae*)	Deka et al. (2011b)
Lemongrass (*Cymbopogon flexuosus*) and citronella (*C. winterianus*) distillation waste	Vermicomposting (*E. fetida*)	Singh et al. (2012)
Solid herbal pharmaceutical industrial waste (waste from extraction/distillation and unused parts of herbs)	Vermicomposting (*E. fetida*)	Singh and Suthar (2012b)

molecules with antimicrobial properties (Moisa et al., 2018; Vasileva et al., 2018) that can hinder the microbial decomposition of OM (Greff et al., 2021ab). For instance, Veličković et al. (2008) observed that the spent material of *S. officinalis* contained various components, including camphene, *n*-decane, 1,8-cineole, *trans*-decahydro naphthalene, *n*-undecane, camphor, *cis*-thujone, *trans*-thujone, carene-3-on, borneol, *n*-dodecane, 2,6-dimethyl undecane, *n*-tridecane, *n*-tetradecane, α-humulene, γ-muurolene, (*E*)-caryophyllene, ethyl hexadecanoate, epi-13-manool, *trans*-totarol, and viridiflorol. In a study by Lesage−Meessen et al. (2018), the cyclohexanic and ethyl acetate extracts of distilled lavender/lavandin flower straws and distilled lavender/lavandin stem straws were characterised by various terpene-derivatives (e.g., τ-cadinol, β-caryophyllene, etc.), phenolics (mainly derivatives of cinnamic and benzoic acids), and lactones (coumarin, herniarin). Similar results were obtained by Mohamed et al. (2018): *Coriandrum sativum* waste contained 17 major essential oil constituents, such as *trans*-anethole, linalool, estragole, longifolene, and carvacrol. After steam distillation, Moisa et al. (2018) determined the phenolic composition of deodorised *Origanum vulgare* var. *aereum*, *Thymus vulgaris* var. *Doone Valley,* and *Satureja hortensis*. The biomasses were rich in pyrogallol, pyrocatechol, rutin, quercetin, campherol, catechin, *p*-coumaric acid, caffeic acid, vanillic acid, gallic acid, and syringic acid. Drinić et al. (2020) found that the total phenolic content of *O. vulgare* L. spp. *hirtum* residual extracts obtained after hydrodistillation and microwave-assisted hydrodistillation ranged from 9.34 to 17.74 μg of gallic acid equivalents per gram of plant dry weight. Rosmarinic acid was identified as the most abundant phenolic compound in all extracts. Navarrete et al. (2011) extracted oil-free *R. officinalis* plant material with ethanol. The residue from the solvent-free microwave extraction contained rosmarinic acid, carnosol, carnosic acid, caffeic acid, chlorogenic acid, and *p*-coumaric acid. Chávez-González et al. (2016) found that orange (*Citrus sinensis* cv. valencia), lime (*Citrus limonium* cv. colima), and grapefruit (*Citrus paradisi* cv. doble rojo) peel wastes were characterised by higher levels of limonene (78.7 mass%–89.1 mass%). Similar results were obtained by Wu et al. (2017a).

Although essential oils and phenolic compounds can be metabolised by special microorganisms that have the enzymatic capacity to break down these organic compounds in the feedstock (Filannino et al., 2015; Hassiotis and Lazari, 2010), the composting process of medicinal and aromatic plant residues faces several disadvantages (Lesage–Meessen et al., 2018; Teigiserova et al., 2021) as the presence of bioactive substances incorporated into the composting material may decrease the efficiency of the biodegradation process by affecting the microbiological parameters (Cano et al., 2016). For instance, citrus by-product, as sole feedstock material, cannot be considered for composting due to its essential oil and phenol contents, low pH, and high protein/carbohydrate ratio (1:14) (Isibika et al., 2021; Siles et al., 2016). Likewise, the addition of a larger amount of peel waste to the feedstock (40%–50%) can inhibit the composting process (Toledo et al., 2018).

In such a situation, the supplementation of organic materials rich in nitrogen (co-composting and co-vermicomposting), inorganic amendments, microbial inoculants, or the combination of these techniques may minimise the drawbacks of traditional composting/vermicomposting and promote the formation of stable and nutrient-rich products (Table 6.3).

During composting, Sharma et al. (2018) optimised the combinations of floral waste containing mostly rose, marigold, and lotus, cow manure, and sawdust using response surface methodology. The ideal mixture of these materials (65 kg of flower waste, 25 kg of cattle manure, and 10 kg of sawdust) had low total organic carbon (TOC) content, reduced final C/N ratio, and increased GI% after the completion of 30 days. The results also suggested that the biodegradable content and higher microbial activity in cattle manure facilitated the assimilation of resistant organic compounds in the biomass. Dimitrijević et al. (2017) used a mesophilic starter culture to promote the

TABLE 6.3 Improving the composting/vermicomposting process of medicinal and aromatic plant residues by various additives.

Plant biomass	Method	Source
Mixed medicinal plant waste (Equiseti Herba, waste of Valerianae Radix, Primulae Radix, and Gentianae Radix, waste of herbal tea production, and mixed herb waste from extraction)	Composting with the addition of microbiological amendment (*Bacillus, Hymenobacter, Paenybacillus,* and *Streptomyces* spp.)	Dimitrijević et al. (2017)
Mint-residue from steam distillation	Composting with the addition of urea and diammonium phosphate in combination with microbiological amendment (*Trichoderma viride*) or soil suspension	Chattopadhyay et al. (1993)
Flower waste containing mainly marigold, lotus and rose	Co-composting with cow manure and sawdust	Sharma et al. (2018)
Mixed herbal waste from extraction	Co-composting with buffalo manure or vegetable waste/ Composting with the addition of a blend of enzymes, probiotics, and organic catalyst inocula	Ali et al. (2012)
Orange peel waste	Co-composting with organic fraction of municipal solid waste	Toledo et al. (2018)
Orange peel waste	Co-composting with organic fraction of municipal solid waste	Siles et al. (2016)
Lavender (*Lavandula angustifolia* Mill.) distillation waste	Co-composting with cow manure, barley star, and microbiological amendment (*Streptomyces viridosporus* and *Cellulomonas flavigena*)	Greff et al. (2021b)
Oil-free residues of *Ocimum basilicum* L., *Rosmarinus officinalis* L., or *Salvia officinalis* L.	Composting with the addition of 2-year-old commercial urban waste compost	Zaccardelli et al. (2021)
Java citronella (*Cymbopogon winterianus* Jowitt.) distillation waste	Co-composting with cow manure	Deka et al. (2011a)

(*Continued*)

TABLE 6.3 (Continued)

Plant biomass	Method	Source
Mixed medicinal herbal residues (40% *Artemisia aplacea*, 20% Honeysuckle, 20% Tuckahoe, Cassia twig, Moutan bark, and Peach kernel, 20% Rhizoma Corydalis, *Ligusticum chuanxiong* Hort., Radix Angelicae Pubescentis, and *Fritillaria ussuriensis*)	Composting with the addition of urea and microbiological amendment (*Bacillus subtilis, Aspergillus niger, Saccharomyces cerevisiae, Myceliophthora thermophila, Streptomyces pratensis, Streptomyces violascens*; single-phase and multi-phase inoculation)	Lu et al. (2021)
Patchouli (*Pogostemon cablin* Benth.) distillation waste	Composting pre-treated (*Trichoderma harzianum, Pseudomonas monteilii, Bacillus megaterium,* and *Azotobacter chroococcum* alone or in combination) distillation waste	Singh et al. (2013)
Solid herbal pharmaceutical industrial waste (waste from extraction/distillation and unused parts of herbs)	Vermicomposting (*Eisenia fetida*) with cow manure	Singh and Suthar (2012b)
Palo Santo (*Bursera graveolens*) distillation waste	Vermicomposting (*E. fetida*) with pig manure/ kitchen leftovers/goat manure and King Grass residue	Carrión-Paladines et al. (2016)
Java citronella (*C. winterianus* Jowitt.) distillation waste	Vermicomposting (*Perionyx excavatus*) with cow manure	Deka et al. (2011a)
Java citronella (*C. winterianus* Jowitt.) distillation waste	Vermicomposting (*Eudrilus eugeniae*) with cow manure	Deka et al. (2011b)
Patchouli (*Pogostemon cablin* Benth.) distillation waste	Vermicomposting (*E. fetida*) pre-treated (*T. harzianum, P. monteilii, B. megaterium,* and *A. chroococcum* alone or in combination) distillation waste	Singh et al. (2013)
Industrial herbal waste (*Punica granatum, Symplocos racemosa, Andrographis paniculata, Woodfordia fruticosa, Salmalia malabarica, Berberis aristata,* and *Aegle marmelos*)	Vermicomposting (*E. eugeniae*) with cow manure	Kumari et al. (2011)

degradation of mixed medicinal plant waste. The addition of *Bacillus*, *Hymenobacter*, *Paenybacillus,* and *Streptomyces* strains shortened the composting period and significantly improved the quality of the final product (GI%: 106%−190%). Singh and Suthar (2012b) vermicomposted cattle manure and solid herbal waste with *E. fetida*. Compared to the control compost (100% of herbal pharmaceutical industrial waste), the addition of cow dung (25%) increased the earthworm total biomass, total earthworm population, and total cocoon production and decreased the final carbon-to-phosphate (C/P) ratio of the herbal vermicompost. Chattopadhyay et al. (1993) found that the application of urea and diammonium phosphate combined with *Trichoderma viride* or soil suspension reduced the final C/N and C/P ratios of mint-residue compost. After 90 days, the end products were rich in plant nutrients and humus. Ali et al. (2012) revealed that drum-composting (one rotation per day) herbal residue with buffalo manure or vegetable waste could provide favourable composting conditions. Similarly, Carrión-Paladines et al. (2016) showed that the waste from *B. graveolens* fruit distillation is an excellent feedstock material for vermicomposting with *E. fetida*. However, the final product prepared from pure herbal waste was characterised by high OM, total C (TC: 40.8%), acid detergent lignin, total extractable polyphenol contents, and C/N ratio (35.0), while the total N content (TN: 1.16%) was reduced. On the other hand, the addition of pig manure improved the vermicompost quality significantly (e.g., TC: 35.0%; TN: 2.63%, etc.). Toledo et al. (2018) co-composted the organic fraction of municipal solid waste (OFMSW) mixed with 10% of orange peel waste (OPW). The lower dynamic respirometric index values indicated that the OPW in OFMSW-OPW hindered the microbial activity compared to the control OFMSW compost. Nonetheless, Siles et al. (2016) pointed out that co-composting OPW with OFMSW can be a feasible way to use these materials simultaneously. The mixture composed of 17% of peel waste and 83% of OFMSW resulted in an ideal compost that was characterised by an acceptable oxidable organic carbon-to-total Kjeldahl nitrogen (TKN) ratio (22.3) and reduced odour generation. Greff et al. (2021b) co-composted extracted lavender (*Lavandula angustifolia* Mill.) waste with cattle manure and barley straw. Although the addition of aromatic plant residues extended the thermophilic stage and accelerated the degradation of OM, adverse effects were also observed (lower GI%, higher C/N ratio). Inoculation with *Streptomyces viridosporus* and *Cellulomonas flavigena* elevated the number of beneficial microorganisms in the feedstock material, and consequently, improved the efficiency of the composting process. Compared to the compost containing lavender and cattle manure only, the quality of the bioaugmented compost was significantly better regarding OM content (76.2%), TN content (2.5%), C/N ratio (16.9), and Klason lignin content (12.7). To facilitate the composting process of oil-free basil, rosemary, and sage biomass, Zaccardelli et al. (2021) used 2% of a 2-year-old commercial urban-waste compost as a starter. After 4 months of composting, the products were stable and mature.

Deka et al. (2011a) used citronella (*C. winterianus* Jowitt.) residues and cow manure for co-composting and co-vermicomposting. The application of *Perionyx excavatus* improved the ash content (795.5 g kg^{-1} on a dry weight basis), TOC content (134.7 g kg^{-1} on a dry weight basis), C/N ratio (7.4), and total nutrient content (TKN, total K, Ca, Mg, and available P) of the finished vermicompost. Lu et al. (2021) used single and multi-stage inoculation techniques to facilitate the composting process of mixed medicinal herbal residues supplemented with urea. During the composting process, the multi-stage treatment with *Bacillus subtilis*, *Aspergillus niger*, *Saccharomyces cerevisiae*, *Myceliophthora thermophila*, *Streptomyces pratensis,* and *Streptomyces violascens* elevated the peak temperature during the thermophilic stage, improved the enzyme production and humification process, accelerated the degradation of lignin and cellulose, and influenced the fungal community composition. The finished product was free from phytotoxic compounds (GI: 102.7%). Singh et al. (2013) found that pre-treating patchouli distillation waste with bioinoculant comprised of *Trichoderma harzianum*, *Pseudomonas monteilii*, *Bacillus megaterium*, and *Azotobacter chroococcum* facilitated the decomposition of lignocellulosic residues in terms of cellulose, hemicellulose, and lignin degradation. Moreover, the addition of beneficial microorganisms expedited the degradation of TOC (final TOC: 17.8%) and significantly increased the TKN (2.3%), TP (0.3%), and TK (0.7%) contents of the mature vermicompost. Kumari et al. (2011) vermicomposted industrial herbal waste and cow dung (1:1) with *E. eugeniae* for about 2 months. This mixture resulted in a nutrient-rich vermicompost that was characterised by significantly improved TKN (26.3 g kg^{-1}), TOC (180 g kg^{-1}), and C/N ratio (6.8).

In conclusion, the application of co-composting/co-vermicomposting technologies and the use of microbial amendments can improve the degradation processes of medicinal and aromatic plant residues by providing nutrients, easily degradable OM, and beneficial microorganisms.

6.5 The applicability of medicinal and aromatic plant compost as soil amendment

Agricultural soils are often more susceptible to desertification, erosion, and climate change due to their low OM content (Epelde et al., 2018). The addition of organic amendments such as compost can improve the biological, physical, and chemical properties of soil, sustain crop productivity, and thereby, provide a healthy environment (Debernardi-Vázquez et al., 2020; Verma et al., 2014). Herbal residues serve as an important source of nutrients (e.g. nitrogen, potassium, phosphorus, etc.) (Ma et al., 2021; Wang et al., 2021), while the compost can affect soil processes, features, and productivity by increasing nutrient and water retention, improving the organic carbon content, and providing essential nutrients for plant growth (Balaguer et al., 2015; Kadoglidou et al., 2014; Zhang et al., 2021a). Although they can partially replace chemical fertilisers in the agro-industry (Tao et al., 2021), Zaccardelli et al. (2021) highlighted that the effectiveness of these composts is dose-dependent, and in a higher concentration, they may have a phytotoxic effect on crops. Furthermore, Chalkos et al. (2010) reported that the addition of *Salvia fruticosa* Mill. or *Mentha spicata* L. composts could affect soil productivity. The combined application of aromatic plant composts with organic or synthetic fertilisers increased the abundance of soil microorganisms and stimulated the growth of tomato plants in most treatments. Meanwhile, the emergence of weeds was reduced, but only in the soils supplemented with spearmint compost. Verma et al. (2014) combined inorganic fertilisers with vermicompost derived from geranium distillation waste to improve the growth of *Pelargonium graveolens* L. crops. The greatest growth (plant canopy and leaf-to-stem ratio) was achieved by the addition of vermicompost and $N_{100}P_{60}K_{60}$ inorganic fertiliser. Furthermore, the treatment significantly increased the essential oil yield of geranium (62.6 kg ha^{-1}) compared to the control (23.3 kg ha^{-1}). Their results suggested that this method could also improve soil health (e.g., available N, P, and K). To decrease the damage caused by *Meloidogyne incognita*, Pandey (2005) mixed vermicomposts prepared from distillation wastes to the soil of *Artemisia pallens* Wall. The use of *Mentha arvensis* and *C. winterianus* amendments enhanced the growth development and oil yield of davana plants, while the nematode proliferation was inhibited. Under field conditions, Singh et al. (2013) found that enriched patchouli vermicompost enhanced the growth (plant height, plant spread, and the number of branches) and essential oil yield of *Pogostemon cablin* Benth. var. “Johore,” improved the physicochemical properties of rhizospheric soil (e.g., bulk density, water holding capacity, available N, P, and K contents, etc.), and reduced the disease severity of *Rhizoctonia* root-wilt more effectively than most of the other treatments.

Over the years, several indigenous microorganisms have been isolated from composts that can promote the suppression of soil-borne diseases. El-Helw et al. (2018) showed that thermophilic microorganisms from aromatic plant compost had an antagonistic effect on the growth of certain Gram-positive bacteria (*Staphylococcus aureus*, *Enterococcus faecalis,* and *Mycobacterium phlei*) by excreting bioactive compounds. In another study by Zaccardelli et al. (2020), biocontrol agents related to *Bacillus* species were isolated from aromatic plant waste composts. The most effective bacteria showed suppression against *Sclerotinia*-damping off on rocket salad (*Diplotaxis tenuifolia* L.), but they had no biocontrol activity in the *Rhizoctonia solani*/rocket pathosystem.

All things considered, the application of composts derived from medicinal and aromatic plant residues can improve the physical, chemical, and biological properties of agricultural soil by providing important nutrients, bioactive compounds, and antagonistic microorganisms.

References

Adebooye, O.C., Alashi, A.M., Aluko, R.E., 2018. A brief review on emerging trends in global polyphenol research. J. Food Biochem. 42 (4), e12519. Available from: https://doi.org/10.1111/jfbc.12519.

Ali, M., Duba, K.S., Kalamdhad, A.S., Bhatia, A., Khursheed, A., Kazmi, A.A., et al., 2012. High rate composting of herbal pharmaceutical industry solid waste. Water Sci. Technol. 65 (10), 1817−1825. Available from: https://doi.org/10.2166/wst.2012.082.

Arcury, T.A., Grzywacz, J.G., Bell, R.A., Neiberg, R.H., Lang, W., Quandt, S.A., 2007. Herbal remedy use as health self-management among older adults. J. Gerontol. Ser. B 62 (2), 142−149. Available from: https://doi.org/10.1093/geronb/62.2.S142.

Azim, K., Soudi, B., Boukhari, S., Perissol, C., Roussos, S., Alami, I.T., 2018. Composting parameters and compost quality: a literature review. Org. Agric. 8, 141−158. Available from: https://doi.org/10.1007/s13165-017-0180-z.

Balaguer, M.P., Villanova, J., Cesar, G., Gavara, R., Hernandez-Munoz, P., 2015. Compostable properties of antimicrobial bioplastics basedon cinnamaldehyde cross-linked gliadins. Chem. Eng. J. 262, 447−455. Available from: https://doi.org/10.1016/j.cej.2014.09.099.

Barata, A.M., Rocha, F., Lopes, V., Carvalho, A.M., 2016. Conservation and sustainable uses of medicinal and aromatic plantsgenetic resources on the worldwide for human welfare. Ind. Crop. Prod. 88, 8−11. Available from: https://doi.org/10.1016/j.indcrop.2016.02.035.

Basak, B.B., Saha, A., Sarkar, B., Kumar, B.P., Gajbhiye, N.A., Banerjee, A., 2021. Repurposing distillation waste biomass and low-value mineral resources through biochar-mineral-complex for sustainable production of high-value medicinal plants and soil quality improvement. Sci. Total. Environ. 760, 143319. Available from: https://doi.org/10.1016/j.scitotenv.2020.143319.

Bejarano, J.S.R., Rodrigues, T.S., Sánchez, C.M., Al-Ghanim, K., Al-Saidi, M., 2020. Promoting sustainable businesses for strong local communities: Qatar's wild herbal plants industry. Energy Rep. 6 (8), 80−86. Available from: https://doi.org/10.1016/j.egyr.2020.10.058.

Bernal, M.P., Alburquerque, J.A., Moral, R., 2009. Composting of animal manures and chemical criteria for compost maturity assessment. A review. Bioresour. Technol. 100 (22), 5444−5453. Available from: https://doi.org/10.1016/j.biortech.2008.11.027.

Bernard-Mongin, C., Hoxha, V., Lerin, F., 2021. From total state to anarchic market: management of medicinal and aromatic plants in Albania. Reg. Environ. Change 21, 5. Available from: https://doi.org/10.1007/s10113-020-01727-1.

Cadar, R.L., Amuza, A., Dumitras, D.E., Mihai, M., Pocol, C.B., 2021. Analysing clusters of consumers who use medicinal and aromatic plant products. Sustainability. 13 (15), 8648. Available from: https://doi.org/10.3390/su13158648.

Cano, A.I., Cháfer, M., Chiralt, A., González-Martínez, C., 2016. Biodegradation behavior of starch-PVA films as affected by the incorporation of different antimicrobials. Polym. Degrad. Stab. 132, 11−20. Available from: https://doi.org/10.1016/j.polymdegradstab.2016.04.014.

Carrión-Paladines, V., Fries, A., Gómez-Muñoz, B., García-Ruiz, R., 2016. Agrochemical characterization of vermicomposts produced from residues of Palo Santo (*Bursera graveolens*) essential oil extraction. Waste Manag. 58, 135−143. Available from: https://doi.org/10.1016/j.wasman.2016.09.002.

Celano, R., Piccinelli, A.L., Pagano, I., Roscigno, G., Campone, L., De Falco, E., et al., 2017. Oil distillation wastewaters from aromatic herbs as new natural source of antioxidant compounds. Food Res. Int. 99 (Part 1), 298−307. Available from: https://doi.org/10.1016/j.foodres. 2017.05.036.

Chalkos, D., Kadoglidou, K., Karamanoli, K., Fotiou, C., Pavlatou-Ve, A.S., Eleftherohorinos, I. G., et al., 2010. *Mentha spicata* and *Salvia fruticosa* composts as soil amendments in tomato cultivation. Plant. Soil. 332, 495–509. Available from: https://doi.org/10.1007/s11104-010-0315-4.

Chand, S., Anwar, M., Patra, D.D., Khanuja, S.P.S., 2004. Effect of mint distillation waste on soil microbial biomass in a mint-mustard cropping sequence. Commun. Soil. Sci. Plant. Anal. 35 (1–2), 243–254. Available from: https://doi.org/10.1081/CSS-120027647.

Chattopadhyay, A., Subrahmanyam, K., Singh, D.V., 1993. Recycling of nutrients in Japanese mint - assessment of soil fertility and crop yield. Fertil. Res. 35, 177–181. Available from: https://doi.org/10.1007/BF00750636.

Chávez-González, M.L., López-López, L.I., Rodríguez-Herrera, R., Contreras-Esquivel, J.C., Aguilar, C.N., 2016. Enzyme-assisted extraction of citrus essential oil. Chem. Pap. 70 (4), 412–417. Available from: https://doi.org/10.1515/chempap-2015-0234.

Cheminal, A., Kokkoris, I.P., Strid, A., Dimopoulos, P., 2020. Medicinal and aromatic *Lamiaceae* plants in Greece: linking diversity and distribution patterns with ecosystem services. Forests 11 (6), 661. Available from: https://doi.org/10.3390/f11060661.

Chen, S.L., Yu, H., Luo, H.M., Wu, Q., Li, C.F., Steinmetz, A., 2016. Conservation and sustainable use of medicinal plants: problems, progress, and prospects. Chin. Med. 11, 37. Available from: https://doi.org/10.1186/s13020-016-0108-7.

Chen, Y.X., Chang, S.K.C., Chen, J., Zhang, Q., Yu, H.Y., 2018. Characterization of microbial community succession during vermicomposting of medicinal herbal residues. Bioresour. Technol. 249, 542–549. Available from: https://doi.org/10.1016/j.biortech.2017.10.021.

Christaki, E., Bonos, E., Giannenasm, I., Florou-Paneri, P., 2012. Aromatic plants as a source of bioactive compounds. Agriculture. 2 (3), 228–243. Available from: https://doi.org/10.3390/agriculture2030228.

Christaki, E., Giannenas, I., Bonos, E., Florou-Paneri, P., 2020. Innovative uses of aromatic plants as natural supplements in nutrition. In: Florou-Paneri, P., Christaki, E., Giannenas, I. (Eds.), Feed Additives: Aromatic Plants and Herbs in Animal Nutrition and Health. Academic Press, Elsevier Science Publishing Co Inc., USA, pp. 19–34. Available from: https://doi.org/10.1016/B978-0-12-814700-9.00002-9.

Das, A.J., Kumar, R., 2018. Utilization of agro-industrial waste for biosurfactant production under submerged fermentation and its application in oil recovery from sand matrix. Bioresour. Technol. 260, 233–240. Available from: https://doi.org/10.1016/j.biortech.2018.03.093.

Debernardi-Vázquez, T.D., Aguilar-Rivera, N., Núñez-Pastrana, R., 2020. Composting of byproducts from the orange (*Citrus sinensis* (L.) Osbeck) and sugarcane (*Saccharum* spp. hybrids) agroindustries. Ingeníeria e Investigación 40 (3), 81–88. Available from: https://doi.org/10.15446/ing.investig.v40n3.82877.

Deka, H., Deka, S., Baruah, C.K., Das, J., Hoque, S., Sarma, H., et al., 2011a. Vermicomposting potentiality of *Perionyx excavatus* for recycling of waste biomass of java citronella - an aromatic oil yielding plant. Bioresour. Technol. 102 (24), 11212–11217. Available from: https://doi.org/10.1016/j.biortech.2011.09.102.

Deka, H., Deka, S., Baruah, C.K., Das, J., Hoque, S., Sarma, N.S., 2011b. Vermicomposting of distillation waste of citronella plant (*Cymbopogon winterianus* Jowitt.) employing *Eudrilus eugeniae*. Bioresour. Technol. 102 (13), 6944–6950. Available from: https://doi.org/10.1016/j.biortech.2011.04.027.

Dimitrijević, S.M., Radanović, D.S., Antić-Mladenović, S.B., Milutinović, M.D., Rajilić-Stojanović, M.D., Dimitrijević-Branković, S.I., 2017. Enhanced fertilization effect of a compost obtained from mixed herbs waste inoculated with novel strains of mesophilic bacteria. Hemijska Industrija 71 (6), 503–513. Available from: https://doi.org/10.2298/HEMIND170327013D.

Drinić, Z., Pljevljakušić, D., Živković, J., Bigović, D., Šavikin, K., 2020. Microwave–assisted extraction of *O. vulgare* L. spp. *hirtum* essential oil: comparison with conventional hydro–distillation. Food Bioprod. Process. 120, 158–165. Available from: https://doi.org/10.1016/j.fbp.2020.01.011.

EIP-AGRI Focus Group, 2020. Plant-based medicinal and cosmetic products: Final report. European Commission [WWW Document]. URL https://ec.europa.eu/eip/agriculture/en/publications/eip-agri-focus-group-plant-based-medicinal-and (accessed 03.12.21).

Elallem, K.A., Sobeh, M., Boularbah, A., Yasri, A., 2021. Chemically degraded soil rehabilitation process using medicinal and aromatic plants: review. Environ. Sci. Pollut. Res. 28, 73–93. Available from: https://doi.org/10.1007/s11356-020-10742-y.

El-Helw, N.O., El-Gendy, A.O., El-Gebaly, E., Hassan, H.M., Rateb, M.E., El-Nesr, K.A., 2018. Characterization of natural bioactive compounds produced by isolated bacteria from compost of aromatic plants. J. Appl. Microbiol. 126 (2), 443–451. Available from: https://doi.org/10.1111/jam.14085.

Epelde, L., Jauregi, L., Urra, J., Ibarretxe, L., Romo, J., Goikoetxea, I., et al., 2018. Characterization of composted organic amendments for agricultural use. Front. Sustain. Food Syst. 2, 44. Available from: https://doi.org/10.3389/fsufs.2018.00044.

Eurostat, 2020a. Annual crop statistics handbook [WWW Document]. URL https://ec.europa.eu/eurostat/cache/metadata/Annexes/apro_cp_esms_an1.pdf (accessed 08.05.21).

Eurostat, 2020b. Crop production in EU standard humidity [WWW Document]. URL http://apps-so.eurostat.ec.europa.eu/nui/submitViewTableAction.do (accessed 06.02.23).

Evcan, E., Tari, C., 2015. Production of bioethanol from apple pomace by using cocultures: conversion of agro-industrial waste to value added product. Energy. 88, 775–782. Available from: https://doi.org/10.1016/j.energy.2015.05.090.

Farhadi, N., Babaei, K., Farsaraei, S., Moghaddam, M., Pirbalouti, A.G., 2020. Changes in essential oil compositions, total phenol, flavonoids and antioxidant capacity of *Achillea millefolium* at different growth stages. Ind. Crop. Prod. 152, 112570. Available from: https://doi.org/10.1016/j.indcrop.2020.112570.

Fierascu, R.C., Fierascu, I., Ortan, A., Georgiev, M.I., Sieniawska, E., 2020. Innovative approaches for recovery of phytoconstituents from medicinal/aromatic plants and biotechnological production. Molecules. 25 (2), 309. Available from: https://doi.org/10.3390/molecules25020309.

Filannino, P., Bai, Y.P., Di Cagno, R., Gobbetti, M., Gänzle, M.G., 2015. Metabolism of phenolic compounds by *Lactobacillus* spp. during fermentation of cherry juice and broccoli puree. Food Microbiol. 46, 272–279. Available from: https://doi.org/10.1016/j.fm.2014.08.018.

Fornari, T., Vicente, G., Vázquez, E., García-Risco, M.R., Reglero, G., 2012. Isolation of essential oil from different plants and herbs by supercritical fluid extraction. J. Chromatogr. A 1250, 34–48. Available from: https://doi.org/10.1016/j.chroma.2012.04.051.

Gonçalves, S., Mansinhos, I., Romano, A., 2020. Aromatic plants: a source of compounds with antioxidant and neuroprotective effects. In: Martin, C.R., Preedy, V.R. (Eds.), Oxidative Stress and Dietary Antioxidants in Neurological Diseases. Academic Press, Elsevier Science Publishing Co Inc., USA, pp. 155–173. Available from: https://doi.org/10.1016/B978-0-12-817780-8.00011-6.

Greff, B., Lakatos, E., Szigeti, J., Varga, L., 2021a. Co-composting with herbal wastes: potential effects of essential oil residues on microbial pathogens during composting. Crit. Rev.

Environ. Sci. Technol. 51 (5), 457–511. Available from: https://doi.org/10.1080/10643389.2020.1732780.

Greff, B., Szigeti, J., Varga, Á., Lakatos, E., Sáhó, A., Varga, L., 2021b. Effect of bacterial inoculation on co-composting of lavender (*Lavandula angustifolia* Mill.) waste and cattle manure. 3 Biotech. 11, 306. Available from: https://doi.org/10.1007/s13205-021-02860-2.

Gupta, A.K., Tomar, J.M.S., Kaushal, R., Kadam, D.M., Rathore, A.C., Mehta, H., et al., 2021. Aromatic plants based environmental sustainability with special reference to degraded land management. J. Appl. Res. Med. Aromat. Plants 22, 100298. Available from: https://doi.org/10.1016/j.jarmap.2021.100298.

Hassiotis, C.N., Lazari, D.M., 2010. Decomposition process in the Mediterranean region. Chemical compounds and essential oil degradation from *Myrtus communis*. Int. Biodeterior. Biodegrad. 64 (5), 356–362. Available from: https://doi.org/10.1016/j.ibiod.2010.02.007.

Huang, C., Li, Z.X., Wu, Y., Huang, Z.Y., Hu, Y., Gao, J., 2021. Treatment and bioresources utilization of traditional Chinese medicinal herb residues: recent technological advances and industrial prospect. J. Environ. Manag. 299, 113607. Available from: https://doi.org/10.1016/j.jenvman.2021.113607.

Isibika, A., Vinnerås, B., Kibazohi, O., Zurbrügg, C., Lalander, C., 2021. Co-composting of banana peel and orange peel waste with fish waste to improve conversion by black soldier fly (*Hermetia illucens* (L.), Diptera: Stratiomyidae) larvae. J. Clean. Prod. 318, 128570. Available from: https://doi.org/10.1016/j.jclepro.2021.128570.

Kadoglidou, K., Chalkos, D., Karamanoli, K., Eleftherohorinos, I.G., Constantinidou, H.I.A., Vokou, D., 2014. Aromatic plants as soil amendments: effects of spearmint and sage on soil properties, growth and physiology of tomato seedlings. Sci. Hortic 179, 25–35. Available from: https://doi.org/10.1016/j.scienta.2014.09.009.

Kong, W.P., Huang, C.S., Shi, J., Li, Y., Jiang, X.X., Duan, Q.W., et al., 2019. Recycling of Chinese herb residues by endophytic and probiotic fungus *Aspergillus cristatus* CB10002 for the production of medicinal valuable anthraquinones. Microb. Cell Fact. 18, 102. Available from: https://doi.org/10.1186/s12934-019-1150-9.

Kozlowska, W., Wagner, C., Moore, E.M., Matkowski, A., Komarnytsky, S., 2018. Botanical provenance of traditional medicines from Carpathian Mountains at the Ukrainian-Polish border. Front. Pharmacol. 9, 295. Available from: https://doi.org/10.3389/fphar.2018.00295.

Kumar, V., Sharma, A., Kohli, S.K., Bali, S., Sharma, M., Kumar, R., et al., 2019. Differential distribution of polyphenols in plants using multivariate techniques. Biotechnol. Res. Innov. 3 (1), 1–21. Available from: https://doi.org/10.1016/j.biori.2019.03.001.

Kumari, M., Kumar, S., Chauhan, R.S., Ravikanth, K., 2011. Bioconversion of herbal industry waste into vermicompost using an epigeic earthworm *Eudrilus eugeniae*. Waste Manag. Res. 29 (11), 1205–1212. Available from: https://doi.org/10.1177/0734242X10384432.

Kumla, J., Suwannarach, N., Sujarit, K., Penkhrue, W., Kakumyan, P., Jatuwong, K., et al., 2020. Cultivation of mushrooms and their lignocellulolytic enzyme production through the utilization of agro-industrial waste. Molecules 25 (12), 2811. Available from: https://doi.org/10.3390/molecules25122811.

Lee, Y., Kim, S., Kim, J., Shin, G.A., Lee, C.G., Jung, S., et al., 2020. Catalytic pyrolysis as a technology to dispose of herbal medicine waste. Catalysts 10 (8), 826. Available from: https://doi.org/10.3390/catal10080826.

Lee, Y., Kim, S., Lee, J., 2021. Co-pyrolysis for the valorization of food waste and oriental herbal medicine byproduct. J. Anal. Appl. Pyrolysis 154, 105016. Available from: https://doi.org/10.1016/j.jaap.2021.105016.

Lesage−Meessen, L., Bou, M., Ginies, C., Chevret, D., Navarro, D., Drula, E., et al., 2018. Lavender− and lavandin−distilled straws: an untapped feedstock with great potential for the production of high−added value compounds and fungal enzymes. Biotechnol. Biofuels 11, 217. Available from: https://doi.org/10.1186/s13068-018-1218-5.

Li, Y.Q., Kong, D.X., Fu, Y., Sussman, M.R., Wu, H., 2020. The effect of developmental and environmental factors on secondary metabolites in medicinal plants. Plant. Physiol. Biochem. 148, 80−89. Available from: https://doi.org/10.1016/j.plaphy.2020.01.006.

Liu, F., Wu, X., Zhang, X., Yang, L., Liu, Y., Song, Z.C., 2021. Co-use of organic herbal residue and red mud waste for syngas production by chemical looping gasification. Int. J. Energy Res. 45 (2), 2195−2210. Available from: https://doi.org/10.1002/er.5912.

Lu, Q., Li, C.L., 2021. Comprehensive utilization of Chinese medicine residues for industry and environment protection: turning waste into treasure. J. Clean. Prod. 279, 123856. Available from: https://doi.org/10.1016/j.jclepro.2020.123856.

Lu, X.L., Wu, H., Song, S.L., Bai, H.Y., Tang, M.J., Xu, F.J., et al., 2021. Effects of multiphase inoculation on the fungal community related with the improvement of medicinal herbal residues composting. Environ. Sci. Pollut. Res. 28, 27998−28013. Available from: https://doi.org/10.1007/s11356-021-12569-7.

Lubbe, A., Verpoorte, R., 2011. Cultivation of medicinal and aromatic plants for specialty industrial materials. Ind. Crop. Prod 34 (1), 785−801. Available from: https://doi.org/10.1016/j.indcrop.2011.01.019.

Ma, J.F., Chen, Y.P., Wang, K.B., Huang, Y.Z., Wang, H.J., 2021. Re-utilization of Chinese medicinal herbal residues improved soil fertility and maintained maize yield under chemical fertilizer reduction. Chemosphere 283, 131262. Available from: https://doi.org/10.1016/j.chemosphere.2021.131262.

Mabrouki, H., Duarte, C.M.M., Akretche, D.E., 2018. Estimation of total phenolic contents and *in vitro* antioxidant and antimicrobial activities of various solvent extracts of *Melissa officinalis* L. Arab. J. Sci. Eng. 43, 3349−3357. Available from: https://doi.org/10.1007/s13369-017-3000-6.

Mahajan, M., Kuiry, R., Pal, P.K., 2020. Understanding the consequence of environmental stress for accumulation of secondary metabolites in medicinal and aromatic plants. J. Appl. Res. Med. Aromat. Plants 18, 100255. Available from: https://doi.org/10.1016/j.jarmap.2020.100255.

Mohamed, M.A., Ibrahim, M.E., Wahba, H.E., 2018. Flavoring compounds of essential oil isolated from agriculture waste of coriander (*Coriandrum sativum*) plant. J. Mater. Environ. Sci. 9 (1), 77−82. Available from: https://doi.org/10.26872/jmes.2018.9.1.9.

Moisa, C., Copolovici, L., Bungău, S., Pop, G., Imbrea, I., Lupitu, A., et al., 2018. Wastes resulting from aromatic plants distillation −bio−sources of antioxidants and phenolic compounds with biological active principles. Farmacia 66 (2), 289−295.

Mojzer, E.B., Hrnčič, M.K., Škerget, M., Knez, Ž., Bren, U., 2016. Polyphenols: extraction methods, antioxidative action, bioavailability and anticarcinogenic effects. Molecules 21 (7), 901. Available from: https://doi.org/10.3390/molecules21070901.

Navarrete, A., Herrero, M., Martín, A., Cocero, M.J., Ibáñez, E., 2011. Valorization of solid wastes from essential oil industry. J. Food Eng. 104 (2), 196−201. Available from: https://doi.org/10.1016/j.jfoodeng.2010.10.033.

Nilashi, M., Samad, S., Yusuf, S.Y.M., Akbari, E., 2020. Can complementary and alternative medicines be beneficial in the treatment of COVID-19 through improving immune system function? J. Infect. Public Health 13 (6), 893−896. Available from: https://doi.org/10.1016/j.jiph.2020.05.009.

Onursal, E., Ekinci, K., 2015. Co-composting of rose oil processing waste with caged layer manure and straw or sawdust: effects of carbon source and C/N ratio on decomposition. Waste Manag. Res. 33 (4), 332−338. Available from: https://doi.org/10.1177/0734242X15574560.

Pandey, J., Verma, R.K., Singh, S., 2019. Suitability of aromatic plants for phytoremediation of heavy metal contaminated areas: a review. Int. J. Phytoremediat. 21 (5), 405−418. Available from: https://doi.org/10.1080/15226514.2018.1540546.

Pandey, R., 2005. Management of *Meloidogyne incognita* in *Artemisia pallens* with bio-organics. Phytoparasitica 33, 304−308. Available from: https://doi.org/10.1007/BF02979868.

Pereira, E., Pimenta, A.I., Calhelha, R.C., Antonio, A.L., Verde, S.C., Barros, L., et al., 2016. Effects of gamma irradiation on cytotoxicity and phenolic compounds of *Thymus vulgaris* L. and *Mentha* x *piperita* L. LWT − Food Sci. Technol. 71, 370−377. Available from: https://doi.org/10.1016/j.lwt.2016.04.004.

Pergola, M., Persiani, A., Palese, A.M., Di Meo, V., Pastore, V., D'Adamo, C., et al., 2018. Composting: the way for a sustainable agriculture. 123, 744−750. https://doi.org/10.1016/j.apsoil.2017.10.016.

Perricone, M., Arace, E., Corbo, M.R., Sinigaglia, M., Bevilacqua, A., 2015. Bioactivity of essential oils: a review on their interaction with food components. Front. Microbiol. 6, 76. Available from: https://doi.org/10.3389/fmicb.2015.00076.

Politi, M., Menghini, L., Conti, B., Bedini, S., Farina, P., Cioni, P.L., et al., 2020. Reconsidering hydrosols as main products of aromatic plants manufactory: the lavandin (*Lavandula* × *intermedia*) case study in Tuscany. Molecules 25 (9), 2225. Available from: https://doi.org/10.3390/molecules25092225.

Proestos, C., Varzakas, T., 2017. Aromatic plants: antioxidant capacity and polyphenol characterization. Foods 6 (4), 28. Available from: https://doi.org/10.3390/foods6040028.

Radaelli, M., da Silva, B.P., Weidlich, L., Hoehne, L., Flach, A., da Costa, L.A.M.A., et al., 2016. Antimicrobial activities of six essential oils commonly used as condiments in Brazil against *Clostridium perfringens*. Braz. J. Microbiol. 47 (2), 424−430. Available from: https://doi.org/10.1016/j.bjm.2015.10.001.

Rajan, R., Robin, D.T., Vandanarani, M., 2019. Biomedical waste management in Ayurveda hospitals − current practices and future prospectives. J. Ayurveda Integr. Med. 10 (3), 214−221. Available from: https://doi.org/10.1016/j.jaim.2017.07.011.

Raut, J.S., Karuppayil, S.M., 2014. A status review on the medicinal properties of essential oils. Ind. Crop. Prod 62, 250−264. Available from: https://doi.org/10.1016/j.indcrop.2014.05.055.

Raveau, R., Fontaine, J., Hijri, M., Sahraoui, A.L.H., 2020. The aromatic plant clary sage shaped bacterial communities in the roots and in the trace element-contaminated soil more than mycorrhizal inoculation − a two-year monitoring field trial. Front. Microbiol 11, 586050. Available from: https://doi.org/10.3389/fmicb.2020.586050.

Rezaie-Tavirani, M., Fayazfar, S., Heydari-Keshel, S., Rezaee, M.B., Zamanian-Azodi, M., Rezaei-Tavirani, M., et al., 2013. Effect of essential oil of *Rosa Damascena* on human colon cancer cell line SW742. Gastroenterol. Hepatol. Bed Bench 6 (1), 25−31.

Romero, E., Plaza, C., Senesi, N., Nogales, R., Polo, A., 2007. Humic acid-like fractions in raw and vermicomposted winery and distillery wastes. Geoderma. 139 (3−4), 397−406. Available from: https://doi.org/10.1016/j.geoderma.2007.03.009.

Saha, A., Basak, B.B., 2020. Scope of value addition and utilization of residual biomass from medicinal and aromatic plants. Ind. Crop. Prod 145, 111979. Available from: https://doi.org/10.1016/j.indcrop.2019.111979.

Sadh, P.K., Duhan, S., Duhan, J.S., 2018. Agro-industrial wastes and their utilization using solid state fermentation: a review. Bioresour. Bioprocess. 5, 1. Available from: https://doi.org/10.1186/s40643-017-0187-z.

Samarth, R.M., Samarth, M., Matsumoto, Y., 2017. Medicinally important aromatic plants with radioprotective activity. Future Sci. OA 3 (4), FSO247. Available from: https://doi.org/10.4155/fsoa-2017-0061.

Sánchez, Ó.J., Ospina, D.A., Montoya, S., 2017. Compost supplementation with nutrients and microorganisms in composting process. Waste Manag. 69, 136−153. Available from: https://doi.org/10.1016/j.wasman.2017.08.012.

Sánchez-Vioque, R., Polissiou, M., Astraka, K., de los Mozos-Pascual, M., Tarantilis, P., Herraiz-Peñalver, D., et al., 2013. Polyphenol composition and antioxidant and metal chelating activities of the solid residues from the essential oil industry. Ind. Crop. Prod. 49, 150−159. Available from: https://doi.org/10.1016/j.indcrop.2013.04.053.

Schmidt, E., 2020. Production of essential oils. In: Başer, K.H.C., Buchbauer, G. (Eds.), Handbook of Essential Oils: Science, Technology, and Applications, third ed. CRC Press, Taylor & Francis Group, USA. Available from: https://doi.org/10.1201/9781351246460.

Sharma, D., Yadav, K.D., Kumar, S., 2018. Biotransformation of flower waste composting: optimization of waste combinations using response surface methodology. Bioresour. Technol. 270, 198−207. Available from: https://doi.org/10.1016/j.biortech.2018.09.036.

Sienkiewicz, A., Piotrowska-Niczyporuk, A., Bajguz, A., 2020. Herbal industry wastes as potential materials for biofuel production. Proceedings 51 (1), 6. Available from: https://doi.org/10.3390/proceedings2020051006.

Siles, J.A., Vargas, F., Gutiérrez, M.C., Chica, A.F., Martín, M.A., 2016. Integral valorisation of waste orange peel using combustion, biomethanisation and co-composting technologies. Bioresour. Technol. 211, 173−182. Available from: https://doi.org/10.1016/j.biortech.2016.03.056.

Singh, D., Suthar, S., 2012a. Vermicomposting of herbal pharmaceutical industry waste: earthworm growth, plant-available nutrient and microbial quality of end materials. Bioresour. Technol. 112, 179−185. Available from: https://doi.org/10.1016/j.biortech.2012.02.101.

Singh, D., Suthar, S., 2012b. Vermicomposting of herbal pharmaceutical industry solid wastes. Ecol. Eng. 39, 1−6. Available from: https://doi.org/10.1016/j.ecoleng.2011.10.015.

Singh, R., Soni, S.K., Awasthi, A., Kalra, A., 2012. Evaluation of vermicompost doses for management of root-rot disease complex in *Coleus forskohlii* under organic field conditions. Australas. Plant. Pathol. 41, 397−403. Available from: https://doi.org/10.1007/s13313-012-0134-6.

Singh, R., Singh, R., Soni, S.K., Singh, S.P., Chauhan, U.K., Kalra, A., 2013. Vermicompost from biodegraded distillation waste improves soil properties and essential oil yield of *Pogostemon cablin* (patchouli) Benth. Appl. Soil. Ecol. 70, 48−56. Available from: https://doi.org/10.1016/j.apsoil.2013.04.007.

Slavov, A., Vasileva, I., Stefanov, L., Stoyanova, A., 2017. Valorization of wastes from the rose oil industry. Rev. Environ. Sci. Bio/Technol. 16, 309−325. Available from: https://doi.org/10.1007/s11157-017-9430-5.

Tao, W.Y., Jin, J.J., Zheng, Y.P., Li, S., 2021. Current advances of resource utilization of herbal extraction residues in China. Waste Biomass Valoriz. 12, 5853−5868. Available from: https://doi.org/10.1007/s12649-021-01428-8.

Teigiserova, D.A., Tiruta-Barna, L., Ahmadi, A., Hamelin, L., Thomsen, M., 2021. A step closer to circular bioeconomy for citrus peel waste: a review of yields and technologies for sustainable management of essential oils. J. Environ. Manag. 280, 111832. Available from: https://doi.org/10.1016/j.jenvman.2020.111832.

Thakur, M., Bhattacharya, S., Khosla, P.K., Puri, S., 2019. Improving production of plant secondary metabolites through biotic and abiotic elicitation. J. Appl. Res. Med. Aromat. Plants 12, 1−12. Available from: https://doi.org/10.1016/j.jarmap.2018.11.004.

Toledo, M., Siles, J.A., Gutiérrez, M.C., Martín, M.A., 2018. Monitoring of the composting process of different agroindustrial waste: influence of the operational variables on the odorous impact. Waste Manag. 76, 266−274. Available from: https://doi.org/10.1016/j.wasman.2018.03.042.

Trivellini, A., Lucchesini, M., Maggini, R., Mosadegh, H., Villamarin, T.S.S., Vernieri, P., et al., 2016. *Lamiaceae* phenols as multifaceted compounds: bioactivity, industrial prospects and role of "positive−stress". Ind. Crop. Prod. 83, 241−254. Available from: https://doi.org/10.1016/j.indcrop.2015.12.039.

Turrini, F., Beruto, M., Mela, L., Curir, P., Triglia, G., Boggia, R., et al., 2021. Ultrasound-assisted extraction of lavender (*Lavandula angustifolia* Miller, cultivar Rosa) solid by-products remaining after the distillation of the essential oil. Appl. Sci. 11 (12), 5495. Available from: https://doi.org/10.3390/app11125495.

Tzima, K., Brunton, N.P., Rai, D.K., 2018. Qualitative and quantitative analysis of polyphenols in *Lamiaceae* plants−a review. Plants 7 (2), 25. Available from: https://doi.org/10.3390/plants7020025.

Vasileva, I., Denkova, R., Chochkov, R., Teneva, D., Denkova, Z., Dessev, T., et al., 2018. Effect of lavender (*Lavandula angustifolia*) and melissa (*Melissa officinalis*) waste on quality and shelf life of bread. Food Chem. 253, 13−21. Available from: https://doi.org/10.1016/j.foodchem.2018.01.131.

Veličković, D.T., Milenović, D.M., Ristić, M.S., Veljković, V.B., 2008. Ultrasonic extraction of waste solid residues from the *Salvia* sp. essential oil hydrodistillation. Biochem. Eng. J. 42 (1), 97−104. Available from: https://doi.org/10.1016/j.bej.2008.06.003.

Verma, R.K., Verma, R.S., Rahman, L.U., Yadav, A., Patra, D.D., Kalra, A., 2014. Utilization of distillation waste-based vermicompost and other organic and inorganic fertilizers on improving production potential in geranium and soil health. Commun. Soil. Sci. Plant. Anal. 45 (2), 141−152. Available from: https://doi.org/10.1080/00103624.2013.854803.

Wang, M.H., Liu, Y., Wang, S.Q., Wang, K., Zhang, Y., 2021. Development of a compound microbial agent beneficial to the composting of Chinese medicinal herbal residues. Bioresour. Technol. 330, 124948. Available from: https://doi.org/10.1016/j.biortech.2021.124948.

Wu, F.F., Jin, Y.M., Xu, X.M., Yang, N., 2017a. Electrofluidic pretreatment for enhancing essential oil extraction from citrus fruit peel waste. J. Clean. Prod. 159, 85−94. Available from: https://doi.org/10.1016/j.jclepro.2017.05.010.

Wu, J.Q., Zhao, Y., Qi, H.S., Zhao, X.Y., Yang, T.X., Du, Y.Q., et al., 2017b. Identifying the key factors that affect the formation of humic substance during different materials composting. Bioresour. Technol. 244 (1), 1193−1196. Available from: https://doi.org/10.1016/j.biortech.2017.08.100.

Ye, S.J., Zeng, G.M., Wu, H.P., Liang, J., Zhang, C., Dai, J., et al., 2019. The effects of activated biochar addition on remediation efficiency of co-composting with contaminated wetland soil. Resour. Conserv. Recycl. 140, 278−285. Available from: https://doi.org/10.1016/j.resconrec.2018.10.004.

Youngquist, C.P., Mitchell, S.M., Cogger, C.G., 2016. Fate of antibiotics and antibiotic resistance during digestion and composting: a review. J. Environ. Qual. 45 (2), 537−545. Available from: https://doi.org/10.2134/jeq2015.05.0256.

Zaccardelli, M., Sorrentino, R., Caputo, M., Scotti, R., De Falco, E., Pane, C., 2020. Stepwise-selected *Bacillus amyloliquefaciens* and *B. subtilis* strains from composted aromatic plant waste able to control soil-borne diseases. Agriculture 10 (2), 30. Available from: https://doi.org/10.3390/agriculture10020030.

Zaccardelli, M., Roscigno, G., Pane, C., Celano, G., Di Matteo, M., Mainente, M., et al., 2021. Essential oils and quality composts sourced by recycling vegetable residues from the aromatic plant supply chain. Ind. Crop. Prod. 162, 113255. Available from: https://doi.org/10.1016/j.indcrop.2021.113255.

Zhang, J.P., Zhang, T.T., Ying, Y., Yao, X.H., 2021a. Effects of different additives on the chemical composition and microbial diversity during composting of *Camellia oleifera* shell. Bioresour. Technol. 330, 124990. Available from: https://doi.org/10.1016/j.biortech.2021.124990.

Zhang, M.X., Wang, C.C., Zhang, R., Chen, Y., Zhang, C.H., Heidi, H., et al., 2021b. Comparison of the guidelines on good agricultural and collection practices in herbal medicine of the European Union, China, the WHO, and the United States of America. Pharmacol. Res. 167, 105533. Available from: https://doi.org/10.1016/j.phrs.2021.105533.

Chapter 7

Lignocellulosic biodegradation of solid organic paper mill waste: succession of enzymes and microbial consortium towards waste valorisation

Ram Kumar Ganguly and Susanta Kumar Chakraborty
Department of Zoology, Vidyasagar University, Midnapore, West Bengal, India

7.1 Introduction

The degradation of the environment causes immense economic loss to society each year and damages the health of humans, ecosystems, and food security (Nellemann and Corcoran, 2010). Approximately 60% of the world's ecosystems are considered degraded (Prăvălie, 2021), which continues to serve as a sink for contaminants from many anthropological activities. A legacy of unregulated wastewater discharge from the pulp, paper, and sawmill industries is one of the causes of many environmental pollution problems. One of these is lignocellulose-bearing sediments. Lignocellulosic sediment pollution is a global environmental problem (Hoffman et al., 2019; Owens, 1991). It is common for sediments containing lignocellulose to be heavily polluted by toxic elements (metals and metalloids) and persistent organic pollutants (PCBs, chlorinated pesticides, PAHs). Such contaminants are found in abundance at shallow depths, which pose serious threats to aquatic ecosystems (Apler et al., 2019).

As a result of their persistence, the lignocellulose-bearing sediments present serious pollution problems today. Among their major components, lignocellulosic wastes consist of cellulose (13%–50%), hemicellulose (0.20%–40%), and lignin (16%–45%) (Iqbal et al., 2013). It was found that the cellulose crystallinity and lignin hydrophobicity, as well as lignin-hemicellulose network trapping, attributed to the lignocellulose's integrity and resistance to depolymerisation (Barakat et al., 2013; Singhvi and Gokhale, 2019). Plant cell walls

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00013-X

are primarily composed of cellulose and hemicellulose, which provide mechanical strength. Cellulose is composed of glucose monomers linked by β-1,4-glycosidic bonds. A major component of lignocellulosic biomass, lignin contains three types of aromatic units (i.e., coniferyl alcohol, sinapyl alcohol and p-coumaryl alcohol), which form a protective encapsulation around two other components. The C-C and ether bonds are the major chemical bonding complexes that contribute to lignin's resistance (Bugg et al., 2011). These bonds are highly resistant to hydrolysis, making lignin resistant to biodegradation.

In the pulp and paper industry, waste residues are generated from primary clarifiers, biological treatment, and recycling paper streams (Kaur et al., 2010). The composition of pulp waste and solid residues recovered from the waste stream depends on the origin of the cellulose fibre (wood or recycled paper). In general, paper wastes are composed of lignin (20%–30%), hemicellulose (30%–40%) and cellulose (40%–50%). Furthermore, waste from paper mills is contaminated with fibres, staples and metals, plastics, glass and sand from the glass washers (Méndez et al., 2009; Bajpai, 2015). Therefore, pulp and paper mills require major investment and operating costs because of its high moisture content and poor dewatering properties of organic waste.

Environment plays an important role in life on Earth. Researchers, manufacturers and scientists are finding ways to constrain the release of hazardous pollutants into the atmosphere, from radioactive waste to untreated water. This poses a significant challenge to the health and survival of all life on Earth. The thermal degradation of waste, such as incineration and combustion, are new techniques for waste disposal. Despite their effectiveness in reducing a wide range of chemicals, such techniques do carry some disadvantages. Biological remediation involves the usage of organisms in the degradation, reduction, and conversion of complex hazardous materials into simpler, more readily available forms. Globally, the United States (US) has the highest number of paper manufacturers. The CEPI (2018) stated that by 2030 there would, however, be an increase in paper production of 500 million tons (CEPI, 2018). Production of waste will be in quick ascendancy as a result of the upsurge in manufacturing (Novais et al., 2018). Generally, waste materials generated from paper manufacturing industries are divided into two categories: non-activated or primary waste (P) produced before any physico-chemical treatment and activated or secondary waste (SS) generated after a variety of treatments like charging, settling, inclining and drying of crude waste (Rivera et al., 2016).

Biological systems and human well-being are affected by the manufacture of such inexorable pollutants in nature (Kamali et al., 2016). There have been a variety of regulations based on ecological concerns that aim to increase awareness of the diversity of pollution issues resulting from an all-encompassing approach to waste management (Wenning et al., 2000; Sumathi and Hung, 2006; Ashrafi et al., 2015).

For this reason, numerous categories of sophisticated machines are now being used to detect and mitigate various point sources or non-point sources pollutants (Spadotto and Mingoti, 2019). Thus, various biotechnological procedures are being used today for waste conversion in order to ensure proper ecological management (Onwurah et al., 2006; Wong et al., 2016; Awasthi et al., 2019).

Among the biotechnological tools used to manage hazardous organic wastes for environmental sustainability, vermitechnology emerged as an innovative tool (Nogales et al., 2020; Karmegam et al., 2019). It is characterised by earthworm activity and microbial activity, which assist in the degradation of organics (Yang et al., 2020). The process, however, relies on the release of enzymes extracellularly that turn the decomposing materials into harmless compounds. By using a gut-allied process, complex organic constituents are homogenised into simpler structures (Bhat et al., 2017). The resulting mineralised forms can then be readily absorbed by plants (Sinha et al., 2010). The present study has discussed the biotransformation of lignocellulosic wastes of solid paper mill waste using vermitechnology and reviewed the succession of different enzymes and microbial groups during the process of valorisation.

7.2 Biodegradation of lignocellulosic component of paper mill waste

The presence or absence of various spectral peaks and certain functional assemblies can be discerned as indicators of bioconversion or biodegradation and should be used as a tool to judge the mellowness of a sample (Hussain et al., 2016). The maturity of vermicompost can be measured by the appearance or disappearance of various spectral peaks (Deka et al., 2011; Paul et al., 2020). The impacts of biodegradation on the mineralisation of polyphenolic substances, lignin, cellulose, etc., were evidenced by FTIR spectroscopy (Fig. 7.1).

Spectral peaks among wavenumbers of 3200−3600 cm^{-1} confirm an increase in the number of hydroxyl groups (hydroxyl groups) in acids, alcohols and phenols. It was demonstrated that different compounds interacted with hydrogen bonds through peak widening (Ganguly and Chakraborty, 2020). The widening and diminution of transmittance peaks between 2900 and 2600 cm^{-1} had revealed the breakdown of organic alkenes (Hussain et al., 2016; Ganguly and Chakraborty, 2019). The decrease in transmission strength in ($C=O$) stretching of several esters and anhydrides was conspicuous among vermicomposted samples in comparison to untreated organic wastes (Sudarshan et al., 2016). A well-marked breakdown of different esters was clearly evident among wavenumbers 1740−1880 cm^{-1}. Peak widening at 1425−1450 cm^{-1} had signified the collapse of complex lignins (Ravindran et al., 2008). The non-appearance of spectroscopical bands at

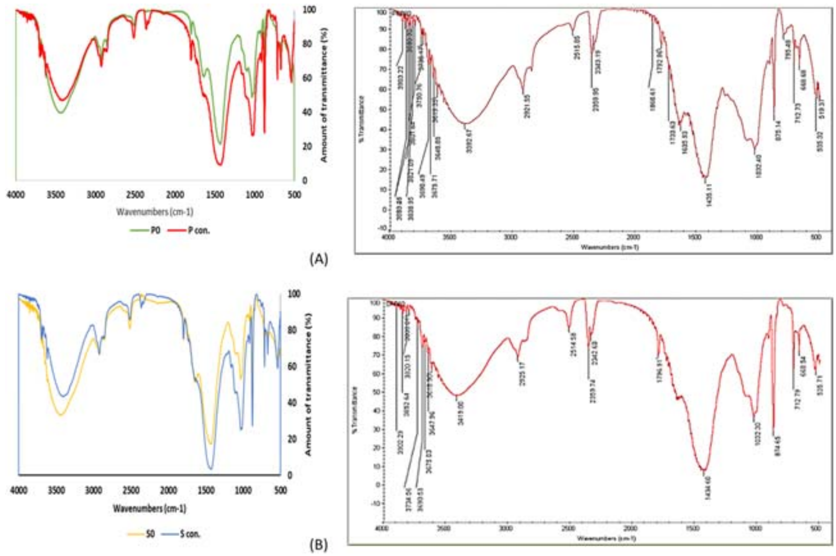
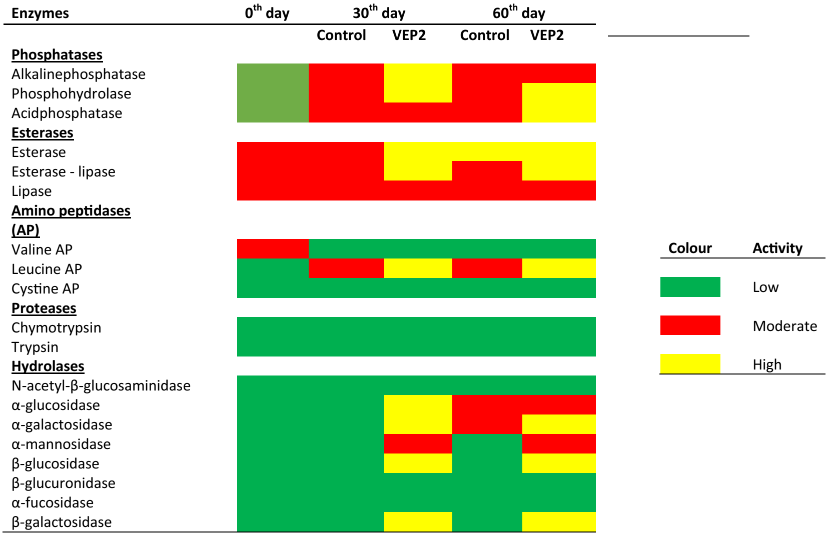


FIGURE 7.1 FTIR spectroscopical peaks of (A) primary and (B) secondary PMW during vermicomposting. P0, P_{con}, P60 characterised primary PMW at the 0th day, control set and 60th day of the procedure. S0, S_{con}, S60 characterised secondary PMW at 0th day, control set and 60th day of the procedure.

1630–1640 cm^{-1} among the primary and secondary PMW samples had represented degradation of amides, which was demonstrated through the rise of peaks at 840–780 cm^{-1} signifying nitrogen mineralisation. Afterwards, a drop in transmission peak at 1200–1100 cm^{-1} was observed, which validated a potential degradation of macromolecular organic polysaccharides and the availability of more naive carbons (Ganguly and Chakraborty, 2021b). An earlier study had revealed a diminishing tendency of peak 2921/1633 proportion was discerned from the compost produced from primary PMW (0.75–0.66) (Fig. 7.2A) and secondary PMW (0.69–0.59).

7.3 Role of microbial enzymes in lignocellulosic degradation

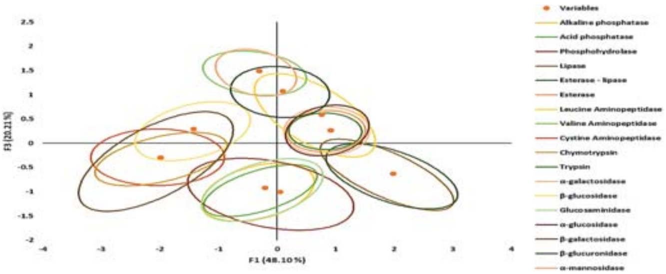
Toxicants can be removed from nature by using several physiochemical methods. In recent years, bioremediation has gained valuable traction due to its eco-friendliness, ease of use, and efficiency in degrading organic contaminants (Ganguly and Chakraborty, 2018). The biodegradation of organic polymers involves the origin of extracellular enzymes that are capable of hydrolysis (Figs. 7.2 and 7.3). Hydrolytic enzymes include lipases, proteases, amino peptidases and glycosyl hydroxylases (Ganguly and Chakraborty, 2021a).



Enzymes	0th day	30th day		60th day	
		Control	VEP2	Control	VEP2
Phosphatases					
Alkalinephosphatase	🟩	🟥	🟨	🟥	🟥
Phosphohydrolase	🟩	🟥	🟨	🟥	🟨
Acidphosphatase	🟩	🟥	🟥	🟥	🟨
Esterases					
Esterase	🟥	🟥	🟨	🟨	🟨
Esterase - lipase	🟥	🟥	🟨	🟥	🟨
Lipase	🟥	🟥	🟥	🟥	🟥
Amino peptidases (AP)					
Valine AP	🟥	🟩	🟩	🟩	🟩
Leucine AP	🟩	🟥	🟨	🟥	🟨
Cystine AP	🟩	🟩	🟩	🟩	🟩
Proteases					
Chymotrypsin	🟩	🟩	🟩	🟩	🟩
Trypsin	🟩	🟩	🟩	🟩	🟩
Hydrolases					
N-acetyl-β-glucosaminidase	🟩	🟩	🟩	🟩	🟩
α-glucosidase	🟩	🟩	🟨	🟥	🟥
α-galactosidase	🟩	🟩	🟨	🟥	🟨
α-mannosidase	🟩	🟩	🟥	🟩	🟥
β-glucosidase	🟩	🟩	🟨	🟩	🟨
β-glucuronidase	🟩	🟩	🟩	🟩	🟩
α-fucosidase	🟩	🟩	🟩	🟩	🟩
β-galactosidase	🟩	🟩	🟨	🟩	🟨

Colour	Activity
🟩	Low
🟥	Moderate
🟨	High

(A)



(B)

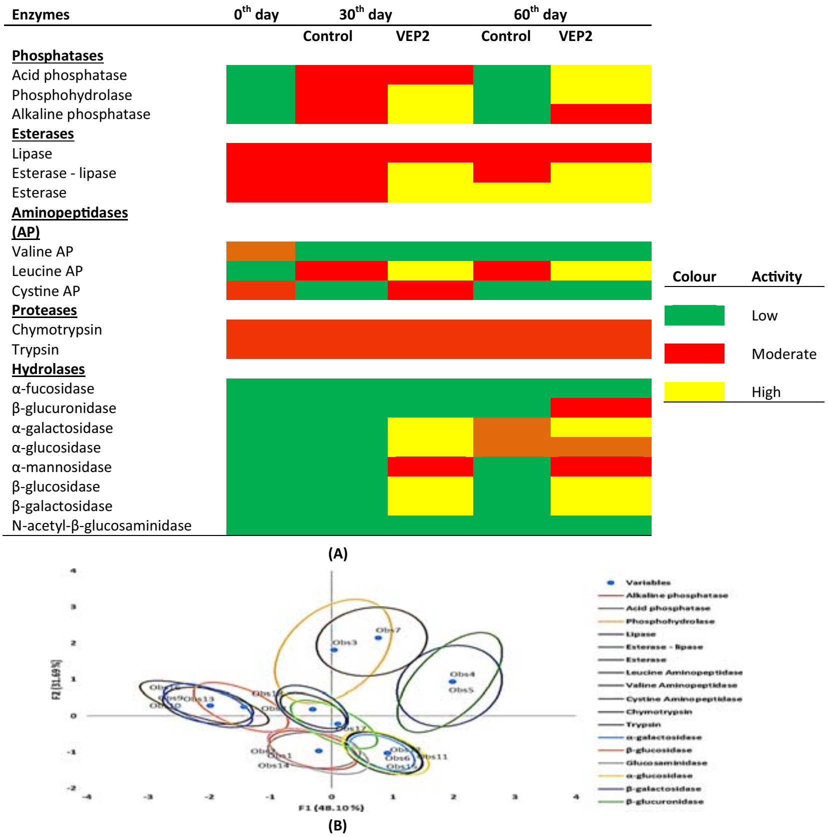
FIGURE 7.2 (A) Characterise succession of enzyme activities in vermicompost samples of primary paper mill waste (B) Principal component analysis of diverse marker enzymes in primary PMW. Bootstrap confidence ellipse validates the degree of variation instigated by the variables.

7.3.1 Microbial lipases

Various types of prokaryotic and eukaryotic lipids can be reduced by lipase. They are highly efficient in destroying complex hydrocarbons in the environment. There are several isozymes of lipase derived from various forms of life, such as bacteria, fungi and animals, which perform various chemical transformations, such as hydrolysis and transesterification. The ubiquitous enzyme lipases are responsible for creating free fatty acids through the hydrolysis of lipids.

7.3.2 Microbial proteases

Peptide chains are broken down by proteases. Because of its application in various industrial settings, microbial sources of protease enzymes have gained



Enzymes	0th day	30th day		60th day	
		Control	VEP2	Control	VEP2
Phosphatases					
Acid phosphatase	🟩	🟥	🟥	🟩	🟨
Phosphohydrolase	🟩	🟥	🟨	🟩	🟨
Alkaline phosphatase	🟩	🟥	🟨	🟩	🟥
Esterases					
Lipase	🟥	🟥	🟥	🟥	🟥
Esterase - lipase	🟥	🟥	🟨	🟥	🟨
Esterase	🟥	🟥	🟨	🟨	🟨
Aminopeptidases (AP)					
Valine AP	🟧	🟩	🟩	🟩	🟩
Leucine AP	🟩	🟥	🟨	🟥	🟨
Cystine AP	🟧	🟩	🟥	🟩	🟩
Proteases					
Chymotrypsin	🟧	🟧	🟧	🟧	🟧
Trypsin	🟧	🟧	🟧	🟧	🟧
Hydrolases					
α-fucosidase	🟩	🟩	🟩	🟩	🟩
β-glucuronidase	🟩	🟩	🟩	🟩	🟥
α-galactosidase	🟩	🟩	🟨	🟧	🟨
α-glucosidase	🟩	🟩	🟨	🟧	🟧
α-mannosidase	🟩	🟩	🟥	🟩	🟥
β-glucosidase	🟩	🟩	🟨	🟩	🟨
β-galactosidase	🟩	🟩	🟨	🟩	🟨
N-acetyl-β-glucosaminidase	🟩	🟩	🟩	🟩	🟩

Colour	Activity
🟩	Low
🟥	Moderate
🟨	High

(A)

(B)

FIGURE 7.3 (A) Characterise succession of enzyme activities in vermicompost samples of secondary paper mill waste (B) Principal component analysis of diverse marker enzymes in secondary PMW. Bootstrap confidence ellipse validates the degree of variation instigated by the variables.

prominence. Proteinases mainly fall into two categories: exopeptidases and endopeptidases. Currently, prevailing research has taken into consideration different types of exopeptidases, such as several aminopeptidase isozymes. During aminopeptidase activity, free amino acids and peptides are released from free N terminals of polypeptides. A large number of extracellular proteases are produced by microorganisms and hence play a crucial role in nitrogen metabolism. So, proteases were deemed important in the biodegradation of organic waste.

7.3.3 Glycosyl hydrolases

Glycosyl hydrolases are the enzymes responsible for carbohydrate degradation. These enzymes breakdown glycosidic bonds in carbohydrates such as cellulose and lignin. The biodegradation of complex polymers was profoundly influenced by the extracellular enzymes produced by the microbial biomass. The prevailing research has thus far examined how microbial enzymes are successively produced among organic waste samples. A number of potential microbial isolates were also screened for enzyme production sensitivity

A decrease in total organic carbon (TOC) content was observed as the result of the extracellular breakdown of carbohydrate polymers. The breakdown of complex cellulosic material to monomers during decomposition was also reported earlier in studies by different microbes and fungi isolates (Gautam et al., 2012; Takizawa et al., 2020). Nitrogen had been transformed into free forms of amino acids through an elemental process in vermicompost as a result of the increase in nitrogen content. Vitek 2 characterisation had indicated that microbes play a crucial role in N-degrading enzyme (arylamidase) manufacture (Tables 7.1 and 7.2). In previous studies, nitrogen content was found to increase after microbe inoculation (Kominoski et al., 2015; Chang et al., 2019). Nevertheless, proteins are not directly assimilated or degraded by bacteria; they rely on enzymes produced either inside or outside the cell for the release of simple amino acids from polymeric organic compounds (Hu et al., 2018).

Vermicomposting is a microbial process; therefore, attention has been paid to the isolation of microbes and their potential for the production of extracellular enzymes that aid in the maintenance of C/N ratios (Qiao et al., 2019; Zhu et al., 2020).

Earlier studies by (Ganguly and Chakraborty, 2020) had revealed the extracellular release of enzymes by several microbial isolates (Fig. 7.4) like *Lysinibacillus sphaericus* (Gen Bank ID: MK910238.1), *Bacillus anthracis* (Gen Bank ID: MK910220.1), *Lysinibacillus macroides* (Gen Bank ID: MK910225.1), *Bacillus cereus* (Gen Bank ID: MK910262.1), *Bacillus thuringiensis* (Gen Bank ID: MK910218.1), *Lysinibacillus fusiformis* (Gen Bank ID: MK910239.1) and *Lysinibacillus* sp. (Gen Bank ID: MK910217.1).

7.4 Microbial succession using PLFA biomembrane markers

Vermicomposting demonstrates the importance of earthworms in developing microbial biomass and the succession of diverse microbial assemblages throughout the process (Klamer and Bååth, 1998; Hussain et al., 2016). Different hydrolytic enzymes are secreted by microbial isolates, which enhance the process of biodegradation. Organic samples are transformed from hazardous forms in such microorganisms into harmless amendments or vermicompost by these organisms (Villar et al., 2016). It is primarily the microbial population responsible for the biochemical disintegration of organic compounds; however, earthworms may also influence this process as they directly feed on the organic substrate and increase the surface area for

TABLE 7.1 Sensitivity of bacterial isolates (screened at mid-point of vermicomposting of primary paper mill waste) towards secretion of extracellular enzymes as obtained from Vitek 2 automated biochemical analyser.

Test	SC1	SC2	SC3	SC4	SC5	SC6	SC7	SC8	SC9	SC10	SC11	SC12
L-lysine arylamidase	0	0	0	0	0	0	0	0	0	0	0	0
L-aspartate arylamidase	0	0	0	0	0	0	0	0	0	1	1	0
Leucine arylamidase	1	1	1	1	1	1	1	1	1	1	1	1
Phenylalanine arylamidase	1	1	1	1	1	1	1	1	1	1	1	1
L-proline arylamidase	1	1	1	1	1	1	1	1	1	0	0	1
Beta galactosidase	1	1	1	1	1	1	1	1	1	1	1	1
L-pyrrolydonyl arylamidase	0	0	0	0	0	0	0	0	0	0	0	0
Alpha galactosidase	1	1	1	1	1	1	1	1	1	1	1	1
Alanine arylamidase	0	0	0	0	0	0	0	0	0	1	1	0
Tyrosine arylamidase	1	1	1	1	1	1	1	1	1	1	1	1
Ala-phe-pro arylamidase	0	0	0	1	0	0	0	1	0	1	1	0
Alpha mannosidase	0	0	0	0	0	0	0	0	0	0	1	0
Glycine arylamidase	0	0	0	0	0	0	0	0	0	0	0	0
Beta-glucosidase	1	1	1	1	1	1	1	1	1	1	1	1
Beta-mannosidase	0	0	0	0	0	0	0	0	0	0	0	0
Alpha-glucosidase	1	1	1	1	1	1	1	1	1	1	1	1

1 and 0 represent the sensitivity and non-sensitivity of the bacterial isolates towards extracellular enzymes. SC followed by number represent the microbial isolates obtained during the study.

TABLE 7.2 Sensitivity of bacterial isolates (screened at mid-point of vermicomposting of secondary paper mill waste) towards secretion of extracellular enzymes as obtained from Vitek 2 automated biochemical analyser.

Test	RKG1	RKG2	RKG3	RKG4	RKG5	RKG6	RKG7	RKG8	RKG9	RKG10	RKG11	RKG12
L-lysine arylamidase	0	0	0	0	0	0	0	0	0	0	0	0
L-aspartate arylamidase	0	0	0	0	0	1	0	0	0	0	1	0
Leucine arylamidase	1	1	1	1	1	1	1	1	1	1	1	1
Phenylalanine arylamidase	1	1	1	1	1	1	1	1	1	1	1	1
L-proline arylamidase	1	1	1	1	1	0	1	1	1	1	0	1
Beta-galactosidase	1	1	1	1	1	1	1	1	1	1	1	1
L-pyrrolydonyl arylamidase	0	0	0	0	0	0	0	0	0	0	0	0
Alpha galactosidase	1	1	1	1	1	1	1	1	1	1	1	1
Alanine arylamidase	0	0	0	0	0	1	0	0	0	0	1	0
Tyrosine arylamidase	1	1	1	1	1	1	1	1	1	1	1	1
Ala-phe-pro arylamidase	1	0	0	0	0	1	1	0	0	0	1	0
Alpha-mannosidase	0	0	0	0	0	1	0	0	0	0	0	0
Glycine arylamidase	0	0	0	0	0	0	0	0	0	0	0	0
Beta-glucosidase	1	1	1	1	1	1	1	1	1	1	1	1
Beta-mannosidase	0	0	0	0	0	0	0	0	0	0	0	0
Alpha-glucosidase	1	1	1	1	1	1	1	1	1	1	1	1

1 and 0 represent the sensitivity and non-sensitivity of the bacterial isolates towards extracellular enzymes. RKG followed by number represents the microbial isolates obtained during the study.

Lysinibacillus sphaericus
GenBank Id: MK910238.1

Bacillus anthracis
GenBank Id: MK910220.1

Lysinibacillus macroides
GenBank Id: MK910225.1

Bacillus cereus
GenBank Id: MK910262.1

Bacillus thuringiensis
GenBank Id: MK910218.1

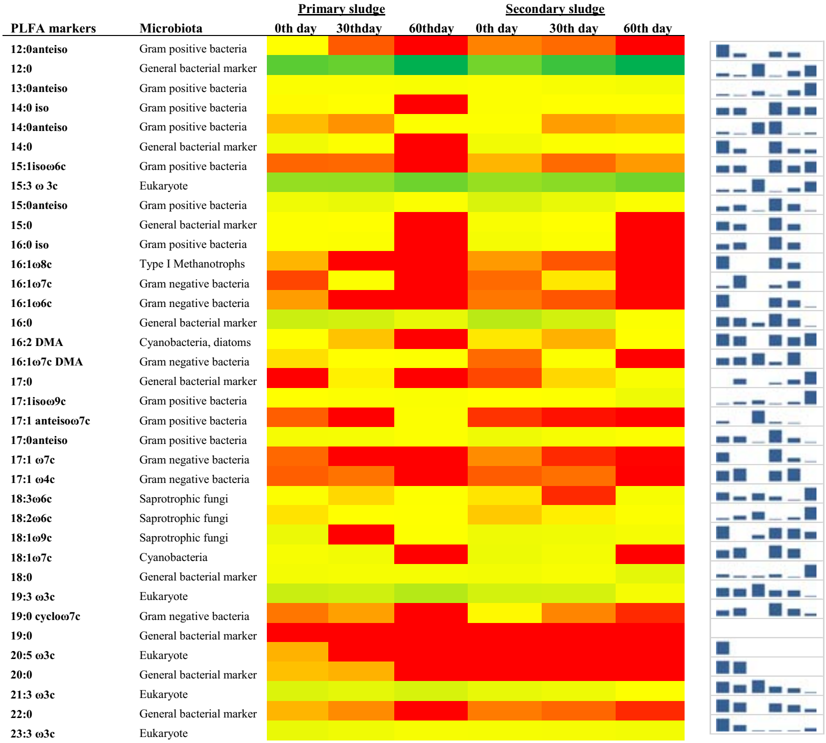
Lysinibacillus fusiformis
GenBank Id: MK910239.1

Lysinibacillus sp.
GenBank Id: MK910217.1

FIGURE 7.4 FE-SEM of bacterial species having a potential role in the secretion of extracellular enzymes for lignocellulosic degradation.

microbial digestion. Vermicomposting facilitates the reconditioning of organic materials and the release of nutrients. It is composed of a whole food web. There is a combination of biotic interactions among decomposers, such as competition, facilitation and inhibition, resulting in higher quality vermicompost. Through such biotic interactions at regular intervals, it is possible to distinguish different isolates of microorganisms.

PLFAs (Phospholipid Fatty Acids) are multifaceted and can be used as a biomarker of environmental stress (Villar et al., 2017). Any alteration in intracellular or extracellular conditions is likely to cause the divergence of fatty acids since they represent the major components of microbial biomass. In this way, PLFA composition reflected successions in microbial biomass (Haei et al., 2011) (Fig. 7.5). Due to their role as the primary component of microbial membranes, they can respond both to intracellular and extracellular environmental conditions. The second reason is that in response to environmental



PLFA markers	Microbiota
12:0anteiso	Gram positive bacteria
12:0	General bacterial marker
13:0anteiso	Gram positive bacteria
14:0 iso	Gram positive bacteria
14:0anteiso	Gram positive bacteria
14:0	General bacterial marker
15:1isoω6c	Gram positive bacteria
15:3 ω 3c	Eukaryote
15:0anteiso	Gram positive bacteria
15:0	General bacterial marker
16:0 iso	Gram positive bacteria
16:1ω8c	Type I Methanotrophs
16:1ω7c	Gram negative bacteria
16:1ω6c	Gram negative bacteria
16:0	General bacterial marker
16:2 DMA	Cyanobacteria, diatoms
16:1ω7c DMA	Gram negative bacteria
17:0	General bacterial marker
17:1isoω9c	Gram positive bacteria
17:1 anteisoω7c	Gram positive bacteria
17:0anteiso	Gram positive bacteria
17:1 ω7c	Gram negative bacteria
17:1 ω4c	Gram negative bacteria
18:3ω6c	Saprotrophic fungi
18:2ω6c	Saprotrophic fungi
18:1ω9c	Saprotrophic fungi
18:1ω7c	Cyanobacteria
18:0	General bacterial marker
19:3 ω3c	Eukaryote
19:0 cycloω7c	Gram negative bacteria
19:0	General bacterial marker
20:5 ω3c	Eukaryote
20:0	General bacterial marker
21:3 ω3c	Eukaryote
22:0	General bacterial marker
23:3 ω3c	Eukaryote

FIGURE 7.5 Profiling of percentage abundance of different PLFA (phospholipid fatty acid) markers during different intervals of vermicomposting of primary and secondary paper mill waste. Each cell is colourised with graded shades of three colours based on the abundance of markers obtained during different times of vermicomposting. Sparkline (*blue* coloured) was used for a better explanation of marker dynamics.

disturbances, the PLFA profile changes with changes in microbial membrane composition, which indicates phenotypic plasticity or succession of microbial species (Devi et al., 2020). PLFA membrane markers are used to indicate community edifice and the appearance of distinct microbial species during vermicomposting (Domínguez et al., 2010).

During the process, a noteworthy decline in microbial burden was discerned at the final phase in contrast to an early period of composting as primary PMW (1953.25−73.25 nmol g^{-1}, F value = 224.15, R^2 value = 0.59) and secondary PMW (1512.35−30.15 nmole g^{-1}, F value = 214.35, $R^2 = 0.68$). Such decrease of microbes was positively validated with the lessening of C/N proportion of the vermicompost as primary PMW (r value = 0.66, R^2 value = 0.76) and secondary PMW (r value = 0.76, R^2 value = 0.72). Earlier investigations had recommended such decline in for other carbon-based wastes and linked it with substrate obtainability along with adaptability and feeding behaviour of soft-bodied annelids (Fernández-Gómez et al., 2012).

At the early period of the procedure, biomembrane markers, such as i14:0, omega i15:1−6c, i16:0, i15:0 anti, i13:0 anti, omega 13:1−3c, omega16:1−8c and omega i17:1anti-7c, etc., were observed at greater abundance, which characterised the predominance of Gram + ve bacterial populaces among wastes samples (Gómez-Brandón et al., 2011; Willers et al., 2015). A comparable drift was also observed in biomembrane markers (Gram-ve bacterial populaces), such as omega 16:1−7c, omega 16:1−6c and omega 16:1−7c, at the primary stage of transformation of primary PMW (Figs. 7.5 and 7.6A). For secondary PMW, numerous Gram −ve biomembrane markers, such as omega 16:1−7c, omega 17:1−7c and omega cyclo19:0−7c, were seen to be maximum in the middle phase of the procedure (Figs. 7.5 and 7.6B). Such an increase in microbial masses had symbolised the thermophilic stage of the procedure (Klamer and Bååth, 1998).

Such noteworthy reduction of bacteria was accredited to the dearth of simple compounds, which confined development and the use of bacterial communities as the source of nutrients by Eisenia fetida (Gómez-Brandón et al., 2011). Previous studies had conveyed the upsurge of eukaryotes and fungi owing to their recolonisation at the maturation period of the procedure (Zhao et al., 2019). The biomembrane markers for the fungal population, such as omega18:2−6c and omega18:1−9c, had displayed advanced abundance at the final phase of bioconversion of primary PMW and middle phase (30th day) for secondary PMW. These distinct fungal communities among PMW can be attributed to three variables: (1) Temperature-dependent growth of fungal populations (Aira et al., 2006) (2) Foraging behaviour of annelids (Fernández-Gómez et al., 2012; Ganguly and Chakraborty, 2021b) and (3) Selection of fungal populations as nitrogen sources (chitin metabolism). It is possible that poor accessibility to food substrate or oxygen is responsible for the diminished eukaryotic and fungal biomass in the secondary PMW set.

During vermicomposting of secondary paper mill waste, fungi and eukaryotes had smaller biomasses due to poor food substrate or oxygen accessibility (Herrmann and Shann, 1997). A significant increase in numerous biomembrane markers belonging to eukaryotes, including omega 19:3−3c, omega 20:5−3c and omega 21:3−3c, was also evident at the final composting period for all PMW that showed maturity or stability of organic PMW (Villar et al., 2017; Ganguly and Chakraborty, 2021b). So, the process of valorisation produced a bottleneck effect that could be credited with paving the way for eukaryotic cells to evolve. In both PMWs, oscillations were less noticeable when comparing the diversity of fungi. In contrast to microbial biomasses, eukaryotes demonstrated deteriorating trends but enhanced at the final phases.

The PC1 axis displayed a maximum variation of 65% among primary PMW (VEP2), which included biomembrane markers like omega15:3−3c,

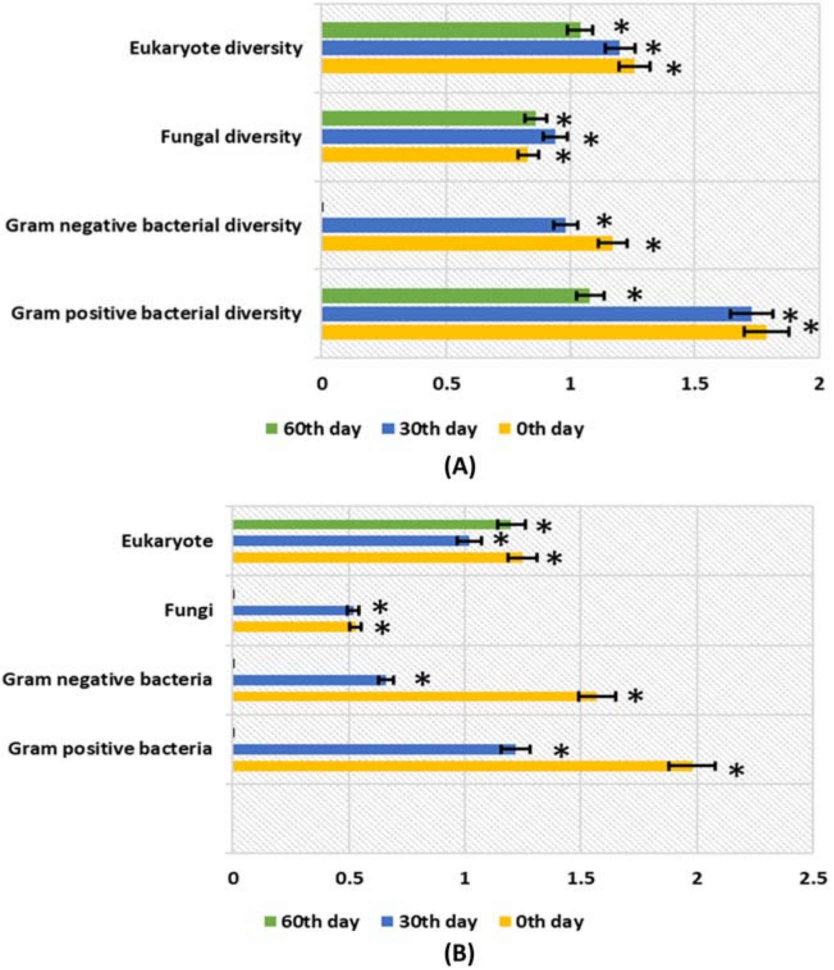


FIGURE 7.6 Shannon-Weiner diversity among microbial groups observed during vermicompost of (A) primary and (B) secondary paper mill wastes. Asterisk (*) designates the significance of the test.

16:0, i15:0 anti, whereas the PC2 axis showed a minimum variation of 32%, which included markers such as omega 19:3−3c, omega 18:1−9c and omega 21:3−3c (Fig. 7.7A). Additionally, such parallel arrays of biomembrane markers also revealed a predominant variation of 77% (PC1) and a negligible variation of 14% (PC2) for secondary PMW. The PCA plots and bootstrap ellipse demonstrated the highest variance in both vermicompost waste forms being accounted for by Gram-positive microbes, saprophytic fungi and eukaryotes (Fig. 7.7B).

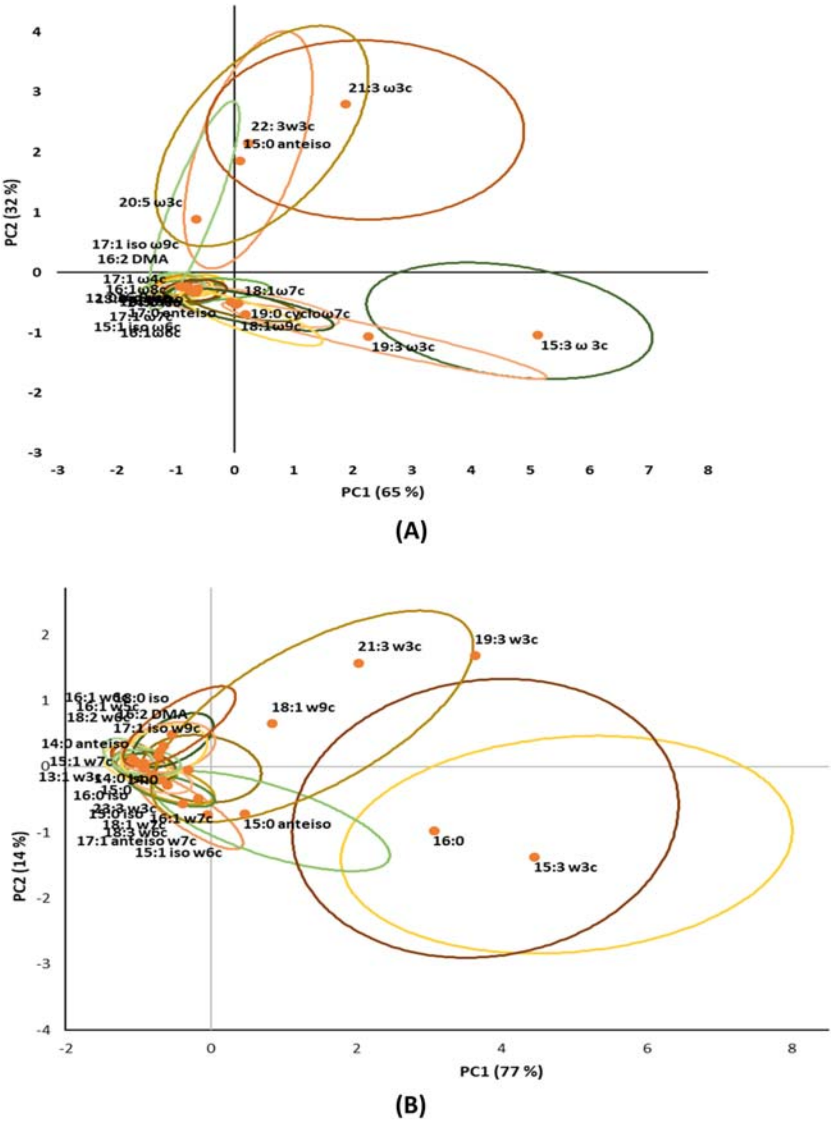


FIGURE 7.7 Principal component analysis of different PLFA markers of microbial groups observed during vermicompost of (A) primary and (B) secondary paper mill wastes. Bootstrap confidence ellipse validates the degree of variation instigated by the variables.

7.5 Conclusion

Vermicomposting of different paper mill waste samples revealed the presence of different nitrogen and carbon degrading enzymes such as esterase, aminopeptidase and glucosidase. The presence of extracellular enzymes was expected to play a significant role in maintaining the C/N ratio in final vermicompost samples.

The bacterial isolates (mutual to earthworm gut and vermicompost) like *Bacillus* spp. and *Lysinibacillus* spp. were sensitive to diverse categories of enzymes and evidenced to be a significant contributor to secretion of extracellular enzymes. The proportion of PLFA biomembrane markers belonging to bacteria decreased during the bioconversion procedure for both types of PMW, while the proportion of eukaryotic and fungal markers increased. Therefore, changes in microbes and their diversity can be attributed to the availability of organic feed and the type used in vermicomposting. Therefore, the study has demonstrated a number of hypotheses-

- Microbial symbionts act as essential mediators in the process of valorisation of organic waste paper mill wastes.
- The sequential production of different extracellular hydrolytic enzymes plays a crucial role in enhancing vermicompost products by generating simpler forms of carbon and nitrogen in an acceptable ratio (C:N).
- Vermicomposting results in a decreased microbial burden, which reduces the noxiousness of waste.
- An increase in microbial biomasses has contributed dramatically to the stability and maintenance of vermicompost's C/N ratio.
- The remarkable diversity of PLFA markers found in Gram-positive bacteria, fungi, and eukaryotes have reflected their potentials to act as biomarkers indicating the mellowness of the vermicompost.

References

Aira, M., Monroy, F., Domínguez, J., 2006. Changes in microbial biomass and microbial activity of pig slurry after the transit through the gut of the earthworm *Eudrilus eugeniae* (Kinberg, 1867). Biol. Fertil. Soils 42 (4), 371–376.

Apler, A., Snowball, I., Frogner-Kockum, P., Josefsson, S., 2019. Distribution and dispersal of metals in contaminated fibrous sediments of industrial origin. Chemosphere 215, 470–481.

Ashrafi, O., Yerushalmi, L., Haghighat, F., 2015. Wastewater treatment in the pulp-and-paper industry: a review of treatment processes and the associated greenhouse gas emission. J. Environ. Manag. 158, 146–157.

Awasthi, A.K., Hasan, M., Mishra, Y.K., Pandey, A.K., Tiwary, B.N., Kuhad, R.C., et al., 2019. Environmentally sound system for E-waste: biotechnological perspectives. Curr. Res. Biotechnol. 1, 58–64.

Bajpai, P., 2015. Management of Pulp and Paper Mill Waste. Springer International Publishing, Switzerland, 191–191.

Barakat, A., de Vries, H., Rouau, X., 2013. Dry fractionation process as an important step in current and future lignocellulose biorefineries: a review. Bioresour. Technol. 134, 362–373.

Bhat, S.A., Singh, J., Vig, A.P., 2017. Amelioration and degradation of pressmud and bagasse wastes using vermitechnology. Bioresour. Technol. 243, 1097–1104.

Bugg, T.D., Ahmad, M., Hardiman, E.M., Singh, R., 2011. The emerging role for bacteria in lignin degradation and bio-product formation. Curr. Opin. Biotechnol. 22 (3), 394–400.

CEPI, 2018. Key statistics European pulp and paper industry.

Chang, R., Li, Y., Li, J., Chen, Q., Zhao, H., Qi, Z., 2019. Influences of the thermophilic period on biodegradation and nitrogen loss in stimulated vegetable waste composting. Glob. Ecol. Conserv. 18, e00623.

Deka, H., Deka, S., Baruah, C., Das, J., Hoque, S., Sarma, H., et al., 2011. Vermicomposting potentiality of Perionyx excavatus for recycling of waste biomass of java citronella-an aromatic oil yielding plant. Bioresour. Technol. 102 (24), 11212−11217.

Devi, J., Deb, U., Barman, S., Das, S., Bhattacharya, S.S., Tsang, Y.F., et al., 2020. Appraisal of lignocellusoic biomass degrading potential of three earthworm species using vermireactor mediated with spent mushroom substrate: compost quality, crystallinity, and microbial community structural analysis. Sci. Total. Environ. 716, 135215.

Domínguez, J., Aira, M., Gómez-Brandón, M., 2010. Vermicomposting: earthworms enhance the work of microbes. Microbes at Work. Springer, pp. 93−114.

Fernández-Gómez, M.J., Nogales, R., Insam, H., Romero, E., Goberna, M., 2012. Use of DGGE and COMPOCHIP for investigating bacterial communities of various vermicomposts produced from different wastes under dissimilar conditions. Sci. Total. Environ. 414, 664−671.

Ganguly, R.K., Chakraborty, S.K., 2018. Assessment of microbial roles in the bioconversion of paper mill sludge through vermicomposting. J. Environ. Health Sci. Eng. 16 (2), 205−212.

Ganguly, R.K., Chakraborty, S.K., 2019. Assessment of qualitative enrichment of organic paper mill wastes through vermicomposting: humification factor and time of maturity. Heliyon 5 (5), e01638.

Ganguly, R.K., Chakraborty, S.K., 2020. Eco-management of industrial organic wastes through the modified innovative vermicomposting process: a sustainable approach in tropical countries. Earthworm Assisted Remediation of Effluents and Wastes. Springer, pp. 161−177.

Ganguly, R.K., Chakraborty, S.K., 2021a. Valorisation of toxic paper mill waste through vermicomposting: an insight towards cleaner engineering through alleviation of wastes. Clean. Eng. Technol. 100070. Available from: https://doi.org/10.1016/j.clet.2021.100070.

Ganguly, R.K., Chakraborty, S.K., 2021b. Qualitative assessment of paper mill waste valorisation through combinatorial PLFA markers and spectroscopical analysis: an ecotechnology towards waste to resource transformation. Environ. Technol. Innov. 101532. Available from: https://doi.org/10.1016/j.eti.2021.101532.

Gautam, S., Bundela, P., Pandey, A., Awasthi, M., Sarsaiya, S., 2012. Diversity of cellulolytic microbes and the biodegradation of municipal solid waste by a potential strain. Int. J. Microbiol. 2012.

Gómez-Brandón, M., Aira, M., Lores, M., Domínguez, J., 2011. Changes in microbial community structure and function during vermicomposting of pig slurry. Bioresour. Technol. 102 (5), 4171−4178.

Haei, M., Rousk, J., Ilstedt, U., Öquist, M., Bååth, E., Laudon, H., 2011. Effects of soil frost on growth, composition and respiration of the soil microbial decomposer community. Soil. Biol. Biochem. 43 (10), 2069−2077.

Herrmann, R., Shann, J., 1997. Microbial community changes during the composting of municipal solid waste. Microb. Ecol. 33 (1), 78−85.

Hoffman, E., Alimohammadi, M., Lyons, J., Davis, E., Walker, T.R., Lake, C.B., 2019. Characterization and spatial distribution of organic-contaminated sediment derived from historical industrial effluents. Environ. Monit. Assess. 191 (9), 1−19.

Hu, Y., Zheng, Q., Zhang, S., Noll, L., Wanek, W., 2018. Significant release and microbial utilization of amino sugars and D-amino acid enantiomers from microbial cell wall decomposition in soils. Soil. Biol. Biochem. 123, 115−125.

Hussain, N., Singh, A., Saha, S., Kumar, M.V.S., Bhattacharyya, P., Bhattacharya, S.S., 2016. Excellent N-fixing and P-solubilizing traits in earthworm gut-isolated bacteria: a vermicompost based assessment with vegetable market waste and rice straw feed mixtures. Bioresour. Technol. 222, 165−174.

Iqbal, H.M.N., Kyazze, G., Keshavarz, T., 2013. Advances in the valorization of lignocellulosic materials by biotechnology: an overview. BioResources 8 (2), 3157−3176.

Kamali, M., Gameiro, T., Costa, M.E.V., Capela, I., 2016. Anaerobic digestion of pulp and paper mill wastes−an overview of the developments and improvement opportunities. Chem. Eng. J. 298, 162−182.

Karmegam, N., Vijayan, P., Prakash, M., Paul, J.A.J., 2019. Vermicomposting of paper industry sludge with cowdung and green manure plants using *Eisenia fetida*: a viable option for cleaner and enriched vermicompost production. J. Clean. Prod. 228, 718−728.

Kaur, A., Singh, J., Vig, A.P., Dhaliwal, S., Rup, P.J., 2010. Cocomposting with and without *Eisenia fetida* for conversion of toxic paper mill sludge to a soil conditioner. Bioresour. Technol. 101 (21), 8192−8198.

Klamer, M., Bååth, E., 1998. Microbial community dynamics during composting of straw material studied using phospholipid fatty acid analysis. FEMS Microbiol. Ecol. 27 (1), 9−20.

Kominoski, J.S., Rosemond, A.D., Benstead, J.P., Gulis, V., Maerz, J.C., Manning, D.W., 2015. Low-to-moderate nitrogen and phosphorus concentrations accelerate microbially driven litter breakdown rates. Ecol. Appl. 25 (3), 856−865.

Méndez, A., Fidalgo, J.M., Guerrero, F., Gascó, G., 2009. Characterization and pyrolysis behaviour of different paper mill waste materials. J. Anal. Appl. Pyrolysis 86 (1), 66−73.

Nellemann, C., Corcoran, E., 2010. Dead planet, living planet: biodiversity and ecosystem restoration for sustainable development. United Nations Environment Programme (UNEP).

Nogales, R., Fernández-Gomez, M., Delgado-Moreno, L., Castillo-Díaz, J., Romero, E., 2020. Ecofriendly vermitechnological winery waste management: a pilot-scale study. SN Appl. Sci. 2 (4), 1−13.

Novais, R.M., Carvalheiras, J., Senff, L., Labrincha, J., 2018. Upcycling unexplored dregs and biomass fly ash from the paper and pulp industry in the production of eco-friendly geopolymer mortars: a preliminary assessment. Constr. Build. Mater. 184, 464−472.

Onwurah, I.N.E., Ogugua, V.N., Otitoju, O.F., 2006. Integrated environmental biotechnology-oriented framework for solid waste management and control in Nigeria. Int. J. Environ. Waste Manag. 1 (1), 94−104.

Owens, J.W., 1991. The hazard assessment of pulp and paper effluents in the aquatic environment: a review. Environ. Toxicol. Chem. An. Int. J. 10 (12), 1511−1540.

Paul, S., Kauser, H., Jain, M.S., Khwairakpam, M., Kalamdhad, A.S., 2020. Biogenic stabilization and heavy metal immobilization during vermicomposting of vegetable waste with biochar amendment. J. Hazard. Mater. 390, 121366.

Prăvălie, R., 2021. Exploring the multiple land degradation pathways across the planet. Earth-Sci. Rev. 220, 103689.

Qiao, C., Penton, C.R., Liu, C., Shen, Z., Ou, Y., Liu, Z., et al., 2019. Key extracellular enzymes triggered high-efficiency composting associated with bacterial community succession. Bioresour. Technol. 288, 121576.

Ravindran, B., Dinesh, S., Kennedy, L.J., Sekaran, G., 2008. Vermicomposting of solid waste generated from leather industries using epigeic earthworm *Eisenia fetida*. Appl. Biochem. Biotechnol. 151 (2−3), 480−488.

Rivera, J.A., López, V.P., Casado, R.R., Hervás, J.-M.S., 2016. Thermal degradation of paper industry wastes from a recovered paper mill using TGA. Characterization and gasification test. Waste Manag. 47, 225–235.

Singhvi, M.S., Gokhale, D.V., 2019. Lignocellulosic biomass: hurdles and challenges in its valorization. Appl. Microbiol. Biotechnol. 103 (23), 9305–9320.

Sinha, R.K., Agarwal, S., Chauhan, K., Valani, D., 2010. The wonders of earthworms & its vermicompost in farm production: Charles Darwin's 'friends of farmers', with potential to replace destructive chemical fertilizers. Agric. Sci. 1 (02), 76.

Spadotto, C.A., Mingoti, R., 2019. Exposure characterization tools for ecological risk assessment. Sustain. Agrochem. Compendium Technol. 321.

Sudarshan, K., Maruthaiya, K., Kotteswaran, P., Murugan, A., 2016. Reduction of pollutants in hardwood pulp mill bleach effluent. Asian J. Chem. 28 (12), 2641–2644.

Sumathi, S., Hung, Y.-T., 2006. Treatment of pulp and paper mill wastes. Waste Treat. Process. Ind. 453–497.

Takizawa, S., Asano, R., Fukuda, Y., Feng, M., Baba, Y., Abe, K., et al., 2020. Change of endoglucanase activity and rumen microbial community during biodegradation of cellulose using rumen microbiota. Front. Microbiol. 11, 3245.

Villar, I., Alves, D., Garrido, J., Mato, S., 2016. Evolution of microbial dynamics during the maturation phase of the composting of different types of waste. Waste Manag. 54, 83–92.

Villar, I., Alves, D., Mato, S., 2017. Product quality and microbial dynamics during vermicomposting and maturation of compost from pig manure. Waste Manag. 69, 498–507.

Wenning, R., Dodge, D., Peck, B., Shearer, K., Luksemburg, W., Della Sala, S., et al., 2000. Screening-level ecological risk assessment of polychlorinated dibenzo-p-dioxins and dibenzofurans in sediments and aquatic biota from the Venice Lagoon, Italy. Chemosphere 40 (9–11), 1179–1187.

Willers, C., Jansen van Rensburg, P., Claassens, S., 2015. Phospholipid fatty acid profiling of microbial communities–a review of interpretations and recent applications. J. Appl. Microbiol. 119 (5), 1207–1218.

Wong, J.W., Tyagi, R.D., Pandey, A., 2016. Current Developments in Biotechnology and Bioengineering: Solid Waste Management. Elsevier.

Yang, W., Hadibarata, T., Mahmoud, A.H., Yuniarto, A., 2020. Biotransformation of pyrene in soil in the presence of earthworm *Eisenia fetida*. Environ. Technol. Innov. 100701.

Zhao, C., Long, J., Liao, H., Zheng, C., Li, J., Liu, L., et al., 2019. Dynamics of soil microbial communities following vegetation succession in a karst mountain ecosystem, Southwest China. Sci. Rep. 9 (1), 1–10.

Zhu, X., Liu, M., Kou, Y., Liu, D., Liu, Q., Zhang, Z., et al., 2020. Differential effects of N addition on the stoichiometry of microbes and extracellular enzymes in the rhizosphere and bulk soils of an alpine shrubland. Plant. Soil. 1–17.

Chapter 8

Waste conversion into biochar: an option for sustainable valorisation

Patrick Nyambo[1], **Leocadia Zhou**[1], **Adornis D. Nciizah**[2,3] and **Hupenyu A. Mupambwa**[4]

[1]*Risk and Vulnerability Science Centre, Faculty of Science and Agriculture, University of Fort Hare, Alice, South Africa,* [2]*Agricultural Research Council—Natural Resources & Engineering, Pretoria, South Africa,* [3]*Department of Agriculture and Animal Health, University of South Africa, Roodepoort, South Africa,* [4]*Sam Nujoma Marine and Coastal Resources Research Centre, Sam Nujoma Campus, University of Namibia, Henties Bay, Namibia*

8.1 Introduction

The projected doubling in the African population from 1 billion to nearly 2.4 billion inhabitants by 2050 (Hamann and Tuinder, 2012) is expected to increase the amount of waste produced. The amount of waste production varies from place to place, but usually, the higher the level of development, the higher the amount of waste generated; however, a lot of waste material is generated annually in Africa and varies among cities and countries. For example, the total annual waste production in Cape Town, South Africa, is around two million tons, while 10 to 15 million tons are produced in Cairo, Egypt (Palczynski and Scotia, 2002). Muniafa and Otiato (2008) reported an average of 260 kg $head^{-1}$ $year^{-1}$ of waste is produced in Nairobi, Kenya. Hunger and Stretz (2006) reported an average of 182 kg $head^{-1}$ $year^{-1}$ in Maputo, Mozambique, and Collivignarelli et al. (2007) reported averages of 110−250 kg $head^{-1}$ $year^{-1}$ in Louga/Senegal. This might be too little relative to other developed countries, but the current growth and developmental rate are likely to increase waste production in the near future exponentially. The majority of African countries have inefficient storage, collection and disposal systems; thus, low waste management standards. Waste materials usually accumulate, and, more often than not, become a huge problem with direct effects on human health, safety, the environment and national/municipal budgets.

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00018-9

Waste can be liquid and/or solid, but this chapter will focus on solid and organic waste materials. According to Bello et al. (2016), solid waste is the unwanted non-liquid or nongaseous products (e.g., trash, junk and refuse) of anthropogenic activities. Other authors define solid waste as any product or substance that has no further use or value for the person or organisation that owns it and which is or will be discarded (Kolekar et al., 2016). Generally, waste is identified depending on its source, for example, municipal, industrial and agricultural solid waste. In all instances, the generated waste poses a threat to urban management, defaces the aesthetics of the country's cities and towns, and is a health hazard to citizens through the blockage of drainage systems, causing erosion and flooding. Furthermore, solid waste provides habitat and encourages the breeding of mosquitoes and is responsible for releasing greenhouse gases that cause global warming. Ensuring effective and proper solid waste management is critical to achieving the Sustainable Development Goals, especially SDG 1 (no poverty), 3n(good health and well-being), 6 (clean water and sanitation), 9 (industry, innovation and infrastructure), 11 (sustainable cities and communities) and 12 (climate action). To attain the ambitious African Union goal of recycling at least 50% of the waste generated on the continent by 2023 (United Nations Environment Programme (UNEP) Africa Waste Management Outlook, 2018), technological developments/solutions are needed to promote sustainable solid waste management.

Approximately 57% of the waste produced in Africa is organic and could be used to improve the socioeconomic conditions of its occupants significantly. Opportunities exist to recycle or convert organic solid waste into useable products that can potentially generate income and protect the environment. Several methods are available for waste disposal; however, they also have shortcomings like inefficiency, high maintenance, the inability to remove pollutants and contribution to global warming (Enaime et al., 2020). Uncontrolled landfills and open burning are the most common waste management practices in most African countries (Adesina et al., 2020) because they are supposedly cost-effective; however, they contribute a great deal to global warming and environmental pollution. The lack of awareness, infrastructure, responsible institutions and landownership further limits the recycling of organic waste materials. Converting organic waste biomass into biochar could be one sustainable and efficient waste management technology. Biochar is a carbon-rich product made from diverse biomass feedstock obtained through pyrolysis of organic biomass in an oxygen-limited environment (Steiner, 2016). During the pyrolysis process, the natural polymeric constituents, lignin, cellulose, fats and starches, are thermally broken down into three different fractions: bio-oil (condensed vapours), char (solid fraction) and noncondensable gases (Mohan et al., 2006). Various researchers have reported the effect of biochar mainly on soil (Nyambo et al., 2018, 2022), crop production (Nyambo et al., 2020a) and wastewater treatment (Obey et al., 2022; Li et al., 2022a,b). The objective of this chapter was to review the potential biochar in organic waste valorisation in Africa.

8.2 Challenges associated with solid waste in Africa

A large percentage of solid waste in Africa is organic, and because of the poor and uncontrolled waste management systems, solid waste dumping causes significant economic, social and environmental impacts (United Nations Environment Programme (UNEP) Africa Waste Management Outlook, 2018). The amount of organic waste varies among African countries (Fig. 8.1). Disposal of organic waste in landfills or open dumps results in a leakage of toxic substances into the environment, underground and above-ground water bodies. Dumpsites provide a habitat for the incubation and proliferation of insects and rodents, vectors of diseases like cholera, malaria, typhoid fever, etc. (Abul, 2010). In some cases, the waste materials can cause blockage of the drainage systems, thus causing flash flooding (Lamond et al., 2012).

Mismanagement of solid wastes can also be a source of global warming causing greenhouse gases. The anaerobic decomposition of organic waste materials releases methane and carbon dioxide into the atmosphere, thus contributing to climate change. Furthermore, the land on which the dumpsites are located becomes inhabitable for decades as the organic materials require decades for complete biodegradation (Medina, 2009). Accordingly, it becomes even more imperative to look at sustainable solutions for dealing with solid waste issues.

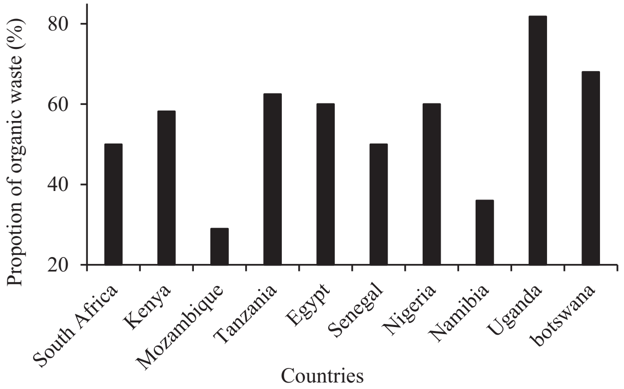


FIGURE 8.1 Proportion of organic waste produced in some African countries (Couth and Trois, 2010). Source*: Adapted from Couth, R., Trois, C., 2010. Carbon emissions reduction strategies in Africa from improved waste management: a review. Waste Manag., 30(11), 2336–2346.*

8.3 Type of waste products

Waste products are classified according to their source, and in this chapter, solid wastes are put into three main categories, that is, agriculture, municipal and industrial waste (Table 8.1).

8.3.1 Municipal waste

According to OECD (2022), waste materials collected and treated by or for municipalities are termed municipal waste.Municipal solid waste consists of various materials ranging from food waste, sewage sludge, furniture, product packaging, grass and tree clippings, etc. For most, this waste is garbage people throw away after use. Many waste materials are discarded from residential, institutional and industrial sources. Kumar et al. (2021) identified food waste and sewage sludge as the two major waste streams in municipal solid waste disposal, which require proper handling to be appropriately handled to promote a sustainable environment. Food waste accounts for approximately 32%–62% of the MSW and is anticipated to rise because of rapid urbanisation and population growth worldwide (Xu et al., 2018).

Municipalities in many African countries dispose of municipal waste in open landfills and/or open burning. Efforts to reduce reliance on landfills to dispose of municipal waste have been unsuccessful for various reasons, including a lack of financing, proper planning and corruption, weak legislation and lack of enforcement, lack of public sensitisation, political instability and lack of political will (Godfrey et al., 2019). According to Mamera et al. (2021),

TABLE 8.1 Types of solid waste.

Source	Typical waste generator	Types of solid wastes
Agricultural waste	Crops, orchards, vineyards, dairies, feedlots, farms. forestry	Food and meat processing solid wastes, waste silage, hazardous wastes (e.g., pesticides), logs, food and meat processing solid wastes, waste silage and green waste.
Municipal waste	Street cleaning, landscaping, parks, beaches, other recreational areas, and water and wastewater treatment plants.	Street sweepings; landscape and tree trimmings; general wastes from parks, beaches and other recreational areas; sewage sludge.
Industrial waste	Food production plants, mines, electrical power and chemical plants.	Housekeeping wastes, packaging, food wastes, construction and demolition materials, hazardous wastes, ashes and special wastes.

shortages in suitable sanitation, poor drainage, and groundwater fluctuations are some of the reasons councils are not emptying pits. For example, in South Africa, the government made the 2001 Polokwane Declaration, which set a target reduction of landfills by 50% by 2021 and 100% by 2022 (Department of Environmental Affairs and Tourism DEAT, 2001). Technological advancement is the way to reduce such disposal methods like landfills.

8.3.2 Agricultural waste

Agricultural solid wastes are produced mainly from farming activities; however, it is not limited to the production but other activities associated with farming and the food chain. The need to feed the ballooning African population instigated increased agricultural production and, consequently, high waste generation (Tripathi et al., 2019). Organic waste from agriculture includes plant residues, animal excreta and other residues from animals. Much of the agricultural wastes are either left in the field or burnt in situ, potentially contributing to the release of greenhouse gases and or environmental pollution. Therefore, it is necessary for waste valorisation by converting it into other by-products, like biochar, which can potentially create employment opportunities, reduce gas emissions and contribute to sustainable agriculture (Nyambo et al., 2020b).

8.4 Biochar

Biochar has attracted interest among researchers as a technology for sequestering carbon, mitigating greenhouse gases and improving soil quality, thus reducing the problems and causes of climate change (Nyambo et al., 2022). Verheijen et al. (2010) describe biochar as a relatively stable C compound created through the heating of biomass (feedstock) at temperatures usually between 300°C and 1000°C, under low (preferably zero) oxygen concentrations. Various techniques used to produce biochar are presented in Fig. 8.2.

Organic materials for the feedstock can be either agricultural wastes (Nyambo et al., 2020a) or municipal or industrial wastes (Zhang et al., 2016). Production can occur at both small and industrial scales. The quality and use of biochar are, however, dependent on the type of biochar used. For example, feedstocks with high levels of toxic substances, such as heavy metals from sewage sludge and industrial waste, are discouraged from being used for making biochar (International Biochar Initiative IBI, 2011).

The key properties such as carbon content, specific surface area and micropores presence contribute to the adsorptive properties of biochar. The biochars' intricate porous structure exposes more active sites for microbial activity and adsorption of organic pollutants, thus enabling catalytic degradation.

The carbon in biochar is mainly found in poly-condensed aromatic structures, which are responsible for the biochar's stability and facilitate long-term

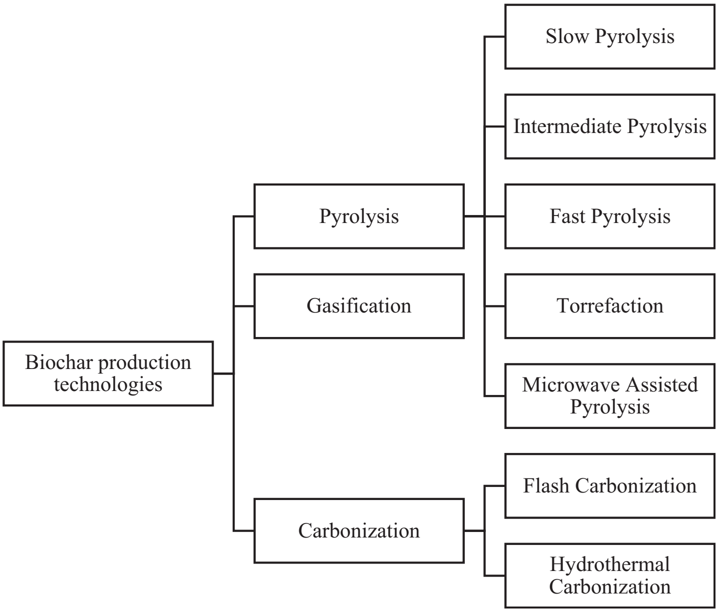


FIGURE 8.2 Biochar production technologies. Source*: Adapted from Gunarathne, V., Ashiq, A., Ramanayaka, S., Wijekon P., Vithanage M., 2019. Biochar from municipal solid waste for resource recovery and pollution remediation. Env. Chem. Lett. 17, 1225–1235. https://doi.org/10.1007/s10311-019-00866-0.*

C sequestration (Steiner, 2016). Biochar can potentially reduce 870 kg CO_2-equivalent of greenhouse gases per ton of dry feedstock, which also means a 12% drop in annual global anthropogenic emission (Roberts et al., 2010). The higher reactivity of biochar surfaces is due to the presence of a range of reactive functional groups (siloxane, OH, COOH, CO, CeO, N), and some are pH-dependent (Roy et al., 2022). Copiousness of small to medium-sized pores in biochar has been discovered to enhance the surface area of materials and with their stability, are keys to ecosystem functioning. Lee et al. (2013) reported that biochar produced from wood and bagasse was found to have pores of various sizes and large surface areas.

The porous structure derived from nanohydroxyapatite self-template is favourable for the catalytic reaction (Alhokbany et al., 2019). In addition, electron transfer is an essential process in the catalytic reaction (Yu et al., 2015). Wang et al. (2015) studied the electrocatalytic performance of fish-bone carbon in the oxygen reduction reaction, showing high electron transfer capacity.

The production of biochar from organic materials is an eco-friendly option as it uses all the waste materials effectively and productively, which successfully helps in reducing the extent of environmental pollution.

8.5 Valorisation of solid waste through the production of biochar

Various studies have highlighted the valorisation of solid waste through biochar production (Table 8.2). The resultant biochar can be an eco-friendly alternative to fossil fuels for cleaner energy production and be helpful in the construction industry and removing contaminants from aqueous environments such as polluted soil, water and waste effluents. Choi et al. (2012) reported replacing cement with biochar 5% (w/w) led to a 12% increase in the strength of a biochar-mortar composite compared to control mortar. Therefore, using biochar converted from organic wastes in construction effectively manages solid organic wastes and stores the carbon in the built environment (Dixit et al., 2019). Furthermore, it can result in savings as it reduces the demand for more cement during construction.

The conversion of organic wastes to biochar can be a renewable energy source. Biochar can be used in heating both at industrial and domestic levels. This is important given that fossil fuel sources are rapidly getting depleted. According to Qambrani et al. (2017), the use of biochar is eco-friendly since no net emissions are released into the atmosphere; only the amount formed during the pyrolysis process is equal to that taken up by the plant during photosynthesis. Furthermore, converting organic waste to biochar reduces the gases released into the atmosphere since the char emits less SOx and NOx than conventional fossil fuels (Kim et al., 2019).

8.5.1 Municipal waste into biochar

There has been a steady increase in work on biochar production from municipal soil waste (MSW) over the past few years. Most of this work has been done in Europe. Africa is still lagging despite the huge potential benefits of this waste management approach. This is especially important given the huge volumes of MWS generated annually. Municipal solid waste can be pyrolysed into biochar and used in agriculture as a soil amendment. Other important uses include green adsorbent for leachate treatment, a permeable reactive barrier material to reduce contamination, and deodorant (Gunarathne et al., 2019). Municipal waste biochar is converted into biochar through pyrolysis, carbonisation and gasification. These thermochemical processes are more efficient than older traditional charcoal production technologies that were not energy efficient (Gunarathne et al., 2019).

The quality of MSW's biochar depends on feedstocks' characteristics, peak temperature and pressure during pyrolysis. Several authors have summarised the characteristics of MSW (Li et al., 2022a,b; Gunarathne et al., 2019). MWS biochar from characteristics from these studies is summarised in Table 8.3. Generally, an increase in pressure and a decrease in

TABLE 8.2 Implications of converting waste into biochar.

Type of waste material	Purpose of biochar	Benefits	Reference
Mixed wood sawdust	biochar was used to replace cement in Ultra-high performance concrete mix at 5% (w/w)	sequesters carbon and reduces costs by reducing the demand for cement.	Dixit et al. (2019)
Invasive species biomass	Biochar was used to test for bioenergy and other value-added applications	The high pH of the biochar makes it a potential soil conditioner. Biochar is a potential energy source; however, the heating values depend on the temperature used during pyrolysis.	Ahmed et al. (2020)
fishbone biowaste	Biochar was used as a persulfate activator for phenol removal.	Biochar saves money by removing toxic compounds as it is cheaper to make than other carbon-based catalysts.	Ren et al. (2021)
Maize residues	Soil amendment	carbon sequestration and soil conditioner	Nyambo et al. (2018)
Pinewood	Phenolic wastewater treatment	removal of toxic compounds.	Kamali et al. (2022)
Matamba shell	Adsorbent for wastewater treatment	Matamba fruit shells have the potential to be used as an eco-friendly and low-cost effective adsorbent for anionic dye removal from the water environment.	Obey et al. (2022)
Sewage sludge	Fuel gas to sustain the pyrolysis process	reduces environmental damage by reducing gas emissions, and heavy metals were strongly immobilised within the biochar	Liu et al. (2018)

temperature results in an increase in biochar yield. Slow pyrolysis at lower temperatures has a high biochar yield of up to 35%, whereas fast pyrolysis at a temperature close to 1000°C has low biochar production but produces bio-oil as a major product (Gunarathne et al., 2019).

TABLE 8.3 Comparison of properties of MSW biochar under different thermal conditions.

Thermochemical process	Pyrolysis temperature (°C)	Mobile matter (%)	Fixed matter (%)	Ash (%)	C (%)	H (%)	O (%)	N (%)	Surface area ($m^2\ g^{-1}$)	Pore volume ($cm^3\ g^{-1}$)
Slow pyrolysis	400–700	6.3–31.6	12.6–78.2	6.1–78.2	15.7–83.8	0.7–12.2	1.9–75.75	1.0–6.57	20.7–380.9	0.013–0.15
Hydrothermal Carbonisation	280	74.2	–	12.5	41.7	5.3	40.1	0.4	–	
Pyrolysis (2 h)	300–750	–	–	10.1–20.6	40.7–61.2	0.01–5.2	38.3–53.6	0.3–0.7	–	
Pyrolysis (4 h)	300–750	–	–	10.3–30.1	45.1–62.5	0.00–4.6	37–49.7	0.5–0.6	5–10	
Pyrolysis (6 h)	300–750	–	–	10.5–30.1	48.4–60.5	0.1–4.5	38.9–46.6	0.5–0.6	140–160	

Adapted from Gunarathne, V., Ashiq, A., Ramanayaka, S., Wijekon, P., Vithanage, M., 2019. Biochar from municipal solid waste for resource recovery and pollution remediation. Env. Chem. Lett. 17, 1225–1235. https://doi.org/10.1007/s10311-019-00866-0.

8.6 Uses of municipal solid waste biochar

Municipal waste biochar has several important environmental and agricultural uses. The most important is reducing landfill volume, especially considering the huge volumes of MSW produced annually. Ecological benefits of MSW include adsorbent for leachate treatment, a permeable reactive barrier material to reduce contamination, deodorant and landfill capping. The use of MSW biochar as leachate treatment in landfills stems from its high adsorption, high surface area and microporous nature. Studies have shown that MSW biochar effectively removes numerous toxic heavy metals and NH3-N (Jayawardhana et al., 2016). The high surface area and adsorption capacity of MWS biochar also provide a huge opportunity for improving the functionality of landfill capping materials. Using MSW biochar in landfill capping materials improves removal rates of CH4 by between 60% and 90% (Gunarathne et al., 2019; Jayawardhana et al., 2016). In addition, MWS biochar enhances the performance of vegetation-based landfill cover due to its improvement of soil's physical and chemical properties. MSW biochar is a good soil amendment like any other biochar due to its high organic carbon content and aromaticity (Gunarathne et al., 2019). Improved crop yield after the application of MSW biochar is due to improved fertility and soil conditions, as well as reduced uptake of harmful elements, which get adsorbed to the biochar. Randolph et al. (2017) studied the effects of MWS biochar on soil quality parameters and reported improvements in soil fertility parameters. In this study, the use of MSW biochar increased Ca, Mg, K and P and higher water retention. Moreover, the leachate from soils amended with MSW biochar had higher pH and electrical conductivity. In this study, however, pyrolysis conditions and feedstock type significantly affected the findings. Hence there is a need for more localised studies.

8.7 Potential of industrial waste as biochar raw materials

Industrial wastes go through a similar pyrolysis process, as shown in Fig. 8.2; however, they can be distinguished from other solids wastes by their relatively high amounts of metallic components (Kwon et al., 2020). Using industrial wastes to produce biochar can take advantage of the specific functionality of metals present in the wastes. Waste materials are mixed with biomass feedstock and co-pyrolyzed to impregnate metallic components into a biochar surface, resulting in engineered biochar. Biochar engineering allows for the tailoring of biochar properties for specific applications and/or under specific conditions (Awasthi, 2022). This would lead to harnessing the favourable features of biochar and enhancing its efficiency. For example, Zhou et al. (2013) synthesised chitosan-modified biochar as a low-cost adsorbent for heavy metal remediation. Furthermore, some industrial wastes can be used to create biochar that has specific adsorption advantages. For instance, Cho et al. (2017)

reported that biochar produced from pyrolysis of industrial wastes containing Fe and Ca (e.g., paper mill sludge or red mud) showed enhanced performances in the adsorption of As(V), Cd(II).

8.8 Potential of agricultural waste as biochar raw materials

Because of the rapid increase in the world population, agriculture has intensified, resulting in the generation of huge quantities of waste that include animal manure and crop residues. From time immemorial, animal manures have, however, been used as sources of soil nutrients and to improve soil quality when applied directly. There is now a growing realisation of the potential environmental challenges that continuous untimed applications of animal manure into soils can cause. Coupled with this is the advent of conservation agriculture, which has increased the demand for plant biomass for soil cover, which has been a significant challenge for smallholder farmers who use this biomass as livestock feed. Because of the challenges caused by the untimed application of manure and the shortage of biomass to apply to soils, there is a growing call for the conversion of these materials into biochar. Converting these materials into biochar creates recalcitrant carbon forms, thus allowing the manures and plant biomass to sequester carbon into the soil more effectively. Conversion of these agricultural wastes has been reported to generate biochar with varying physiochemical properties, and research that has focused on this is summarised in Table 8.4.

TABLE 8.4 A summary of research done on preparing biochar using different agro-based wastes and the general characteristics of the pyrolyzed material.

Agricultural waste	Pyrolysis conditions	Physiochemical properties observed	Reference
Maize cobs, straw and oreganum stalks	500°C in a fluidised bed reactor.	Corn cobs and oreganum stalks yielded all 23% of biochar, with straw yielding 20% of biochar on a weight basis. The pyrolysis process also yielded potentially important gases, which can be used as fuel.	Yanik et al. (2007)

(*Continued*)

TABLE 8.4 (Continued)

Agricultural waste	Pyrolysis conditions	Physiochemical properties observed	Reference
Cherry seed and cherry seed shell	Pyrolyzed in a fixed-bed design at a temperature of 500°C	Biochar yields were 21 and 15 wt.% for cherry seed and cherry seed shell, respectively. The bio-oil yield was a maximum of 44%, indicating great potential for the use of this oil as fuel for combustion systems.	Duman et al. (2011)
Beach cast seaweeds *Laminaria pallida and Gracilariopsis funicularis*	Pyrolysis was done at different temperatures ranging from 200°C, and 800°C and the physical and chemical properties of the biochar were determined.	Temperatures of 400°C resulted in an almost 50% reduction in biochar yield. G. funicularis biochar has the highest concentration of macro-nutrients, while L. pallida had the highest cations, all indicating the fertilizer value of these biochars in agriculture.	Katakula et al. (2020)
Beach collected *Laminaria digitate, Fucus ser-ratus* seaweeds and other lignocellulosic biomass.	A continuous fluidised bed reactor operated at 500°C, and pyrolysis products (char, liquid and gas) were quantified.	The yields of oil from the seaweeds varied from 11% to 17%, with yields of biochar ranging from 29% to 36% for seaweeds and 23%–26% for lignocellulosic biomass.	Yanik et al. (2013)
Agricultural wastes, that is, jatropha, date stone, olive stone, peach stone and Spirulina microalgae species, were used as feed materials.	A temperature-controlled tubular furnace was used at 500°C.	Biochar yields varied among the agricultural wastes ranging from 50% to 33%. Also, the biochar had varying concentrations of carbon from 76% to 43%.	

8.9 Conclusions

The conversion of organic waste into biochar presents an opportunity to increase the value of most organic waste. This is also important in the face of climate change, where carbon sequestration is among the critical processes required in mitigation against climate change. Carbon from biochar has been reported to be a recalcitrant type, which is stored longer in the soil. Apart from biochar, the pyrolysis process also yields gasses and oils that have the potential for use as a source of energy.

Acknowledgments

This work is based on the research supported wholly by the National Research Foundation (NRF) of South Africa and the Dutch Research Council (de Nederlandse Organisatie voor Wetenschappelijk Onderzoek—NWO) Project UID 129352. The NRF and NWO are thanked for their financial contribution. Any opinion, finding, conclusion, or recommendation expressed in this material is that of the author(s), and the NRF and NWO do not accept any liability in this regard.

References

Abul, S., 2010. Environmental and health impact of solid waste disposal at Mangwaneni dumpsite in Manzini: Swaziland. J. Sustain. Dev. Afr. 12 (7), 64–78.

Adesina, O.A., Adesina, A.O., Adeniran, J.A., 2020. Level of polycyclic aromatic hydrocarbon emission in the ambient air and residual ash from a typical municipal solid waste open burning site in Nigeria. Waste Dispos. Sustain. Energy 2 (2), 105–111.

Ahmed, A., Bakar, M.S.A., Hamdani, R., Park, Y.K., Lam, S.S., Sukri, R.S., et al., 2020. Valorisation of underutilised waste biomass from invasive species to produce biochar for energy and other value-added applications. Environ. Res. 186, 109596.

Alhokbany, N., Ahama, T., Naushad, M., Alshehri, S.M., 2019. AgNPs embedded N-doped highly porous carbon derived from chitosan based hydrogel as catalysts for the reduction of 4-nitrophenol. Compos. Part B: Eng. 173, 106950.

Awasthi, M.K., 2022. Engineered biochar: a multifunctional material for energy and environment. Environ. Pollut. 298, 118831.

Bello, I.A., bin Ismail, M.N., Kabbashi, N.A., 2016. Solid waste management in Africa: a review. Int. J. Waste Resour 6 (2), 1–4.

Cho, D.-W., Kwon, G., Yoon, K., Tsang, Y.F., Ok, Y.S., Kwon, E.E., Song, H., 2017. Simultaneous production of syngas and magnetic biochar via pyrolysis of paper mill sludge using CO2 as reaction medium. Energy Convers. Manage 145, 1–9.

Choi, W.C., Do Yun, H., Lee, J.Y., 2012. Mechanical properties of mortar containing biochar from pyrolysis. J. Korea Inst. Struct. Maint. Insp. 16, 67–74.

Collivignarelli, C., Mazzu, A., Sorlini, S., 2007. Appropriate technologies for energy recovery from biomass in Africa. In: Sardinia 2007 Eleventh International Waste Management and Landfill Symposium, October 2007, Santa Margherita di Pula, Cagliari, Sardinia, Italy.

Couth, R., Trois, C., 2010. Carbon emissions reduction strategies in Africa from improved waste management: a review. Waste Manag. 30 (11), 2336–2346.

Department of Environmental Affairs and Tourism (DEAT), 2001. Polokwane Declaration on Waste Management. First National Waste Management Summit held in Polokwane, Northern Province. South Africa.

Dixit, A., Gupta, S., Dai Pang, S., Kua, H.W., 2019. Waste valorisation using biochar for cement replacement and internal curing in ultra-high performance concrete. J. Clean. Prod. 238, 117876.

Duman, G., Okutucu, C., Ucar, S., Stahl, R., Yanik, J., 2011. The slow and fast pyrolysis of cherry seed. Bioresour. Technol 102 (2), 1869−1878. Available from: https://doi.org/10.1016/j.biortech.2010.07.051. ISSN 0960−8524.

Enaime, G., Baçaoui, A., Yaacoubi, A., Lübken, M., 2020. Biochar for wastewater treatment—conversion technologies and applications. Appl. Sci. 10 (10), 3492.

Godfrey, L., Ahmed, M.T., Gebremedhin, K.G., Katima, J.H., Oelofse, S., Osibanjo, O., et al., 2019. Solid waste management in Africa: governance failure or development opportunity. Regional Dev. Afr. 235.

Gunarathne, V., Ashiq, A., Ramanayaka, S., Wijekon, P., Vithanage, M., 2019. Biochar from municipal solid waste for resource recovery and pollution remediation. Env. Chem. Lett. 17, 1225−1235. Available from: https://doi.org/10.1007/s10311-019-00866-0.

Hamann, M., Tuinder, V., 2012. Introducing the Eastern Cape: A Quick Guide to Its History, Diversity and Future Challenges. Stockholm Resilience Centre, Stockholm University, Stockholm.

Hunger, G., Stretz, J., 2006. 'Proof of Service' − an innovative multifunctional tool for sustainable solid waste management strategies in developing countries. In: WasteCon 2006, Eighteenth Waste Management Conference and Exhibition, October 2006, Cape Town, South Africa.

International Biochar Initiative (IBI), 2011. Feedstocks. Available from: http://www.biochar-international.org/technology/feedstocks. (accessed 03.10.18.).

Jayawardhana, Y., Kumarathilaka, P., Herath, I., Vithanage, M., 2016. Municipal solid waste biochar for prevention of pollution from landfill leachate. In: Prasad, M.N.V., Shih, K. (Eds.), Environmental Materials and Waste. Resources Recovery and Pollution Prevention. Academic Press, Cambridge, MA, pp. 117−148.

Kamali, M., Aminabhavi, T.M., Tarelho, L.A., Hellemans, R., Cuypers, J., Capela, I., et al., 2022. Acclimatised activated sludge for enhanced phenolic wastewater treatment using pinewood biochar. Chem. Eng. J. 427, 131708.

Katakula, A.A.N., Gawanab, W., Itanna, F., Mupambwa, H.A., 2020. The potential fertilizer value of Namibian beach-cast seaweed (Laminaria pallida and Gracilariopsis funicularis) biochar as a nutrient source in organic agriculture. Sci. Afr 10, e00592. Available from: https://doi.org/10.1016/j.sciaf.2020.e00592. ISSN 2468-2276.

Kim, S., Tsang, Y.F., Kwon, E.E., Lin, K.A., Lee, J., 2019. Recently developed methods to enhance stability of heterogeneous catalysts for conversion of biomass-derived feedstocks. Kor. J. Chem. Eng. 36, 1−11.

Kolekar, K.A., Hazra, T., Chakrabarty, S.N., 2016. A review on prediction of municipal solid waste generation models. Procedia Environ. Sci. 35, 238−244.

Kumar, M., Dutta, S., You, S., Luo, G., Zhang, S., Show, P.L., et al., 2021. A critical review on biochar for enhancing biogas production from anaerobic digestion of food waste and sludge. J. Clean. Prod. 305, 127143.

Kwon, G., Bhatnagar, A., Wang, H., Kwon, E.E., Song, H., 2020. A review of recent advancements in utilization of biomass and industrial wastes into engineered biochar. J. Hazard. Mater 400, 123242.

Lamond, J., Bhattacharya, N., Bloch, R., 2012. The role of solid waste management as a response to urban flood risk in developing countries, a case study analysis. WIT Trans. Ecol. Environ. 159, 193–204.

Lee, Y., Park, J., Ryu, C., Gang, K.S., Yang, W., Park, Y.K., et al., 2013. Comparison of biochar properties from biomass residues produced by slow pyrolysis at 500 C. Bioresour. Technol. 148, 196–201.

Li, N., He, M., Lu, X., Yan, B., Duan, X., Chen, G., et al., 2022a. Municipal solid waste derived biochars for wastewater treatment: production, properties and applications. Resources, Conserv. Recycling 177, 106003.

Li, R., Wang, B., Niu, A., Cheng, N., Chen, M., Zhang, X., et al., 2022b. Application of biochar immobilised microorganisms for pollutants removal from wastewater: a review. Sci. Total. Environ. 155563.

Liu, X., Chang, F., Wang, C., Jin, Z., Wu, J., Zuo, J., et al., 2018. Pyrolysis and subsequent direct combustion of pyrolytic gases for sewage sludge treatment in China. Appl. Therm. Eng. 128, 464–470.

Mamera, M., van Tol, J.J., Aghoghovwia, M.P., Nhantumbo, A.B., Chabala, L.M., Cambule, A., et al., 2021. Potential use of biochar in pit latrines as a faecal sludge management strategy to reduce water resource contaminations: a review. Appl. Sci. 11 (24), 11772.

Medina, M., 2009. Municipal solid waste management in third world cities: lessons learned and a proposal for improvement. Hum. Settl. Dev. 3, 264.

Mohan, D., Pittman, C.U., Steele, P.H., 2006. Pyrolysis of wood/biomass for bio-oil: a critical review. Energy Fuels 20 (3), 848–889.

Muniafa, M., Otiato, E., 2008. Solid waste management in Nairobi, Kenya. A case for emerging economics. In: WasteCon 2008, Nineteenth Waste Management Conference and Exhibition, October 2008, Cape Town, South Africa.

Nyambo, P., Taeni, T., Chiduza, C., Araya, T., 2018. Effects of maise residue biochar amendments on soil properties and soil loss on acidic Hutton soil. Agronomy 8 (11), 256.

Nyambo, P., Chiduza, C., Araya, T., 2022. Effect of conservation agriculture on selected soil physical properties on a haplic cambisol in Alice, Eastern Cape, South Africa. Arch. Agron. Soil Sci. 68 (2), 195–208.

Nyambo, P., Mupambwa, H.A., Nciizah, A.D., 2020a. Biochar enhances the capacity of climate-smart agriculture to mitigate climate change. Handbook of Climate Change Management: Research, Leadership, Transformation 1–18.

Nyambo, P., Chiduza, C., Araya, T., 2020b. Carbon input and maise productivity as influenced by tillage, crop rotation, residue management and biochar in a semiarid region in South Africa. Agronomy 10 (5), 705.

Obey, G., Adelaide, M., Ramaraj, R., 2022. Biochar derived from non-customised matamba fruit shell as an adsorbent for wastewater treatment. J. Bioresour. Bioprod. 7 (2), 109–115.

OECD, 2022. Municipal waste (indicator). https://doi.org/10.1787/89d5679a-en (accessed 11.06.22.).

Palczynski, R.J., Scotia, W.N., 2002. Study on solid waste management options for Africa. African Development Bank, 131.

Qambrani, N.A., Rahman, M.M., Won, S., Shim, S., Ra, C., 2017. Biochar properties and eco-friendly applications for climate change mitigation, waste management, and wastewater treatment: a review. Renew. Sustain. Energy Rev. 79, 255–273.

Randolph, P., Bansode, R.R., Hassan, O.A., Rehrah, D.J., Ravella, R., Reddy, M.R., et al., 2017. Effect of biochars produced from solid organic municipal waste on soil quality parameters. J. Environ. Manage. 192, 271–280.

Ren, X., Wang, J., Yu, J., Song, B., Feng, H., Shen, M., et al., 2021. Waste valorisation: transforming the fishbone biowaste into biochar as an efficient persulfate catalyst for degradation of organic pollutant. J. Clean. Prod. 291, 125225.

Roberts, K.G., Gloy, B.A., Joseph, S., Scott, N.R., Lehmann, J., 2010. Life cycle assessment of biochar systems: estimating the energetic, economic, and climate change potential. Environ. Sci. Technol. 44, 827−833. Available from: https://doi.org/10.1021/es902266r.

Roy, H., Islam, M.S., Arifin, M.T., Firoz, S.H., 2022. Synthesis, characterization and sorption properties of biochar, chitosan and ZnO-based binary composites towards a cationic dye. Sustainability 14 (21), 14571.

Steiner, C., 2016. Considerations in biochar characterisation. Agric. Environ. Appl. Biochar Adv. Barriers (sssaspecpub63), 87−100.

Tripathi, N., Hills, C.D., Singh, R.S., Atkinson, C.J., 2019. Biomass waste utilisation in low-carbon products, harnessing a major potential resource. Clim. Atmos. Sci. 2 (1), 1−10. Available from: https://doi.org/10.1038/s41612-019-0093-5.

United Nation Environment Programme (UNEP) Africa Waste Management Outlook, 2018. United Nations Environment Programme, Nairobi, Kenya (2021) https://www.unep.org/ietc/resources/publication/africa-waste-management-outlook. (accessed 11.06.22.).

Verheijen, F., Jeffery, S., Bastos, A.C., Van der Velde, M., Diafas, I., 2010. Biochar application to soils. A critical scientific review of effects on soil properties, processes, and functions. Environ. Sci. 24099 (162), 2183−2207.

Wang, H., Wang, K., Song, H., Li, H., Ji, S., Wang, Z., et al., 2015. N-doped porous carbon material made from fish-bones and its highly electrocatalytic performance in the oxygen reduction reaction. RSC Adv 5 (60), 48965−48970.

Xu, F., Li, Y., Ge, X., Yang, L., Li, Y., 2018. Anaerobic digestion of food waste e challenges and opportunities. Bioresour. Technol. 247, 1047−1058.

Yanik, J., Kornmayer, C., Saglam, M., Yüksel, M., 2007. Fast pyrolysis of agricultural wastes: characterization of pyrolysis products. Fuel Process. Technol 88 (2007), 942−947.

Yanik, J., Stahl, R., Troeger, N., Sinag, A., 2013. Pyrolysis of algal biomass. J. Anal. Appl. Pyrolysis 103, 134−141. Available from: https://doi.org/10.1016/j.jaap.2012.08.016. ISSN 0165−2370.

Yu, L., Yuan, Y., Tang, J., Wang, Y., Zhou, S., 2015. Biochar as an electron shuttle for reductive dechlorination of pentachlorophenol by Geobacter sulfurreducens. Sci. Rep. 5 (1), 16221.

Zhang, D., Yan, M., Niu, Y., Liu, X., van Zwieten, L., Chen, D., et al., 2016. Is current biochar research addressing global soil constraints for sustainable agriculture? Agric. Ecosyst. Environ. 226, 25−32.

Zhou, Y., Gao, B., Zimmerman, A.R., Fang, J., Sun, Y., Cao, X., 2013. Sorption of heavy metals on chitosan-modified biochars and its biological effects. Chem Eng J 231, 512−518.

Chapter 9

Solid waste management: challenges and health hazards

Robin[1], **Pardeep Kaur**[2], **Jagdeep Kaur**[3] **and Sartaj Ahmad Bhat**[4]

[1]Agilent Technologies India Pvt. Ltd. & Regional Water Testing Laboratory, Department of Water Supply and Sanitation, Government of Punjab, Amritsar, Punjab, India, [2]Department of Botany, Khalsa College, Amritsar, Punjab, India, [3]Department of Botanical and Environmental Sciences, Guru Nanak Dev University, Amritsar, Punjab, India, [4]River Basin Research Center, Gifu University, Gifu, Japan

9.1 Introduction

Solid waste management (SWM) is an organised system of waste management that ensures ecological safety by reducing environmental and health risks. As the world's population has grown and natural resources have been depleted, waste generation has, however, skyrocketed. Workers in the waste management industry have been demonstrated to have a higher accident rate and musculoskeletal ailments. Numerous waste treatment processes release a wide variety of chemicals, the majority of which are in minuscule amounts and at relatively small levels. Moreover, SWM is an integrated process of several waste optimisation regimes involving technical, socioeconomic and environmental parameters. These multifaceted nonlinear approaches make conventional modelling, predicting and optimising techniques more challenging. Swift urbanisation, unprecedented rise in population and financial improvement have brought extended production of solid waste in nations throughout the world. Ongoing measurements show that 2.01 billion tons of municipal solid waste are being produced annually, which is estimated to grow to 3.40 billion tons by 2050 (Kaza et al., 2018). Out of this, 33% of the waste produced is perilously taken care of, with the waste being dumped in an illegitimate way or unmanaged landfills (Kaza et al., 2018). These ill-fated practices epitomise numerous ecological and health concerns, including groundwater contamination, land decay, lingering cancer growth frequency, infant mortality and birth oddities (Triassi et al., 2015; Shyam et al., 2022).

SWM is an all-inclusive issue influencing everyone on the planet. Studies have shown that a lack of strategic planning and insufficient actions are

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00002-5

significant reasons for miserable waste management (Hannan et al., 2013; Yukalang et al., 2017). Direct or indirect emissions of greenhouse gases (GHG) have been linked to waste treatment, disposal regime and recovery. Unduly managed waste can lead to exacerbated GHG emissions (United Nations Environment Programme, 2010). Management of solid waste is a widespread issue affecting each individual in the world. Every person's and governmental decisions about resource usage and waste monitoring affect the day-to-day health, productivity and personal hygiene of communities. Most governments in the world recognise the need for an effective and long-term waste management system that considers the possible effects on human health and the environment. This chapter explores the impact on human health through various waste management regimes. The following chapter is not intended to be a complete analysis of every waste management approach's health effect but rather to highlight the spectrum of potential advantages and drawbacks of the major management strategies backed by evidence from the literature.

9.1.1 Solid waste and its characterisation

Solid wastes include sludge from various treatment plant facilities, as well as unwanted rejected materials from industries, commercial and community activities, mining set-ups and agricultural processes (United States Environment Protection Agency, 2022). These wastes can be nonhazardous (garbage, trash, sludge, municipal waste), hazardous (chemical sludges, heavy metals, solvents, pesticide/insecticide residues, incinerator ash, corrosive acids, coating/plating solutions), radioactive (natural, anthropogenic) or mixed. Domestic waste makes up 67.8% of the composition of solid waste in urban areas, and commercial waste makes up 23.5%, whereas domestic waste makes up 92.4% of the composition of solid waste in rural areas. Urban solid waste is composed of 50% decomposable waste and 50% non-putrescible waste, whereas rural solid waste is composed of 63.5% degradable waste and 36% non-putrescible waste. A significant issue in rural areas is open disposal, which is used by about 78% of the population there to store and collect solid waste (Boateng et al., 2016; Patwa et al., 2020).

9.1.2 Solid waste management practices

The main motive of solid waste treatment procedures is to recover and use as many of the components found in the discarded wastes as a resource as feasible while lowering the quantity of solid waste that has to be landfilled. Different techniques are used to handle solid waste, and the best technique to use depends on the qualities of the trash components, the amount of land that is available for treatment and the expense of disposal of waste. A typical solid waste management regime involves waste recycling, heat treatment (incineration/pyrolysis), landfill and composting. A number of undesirable domestic or industrial solid materials constitute solid wastes. Various types of solid waste include rubbish/garbage, animal/agricultural waste, treated wastewater sludge, ash, industrial waste, hazardous waste, construction/demolition waste, etc. Food and green wastes are the primary waste category on a worldwide scale, accounting for 44% of total waste, and about 38% of waste comprises recyclable dry materials like paper and cardboard, plastic, glass and metals (Kaza et al., 2018). Most industrialised nations today strictly regulate waste management, which encompasses waste creation, collection, processing, transportation and disposal. Furthermore, waste site remediation is an important concern, both to decrease dangers while the plant is functional and to adapt the site for a transition of its use.

9.2 Impact of waste management practices

Solid waste can be managed by commonly used techniques/methods like waste recycling, incineration/pyrolysis, landfilling and composting/vermicomposting, with each method having its own merits and demerits (Table 9.1). Moreover, any waste management approach emits GHGs, both directly (increase in emission levels from the processing) and indirectly (through the consumption of energy). Local officials have failed to reach out to the community and educate individuals on the fundamentals of waste management, including basic methods for keeping garbage in their own dumpsters at the home, store and enterprise levels. Because of the lack of a basic infrastructure for trash collection at the source, inhabitants are prone to depositing garbage on nearby streets, open spaces, sewers and water bodies, resulting in unsanitary circumstances. The linear, non-cyclical approach to dealing with and treating waste, both on paper and in reality, is a major issue.

9.2.1 Landfill and open dumping

9.2.1.1 Health impact of landfill and open dumping

Pukkala and Pönkä (2001) conducted a study on individuals with habitation in households built on a previous landfill dump site of industrial and domestic wastes to observe the impact of cancers and respiratory diseases. This includes a retrospective cohort of 2014 individuals, including 957 males and 1057 females, with 2028 persons as a reference cohort. The study revealed that colorectal cancer was found completely amongst people who had lived at the landfill site for at least 5 years and asthma prevalence was substantially greater in the trash cohort than in all of the residents of the reference households. (Pukkala and Pönkä 2001). The relative risk increased with a living timespan at the site. Michelozzi et al. (1998) carried out a study on individuals in the vicinity of various pollution sources like a disposal site,

TABLE 9.1 Commonly used waste management techniques/methods with key benefits and drawbacks.

Waste management technique/ method	Operational regime	Benefits	Drawbacks
Waste recycling	The recovery of components from items once they've been consumed by users	1. Resource conservation 2. Source of industrial raw materials 3. Waste reduction for landfill and incineration disposal	1. A wide range of procedures 2. Recycling process emissions 3. May consume more energy in recycling processes 4. Products have low demand 5. Necessitates individuals co-operation
Open/illegal dumping	Heaving solid waste in open sites or areas, unoccupied plots, exposed trenches, roadsides, etc.	1. Only merit is that it is the most inexpensive method of waste disposal	1. Landscape pollution 2. Odour or awful stench 3. Breeding place of various disease vectors of diarrhoea, cholera, malaria, dengue, tuberculosis, etc. 4. One of the main sources of air, aquatic and soil/land pollution
Landfill	The disposal of solid waste in a specifically defined site, which in contemporary settings comprises of a pre-built "cell or compartment" enclosed in an impervious man-made or natural layer with inbuilt emission panels.	1. Economical method of disposal 2. Waste is used to backfill quarries 3. Renewable supply of energy from emitted gas	1. Contamination of water from leachate and runoff 2. The anaerobic breakdown of organic matter produces CH_4, CO_2, nitrogen, sulphur and other volatile organic compounds (VOCs), which pollute the atmosphere.

			3. Carcinogens or teratogens emission (e.g. As, Ni, Cr, dioxins, benzene, polycyclic aromatic hydrocarbons, vinyl chloride, etc.) 4. Biological vectors (birds, flies, pests) for various diseases 5. Odour, dirt, road traffic complications
Incineration/ pyrolysis	A combustion process meant to recover energy and decrease the volume of garbage disposed away	1. Waste volume and weight reduction and recovery of materials from approximately 30% leftover ash 2. Minimise the possible contagiousness of health care wastes 3. Generates energy for the electricity production	1. Generates hazardous wastes 2. Releases contaminated wastewater 3. Emission of toxic chemicals, heavy metals and waste incineration end products
Composting/ vermicomposting	Decomposition of organic substances by aerobic biological mechanisms	1. Waste reduction prior to its disposal by incineration and landfill 2. Soil amendment by useful end products 3. Create possibilities for employment	1. Odours and vermin nuisance 2. Bio-aerosols via microbes and fungal spores 3. VOCs emission 4. Ecological concerns for potential entry of contaminants in food chain

incinerator plant or oil refinery plant to perceive the impact of various cancers. This includes mortality/geographical analysis of the years 1987–93 with 1,341,389 residents, including 165,074 males and 176,315 females. In either sex, there was statistically no general increase or decrease in threat with the distance for various cancers at 0–3 km distance range from polluted sources. Goldberg et al. (1999) conducted a study on habitation nearby the Miron quarry municipal solid waste landfill to identify the cancer prevalence in individuals. The study comprised a case-control analysis of cancer diagnosed between 1979 and 1985 with 41 liver and intrahepatic bile duct cancer cases, 146 kidney cancer cases, 168 Non-Hodgkin's lymphoma cases, 101 pancreatic cancer cases and 367 prostate cancer cases. The reference control group was 533 male individuals aged 35–70 years. The cancers were shown to be more common in the exposure region closest to the location with an elevated risk at 0–1.5 km. Jarup et al. (2002) performed a study on individuals with habitation between 0–2 km of 9565 waste landfill regions operational in 1982–97 out of a total of 19,196 sites. The geographical study included 341,856,640 people with 89,786 cases of bladder cancer in adults, 113,631,433 people with 3973 cases of leukaemia in children, 341,856,640 people with 36,802 case conditions of a cancerous brain, 21,773 case conditions of hepatobiliary malignancies and 37,812 case conditions of leukaemia in adults. No significant risk was observed in people residing within 0–2 km of a landfill. Further, Jarup et al. (2007) carried out a geographical analysis of 4,584,541 births in individuals with habitation between 0–2 km of 6829 waste landfill regions operational in 1987–96 out of a total of 10,923 sites. There was no statistically substantial surge in the frequency of Down syndrome in communities residing within 0–2 km of the dump site. Fielder et al. (2000) conducted a geographical examination of individuals within five wards nearly 3 km from the Nanty-Gwiddon landfill area who had formally reported smells (exposed population), and comparative populations around 22 wards under the same authority. It was observed that residents closer to the area had maternal high-risk factors associated with congenital disorders in babies. No persistent alterations were detected among the two groups in terms of mortality, hospitalisation rates, or the percentage of low-weight births. Elliott et al. (2001) conducted the geographical analysis of individuals with habitation within a 2 km radius of 9565 landfills out of a total of 19,196 sites that were operating during the years 1982–97. The locations comprised 774 sites with special (hazardous) trash, 7803 sites with non-hazardous waste and 988 sites with undetermined waste. The analysis includes nearly 8.2 million babies born, 43,471 infant deaths during delivery, and 124,597 congenital abnormalities. The risks of birth defects with all abnormalities combined, birth weight anomalies and stillbirths were associated with landfill operations. Kloppenborg et al. (2005) carried out a geographical analysis of 2477 births with congenital anomalies during 1997–2001 in maternal dwellings in close proximity to 48 Danish waste landfills versus those residing further apart. There was a negligible link between trash dump landfill sites and congenital abnormalities combined with nervous system anomalies; however, there was a minor statistically significant additional risk for congenital cardiovascular abnormalities. Palmer et al. (2005) performed a geographical analysis of 542,682 births with at least one congenital abnormality on record in maternal dwellings within 2 km of 24 waste landfill areas. The ratio of observed to predicted rates of congenital abnormalities rose when landfills opened, according to data from expanded congenital malformation surveillance gathered between 1998 and 2000. Gilbreath and Kaas (2006) conducted a study that includes the retrospective cohort analysis of 10,073 live births during the years 1997–2001 on maternal residency in 197 Alaska native communities within potentially harmful waste disposal sites to health and the environment. It was observed that newborns born to mothers living near intermediate and high-hazard dumpsite areas had a greater proportion of low birth weight children, and children born to mothers in medium and high-hazard areas were affected by intrauterine growth restriction. Table 9.2 summarises the exposure scenario of landfill sites and probable health consequences.

9.2.1.2 GHG impact of landfill and open dumping

Controlled and unmanaged landfilling and open dumping of unprocessed garbage is the predominant disposal method in the vast majority of countries across the world. The emission of methane in the atmosphere from landfills is the most significant source of greenhouse gas from the solid waste controlling and management sector, accounting for approximately 700 Mt CO_2-e (United Nations Environment Programme, 2010; Njoku et al., 2019). Landfills may potentially be a contributor of nitrous oxide, but their contribution to total greenhouse gas emissions is small and is tied to the treatment of both wastewater bio-solids and leachates from landfills (United Nations Environment Programme, 2010). The GHG emission could be direct or indirect. Direct emission accounts for landfilling operating processes, that is leaks or other irregular liberation of gases (methane, trace non-methane volatile organic compounds, nitrous oxide and halogen-containing gases), the release of biogenic CO_2 from the decomposition of waste, emission of carbon dioxide, methane, nitrous oxide, trace carbon monoxide and non-methane volatile organic compounds, from fuel combustion in equipment and biogenic carbon dioxide, methane, nitrous oxide and carbon dioxide from the treatment of landfill leachate (United Nations Environment Programme, 2010). Indirect emission accounts for the release of GHGs from landfilling by upstream or downstream processes. Upstream emission of carbon dioxide, methane and nitrous oxide commences during onsite fuel production, consumption of electricity and materials production like liners and soils. Downstream liberation of GHG occurs from the combustion of the

TABLE 9.2 Landfill and incineration health hazards.

Waste handling technique	Design and population of study	Exposure conditions	Health perspective and study results	References
Landfill	Retrospective cohort study of 2014 individuals, including 957 males and 1057 females with a 2028 persons' reference cohort.	Habitation in households built on a previous landfill dump site of industrial and domestic wastes.	1. Cancers: Colorectal, Lung, Kidney, Pancreatic and Skin - The relative risk increased with living timespan at the site. Colorectal cancer was found completely amongst people who had lived at the landfill site for at least 5 years. 2. Respiratory diseases or symptoms - Asthma prevalence was substantially greater in the trash cohort than in all of the residents of the reference households.	Pukkala and Pönkä (2001)
	Mortality/geographical study of year 1987–93 with 1,341,389 residents, including 165,074 males and 176,315 females.	At vicinity of various pollution sources: a disposal site, incinerator plant and oil refinery plant.	1. Liver cancer - There was no net upsurge or decrease in the possibility of liver cancer with the distance for either sex. Within 0–3 km, no fatalities from hepatocellular carcinoma were detected. 2. Larynx cancer - Males had a non-significant higher risk of laryngeal cancer between 0–3 km, whereas females had no cases at 0–3 km. Men's death from laryngeal cancer decreased significantly with remoteness. 3. Lung cancer - In either sex, there was no general increase or decrease in threat with the distance for lung cancer. 4. Kidney cancer – The male kidney mortality rate was higher but not significant as anticipated, within 0–3 km radius. The females' mortality was considerably greater rate within the 3–8 km range. There have been no instances detected inside the 0–3 km range. 5. Hodgkin's and Non-Hodgkin's lymphoma - In either sex, there was no general increase or decrease in risk with the distance for lung cancer.	Michelozzi et al. (1998)

	Case-control study of cancer diagnosed between 1979 and 1985 with 41 liver and intrahepatic bile ducts cancer cases, 146 kidney cancer cases, 168 Non-Hodgkin's lymphoma cases, 101 pancreatic cancer cases and 367 prostate cancer cases. The reference control group was 533 male individuals aged 35–70 years.	Habitation nearby the Miron quarry municipal solid waste landfill	1. Liver cancer - It was shown to be more common in the exposure region closest to the location with an elevated risk at 0–1.5 km. 2. Kidney cancer - In the closest exposure region to the location, the higher-than-expected risks of kidney cancer were observed. 3. Non-Hodgkin's lymphoma - In the closest exposure region to the location, there were greater chances for Non-Hodgkin's lymphoma, than anticipated. 4. Pancreas and Prostate cancer - Cancers of the pancreatic and prostate were shown to be more common in the exposure radius closest to the site with increased risks of pancreatic cancer within 0–1.25 km discovered, but not of prostate cancer.	Goldberg et al. (1999)
	Geographical study of 341,856,640 people with 89,786 cases of bladder cancer in adults, 113,631,433 people with 3973 cases leukaemia in children, 341,856,640 people with 36,802 case conditions of cancerous brain, 21,773 case conditions of hepatobiliary malignancies and 37,812 case conditions of leukaemia in adults.	Habitation between 0–2 km of 9565 waste landfill regions operational in 1982–97 out of total 19,196 sites.	1. Bladder cancer - There was no upsurge in the risk of cancer of bladder in people residing within 0–2 km of a landfill. 2. Childhood Leukaemia - There was no proliferation in the risk of childhood leukaemia in the people residing within 0–2 km of a landfill. 3. Brain and Hepatobiliary cancer and adult Leukaemia - There was no increase in the risk of brain and hepatobiliary cancer and leukaemia in adults.	Jarup et al. (2002)

(*Continued*)

TABLE 9.2 (Continued)

Waste handling technique	Design and population of study	Exposure conditions	Health perspective and study results	References
	Geographical analysis of 4,584,541 births.	Habitation between 0–2 km of 6829 waste landfill regions operational in 1987–96 out of total 10,923 sites.	1. Down syndrome - There was no statistically substantial surge in the frequency of Down syndrome in communities residing within 0–2 km of the dump site.	Jarup et al. (2007)
	The geographical examination of individuals within five wards near 3 km from the landfill area who had formally reported smells (exposed population), and comparative populations around 22 wards under the same authority.	Habitation adjoining the Nanty-Gwiddon landfill.	1. Birth defects - Residents closer to the area had maternal high-risk factors associated with congenital disorders in babies. 2. Birth weight - No persistent alterations were detected among the two groups in terms of mortality, hospitalisation rates, or the percentage of low-weight births.	Fielder et al. (2000)
	Geographic analysis of nearly 8.2 million babies born, 43,471 infant deaths during delivery and 124,597 congenital abnormalities.	Habitation within a 2 km radius of 9565 landfills out of a total of 19,196 sites that were operating during the years 1982–97. The locations comprised of 774 sites with special (hazardous) trash, 7803 sites with non-hazardous waste and 988 sites with undetermined waste.	1. Birth defects - The risks with all abnormalities combined were mostly for neural tube deformities, gut wall deformities, clinical repair of gastroschisis and exomphalos and hypospadias and epispadias. 2. Birth weight - The hazards were associated with low and extremely low birth weights. There was no increase in the chance of stillbirth. However, after opening, the chances for low birth weight and stillbirths increased.	Elliott et al. (2001)

	Geographical analysis of 2477 births with congenital anomalies amid 1997–2001.	Maternal dwellings in close proximity to 48 Danish waste landfills versus those residing further apart	1. Birth defects - There was negligible link between trash dump landfill site and congenital abnormalities combined with nervous system anomalies; there was a minor statistically significant additional risk for congenital cardiovascular abnormalities.	Kloppenborg et al. (2005)
	Geographical analysis of 542,682 births with at least one congenital abnormality on record	Maternal dwelling in between 2 km of 24 waste landfill areas	1. Birth defects - The ratio of observed to predicted rates of congenital abnormalities rose when landfills opened, according to data from expanded congenital malformation surveillance gathered between 1998 and 2000.	Palmer et al. (2005)
	Study includes the retrospective cohort analysis of 10,073 live births within year 1997–2001.	Maternal residency in 197 Alaska native communities within potentially harmful waste disposal sites to health and the environment	1. Birth weight - Newborns born to mothers living near intermediate and high hazard dumpsite areas had a greater proportion of low birth weight children; children born to mothers in medium and high hazard areas were affected by intrauterine growth restriction.	Gilbreath and Kaas (2006)
Incineration	Geographical analysis of above 14 million individuals with cancer diagnosed amid 1974–86	Habitation close to 72 solid waste incinerator plants	1. Stomach cancer - The study revealed a statistically substantial ($P < .05$) decrease in the risk of stomach cancer with increasing the distance throughout the course of the two stages of the trial. 2. Colorectal cancer - For the two stages trial, a statistically significant ($P < .05$) decrease in possibility of colorectal cancer was observed with increasing distance from the incinerators. 3. Liver cancer - For entire study phases, a statistically significant ($P < .05$) decrease in the risk of liver cancer was observed with increasing distance from the incinerators. 4. First Stage larynx cancer - Overall, there was no proof that danger decreased with distance from the incinerators. 5. Lung cancer – The study revealed a statistically significant ($P < .05$) decrease in the risk of lung cancer with increasing distance from incinerators throughout the course of the two stages of the research. 6. Soft-tissue sarcoma - Overall, there was no indication that the incidence of connective tissue cancer decreased with the distance to incinerators.	Elliott et al. (1996)

(*Continued*)

TABLE 9.2 (Continued)

Waste handling technique	Design and population of study	Exposure conditions	Health perspective and study results	References
			7. Bladder cancer - There was no general increase or decrease in risk of bladder cancer with distance. 8. Non-Hodgkin's lymphoma - Overall, there was no evidence supporting a decrease in risk with increasing distance from incineration plants for non-Hodgkin lymphomas. 9. Nasal and nasopharyngeal cancers - Overall, there was little proof that danger decreased with the distance to incinerators.	
	Geographical study of 235 liver cancer cases diagnosed between 1974 and 1986. Sex: 155 males. Age: 82 cases <65 years	Residence near 52 solid waste incinerator plants	1. Liver cancer - 66 (55%) out of the 119/235 (51%) patients analysed had primary liver cancer verified, while 21 (18%) had proven secondary malignancies. According to updated estimates, there were 0.53–0.78 more cases per 105 each year within 1 kilometre, with the percentage of actual primary ranging from 55% to 82% (i.e. omitting secondary malignancies).	Elliott et al. (2000)
	Case control study of 755 case-control pairs, determined by age (755 histologically confirmed cases of lung cancer, who died from 1979 to 1981 or from 1985 to 1986). Sex: 755 males.	Residence near various sources of environmentally polluted sites: shipyard, iron foundry, incinerator and city centre	1. Lung cancer - Lung cancer risk was correlated with the incinerator, with a disproportionately high risk at the site and a sharp decline distant from the incinerator.	Biggeri et al. (1996)

	Geographical study of lung cancer deaths from the Liguria mortality registry; whole Province population as referent. Age: between 35 and >65 years.	Residence in two areas exposed to environmental pollution emitted by a coal-fired power station and other industrial sources, including a waste incinerator	1. Lung cancer - The greatest rates were seen among urban residents, of both sexes. After eliminating rural and semi-rural areas from the analysis, no statistical significance of risk increase was discovered among men in the two exposed sites. In both exposed regions, a risk excess for females persisted after being restricted to urban and semi-urban municipalities after correcting for deprivation characteristics ($P < .05$). The rural/urban gradients and the elevated risk seen in females near industrial locations were also verified by Bayesian mapping.	Parodi et al. (2004)
	Case-control analysis Cases: 803 non-Hodgkin lymphoma cases and 110 soft-tissue sarcoma cases (clinically diagnosed in 1980–95) Controls: 176 Hodgkin's disease cases	Residence next to a MSW incinerator (16.3 ng international toxic equivalence factor/m^3) with excessive dioxin emissions	1. Non-Hodgkin's lymphoma and soft-tissue sarcoma - The most likely and very significant clusters were discovered in the vicinity of the disposal site of the solid waste incinerator. 2. Hodgkin's lymphoma - The number of identified cases was less than expected, so there was no cluster near the MSW incinerator, and the most probable cluster in the entire department was made up of four cantons situated south of Besancon, although the risk was not particularly high. Hodgkin's disease did not have a specific spatial distribution.	Viel et al. (2000)
	Case-control study of 37 primary malignant sarcomas cases (diagnosed between 1989 and 1998) and 171 controls; 17 cases of males and 20 cases of females of 26–85 years age. Controls of 24–90 years age.	Residence near an industrial waste incinerator: residence's distance (in rings of 1 kilometre) from incineration.	1. Soft-tissue sarcoma - Based on 5 exposed instances, the odds ratio for living within two kilometres, normalised by age and sex, was 31.4. The risk dropped very quickly at higher separations, fluctuating towards the null value of 1.	Comba et al. (2003)

(*Continued*)

TABLE 9.2 (Continued)

Waste handling technique	Design and population of study	Exposure conditions	Health perspective and study results	References
	222 NHL patients (diagnosed in 1980 to 1995) and subjects were matched 10-to-1 in a case-control study that included both groups.	Residential dwelling close to an MSW incinerator with elevated levels of dioxin emissions	1. Non-Hodgkin's lymphoma - Habitation in the region with the strongest concentration of dioxin (simulated ground-level dioxin density 0.0004–0.0016 pg m^{-3} zone) increased one's risk of development of NHL by 2.3 times compared to living in regions with the lowest concentration (simulated ground-level dioxin density <0.0001 pg m^{-3} zone). The moderate dioxin exposure groups were not associated with an elevated risk. The outcomes were unchanged after accounting for a wide variety of socioeconomic traits at the block group level.	Floret et al. (2003)
	Geographical/mortality study of 9224 childhood deaths due to cancer befalling before the 16th birthdate in 1953–80	Residence near 70 municipal incinerators (307 hospitals incinerators and 460 landfills for hazardous waste)	1. Childhood cancer - The statistics on paediatric cancer and leukaemia revealed a significantly substantial over-migration away from birth places near municipal incinerators. Over 5.0 km of these locations, the relative hazards were roughly 2:1. The ratio was 2.27 when the subgroup was limited to people whose two residences were within 15 kilometres of same incinerator.	Knox (2000)
	Case study (CS) /geographical study (GS): six cleft lip and palate children for CS; 18 regions with solid waste incinerators for GS	Residence near waste incinerators	1. Birth defects - After the commencement of incineration, CS and GS revealed no higher risk of cleft lip and palate in the tested area.	Jansson and Voog (1989)
	Geographical comparison of 194 exposed community surrounding 70 MSW incinerators (within 10 km of the plants); 2678 unexposed.	Community where mothers lived at the time of childbirth or a medical abortion	1. Birth defects - When compared to populations that were not exposed, the prevalence of congenital abnormalities was not noticeably greater. Particularly the facial cleft and renal dysplasia categories of severe defects were more prevalent in the impacted communities. For obstructive uropathies, exposed populations showed a dose-response relationship of risk with rising exposure. With an increase in vehicle traffic density, the incidence of cardiac abnormalities, obstructive uropathies and cutaneous anomalies rose linearly.	Cordier et al. (2004)

	A geographical comparison of 7242 perinatal mortality, 1796 infant mortality and 489,154 live births between 1997 and 1998	63 solid wastes incineration plants with elevated dioxin pollution content (over 80 nanogram international toxic equivalent, TEQ/m^3) were within 10 km of the maternal domicile.	1. Birth defects and birth weight - Around 2 km of the incinerators, none of the studied birth outcomes displayed a statistically significant increase for birth defects and birth weight.	Tango et al. (2004)
	The geographical analysis of 244,758 births to Cumbrian women between 1956 and 1993 that included 3234 stillbirths, 2663 neonatal mortality and 1569 deadly congenital abnormalities	Mother's dwelling was next to four incinerators that were active during the research period	1. Birth defects - Increased chance of a fatal congenital abnormality, including heart and spina bifida problems.	Dummer et al. (2003)
	Geographical analysis of birth to twins in 1975–83	Maternal dwelling next to an incinerator for both chemicals and municipality trash	1. Sex ratio and birth twinning - In places where the emissions posed the most threat, twinning occurred more frequently. In high-risk locations between 1976 and 1979, there were 7.3 and 12.4 twins per 1000 births, compared to 7.1 per 1000 in background areas. 1980–83: 7.6 per 1000 births in background regions against 16.0 and 19.9 per 1000 births in high-risk areas.	Lloyd et al. (1988)
	Geographical study of births between 1975 to 1979, and between 1981 to 1983	Residence in at-risk areas near two waste incinerators	i.Sex ratio and twinning - There were no variations between entire areas at-risk and comparative locations, but the district with the highest risk has a notable surplus of female births.	Williams et al. (1992)

(*Continued*)

TABLE 9.2 (Continued)

Waste handling technique	Design and population of study	Exposure conditions	Health perspective and study results	References
	Cross-sectional research of 382 primary school students in a cross-sectional design, aged 9 to 11 years.	Habitation in three polluted areas in the vicinity of an incinerator	1. Respiratory diseases or symptoms - The frequency of respiratory function aberration was higher in the polluted regions than in the non-polluted regions ($P < .05$), and the average value of FVC and FEV_1 were considerably lower in the polluted sites than in the non-polluted zone ($P < .05$).	Hsiue et al. (1991)
	Case-control research using three matched comparator communities and a sample of normal and responsive inhabitants from three waste incineration plants (biomedical, hazardous, and municipal). Infants and kids under the age of eight were not included in this research.	Residence near waste incinerators	1. Respiratory diseases or symptoms - Between incineration and comparator communities, there were no discernible changes in the incidence of chronic or acute respiratory complaints, basal pulmonary function tests, or the mean peak expiratory flow rate recorded during a 35-day period.	Shy et al. (1995)
	756 participants in cross-sectional research. In comparison to male participation, there were more female participants. Ages ranged from 8 to 81, with a mean of 39.1 years.	Residence near an incinerator	1. Respiratory diseases or symptoms - There were no variations in the individuals' pulmonary health between an incineration community and its comparative community.	Lee and Shy (1999)
	450,807 primary school students, ages 6 to 12, were included in a cross-sectional research.	Distances between all 37 municipal trash incineration facilities with each of the 996 public elementary schools.	1. Pulmonary and skin ailments or symptoms - Decrease in the distance of schools from the nearest MSW incinerator plant were independently associated with an amplified pervasiveness of wheezing, headache and stomach ache.	Miyake et al. (2005)

captured landfill methane substituting fossil energy (CO_2) and from carbon stored for a long term in the landfill (United Nations Environment Programme, 2010).

9.2.2 Incineration and waste thermal treatment

9.2.2.1 Health impact of incineration and waste thermal treatment

Elliott et al. (1996) conducted a geographical analysis of above 14 million individuals with cancer diagnosed from 1974 to 1986. The habitation has close to 72 solid waste incinerator plants. The incidence of stomach, colorectal, liver and lung cancers decreased significantly at $P < .05$ with increasing distance from the incinerators; however, no associated risk or indication was observed for soft-tissue sarcoma, first stage larynx cancer, bladder cancer, non-Hodgkin's lymphoma, nasal and nasopharyngeal cancers. Further, Elliott et al. (2000) conducted a geographical study of 235 liver cancer cases diagnosed between 1974 and 1986. Sex: 155 males with 82 cases <65 years' age on residences near 52 solid waste incinerator plants. It was observed that there were 0.53–0.78 more cases per 105 each year within 1 kilometre, with the percentage of actual primary ranging from 55% to 82% (i.e. omitting secondary malignancies). Biggeri et al. (1996) conducted a case-control study of 755 case-control pairs, determined by age (755 histologically confirmed cases of lung cancer who died from 1979 to 1981 or from 1985 to 1986) on 755 males residing near various sources of environmentally polluted sites: shipyard, iron foundry, incinerator and city centre. Lung cancer risk was correlated with the incinerator, with a disproportionately high risk at the site and a sharp decline distant from the incinerator. Parodi et al. (2004) carried out a geographical study of lung cancer deaths from the Liguria mortality registry, with the whole province population as reference. Ages were between 35 and >65 years. Individual residences in two areas were exposed to environmental pollution emitted by a coal-fired power station and other industrial sources, including a waste incinerator. The greatest rates were seen among urban residents of both sexes. After eliminating rural and semi-rural areas from the analysis, no statistical significance of risk increase was discovered among men in the two exposed sites. In both exposed regions, a risk excess for females persisted after being restricted to urban and semi-urban municipalities after correcting for deprivation characteristics ($P < .05$). The rural/urban gradients and the elevated risk seen in females near industrial locations were also verified by Bayesian mapping. Viel et al. (2000) conducted a case-control analysis. There were 803 case of non-Hodgkin lymphoma and 110 soft-tissue sarcoma cases (clinically diagnosed in 1980–95) with controls of 176 Hodgkin's disease cases. Residences were next to an MSW incinerator (16.3 ng international toxic equivalence factor/m^3) with excessive dioxin emissions. In the case of non-Hodgkin's lymphoma and soft-tissue sarcoma, the most likely and very significant clusters were discovered in the vicinity of the disposal site of the solid waste incinerator as compared to Hodgkin's lymphoma. Comba et al. (2003) carried out a case-control study of 37 primary malignant sarcomas cases (diagnosed between 1989 and 1998) and 171 controls; 17 cases of males and 20 cases of females aged 26–85 years with controls aged 24–90 years residing near an industrial waste incinerator; within a radius of one kilometre distance from the incinerator. Based on five exposed instances, the odds ratio for living within two kilometres, normalised by age and sex, was 31.4. The risk dropped very quickly at higher separations, fluctuating towards the null value of 1. Floret et al. (2003) performed a study on 222 non-Hodgkin's lymphoma (NHL) patients (diagnosed in 1980–95), and subjects were matched 10-to-1 in a case-control study that included both groups with residential dwellings close to an MSW incinerator with elevated levels of dioxin emissions. Habitation in the region with the strongest concentration of dioxin (simulated ground-level dioxin density 0.0004–0.0016 pg m^{-3} zone) increased one's risk of development of NHL by 2.3 times compared to living in regions with the lowest concentration (simulated ground-level dioxin density <0.0001 pg m^{-3} zone). The moderate dioxin exposure groups were not associated with an elevated risk. The outcomes were unchanged after accounting for a wide variety of socioeconomic traits at the block group level. Knox (2000) conducted a geographical/mortality study of 9224 childhood deaths due to cancer falling before the 16th birthdate in 1953–80 at residences near 70 municipal incinerators (307 hospital incinerators and 460 landfills for hazardous waste). The statistics on paediatric cancer and leukaemia revealed a significantly substantial over-migration away from birthplaces near municipal incinerators. Over 5.0 km of these locations, the relative hazards were roughly 2:1. The ratio was 2.27 when the subgroup was limited to people whose two residences were within 15 kilometres of the same incinerator. Jansson and Voog (1989) carried out a case study (CS)/geographical study (GS): six cleft lip and palate children for CS; 18 regions with solid waste incinerators for GS. After the commencement of incineration, CS and GS revealed no higher risk of cleft lip and palate in the tested area. Cordier et al. (2004) conducted a geographical comparison of 194 exposed communities surrounding 70 MSW incinerators (within 10 km of the plants), with 2678 unexposed. Communities include mothers who lived at the time of childbirth or medical abortion. When compared to populations that were not exposed, the prevalence of congenital abnormalities was not noticeably greater. Particularly the facial cleft and renal dysplasia categories of severe defects were more prevalent in the impacted communities. For obstructive uropathies, exposed populations showed a dose-response relationship of risk with rising exposure. With an increase in vehicle traffic density, the incidence of cardiac abnormalities, obstructive uropathies and cutaneous anomalies rose linearly. Tango et al.

(2004) carried out a geographical comparison of 7242 perinatal mortalities, 1796 infant mortalities and 489,154 live births between 1997 and 1998; around 63 solid wastes incineration plants with elevated dioxin pollution content (over 80 ng of international toxic equivalent, TEQ/m^3) are within 10 km of the maternal domicile. Within a radius of around 2 km of the incinerators, none of the studied birth outcomes displayed a statistically significant increase in birth defects and birth weight. Dummer et al. (2003) performed a geographical analysis of 244,758 births to Cumbrian women between 1956 and 1993 that included 3234 stillbirths, 2663 neonatal mortalities and 1569 deadly congenital abnormalities. Mother's dwellings were next to four incinerators that were active during the research period. An increased chance of a fatal congenital abnormality, including heart and spina bifida problems, was observed in the study. Lloyd et al. (1988) conducted a geographical analysis of the birth of twins during 1975−83 with maternal dwellings next to an incinerator for both chemicals and municipality trash. In places where the emissions posed the most threat, twinning occurred more frequently. In high-risk locations between 1976 and 1979, there were 7.3 and 12.4 twins per 1000 births, compared to 7.1 per 1000 in background areas; during1980−83, there were 7.6 per 1000 births in background regions against 16.0 and 19.9 per 1000 births in high-risk areas. Williams et al. (1992) performed a geographical study of births between 1975 to 1979, and between 1981 to 1983 on residence in at-risk areas near two waste incinerators. There were no variations between entire areas at-risk and comparative locations, but the district with the highest risk has a notable surplus of female births. Hsiue et al. (1991) conducted a cross-sectional research of 382 primary school students aged 9 to 11 years in a cross-sectional design, with habitation in three polluted areas in the vicinity of an incinerator. The frequency of respiratory function aberration was higher in the polluted regions than in the non-polluted regions ($P < .05$), and the average value of FVC and FEV$_1$ were considerably lower in the polluted sites than in the non-polluted zone ($P < .05$). Shy et al. (1995) carried out a case-control research using three matched comparator communities and a sample of normal and responsive inhabitants from three waste incineration plants (biomedical, hazardous and municipal). Infants and kids under the age of eight were not included in this research. Between incineration and comparator communities, there were no discernible changes in the incidence of chronic or acute respiratory complaints, basal pulmonary function tests, or the mean peak expiratory flow rate recorded during a 35-day period. Lee and Shy (1999) included 756 participants in cross-sectional research. In comparison to male participation, there were more female participants. Ages ranged from 8 to 81, with a mean of 39.1 years with residence near an incinerator. It was observed that there are no variations in the individuals' pulmonary health between an incineration community and its comparative community. Miyake et al. (2005) included 450,807 primary school students aged 6−12 in a cross-sectional research. This included distances between all 37 municipal trash incineration facilities and each of the 996 public elementary schools. The decrease in the distance of the schools from the nearest MSW incinerator plant was independently associated with an amplified pervasiveness of wheezing, headache and stomach ache. The health impact of thermal waste treatment is methodically summarised in Table 9.2.

9.2.2.2 GHG impact of incineration and waste thermal treatment

Incineration has a lower global environmental effect than landfilling, generating approximately 40 Mt CO_2-e for a given year (Bogner et al., 2007). The direct releases from establishments are mostly made up of fossil or biogenic carbon dioxide. The quantities of fossil, as well as the biogenic carbon of the waste inputs, differ tremendously among nations, regions and perhaps even installations (Astrup et al., 2009). Because only the fossil carbon dioxide is often recorded as greenhouse emissions from the incinerator, the total climatic impact of combustion will be heavily impacted by the input waste's fossil carbon concentration. Based on the energy anticipated to be replenished, downstream indirect greenhouse gas savings through energy production can dominate an estimation of the emissions from incinerators.

9.3 Discussion

The increase in population, lifestyle and urbanisation has led to aggravated waste generation over the years. More waste needs additional landfilling sites for the appropriate management. Moreover, injudicious land use patterns have caused exacerbated GHG and toxic gases emission from overloaded waste landfilling sites. The consequential health effects can be direct or indirect. Direct effects include explosion and asphyxiation and mental hazards from emission of toxic gases, like particulate matter, furans, sulphides, VOCs, dioxins, ammonia, carbon dioxide, carbon monoxide and methane (Silverman and Ito, 2010; Nanda and Berruti, 2021). Toxic chemical emissions from open dumping and burning due to unmanaged or poorly managed waste dumps. In landfills, the decomposition of organic solid waste is associated withGHG emissions along with pollution of soil and water in surrounding areas. Illegitimate solid waste disposal is the main malefactor for drain clogging, enabling a suitable environment for the breeding of disease vectors in stagnant water bodies (Reinhart, 1993; Srivastava et al., 2015). The most probable cause of disease outbreaks (respiratory, gastrointestinal, hepatic and kidney infections, allergies, malaria, cancers, psychological problems, etc.) results from exposure of disease vectors, pollutants and toxins to the people living in the garbage-strewn vicinity (Gouveia and Prado, 2010; Zohoori and Ghani, 2017). Children and elderly individuals, especially those with existing respiratory and heart ailments, tend to demonstrate more susceptibility to bronchitis and other respiratory infections, short lifespan, premature births/deaths, cancers, etc. (Rushton, 2003; Porta et al., 2009; He et al., 2015).

Thermal waste processing and treatment includes systematic incineration, co-incineration (replacement of non-renewable fossil fuel with refuse-derived fuel (RDF) in typical industrial applications such as cement kilns), mass-burn pyrolysis and the gasification. There is little success in India with thermally treated waste programmes, which typically emphasise converting MSW into refuse-derived fuel (RDF), also known as "fluff," to be burned for electricity generation or to augment fuel in the cement kilns. The nature of the informal recycling industry in the Indian cities retrieves most of the dried, higher heating material from the MSW, generating a wet remnant with a significant green waste composition that is inappropriate for the creation of flammable "fluff" if deprived of significant pre-treatment procedure like drying. These issues, for instance, are documented for establishments in Indian cities like Chandigarh and Hyderabad (Yadav, 2009; Joseph, 2007). Like several underdeveloped countries, India lacks suitable lab amenities to properly monitor the operation of the thermal systems. The possible health impacts of a large range of toxic chemicals generated by waste incineration could be assessed by analysing the effects of particular pollutants or by conducting more comprehensive studies of neighbourhood residents and the incinerator workforce (National Research Council US Committee on Health Effects of Waste Incineration, 2000). Particulate exposure is linked to acute health impacts such as increased total mortality and emergency hospital admissions, notably cardiovascular and respiratory mortality and morbidity, according to epidemiological data (Dockery and Pope, 1994, Hu and Shy, 2001). The direct or indirect emissions of greenhouse gas (GHG) are also linked to the waste treatment, disposal regime and recovery. Unduly managed solid waste can lead to exacerbated GHG emissions. International institutions, notably UNEP, are now leading a variety of waste and climate change-related operations. There is indeed a lack of a unified strategy that has, however, led to gaps, redundancy and geographical disparities in the programmes available.

9.4 Conclusion

A centralised mechanism is required to work collaboratively with established organisations to ensure easy accessibility and diffusion of pertinent data, efficient resource use to accomplish climate gain via integrated waste management, encouragement of the best practices, and the rapid exchange of a simple, efficacious and proven knowledge and technology to the developing nations. The existing literature reflects that more comprehensive efforts are required to overcome various facets and barriers for implementing effective and efficient managerial regimes for solid waste management with minimal environmental and health risks.

References

Astrup, T., Mollee, J., Fruergaard, T., 2009. Incineration and co-combustion of waste: accounting of greenhouse gases and global warming contributions. Waste Manag. Res. 27, 789–799.

Biggeri, A., Barbone, F., Lagazio, C., Bovenzi, M., Stanta, G., 1996. Air pollution and lung cancer in Trieste, Italy: spatial analysis of risk as a function of distance from sources. Environ. Health Perspect. 104, 750–754.

Boateng, S., Amoako, P., Appiah, D.O., Poku, A.A., Garsonu, E.K., 2016. Comparative analysis of household's solid waste management in rural and urban Ghana. J. Environ. Public Health 2016, 5780258. Available from: https://doi.org/10.1155/2016/5780258.

Bogner, J., Abdelrafie Ahmed, M., Diaz, C., Faaij, A., Gao, Q., Hashimoto, S., et al., 2007. Waste Management. In: Metz, B., Davidson, O.R., Bosch, P.R., Dave, R., Meyer, L.A. (Eds.), Climate Change 2007: Mitigation. Contribution of Working Group III to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Comba, P., Ascoli, V., belli, S., Benedetti, M., Gatti, L., Ricci, P., et al., 2003. Risk of soft tissue sarcomas and residence in the neighborhood of an incinerator of industrial wastes. Occup. Environ. Med. 60, 650–683.

Cordier, S., Chevrier, C., Robert-Gnansia, E., Lorente, C., Brula, P., Hours, M., 2004. Risk of congenital anomalies in the vicinity of municipal solid waste incinerators. Occup. Environ. Med. 61, 8–15.

Dockery, D.W., Pope III, C.A., 1994. Acute respiratory effects of particulate air pollution. Annu. Rev. Public. Health 15, 107–132.

Dummer, T.J., Dickinson, H.O., Parker, L., 2003. Adverse pregnancy outcomes around incinerators and crematoriums in Cumbria, north west England, 1956-93. J. Epidemiol. Community Health 57, 456–461.

Elliott, P., Shaddick, G., Kleinschmidt, I., Jolley, D., Walls, P., Beresford, J., et al., 1996. Cancer incidence near municipal solid waste incinerators in Great Britain. Br. J. Cancer 73, 702–710.

Elliott, P., Eaton, N., Shaddick, G., Carter, R., 2000. Cancer incidence near municipal solid waste incinerators in Great Britain. Part 2: histopathological and case-note review of primary liver cancer cases. Br. J. Cancer 82 (5), 1103–1106.

Elliott, P., Briggs, D., Morris, S., de Hoogh, C., Hurt, C., Jensen, T.K., et al., 2001. Risk of adverse birth outcomes in populations living near landfill sites. Br. Med. J. 323, 363–368.

Fielder, H.M., Poon-King, C.M., Palmer, S.R., Moss, N., Coleman, G., 2000. Assessment of impact on health of residents living near the Nant-y-Gwyddon landfill site: retrospective analysis. BMJ. 320 (7226), 19–22.

Floret, N., Mauny, F., Challier, B., Arveux, P., Cahn, J.Y., Viel, J.F., 2003. Dioxin emissions from a solid waste incinerator and risk of non-Hodgkin lymphoma. Epidemiology 14 (4), 392–398.

Gilbreath, S., Kaas, P.H., 2006. Adverse birth outcomes associated with open dumpsites in Alaska native villages. Am. J. Epidemiol. 164, 518–528.

Goldberg, M.S., Siemiatyck, J., DeWar, R., Dèsy, M., Riberdy, H., 1999. Risk of developing cancer relative to living near a municipal solid waste landfill site in Montreal, Quebec, Canada. Arch. Environ. Health 54, 291–296.

Gouveia, N., Prado, R.R.D., 2010. Health risks in areas close to urban solid waste landfill sites. Rev. de. Saude Publica 44, 859–866. Available from: https://doi.org/10.1590/S0034-89102010005000029.

Hannan, M.A., Arebey, M., Begum, R.A., Mustafa, A., Basri, H., 2013. An automated solid waste bin level detection system using Gabor wavelet filters and multilayer perception. Resour. Conserv. Recycl. 72, 33–42. Available from: https://doi.org/10.1016/j.resconrec.2012.12.002.

He, Z., Li, G., Chen, J., Huang, Y., An, T., Zhang, C., 2015. Pollution characteristics and health risk assessment of volatile organic compounds emitted from different plastic solid waste recycling workshops. Environ. Int. 77, 85–94. Available from: https://doi.org/10.1016/j.envint.2015.01.004.

Hsiue, T.R., Lee, S.S., Chen, H.I., 1991. Effects of air pollution resulting from wire reclamation incineration on pulmonary function in children. Chest 100, 698–702.

Hu, S., Shy, C.M., 2001. Health effects of waste incineration: a review of epidemiologic studies. J. Air Waste Manag. Assoc. 51, 1100–1109.

Jansson, B., Voog, L., 1989. Dioxin from Swedish municipal incinerators and the occurrence of cleft lip and palate malformations. Int. J. Environ. Stud. 34, 99–104.

Jarup, L., Briggs, D., de Hoogh, C., Morris, S., Hurt, C., Lewin, A., et al., 2002. Cancer risks in populations living near landfill sites in Great Britain. Br. J. Cancer 86, 1732–1736.

Jarup, L., Morris, S., Richardson, S., Briggs, D., Cobley, N., de Hoog, C., et al., 2007. Down syndrome in births near landfill sites. Prenat. Diagn. 27, 1191–1196.

Joseph K., 2007. Lessons from municipal solid waste processing initiatives in India. In: Proceedings of the International Symposium MBT 2007.

Kaza, S., Yao, L., Bhada-Tata, P., Woerden, F.V., 2018. What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050. World Bank, Urban Development; Washington, DC. Available from: https://openknowledge.worldbank.org/handle/10986/30317License: CC BY 3.0IGO.

Kloppenborg, S.Ch, Brandt, U.K., Gulis, G., Ejstrud, B., 2005. Risk of congenital anomalies in the vicinity of waste landfills in Denmark; an epidemiological study using GIS. Cent. Eur. J. Public. Health 13, 137–143.

Knox, E., 2000. Childhood cancers, birthplaces, incinerators and landfill sites. Int. J. Epidemiol. 29, 391–397.

Lee, J.T., Shy, C.M., 1999. Respiratory function as measured by peak expiratory flow rate and PM10: six communities study. J. Expo. Anal. Environ. Epidemiol. 9, 293–299.

Lloyd, O.L., Lloyd, M.M., Williams, F.L., Lawson, A., 1988. Twinning in human populations and in cattle exposed to air pollution from incinerators. Br. J. Ind. Med. 45, 556–560.

Michelozzi, P., Fusco, D., Forastiere, F., Ancona, C., Dell'Orco, V., Perucci, C.A., 1998. Small area study of mortality among people living near multiple sources of air pollution. Occup. Environ. Med. 55, 611–615.

Miyake, Y., Yura, A., Misaki, H., Ikeda, Y., Usui, T., Iki, M., et al., 2005. Relationship between distance of schools from the nearest municipal waste incineration plant and child health in Japan. Eur. J. Epidemiol. 20, 1023–1029.

Nanda, S., Berruti, F., 2021. Municipal solid waste management and landfilling technologies: a review. Environ. Chem. Lett. 19 (2), 1433–1456. Available from: https://doi.org/10.1007/s10311-020-01100-y.

National Research Council (US) Committee on Health Effects of Waste Incineration, 2000. Waste Incineration and Public Health. National Academy Press, Washington, DC.

Njoku, P.O., Edokpayi, J.N., Odiyo, J.O., 2019. Health and environmental risks of residents living close to a landfill: a case study of Thohoyandou Landfill, Limpopo Province, South Africa. Int. J. Environ. Res. Public. Health 16 (12), 2125. Available from: https://doi.org/10.3390/ijerph16122125.

Palmer, S.R., Dunstan, F.D., Fielder, H., Fone, D.L., Higgs, G., Senior, M.L., 2005. Risk of congenital anomalies after the opening of landfill sites. Environ. Health Perspect. 113 (10), 1362−1365.

Parodi, S., Baldi, R., Benco, C., Franchini, M., Garrone, E., Vercelli, M., et al., 2004. Lung cancer mortality in a district of La Spezia (Italy) exposed to air pollution from industrial plants. Tumori 90, 181−185.

Patwa, A., Parde, D., Dohare, D., Vijay, R., Kumar, R., 2020. Solid waste characterization and treatment technologies in rural areas: an Indian and international review. Environ. Technol. Innov. 20, 101066.

Porta, D., Milani, S., Lazzarino, A.I., Perucci, C.A., Forastiere, F., 2009. Systematic review of epidemiological studies on health effects associated with management of solid waste. Environ. Health 8 (1), 1−14.

Pukkala, E., Pönkä, A., 2001. Increased incidence of cancer and asthma in houses built on a former dump area. Environ. Health. 109, 1121−1125.

Reinhart, D.R., 1993. A review of recent studies on the sources of hazardous compounds emitted from solid waste landfills: a US experience. Waste Manag. Res. 11 (3), 257−268.

Rushton, L., 2003. Health hazards and waste management. Br. Med. Bull. 68 (1), 183−197. Available from: https://doi.org/10.1093/bmb/ldg034.

Shy, C.M., Degnan, D., Fox, D.L., Mukerjee, S., Hazucha, M.J., Boehlecke, B.A., et al., 1995. Do waste incinerators induce adverse respiratory effects? An air quality and epidemiological study of six communities. Environ. Health Perspect. 103, 714−724.

Shyam, H.S., Gupta, M., Gupta, H., 2022. Circular route to the management of solid waste: a case study of EcoSage Enviro. Mater. Today Proc. 49 (8), 3650−3653. Available from: https://doi.org/10.1016/j.matpr.2021.08.259.

Silverman, R.A., Ito, K., 2010. Age-related association of fine particles and ozone with severe acute asthma in New York City. J. Allergy Clin. Immunol. 125 (2), 367−373. Available from: https://doi.org/10.1016/j.jaci.2009.10.061.

Srivastava, V., Ismail, S.A., Singh, P., Singh, R.P., 2015. Urban solid waste management in the developing world with emphasis on India: challenges and opportunities. Rev. Environ. Sci. Bio/Technol. 14 (2), 317−337. Available from: https://doi.org/10.1007/s11157-014-9352-4.

Tango, T., Fujita, T., Tanihata, T., Minowa, M., Doi, Y., Kato, N., et al., 2004. Risk of adverse reproductive outcomes associated with proximity to municipal solid waste incinerators with high dioxin emission levels in Japan. J. Epidemiol. 14, 83−93.

Triassi, M., Alfano, R., Illario, M., Nardone, A., Caporale, O., Montuori, P., 2015. Environmental pollution from illegal waste disposal and health effects: a review on the ‘‘Triangle of Death.”. Int. J. Environ. Res. Public. Health 12, 1216−1236. Available from: https://doi.org/10.3390/ijerph120201216.

United Nations Environment Programme, 2010. Waste and Climate Change: Global Trends and Strategy Framework. Division of Technology, Industry and Economics, International Environmental Technology Centre (IETC), Osaka/Shiga. Available from: https://wedocs.unep.org/20.500.11822/8648.

United States Environment Protection Agency, 2022. Criteria for the Definition of Solid Waste and Solid and Hazardous Waste Exclusions. Retrieved September 09, 2022 from https://www.epa.gov/hw/criteria-definition-solid-waste-and-solid-and-hazardous-waste-exclusions.

Viel, J.F., Arveux, P., Baverel, J., Cahn, J.Y., 2000. Soft-tissue sarcoma and non- Hodgkin’s lymphoma clusters around a municipal solid waste incinerator with high dioxin emission levels. Am. J. Epidemiol. 152, 13−19.

Williams, F.L., Lawson, A.B., Lloyd, O.L., 1992. Low sex ratios of births in areas at risk from air pollution from incinerators, as shown by geographical analyis and 3-dimensional mapping. Int. J. Epidemiol. 21, 311−319.

Yadav D., 2009. Garbage disposal plant mired in controversy. India Times, TNN, 19 Feb 2009.

Yukalang, N., Clarke, B., Ross, K., 2017. Barriers to effective municipal solid waste management in a rapidly urbanizing area in Thailand. Int. J. Environ. Res. Public. Health 14 (9), 1013. Available from: https://doi.org/10.3390/ijerph14091013.

Zohoori, M., Ghani, A., 2017. Municipal solid waste management challenges and problems for cities in low-income and developing countries. Int. J. Appl. Sci. Eng. 6 (2), 39−48.

Chapter 10

Production, characteristics and applications of biochar for environmental sustainability

Huijuan Shao[1] **and Sartaj Ahmad Bhat**[2]

[1]*College of Resources and Environment, Shandong Agricultural University, Tai'an, P.R. China,*
[2]*River Basin Research Center, Gifu University, Gifu, Japan*

10.1 Introduction

The world's population is currently increasing at a rate of about 1.05% annually, and it is expected to surpass 10 billion people in 2057 (Worldometers.info, 2021). This demographic growth plays an important role in the production of a notable quantity of solid waste, posing a serious challenge towards environmental sustainability. Meanwhile, the rising energy costs and global climate change caused by high levels of greenhouse gas emissions have prompted a lot of research into converting biomass into biofuels and other renewable products (Hossain et al., 2011). Solid waste treatment and disposal is a global challenge for the development of a sustainable environment. The problem has been worsened by the rise in waste products as a result of population growth, urbanisation and economic development. It is estimated that 1/4 to 1/3 of produced food is wasted worldwide (Bellemare et al., 2017). In addition, the world generates around 2.01 billion tons of municipal solid waste annually, and 33% of them are not managed appropriately. These data demonstrate the insistent demand for strategies to address the growing rate of solid waste generation worldwide.

As a result of the constant growth of urban and industrial waste volume and complexity, waste management has been identified as one of the urgent and intractable problems. Along with adverse environmental implications, the mismanagement of solid waste also poses risks to community welfare and creates numerous societal and economic issues (Istrate et al., 2020). The traditional landfill method is no longer suitable for the current urban development conditions because of the large occupying land area and the limited treatment capacity; additionally, landfill generates a large amount of

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00017-7

greenhouse gases. The over-reliance on landfill and inadequate waste disposal has continually troubled people with financial, health and safety concerns (Shah et al., 2021). The other common treatment technologies to dispose solid wastes are composting or open burning, which has the disadvantages of producing greenhouse and pollution gases and reducing the usage value of organic waste (Dunnigan et al., 2018; Yang et al., 2021).

Since most of the solid waste produced by living and production activities is rich in organic matter and nutrients, the technology of producing biochar through pyrolysis has many advantages, including wide sources, large quantities, and large application fields (Xiao et al., 2018). In an oxygen-limited environment, pyrolysis converts trees, grasses, crop residues or animal faeces into biochar, with twofold higher carbon content than ordinary biomass, and biochar locks up carbon in plant biomass in more durable forms (Baldock and Smernik, 2002). Under the temperature range of 300°C–1000°C, the pyrolysis process can effectively reduce the volume of solid wastes, kill intrinsic pathogens and parasites in animal faeces, and produce bio-oil, non-condensable gases (e.g. CO, CO_2, CH_4 and H_2), and carbon-rich biochar (Ahmad et al., 2012; Cantrell et al., 2012; Xu et al., 2019). This technology can mitigate global climate change since biochar is believed to store carbon in the soil for hundreds to thousands of years, potentially leading to a significant reduction in atmospheric greenhouse gas levels (Lehmann, 2007). The application of biochar to soil is also reported to be able to improve soil properties (such as pH, water retention and hydraulic conductivity) and plant growth (Van Zwieten et al., 2010). Moreover, biochar is widely used in wastewater treatment and soil remediation, as well as the catalysts for chemical synthesis and biofuel production (O'Connor et al., 2018; Xiong et al., 2017). Therefore, biochar production could be a win-win technology for energy, carbon storage and environmental functions (Lehmann, 2007).

This chapter reviews the production and physicochemical properties of biochar; the effects of production conditions, including pyrolysis temperature and duration, as well as activation/modification methods, and the applications of biochar to the environment are also explained.

10.2 Production of biochar

Many thermal technologies have been developed for biochar production, such as slow and fast pyrolysis, gasification, and hydrothermal carbonisation (HTC) (Kambo and Dutta, 2015). Pyrolysis is a mature technique for producing biochar with a high yield under an oxygen-limited environment at high temperatures (300°C–1000°C). Manara and Zabaniotou (2012) reported that low temperature was not enough to break down the chemical structure of organics. High temperatures can supply enough energy for secondary pyrolysis but may reduce the biochar yield (Xiao et al., 2022). Yuan et al. (2015) observed that the biochar yield decreased from 83.3% to 65.0%, and the pore structure was developed when the temperature increased from 300°C to 700°C. In addition, slow pyrolysis has typical heating rates between 1°C and 30°C min^{-1} (Lua et al., 2004) and is performed at longer residence time and moderate temperature (350°C–550°C), which was reported to obtain a higher yield of biochar (30%) than the fast pyrolysis (12%) or gasification (10%) (Inyang and Dickenson, 2015). Therefore, slow pyrolysis is often considered as the most feasible thermal treatment method for biochar production. Gasification is considered as a technology for recovering energy from syngas, as well as obtaining biochar, which has positive effects in sorption and electrochemical applications (You et al., 2017). Gasification transfers the heat value from biomass to syngas, bio-oil and biochar at high temperatures ($>$600°C) in the presence of gasifying agents.

HTC is a newly developing low-temperature (180°C–260°C) approach for producing hydrothermal char (hydrochar), which can dispose wet materials directly (Cui et al., 2022). The yield and properties of hydrochar obtained from HTC are obviously different from that of pyrolysis-derived biochar and are mainly governed by temperature (Kambo and Dutta, 2015). Compared with dry carbonisation, HTC retains more organic carbon and has obvious advantages in the enrichment of effective elements (such as N and P) and the fixation of heavy metals.

10.3 Characteristics of biochar

Biochar derived from different feedstocks has different physicochemical properties. For instance, animal manure-derived biochar has a relatively lower specific surface area (SSA) and richer nutrient content than that derived from wood and plant (Agegnehu et al., 2017). Biochar has a very complex pore structure with different sizes, and the pores can be divided into large pores ($>$ 50 nm), small pores ($<$ 0.9 nm) and micropores ($<$ 2 nm) (Ahmad et al., 2014). The SSA of biochar is usually determined by its porosity, and biochar usually has a high SSA. The abundant pores and surface functional groups on biochar make it perfect for holding capacity for water and nutrients. The pore structure of biochar can also provide a place for beneficial microorganisms, such as mycorrhizas and bacteria, to live and reproduce.

Biochar is usually alkaline (pH = 5–12), and as a result, it can be used as an acid-correcting soil conditioner. Biochar contains many oxygen-containing functional groups such as carboxyl group, carbonyl group, phenolic hydroxyl group and lactone group on its surface. The rich oxygen-containing functional groups on the surface of biochar generate high adsorption capacity, hydrophilic or hydrophobic characteristics, and buffering ability against acid and base. Biochar has a high cation exchange capacity (CEC) because of the high negative charge density on its surface, which is due to the oxygen-containing active groups. Biochar contains a large number of nutrients required by plants. In addition to the high content of C, the contents of N, P, K, Ca and Mg are also high. The content of C and N decreases with the increase of pyrolysis temperature due to combustion and volatilisation, while the contents of K, Ca, Mg and P increase with the increase in temperature (Cao and Harris, 2010). The chemical composition of raw materials has an important effect on the elemental composition and content of biochar. Because of the concentration and enrichment of some nutrients during pyrolysis, the content of P, K, Ca and Mg in biochar is higher than that in the raw materials. Biochar has strong biological inertia and chemical stability (not easy to degrade).

10.4 Factors influencing the characteristics of biochar

The characterisation of biochar is of great importance for its potential applications. Biochar can show distinctly different physicochemical properties, such as SSA, polarity, pH, CEC, atomic ratio and element composition, which are feedstock types and processing conditions (temperature and duration) dependent (Keiluweit et al., 2010; Uchimiya et al., 2013; Zhao et al., 2013). Table 10.1 shows the properties of some biochars derived from different feedstocks under different pyrolysis temperatures and durations.

10.4.1 Feedstocks

Biochar is produced from various organic feedstocks that possess different physicochemical properties. Extensive biomass feedstocks have been used in the production of biochar, including bioenergy crops (willows, miscanthus and switchgrass), forest residues (wood chips and nut shells), agricultural wastes, animal manure, kitchen wastes and industrial waste (sewage sludge) (Angin, 2013; Cao and Harris, 2010; Nartey and Zhao, 2014). The properties of biochar vary with the types of feedstocks. For example, despite the abundant and valuable biomass, most wood wastes are directly disposed at landfills without recycling, which consumes precious land resources and produces a large amount of greenhouse gas emissions (USEPA, 2018). Biochar derived from lignocellulosic-rich wood biomass exhibits an easily tunable porous structure, aromatic carbon domain, and surface functionality for various environmental applications (Rangabhashiyam and Balasubramanian, 2019). Wood waste-derived biochar has a great potential to deal with various environmental issues with concurrent attainment of a circular economy (Hu et al., 2021). Li et al. (2019) found that at the same pyrolysis temperature, the pH value of agricultural waste-derived biochar was the highest, while the wood and herbal raw materials derived biochars were neutral or acidulous. Small aquatic plant-derived biochars were reported to have a larger ash contents than that made from emergent macrophytes and crop straw (Ahmad et al., 2014). Wang et al. (2020a,b) compared the biochar produced from raw rice straw, and the pellet biomass-derived biochar had a higher yield, with more mesopores and fewer macropores, indicating that pelletising can enhance the yield and porous

TABLE 10.1 Properties of biochar derived from different feedstocks under different pyrolysis temperatures and durations.

Feedstock	Temperature (°C)	Duration	Biochar properties	References
Rice straw	250–450	2–8 h	Increasing temperature and duration decreased biochar yield and volatile matter content but increased C, K, P contents and SSA, reduced O, H and aliphatic C functional groups, but concentrated aromatic C.	Peng et al. (2011)
Wood waste	450–950	30–120 min	Applying a higher pyrolysis temperature (> 750°C) or longer duration (> 60 min) is recommended to enhance the pore structure of biochar.	He et al. (2021)
Wood, wheat straw, dried algae	300–750	10–60 min	The pH in a solution, higher heating value and BET SSA positively correlated with pyrolysis temperature.	Ronsse et al. (2013)
Lychee branches	500	6 h	pH = 4.38, CEC = 12.2 cmol kg^{-1}	Liu et al. (2020)
Wetland plant–*Arundo donax*	300–600	1 h	As the temperature increased, the pH increased from 6.93 to 8.85, and the SSA increased from 10.65 to 281.15 m^2 g^{-1}.	Li et al. (2018)
Canna indica	300–600	2 h	As the temperature increased, the pH increased from 9.69 to 10.47, and the SSA increased from 3.46 to 10.40 m^2 g^{-1}.	Cui et al. (2016)
Sewage sludge	300–700	3 h	Higher-temperature biochar contained less N but more P and K.	Yuan et al. (2016)

CEC, Cation exchange capacity; *SSA*, specific surface area.

structure of biochar. The feedstock also plays an important role in the CEC, SSA and microporosity of biochar (Cui et al., 2022).

Microalgal biomass is suitable for its conversion into biochar, which is used as fertilizer and adsorbent (Law et al., 2021). Microalgae exist in fine particle form owing to their cellular structure, and their biomasses primarily contain carbohydrates, proteins, fats, ash and water, and particularly contain little or no lignin, which requires simpler pretreatment (Law et al., 2021). Microalgal biochar typically has more N and O content, as well as less C content compared to lignocellulosic biochar (Bird et al., 2011). Microalgal biochar consists of large aggregates with irregular porosity (Yu et al., 2017). Biochars made from microalgae normally have a lower surface area than lignocellulosic biomass; however, it could be improved by increasing pyrolytic temperature (Law et al., 2021). Algal-derived biochar can be used not only for soil amendment for agriculture purposes but can also be used as an adsorbent in wastewater treatment for the removal of organic or inorganic contaminants due to its high nutrient content and ion-exchange capacity (Yu et al., 2017).

10.4.2 Pyrolysis temperature

Pyrolysis temperature is the most significant factor affecting the aromaticity and stability of biochar (Amen et al., 2020). With the temperature increasing, both H/C and O/C ratios decrease, indicating a greater degree of aromaticity and thus influencing the contribution to CO_2 sequestration and soil fertility (Peng et al., 2011). The pH value of biochar usually shows an increase in tendency with increasing pyrolysis temperature due to the decomposition of acid functional groups, the enrichment of ash content, as well as the release of alkali salts (Windeatt et al., 2014; Irfan et al., 2016). Additionally, the hydrophobicity, aromaticity, and SSA of biochar also increased with the increasing temperature, and the increased SSA of biochar can provide more adsorption sites and space for nutrients and water retention (Keiluweit et al., 2010). A decrease in SSA was, however, found in some biochars produced at relatively high temperatures ($>$ 700°C), probably because of the collapse of micropores and blockage by minerals (Cui et al., 2022). The higher the pyrolysis temperature, the weaker the water-holding capacity of biochar. This is because the higher pyrolysis temperature makes the polar functional groups on the biochar surface become less, thereby enhancing its hydrophobicity and making it difficult to maintain soil interstitial water.

He et al. (2021) revealed that the DOC content considerably decreased from 226 to 10.4 $\mathrm{mg\,L^{-1}}$ with the increase of pyrolysis temperature from 450°C to 950°C, indicating the increase of fixed carbon contents and the removal of incomplete combustion by-products. Biochar-derived DOM plays a crucial role in the nutrient cycle and serves as potential substrates for indigenous microorganisms and competitive adsorption of contaminants, as well as surface blockage; therefore, it is important to understand the DOM characteristics of biochar (Sun et al., 2021b; He et al., 2021).

10.4.3 Pyrolysis duration

He et al. (2021) reported that a longer pyrolysis duration enhanced the surface area and stability of biochar, indicating that longer pyrolysis duration was conducive to the development of biochar inner structure. By increasing the pyrolysis time, the volume of both mesopores and micropores in the biochar could be enlarged, probably due to the driven-off of pore-blocking compounds initially formed during the incomplete biomass decomposition (He et al., 2021). A longer duration could also probably facilitate the graphitisation process.

10.4.4 Other factors

Apart from the temperature and duration, Demirbas (2004) reported that the biochar yield depended on particle size, heating rate and the quality of organic materials (such as lignin). Biomass with more lignin and less cellulose can generally produce a high yield of biochar; meanwhile, the porosity of biochar increases with the content of lignin in biomass (Nartey and Zhao, 2014). The volatile component, water content, particle size and shape of biomass feedstock will also affect the characteristics of obtained biochar (Nartey and Zhao, 2014).

The modification of biochar can alter its physicochemical characteristics. Through modification, the specific surface area and porosity of biochar can be increased, and various functional groups can be introduced on the biochar surface (Xiao et al., 2022). The modified methods mainly include acid (H_3PO_4, H_2SO_4, HCl, HF), alkali (NaOH, KOH), salt ($ZnCl_2$) activation and other activators (Xiao et al., 2022).

10.5 Environmental applications of biochar

Biochar is a structurally stable solid material with a high degree of aromatisation and anti-decomposition ability (Zhao et al., 2021). Environmental remediation is one of the primary applications of biochar. Because of its large specific surface area and abundance of oxygen-containing functional groups (Ali et al., 2020), biochar is emerging as an economical material to remove various organic contaminants such as agrochemicals, antibiotics, PAHs, PCBs, volatile organic compounds and aromatic dyes, and a series of inorganic contaminants, such as heavy metals, ammonia, nitrate, phosphate, and sulphide from aqueous, gaseous and/or solid phases (Ahmad et al., 2014; Oliveira et al., 2017; Xu et al., 2012). The pore structure, specific surface area and surface charge density of biochar determine its adsorption ability, especially for cationic. The main mechanism is surface adsorption, which can be achieved through a combination of chemical bonds or electrostatic attraction.

10.5.1 Soil amelioration and remediation

Biochar is widely used as a soil amendment to improve the soil structure and quality for crop growth (Sun et al., 2021a). Many researchers concluded that the application of biochar to soil generally increased soil pH, soil nutrients, soil microbial community and crop yield (Biederman and Harpole, 2013). Specifically, biochar application can: (1) neutralise acidic soils by increasing soil pH (Cao and Harris, 2010); (2) deepen soil colour, reduce soil surface reflectance, and promote soil heating; (3) control soil water distribution to improve soil water holding capacity; (4) decrease the volume-weight of soil and improve soil porosity, promote the transformation of soil into aggregates, reduce nutrient leaching, which is conducive to the growth and extension of plant roots and promote the absorption of nutrients by plants; (5) retain toxic compounds such as heavy metals and pesticides, and remediate agricultural non-point source pollution; (6) increase the contents of organic matter, total N and P, available N and P in soil, directly supply organic carbon, and maintain soil fertility. Therefore, due to the favourable characteristics in terms of pH, CEC, porosity and SSA, proper biochar can ameliorate degraded and low-fertility soils by providing or retaining plant nutrients. It is remarkable that the effect of biochar application on soil quality is greatly dependent on biochar types, application rates, soil properties and climatic conditions (Cui et al., 2022).

In the biochar-amended soil, the reproductive ability of microorganisms in the rhizosphere can be enhanced, and the microbial community structure is also changed, which has an important effect on the physiological and biochemical processes of plants (Steinbeiss et al., 2009); the seed germination, early growth of seedlings and crop yield can also be promoted (Solaiman et al., 2012; Hale et al., 2013). In addition, biochar application can promote root growth and plant hormone production and improve the ability of cold and drought resistance. The porous structure and adsorbed nutrients of biochar provide a suitable habitat for the soil microbial community, restore the health of the soil ecosystem, and improve the number and activity of microorganisms, especially those related to nitrogen cycling. Sun et al. (2021a) conducted a six-year field experiment to evaluate the long-term effects of different biochar application rates on soil aggregate stability and biological binding agents in brown soil; the results indicated that biochar increased small macroaggregates (0.25–2 mm) and the soil aggregate stability, increased the amount of biological binding agents and the soil pH, EC, CEC, exchangeable K^+, Na^+ and Mg^{2+} levels; therefore, the application of biochar at a suitable rate might improve the soil structure (Fig. 10.1).

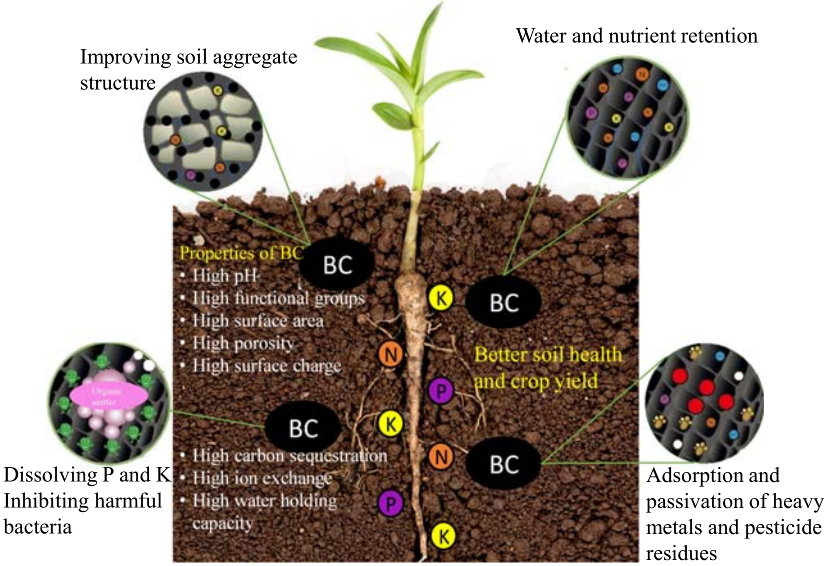


FIGURE 10.1 Physicochemical properties and soil amelioration of biochar (BC).

For the contaminated soil, biochar can adsorb or passivate contaminants in the soil and has become the forefront of in situ remediation due to its large SSA, abundant active functional groups, good complexation and adsorption properties (Zheng et al., 2013; Shao et al., 2021). As shown in Fig. 10.2, the biochar fixed heavy metals in the soil through physical and chemical adsorption, aiming to change the specific chemical forms of heavy metals in contaminated soil, and inhibiting the activity and bioavailability of heavy metals (Lu et al., 2014). For instance, Liu et al. (2020) revealed that the application of lychee biochar (2.5%−5%) enhanced sunflower growth and the remediation of heavy metals in contaminated soil. Tang et al. (2022) conducted field experiments to explore the remediation effects of ordinary biochar and polyethyleneimine (PEI)-modified biochar on the Cd-contaminated yellow-brown soil; the results indicated that both ordinary biochar and PEI-modified biochar could passivate Cd in soil and improve the soil environmental quality effectively; the PEI-modified biochar performed better than the ordinary biochar. Meanwhile, the remediation ability of biochar to heavy metal pollution is affected by pyrolysis temperature, application amount, biochar type and other factors.

10.5.2 Carbon sequestration

In order to mitigate the global climate change problem, the greenhouse gas emission must be reduced. Carbon sequestration is the capture and storage of

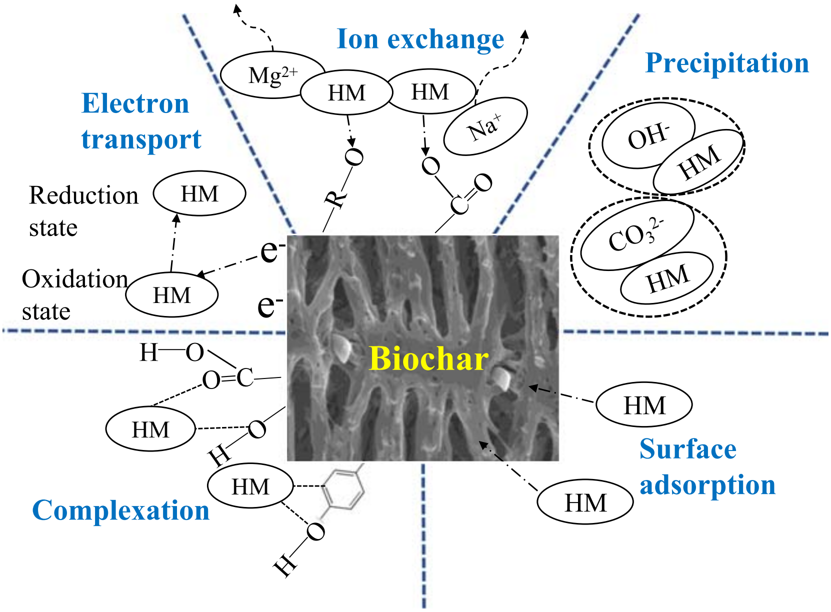


FIGURE 10.2 Main immobilisation mechanisms of heavy metal (HM) by biochar.

carbon to prevent it from releasing to the atmosphere. Biochar, as a highly stable carbon-rich material, can lock the carbon in its biomass during generation and storage, and avoid carbon entering the atmosphere through microbial decomposition, thus playing a positive role in reducing greenhouse gas emissions (CO_2, N_2O, CH_4, etc.) and affecting climate change. CO_2 is fixed in biomass through photosynthesis and is released by biomass decomposition. Biochar indirectly prevents the decomposition process of biomass, which makes carbon stored in a stable form and prevents carbon from being re-released to the atmosphere. This is the so-called "carbon-negative effect" of biochar. Kim et al. (2011) showed that the amount of CO_2 released by soil without biochar was significantly higher than that of soil with biochar. Biochar is a win-win strategy for mitigating global warming and addressing global food security. Therefore, the application of biochar to soil has double benefits, including long-term carbon sequestration and potentially positive soil change.

10.5.3 Water purification and treatment

Biochar is mainly used to remove pesticides, heavy metal ions and other organic solvents from wastewater in the field of wastewater purification. Table 10.2 shows the adsorption capacities of some biochar for various contaminants from water. The key to controlling water eutrophication is to control N and P. Among the commonly used methods, the adsorption method is a fast, efficient, no secondary pollution and low-cost method, and biochar can be used as an effective adsorbent. Biochar adsorbs ammonia nitrogen by electrostatic attraction and phosphorus, usually by a chemical bonding mechanism.

Biochar properties play a critical role in contaminant removal, which is often governed by pyrolysis temperature and feedstock type (Oliveira et al., 2017). Fully carbonised biochar produced at a relatively high pyrolysis temperature ($>500°C$) has more affinity for organic contaminants due to its greater surface area, microporosity, hydrophobicity, high C/N, high pH and low dissolved organic carbon (DOC) (Ahmad et al., 2014; Keiluweit et al., 2010; Ronsse et al., 2013; Uchimiya et al., 2013). On the other hand, partly carbonised biochar produced at a relatively low pyrolysis temperature ($<500°C$) is more appropriate for removing inorganic contaminants since it contains a relatively high content of DOC and oxygen-containing functional groups, low porosity and C/N (Ahmad et al., 2014; Keiluweit et al., 2010).

Sewage sludge from wastewater treatment plants usually contains numerous hazardous organic matter and heavy metals; these poisonous substances exhibit strong biological toxicity, which can easily cause a serious unknown threat to the environment and human health (Sun et al., 2020; Wang et al., 2021). Pyrolysis into biochar is one of the most promising thermochemical processes for the safe disposal of sewage sludge and possesses several advantages over sanitary landfill and incineration because of its energy recovery, nutrient recycling and heavy metal immobilisation (Ahmed and Hameed, 2020; Fan et al., 2020).

10.5.4 Other applications

Biochar can be used as a substrate to prepare fertilizer by adding different organic or inorganic components. Biochar-based fertilizer is rich in organic matter, such as humic acid, protein, nucleic acid, N, P, K and medium, and trace elements such as Ca, Mg, S, Fe, Mn, Cu, Zn. In addition, biochar-based slow-release fertilizers can slow down the release rate of fertilizer nutrients, improve the fertilizer usage rate of crops, improve soil ecology and reduce environmental pollution from agricultural production.

Biochar also has a strong adsorption capacity for some gases such as NH_3, CO, SO_2, H_2S and some gaseous Hg. The adsorption of harmful gases by biochar has the advantages of simple operation, economic feasibility and good effect. Therefore, biochar can be used in the field of gas pollution control. Shang et al. (2012) found that biochar made from camphor tree branches can effectively remove hydrogen sulphide from odorous gases.

The physicochemical properties, such as porous structure, aromatisation level and surface functionality of biochar, should be carefully tailored for different targeted applications (He et al., 2021). For example, (1) biochar with

TABLE 10.2 Adsorption capacity of various biochar (BC) to remove contaminants from wastewater.

BC type	Target contaminant	Adsorption capacity (mg g^{-1})	References
Modified yak dung BC	Fluoride and arsenic	3.928 and 2.926 for F_2 and As (V), respectively	Luo et al. (2018)
Oak wood and oak bark chars	Cr (VI)	3.03 for oak wood char and 4.61 for oak bark char	Mohan et al. (2011)
Municipal sewage sludge BC	Cd^{2+}	40 (BC production at 900°C)	Chen et al. (2015)
Hardwood char	Zn^{2+}	4.54 (BC production at 600°C)	Chen et al. (2011)
Taihu blue algae BC	Ni^{2+}	2.2 (BC production at 800°C)	Wang et al. (2020a,b)
Phyllostachys pubescens BC	Pb^{2+}	67.4 (BC production at 450°C)	Chao et al. (2017)
Durian rind BC	Congo red	87.32 (BC production at 800°C)	Thines et al. (2017)
BC-supported Mg $(OH)_2$/bentonite composite	Phosphate, ammonium, and humic acid	125.36 for phosphate, 58.20 for ammonium, 34.57 for humic acid	Jing et al. (2019)
Various types of BCs	Various pesticides	0.02–23 mg g^{-1} for pesticides removal	Inyang and Dickenson (2015)

a well-developed pore structure facilitated organic contaminants removal but suffered a lower yield; (2) biochar with a high energy value could be used as a solid fuel but was limited by energy consumption during high-temperature pyrolysis (Maksimuk et al., 2020); (3) biochar with abundant surface functional groups exhibited better metal removal capacity but might compromise its carbon stability in the environment (Xu et al., 2021).

10.6 Conclusions and future perspectives

Biochar has the characteristics of extensive raw materials and green and sustainable development. In today's world, where resources are increasingly scarce and environmental pollution is becoming more and more serious, using biomass waste materials with high carbon content to prepare biochar can avoid not only environmental pollution but also generate new energy, which is also a good way to recycle waste. In the resource-oriented

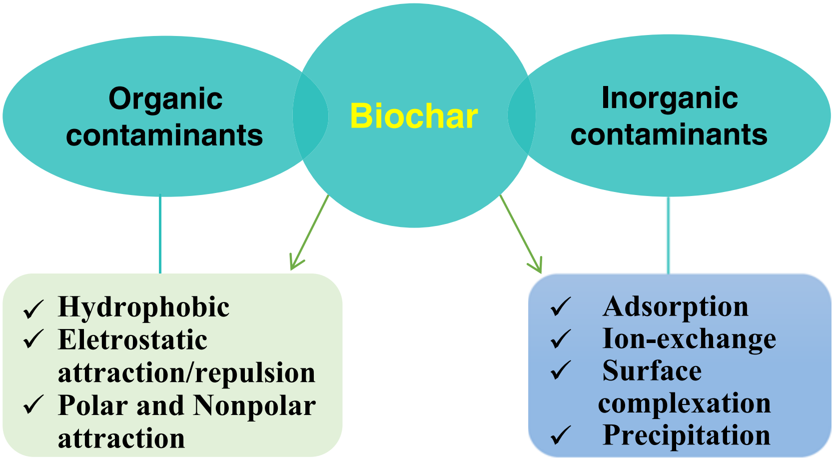


FIGURE 10.3 Mechanism of biochar interaction with organic and inorganic contaminants.

comprehensive waste treatment process, the production and use of biochar can not only solve the problem of harmless disposal of solid waste but also realise the final consumption through agricultural cultivation. It is also of great significance to the improvement of rural living and health conditions and agricultural production technology.

Previous investigations on biochar remained at the experimental stage, while applications to farmlands have not been widely carried out. Therefore, in future studies on biochar applied to soils, both short-term pot experiments and long-term (more than 5 years) field studies are required. In addition, practical biochar production can be affected by feedstock variation, equipment selection, operation mode and so on. Inappropriate production condition selection would inevitably hinder the economic feasibility and treatment efficiency, and even cause adverse environmental impacts of biochar applications. Biochar can introduce organic and inorganic pollutants to the environment under some circumstances. Therefore, in order to produce safe biochar, it is important to have recommendations for feedstock selection; local standards and performance specifications are also necessary for the expanding market and emerging biochar applications (Fig. 10.3).

References

Agegnehu, G., Srivastava, A.K., Bird, M.I., 2017. The role of biochar and biochar-compost in improving soil quality and crop performance: a review. Appl. SoilEcol. 119, 156–170.

Ahmad, M., Lee, S.S., Dou, X., Mohan, D., Sung, J.K., Yang, J.E., et al., 2012. Effects of pyrolysis temperature on soybean stover and peanut shell-derived biochar properties and TCE adsorption in water. Bioresour. Technol. 118, 536–544.

Ahmad, M., Rajapaksha, A.U., Lim, J.E., Zhang, M., Bolan, N., Mohan, D., et al., 2014. Biochar as a sorbent for contaminant management in soil and water: a review. Chemosphere 99, 19–33.

Ahmed, M.J., Hameed, B.H., 2020. Insight into the co-pyrolysis of different blended feedstocks to biochar for the adsorption of organic and inorganic pollutants: a review. J. Clean. Prod. 265, 121762.

Ali, A., Shaheen, S.M., Guo, D., Li, Y., Xiao, R., Wahid, F., et al., 2020. Apricot shell- and apple tree-derived biochar affectthe fractionation and bioavailability of Zn and Cd as well as the microbial activity in smelter contaminated soil. Environ. Pollut. 264, 114773.

Amen, R., Bashir, H., Bibi, I., Shaheen, S.M., Niazi, N.K., Shahid, M., et al., 2020. A critical review on arsenic removal from water using biochar-based sorbents: the significance of modification and redox reactions. Chem. Eng. J. 396, 125195.

Angin, D., 2013. Effect of pyrolysis temperature and heating rate on biochar obtained from pyrolysis of safflower seed press cake. Bioresour. Technol. 128, 593−597.

Baldock, J.A., Smernik, R.J., 2002. Chemical composition and bioavailability of thermally altered Pinus resinosa (Red pine) wood. Org. Geochem. 33, 1093−1109.

Bellemare, M.F., Çakir, M., Peterson, H.H., Novak, L., Rudi, J., 2017. On the measurement of food waste. Am. J. Agric. Econ. 99 (5), 1148−1158.

Biederman, L.A., Harpole, W.S., 2013. Biochar and its effects on plant productivity and nutrient cycling: a meta-analysis. GCB Bioenergy 5, 202−214.

Bird, M.I., Wurster, C.M., de Paula Silva, P.H., Bass, A.M., de Nys, R., 2011. Algal biochar−-production and properties. Bioresour. Technol. 102, 1886−1891.

Cantrell, K.B., Hunt, P.G., Uchimiya, M., Noval, J.M., Ro, K.S., 2012. Impact of pyrolysis temperature and manure source on physicochemical characteristics of biochar. Bioresour. Technol. 107, 419−428.

Cao, X., Harris, W., 2010. Properties of dairy-manure-derived biochar pertinent to its potential use in remediation. Bioresour. Technol. 101 (14), 5222−5228.

Chao, Z., Shan, B., Tang, W., Zhu, Y., 2017. Comparison of cadmium and lead sorption by Phyllostachys pubescens biochar produced under a low-oxygen pyrolysis atmosphere. Bioresour. Technol. 238, 352−360.

Chen, X., Chen, G., Chen, L., Chen, Y., Lehmann, J., McBride, M.B., et al., 2011. Adsorption of copper and zinc by biochars produced from pyrolysis of hardwood and corn straw in aqueous solution. Bioresour. Technol. 102, 8877−8884.

Chen, T., Zhou, Z., Han, R., Meng, R., Wang, H., Lu, W., 2015. Adsorption of cadmium by biochar derived from municipal sewage sludge: impact factors and adsorption mechanism. Chemosphere 134, 286−293.

Cui, X., Fang, S., Yao, Y., Li, T., Ni, Q., Yang, X., et al., 2016. Potential mechanisms of cadmium removal from aqueous solution by *Canna indica* derived biochar. Sci. Total. Environ. 562, 517−795.

Cui, X., Wang, J., Wang, X., Khan, M.B., Lu, M., Khan, K.Y., et al., 2022. Biochar from constructed wetland biomass waste: a review of its potential and challenges. Chemosphere 287, 132259.

Demirbas, A., 2004. Effects of temperature and particle size on bio-char yield from pyrolysis of agricultural residues. J. Anal. Appl. Pyrolysis 72, 243−248.

Dunnigan, L., Morton, B.J., Ashman, P.J., Zhang, X.P., Kwong, C.W., 2018. Emission characteristics of a pyrolysis-combustion system for the co-production of biochar and bioenergy from agricultural wastes. Waste Manag 77, 59−66.

Fan, J., Li, Y., Yu, H., Li, Y., Yuan, Q., Xiao, H., et al., 2020. Using sewage sludge with high ash content for biochar production and Cu (II) sorption. Sci. Total. Environ. 713, 136663.

Hale, S.E., Alling, V., Martinsen, V., Mulder, J., Breedveld, G.D., Cornelissen, G., 2013. The sorption and desorption of phosphate-P, ammonium-N and nitrate-N in cacao shell and corn cob biochars. Chemosphere 91 (11), 1612−1619.

He, M., Xu, Z., Sun, Y., Chan, P.S., Lui, I., Tsang, D.C.W., 2021. Critical impacts of pyrolysis conditions and activation methods on application-oriented production of wood waste-derived biochar. Bioresour. Technol. 341, 125811.

Hossain, M.K., Strezov, V., Chan, K.Y., Ziolkowski, A., Nelson, P.F., 2011. Influence of pyrolysis temperature on production and nutrient properties of wastewater sludge biochar. J. Environ. Manage. 92, 223−228.

Hu, Q., Jung, J., Chen, D., Leong, K., Song, S., Li, F., et al., 2021. Biochar industry to circular economy. Sci. Total. Environ. 757, 143820.

Inyang, M., Dickenson, E., 2015. The potential role of biochar in the removal of organic and microbial contaminants from potable and reuse water: review. Chemosphere 134, 232−240.

Irfan, M., Chen, Q., Yue, Y., Pang, R., Lin, Q., Zhao, X., et al., 2016. Co-production of biochar, bio-oil and syngas from halophyte grass (*Achnatherum splendens* L.) under three different pyrolysis temperatures. Bioresour. Technol. 211, 457−463.

Istrate, I.-R., Iribarren, D., Gálvez-Martos, J.-L., Dufour, J., 2020. Review of life-cycle environmental consequences of waste-to-energy solutions on the municipal solid waste management system. Resour. Conserv. Recycl. 157, 104778.

Jing, H.P., Li, Y., Wang, X., Zhao, J., Xia, S., 2019. Simultaneous recovery of phosphate, ammonium and humic acid from wastewater using a biochar supported $Mg(OH)_2$/bentonite composite. Environ. Sci. Water Res. Technol. 5, 931−943.

Kambo, H.S., Dutta, A., 2015. A comparative review of biochar and hydrochar in terms of production, physico-chemical properties and applications. Renew. Sustain. Energ. Rev. 45, 359−378.

Keiluweit, M., Nico, P.S., Johnson, M.G., Kleber, M., 2010. Dynamic molecular structure of plant biomass-derived black carbon (Biochar). Environ. Sci. Technol. 44, 1247−1253.

Kim, Y., Makoto, K., Takakai, F., et al., 2011. Greenhouse gas emissionsafter a prescribed fire in white birch-dwarf bamboo stands in northernJapan, focusing on the role of charcoal. Eur. J. Forest Res 130 (6), 1031−1044.

Law, X.N., Cheah, W.Y., Chew, K.W., Ibrahim, M.F., Park, Y.-K., Ho, S.-H., et al., 2021. Microalgal-based biochar in wastewater remediation: its synthesis, characterization and applications. Environ. Res. 111966.

Lehmann, J., 2007. A handful of carbon. Nature 447, 143−144.

Li, J., Fan, J., Zhang, J., Hu, Z., Liang, S., 2018. Preparation and evaluation of wetland plant-based biochar for nitrogen removal enhancement in surface flow constructed wetlands. Environ. Sci. Pollut. R. Int. 25, 13929−13937.

Li, S., Harris, S., Anandhi, A., Chen, G., 2019. Predicting biochar properties and functions based on feedstock and pyrolysis temperature: a review and data syntheses. J. Clean. Prod. 215, 890−902.

Liu, J., Huang, W., Mo, A., Ni, J., Xie, H., Hu, J., et al., 2020. Effect of lychee biochar on the remediation of heavy metal-contaminated soil using sunflower: a field experiment. Environ. Res. 188, 109886.

Lu, K., Yang, X., Shen, J., Robinson, B., Huang, H., Liu, D., et al., 2014. Effect of bamboo and rice straw biochars on the bioavailability of Cd, Cu, Pb and Zn to Sedum plum bizincicola. Agric. Ecosyst. Environ. 191, 124−132.

Lua, A.C., Yang, T., Guo, J., 2004. Effects of pyrolysis conditions on the properties of activated carbons prepared from pistachio-nut shells. J. Anal. Appl. Pyrol. 72, 279−287.

Luo, C., Tian, J., Zhu, P., Zhou, B., Bu, D., Lu, X., 2018. Simultaneous removal of fluoride and arsenic in geothermal water in Tibet using modified yak dung biochar as an adsorbent. R. Soc. Open. Sci. 5, 181266.

Maksimuk, Y., Antonava, Z., Krouk, V., Korsakova, A., Kursevich, V., 2020. Prediction of higher heating value based on elemental composition for lignin and other fuels. Fuel 263, 116727.

Manara, P., Zabaniotou, A., 2012. Towards sewage sludge based biofuels via thermochemical conversion – a review. Renew. Sustain. Energy Rev. 16, 2566–2582.

Mohan, D., Rajput, S., Singh, V.K., Steele, P.H., Pittman, C.U., 2011. Modeling and evaluation of chromium remediation from water using low cost bio-char, a green adsorbent. J. Hazard. Mater. 188, 319–333.

Nartey, O.D., Zhao, B., 2014. Biochar preparation, characterization, and adsorptive capacity and its effect on bioavailability of contaminants: an overview. Adv. Mater. Sci. Eng. 715398.

O'Connor, D., Peng, T., Zhang, J., Tsang, D.C.W., Alessi, D.S., Shen, Z., et al., 2018. Biochar application for the remediation of heavy metal polluted land: a review of in situ field trials. Sci. Total. Environ. 619–620, 815–826.

Oliveira, F.R., Patel, A.K., Jaisi, D.P., Adhikari, S., Lu, H., Khanal, S.K., 2017. Environmental application of biochar: current status and perspectives. Bioresour. Technol. 246, 110–122.

Peng, X., Ye, L.L., Wang, C.H., Zhou, H., Sun, B., 2011. Temperature- and duration-dependent rice straw-derived biochar: characteristics and its effects on soil properties of an Ultisol in southern China. Soil. Till. Res. 112, 159–166.

Rangabhashiyam, S., Balasubramanian, P., 2019. The potential of lignocellulosic biomass precursors for biochar production: performance, mechanism and wastewater application-a review. Ind. Crop. Prod. 128, 405–423.

Ronsse, F., Hecke, S., Dickinson, D., Prins, W., 2013. Production and characterization of slow pyrolysis biochar: influence of feedstock type and pyrolysis conditions. GCB. Bioenerg. 104–115.

Shah, A.V., Srivastava, V.K., Mohanty, S.S., Varjani, S., 2021. Municipal solid waste as a sustainable resource for energy production: state-of-the-art review. J. Environ. Chem. Eng. 9 (4), 105717.

Shang, G., Shen, G., Wang, T., et al., 2012. Effectiveness and mechanismsof hydrogen sulfide adsorption by camphor-derived biochar. J. Air Waste Manag. Assoc. 62 (8), 873–879.

Shao, H., Wei, Y., Wei, C., Zhang, F., Li, F., 2021. Insight into cesium immobilization in contaminated soil amended with biochar, incinerated sewage sludge ash and zeolite. Environ. Technol. Inno. 23, 101587.

Solaiman, Z.M., Murphy, D.V., Abbott, L.K., 2012. Biochars influence seed germination and early growth of seedlings. Plant. Soil. 353 (1–2), 273–287.

Steinbeiss, S., Gleixner, G., Antonietti, M., 2009. Effect of biochar amendment on soil carbon balance and soil microbial activity. Soil. Biol. Biochem. 41 (6), 1301–1310.

Sun, Y., Chen, G., Yan, B., Cheng, Z., Ma, W., 2020. Behaviour of mercury during Co-incineration of sewage sludge and municipal solid waste. J. Clean. Prod. 253, 119969.

Sun, Q., Meng, J., Lan, Y., Shi, G., Yang, X., Cao, D., et al., 2021a. Long-term effects of biochar amendment on soil aggregate stability and biological binding agents in brown earth. CATENA 205, 105460.

Sun, Y., Xiong, X., He, M., Xu, Z., Hou, D., Zhang, W., et al., 2021b. Roles of biochar-derived dissolved organic matter in soil amendment and environmental remediation: a critical review. Chem. Eng. J. 424, 130387.

Tang, B., Xu, H., Song, F., Ge, H., Chen, L., Yue, S., et al., 2022. Effect of biochar on immobilization remediation of Cd-contaminated soil and environmental quality. Environ. Res. 204, 111840.

Thines, K.R., Abdullah, E.C., Mubarak, N.M., 2017. Effect of process parameters for production of microporous magnetic biochar derived from agriculture waste biomass. Microporous Mesoporous Mater. 253, 29–39.

Uchimiya, M., Ohno, T., He, Z., 2013. Pyrolysis temperature-dependent release of dissolved organic carbon from plant, manure, and biorefinery wastes. J. Anal. Appl. Pyrol. 104, 84–94.

USEPA, 2018. Facts and figures about materials, waste and recycling (wood: mateiral-specific data). United States Environmental Protection Agency.

Van Zwieten, L., Kimber, S., Morris, S., Chan, K.Y., Downie, A., Rust, J., et al., 2010. Effects of biochar from slow pyrolysis of papermill waste on agronomic performance and soil fertility. Plant. Soil. 327, 235–246.

Wang, H., Wang, H., Zhao, H., Yan, Q., 2020a. Adsorption and Fenton-like removal of chelated nickel from Zn-Ni alloy electroplating wastewater using activated biochar composite derived from Taihu blue algae. Chem. Eng. J. 379, 122372.

Wang, T., Meng, D., Zhu, J., Chen, X., 2020b. Effects of pelletizing conditions on the structure of rice straw-pellet pyrolysis char. Fuel 264, 116909.

Wang, X., Chang, V.W.-C., Li, Z., Chen, Z., Wang, Y., 2021. Co-pyrolysis of sewage sludge and organic fractions of municipal solid waste: synergistic effects on biochar properties and the environmental risk of heavy metals. J. Hazard. Mater. 412, 125200.

Windeatt, J.H., Ross, A.B., Williams, P.T., Forster, P.M., Nahil, M.A., Singh, S., 2014. Characteristics of biochars from crop residues: potential for carbon sequestration and soil amendment. J. Environ. Manag. 146, 189–197.

Worldometers.info, World Population Clock: 7.9 Billion People, 2021 - Worldometer, 2021. [online] <https://www.worldometers.info/world-population/#> (accessed 17.08.21).

Xiao, Y., Igalavithana, A.D., Oh, S.-E., Nam, H., Zhang, M., Wang, C.-H., et al., 2018. Characterization of bioenergy biochar and its utilization for metal/metalloid immobilization in contaminated soil. Sci. Total. Environ. 640–641, 704–713.

Xiao, Y., Raheem, A., Ding, L., Chen, W.-H., Chen, X., Wang, F., et al., 2022. Pretreatment, modification and applications of sewage sludge-derived biochar for resource recovery- a review. Chemosphere 287, 131969.

Xiong, X., Yu, I.K.M., Cao, L., Tsang, D.C.W., Zhang, S., Ok, Y.S., 2017. A review of biochar-based catalysts for chemical synthesis, biofuel production, and pollution control. Bioresour. Technol. 246, 254–270.

Xu, T., Lou, L., Luo, L., Cao, R., Duan, D., Chen, Y., 2012. Effect of bamboo biochar onpentachlorophenol leachability and bioavailability in agricultural soil. Sci. Total. Environ. 414, 727–731.

Xu, Y., Qi, F., Bai, T., Yan, Y., Wu, C., An, Z., et al., 2019. A further inquiry into co-pyrolysis of straws with manures for heavy metal immobilization in manure-derived biochars. J. Hazard. Mater. 380, 120870.

Xu, Z., He, M., Xu, X., Cao, X., Tsang, D.C.W., 2021. Impacts of different activation processes on the carbon stability of biochar for oxidation resistance. Bioresour. Technol. 338, 125555.

Yang, Q.S., Mašek, O., Zhao, L., Nan, H.Y., Yu, S.T., Yin, J.X., et al., 2021. Country-level potential of carbon sequestration and environmental benefits by utilizing crop residues for biochar implementation. Appl. Energy. 282, 116275.

You, S., Ok, Y.S., Chen, S.S., Tsang, D.C.W., Kwon, E.E., Lee, J., et al., 2017. A critical review on sustainable biochar system through gasification: energy and environmental applications. Bioresour. Technol. 246, 242–253.

Yu, K.L., Lau, B.F., Show, P.L., Ong, H.C., Ling, T.C., Chen, W.H., et al., 2017. Recent developments on algal biochar production and characterization. Bioresour. Technol. 246, 2−11.

Yuan, H., Lu, T., Huang, H., Zhao, D., Kobayashi, N., Chen, Y., 2015. Influence of pyrolysis temperature on physical and chemical properties of biochar made from sewage sludge. J. Anal. Appl. Pyrol. 112, 284−289.

Yuan, H., Lu, T., Wang, Y., Chen, Y., Lei, T., 2016. Sewage sludge biochar: nutrient composition and its effect on the leaching of soil nutrients. Geoderma 267, 17−23.

Zhao, L., Cao, X., Mašek, O., Zimmerman, A., 2013. Heterogeneity of biochar properties as a function of feedstock sources and production temperatures. J. Hazard. Mater. 256−257, 1−9.

Zhao, R., Wang, B., Theng, B.K.G., Wu, P., Liu, F., Lee, X., et al., 2021. Fabrication and environmental applications of metal-containing solid waste/biochar composites: a review. Sci. Total. Environ. 799, 149295.

Zheng, R., Chen, Z., Cai, C., Wang, X., Huang, Y., Xiao, B., et al., 2013. Effect of biochar from rice husk, bran, and straw on heavy metal uptake by pot-grown wheat seedling in a historically contaminated soil. Biol. Res. 8, 5965−5988.

Chapter 11

Biochar: a sustainable solution for organic waste management a way forward towards circular economy

J.S. Sudarsan[1], **K. Prasanna**[2], **R. Shiam Babu**[2] **and V.M.V. Sai Krishna**[2]
[1]*School of Energy and Environment, NICMAR University, Pune, Maharashtra, India,*
[2]*Department of Civil Engineering, College of Engineering and Technology, SRM Institute of Science and Technology, Kattankulathur, Tamil Nadu, India*

11.1 Introduction

11.1.1 Origination

High energy and product consumption rates, along with rising population growth and living standards, result in great capacities for municipal solid waste generation, which pose severe environmental dangers if not properly disposed of or recycled. Garbage, trash, and refuse are all examples of municipal solid waste. Currently, globally, around 2 billion tonnes of municipal solid trash are produced, with approximately 33% of this rubbish going uncollected by municipalities. The normal waste generation rate of garbage per person daily is estimated to be 0.70–0.75 kg of garbage in the cities (Sunmathi et al., 2022; Sudarsan et al., 2022). The output of municipal solid garbage is predicted to increase by 2050, and 3.4 billion tonnes are expected according to the World Bank (Sunmathi et al., 2022; Sudarsan et al., 2022). Municipal solid garbage is disposed in landfills or dumpsites in about 70% of cases, while 19% is reprocessed and energy recovery accounts for 11% of the total (Sudarsan et al., 2022).

An urbanised country like India is promptly transitioning from farming to a business and service-related country, with about 34% of its inhabitants residing in cities. Despite this, its metropolitan population is the world's second-largest (Nanda and Berruti, 2021) established on a UN evaluation of 440 million urban residents and a normal per capita garbage productivity of

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00005-0

0.45 kg per day. By 2031, annual garbage creation is predicted to exceed 125 million metric tonnes, and by 2050, 300 million metric tonnes (Bing et al., 2016; Ahluwalia and Patel, 2018). Several researchers, based on their analysis and research studies, highlight the usage of biochar in waste management, especially organic waste management, gaining momentum. But with respect to commercial growth, the usage of biochar in various fields seems to be less. One of the market research teams from the transparency research market highlighted that biochar usage in different countries, especially American and Asian countries, is high compared to other countries like Europe middle east and Africa. The same is depicted in Fig. 11.1A. It also highlighted that the agriculture sector is one of the business sectors where biochar usage is high, as represented in Fig. 11.1B. Suppose Indian ULBs carry on to put their trust in landfills for unwanted waste administration, then in that case, about 1450 km^2 of land will be needed by 2050 to discard MSW in an efficient manner at the present rate of higher 1.3% per year (Bing et al., 2016; Ahluwalia and Patel, 2018). As a result, it is critical to handle organic solid waste in an environmentally responsible manner. Despite the widespread use of organic waste management processes, such as animal feeding, recycling, incineration, composting, biogas production and landfilling, nearly half of all organic wastes are left to decay in open areas, resulting in critical environmental effects, such as global warming, arousal and spread of xenobiotic oregano pollutants, pandemics and much more. As a result, a more practical method for efficiently managing organic wastes should be established. Biochar may be an appropriate conversion technique for effectively using energy from organic solid waste (Joshi and Ahmed, 2016).

11.2 Biochar (carbon-rich solid)

Carbon-rich solid is a high-carbon value product generated from natural waste feedstocks such as agricultural waste and municipal sewage sludge. This solid derivate was generated through the thermal breakdown process of biomass happening in the absence of oxygen, which is made from natural wood waste feedstock, adopted in different applications, and used for thousands of years. It can be created from any biogenic substance, despite the fact that charcoal is the well-known type. The technic adopted to produce this carbon-rich compound through thermochemical reaction with the distractive change of fuel at high temperatures in the absence of O_2 from the environment. The first step in this process is drying the biomass (Yang et al., 2007; Sudarsan and Srihari, 2019; Deeptha et al., 2015). Heating of carbon-rich substance at extreme temperature also release volatile components and stable gases when they react (e.g., methanol and acetic acid). Cracking and polymerisation are two gas-phase processes that can affect the entire product

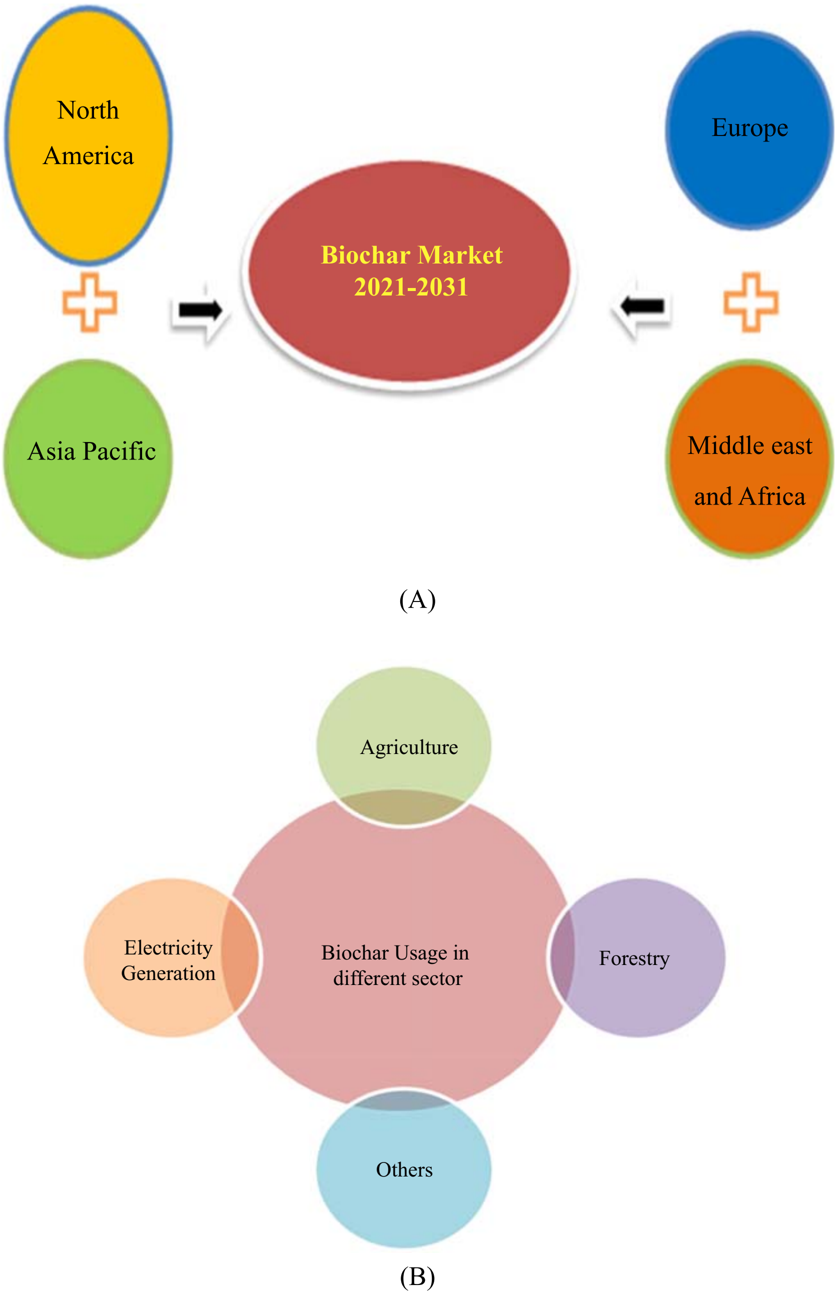


FIGURE 11.1 (A) Biochar market counties. (B) Biochar utility sector.

range. The three distinct products are lasting gases and single or additional liquid states (Sudarsan and Srihari, 2019).

The response paths to these various products are rather competitive, and process variables, such as process temperature and residence time, may alter product distribution. Several broad criteria depending on the process's desired output may be defined according to a definite product group (Deeptha et al., 2015). The efficiency of carbon-rich solid compounds depends on the process of production of the same. The entire process is carried out in a five-step process; in the first step, organic wood waste is collected, and in the meantime, the fabrication of the unit to execute the trial is also developed with plastic or acrylic material (Jayasanthakumari et al., 2015; Sudarsan et al., 2015). Then, collected biochar (carbon-rich solid) was produced by the pyrolysis process, and the different temperatures will be considered for a different type of product in making biochar. Then it is applied to the fabricated proto-type unit that processes the organic waste. A series of trials are carried out before it is applied on the field. Biochar produced from wood waste act as a good supplement to react with the organic waste and convert it into manure with rich nutrients. Manure produced by this process is nutrient-rich and best suited to use as organic fertilizer for crops. The schematic process of application and usage of biochar in waste management is depicted in Fig. 11.2.

Step 1: Selection of site and literature collection
Step 2: Selection of waste for the production of biochar
Step 3: Shredding and segregation of waste.
Step 4: Preparation of carbon-rich solid (Biochar) from waste and fabrication of rotary drum reactors
Step 5: Setting up two lab-scale rotary drum reactors
Step 6: Starting of trials in the two reactors (One reactor with the normal composting method and another reactor with biochar-mixed waste)
Step 7: Collection of samples and analysis for basic nutrient status and checking the efficiency of compost.
Step 8: Interpretation of results of different trials and field trials with the compost.
Step 9: Training the stakeholders with the developed concept.

11.3 Conventional production of carbon-rich solid

On the other hand, carbon-rich solid production primarily focuses on a carbonaceous solid product. The percentage of stable carbon content that forms dense compounds becomes greater when water evaporates, releasing volatile components. Organic polymerisation molecules in vapour and gases are thought to result in subordinate char production and higher solid relent. The pace of heating is short, and the stay is prolonged. In the old-style charcoal

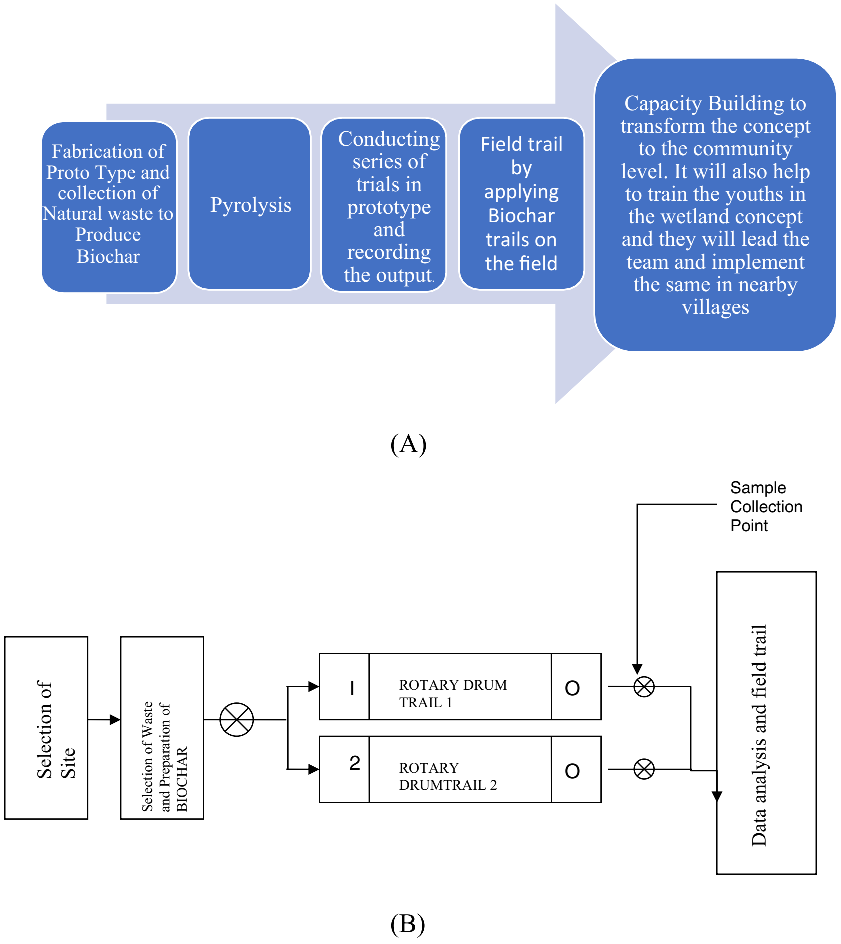


FIGURE 11.2 (A) Methodology of production of biochar manure. (B) Schematic representation of production of biochar-based manure.

manufacturing method, carbonisation can take weeks (charcoal pit). In the measured pyrolysis process, the temperature is typically approximately 500°C, but this varies depending on the wanted product grade. Treatment temperatures of up to 1000°C may be required for high-carbon content feedstock, which is achievable in woody feedstock, easier to work with, whereas in agriculture, waste and supplementary constituents with low ash melting temperatures are highly challenging. As a result, temperatures beyond 700°C are rarely reached (Gupta et al., 2015a,b,c; Xie et al., 2015; Tan et al., 2015). Refraction is the term for pyrolysis that occurs between 200 and 300°C. The main determination is to keep and focus the majority of the power in the compacted substance so that it will attain its full strength.

It also pointedly increases the mechanical qualities of the biomass (such as grind ability), which might be limiting in some adaptability of the product. Aside from development parameters, feedstock attributes have an impact on the conversion procedure and product characteristics. The three organic molecules cellulose, hemicelluloses, and lignin make up the majority of biomass (Prasanna et al., 2017; Xu et al., 2013). Because they perform differently throughout the hotness action, the configuration of the biomass has a direct impact on product production and attributes. Hemicelluloses are polysaccharides having a divided sequence structure. It decomposes at temperatures varying from 220°C to 315°C, making it the most reactive of the three major components (Cantrell et al., 2012).

The key mechanism in refraction is the breakdown of hemicelluloses in biomass. The most important changes in practically all parameters occur in the short hotness range of hemicellulose breakdown, making refraction highly sensitive and difficult to regulate the process. Cellulose is likewise a polysaccharide, but unlike hemicelluloses, it has an unbranched structure. It is more thermally constant, disintegrating at temperatures ranging from 280°C to 400°C (Xu and Wang, 2015). Because cellulose is the most prevalent organic substance on the planet, it has been researched for over 10 years. Nonetheless, the thermal breakdown is not completely recognised. Lignin is a very small complicated three-dimensional, molecule with many chemical connections. As a result, unlike hemicellulose and cellulose, breakdown does not occur within a limited temperature range. Thermal deterioration begins at 200°C and can last up to 900°C (depending on residence time) (Cantrell et al., 2012). Because of the varying thermal stabilities, the amount of lignin compound, cellulose, and semi-cellulose dictate the actual action temperature so that good quality product is the outcome with mass production for any given combination of conditions (Xu and Wang, 2015; Chen et al., 2015).

11.4 Modern applications

Carbon-rich solid is used to absorb massive metals, organic contaminants, nitrogen, and phosphorus impurities, as well as other pollutants from aqueous solutions (Deeptha et al., 2015; Jayasanthakumari et al., 2015). It was revealed that faeces char is more essential than rice husk char at separating metals, and it is also used as an adsorbent and manure in many situations. Heavy metals were absorbed on ionised hydroxyl-O-1 groups by faeces char; however, only O-groups were absorbed by rice husk char, demonstrating that mineral compounds and oxygen-functional groups are significant in heavy metal sorption on carbon-rich solid surfaces. Several researchers adopted microorganisms mixed with biochar in the process of treatment. Many researchers have tried *Ipomoea fstulosa* biochar to extract cadmium (Cd) from an aqueous solution at 350°C−550°C (Sudarsan et al., 2015; Lalander et al., 2013). The carbon-rich solid that had been activated by potassium hydroxide solution to increase surface area resulted in improved Cd separate efficiency (Liu et al., 2017). Carbon-rich solid was made from organic macroalga via gradual pyrolysis. Mist activation interchange cation exchange capacity and particular surface area, increasing copper detach efficiency. In contrast, potassium hydroxide solution-activated biochar increases pore fraction and decreases cation conversation capacity. This helps in copper removal efficiency. The physicochemical properties of carbon-rich solids made from empty oil palm fruit bunches were studied. Empty fruit bunch carbon-rich solid absorbed more Zn, Pb, and Cu than rice husk carbon-rich solid, although with a smaller surface area. The oxygen-restraining functional group, polarity index, and O/C molar ratio, according to their findings, are more relevant for heavy metal adsorption than surface area. Sudarsan et al. (2015) synthesised a carbon-rich solid from eucalyptus sawdust and activated it using tartaric, acetic and citric acids. Methylene blue was then extracted from the aqueous solution using activated charcoal. According to the results of the tests, citric acid-activated biochar was used to activate good efficiency carbon-rich solid made from eucalyptus sawdust. As a citric acid-activated, carbon-rich solid has a higher carboxyl group (COOH) concentration than other types of biochar. Every year, 0.5 million people die as a result of poor sanitation, filthy water and disease transmission [WHO, 7 February 2018]. Access to safe, affordable, and reliable drinking water is critical for public welfare in impoverished countries. Carbon-rich solids made from agricultural waste, forest waste or other biomass could be a good way to clean water. According to research, harmful bacteria (Gupta et al., 2015a,b,c), organics (Xie et al., 2015), inorganic, arsenic, phosphate and nitrate can all be removed by carbon-rich solid and its derivatives.

Researchers are developing new technology for electrical energy storage while also improving existing choices, such as batteries and supercapacitors, carbon-rich solid made from biomass has proven to be an effective sorbent for heavy metal removal from air and water. After this procedure, disposing of heavy metal-laden carbon-rich solid is a difficult undertaking (Prasanna et al., 2017; Xu et al., 2013; Cantrell et al., 2012). The physical and electrochemical characteristics of carbon-rich solids were discovered to be dependent on the manufacturing process, source and activation technique (Xu et al., 2013; Cantrell et al., 2012; Xu and Wang, 2015). Extensive study has been undertaken to evaluate the effectiveness of microbial fuel cells employing carbonaceous materials, such as start carbon and carbon nanomaterials as electrodes (Chen et al., 2015; Goswami et al., 2016; Ogbonnaya and Semple, 2013). Carbon-rich solid capacitors have substantially greater capacitance than the original carbon-rich solid capacitor. Lia et al. carbonised ash tree waste at 700°C to form a carbon-rich solid table, which was then chemically operated in an Ag_2SO_4/HNO_3 solution (Cao et al., 2009; Ter Laak et al., 2006; Ogbonnaya and Semple, 2013). When compared to the same carbon-rich solid activated with HNO_3, the Ag− carbon-rich solid demonstrated superior performance and stability (Dai et al., 2017).

For high capacitance, carbon-rich solid activation is essential. The charcoal is collaborated with a strong base and cooked at a very high temperature (> 900°C) in the classic method (Liu et al., 2017). This procedure, on the other hand, took a lot longer. Major (2017) used pine carbon-rich solid to create and test an energy-efficient low-temperature plasma activation process. Low-temperature plasma activation yielded a high capacitance biochar. Thines et al. (2017), Han et al. (2016) made magnetic biochar with three various metallic salts. During the polymerisation phase, the magnetic charcoal was implanted with polyp role (PPY) with increased specific capacitance and magnetic biochar. This substance has the ability to be employed in the development of low-cost supercapacitors in the future. Hydrogen is a clean, renewable energy source that has the potential to replace present fossil-fuel-based technology in a variety of industries. Hydrogen has a higher energy efficiency (60%) than gasoline (22%) or diesel (45%) and is expected to become more popular in the future (Yavari et al., 2015). The safe and cost-effective storage of hydrogen is the bottleneck in the hydrogen fuel industry; finding a cost-effective hydrogen storage mechanism has proven difficult thus far. Molecular adsorption, diffusion, chemical bonding, Vander Wall attraction and dissociation are all methods for storing hydrogen Hilber et al. (2017a,b). Adsorption on carbon-based materials is the most appropriate technique because of its chemical stability, reversibility and cost-effectiveness Hilber et al. (2017a,b). Hydrogen could be stored using activated carbon.

By using both conventional and microwave pyrolysis, a carbon-rich solid was made from tamarind seeds. After that, the carbon-rich solid was thermally powered by a potassium hydroxide solution, and hydrogen storage was evaluated. To boost high hydrogen storage capacity, they looked at a number of variables, including potassium hydroxide solution, pyrolysis condition and hydrogen adsorption condition. They discovered that microwave pyrolysis produced more micropores in carbon-rich solids, whereas conventional pyrolysis produced more mesopores in carbon-rich solids (Nag et al., 2011). They discovered that surface area and pore volume have a big impact on hydrogen storage capacity. Carbon-rich solid from palmyra sprout was produced using pyrolysis in a muffle furnace at 350°C with a heating rate of 5°C/min to make this type of biochar from natural products (Ter Laak et al., 2006; Logan et al., 2007). During the process of wastewater and water treatment, the fine carbon-rich solid is difficult to remove from a liquid media, as was the case when carbon-rich solid was used in water purification (Kouchachvili and Entchev, 2017); it has a sorption capacity for heavy metal removal from aqua solution. As a result, getting the carbon-rich solid particle out of the aqua solution requires a costly filtration or centrifugation procedure. Magnetic carbon-rich solids can be used to collect toxic metals from aqua solutions at a low cost. Magnetic carbon-rich solids can be made by mixing metal particles into the charcoal matrix (Kouchachvili and Entchev, 2017).

To isolate the magnetic carbon-rich solid grain from the aqua solution, a magnetic field can be applied. Cr (VI) was removed from an aqua solution using a magnetic carbon-rich solid (Kouchachvili and Entchev, 2017). A comparison of magnetic carbon-rich solids and normal carbon-rich solids was conducted. Although magnetic carbon-rich solid has many benefits, it is difficult to produce the same in the conventional way (Logan et al., 2007). So, it can be concluded that magnetic carbon-rich can be used based on the specific purpose and application. But conventional carbon-rich solids can be widely used in several applications, and it can also be produced in large quantities.

11.5 Biochar (carbon-rich solid) as a catalyst for organic waste treatment

Based on the above-mentioned procedure, organic biochar was produced. Mainly this biochar was manufactured using coconut shell waste as the main raw material. It was manufactured at a controlled temperature by the modern pyrolysis process (Kouchachvili and Entchev, 2017; Jin et al., 2013; Das et al., 2020; Anand et al., 2022). This carbon-rich solid was produced, and it was used as a mixed soil substrate. In this trial study, the biochar-amended soil is mixed with the contaminated soil in the rotary kiln. Then it must be mixed properly (Combination of biochar with soil) with the unit in equal proportion so that soil reclamation will be effective (Anand et al., 2022; Gupta et al., 2015a,b,c; Thines et al., 2017). Several trials were conducted, and the results are tabulated in the following Table 11.1. The average results of the several trials are tabulated below at different time intervals. Biochar amended soil unit (BAS) and Normal System (NS) without biochar. By the anaerobic decomposition of the organic waste by the action of microorganisms, the degradation process starts in one day and reaches the maximum by

TABLE 11.1 Contaminated soil reclamation with biochar @retention time of 120 h.

Parameters (mg/L) Except pH	One day		Second day		Third day		Fourth day	
	BAS	**NS**	**BAS**	**NS**	**BAS**	**NS**	**BAS**	**NS**
pH	7.5	7.1	7.4	7.2	7.6	7.4	7.9	7.4
TDS	2455	2560	1800	2100	1300	1500	500	750
TN	61.6	56.2	54.3	41	36	26	19	12.1
Chromium	51.5	55	39	43	32	36	25	29

TABLE 11.2 Biochar-based organic waste treatment by anaerobic digestion (AD) (after treatment).

Parameters (wt.%)	Anaerobic digestion (AD) treatment
p	2.7
K	4.8
N	2.7
S	0.7
C	27

the fourth day of the process, and the samples were collected every 24 hours till 120 hours as depicted in Tables 11.1 and 11.2.

Based on the comparison of the results with the normal soil composites, we can infer that there is around a 40% increase in the major nutrient status of biochar-amended soil after treating the organic waste. Based on the outcome of the above results, it was proved that compared to the normal system (NS), the biochar-amended unit (BAS) treatment efficiency was good (Prasanna et al., 2017; Xu et al., 2013; Cantrell et al., 2012). It is also evident that the reduction of pollutants takes place from day two of the trial, and it reaches its peak on day four of the trial. So, it is concluded that maximum treatment efficiency can be achieved at 96 hours, that is, day four in the treatment unit. Although in the above table, it represented a good reduction in chromium, it is difficult to reclaim the metals from the soil as it is costly to process and is influenced by several factors. The influencing factors lead to both positive and negative benefits of this treatment process (Cantrell et al., 2012; Xu and Wang, 2015). So detailed preliminary study needs to be carried out before adopting the process in a large scale. It is advisable to adopt this strategy to treat organic waste or reclaim the soil contaminated by organic contaminants.

11.6 Limitations of biochar technology

When related to the testing site or greenhouse carbon-rich solid applications, outdoor carbon-rich solid applications often result in lower pollutant sorption due to the influence of several external factors. Carbon-rich solid is administered directly in the field under natural settings; it appears to have little effect on soil optimisation. This could be due to the difficulty of controlling environmental conditions, such as temperature, breeze, humidity, precipitation, species of bacteria, ionic strength and so on (Tan et al., 2015; Prasanna et al., 2017). Aside from these environmental problems, researchers are concerned that carbon-rich solids may interfere with the effects of some beneficial compounds and function as a source of pollution itself. The addition of carbon-rich solids has also been shown to affect the efficiency of DNA extraction in soil. This may affect the precision with which soil microbial abundance and diversity are assessed and compared; however, this was affected by pyrolysis temperature, extractable C, and soil incubation (Xu et al., 2013). Some laboratory or greenhouse studies have reported on the influence of temperature on carbon-rich solid sorption capacity. (Cantrell et al., 2012) Rice straw carbon-rich solid produced at 450°C at 25°C had a good adsorption property of 15 mg g^{-1} for heavy metals like arsenic. The sorption capacity improved to 18 mg g^{-1} when the temperature was raised to 45°C. The temperature plays an important role in achieving effective biochar efficiency. The temperature of the soil fluctuates according to the time of day, night, and season. Despite the lack of field studies on the impact of temperature on carbon-rich solid presentation, laboratory studies reveal that changing temperatures have an impact on a carbon-rich solid performance. In the field, several external factors influence biochar sorption capacity efficiency. Heavy precipitation and runoffs may induce biochar compounds to penetrate deeper into the soil or bigger particles to float away, distancing them from impurities and reducing their chances of interaction and sorption (Xu and Wang, 2015). By degrading biochar particles, bacteria in the soil can impact carbon-rich solid performance in the short and medium haul. When bacterial growth decompose carbon-rich solids, pollutants may be transferred deep into the soil.

As biochar is strongly bound to carbon-rich solid matrices through interactions, these compounds are usually inaccessible for microbial breakdown (Goswami et al., 2016). The concentration of these compounds in carbon-rich solids is determined by the kind of pyrolysis technique (Ogbonnaya and Semple, 2013; Dai et al., 2017). Several researchers recommended that the study of a detailed characteristic needs to be carried out for this to be considered before using carbon-rich solids for pollutant cleaning (Liu et al., 2017; Major, 2017). An additional disadvantage of using carbon-rich solids in the farm is that it may diminish the efficacy of certain herbicides and pesticides used in the soil, such as atrazine (Goswami et al., 2016; Lalander et al., 2013). As previously noted, carbon-rich solid has a good sorption empathy for atrazine, which is readily restrained onto charcoal by organic and unit operation process of sorption (Han et al., 2016; Yavari et al., 2015). The entrapped herbicide or pesticide's efficiency in weed or insect control in soil may be greatly reduced (Cao et al., 2009; Yavari et al., 2015; Hilber et al., 2017a,b). The addition of 1% wheat straw to carbon-rich solid generated at 450°C raised the stunning dosage needed to eliminate ryegrass weed organic matter by 50% and 3.5 times, respectively (Logan et al., 2007; Kouchachvili and Entchev, 2017). This shows that using carbon-rich solids could lead to increased herbicide use and application expenditures, and also the spread of herbicide-resistant weeds (Kouchachvili and Entchev, 2017; Jin et al., 2013).

11.7 Future perspectives and conclusion

In recent years, biochar (Carbon-rich solid) based research has increased, emphasising its characteristics and how they affect its potential to encapsulate carbon-based and inert pollutants. Carbon-rich solid is still an effective soil remediation tool. Although the majority of carbon-rich solids give considerable sorption consequences greater than 45%–50% when adopted in the agricultural field without a preliminary investigation, a few carbon-rich solids may be unsuccessful in soil pollutant removal, especially in soil remediation (Logan et al., 2007). Electric force, rainfall, complex formation, and microbial activity dictate the interaction between carbon-rich solids and pollutants, and several factors highly influence these processes. Materials used for producing carbon-rich solids also play an important role. The third factor that needs to be looked into is the pyrolysis temperature. Finally, the contaminant's properties like hygroscopicity and polarity. During the proto-type trial, the sorption of pollutants is usually much higher than sorption in the ground (agricultural field) under normal situations. Starting materials for carbon-rich solid production, pyrolysis temperature, and contaminants properties like polarity. In the laboratory, Many investigations have been carried out on the combination of carbon-rich solids in soil (Kouchachvili and Entchev, 2017; Jin et al., 2013; Das et al., 2020; Anand et al., 2022). Only a few of these investigations have been conducted in the field. To completely study the dynamics of carbon-rich solids under such conditions, more research is needed on contaminated soil reclamation under optimal conditions (Jin et al., 2013). Significant effort is necessary to ensure case-specific and accurate carbon-rich solid usage in terms of carbon-rich solid type, the process of the proposed application rate, application rates and recovery processes. More research is needed to determine how much and what kind of carbon-rich solid a particular soil type can tolerate without compromising its normal function (Anand et al., 2022; Gupta et al., 2015a,b,c; Thines et al., 2017). While it is necessary to comprehend the processes of contaminant ion retention on carbon-rich solid, it is as crucial to comprehend how these mechanisms interact (i.e., whether their activities are well planned and they exertion self-sufficiently on each other) (Choi and Park, 2015; Niaz et al., 2015; Züttel et al., 2010). More effort will be required to clarify this issue. Given the high resistance of carbon-rich solids to biological degradation, nothing is known about what happens to individual carbon-rich solid particles over time (Das et al., 2020).

This study examined the different pyrolysis processes and reactor designs now in use or newly developed for carbon-rich solid synthesis (Choi and Park, 2015; Ramesh et al., 2015; Samantaray et al., 2019). Carbon-rich solid technologies, such as purification of water, wastewater, soil improvement, chemicals and contemporary solicitation, such as supercapacitors, fuel cells and hydrogen storage, among many others, are covered in the research.

The outcome efficiency of the trial is based on its acceptability for each application (Zhang et al., 2016, 2012; Kołodyńska and Bąk, 2018). As a result, several physical and chemical activation processes are used to improve the surface characteristics of carbon-rich solids. Carbon-rich solid adsorption was shown to be more cost-effective and efficient than other existing strategies for removing pollutants from aqueous solutions in the study (Choi and Park, 2015; Ramesh et al., 2015; Samantaray et al., 2019; Chen et al., 2011). As a result, carbon-rich solid-based wastewater treatment can be upgraded. Activated charcoal is gaining favour in the cosmetics sector as a unique ingredient (Zhang et al., 2012; Kołodyńska and Bąk, 2018). Other new carbon-rich solid applications include active medicinal ingredient carriers, thermal insulation for functional clothing, sole shoe deodorants, bad-odour filters in the food business, electromagnetic wave barriers, humidity control in the construction sector, and so on. Carbon-rich solid technology has had various success stories worldwide but general adoption and implementation of the technology have yet to materialise (Zhang et al., 2012; Kołodyńska and Bąk, 2018; Yang et al., 2017). The advantages of carbon-rich solids in various applications are well known, and several lab-scale and proto-type studies have also proven the same. It is high time to think about the business model and a constant supply chain to promote carbon-rich solid use in industries and other sectors. Furthermore, cradle-to-grave analysis and rating will substantially boost the sector, bolstering the prospects of carbon-rich solid as a renewable product.

References

Ahluwalia, I.J., Patel, U., 2018. Solid waste management in India: an assessment of resource recovery and environmental impact (No. 356). Working Paper. New Delhi: Indian Council for Research on International Economic Relations (ICRIER).

Anand, A., Kumar, V., Kaushal, P., 2022. Biochar and its twin benefits: crop residue management and climate change mitigation in India. Renew. Sustain. Energy Rev. 156, 111959. Available from: https://doi.org/10.1016/j.rser.2021.111959.

Bing, X., Bloemhof, J.M., Ramos, T.R.P., Barbosa-Povoa, A.P., Wong, C.Y., van der Vorst, J. G., 2016. Research challenges in municipal solid waste logistics management. Waste Manag. 48, 584–592.

Cantrell, K.B., Hunt, P.G., Uchimiya, M., Novak, J.M., Ro, K.S., 2012. Impact of pyrolysis temperature and manure source on physicochemical characteristics of biochar. Bioresour. Technol. 107, 419–428. Available from: https://doi.org/10.1016/j.biortech.2011.11.084.

Cao, X., Ma, L., Gao, B., Harris, W., 2009. Dairy-manure derived biochar effectively sorbs lead and atrazine. Environ. Sci. Technol. 43 (9), 3285–3291. Available from: https://doi.org/10.1021/es803092k.

Chen, B., Chen, Z., Lv, S., 2011. A novel magnetic biochar efficiently sorbs organic pollutants and phosphate. Bioresour. Technol. 102 (2), 716–723. Available from: https://doi.org/10.1016/j.biortech.2010.08.067.

Chen, Z., Hu, T.Q., Jang, H.F., Grant, E., 2015. Modification of xylan in alkaline treated bleached hardwood kraft pulps as classified by attenuated total-internal-reflection (ATR)

FTIR spectroscopy. Carbohydr. Polym. 127, 418–426. Available from: https://doi.org/10.1016/j.carbpol.2015.03.084.

Choi, Y.K., Park, S.J., 2015. Hydrogen storage capacity of highly porous carbons synthesized from biomass-derived aerogels. Carbon Lett. 16 (2), 127–131. Available from: https://doi.org/10.5714/CL.2015.16.2.127.

Dai, Z., Webster, T.M., Enders, A., Hanley, K.L., Xu, J., Thies, J.E., et al., 2017. DNA extraction efficiency from soil as affected by pyrolysis temperature and extractable organic carbon of high-ash biochar. Soil. Biol. Biochem. 115, 129–136.

Das, S.K., Ghosh, G.K., Avasthe, R., 2020. Application of biochar in agriculture and environment, and its safety issues. Biomass Convers. Biorefinery 1–11. Available from: https://doi.org/10.1007/s13399-020-01013-4.

Deeptha, V.T., Sudarsan, J.S., Baskar, G., 2015. Performance and cost evaluation of constructed wetland for domestic waste water treatment. J. Environ. Biol. 36 (5), 1071.

Goswami, R., Shim, J., Deka, S., Kumari, D., Kataki, R., Kumar, M., 2016. Characterization of cadmium removal from aqueous solution by biochar produced from Ipomoea fistulosa at different pyrolytic temperatures. Ecol. Eng. 97, 444–451. Available from: https://doi.org/10.1016/j.ecoleng.2016.10.007.

Gupta, N., Yadav, K.K., Kumar, V., 2015a. A review on current status of municipal solid waste management in India. J. Environ. Sci. 37, 206–217.

Gupta, P., Ann, T.W., Lee, S.M., 2015b. Use of biochar to enhance constructed wetland performance in wastewater reclamation. Environ. Eng. Res. 21 (1), 36–44.

Gupta, R.K., Dubey, M., Kharel, P., Gu, Z., Fan, Q.H., 2015c. Biochar activated by oxygen plasma for supercapacitors. J. Power Sources 274, 1300–1305. Available from: https://doi.org/10.1016/j.jpowsour.2014.10.169.

Han, L., Ro, K.S., Sun, K., Sun, H., Wang, Z., Libra, J.A., et al., 2016. New evidence for high sorption capacity of hydrochar for hydrophobic organic pollutants. Environ. Sci. Technol. 50 (24), 13274–13282.

Hilber, I., Bastos, A.C., Loureiro, S., Soja, G., Marsz, A., Cornelissen, G., et al., 2017a. The different faces of biochar: contamination risk versus remediation tool. J. Environ. Eng. Landsc. Manag. 25 (2), 86–104.

Hilber, I., Mayer, P., Gouliarmou, V., Hale, S.E., Cornelissen, G., Schmidt, H.P., et al., 2017b. Bioavailability and bioaccessibility of polycyclic aromatic hydrocarbons from (post-pyrolytically treated) biochars. Chemosphere 174, 700–707.

Jayasanthakumari, H., Krishnamoorthy, P., Arumugam, T.K., 2015. Kinetics, thermodynamics and isotherm studies on the removal of chromium by *Typha latifolia* bioremediator stem carbon. J. Mater. Environ. Sci. 6 (6), 1532–1541.

Jin, H., Wang, X., Gu, Z., Polin, J., 2013. Carbon materials from high ash biochar for supercapacitor and improvement of capacitance with HNO_3 surface oxidation. J. Power Sources 236, 285–292. Available from: https://doi.org/10.1016/j.jpowsour.2013.02.088.

Joshi, R., Ahmed, S., 2016. Status and challenges of municipal solid waste management in India: a review. Cogent Environ. Sci. 2 (1), 1–18.

Kołodyńska, D., Bąk, J., 2018. Use of three types of magnetic biochar in the removal of copper (II) ions from wastewaters. Sep. Sci. Technol. 53 (7), 1045–1057. Available from: https://doi.org/10.1080/01496395.2017.1345944.

Kouchachvili, L., Entchev, E., 2017. Ag/Biochar composite for supercapacitor electrodes. Mater. Today Energy 6, 136–145. Available from: https://doi.org/10.1016/j.mtener.2017.09.002.

Lalander, C., Dalahmeh, S., Jönsson, H., Vinnerås, B., 2013. Hygienic quality of artificial greywater subjected to aerobic treatment: a comparison of three filter media at increasing organic loading rates. Environ. Technol. 34 (18), 2657−2662. Available from: https://doi.org/10.1080/09593330.2013.783603.

Liu, S., Huang, B., Chai, L., Liu, Y., Zeng, G., Wang, X., et al., 2017. Enhancement of As (V) adsorption from aqueous solution by a magnetic chitosan/biochar composite. RSC Adv. 7 (18), 10891−10900.

Logan, B., Cheng, S., Watson, V., Estadt, G., 2007. Graphite fiber brush anodes for increased power production in air-cathode microbial fuel cells. Environ. Sci. Technol. 41 (9), 3341−3346. Available from: https://doi.org/10.1021/es062644y.

Major, J., 2017. Guidelines on practical aspects of biochar application to field soil in various soil management systems. Int. Biochar Initiative 8, 5−7.

Nag, S.K., Kookana, R., Smith, L., Krull, E., Macdonald, L.M., Gill, G., 2011. Poor efficacy of herbicides in biochar-amended soils as affected by their chemistry and mode of action. Chemosphere 84 (11), 1572−1577.

Nanda, S., Berruti, F., 2021. Municipal solid waste management and landfilling technologies: a review. Environ. Chem. Lett. 19 (2), 1433−1456.

Niaz, S., Manzoor, T., Pandith, A.H., 2015. Hydrogen storage: materials, methods and perspectives. Renew. Sustain. Energy Rev. 50, 457−469. Available from: https://doi.org/10.1016/j.rser.2015.05.011.

Ogbonnaya, U., Semple, K.T., 2013. Impact of biochar on organic contaminants in soil: a tool for mitigating risk? Agronomy 3 (2), 349−375.

Prasanna, K., Sudarsan, J.S., Nithiyanantham, S., 2017. Wastewater treatment using combined biological and constructed wetlands technique in paper mills. Sustain. Water Resour. Manag. 3 (4), 431−439.

Ramesh, T., Rajalakshmi, N., Dhathathreyan, K.S., 2015. Activated carbons derived from tamarind seeds for hydrogen storage. J. Energy Storage 4, 89−95. Available from: https://doi.org/10.1016/j.est.2015.09.005.

Samantaray, S.S., Mangisetti, S.R., Ramaprabhu, S., 2019. Investigation of room temperature hydrogen storage in biomass derived activated carbon. J. Alloy. Compd. 789, 800−804. Available from: https://doi.org/10.1016/j.jallcom.2019.03.110.

Sudarsan, J.S., Srihari, V., 2019. Evaluation of adsorption capacity of biochar mixed substrate to treat tannery wastewater by constructed wetland, AIP Conference Proceedings, Vol. 2112. AIP Publishing LLC, p. 020176, No. 1.

Sudarsan, J.S., Roy, R.L., Baskar, G., Deeptha, V.T., Nithiyanantham, S., 2015. Domestic wastewater treatment performance using constructed wetland. Sustain. Water Resour. Manag. 1 (2), 89−96.

Sudarsan, J.S., Vaishampayan, S., Parija, P., 2022. Making a case for sustainable building materials to promote carbon neutrality in Indian scenario. Clean. Technol. Environ. Policy 1−9.

Sunmathi, N., Padmapriya, R., Sudarsan, J.S., 2022. Feasibility study of tannery waste as an alternative for fine aggregate in concrete. Sustainable Construction Materials. Springer, Singapore, pp. 397−405.

Tan, X., Liu, Y., Zeng, G., Wang, X., Hu, X., Gu, Y., et al., 2015. Application of biochar for the removal of pollutants from aqueous solutions. Chemosphere 125, 70−85.

Ter Laak, T.L., Barendregt, A., Hermens, J.L., 2006. Freely dissolved pore water concentrations and sorption coefficients of PAHs in spiked, aged, and field-contaminated soils. Environ. Sci. Technol. 40 (7), 2184−2190.

Thines, K.R., Abdullah, E.C., Mubarak, N.M., Ruthiraan, M., 2017. In-situ polymerization of magnetic biochar−polypyrrole composite: a novel application in supercapacitor. Biomass Bioenergy 98, 95−111. Available from: https://doi.org/10.1016/j.biombioe.2017.01.019.

Xie, T., Reddy, K.R., Wang, C., Yargicoglu, E., Spokas, K., 2015. Characteristics and applications of biochar for environmental remediation: a review. Crit. Rev. Environ. Sci. Technol. 45 (9), 939−969.

Xu, F., Wang, D., 2015. Analysis of lignocellulosic biomass using infrared methodology. Pretreatment of Biomass. Elsevier, pp. 7−25. Available from: https://doi.org/10.1016/b978-0-12-800080-9.00002-5.

Xu, X., Cao, X., Zhao, L., 2013. Comparison of rice husk-and dairy manure-derived biochars for simultaneously removing heavy metals from aqueous solutions: role of mineral components in biochars. Chemosphere 92 (8), 955−961. Available from: https://doi.org/10.1016/j.chemosphere.2013.03.009.

Yang, H., Yan, R., Chen, H., Lee, D.H., Zheng, C., 2007. Characteristics of hemicellulose, cellulose and lignin pyrolysis. Fuel 86 (12-13), 1781−1788. Available from: https://doi.org/10.1016/j.fuel.2006.12.013.

Yang, P., Guo, D., Chen, Z., Cui, B., Xiao, B., Liu, S., et al., 2017. Removal of Cr (VI) from aqueous solution using magnetic biochar synthesized by a single step method. J. Dispers. Sci. Technol. 38 (11), 1665−1674. Available from: https://doi.org/10.1080/01932691.2016.1272058.

Yavari, S., Malakahmad, A., Sapari, N.B., 2015. Biochar efficiency in pesticides sorption as a function of production variables—a review. Environ. Sci. Pollut. Res. 22 (18), 13824−13841.

Zhang, M., Gao, B., Yao, Y., Xue, Y., Inyang, M., 2012. Synthesis of porous MgO-biochar nanocomposites for removal of phosphate and nitrate from aqueous solutions. Chem. Eng. J. 210, 26−32.

Zhang, J., Zhang, C., Zhou, N., Chen, W., 2016. The influence of microwave drying on leather properties. International Conference on Advanced Materials and Systems (ICAMS). The National Research & Development Institute for Textiles and Leather-INCDTP, pp. 189−194. Available from: https://doi.org/10.24264/icams-2016.i.29.

Züttel, A., Remhof, A., Borgschulte, A., Friedrichs, O., 2010. Hydrogen: the future energy carrier. Philos. Trans. R. Soc. A Math. Phys. Eng. Sci. 368 (1923), 3329−3342. Available from: https://doi.org/10.1007/s10800-008-9585-4.

Chapter 12

Solid waste negligence as an emerging environmental threat to ruminant health in resource-limited countries: a narrative review

Ishmael Festus Jaja[1,2], **Mhlangabezi Slayi**[3], **Leocadia Zhou**[3] **and James Wabwire Oguttu**[2]

[1]Department of Livestock and Pasture Science, University of Fort Hare, Alice, South Africa, [2]Department of Agriculture and Animal Health, University of South Africa, Johannesburg, South Africa, [3]Risk and Vulnerability Science Centre, Faculty of Science and Agriculture, University of Fort Hare, Alice, South Africa

12.1 Introduction

Solid waste accumulation is a global problem arising from human activities (Nongcula et al., 2017), which requires urgent intervention from governments around the world. In many developing countries, it is, however, evident that local governments are failing their mandate to collect and dispose of household waste in a specifically designated area (Kebede et al., 2020). For instance, in developing nations such as South Africa, more than 60% of the municipalities do not meet the minimum waste management standards, including minimisation, treatment and recycling before landfill disposal (Teshome et al., 2017). At the same time, developing countries are characterised by small towns that share boundaries with rural communities which leads to livestock wandering and scavenging waste dumps, and South Africa is not exceptional (Adane and Muleta, 2011). Solid waste mismanagement became worse as the cities grew rapidly to accommodate the growing population. As a result, the conditions became worse for these cramped communities (Tiruneh and Yesuwork, 2010). The migration of people to urban cities has resulted in increased occupancy of vacant lands through illegal home-dwelling units in the form of shacks (Ferronato and Torretta, 2019). Illegal home dwellings have resulted in an increase in unmonitored dumping sites

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00010-4

as the respective local governments do not recognise the areas (Akinbobola et al., 2016). Failure to comply with such standards explains the poor solid waste management, and the resultant environmental pollution (Dodia et al., 2014).

Because of the harmful effects caused by environmental pollution, communities should be given the opportunity to practice efficient solid waste disposal for their own benefit (Abdullah et al., 2014). Poor solid waste management has implications for human livelihood, the environment and livestock health, as the invaded areas are those previously used for livestock grazing in some developing countries (Fromsa and Mohammed, 2011; Mushonga et al., 2015; Negash et al., 2015). The impact of solid waste on livestock health, specifically on ruminants, has been an area of concern in the last ten years throughout the world. Livestock farmers in western Kenya and South Africa perceived that ingesting waste material, such as plastics was among other causes of illnesses in cattle (Nongcula et al., 2020; Nyamanga et al., 2006). In Ghana, Ethiopia and Rwanda, it was documented that livestock reared extensively tend to scavenge for food in areas such as waste dumps (Jebessa et al., 2018; Mushonga et al., 2015; Okai et al., 2007). The clinical impact of scavenging at waste dumps includes but is not limited to disruption of the digestive functions of the stomach, pain, rumenitis, polybezoars, reticuloperitonitis and abomasum impaction (Mushonga et al., 2015). Ingestion of foreign bodies in cattle is a pathological condition of economic importance, leading to severe economic losses due to high morbidity and mortality rates (Ramaswamy and Sharma, 2011). The present paper describes the impact of solid waste material on ruminant health and production in developing countries.

12.2 Vulnerability of ruminants to solid waste mismanagement

Environmental contamination with solid wastes from domestic and commercial sources is common in developing countries and is attributed to a poor refuse disposal system (Nongcula et al., 2017). Solid wastes may range from metallic to non-metallic types, and recently, plastic bags have been common in various peri-urban and urban areas (Teshome et al., 2017). Unfortunately, cattle commonly ingest the indigestible foreign bodies (IFB) due to their indiscriminate feeding habits (Mushonga et al., 2015). The IFB may lodge in different parts of the gastrointestinal tract, especially in the forestomachs (Bwala et al., 2016). Apart from causing pathological lesions, the presence of IFB reduces the volume of the forestomachs, which are otherwise supposed to be filled with feed (Bwatota et al., 2018). When the IFB occupies space in the forestomachs, it ultimately leads to reduced feed intake (Nongcula et al., 2020). The consequences of these are economic losses associated with severe loss of production and increased mortality rates (Akinbobola et al., 2016). Several authors have documented several factors that predispose cattle to ingestion of IFB, and those include: industrialisation, agriculture mechanisation, food scarcity, poor farming management and diseases that cause pica (Teshome et al., 2017; Kebede et al., 2020) (Fig. 12.1).

12.2.1 Traumatic reticulopericarditis

In most instances, nails, wires, or other sharp hard objects are wrapped in poly bags and dropped in waste pits (Adane and Muleta, 2011). While grazing on such waste, animals ingest these sharp objects and plastic bags (Teshome et al., 2017). Because of the honeycomb pattern of the reticulum, these sharp objects get trapped in the reticulum, causing damage (Tiruneh and Yesuwork, 2010). Over time, these sharp objects penetrate the wall of the reticulum and diaphragm and invade the heart, leading to traumatic reticulopericarditis (Jebessa et al., 2018).

12.2.2 Ruminal tympany

Polythenes present in the rumen and reticulum will wholly or partially occlude the orifice of the reticulum and omasum, leading to the accumulation of gases in the rumen (Negash et al., 2015). The situation worsens if such animals are fed with legumes or other gas-forming feed/concentrates (Kebede et al., 2020). Accumulation of gases in the rumen gives rise to bloat or tympany, which becomes fatal if the gases are not properly removed (Bwatota et al., 2018). Sometimes the poly bags present in the rumen may also occlude the oesophageal orifice leading to hindrance in eructation (Bwala et al., 2016). This gives rise to dyspnoea and death (Akinbobola et al., 2016).

12.2.3 Local or diffused peritonitis

Sometimes the animals accidentally ingest sharp objects, such as needles, nails and wires wrapped in plastic bags and covers (Fasil, 2016). Over a

FIGURE 12.1 Cattle grazing on a solid waste dumping site.

period of time, these sharp materials are released into the rumen, causing ruminal wall puncture and the development of local or diffused peritonitis (Kebede et al., 2020).

12.2.4 Ruminal impaction

Initially, due to ruminal contractions, polythene bags/plastics that accumulate in the rumen get entangled, leading to the formation of a hard mass (Teshome et al., 2017). Later on, this hard plastic mass obstructs the orifice between the reticulum and omasum, thereby causing hindrance to ruminal movements (Otsyina et al., 2015). Over time, this hard plastic mass decreases rumen motility and causes ruminal atony and ruminal impaction (Bwala et al., 2016). Ruminal impaction by a plastic foreign body is asymptomatic and is diagnosed only after the accumulation of huge quantities of plastic materials in the rumen (Jebessa et al., 2018).

12.3 Risk factors for ingestion of foreign objects

12.3.1 Species

Although ruminal impaction due to plastic materials occurs in all domestic and wild ruminants (Tiruneh and Yesuwork, 2010), cattle are the most susceptible livestock species among domestic ruminants. This is followed by sheep and goats (Bwatota et al., 2018). This difference in susceptibility can be attributed to the prehensile characteristics of the different animals. For example, prehensile organs such as the lips and tongues of cattle are not highly sensitive and, as a result, have an indiscriminate sense of taste. This makes cattle indiscriminate feeders (Jebessa et al., 2018). In addition, cattle tend to graze close to the ground, making them more vulnerable to the ingestion of foreign bodies such as plastics (Negash et al., 2015). On the other hand, goats are less susceptible to ingestion of foreign bodies because of their well-developed sensitive prehensile organs and their browsing and selective feeding behaviour (Tiruneh and Yesuwork, 2010). Fodder scarcity, environmental contamination and poor standards of animal rearing are, however, forcing goats to ingest and accumulate plastic foreign bodies that get stuck in their rumen (Ngoshe, 2012). Cases of ingesting indigestible foreign objects have been reported in ruminants due to the contamination of their habitat.

12.3.2 Breed

It has been observed that crossbreed cattle are more susceptible to ingestion of solid waste materials compared to local cattle breeds (Berrie et al., 2015). This could be attributed to crossbreed cattle having higher requirements for feed and fodder. Therefore, the need to compensate for the higher demand predisposes them to the ingestion of foreign bodies (Otsyina et al., 2015).

12.3.3 Sex

The sex of the animal has a strong influence on the likelihood of the animal ingesting foreign bodies (Fromsa and Mohammed, 2011). The risk of ingesting solid waste objects is higher in female animals as compared to male animals (Tassie and Endalew, 2020). Various physiological factors in female animals contribute to the ingestion of foreign bodies (Dodia et al., 2014). Some of these physiological factors include pregnancy and lactation stages, which place greater nutritional demand, create negative balance and/or lead to mineral deficiency (Ngoshe, 2012). These conditions, therefore, can increase the appetite of these animals, leading to development of pica and thus increased risk of consuming foreign bodies that they would not normally consume. In addition, farmers tend to keep female animals longer than male animals because of their long reproductive period (Adane and Muleta, 2011). Therefore, female animals are at higher risk of exposure to ingesting and accumulating foreign bodies in their rumen (Ibikunle et al., 2021). Abandoning of unproductive female animals and banning the slaughter of female cattle in countries such as India have compelled them to graze on roadsides and waste disposal areas, leading to a heightened risk of ingestion of foreign bodies (Desiye and Mersha, 2012).

12.3.4 Body condition status

The risk of ingesting foreign materials is higher in animals with a poor body condition score than in animals with a good body condition (Akinbobola et al., 2016). The poor nutritional status and negative energy balance of animals with poor body conditions cause capricious appetite in animals leading to the ingestion of indigestible food materials and the development of ruminal disease associated with ingesting plastic and hardware materials (Fasil, 2016).

12.3.5 Age

Old animals are more susceptible to the development of ruminal impaction caused by the ingestion of plastic materials than young animals (Nongcula et al., 2017). Cattle above 10 years of age and sheep and goats above 4 years have been observed to more frequently ingest indigestible materials than the other age groups (Tiruneh and Yesuwork, 2010). This could be attributed to increased exposure throughout life, and animals gradually ingest and accumulate foreign bodies over time (Mushonga et al., 2015).

12.3.6 Season

The risk of ingesting indigestible material is higher during the dry period, that is, between March and July in the case of Ethiopia (Kebede et al., 2020). Fodder scarcity is a major problem in dryland and rainfed agriculture (Teshome et al., 2017). In the dry season, animals will roam from one place to another in search of feed and fodder (Slayi et al., 2021). This makes them prone to ingestion of foreign bodies (Otsyina et al., 2015). During lean seasons of agriculture, livestock owners and their animals migrate to urban areas in search of jobs (Negash et al., 2015). In urban areas, they leave their livestock roam to find food for themselves (Bwala et al., 2016). Because of the non-availability of feed and indiscriminate disposal of polythene materials with some food remnants, livestock graze on these waste damps, leading to the ingestion of plastic bags and other solid waste materials (Berrie et al., 2015).

12.4 Impact of solid waste on ruminant health

Exposure of ruminants to indigestible materials results in high economic losses (Berrie et al., 2015). Most of these foreign bodies get lodged in the forestomachs, which is the site of most pathological conditions (Tiruneh and Yesuwork, 2010). Lodging of indigestible materials in the forestomachs may lead to various complications such as rumenitis, rumen impaction, traumatic pericarditis and traumatic reticuloperitonitis (Dodia et al., 2014). Internal injury resulting from sharp objects lodged in the forestomachs leads to perforation of the reticulum wall, which allows leakage of ingesta and rumen micro-flora, which contaminates the peritoneal cavity, resulting in inflammation of the peritoneum. If penetration of the pleural cavity occurs, it causes pleuritis and pneumonia. But if the foreign object penetrates the pericardial sac, it causes pericarditis (Dodia et al., 2014). Lodging of indigestible material in the forestomachs may ultimately lead to rumen distension, absence of defecation, reduced feed intake, failure to absorb volatile fatty acids, reduced weight gain and consequently death (Remi-Adewunmi et al., 2004; Mushonga et al., 2015).

12.5 Indigestible materials resulting from poor environmental management

The indigestible materials that ruminants are exposed to vary in type and are discussed in the following paragraphs. Fig. 12.2 below shows a proportion of solid waste material found in animals and relevant photos of foreign objects found in the rumen after slaughter. Foreign bodies found in animals were grouped into polybezoars, penetrating and non-penetrating objects. These are discussed in various sections below.

FIGURE 12.2 Proportion of foreign objects found in slaughtered animals and types of solid material (Nongcula et al., 2020).

12.5.1 Polybezoars

Hard stone-like masses formed by the deposition of salts around polythenes in the digestive tract are known as polybezoars (Mushonga et al., 2015). These polybezoars cause obstruction of food passage and lead to rumen pain and inflammation (Desiye and Mersha, 2012).

12.5.2 Hardware or penetrating foreign bodies

The ingestion of penetrating foreign bodies, such as nails, wires and needles remains a problem in cattle farming worldwide (Dodia et al., 2014). Penetrating foreign bodies are observed mainly in semi-intensive and extensive livestock production systems (Teshome et al., 2017). Animals reared under these production systems get exposed to waste disposal materials as they graze freely around the roadside (Adane and Muleta, 2011). As a result, penetrating objects such as wires account for approximately 70% of ingested foreign bodies, and nails and steel objects make up 30% (Desiye and Mersha, 2012). Another reason wires and nails are the most common inciting agents could be because animals are exposed to construction sites or deteriorating buildings and fences (Fromsa and Mohammed, 2011).

12.5.3 Non-penetrating foreign bodies

Non-penetrating foreign bodies such as leather, hair, plastic, rope and sack thread have been documented by various studies around the world (Nugusu et al., 2013; Mushonga et al., 2015). The reason for the increased prevalence of ingestion of non-penetrating foreign objects could be attributed to the fact these materials are used for storing wastes (Bwala et al., 2016), as shopping bags (Jebessa et al., 2018), and for packing food items (Kebede et al., 2020) and tend to be disposed of everywhere after they have been used. Hence they get eaten by the free-grazing animals, especially in towns and villages (Berrie et al., 2015).

12.6 Previous research conducted on the impact of solid waste on ruminant health

Ingestion of foreign objects in ruminants has been reported in many countries (Kebede et al., 2020). Other than hardware, plastic bags are the most common type of foreign body found in ruminants suffering from ill health (Bwatota et al., 2018). Different workers have reported the slaughterhouse prevalence of plastics and wires among other foreign bodies (Akinbobola et al., 2016; Bwala et al., 2016). In Tanzania, slaughterhouse prevalence was around 74% (Bwatota et al., 2018), and in Rwanda, it was about 62.5% (Mushonga et al., 2015). Although studies indicate that the prevalence of indigestible objects in ruminants is lacking in a large proportion of countries such as South Africa, Egypt, Zimbabwe and others, many research scientists have reported case studies regarding the occurrence of ruminal disorders due to indigestible materials in ruminants from these countries (Table 12.1).

TABLE 12.1 Summary of research findings on the impact of ingesting foreign objects on ruminant health in developing countries.

Organ affected	Main findings	Ruminant species	Country	Literature source
Rumen and reticulum	The prevalence of foreign bodies was significantly ($\chi^2 = 17.53$, $P < .05$) higher in sheep (59.3%) and goats (56.7%) than in cattle (43.4%). The prevalence of foreign bodies in study animals with poor body conditions was significantly higher ($\chi^2 = 38.57$, $P < .05$) than in medium and good body conditions. A higher percentage of foreign bodies	Cattle, sheep and goats	Ethiopia	Negash et al. (2015), Teshome et al. (2017), Jebessa et al. (2018), Kebede et al. (2020), Tiruneh and Yesuwork. (2010)

(*Continued*)

TABLE 12.1 (Continued)

Organ affected	Main findings	Ruminant species	Country	Literature source
	occurred in the rumen alone (87.9%) than in the reticulum alone (5.0%), with the rest present in both. Significantly higher proportions of foreign bodies were observed in the rumen of cattle ($\chi^2 = 332$, $P < .05$), sheep ($\chi^2 = 193$, $P < 0.05$) and goats ($\chi^2 = 285.0$, $P = .000$) than in the reticulum. Plastic was the most commonly encountered (79.2%) foreign body, followed by cloth (15.3%) and rope (12.3%). In addition, metal (0.9%), calcified material and/or stone (1.0%) were found in the reticulum of cattle.			
Rumen	The results show an overall occurrence of 17.4% foreign bodies in cattle. The highest occurrence (25.3%) was recorded in June (the driest month). Results further show that the majority of the foreign bodies were plastics (65.0%). More foreign bodies (29.5%) were found in older animals (5 years and above) than in younger and middle-aged animals (16.5% and 6.0%, respectively). There was a higher prevalence of foreign bodies in female cattle (20.0%) than in males (15.7%). As observed in the study, the presence of cassette tapes has not been reported elsewhere. The high representation of plastics in animals (65.5%) in the light of a government plastic bag ban in supermarkets presents a significant challenge to livestock production in Rwanda.	Cattle	Rwanda	Mushonga et al. (2015)

(*Continued*)

TABLE 12.1 (Continued)

Organ affected	Main findings	Ruminant species	Country	Literature source
Rumen and reticulum	The study revealed that metallic and non-metallic indigestible objects had an overall prevalence of 63% in cattle slaughtered in Queenstown abattoir (QTA, ($n = 1906$)) and 64.8% at the East London abattoir (ELA, ($n = 2518$)). Most IFOs were found in the rumen (64.2% and 70.8%) and reticulum (28.5% and 20.6%) at QTA and ELA, respectively. The leading IFOs in the stomach of cattle at QTA were plastics (27.7%), polybezoars (10.7%) and ropes (10.7%), while polybezoars (19.8%), ropes (17.6%) and stones (10.7%) were the main IFOs seen in cattle at ELA. The study showed a statistical significance ($P < .05$) between body condition score and the prevalence of indigestible objects in cattle.	Cattle	South Africa	Nongcula et al. (2017), Nongcula et al. (2020)
Rumen and reticulum	Out of 387 examined cattle, 93 (24.03%) had IFB in their forestomachs. The observed IFB were plastic bags, fruit seeds, clothing materials, ropes, hairballs, leather materials, stones, metallic nails and wire. Plastic bags were the most frequently (50.5%) observed IFB, followed by fruit seeds (18.3%). A significantly ($P < .05$) high proportion of old animals (31.7%) had IFB compared to the young animals (21.2%). Similarly, the frequency of occurrence of IFB was significantly high ($P < .05$) in crossbred dairy cattle (42.3%) compared to local breeds (22.7%). Cattle that appeared with poor body condition (37.8%) were found to be more affected ($P < .05$) by IFB than those with good body condition (15.9%). In 91.4% of animals with IFB, all the materials were located in the rumen.	Cattle	Tanzania	Bwatota et al. (2018)

(*Continued*)

TABLE 12.1 (Continued)

Organ affected	Main findings	Ruminant species	Country	Literature source
Rumen	Out of the total number, 15 (16.5%) were found to have foreign bodies, of which 13 (86.7%) were females while 2 (13.3%) were males. The Foreign bodies identified were an admixture of polythene, plastic materials (shopping bags, mats and ropes), and pieces of wood. Breed distribution of the foreign body impaction showed that the highest percentage was recorded in crossbreeds (20%) while the lowest was in Uda breed (8.3%). The indigestible foreign bodies found in the rumen of slaughtered cattle were predominantly polythene bags, wires and stones, with a prevalence rate of 9%, 1% and 2%, respectively. The overall prevalence of foreign bodies in the rumen was 12%. Among the different age groups of cattle examined, those within 18–24 months had the highest prevalence (28.57%), while those under 12 months had a prevalence rate of 0%. It was also observed that the prevalence was higher in females (18.42%) than in males (8.06%). The study also indicated that animals with poor, medium and excellent body conditions had 27.78%, 10.45% and 0% prevalence rates, respectively.	Cattle and sheep	Nigeria	Akinbobola et al. (2016), Bwala et al. (2016)

12.7 Conclusion

The scientific literature on the impact of solid waste on ruminant health is predominantly in the form of case reports or slaughterhouse incidents. Moreover, studies done at slaughter houses show that generally more than half of the animals slaughtered carry ingestible waste material with potential to negatively affect the health and performance of the animals. A lack of detailed studies under farm conditions hinders early diagnosis of the problem. In addition, it was noted that there is unequal research attention given to the impact of solid waste on ruminant health in developing countries. In developing countries, more basic research is warranted to explore the pathophysiology, diagnosis and treatment of solid waste in different animal species.

References

Abdullah, N., Sakawi, Z., Ismail, L., 2014. The perception of the Langkawi community on solid waste management. Curr. World Environ. 9, 237–243.

Adane, L., Muleta, D., 2011. Survey on the usage of plastic bags, their disposal and adverse impacts on environment: a case study in Jimma City, Southwestern Ethiopia. J. Toxicol. Envtal. Health Sci 3, 234–248.

Akinbobola, J.S., Omeje, J.N., Akinbobola, R.I., Ayoade, R.I., Ajagbe, O.A., Okaiyeto, S.O., 2016. Prevalence of indigestible foreign bodies in the rumen of cattle slaughtered at Gwagwalada Abattoir, Abuja, Nigeria. Int. J. Livest. Res. 6 (6), 25–31.

Berrie, K., Tadesse, E., Mossie, B., 2015. Study on rumen and reticulum foreign body in slaughtered cattle at Gondar Elfora Abattoir. World J. Biol. Med. Sci. 2, 133–150.

Bwala, D.A., Peter, I.D., Eze, C.A., Bukar-Kolo, Y.M., Bukar, M.M., 2016. A study on rumen foreign body impaction in sheep slaughtered at the Maiduguri Metropolitan Abattoir, Maiduguri, Nigeria. Int. J. Livest. Res. 6 (3), 16–23.

Bwatota, S.F., Makungu, M., Nonga, H.E., 2018. Occurrences of indigestible foreign bodies in cattle slaughtered at morogoro municipal slaughterhouse. Tanzania. Hindawi J. Vet. Med. 3 (4), 1–6.

Desiye, T., Mersha, C., 2012. Study on rumen and reticulum foreign bodies in cattle slaughtered at Jimma Municipal Abattoir, South West Ethiopia. Am. Eur. J. Sci. Res 4, 160–167.

Dodia, V.D., Kelawala, N.H., Suthar, D.N., Prajwalita, S., 2014. Haematological and serum biochemical profile of cattle affected with plastic foreign bodies. Int. J. Sci. Res. Publica 4, 1–2.

Fasil, N., 2016. Assessment of sheep and goat foreign bodies in rumen and reticulum in the Jigjiga municipal Abattiar. J. Adv. Dairy. Res 4, 1–6.

Ferronato, N., Torretta, V., 2019. Waste mismanagement in developing countries: a review of global issues. Int. J. Environ. Res. Public Health 16 (6), 1060.

Fromsa, A., Mohammed, N., 2011. Prevalence of indigestible foreign body ingestion in small ruminants slaughtered at Luna Export Abattoir, East Shoa, Ethiopia. J. Anim. Vet. Adv 10, 1598–1602.

Ibikunle, R., Titiladunayo, I., Dahunsi, S., Akeju, E., Osueke, C., 2021. Characterization and projection of dry season municipal solid waste for energy production in Ilorin metropolis. Nigeria. Waste Manag. Res. 39 (8), 1048–1057.

Jebessa, D., Lemma, F., Kabeta, T., Sibhat, B., Tarefe, Y., 2018. Survey on indigestible foreign bodies in the rumen and reticulum of cattle slaughtered at Nekemte municipal abattoir. Nekemte, Ethiopia. Ethiop. Vet. J 22 (1), 11–25.

Kebede, S., Bekele, T., Fesseha, H., 2020. Prevalence of indigestible rumen and reticulum foreign bodies in cattle slaughtered at Kombolcha Elfora Abattoir, Kombolcha Town, Amhara Regional State, Ethiopia. Int. J. Rec. Biotech. 8 (1), 25–34.

Mushonga, B., Habarugira, G., Musabyemungu, A., Udahemuka, J.C., Jaja, F.I., Pepe, D., 2015. Investigations of foreign bodies in the fore-stomach of cattle at Ngoma Slaughterhouse, Rwanda. J. South. Afr. Vet. Assoc. 86 (1), 1–6.

Negash, S., Sibhat, B., Sheferaw, D., 2014. A postmortem study on indigestible foreign bodies in the rumen and reticulum of ruminants, eastern Ethiopia. Onderstepoort. J. Vet. Res. 1, 1–5.

Ngoshe, A.A., 2012. Incidence of polythene bag ingestion by ruminant animals at Maiduguri Central Abattoir. Ramat J. Manag. Sci. Technol. 1, 12–16.

Nongcula, V.V., Jaja, I.F., Nhundu, K., Zhou, L., 2020. Prevalence, perception and implication of solid waste in cattle slaughtered in Eastern Cape Province, South Africa. Adv. Anim. Vet. Sci. 8 (7), 692–700.

Nongcula, V.V., Zhou, L., Nhundu, K., Jaja, I.F., 2017. Association between the prevalence of indigestible foreign objects in the gastrointestinal tract of slaughtered cattle and body condition score. Animals 7 (80), 1–10.

Nugusu, S., Velappagounder, R., Unakal, C., Nagappan, R., 2013. Study on foreign body ingestion and related complication in ruminants associated with inappropriate solid waste disposal in Gondar town, North West Ethiopia. Int. J. Anim. Vet. Adv. 5 (2), 67–74.

Nyamanga, P.A., Suda, C.A., Aagaard-Hansen, J., 2006. Ethnoveterinary perceptions of cattle illnesses in Western Kenya. J. Inst. African Stud. 7, 1–7.

Okai D.B., Boateng M., Opoku-Agyemang H. (2007). The incidence of plastics in the rumens of cattle. In: Proceedings of the Fifteenth (15th) GSAP Conference, 2007. pp. 119–123.

Otsyina, H.R., Nguhiu-Mwangi, J., Mogoa, E.G.M., Mbuthia, P.G., Ogara, W.O., 2015. Prevalence of indigestible rumen foreign bodies in sheep and goats at Dagoretti and Kiserian Abattoirs, Kenya. Int. J. Vet. Sci 4, 75–80.

Ramaswamy, V., Sharma, H.R., 2011. Research article plastic bags—Threat to environment and cattle health : a retrospective study from Gondar city of Ethiopia. Inst. Intergrative Omi. Appl. Biotechnol. 2, 7–12.

Remi-Adewunmi, B.D., Gyang, G., Osinowo, O.A., 2004. Abattoir survey of foreign body, rumen impaction in small ruminants. Nig. Vet. J 25, 32–38.

Slayi, M., Muchenje, V., Njisane, Y.Z., 2021. Behavioral and haemato-biochemical responses of Nguni and Boran steers post relocation and herd regrouping in a novel environment. J. Appl. Anim. Welf. Sci 24 (1), 39–55.

Tassie, K., Endalew, B., 2020. Willingness to pay for improved solid waste management services and associated factors among urban households: one and one half bounded contingent valuation study in Bahir Dar city, Ethiopia. Cogent Environ. Sci 6 (1), 1807275.

Teshome, E., Abdela, N., Hassan, A., 2017. Postmortem study on indigestible foreign bodies in rumen and reticulum of ruminants slaughtered at Asella Municipal Abattoir, Southeastern Ethiopia. J. Vet. Sci. Technol 8 (3), 1–6.

Tiruneh, R., Yesuwork, H., 2010. Occurrence of rumen foreign bodies in sheep and goats slaughtered at the Addis Ababa Municipality Abattoir. Ethiop. Vet. J 14, 91–100.

Chapter 13

Biowaste valorisation in a circular economy

Wentao Chen[1,2], Yong Li[1], Qianlong Tan[1,2], Lingli Xie[1,2], Yuanyuan Hou[1,2], Wenxi Jiang[1,2], Dan Peipei[1,2], Haimei Wu[1,2], Hong Liu[1,2], Zekai Chen[1,2], Can Mao[1,2] and Taimoor Hassan Farooq[1]

[1]College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China, [2]National Engineering Laboratory for Applied Technology of Forestry and Ecology in South China, Central South University of Forestry and Technology, Changsha, Hunan, China

13.1 Introduction

As the world population booms and the anthropic activities extend in the past century, the massive extent increase in resource demand and consumption causes one serious problem to ecosystems and biodiversity – the released biowastes, which present global issues like the way and cost of disposal and storage, pollution of land, water, air and hazard to animal's and human's health, etc. The role of valorisation of biowaste comes to be expected to alleviate these problems for the transition from fossil-based economy to a circular bioeconomy. This chapter focuses on biowaste valorisation in a circular economy (Fig. 13.1) and proposes several suggestions and disposal methods to convert biowaste residues.

13.1.1 What is biowaste?

First, let us talk about biowaste, which is not yet unambiguously defined. Still, some similar concepts are posed and clearly defined, such as organic waste, biodegradable waste or residue, and biomass waste. To waste, distinguished from other residues by the United Nations Statistics Division, is as follows: 'waste includes materials which are not prime products and the generator has no further use for these resources and discards them, or intends or is required to discard them' (United Nations Statistics Division, 2022).

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00011-6

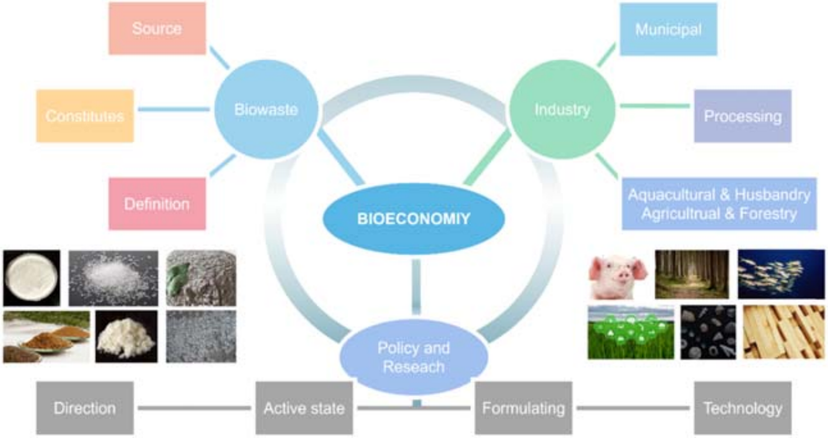


FIGURE 13.1 Aspects covered in the concept of biowaste valorisation in a circular economy.

Furthermore, according to the European Commission, biowaste is defined as ‘biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises, and comparable waste from food processing plants’ (European Commission, 2022), while in a broader sense, it also encompasses sewage and domestic wastes like food/manure wastes, or fisheries/forestry/agriculture residues (Aliyu and Bala, 2011). Aquacultural waste, biomass from forestry/agriculture and animal husbandry, municipal solid waste residues generation vastly, persistence ubiquitously, and high chemical richness, becoming promising business models of biowaste valorisation for economic profits and environmental impacts (Xu et al., 2019).

13.2 Industrial residues and constitution of biowaste

A circular bioeconomy involves a lot of industrial sectors, thereinto many primary sectors like ‘agriculture, food, beverage and tobacco’, including animal byproduct industries and meat processing. Currently, these sectors lead the bioeconomy to take up employment (80%) and turnover (~75%) in the European Union and also expand to these sectors: sustainable biological resources production, residues and side-streams usage for valorised products in feed, food, chemicals, cosmetics, pharmaceuticals, bio-based materials, and energy. There are some examples as shown below:

13.2.1 Agroforestry and its driven processing industry waste

Valorisation of the forestry-based value chains should highlight the residues and side-streams from the manufacturing of wooden furniture or construction, production of pulp, paper, lignocellulosic chemicals or materials, and solid bioenergy or liquid biofuels. Thereinto, we need to divide the residues into three main groups (Rodsrud et al., 2012):

1. Primary residues: inside and outside forests generate by logging/cultivation/harvesting activities from trees;
2. Secondary residues: bark/sawdust/black liquor, etc., generated by wood processing;
3. Tertiary residues: castoff/demolition wood/discarded furniture/household waste/other end-of-life wood from trade and industrial, considered organic waste.

Based on these residues, value creation is empowered with high potential. Besides its transferable processing property, soil nutrients will not be depleted in the forest if the leaves/bark are left behind. At first, the bark will be removed from the plant and become a residue for other operational use as fuel.

When we mention fuel, there is another way to make valorisation.

Fly ash from coal or oil extraction residues mixed with co-firing biomass like forestry and agricultural industries residues or cereal production generates co-combustion fly ash, which reduces pollutant emissions of coal, saving space and energy, and use in various fields due to raw materials and production cost potentially reduction (Chen et al., 2018; Salah et al., 2019; Visa et al., 2015). In this area, there are some of the most important examples encompass carbon nanotubes (CNTs) (Salah et al., 2017), carbon nanofibers (Muthu et al., 2018) and carbon nanomaterials (CNMs) (Hintsho et al., 2016), but due to the expensive precursors and catalysts usage, the production process is costly.

The brewing industry is also considered somewhat bio-circular since spent grain, organic side-stream, as animal feed returns to the ecosystem. Among these, high protein content and other nutritional assets properties for spent grain as raw materials used in various industries, encompassing biofuels/food/livestock feed/nutrition/pharmaceuticals/chemicals (Mussatto, 2014), but due to the research results deployment in low levels, even with technical potential, only a few progressive uses example on an industrial scale (Aliyu and Bala, 2011; Mussatto, 2014).

13.2.2 Aquaculture and meat processing waste

Aquaculture and meat processing waste have several potentially high-value molecules sources encompassing amino acids, proteins, bioactive peptides, collagen, gelatin, chitin, pigments, oils, and other biopolymers (Aspevik et al., 2017), which usually contain 20%–30% chitin, 30%–40% protein, 0%–14% lipids and inorganic salts mainly calcium carbonate and 30%–50% phosphate. Via deproteinisation, biological fermentation and demineralisation (remove inorganic salts) achieve discolouration by solvent extraction at room temperature (Jayakumar et al., 2011).

In the fishing industry, crustacean wastes (crab, shrimp, lobster, krill and prawn shells are major sources) (Devi and Dhamodharan, 2018; Verlee et al., 2017; Yang et al., 2019) are always used for the extraction of commercial chitin, which is natural amino polysaccharides found together in nature, analogous to collagen inside higher animals and cellulose inside plants, existing inside the exoskeleton of crustacean and arthropod, and cell walls of fungi and yeast; as an ordered crystalline microfibrils form structural components (Pillai et al., 2009).

13.2.3 Animal husbandry-related industries waste

Acid whey, a strong pollutant residue from dairies and a valuable substance in whey, is becoming a worldwide waste disposal issue but possible to valorise, including lipids/vitamins/minerals/lactose/functional proteins and peptides (Smithers, 2015). In sweet whey, in recent decades, dairies worldwide have developed new business models/products/technologies and processing capacities for using these feedstocks for animal feed, biogas, bakery, whey beverages, and nutritional products. Similarly, acid whey will become more useful if appropriately dealt with the taste and processing issues (Chandrapala et al., 2016).

Pig bristles, consisting of 90% valuable protein but hard-to-degrade moiety therein, like keratin, caused these bio-residues to reduce to a low value. Fortunately, a Danish project called *keratin2protein* proposed to feed fish by degrading them, which offers a new sight and exploration to deal with the problem (Gonzalez-Ballesteros et al., 2018; Lipkowski et al., 2009).

Cattle manure and derived waste are others of the less valuable animal debris that will cause detrimental environmental effects like contaminating soil and water resources and increasing greenhouse gas emissions if management is inappropriate and of excessive application. But converting into biomethane or bio-oil to generate power by anaerobic digestion is a good choice for neutral climate (Monteiro et al., 2011; Nasir et al., 2012; Pituello et al., 2015).

13.2.4 Municipal solid waste

Household waste, or call it residential waste or domestic waste, mainly include paper and organic materials, and other heterogeneous composition like plastic/glass/metals/electronic waste, calls for huge environmental/economic/social problems for the local authority, which needs to handle the physical infrastructures effectively and organisational capabilities' lacking, and challenges existing institutions/policies/technologies/business models (Paledal et al., 2018).

13.3 Constitutions

Cellulose/hemicellulose/lignin/carbohydrates/protein/lipids are common components of different styles of biowaste. Among these, the largest compound ingredient is cellulose, lignin made of aromatic polymers, while sugar compounds like cellulose and hemicellulose can make up macromolecules (Saidur et al., 2011). Moreover, hemicellulose combined with lignin is called 'lignocellulose' (Chen et al., 2015). Lignocellulose, as a kind generator of forestry and agricultural industries biowaste, earth's most abundant sustainable biomass, is involved in terrestrial ecosystems carbon cycle, experienced decomposition and transformations of microbial, generate water, and CO_2 at last, due to the content of proteins/polysaccharides/phenolics, can be used as prospect sources for food/energy/materials/chemicals.

Thereinto, lignin, cellulose and hemicellulose can be, respectively, transformed into high-value substrate furfural to produce chemicals and carbon (battery anodes/carbon fibres/foam) or soluble pulp to produce the chemicals and textile fibres (Alonso et al., 2017).

13.4 Disposal and reuse of biowaste

Based on the source mentioned above, we expand the discussion to biowaste transformation. Biowaste can be physically, chemically or biologically transformed into useful products and materials. There is a massive economic potential for converting low-value biowaste into valorised materials and products; thereinto, the commercialisation of technology and industry are essential, which rely on enterprises integrating biowaste processing within biorefinery schemes. Several techniques now valorise biomass waste into value-added products/high-quality biofuel (biochar); disposal methods commonly include sending to a landfill, thermal degradation (pyrolysis /torrefaction /gasification /liquefaction), microbial decomposition (aerobic/anaerobic) as described below: (Cheng et al., 2020).

13.4.1 Microbial decomposition of biowaste

Anaerobic digestion of biological treatment based on several substrates (organic waste from municipal, manure, and food processing industry) output biogas and bio-digest. The former can replace fossil fuels to use in transport, which produce easily and are especially profitable if transportation costs are covered; the latter can replace artificial fertilizer to get nutrients back into the soil (Klitkou and Bolwig, 2019). Microorganisms' energies by biochemical processes (enzymatic hydrolysis/anaerobic digestion) work on biowaste; thereinto, cellulose content will ferment into sugars, through the biochemical process biosynthesise fine chemicals, which will increase the carbohydrate potential of biowaste materials (Cheng et al., 2020).

13.4.2 Biorefineries of biowaste

Biorefineries' treatment can be classified by applied technology as thermochemical (based on gasification and/or pyrolysis) or biochemical, which is also based on different types of feedstock like residues from food/fish processing/animal husbandry/aquaculture/forestry and forest-based industries, wastewater treatment sludge, straw and stover from plant production, the main intermediate products are syngas, sugar and lignin.

Although we use some biowaste as fuel to directly combust, it emits and pollutes the air (Chen et al., 2019), and the lignocellulose-rich property means low calorific value, large volume and high moisture content (Lu et al., 2013); therefore, improved value performance of biowaste becomes such a significant conversion process. So biorefineries must be optimised to efficiently recover valuable compounds like proteins or phosphorus, use bioresources or energy in the materials and chemicals production, which are able to replace fossil resources with renewable, organic resources, and moreover, produce new material types, the qualities of which are different to fossil-based materials (Klitkou and Bolwig, 2019).

13.4.3 Biowaste to biomaterials

Waste valorisation aims to increase residues and side-streams value by changing the physical properties of these materials; several examples are shown here:

13.4.3.1 Collagen and collagen-based biopolymers

Collagen found in bodies of animals and humans are the most abundant fibrous protein and shows superior properties of bioactive, biocompatibility, biodegradability, low antigenicity and non-immunogenicity, which means broad feasibility applications in sectors of medicine, food, tissue engineering and cosmetics used as a biomaterial (Lee et al., 2001; Parenteau-Bareil et al., 2010). When exploring methods for improving substance yields, especially the use of an organic fraction of fish biowaste discards, as an eco-friendly and low-cost source of collagen can be extracted through ionic liquids, ultrasonication or extrusion-hydro-extraction process (Klitkou and Bolwig, 2019).

13.4.3.2 Chitin and chitosan-derived biomaterials

Crystalline chitin display low immunogenicity, biocompatibility, antimicrobial activity, eco-safety and biodegradability, which highly attract the biomaterials field (Jayakumar et al., 2011). One of the applications is to synthesise multifunctional hybrid bio-aerogels decorated with chitin nanocrystals (CNCs) based on cellulose nanofibers (CNFs) (Gopi et al., 2017); another remarkable application is the nanofibers fabrication with diameters less than 100 nm have been reported (Ding et al., 2014). What is more, chitosan will generate by the deacetylation of chitin with the acetamide groups alkaline hydrolysis.

13.4.3.3 Bioplastics

Various materials like biodegradable, bio-based (biomass synthesised), or both biodegradable and bio-based plastics are all counted, made by cellulose mostly, manufactured by diverse sources like wood pulp, crops (cotton, corn, sugar cane), fungi, and other bio-based raw materials produced with microbes' or algae's work, production encompassing viscose rayon (fibre), celluloid and cellophane (film) (Altman, 2021).

European Bioplastics shows that in 2022, the production capacity of bioplastics is increasing to almost 2.44 million tonnes; thereinto, the main drivers of this growth are biodegradable or fully bio-based biopolymers like PHAs (polyhydroxyalkanoates) and PLA (polylactic acid) (Xu et al., 2019). However, sourced from biomass, nonbiodegradable polymers will follow different fates, which constitute presently almost 56% of bioplastics production (1.2 million tonnes) worldwide, like bio-PET (polyethylene terephthalate) and bio-PE (polyethylene), in recent years, bio-based PE manufacturing will continue growing, but bio-based PET will not substitute by 100% bio-based like bio-PEF (polyethylene furanoate), which thermal and barrier properties had improved, used for food, drinks and non-food products' packaging (Xu et al., 2019).

13.4.3.4 Silica and silicates

Rice husks are one of the most silica-rich sources, largely available among biowastes, typically account for 20%–22% of rice grains; within it, biosilica and its derivatives applications are becoming attractive (Wang et al., 2012). Biogenic silica extract approaches from rice husks mostly rely on acidic pretreatments to remove trace amounts of metals; next pyrolytic procedures operate respectively at variable times and temperatures, usually within the scope of 8–24 hours and 500°C–700°C (Patil et al., 2017).

13.4.3.5 Additional miscellaneous examples

In addition to the above, the conversion of the other materials is also shown in the following (Table 13.1).

13.5 The conception of circular bioeconomy

Based on these technologies mentioned above, how to make the directions of society formulating systematically changing to a sustainable active state comes to an executive problem, which appeals to us to clarify the conception of circular bioeconomy.

TABLE 13.1 Different biomaterials derived from biowastes (Xu et al., 2019).

Biowaste	Prepared biomaterials
Bovine bones	Mineralised collagen
Silver carp skin	Collagen
Mantis shrimp muscle	Collagen
Pufferfish skin	Collagen
Cuttlefish skin	Collagen
Brown-backed toadfish skin	Collagen
Fish (*Lates calcarifer*) scales	Collagen sheet
Loligo uyii skin	Type V collagen
Eel fish skin	Type-I collagen
Milkfish scales	Collagen
Prawn shells	Chitosan
R. oryzae fungi on potato peels	Chitosan
Goatskin	Collagen-chitosan
Fish (*Latbeo rohita*) scales	Chitin and chitosan
Shellfish	Chitosanases
Crustacean	Chitin
Blue crab	Chitosan
Crab shells	Chitin and chitosan
Shrimp shells	Chitosan and Chitooligosaccharides
Beetle (*Catharisus molossus*)	Chitosan
Fish waste	Gelatin
Fish fin and chicken feather waste	Protein
Fishbone	Calcium phosphates
Groundnut and coconut shell, rice husk, palm fruit bunch and palm fruit stalk	Cellulosic fibres
Pig bones and teeth	Hydroxyapatite
Eggshells	Nanostructured hydroxyapatite
Eggshells	Mesoporous hydroxyapatite NPs

(*Continued*)

TABLE 13.1 (Continued)

Biowaste	Prepared biomaterials
Fish scales	Hydroxyapatite scaffolds
Shrimp shells	Bioplastic
Chicken feathers	Bioplastic
Fish scales	Bioplastics
Rice husk	Polyester bioplastic
Food industry	Bioplastic
Fruit peel	Bioplastic
Wood mill effluents	Bioplastic
Rice husk	Porous SiO_2
Rice husk	Nanosilica
Rice and coffee husks	Cellulose nanocrystals
Cotton linters and kraft pulp	Cellulose nanocrystals
Croaker fish skin	Gelatin

Human activities indeed impact the environment. It should be sustainable, the same as the economy founded by human activities and represent the highest relation of the human. No part of the activity should restrict other people's possibilities for their needs. But that is the thing when we mention general and abstract levels, sustainability is easy to agree upon, but when it means in practice, it is harder to judge (Klitkou and Bolwig, 2019). The bioeconomy conception aims to address the challenge of annually generated millions of tonnes of biowaste worldwide. Intending to ensure food and energy security, improve public health, mitigate climate change, and promote industrial restructuring, strategies for replacing fossil-based with organic-waste-based resources products and the economy have become important. The assessment of the bioeconomy is embodied in designing innovative products, setting new business models, and changing consumer behaviour, to achieve a rational circle. To transform into a circular economy and estimate environmental sustainability, LCA should call for broader use as a method or tool (Finnveden et al., 2009).

The International Organization for Standardization (ISO) gives LCA a definition as 'compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle'. The breadth and depth of the LCA definition have developed in recent years; apart from directly impacting the natural environment, sustainability dimensions expanded and modelled on more specific impacts.

Moreover, the difficulty of transformation to a circular economy keeps exists, here poses several challenges:

13.5.1 Policy

Although many governments worldwide embraced this bioeconomy concept for responding to diverse societal challenges, the notion is omnifarious. It encompasses several meanings and sectors, including dissimilar 'rationales or visions of the underlying values, directions and drivers of the bioeconomy' (Staffas et al., 2013). Involved industries and policy area's diversity may reflect opposing rationales, so across sectors, a mixed horizontal policy is needed; moreover, sociotechnical transitions should not just be theorising; an active state of governance, direction establishing and societal formulating are all needed (Bugge et al., 2016). The urgent need for policies about sustainable innovation requires a design to fix market failures and improve coordination and clear strategies to reduce the risks and uncertainties (Schneir, 2015).

13.5.2 Research

Advanced economies should get the ability to use new scientific and technological knowledge to generate and disseminate; although knowledge-based economies are characterised as modern economies, the innovation of sustainability should not yet stagnate.

The systemic innovation approach should focus on three types of knowledge and concepts:

(1) Research: conduct developed by research institutes, universities and enterprises; (2) Empowerment: develop human capital through omnifarious education and training; (3) Innovation: knowledge-based asset manipulated by enterprises. Only by conquering these constantly generating new challenges we accomplish the goal of biowaste valorisation in a circular economy.

13.6 Conclusions and prospects

Biowaste streams from households and industry as an expense to enterprises and a burden for society are a valuable misplaced resource asset to today's economies, like many applications, which transfer biomass into high-value products like bioenergy/biomaterials/biochemicals/biopharmaceuticals (Cheng et al., 2020). But the main problem is precisely focused on how to position the bioeconomy transforming. Looking forward, the ability to address society's daunting grand challenges is not just about developing innovation in research institutes, universities and enterprises; moreover, when using the current economy's different parts investments, it is increasingly dependent on systems innovation efficiency improvement.

There are perspectives focused on roughly three different aspects of bioeconomy:

(1) The OECD and the United States: using biotechnology and life sciences to convert raw material into high-value products; (2) The European Union: use of waste and biological resources as inputs for industrial products, feed, food and energy; (3) Environmental scientists/NGO: planetary boundaries and sustainability.

No matter through which perspective, around the world, the discussion of political/academia/business circles is propelling, while only when certain challenges (growing demand for food/energy/materials, sustainable use of resources, decoupling economic growth from environmental degradation) are addressed, transition to a sustainable economy will emerge.

Declarations

Ethics approval and consent to participate: Not applicable
Consent for publication: Not applicable
Conflicts of Interest: The authors declare that they have no known competing financial or personal interests.
Funding: No funding was received for this work.
Author Contribution: This collaboration work was carried out among all the authors.

References

Aliyu, S., Bala, M., 2011. Brewer's spent grain: a review of its potentials and applications. Afr. J. Biotechnol. 10 (3), 324–331. < Go to ISI>://WOS:000286397500003.

Alonso, D.M., Hakim, S.H., Zhou, S.F., Won, W.Y., Hosseinaei, O., Tao, J.M., et al., 2017. Increasing the revenue from lignocellulosic biomass: maximizing feedstock utilization. Sci. Adv. 3 (5), e1603301. Available from: https://doi.org/10.1126/sciadv.1603301.

Altman, R., 2021. The myth of historical bio-based plastics early bio-based plastics, which were neither clean nor green, offer lessons for today. Science 373 (6550), 47–49. Available from: https://doi.org/10.1126/science.abj1003.

Aspevik, T., Oterhals, A., Ronning, S.B., Altintzoglou, T., Wubshet, S.G., Gildberg, A., et al., 2017. Valorization of proteins from co- and byproducts from the fish and meat industry. Top. Curr. Chem. 375 (3), 53. Available from: https://doi.org/10.1007/s41061-017-0143-6.

Bugge, M.M., Hansen, T., Klitkou, A., 2016. What is the bioeconomy? A review of the literature. Sustainability 8 (7), 691. Available from: https://doi.org/10.3390/su8070691.

Chandrapala, J., Chen, G.Q., Kezia, K., Bowman, E.G., Vasiljevic, T., Kentish, S.E., 2016. Removal of lactate from acid whey using nanofiltration. J. Food Eng. 177, 59–64. Available from: https://doi.org/10.1016/j.jfoodeng.2015.12.019.

Chen, H.L., Forbes, E.G.A., Archer, J., De Priall, O., Allen, M., Johnston, C., et al., 2019. Production and characterization of granules from agricultural wastes and comparison of combustion and emission results with wood based fuels. Fuel 256, 115897. Available from: https://doi.org/10.1016/j.fuel.2019.115897.

Chen, W.H., Peng, J.H., Bi, X.T.T., 2015. A state-of-the-art review of biomass torrefaction, densification and applications. Renew. Sustain. Energy Rev. 44, 847−866. Available from: https://doi.org/10.1016/j.rser.2014.12.039.

Chen, Z.S., Xing, J.L., Pu, Z.X., Wang, X.X., Yang, S.Y., Wei, B.B., et al., 2018. Preparation of nano-Fe-0 modified coal fly-ash composite and its application for U(VI) sequestration. J. Mol. Liq. 266, 824−833. Available from: https://doi.org/10.1016/j.molliq.2018.05.118.

Cheng, S.Y., Tan, X.F., Show, P.L., Rambabu, K., Banat, F., Veeramuthu, A., et al., 2020. Incorporating biowaste into circular bioeconomy: a critical review of current trend and scaling up feasibility. Environ. Technol. Innov. 19, 101034. Available from: https://doi.org/10.1016/j.eti.2020.101034.

Devi, R., Dhamodharan, R., 2018. Pretreatment in hot glycerol for facile and green separation of chitin from prawn shell waste. ACS Sustain. Chem. Eng. 6 (1), 846−853. Available from: https://doi.org/10.1021/acssuschemeng.7b03195.

Ding, F.Y., Deng, H.B., Du, Y.M., Shi, X.W., Wang, Q., 2014. Emerging chitin and chitosan nanofibrous materials for biomedical applications. Nanoscale 6 (16), 9477−9493. Available from: https://doi.org/10.1039/c4nr02814g.

European Commission. Biodegradable waste, 2022. https://ec.europa.eu/environment/topics/waste-and-recycling/.

Finnveden, G., Hauschild, M.Z., Ekvall, T., Guinee, J., Heijungs, R., Hellweg, S., et al., 2009. Recent developments in life cycle assessment. J. Environ. Manag. 91 (1), 1−21. Available from: https://doi.org/10.1016/j.jenvman.2009.06.018.

Gonzalez-Ballesteros, N., Rodriguez-Gonzalez, J.B., Rodriguez-Arguelles, M.C., 2018. Harnessing the wine dregs: an approach towards a more sustainable synthesis of gold and silver nanoparticles. J. Photochem. Photobiol. B-Biol. 178, 302−309. Available from: https://doi.org/10.1016/j.jphotobiol.2017.11.025.

Gopi, S., Balakrishnan, P., Divya, C., Valic, S., Bajsic, E.G., Pius, A., et al., 2017. Facile synthesis of chitin nanocrystals decorated on 3D cellulose aerogels as a new multifunctional material for waste water treatment with enhanced anti-bacterial and anti-oxidant properties. N. J. Chem. 41 (21), 12746−12755. Available from: https://doi.org/10.1039/c7nj02392h.

Hintsho, N., Shaikjee, A., Triphati, P.K., Masenda, H., Naidoo, D., Franklyn, P., et al., 2016. Effect of nitrogen and hydrogen gases on the synthesis of carbon nanomaterials from coal waste fly ash as a catalyst. J. Nanosci. Nanotechnol. 16 (5), 4672−4683. Available from: https://doi.org/10.1166/jnn.2016.12434.

Jayakumar, R., Prabaharan, M., Kumar, P.T.S., Nair, S.V., Tamura, H., 2011. Biomaterials based on chitin and chitosan in wound dressing applications. Biotechnol. Adv. 29 (3), 322−337. Available from: https://doi.org/10.1016/j.biotechadv.2011.01.005.

Lee, C.H., Singla, A., Lee, Y., 2001. Biomedical applications of collagen. Int. J. Pharm. 221 (1−2), 1−22. Available from: https://doi.org/10.1016/s0378-5173(01)00691-3.

Lipkowski, A.W., Gajkowska, B., Grabowska, A., Kurzepa, K., 2009. Keratin-associated protein micromaterials for medical and cosmetic applications. Polimery 54 (5), 386−388. Available from: https://doi.org/10.14314/polimery.2009.386.

Lu, K.M., Lee, W.J., Chen, W.H., Lin, T.C., 2013. Thermogravimetric analysis and kinetics of co-pyrolysis of raw/torrefied wood and coal blends. Appl. Energy 105, 57−65. Available from: https://doi.org/10.1016/j.apenergy.2012.12.050.

Klitkou, A., Bolwig, S., 2019. Adding value to side-streams in the food and beverage industry: lessons for the circular bioeconomy.

Monteiro, E., Mantha, V., Rouboa, A., 2011. Prospective application of farm cattle manure for bioenergy production in Portugal. Renew. Energy 36 (2), 627−631. Available from: https://doi.org/10.1016/j.renene.2010.08.035.

Mussatto, S.I., 2014. Brewer's spent grain: a valuable feedstock for industrial applications. J. Sci. Food Agric. 94 (7), 1264−1275. Available from: https://doi.org/10.1002/jsfa.6486.

Muthu, J., Bradley, P., Jinasena, I.I., Durbach, S., Moya, A., Paskaramoorthy, R., 2018. The effects of carbon nanofiber on the mechanical properties of glass/coir fiber reinforced polyester hybrid composites. Polym. Compos. 39 (2), 318−328. Available from: https://doi.org/10.1002/pc.23938.

Nasir, I.M., Ghazi, T.I.M., Omar, R., 2012. Anaerobic digestion technology in livestock manure treatment for biogas production: a review. Eng. Life Sci. 12 (3), 258−269. Available from: https://doi.org/10.1002/elsc.201100150.

Paledal, S.N., Hellman, E., Moestedt, J., 2018. The effect of temperature, storage time and collection method on biomethane potential of source separated household food waste. Waste Manag. 71, 636−643. Available from: https://doi.org/10.1016/j.wasman.2017.05.034.

Parenteau-Bareil, R., Gauvin, R., Berthod, F., 2010. Collagen-based biomaterials for tissue engineering applications. Materials 3 (3), 1863−1887. Available from: https://doi.org/10.3390/ma3031863.

Patil, P.O., Bhandari, P.V., Deshmukh, P.K., Mahale, S.S., Patil, A.G., Bafna, H.R., et al., 2017. Green fabrication of graphene-based silver nanocomposites using agro-waste for sensing of heavy metals. Res. Chem. Intermed. 43 (7), 3757−3773. Available from: https://doi.org/10.1007/s11164-016-2844-9.

Pillai, C.K.S., Paul, W., Sharma, C.P., 2009. Chitin and chitosan polymers: chemistry, solubility and fiber formation. Prog. Polym. Sci. 34 (7), 641−678. Available from: https://doi.org/10.1016/j.progpolymsci.2009.04.001.

Pituello, C., Francioso, O., Simonetti, G., Pisi, A., Torreggiani, A., Berti, A., et al., 2015. Characterization of chemical-physical, structural and morphological properties of biochars from biowastes produced at different temperatures. J. Soils Sediment. 15 (4), 792−804. Available from: https://doi.org/10.1007/s11368-014-0964-7.

Rodsrud, G., Lersch, M., Sjode, A., 2012. History and future of world's most advanced biorefinery in operation. Biomass Bioenergy 46, 46−59. Available from: https://doi.org/10.1016/j.biombioe.2012.03.028.

Saidur, R., Abdelaziz, E.A., Demirbas, A., Hossain, M.S., Mekhilef, S., 2011. A review on biomass as a fuel for boilers. Renew. Sustain. Energy Rev. 15 (5), 2262−2289. Available from: https://doi.org/10.1016/j.rser.2011.02.015.

Salah, N., Abdel-Wahab, M.S., Alshahrie, A., Alharbi, N.D., Khan, Z.H., 2017. Carbon nanotubes of oil fly ash as lubricant additives for different base oils and their tribology performance. RSC Adv. 7 (64), 40295−40302. Available from: https://doi.org/10.1039/c7ra07155h.

Salah, N., Alshahrie, A., Alharbi, N.D., Abdel-wahab, M.S., Khan, Z.H., 2019. Nano and micro structures produced from carbon rich fly ash as effective lubricant additives for 150SN base oil. J. Mater. Res. Technol. 8 (1), 250−258. Available from: https://doi.org/10.1016/j.jmrt.2017.12.003.

Schneir, J.R., 2015. The entrepreneurial state: debunking public vs. private sector myths. Telecommun. Policy 39 (7), 623−624. Available from: https://doi.org/10.1016/j.telpol.2015.04.002.

Smithers, G.W., 2015. Whey-ing up the options - yesterday, today and tomorrow. Int. Dairy. J. 48, 2−14. Available from: https://doi.org/10.1016/j.idairyj.2015.01.011.

Staffas, L., Gustavsson, M., McCormick, K., 2013. Strategies and policies for the bioeconomy and bio-based economy: an analysis of official national approaches. Sustainability 5 (6), 2751−2769. Available from: https://doi.org/10.3390/su5062751.

United Nations Statistics Division, 2022. https://unstats.un.org/.

Verlee, A., Mincke, S., Stevens, C.V., 2017. Recent developments in antibacterial and antifungal chitosan and its derivatives. Carbohydr. Polym. 164, 268−283. Available from: https://doi.org/10.1016/j.carbpol.2017.02.001.

Visa, M., Andronic, L., Duta, A., 2015. Fly ash-TiO_2 nanocomposite material for multi-pollutants wastewater treatment. J. Environ. Manag. 150, 336−343. Available from: https://doi.org/10.1016/j.jenvman.2014.10.026.

Wang, W.X., Martin, J.C., Fan, X.T., Han, A.J., Luo, Z.P., Sun, L.Y., 2012. Silica nanoparticles and frameworks from rice husk biomass. ACS Appl. Mater. Interfaces 4 (2), 977−981. Available from: https://doi.org/10.1021/am201619u.

Xu, C.P., Nasrollahzadeh, M., Selva, M., Issaabadi, Z., Luque, R., 2019. Waste-to-wealth: bio-waste valorization into valuable bio(nano)materials. Chem. Soc. Rev. 48 (18), 4791−4822. Available from: https://doi.org/10.1039/c8cs00543e.

Yang, H.Y., Gozaydin, G., Nasaruddin, R.R., Har, J.R.G., Chen, X., Wang, X.N., et al., 2019. Toward the shell biorefinery: processing crustacean shell waste using hot water and carbonic acid. ACS Sustain. Chem. Eng. 7 (5), 5532−5542. Available from: https://doi.org/10.1021/acssuschemeng.8b06853.

Chapter 14

Biogas production as a sustainable waste valorisation technology: perspectives from Namibia

Andreas S. Namwoonde[1], **Pineas Tomas**[2] and
Hupenyu A. Mupambwa[1]
[1]Sam Nujoma Marine and Coastal Resources Research Centre, Sam Nujoma Campus, University of Namibia, Henties Bay, Namibia, [2]Multidisciplinary Research; Center for Research Services, Main Campus, University of Namibia, Windhoek, Namibia

14.1 Introduction

The shift from nomadic food production to a fixed system has seen the intensified use of land resources enabling the production of high food quantities in small areas; however, this has seen agricultural systems like animal production generating huge quantities of animal waste due to the confined handling of animals like cows, chickens, pigs, goats and sheep. Under smallholder farming systems, most farmers use animal manure as nutrient sources in their fields, where unprocessed manure is applied indiscriminately to the fields (Mupambwa and Mnkeni, 2018). Such disposal carries environmental challenges, such as the introduction of harmful trace metals, inorganic salts and pathogens into the soil (Lazcano et al., 2008). Apart from technologies like composting and vermicomposting that have been promoted in animal manure valorisation, biogas production is another valorisation technology that is not new but has, of late, been promoted for uptake by resource-poor smallholder farmers in Africa. Biogas is being promoted due to its limited environmental footprint as it generates clean energy, while the products of this process, called the 'digestate', also have potential for use as an organic source of nutrients in agriculture (Mupambwa et al., 2019). The management of animal manures through biogas production is being widely promoted as an important renewable energy source (Sieling et al., 2013).

The process of biogas production involves various processes that all take place under anaerobic conditions resulting in the production of mostly

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00001-3

methane gas and other minor gases like carbon dioxide, nitrogen, hydrogen sulphide and hydrogen (Goswami et al., 2016). Methane, the product of anaerobic digestion of organic materials, is considered as an important greenhouse gas that is responsible for increasing the greenhouse effect. It is, therefore, important that during biogas production, all the gas is fully harvested to avoid its release into the atmosphere if the process is to be termed clean. Apart from the anaerobic process, biogas is also, however, produced by various natural processes, such as termite activity, wetlands, oceans and anthropogenic activities, such as livestock production, rice cultivation and mining, among others.

Although the biogas generation process is not a new process, the global production of biogas has been increasing, improving by 4% over 9 years between 2010 and 2018 (Abanades et al., 2022). The use of this source of energy has been more pronounced in developed countries where large-scale advanced plants have been established where they generate biogas that is used to generate electricity and heat for municipal or national grids (Abanades et al., 2022). In developing countries in Africa and Asia, most biogas installations are, however, family-sized plants that generate energy for household use only (Kemausuor et al., 2018). In Africa, there is a huge potential for biogas energy generation, although this has not been exploited, although renewable energy development, which includes biogas, is among the critical areas of the African Unions Agenda 2063 (Kemausuor et al., 2018). Our chapter seeks to further stimulate literacy on the potential of biogas production as a waste valorisation technique and its capacity to generate energy and organic fertilizers for smallholder resource-limited farmers.

14.2 The energy case of Namibia

Namibia is a less and sporadically populated country, with many inhabitants living in remote areas, making it hard to distribute electricity to all citizens, especially in rural areas. Although the Namibian government has a policy on rural electrification, still many rural settlements do not have access to the grid electricity, mainly due to the fact that they live far from the power lines or cannot afford to pay for the high connection fees. According to the Namibia Household Income and Expenditure Survey (NSA, 2018) '*access to energy is one of the indicators of socio-economic status of the household*', and over 85% of rural households are using firewood as a primary source of energy. Apart from deforestation and triggering climate change, firewood consumption is associated with household indoor pollution, while women and children spend their time unproductively looking for firewood, which could be spent on doing homework and studies. Only over 9% of rural households in Namibia use electricity and less than 4% use imported liquefied petroleum gas (LPG); however, the affordability of electricity and LPG remain a burden to individual end-users. Namibia has limited electricity generation; LPG and paraffin are not produced locally, and their accessibility depends on geographical location; hence, the final end price depends on value-added tax and transportation costs, making them unaffordable to many low-income earners. Because of the availability of biomass for biogas generation in rural areas, biogas production can relieve the communities from energy constraints with clean, sustainable energy source by portable biogas digesters. In Namibia biogas is not yet applied on a mass scale, and a few demonstration scale biodigesters were built in a few parts of the country.

14.3 Biogas technology

Anaerobic digestion as a means of alternative energy production can substitute part of fossil fuel demands in many farming communities due to the availability of animal and plant waste. Anaerobic digestion of organic substrate takes place in the absence of atmospheric oxygen, during which complex organic matter goes through four main steps, as indicated in Fig. 14.1.

The chamber where anaerobic digestion takes place is referred to as the biogas digester, or in short, biodigester. The digester type depends mostly on the type and amount of waste to be treated and geographical location. The domestic biogas digesters differ completely from the industrial waste treatment digesters; however, they all serve the common purpose of providing an anaerobic environment and allowing the storage of the produced biogas. The biogas digesters mainly belong to three groups: fixed domes, floating drums and bio-bag digesters.

The fixed dome digester (Fig. 14.2A) is also called the 'hydraulic or Chinese digester' (Obileke et al., 2021) and is made up of a gasholder dome and slurry chamber built underground; this digester type can withstand high-pressure build up and experience less temperature variation, but it can easily be flooded if it is not well placed. Since this type of digester is built underground, it is protected from physical damage and temperature fluctuations during cold nights, as temperature fluctuations influence microbial activity, thus affecting biogas generation (Kossmann et al., 1999).

The floating drum digester (Fig. 14.2B) has a gasholder made of a drum that rises with the rise in biogas pressure; its slurry chamber can be dug in the ground or made from a portable tank, making it possible to be placed in different locations. Floating drum digesters are made from flexible plastic materials that are durable and UV resistant, and they are also portable and fast to install, but can be affected by ambient temperature variations. The design is such that the top inverted drum floats on top of the digesting materials to capture the gas, and this floating drum rises and falls with changes in gas pressure (Obileke et al., 2021).

The balloon digester (Fig. 14.2C) is designed such that the inlet and outlet are attached to the inflatable structure, which is not rigid but expands as the gas accumulates. As this type of digester is placed above ground and not

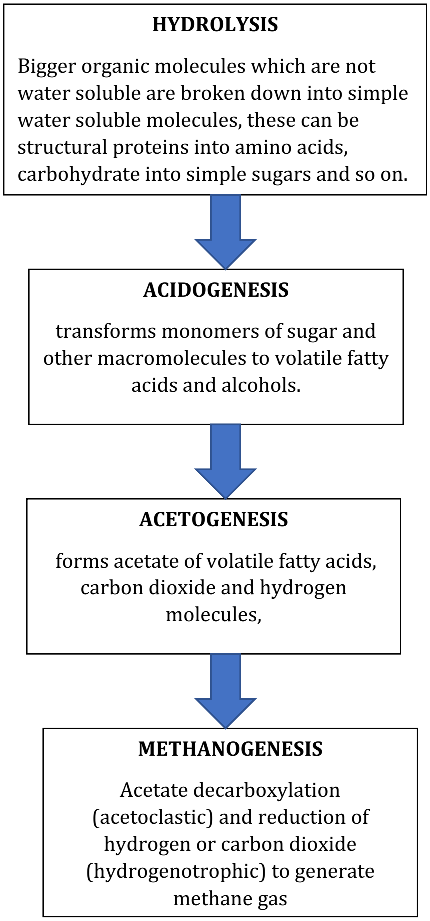


FIGURE 14.1 Flow diagram showing the various stages involved in the biogas generation process.

underground, the material used must be UV protected and be able to resist the elements (Obileke et al., 2021).

The quantity and quality of biogas, apart from the substrate type, can also depend on various factors as the digestion process involves various microbial processes, as indicated in Fig. 14.2. These factors include the temperature of the digester, carbon to nitrogen ratio of materials, the concentration of organics, nutritional content and the pH (Goswami et al., 2016). The biogas potential of a substrate is a measure of biogas volume produced per unit mass of a volatile solid of a specific biomass during a given time period at a specific temperature (Kuo and Dow, 2017). Anaerobic digestion is a temperature-sensitive process, and there are two temperature ranges where the production

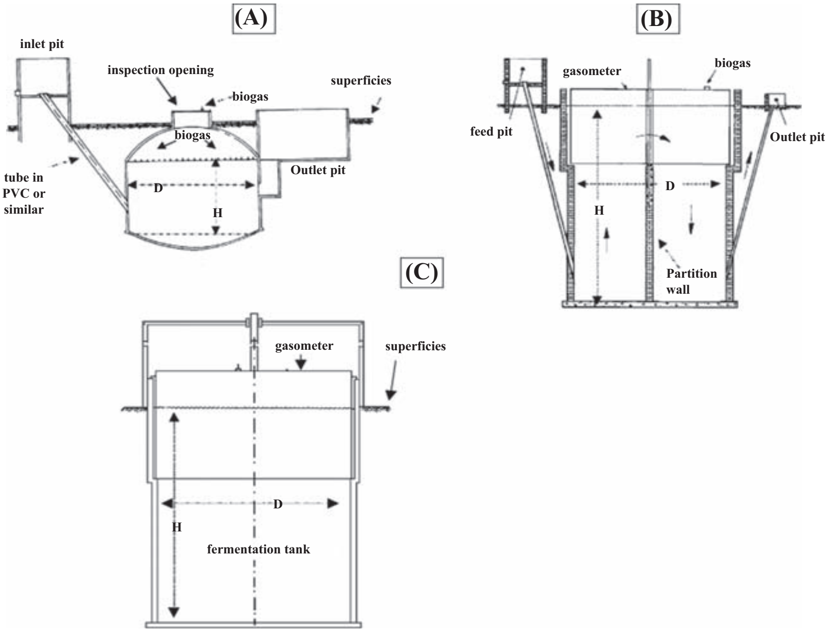


FIGURE 14.2 Structure of the three main types of biogas digesters i.e. (A) fixed dome, (B) floating drum and (C) ballon (Florentino, 2003). Source*: From Florentino H. Mathematical tool to size rural digesters. Sci. Agric. (Piracicaba, Braz.). 2003;60:185–190.*

of biogas is appropriate, i.e. mesophilic (32°C–43°C) and thermophilic (48°C–60°C). Mesophilic temperatures are favoured mostly due to the low energy demand required to keep the process running; however, thermophilic temperatures result in high biogas yield and low biomass retention time, which means there should be constant refilling of the biodigester, and the high temperature can lead to high energy bills to generate the thermophilic temperatures, thus defeating the purpose of biogas production as sustainable energy generation process.

Biogas is made up of 40%–75% of biomethane, and carbon dioxide can take up 15%–60%, while other components like hydrogen sulphide, nitrogen, oxygen and water vapour can exist in trace forms, which heavily depend on the types of substrate used (Bhatt and Tao, 2020). The application of biogas varies widely; it can be supplemented in the natural gas pipelines, especially in developed countries. Well-cleaned biogas can be used in powering engines for vehicles and electric generators, replacing diesel, petrol and paraffin. In developing countries, biogas is commonly used for heating and cooking, which reduces the risk of indoor pollution associated with the burning of firewood using traditional stoves (Iqbal et al., 2021). The quality of biogas can be improved by applying various pretreatment methods to the substrate during anaerobic digestion, however, there are many methods of treating biogas after production to remove carbon dioxide, hydrogen sulphide, water vapour and other components to increase the concentration of methane. Removal of hydrogen sulphide is key if the biogas will be used internal combustion engines, hydrogen sulphide is very corrosive, which can be very detrimental to the engine's efficiency.

14.4 Animal waste processing into biogas

Animal farming is one of the important agricultural sector in developing countries like Namibia, with cattle farming contributing significantly to the GDP of the country through beef export to markets in America, Europe and Asia. Furthermore, other livestock like goats, sheep and chicken are farmed mainly for domestic consumption.

Commonly, in animal husbandry, cattle heads and other big animals are allowed to wander freely on the grazing land and only sleep in the enclosure during the night, allowing the accumulation of animal dung and dropping in one area. If there are no further use of animal dung at the farm, waste management problems will arise (Bhatt and Tao, 2020), posing dangers to the health of not only the farm inhabitants but also to those of the animals and the general hygiene. Further processing of animal waste for biogas production does not only provide much needed alternative energy but also reduce release of methane into the atmosphere that contribute to global warming (Kuo and Dow, 2017).

Animal waste presents a great opportunity in the waste-to-energy concept as a way of waste management and value addition to animal waste around the farm, helping to attain sustainable energy development in many African countries where access to clean energy still remains a great challenge. According to the International Energy Agency report (2018), the global energy demand increased by more than 2% in the past years. Of that demand, however, over 70% was produced from fossil fuel, leading to the high emission of greenhouse gases, which causes global warming and climate change (Table 14.1).

14.5 Biogas digester design: experiences from Namibia

In Namibia, biogas production was mostly at wastewater treatment plants with very few functional industrially sized biogas systems being available in the country. Research has, however, been done at the University of Namibia's Sam Nujoma Marine and Coastal Resources Research Centre, Sam Nujoma Campus, located in the coastal town of Henties Bay in Namibia, with some of the experiences being presented below.

An experiment was carried out, and the project was used as a case study and a wider demonstration of biogas uses for domestic purposes in Namibia. The portable biodigesters were used in this study and these are suitable for

TABLE 14.1 Summary of research done on biogas using various organic materials.

Manure or organic matter sources	Experimental conditions	Main results	References
Pig manure (PM), fish waste and biodiesel waste.	Co-digestion of pig manure with fish waste (FW was delivered by a canning industry, and it consisted of heads, tails, bones and viscera of tuna fish) and biodiesel waste evaluated in a continuously stirred digester at 160 rpm, reactors were operated at 35°C by hot water recirculation temperature 35. Digestion done between 150 and 200 days, with sequential addition of new materials. Monitoring twice for temperature, pH, stirring speed and biogas production were monitored online with solids, chemical oxygen demand (COD), alkalinity, volatile fatty acids, ammonium and biogas production.	Pig waste and fish waste co-digestion (90:10 and 95:5, w/w) was possible at an organic loading rate (OLR) of 1–1.5 g chemical oxygen demand (COD) L^{-1} day, resulting in biogas production rates of 0.4–0.6 L L^{-1} day and COD removal efficiencies of 65%–70%. BW, good results (biogas production of 0.9 L L^{-1} day and COD elimination of 85%) were achieved with less than a 5% feeding rate. A protein-rich substrate (FW) was more problematic to co-digest with PM due to their similar characteristics (high nitrogen content), but good results were achieved by keeping the FW in the feeding mixture below 10%.	Regueiro et al. (2012)
Slaughterhouse waste, which included paunch (PA), soft offal (SO) (intestinal residues, fat and meat trimmings and some blood), as well as dissolved air flotation sludge (DAF) from the wastewater treatmentfacility onsite.	The three selected waste streams SO, PA and DAF on an individual basis and also as a mixture according to their annual production ratios (1:2.55:3.22-PA:DAF: SO), i.e. SHWM.	Highest methane yield was achieved from the SO at 651 mL methane per g of volatile solids followed closely by SHWM at 642 mL methane, PA accumulating less than half of this with a yield of 229 mL methane, and finally, the DAF had a much lower yield of 50 mL methane per g of volatile solids. The net energy analysis (both the thermal and electrical) of the slaughterhouse that culls up to 52,000 cows could be met from the energy generated through the combustion of the biogas in a combined heat and power unit with electrical and thermal efficiencies of 41% and 49%, respectively.	Ware and Power (2016)

(*Continued*)

TABLE 14.1 (Continued)

Manure or organic matter sources	Experimental conditions	Main results	References
Food waste (FW), both edible and non-edible parts; wastewater sludge (WWS), meat and bone meal (MBM).	Two experiments where the substrates (FW and MBM and FW and WWS) were hydrolysed at 35°C for 5 days, and the anaerobic digestion was performed on pretreated substrates. In the first stage, MBM and WWS were added to FW in the amounts of 0%, 5%, 10% and 15% on a TS basis. In the second stage, two mixtures were selected for anaerobic digestion, i.e. a control mixture (with no added co-substrate, thus only FW) and a mixture with the overall best parameters according to the selection criteria.	Thermal pretreatment of these mixtures at a temperature of 35°C for a 5-day duration showed no impact on the pH, while concentrations of both chemical oxygen demand and ammonia increased. Anaerobic digestion of both samples containing MBM or WWS causes antagonistic effects in terms of biogas production when added to FW. Adding 5% MBM to FW decreased biogas production by 12%, while adding 5% WWS to FW decreased biogas production by 23%. The study concluded that the usage of waste and residue materials to produce renewable energy, like biogas, is more complex and requires higher level analysis, compared to the use of common substrates to produce biogas, e.g., cultivated energy crops.	Bedoić et al. (2020)
Fruit-juice industrial waste (FW) and municipal sewage sludge (MSS).	Samples of apple waste (AW), pomegranate waste (PW), and black carrot waste (BCW) were obtained from a food industry were used. The municipal sewage sludge (MSS was collected from a municipal wastewater treatment plant). Digestate from a lab-scale batch digester fed with cattle manure was used as the inoculum. The mixture was at 75% FW and 25% MSS, and these were subjected to various pre-treatments, i.e. ultrasonic pretreatment; microwave pretreatment; acid pre-treatment; alkaline pretreatment; thermal pretreatment; ultrasonic acid and alkaline pretreatment. Batch experiment undertaken.	The pretreatment of the substrates with ultra-sonication, microwave and weak alkali-acid caused an increase in cumulative biogas production of approximately 21%; 15%; 8% and 5%, respectively. Thermal and strong acid-alkali pretreatment reduced biogas production. The highest cumulative biogas and methane yield was increased with hybrid pretreatment, which contains ultra-sonication and alkali pretreatment by 36% and 49%, respectively. This increase in biogas production in the hybrid pre-treatment was attributed to the increase in pore sizes of organic compounds, which increased the contact area for the microorganisms and enzymatic activity.	Gulsen Akbay et al. (2021)

Grease trap waste collected from the aerated grit chamber from a wastewater treatment plant (GTW) and sewage sludge (SS).	A mixture of primary and secondary thickened sludge and anaerobically digested sludge with the SS being used as inoculum. Laboratory alkali pre-treatment using sodium hydroxide. The pre-treatment was conducted at 25°C for 20 h. Samples with different alkali: volatile solids ratios (20–25 $\text{gNaOH} \cdot \text{kg VS}^{-1}$)	The results confirmed that alkali pre-treatment of GTW at low temperatures is a technically and economically viable solution to integrate GTW management in wastewater treatment plants. Results also showed that the optimal conditions for optimising grease extraction and GTW biodegradability are NaOH doses of >20 g NaOH per kg of volatile solids and a pre-treatment time of >6 h.	Romero-Güiza et al. (2021)

use in Namibian rural areas where families keep a wide range of animals, which generate large quantities of manure, and temperatures are also favourable for biogas production as the temperatures range around 30°C most times of the year. A floating biogas digester was used and this contains two main components, which are sludge and gasholder. It is the gasholder that keeps moving up and down due to the pressure of gas accumulation. In this design, the sludge holder is portable and can be moved from place to place, making the digester easy to use under different conditions.

14.5.1 Sludge holder

This project employed a water tank with a working capacity of 1000 L made from polyethylene plastic, which was converted into a sludge holder. The tank top cover was removed using an electric saw right where it joined the tank's wall, leaving it open from the inner diameter. At 0.3 m above the tank's bottom, a substrate inlet was connected using a 90 mm tank connector, a 90 degrees elbow connector pointing upward was attached to the tank connector, and thereafter, a 90 mm PVC pipe was connected, which gave the total height of the feeder from the tank's base to 0.8 M. Equally, the same materials were used to for digestate outlet but were placed on the opposite side of the inlet facing downward at 0.8 m height from the bottom. To calculate the working volume of the sludge holder, the formula (Eq. 14.1) finding the volume of a cylindrical object was used:

$$\begin{aligned} &V = \pi r^2 h \\ &h = 0.8\ \text{m},\ r = 0.57\ \text{m} \end{aligned} \tag{14.1}$$

From these parameters, the maximum sludge volume that the digester can hold is roughly 0.8 m^3. In addition to the inlet and outlet, a ball valve with a 57 mm ball valve was fitted at the bottom side of the digester for the complete draining of the sludge.

14.5.2 Gasholder

A handmade gasholder was used, made from fibre-glass and consisted of two figurative shapes: a cylindrical base and a cone-shaped head, and these two shapes contributed to the total volume of the gasholder. On top of the side of the cone head, a gas outlet was attached and controlled using a metallic ball valve. Additionally, to keep the digestate well mixed, a homogeniser was fitted inside the gasholder, made from 30 mm PVC pipes attached with cable tied to the bottom of the digester; it mixes the content during the lifting movement of the gasholder when it is filled with gas and down movement during biogas usage, the floating digester can also be turned around manually for additional mixing.

The cylindrical volume of the gasholder was obtained using equation 1 while the conical volume was calculated using Eq. (14.2).

$$V = \pi r^2 h/3 \; r = 0.5, \; h = 0.3 \text{ m}, \tag{14.2}$$

The cylindrical volume of the gasholder was roughly 0.55 m^3 while that of the conical part was 0.08 m^3, this resulted in a volume of 0.63 m^3 for the gasholder. The total mass for the gasholder was 65 kg.

14.5.3 Digestate preparation and cooking time

Fresh cattle dung of dairy cows was collected from Neudamm campus of the University of Namibia in sealed plastic drums and transported to SANUMARC. The manure was mixed with water in the ratio of 1:1 and fed into the digester at 8% TS digestate concentration. The digester was set up in a batch system, whereby there was no new feed added to the system until the retention time was reached. Because of the volume of the digester and lack of adequate volume measuring tools, it was decided to measure only cooking time using a one-plate camping gas stove and the gas flow rate during cooking. The measurements were carried out at peak gas production, the highest burning time was 90 minutes with the lowest at 60 minutes and average at 72 minutes. The average gas flow rate was 3.5 L min^{-1} and the digester model is indicated in Fig. 14.3.

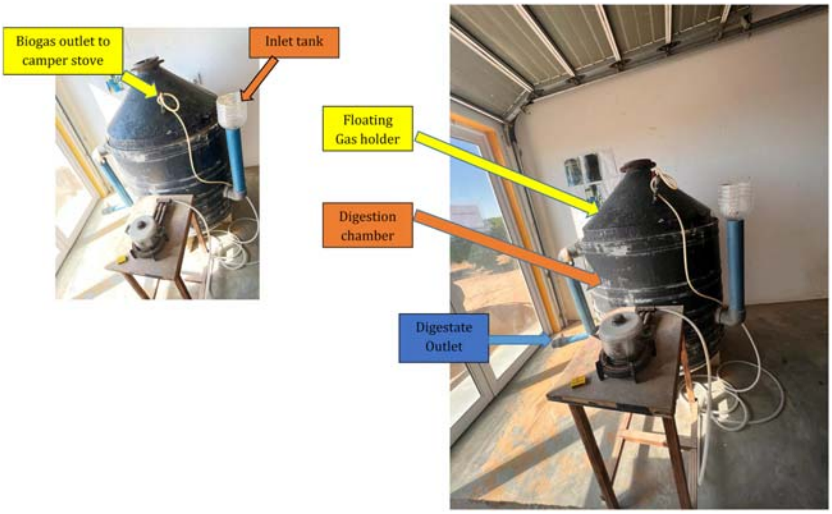


FIGURE 14.3 Schematic image showing major parts of a floating drum biogas digester at the University of Namibia. Source*: Images by A.S. Namwoonde.*

14.6 Biogas digestates in agriculture

With biogas production being promoted as a clean source of energy, the other main challenge is that it still generates solid waste in the form of digestate that needs to be managed during disposal. As a way of addressing this, several researchers have looked into the possibility of using the digestate from biogas as a nutrient source in agriculture. Although depending on organic materials used, studies have shown that the digestate from biogas are high quality nutrient materials rich in macro nutrients, such as P, K and in particular, nitrogen (Chen et al., 2017). The liquid fraction of the biogas digestates have thus been evaluated as potential fertilizers sources in soils and hydroponic systems, with the major challenge being their potential phytotoxicity, heavy metal and pathogen addition to the soil and plants (Lencioni et al., 2016; Krishnasamy et al., 2012). In a study, Krishnasamy et al. (2012) evaluated the potential of the diluted biogas digestates as a nutrient solution for silverbeet in a hydroponic system. It was observed that silverbeet survival was best at 20% digestate, while at 50%, silverbeet survival was negatively affected due to ammonia toxicity and low oxygen (Krishnasamy et al., 2012). Lencioni et al. (2016) evaluated the phytotoxicity of pig digestate, which had been diluted from 5% to 30% on different plants, reporting that digestate concentrations that stimulated germination and early seedling growth were as low as 2%−3%, while 20%−30% could be used for the advanced stages with limited negative effects. What is clear in most of these studies is that the use of biogas digestate as a direct source of nutrients might be a challenge and some authors have done research to evaluate these digestates either as a nutrient source for hydroponics or directly as a soil conditioner, with Table 14.2 summarising this research.

TABLE 14.2 Summary of research done on biogas digestate use in agriculture.

Digestate organic source	Experimental conditions	Main results	References
Cow manure	Digestate from cow manure was used as a hydroponic nutrient source for tomatoes. For crop phytotoxicity, digestate was diluted with deionised water to give the following treatments: undiluted biogas digestate; 10%; 20% and 40% biogas digestate and tested on tomato, spinach, carrot, beetroot and cabbage. A 10% diluted digestate solution was used and this was supplemented with commercial hydroponic fertilizer at 20%; 40% and 60%.	The highest relative seed germination was recorded in the 10% biogas digestate mixture, while the 40% treatment gave the lowest RSG. For the crop growth study, relative to the control, the treatments with 20%, 40% and 60% mineral fertilizer substitution resulted in 39.4%; 22.8% and 8.7% lower chlorophyll content. On average, the treatments with biogas slurry, although substituted with mineral fertilizers, resulted in a 275% lower fresh fruit yield compared to the control treatment. The results of the study demonstrated that cow based digestates are not a suitable nutrient media for hydroponic tomato production.	Mupambwa et al. (2019)
Digestates made from various organic materials that included slaughterhouse waste, pig slurry, cow manure, food processing waste and household waste.	Twenty digestate samples were compared to pig slurry and cow manure. Various chemical properties of the organic mixtures were determined but not tested from on crops.	The digestates all contained significantly higher concentrations of ammonium, while the concentrations of total carbon and volatile fatty acids were higher in the pig slurry and cow manure. The digestate showed both stimulating and inhibiting effects on potential ammonia oxidation rate (PAO), while all cow manure and pig slurry showed inhibiting effects on PAO. Overly, the digestates were different compared with pig slurry and cow manure but without posing a higher risk with respect to their impact on soil microbial activity.	Risberg et al. (2017)

(*Continued*)

TABLE 14.2 (Continued)

Digestate organic source	Experimental conditions	Main results	References
Poultry biogas slurry	Assessing the feasibility of using poultry biogas slurry used in different proportions with mineral fertilizers in a hydroponics experiment with lettuce. Four treatments were established: HS (half-strength Hoagland solution), BS (2.6% biogas slurry), BS + HS (1.3% biogas slurry + quarter-strength Hoagland solution), and BS + MF (2.6% biogas slurry + mineral fertilizers).	Mixture of poultry biogas slurry with half strength Hoagland solution did not significantly affect lettuce growth but significantly increased the soluble sugar concentration, reduced the nitrate concentration, while the concentrations of heavy metals were still within the safety standards. Biogas slurry could be used to replace mineral fertilizer at 50% for lettuce production although there is need to carefully monitor electrical conductivity and missing nutrients need to be replenished especially magnesium.	Wang et al. (2019)
Biogas slurry from plant residues obtained from local agriculture	Biogas slurry was compared to normal Spirulina (*Arthrospira platensis*) media. The biogas slurry was filtered in polyamide filters in several steps with a final mesh size of 5 μm to remove particles.	Equal biomass production was observed and no significant differences were observed between the treatments in protein amount, amino acid composition, and total lipid concentration of Spirulina. The concentration of the pigment phycocyanin differed significantly between spirulina medium (63 mg g^{-1}) and the effluent-based medium (86 mg g^{-1}). It is important to control heavy metal concentrations in the biogas slurry fed to spirulina.	Hultberg et al. (2017)

14.7 Conclusions

The technology of biogas digestion is not new but it has been hardly adopted under smallholder resource-poor farmers who still rely on environment polluting and damaging methods of heating, such as use of firewood and other liquid fuels like paraffin. Our chapter highlights the potential biogas digestion has in converting household animal manures into useful energy sources with the same method being capable of being used in dealing with other wastes like human wastes in places where the pit latrine is still used. More research on this is, however, required, as well as research on the use of the digestate in agriculture fertilisation as research has shown that the digestates may increase the electrical conductivity of the planting medium. With the technology being simple, there is need for efforts that drive the successful adoption of such technologies among resource-poor farming sectors, thus contributing to the mitigation actions towards climate change.

References

Abanades, S., Abbaspour, H., Ahmadi, A., et al., 2022. A critical review of biogas production and usage with legislations framework across the globe. Int. J. Environ. Sci. Technol. 19, 3377–3400. Available from: https://doi.org/10.1007/s13762-021-03301-6.

Akbay, Habibe Elif Gulsen, Dizge, Nadir, Kumbur, Halil, 2021. Enhancing biogas production of anaerobic co-digestion of industrial waste and municipal sewage sludge with mechanical, chemical, thermal, and hybrid pretreatment. Bioresour. Technol. 340 (2021), 125688. Available from: https://doi.org/10.1016/j.biortech.2021.125688. ISSN 0960–8524.

Bedoić, Robert, Špehar, Ana, Puljko, Josip, Čuček, Lidija, Ćosić, Boris, Pukšec, Tomislav, et al., 2020. Opportunities and challenges: experimental and kinetic analysis of anaerobic co-digestion of food waste and rendering industry streams for biogas production. Renew. Sustain. Energy Rev. 130 (2020), 109951. Available from: https://doi.org/10.1016/j.rser.2020.109951. ISSN 1364-0321.

Chen, G., Zhao, G., Zhang, H., Shen, Y., Fei, H., Cheng, W., 2017. Biogas slurry use as N fertilizer for two-season *Zizania aquatica* Turcz. in China. Nutrient Cycl. Agroecosystems 107, 303–320.

Florentino, H., 2003. Mathematical tool to size rural digesters. Sci. Agric. (Piracicaba, Braz.) 60, 185–190.

Goswami, Ramansu, Chattopadhyay, Pritam, Shome, Arunima, Nath Banerjee, Sambhu, Chakraborty, Amit Kumar, Mathew, Anil K., Chaudhury, Shibani, 2016. An overview of physico-chemical mechanisms of biogas production by microbial communities: a step towards sustainable waste management. 3 Biotech 6, 72. Available from: https://doi.org/10.1007/s13205-016-0395-9.

Hultberg, Malin, Lind, Olle, Birgersson, G.öran, Asp, Hakan, 2017. Use of the effluent from biogas production for cultivation of Spirulina. Bioprocess. Biosyst. Eng. 40 (2017), 625–631. Available from: https://doi.org/10.1007/s00449-016-1726-2.

Kemausuor, Francis, Adaramola, Muyiwa S., Morken, John, 2018. A review of commercial biogas systems and lessons for Africa. Energies 11, 2984. Available from: https://doi.org/10.3390/en11112984.

Kossmann, W., Pönitz, U., Habermehl, S., Hörz, T., Krämer, P., Klingler, B., et al., 1999. Biogas Digest (Volume II) - Biogas - Application and Product Development. GTZ.

Krishnasamy, K., Nair, J., Bauml, B., 2012. Hydroponic system for the treatment of anaerobic liquid. Water Sci. Technol. 65, 1164−1171.

Lazcano, C., Gomez-Brandon, M., Dominguez, J., 2008. Comparison of the effectiveness of composting and vermicomposting for the biological stabilization of cattle manure. Chemosphere 72, 1013−1019.

Lencioni, G., Imperiale, D., Cavirani, N., Marmiroli, N., Marmiroli, M., 2016. Environmental application and phytotoxicity of anaerobic digestate from pig farming by in vitro and in vivo trials. Int. J. Environ. Sci. Technol. 13, 2549−2560.

Mupambwa, H.A., Mnkeni, P.N.S., 2018. Optimizing the vermicomposting of organic waste materials for production of nutrient rich organic fertilizers: a review. Environ. Sci. Pollut. Res. 25, 10577−10595. Available from: https://doi.org/10.1007/s11356-018-1328-4.

Mupambwa, Hupenyu A., Namwoonde, Andreas S., Liswaniso, Gadaffi M., Hausiku, Martha K., Ravindran, Balasubramani, 2019. Biogas digestates are not an effective nutrient solution for hydroponic tomato (*Lycopersicon esculentum* L.) production under a deep water culture system. Heliyon 5 (2019), e02736.

Obileke, K.C., Onyeaka, H., Nwokolo, N., 2021. Materials for the design and construction of household biogas digesters for biogas production: a review. Int. J. Energy Res. 45, 3761−3779. Available from: https://doi.org/10.1002/er.6120.

Regueiro, L., Carballa, M., Álvarez, J.A., Lema, J.M., 2012. Enhanced methane production from pig manure anaerobic digestion using fish and biodiesel wastes as co-substrates. Bioresour. Technol. 123, 507−513. Available from: https://doi.org/10.1016/j.biortech.2012.07.109.

Risberg, Kajsa, Cederlund, Harald, Pell, Mikael, Arthurson, Veronica, Schnürer, Anna, 2017. Comparative characterization of digestate versus pig slurry and cow manure − chemical composition and effects on soil microbial activity. Waste Manag. 61, 529−538. Available from: https://doi.org/10.1016/j.wasman.2016.12.016. ISSN 0956-053X.

Romero-Güiza, M.S., Palatsi, J., Tomas, X., Icaran, P., Rogalla, F., Monsalvo, V.M., 2021. Anaerobic co-digestion of alkaline pre-treated grease trap waste: laboratory-scale research to full-scale implementation. Process. Saf. Environ. Prot. 149, 958−966. Available from: https://doi.org/10.1016/j.psep.2021.03.043. ISSN 0957−5820.

Sieling, K., Herrmann, A., Wienforth, B., Taube, F., Ohl, S., Hartung, E., et al., 2013. Biogas cropping systems: short term response of yield performance and N use efficiency to biogas residue application. Eur. J. Agron. 47, 44−54.

Wang, Lei, Guo, Shirong, Wang, Ying, Yi, Dandan, Wang, Jian, 2019. Poultry biogas slurry can partially substitute for mineral fertilizers in hydroponic lettuce production. Environ. Sci. Pollut. Res. 26 (2019), 659−671. Available from: https://doi.org/10.1007/s11356-018-3538-1.

Ware, Aidan, Power, Niamh, 2016. Biogas from cattle slaughterhouse waste: energy recovery towards an energy self-sufficient industry in Ireland. Renew. Energy 97 (2016), 541−549.

Chapter 15

Environmental waste regarding soft foreign body (plastic) – a threat to livestock health

Taimoor Hassan Farooq[1], **Vinod Bhateshwar**[2], **Awais Shakoor**[3],
Uttam Kumar[4], **Norela C.T. Gonzalez**[1], **Dinesh Chandra Rai**[2],
V.P. Aparnna[2] **and Yong Li**[1]

[1]*College of Life Science, Central South University of Forestry and Technology, Changsha, Hunan, China,* [2]*Department of Dairy Science and Food Technology, Institute of Agricultural Sciences, Banaras Hindu University, Varanasi, Uttar Pradesh, India,* [3]*Environment, Soils and Land Use Department, Teagasc, Johnstown Castle, Wexford, Ireland,* [4]*College of Plant Protection, Fujain Agriculture and Forestry University, Fujian, China*

15.1 Introduction

Uncontrolled waste disposal has negative consequences for the environment and human and animal health. It has been observed that animals, particularly ruminants, ingest indigestible materials (such as plastics, nails, rubber, wire, wood, and ropes), especially during droughts (Atawalna et al., 2015; Sheferaw et al., 2014). Animals consuming indigestible material are known to occur due to a lack of available feed. Furthermore, if their food does not contain enough critical minerals such as phosphorus (P), animals will ingest indigestible items such as plastic bags and metallic objects (Nongcula et al., 2017). In developing nations, especially those with a shortage of recycling businesses, environmental trash consists mostly of solid waste from home and commercial sources, particularly prevalent. Solid wastes can range from metallic to non-metallic in composition, and plastic wastes have lately become popular in various rural and urban settings (Mushonga et al., 2015; Semieka, 2010).

The term ‘plastic’ derives from the Greek word ‘plastikos’, which means ‘mouldable into many shapes’ (Joel, 1995). They are long-chain polymeric molecules (Zheng et al., 2005). Plastics are a broad category of chemical compounds that are either synthetic or semi-synthetic. They are mostly employed in the plastics manufacturing business and represent a hazard to animals and the environment (Reddy and Sasikala, 2012). The influence of

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00016-5

plastic wastes on bovine (Vanitha et al., 2010a,b; Abu-Seida and Al-Abbadi, 2014) and ovine (Pitroda et al., 2010). Additional nutritional deficiencies are caused by malnutrition and imbalanced diets, including pica and the consumption of things other than food, such as wastes (Ghurashi et al., 2009). The buildup of these wastes in the rumen of grazing animals may negatively affect their health (Ramaswa and Sharma, 2011).

Plastics can travel through the digestive system and end up in various regions of the digestive system, particularly in the forestomach. In addition to causing pathological lesions, plastics reduce the capacity of the fore stomachs, which are supposed to be loaded with food. Reduced feed intake results from plastic taking up space in the environment (Bwatota et al., 2018). Gastric problems in ruminants are most commonly caused by rumen tympanic metal or non-metal (mainly plastic) particles (Radostatis et al., 1994). This results in economic losses due to a significant decrease in cattle productivity and an increase in animal mortality (Blood et al., 2007; Muhairwa et al., 1995). Compared to cattle, sheep and goats are far more discriminating feeders due to their generally wide manner of rearing. As a result, they consume far lesser foreign bodies (Hailat et al., 1996). A high death rate and significant productivity loss might result from cattle ingesting foreign bodies (Radostits et al., 2007).

Non-metallic foreign bodies (mostly made of plastic material) have been studied by many researchers, most notably in cattle (Boodura et al., 2010; Khore et al., 2010) and buffaloes (Ramprabu et al., 2003; Boodura et al., 2010; Semieka, 2010; Athar et al., 2010). The grazing grounds available to animals raised in urban and suburban regions are degraded by plastics, ropes, hair, wool, and metals, especially for animals raised in urban and suburban locations. This pollution may probably become a bigger problem for grazing animals because of the ineffective waste management system and the lack of feed during the long dry season (Tiruneh and Yesuwork, 2010).

Animal lives might be in danger if plastics obstruct the aperture between the reticulum and omasum. Animals cannot digest or transfer these plastic wastes through their faeces in their natural state. On haematological and biochemical evaluation, there were no or few abnormalities in the majority of instances of plastic trash impaction (Raoofi et al., 2011). Foetuses and reproductively sound dams are often discarded because of the need to fill the deficit in the protein supply. This has a detrimental impact on animal development capacity (Cadmus and Adesokan, 2009). There are numerous reports of pregnant animals being slaughtered, resulting in foetal wastage in the majority of the world's countries, including Iran (Borji et al., 2011), Zambia (Zulu et al., 2013), Ethiopia (Simenew et al., 2011); for animals in particular, camels (Bello et al., 2008), small ruminants (Muhammad et al., 2007; Bokko, 2011) and cattle (Abdulkadir et al., 2008; Idahor et al., 2009; Alhaji, 2011). This approach has harmed the populace's access to animal protein. The current chapter discusses ruminal impaction in animals caused by plastic materials in detail.

15.2 Causes of plastic ingestion

These plastic materials are being ingested by animals because of their erratic eating habits, confusion about food, and efforts to eat leftover feed products that have been wrapped in plastic (Ngoshe, 2012). A condition known as ruminal impaction develops in animals that have consumed large amounts of indigestible plastic materials over time (Kohli et al., 1998).

15.3 Predisposing factors

The ingestion of plastic garbage by animals is predisposed by several conditions, each of which plays an essential part in this predisposition.

15.3.1 Mode of grazing

The inclusion of plastic components in the food of animals raised in an extended agricultural system increases their risk of developing ruminal impaction (Ngoshe, 2012; Reddy et al., 2004). Because these animals are allowed to graze freely, they are at risk of becoming ill from consuming plastic garbage. No waste is present in the environment where animals are raised in intensive agricultural systems. As a result, these animals are far less likely to acquire plastic foreign body syndrome.

15.3.2 Mineral deficiencies

An animal's appetite will become unpredictable if they are deficient in calcium (Ca) and phosphorus (P). Animals begin to consume inanimate items in order to fulfil their hunger, which results in the development of foreign body syndrome (Ngoshe 2012; Tiruneh and Yesuwork, 2010).

15.3.3 Negative energy balance

Animals in a negative energy balance state are caused by inadequate dietary supplements, increased energy requirements during pregnancy and lactation, and higher metabolic rates. Animals ingest inanimate items to fulfil their energy requirements, resulting in the development of foreign body syndrome (Ngoshe, 2012; Berrie et al., 2015).

15.3.4 Urbanisation

Animal supplies and grazing pastures have been depleted as a result of rapid urbanisation. Furthermore, substantial building activity is done in metropolitan areas and the correct disposal of plastic garbage is not carried out. As a result, grazing animals consume waste that has been placed wherever, including on roadways and near fences, resulting in ruminal impaction due to the presence of plastic compounds in the waste (Tiruneh and Yesuwork, 2010). As a result, animals are more prone to this illness in metropolitan and peri-urban regions than in rural areas (Berrie et al., 2015).

15.3.5 Draught and flood

In drought and flood-prone locations, plastic-related ruminal impaction is prevalent. There will be a lack of fodder in certain locations due to prolonged periods of drought and flooding. There was no previous planning for scenarios that may impair the supply of grass for cattle. Because of this, sheep graze indiscriminately and develop foreign body syndrome (Berrie et al., 2015).

15.4 Types of plastic

There are different types of plastics based on their constituents and the type of materials used in their production. Table 15.1 shows the different types of plastics, their properties and their common uses (Alabi et al., 2019).

Ingestion of polythene bags, plastic covers, and other plastic materials used in food packaging is the most prevalent cause of ruminal impaction caused by plastics (Ngoshe, 2012; Remi-Adewunmi et al., 2004; Ramaswamy and Sharma, 2011).

15.5 Ruminal impaction caused by plastic materials: epidemiology

15.5.1 Prevalence

Plastic-related ruminal impaction has been documented in several nations. A wide range of foreign bodies can be discovered in ruminants with impaction, although plastic bags are the most prevalent. Various slaughterhouse employees worldwide have reported the presence of plastics and other foreign bodies in ruminal impaction. The total frequency of foreign bodies in Gondar Town, North West Ethiopia, was 8.6% (Nugusu et al., 2013). When comparing prevalence across sexes, breeds, and age groups, as well as between different body condition scores and animals from different areas in South West Ethiopia, there was a higher prevalence of foreign bodies in females and crossbred animals older than 10 years and with poor body condition scores (Tesfaye and Chanie, 2012). Foreign body prevalence was 16.3% in small ruminants (11.9% in goats and 20.6% in sheep), and cattle killed at the Bahir-Dar municipal slaughterhouse and hotels in Amhara Region, Ethiopia. According to Roman and Hiwot (2010), there has been a rise in the incidence of bovine tuberculosis in small ruminants (23.4%). According to Remi-Adewunmi et al. (2004), butchered sheep and goats in Nigerian urban areas showed a much greater frequency of foreign bodies (97%). Numerous locations throughout Ethiopia and

TABLE 15.1 Types of plastics, their properties and common uses (Alabi et al., 2019).

S. N.	Types of plastics	Common uses	Properties	Recycled into
1	Polyethylene terephthalates	Soft drinks, water bottles, containers, salad dressing, biscuit trays and salad domes.	Clear, tough, solvent resistant, a barrier to gas and moisture, softens at 80°C.	Pillow and sleeping bag filling, clothing, soft drink bottles, carpeting, building insulation
2	High-density polyethylene (HDPE)	Shopping bags, freezer bags, buckets, shampoo, milk bottles, ice cream containers, juice bottles, chemical and detergent bottles, rigid agricultural pipes and crates.	Hard to semi-flexible, resistant to chemicals and moisture, waxy surface, opaque, softens at 75°C, easily coloured, processed and formed	Recycling bins, compost bins
3	Polyvinyl Chloride (PVC) Plasticised Polyvinyl chloride PVC-P.	Cosmetic container, plumbing pipes and fittings, electrical conduits, blister packs, wall cladding, roof sheeting, bottles, garden hose, shoe soles, cable sheathing, blood bags and tubing.	Strong, tough, softens at 80°C, can be clear, can be solvent welded. Flexible, clear, elastic, can be solvent welded.	Compost bin
4	Low-density polyethylene (LDPE)	Refuse bags, irrigation tubings, mulch film, cling wrap, garbage bags and squeeze bottles.	Soft flexible, waxy surface, translucent, softens at 70°C, scratches easily.	Bin liners, pallet sheets
5	Polypropylene (PP)	Microwave dishes, lunch boxes, packaging tape, garden furniture, kettles, bottles and ice cream tubs, potato chip bags and straws.	Hard and translucent, softens at 140°C, translucent, withstands solvents, versatile.	Pegs, bins, pipes, pallet sheets

(*Continued*)

TABLE 15.1 (Continued)

S. N.	Types of plastics	Common uses	Properties	Recycled into
6	Polystyrene (PS) Expanded polystyrene (PS-E)	CD cases, plastic cutlery, imitation glassware, low cost brittle toys, video cases/foamed polystyrene cups, protective packaging, building and food insulation	clear, glassy rigid, opaque, semi-tough, softens at 95°C, Affected by fat, acids and solvents, but resistant to alkalis, salt solutions, low water absorption, is clear when not pigmented, is odour and taste free. Special types of polystyrene (PS) are available for special applications.	Recycle bin
7	Other	Automotive and appliance components, computers, electronics, cooler bottles, packaging	includes all resins and multimaterials (e.g., laminates) properties dependent on plastic or combination of plastics	Recycle bins

beyond have also reported decreased levels of foreign body pollution (Abebe and Nuru, 2011; Hailat et al., 1997). Foreign bodies may be more abundant in some regions than others due to differences in animal management methods and the quality of foreign body control between rural and urban areas. According to Omidi et al. (2012), plastic items made up the bulk of foreign bodies detected in sheep and goats in Birjand, Iran (27.5% and 24.3%, respectively), and 40% of these animals were pregnant. Drought and a lack of sufficient pastures have been major factors in sheep and goats eating unusual stuff during the last few years.

Plastic-related ruminal impaction has been documented in a number of nations throughout the world. A wide range of foreign bodies can be discovered in ruminants with impaction, although plastic bags are the most prevalent. Slaughterhouse employees across the world have reported a high incidence of plastics and other foreign materials in ruminal impaction. 74% of slaughterhouses in Jordan, 62.5% in Pakistan, 81.6% and 85% in Nigeria, 50% in Ethiopia and 72.3% in Kenya were found to have prevalence rates of more than 80% (Negash et al., 2015). Numerous researchers have reported occurrences of ruminal impaction caused by plastic materials in ruminants in countries such as India, Egypt and Yemen, despite the fact that these nations lack data on the prevalence of ruminal impaction (Vanitha et al., 2010a,b; Reddy and Sasikala, 2012; Abdelaal and EL-Maghawry, 2014).

15.6 Type of damage by plastic ingestion

In and around metropolitan areas, the majority of instances of ruminal impaction caused by plastic materials occur on an irregular basis and cause severe damage.

15.6.1 Economic loss

Ruminal impaction produced by plastic materials costs farmers money through decreased milk production, poor weight gain, lower draft capacity, different comorbid sickness issues and mortality, among other repercussions. Plastic impaction in sheep is estimated to have cost Jordan 15 million dollars in terms of output and general health (Al-Dwery, 1994).

15.6.2 Pathogenesis

When animals are confused about where to get food, they will munch on plastic waste products, such as polybags and plastic covers (Fig. 15.1)

FIGURE 15.1 Grazing of cattle on plastic waste materials. Source*: Modified from https://karunasociety.org/the-plastic-cow-project.*

(Ramaswamy and Sharma, 2011). Because these polymeric components are indigestible, they become stuck on the rumen and then move to the reticulum and omasum (Figs. 15.2 and 15.3) (Ghurashi et al., 2009). Numerous pathological disorders in animals can be produced by eating various types of plastic garbage, depending on the type of plastic waste, the duration of the waste's stay in the stomach, and the foreign body's location in the digestive system. According to Singh (2005), the most prevalent clinical symptoms in animals with ruminal impaction induced by plastic materials include impaction, tympany, polybezoars, traumatic reticulopericarditis, chemical leaching and immunological suppression. Heavy metal toxicity, endocrine disruption, carcinogenicity, teratogenicity and urolithiasis may also occur in ruminants as a result of ruminal impaction with plastic materials. Even still, these circumstances have remained hidden from the public until now. Fig. 15.4 illustrates all of the scenarios that might result in ruminal impaction as a result of plastics.

FIGURE 15.2 Rumen impacted by plastic waste materials. Source*: From* *https://www.researchgate.net*.

FIGURE 15.3 Impacted plastic waste materials in the rumen. Source*: From* https://www.researchgate.net.

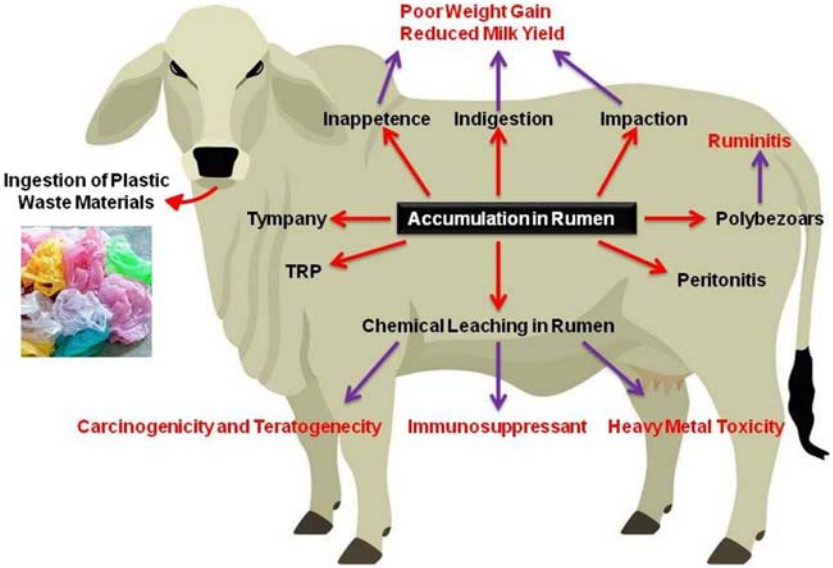


FIGURE 15.4 Pathophysiology of ruminal impaction due to plastic materials in ruminants (Conditions mentioned in yellow words indicate the probable outcome after chronic exposure to the products of chemical leaching in the rumen. However, no studies are conducted in this direction). Source*: Designed by V. Bhateshwar.*

15.6.3 Lack of appetite

Polythene and other plastic materials do not degrade in the rumen/reticulum and remain in their original state of composition. Because of their physical presence in the rumen, the rumen's cranial sac will be strained, causing it to rupture. As a result, the ventromedial hypothalamus and satiety region will be stimulated, resulting in a decrease in appetite (Ghurashi et al., 2009; Reece, 2005).

15.6.4 Simple indigestion

Animals are unable to digest or pass plastic bags and other plastic materials through their faeces in their natural state (Reece, 2005). Their continued presence in the rumen will result in the atrophy of ruminal papillae, which will interfere with the normal digestive and fermentation processes in the animal (Sheferaw et al., 2014). In the rumen, plastics become entangled with one another as a result of the churning action of the contractions. Feed components become caught between the plastic materials in the rumen, making them inaccessible to the rumen protozoa, which are responsible for the digestive and fermentative processes. This has an effect on the function of the rumen bacteria, resulting in indigestion (Singh, 2005).

15.6.5 Ruminal impaction

Polythene bags/plastics in the rumen will initially become entangled with one other due to ruminal contractions, resulting in a hard mass being formed. As time passes, this hard plastic mass blocks the reticulum-omasum opening, slowing or stopping ruminative movements. Ruminal atony and impaction occur as a result of decreased rumen motility caused by this hard plastic mass over time (Singh, 2005; Abdullahi et al., 1984). Symptoms of ruminal impaction produced by plastic foreign bodies are non-existent, and the condition is only discovered when the rumen's digestive tract has been overburdened with plastic (Vanitha et al., 2010a,b).

15.6.6 Ruminal tympany

Polythenes can restrict the reticulum and omasum orifices in the rumen and reticulum, resulting in gas accumulation (Remi-Adewunmi et al., 2004; Igbokwe et al., 2003). When an animal is fed a lot of legumes and other gas-forming foods, the situation just becomes worse. Toxic gases build up in the rumen and can be dangerous if they aren't expelled in the proper manner. Eructation may be hindered by the polybags present in the stomach and rumen. Dyspnoea and mortality are the inevitable consequences of this (Singh, 2005).

15.6.7 Polybezoars

Polybezoars are hard stone-like aggregates that develop when salts collect around polythenes in the digestive system. Not only can polybezoars impair digestion, but they also induce discomfort and irritation in the rumen (Singh, 2005).

15.6.8 Traumatic reticulopericarditis

Nails, wires, and other brittle objects are frequently disposed of in landfills after being bagged in polypropylene. Animals ingest these sharp objects and plastic bags when foraging for food in waste dumps. The honeycomb design causes these sharp objects to become caught in the reticulum and inflict harm. Traumatic reticulopericarditis occurs when the reticulum and diaphragm are perforated and the heart is invaded by these sharp objects over time (Singh, 2005).

15.6.9 Local or diffused peritonitis

Sharp items, such as needles, nails, and wires wrapped in plastic bags or covers, could inadvertently wind up in the stomachs of animals. Over a period of time, this sharp debris is expelled into the rumen generating ruminal wall perforation and the development of local or widespread peritonitis (Tyagi and Singh, 2004; Chanie and Tesfaye, 2012).

15.6.10 Reduction in production

Plastic foreign elements physically present in the rumen and reticulum impair the absorption of volatile fatty acids in the rumen and reticulum. As a result, foreign plastic bodies may cause animals to produce less milk and fatten at a slower rate (Igbokwe et al., 2003; Sheferaw et al., 2014; Tyagi and Singh, 2004).

15.6.11 Heavy metal toxicity

In the production of plastics, metal salts and complexes such as stearates and phthalates and cobalt, lead, mercury, cadmium and chromium are employed. Slowly, the rumen releases these heavy metals into the bloodstream. These hazardous metals have a long half-life because of their sluggish bioaccumulation in vital organs and long-term harmful effects. These heavy metals can also be present in animal products like meat and milk, posing a further threat to human health. The presence of these heavy metals in stray animals grazing on rubbish has been confirmed by several studies, although there is no evidence of their effects on the host yet (Muleke et al., 2013; Yasotha, 2014).

15.6.12 Oestrogenic activity and reproductive problems

Numerous compounds with oestrogenic activity are released by plastic materials, including bisphenol A (BPA), di-(2-ethylhexyl) phthalate (DEHP), and triphenyl phosphate (TPP) (Bittner et al., 2014). More than one oestrogen receptor subtype is frequently harmed by these xenobiotic compounds in animals, which are affected by a wide range of biological and physiological impacts (Gandolfi et al., 2002; Vandenberg et al., 2012). Male and female reproductive systems can be disrupted by a number of physiological causes during the course of a woman's life, resulting in conditions such as cystic ovarian disease, low sperm count, early embryonic death, and even an earlier onset of puberty in some cases (Vandenberg et al., 2012; Oskam et al., 2005; Halden, 2010). The physiological repercussions of wandering ruminants grazing on plastic waste products are mainly unclear. However, the impact of exposure to these chemicals through food and drinking water on reproductive issues in high-yielding dairy calves cannot be ignored since oestrogenic hormones and phthalates are present at such high levels in grazing ruminants (Brevini et al., 2005).

15.7 Clinical signs

Depending on the amount and length of time foreign plastics are ingested by the animals, they display different clinical signs (Reddy and Sasikala, 2012; Singh, 2005). If an animal has ingested a small amount of foreign plastic, it may not exhibit any symptoms for months before showing indications of sickness. Clinical symptoms such as diarrhoea and vomiting might be seen when plastics are swallowed and disrupt the normal functioning of the rumen (Ngoshe, 2012; Ramaswamy and Sharma, 2011). It has been shown that animals suffering from this ailment often display signs and symptoms, such as sadness, partial or complete anorexia, frequent abdominal distention, decreased milk production, weight loss and increased susceptibility to other illnesses (Dodia et al., 2014; Vanitha et al., 2010a,b). An animal's condition might lead to its death in road accidents or slaughter due to ruminal impaction, which is a disease that affects stray animals. As a result, additional signs, such as impaired fertility and malignancies, go unnoticed (Boerjan et al., 2002).

15.8 Detection and treatment

It is difficult to diagnose ruminal impaction produced by plastic materials since the clinical symptoms in animals suffering from it are general. The animals' frequent bloats and persistent ruminal impactions may be caused by ruminal impaction produced by plastic materials in their digestive systems (Dodia et al., 2014; Vanitha et al., 2010a,b). More research is needed before these compounds may be used for diagnostic purposes in ruminal fluid and milk samples, despite a few studies having done this (Vanitha et al., 2010a,b). Plastic foreign body syndrome cannot be effectively treated with traditional methods such as anti-bloat medications or purgatives. As a result, veterinarians and other healthcare professionals still rely on rumenotomy to diagnose and treat ruminal impaction caused by animals' use of plastic items (Tyagi and Singh, 2004).

15.9 Control and preventive measures

Ruminal impaction induced by plastic debris is a result of improper waste management and animal husbandry practices (Singh, 2005; Otsyina et al., 2017). Appropriate waste disposal and husbandry practices may thus be required to keep environmental pollution under control and to prevent animals from acquiring indigestible foreign bodies (Ngoshe, 2012; Vanitha et al., 2010a,b). Proper animal husbandry procedures include providing adequate feed, water, shelter, and mineral supplements on a timely manner. The development of fodder banks, grazing centres, and water facilities can help mitigate the dry season's detrimental effects by preventing stray animals from congregating on roadsides and in the trash in search of food and water during this time.

To correctly manage plastic waste, it is vital to adhere to the three R's (Reduce, Reuse, Recycle). For impoverished nations, the most practical options are to reduce plastic waste production and repurpose plastic waste that has already been generated (Ramaswamy and Sharma, 2011). It is feasible to minimise plastic use by replacing alternate polybag materials, such as jute, cotton, or paper bags. Municipalities and other sanitation agencies should take an active part in collecting plastic debris along roadsides and in open spaces and properly disposing of it to avoid animal consumption. The public, particularly livestock owners, should be informed about the negative impacts of plastic on animal health and hence, on human health. They should be educated on basic animal husbandry techniques, as well as safe plastic waste disposal (Abu-Seida and Al-Abbadi, 2016).

15.10 Conclusion

Plastics have invaded every facet of modern life. Plastic materials are used in a broad number of applications and are widespread due to their low cost, ease of availability, and a multitude of other advantages. It is not the use of plastics in and of itself that is causing harm to humans, animals, and environmental health, but rather the improper disposal of plastic waste. Ruminal impaction induced by plastic products is one of the most important, though generally disregarded, health concerns impacting stray ruminants in impoverished nations' metropolitan areas. The sole source of knowledge on this subject is case reports and slaughterhouse incidents. Much more basic study is required to better understand the pathophysiology, diagnosis, and treatment of ruminal impaction in animals induced by plastic materials. Only a few studies have been undertaken to determine the presence of plastic residues in the milk and meat products of animals that have had ruminal impaction as a result of the usage of plastic materials. It is vital to perform an extensive study on the rumen's chemical leaching processes. Along with improving basic animal husbandry practices and proper waste disposal, government intervention, particularly in the form of slaughter policies for unproductive animals, the establishment of fodder banks/grazing centres/watering facilities for animals, the implementation of public health awareness programs, the implementation of strict laws requiring proper waste disposal, and other measures will aid in the control of ruminal impaction caused by plastic materials in animals.

Competing interests

The authors declare that they have no competing interests.

References

Abdelaal, A.M., EL-Maghawry, S., 2014. Selected studies on foreign body impaction in goats with special reference to ultrasonography. Vet. World 7, 522–527.

Abdulkadir, U., Jiya, E.Z., Kosu, S.A., 2008. Survey of foetal wastages: a case study of Makurdi Abattoir in benue state from 1997 to 2002. Pak. J. Nutr. 7 (3), 450–452.

Abdullahi, U., Usman, S., Mshelia, T., 1984. Impaction of rumen with indigestible garbage in cattle and sheep reared within urban and suburban environment. Nigerian Vet. J. 13, 89–95.

Abebe, F., Nuru, M., 2011. Prevalence of indigestible foreign body ingestion in small ruminants slaughtered at Luna export abattoir, East Shoa, Ethiopia. J. Anim. Vet. Adv. 10 (12), 1598–1602.

Abu-Seida, A., Al-Abbadi, O., 2016. Recent advances in the management of foreign body syndrome in cattle and buffaloes: a review. Pak. Vet. J. 36, 385–393.

Abu-Seida, A.M., Al-Abbadi, O.S., 2014. Recurrent rumen tympany caused by Trichobezoars in buffaloes. A Ser. Rep. 44, 147–151.

Al-Dwery, M., 1994. Agricultural conference of the national center for research and development. Jordan, Personal. Commun.

Alabi, O.A., Ologbonjaye, K.I., Awosolu, O., Alalade, O.E., 2019. Public and environmental health effects of plastic wastes disposal: a review. J. Toxicol. Risk Assess. 5 (021), 1–13.

Alhaji, N.B., 2011. Prevalence and economic implications of calf foetal wastage in an abattoir in Northcentral Nigeria. Trop. Anim. Health Prod. 43, 587–590.

Atawalna, J., Kotoku, V., Ewura, S., 2015. Prevalence of indigestible foreign materials in small ruminants slaughtered at the Kumasi Abattoir of Ghana. Int. J. Livest. Res. 5, 1–7.

Athar, H., Mohindra, J., SIingh, Singh, T.K., 2010. Indian. Polyvet 119 (2), 180–183.

Bello, M.B., Garba, H.S., Sonfada, M.L., 2008. Foetal wastage in camels slaughtered at Sokoto municipal abattoir. Sokoto J. Vet. Sci. 7 (1), 46–49.

Berrie, K., Tadesse, E., Mossie, B., 2015. Study on rumen and reticulum foreign body in slaughtered cattle at Gondar Elfora Abattoir. World J. Biol. Med. Sci. 2, 133–150.

Bittner, G.D., Denison, M.S., Yang, C.Z., Stoner, M.A., He, G., 2014. Chemicals having estrogenic activity can be released from some bisphenol a-free, hard and clear, thermoplastic resins. Environ. Health 13, 1–18.

Blood, D.C., Radostits, O.M., Gay, C.C., Hinchclif, K.W., Constable, P.D., 2007. Veterinary Medicine E-Book: A Text Book of Diseases of Cattle, Horses, Sheep, Pigs, and Goats, Saunders, Philadelphia, Pennsylvania, (10th edi.). Elsevier Health Sciences.

Boerjan, M.L., Freijnagel, S., Rhind, S.M., Meijer, G.A.L., 2002. The potential reproductive effects of exposure of domestic ruminants to endocrine disrupting compounds. Anim. Sci. 1, 3–12.

Bokko, P.B., 2011. Pregnancy wastage in sheep and goat in the Sahel region of Nigeria. Nigerian Vet. J. 32, 120–126.

Boodura, P., Sivaprakash, B.V., Kasaralivar, V.R., Dilip, D., 2010. Indian. Polyvet 11 (2), 184–188.

Borji, H., Azizzadeha, M., Kamelli, M., 2011. A retrospective study of abattoir condemnation due to parasitic infections: economic importance in Ahwaz, Southwestern. Iran. J. Parasitol. 98 (5), 954–957.

Brevini, T.A., Cillo, F., Antonini, S., Gandolfi, F., 2005. Effects of endocrine disrupters on the oocytes and embryos of farm animals. Reprod. Domestic Anim. 40 (4), 291–299.

Bwatota, S.F., Makungu, M., Nonga, H., 2018. Occurrences of indigestible foreign bodies in cattle slaughtered at Morogoro municipal slaughterhouse, Tanzania. J. Vet. Med. 1–6.

Cadmus, S.I.B., Adesokan, H.K., 2009. Bovine fetal wastage in southwestern Nigeria: a survey of some abattoirs. Trop. Anim. Health Prod. 42, 617–621.

Chanie, M., Tesfaye, D., 2012. Clinicopathological findings of metallic and non-metallic foreign bodies in dairy cattle: a review. Acad. J. Anim. Dis. 3, 13–20.

Dodia, V.D., Kelawala, N.H., Suthar, D.N., Prajwalita, S., 2014. Haematological and serum biochemical profile of cattle affected with plastic foreign bodies. Int. J. Sci. Res. Publ. 4, 1–2.

Gandolfi, F., Pocar, P., Brevini, T.A.L., Fischer, B., 2002. Impact of endocrine disrupters on ovarian function and embryonic development. Domestic Anim. Endocrinol. 23, 189–201.

Ghurashi, M.A., Seri, H.I., Bakheit, A.H., Ashwag, E.A., 2009. Effect of surgical removal of foreign body from goat's rumen with special reference to prevalence of foreign body in goats in Southern Darfur. Aust. J. Basic. Appl. Sci. 3 (2), 664–668.

Hailat, N., Nouh, S., Al-Darraji, A., Lafi, S., Al-Ani, F., Al-Majali, A., 1997. Prevalence and pathology of foreign bodies (plastics) in Awassi sheep in Jordan. Small Rumin. Res. 24, 43–48.

Hailat, N., South, S., Al-Darraji, A., Al-Majali, A., 1996. Prevalence and pathology of foreign bodies (plastics) in Awassi sheep in Jordan. Small Rumin. Res. 24, 43–48.

Halden, R.U., 2010. Plastics and health risks. Annu. Rev. Public. Health 31, 179–194.

https://karunasociety.org/the-plastic-cow-project.

https://www.researchgate.net.

Idahor, K.O., Omeje, J.N., Aya, V.E., Audi, P., Daniel, S.R., Luka, B.D., 2009. Awareness of foetal losses from ruminants slaughtered at Lafia Abattoir. Acta SATECH 3, 42–46.

Igbokwe, I.O., Rolo, M.Y., Egwu, G.O., 2003. Rumen impaction in sheep with Indigestible foreign bodies in the semi-arid of Nigeria. Small Rumin. Res. 49, 141–146.

Joel, F.R., 1995. Polymer Science & Technology: Introduction to Polymer Science, third ed. Prentice Hall PTR Inc, Upper Saddle River, NJ, pp. 4–9.

Khore, K.A., Jodhar, P.A., Mahjan, V.E., 2010. Intas polyvet 11 (2), 189–190.

Kohli, M.R., Nadaff, H., Ghadrdan, A., 1998. Bovine indigestion due to chronic Ruminal engorgement associated with ingestion of plastic material: a retrospective study of 54 cases. Indian J. Vet. Surg. 19 (2), 105–106.

Muhairwa, A.P., Maselle, R.M., Fyumagwe, R.D., Masambu, J.G., Mpelumbe, I.S., 1995. Sharp foreign bodies in the forestomaches of cattle. Diagnosis and treatment of ten cases in the city of Dares Salaam, Tanzania. Tanzan Vet. J. 15 (3–4), 138–143.

Muhammad, I.R., Ashiru, R., Abdullahi, A.Y., 2007. Implications of the slaughter of Pregnant Ewes and Does to future stock in the Semi-arid Urban Abattoir. J. Anim. Vet. Adv. 6, 819–822.

Muleke, I.C., Maina, J.N., Osuga, M.I., Mutai, J.K.O., Karubiu, N., Bebe, B., 2013. Lead and copper levels in the soil, water, serum and tissues of livestock feeding on dumpsite waste in urban slums of industrial towns in western Kenya. Egerton J. Sci. Technol. 13, 11–20.

Mushonga, Habarugira, G., Musabyemungu, A., Udahemuka, J.C., Jaja, F.I., Pepe, D., 2015. Investigations of foreign bodies in the fore-stomach of cattle at ngoma slaughterhouse, Rwanda. J. South Afr. Vet. Assoc. 86 (1), 1233.

Negash, S., Sibhat, B., Sheferaw, D., 2015. A postmortem study on indigestible foreign bodies in the rumen and reticulum of ruminants, eastern Ethiopia. Onderstepoort Onderstepoort J. Vet. Res. 82, 881–886.

Ngoshe, A.A., 2012. Incidence of polythene bag ingestion by ruminant animals at Maiduguri central abattoir. Ramat J. Manage. Sci. Technol. 1, 12–16.

Nongcula, V.V., Zhou, L., Nhundu, K., Jaja, I.F., 2017. Association between the prevalence of indigestible foreign objects in the gastrointestinal tract of slaughtered cattle and body condition score. Animals 7 (11), 80.

Nugusu, S., Velappagounder, R., Unakal, C., Nagappan, R., 2013. Studies on foreign body ingestion and their related complications in ruminants associated with inappropriate solid waste disposal in Gondar Town, North West Ethiopia. Int. J. Anim. Vet. Adv. 5 (2), 67–74.

Omidi, A., Naeemipoor, H., Hosseini, M., 2012. Plastic debris in the digestive tract of sheep and goats: an increasing environmental contamination in Birjand, Iran. Bull. Environ. Contam. Toxicol. 88 (5), 691–694.

Oskam, I.C., Lyche, J.L., Krogenæs, A., Thomassen, R., Skaare, J.U., Wiger, R., 2005. Effects of long-term maternal exposure to low doses of PCB126 and PCB153 on the reproductive system and related hormones of young male goats. Reproduction 130, 731–742.

Otsyina, H.R., Mbuthia, P.G., Nguhiu-Mwangi, J., Mogoa, E.G.M., Ogara, W.O., 2017. Gross and histopathologic findings in sheep with plastic bags in the rumen. Int. J. Vet. Sci. Med. 5 (2), 152–158.

Pitroda, A.H., Tiwar, D.K., Dar, M., Patil, D.B., Parikh, P.V., 2010. Ultrasonographic diagnosis and treatment of rumen impaction in a goat. Intas Polivet 11 (2), 251–252.

Radostatis, O.M., Blail, D.C., Gug, G.C., 1994. Veterinary Medicine, eighth ed. *Baillierv Tindall*, *London*, pp. 270–278.

Radostits, O.M., Blood, C.C., Hinchcliff, K.W., Constable, P.D., 2007. Veterinary Medicine: A Text Book of Disease of Cattle, Horse, Sheep, Pig and Goat, tenth ed. *Saunders Elsevier*, England, London, pp. 112–113.

Ramaswamy, V., Sharma, H.R., 2011. Plastic bags–threat to environment and cattle health: a retrospective study from Gondar City of Ethiopia. IIOAB J. 1, 7–12.

Ramprabu, R., Dhangralan, P., Prabhuban, S., 2003. Irrrak. Vet. Rech. Alloc. 58, 2–3.

Raoofi, A., Namjoo, A., Karimi, A.H., Esfahani, M.A., 2011. A study of clinical signs, hematological changes and pathological findings of experimental ingestion of soft foreign body (plastic rope) in goats. Small Rumin. Res. 105 (1–3), 351–354.

Reddy, Y.R., Rao, S.T.V., Naidu, P.T., 2004. Foreign bodies in rumen and reticulum of Punganur cattle. Indian Vet. J. 81, 1063–1065.

Reddy, M.V.B., Sasikala, P., 2012. A review on foreign bodies with special reference to plastic pollution threat to live stock and environment in Tirupati rural areas. Int. J. Sci. Res. Publ. 2 (12), 1–8.

Reece, O.W., 2005. Functional Anatomy and Physiology of Domestic Animals. *Ames, Iowa University Press*, *Ithaca, USA*, pp. 357–358.

Remi-Adewunmi, B.D., Gyang, E.O., Osinowo, A.O., 2004. Abattoir survey of foreign body rumen impaction in small ruminants. Nigerian Vet. J. 25, 32–38.

Roman, T., Hiwot, Y., 2010. Occurrence of rumen foreign bodies in sheep and goats slaughtered at the Addis Ababa Municipality Abattoir. Ethiopian Vet. J. 14 (1), 91–100.

Semieka, M.A., 2010. Radiography of unusual foreign body in ruminants. Vet. World 3 (10), 473–475.

Sheferaw, D., Gebru, F., Asrat, M., 2014. Ingestion of indigestible foreign materials by free grazing ruminants in Amhara Region, Ethiopia. Trop. Anim. Health Prod. 46 (1), 247–250.

Simenew, K., Bekana, M., Fikre, L., Tilahun, Z., Wondu, M., 2011. Major gross reproductive tract abnormalities in female cattle slaughtered at Sululta Slaughterhouse in Ethiopia. Glob. Vet. 6 (6), 506–513.

Singh, B., 2005. Harmful effect of plastic in animals. Indian. Cow. 4, 10–18.

Tesfaye, D., Chanie, M., 2012. Study on rumen and reticulum foreign bodies in cattle slaughtered at Jimma Municipal Abattoir, South. West. Ethiopia. Am.-Eurasian J. Sci. Res. 7 (4), 160–167.

Tiruneh, R., Yesuwork, H., 2010. Occurrence of rumen foreign bodies in sheep and goats slaughtered at the Addis Ababa Municipality Abattoir. Ethiopian Vet. J. 14 (1), 91–100.

Tyagi, R.P.S., Singh, J., 2004. Ruminant Surgery. *CBS Publishers and Distributors*, *New Delhi*, pp. 198–204.

Vandenberg, L.N., Colborn, T., Hayes, T.B., Heindel, J.J., Jacobs, D.R., Lee, D.H., et al., 2012. Hormones and endocrine-disrupting chemicals: low dose effects and nonmonotonic dose responses. Endocr. Rev. 33, 378–455.

Vanitha, V., Chandra, G.S., Nambi, A.P., 2010a. Polychlorinated biphenyls in milk and rumen liquor of stray cattle in Chennai. Tamil Nadu J. Vet. Anim. Sci. 6, 71–74.

Vanitha, V., Nambi, A.P., Gowri, B., Kavitha, S., 2010b. Rumen impaction in cattle with indigestible foreign bodies in Chennai. Tamil Nadu J. Vet. Anim. Sci. 6 (3), 138–140.

Yasotha, A., 2014. Bio-monitoring of heavy metals in cattle reared around industrial areas of Tamil Nadu. Ph.D. thesis submitted to Tamil Nadu Veterinary and Animal Sciences University, Chennai.

Zheng, Y., Yanful, E.K., Bassi, A.S., 2005. A review of plastic waste biodegradation. Crit. Rev. Biotechnol. 25 (4), 243–250.

Zulu, V.C., Mwanza, A.M., Banda, F.C., Yasuda, J., Oshida, M.Y., 2013. Cattle reproductive wastage in Zambia: a case of Mongu abattoir. Bull. Faculty Agric. Kagoshima. 63, 49–54.

Chapter 16

Temple floral waste for various bio-products in India

Sachi Gupta[1], **Shri Krishna Tewari**[1] **and Sanjay Pathak**[2]
[1]Botanic Garden, PCAT, CSIR-National Botanical Research Institute, Lucknow, Uttar Pradesh, India, [2]College of Horticulture and Forestry, Acharya Narendra Deva University of Agriculture and Technology, Kumarganj, Uttar Pradesh, India

16.1 Introduction

In the entire world, India is one of the most well-known countries for its spirituality. Here, where worship is one of the ways of life, several religions are fused. People make offerings to the gods, which often consist of flowers, leaves, fruits, etc. Large quantities of flowers are offered in Indian temples, resulting in a lot of floral waste that pollutes the environment and raises health hazards. Indian cities and towns are suffering from the environmental consequences of solid waste management, just like many other developing nations (Kiran and Kavitha, 2020). The garbage from these temples can be totally biodegraded. These wastes' organic makeup makes it easier to handle them through processes like vermicomposting, turning them into useful goods, extracting the oil, creating dyes, etc., instead of uncontrolled and open dumping (Yadav et al., 2015).

Every year, about tonnes of floral waste are produced, clogging rivers and other waterways and serving as breeding sites for numerous diseases. Many of us avoid disposing of flowers and other materials used for prayers in the waste because of our religious convictions; instead, we place them in plastic bags and dump them into bodies of water. In addition to this, flowers are also preserved in sacred trees, which is nearly an improper method of disposal. For example, Banaras, one of the most sacred places in the nation, lacks a policy for disposing of the tonnes of waste that are generated by its numerous temples. Waste materials weighing 3.5−4 tonnes are abandoned each day just behind the temples (Mishra, 2013).

Compared to the degradation of kitchen waste, the degradation of floral waste is a somewhat slower process (Jadhav et al., 2013). Therefore, a proper and environmentally friendly method of treating flower waste is required.

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00008-6

In several research, the management and usage of floral waste have been done. The Kashi Vishwanath temple is one such instance, which attracts the greatest number of devotees throughout the year, particularly in the month of Shravan. It has a system in place to dispose of the hundreds of kilogrammes of waste generated by the devotee offerings. The floral waste generated from the temple is turned into manure (Mishra, 2013).

For example, burning incense sticks produce smoke (fumes) that contain particulate matter (PM), gas products, and numerous organic compounds. As a result, flower petals collected from temples can be used to construct herbal incense sticks, avoiding the negative impacts brought on by their disposal (Waghmode et al., 2016). Incense sticks are made from flowers like marigolds, hibiscus, and others, while roses are used to make rose water. According to a 2013 article in The Hindu, roses are used to manufacture rose water, while flowers like gainda (Tagetes spp.) are used to make incense sticks (Mishra, 2013). In addition to rosewater and incense sticks, the blooms can be used in herbal items like herbal colours, natural dyes, etc. (Ravishankar et al., 2014). This flower waste can be used again for a variety of things. Each day, these temples, dargahs, gurdwaras, etc., released 15–18 quintals of flower debris. The majority of temple garbage is made up of organic materials like flowers, leaves, coconut shells, fruits, etc., that eventually make their way into trash cans or certain water bodies and cause pollution and hygiene issues.

There are many uses of temple floral waste, summarised as under (Table 16.1):

1. Different colourful dyes are extracted from these flowers, which are used in
   - Food Industry for colouring vegetables, eggs, etc.
   - Textile industry for dyeing of materials
   - Candle industry to provide colour to candles and also in wax-based artefacts.

TABLE 16.1 Summary of various bio-products, developed from temple floral waste in India.

Sl./ no.	Technologies available	Product/process	Technology provider
1.	Technology for usage of waste	flower-based Incense sticks	CSIR – CIMAP, Lucknow Website: http://www.cimap.res.in
2.	Dehydration of flowers & foliage technologies	Artefacts like greeting cards, wall plates, landscapes, three-dimensional interior decorative items, etc.	CSIR – NBRI, Lucknow Website: http://www.nbri.res.in

(*Continued*)

TABLE 16.1 (Continued)

Sl./ no.	Technologies available	Product/process	Technology provider
3.	Natural dye	Eco-friendly dyeing and antibacterial finishing of soya bean protein fabric using waste flowers from temples	Department of Fibres and Textile Processing Technology, Institute of Chemical Technology, Matunga (E), Mumbai-400019, India mdt9pub@gmail.com; javed_uict@yahoo.co.in
4.	Production of vermicompost from temple floral waste	Vermicomposting of temple waste is an excellent and eco-friendly method of temple waste management.	Department of Microbiology, K. W. College, Sangli, Maharashtra and Department of Biotechnology, Fergusson College, Maharashtra
5.	Usage of temple waste flower for dyeing	Dyeing of cotton, wool and silk at an industrial scale	IIT, Kanpur

2. The use of biological waste is a solution that recovers valuable fertilizer components.
3. Its extracts are also used for the preparation of herbal *gulal*.
4. Some flowers can also be used as veterinary feed; marigold is used in poultry feed as a food additive because it is a rich source of carotenoids.
5. Various flowers, such as *hibiscus* and marigold, are also used for different medicinal purposes.
6. Extraction of essence from flowers like tuberose, jasmine and rose.

16.2 Utilization of temple floral waste

16.2.1 As biofertilizers or vermicompost

Numerous scientists have put in a tremendous amount of effort to properly use flower waste material in compost. Composting is a management strategy that has proven to be quite successful, in which they used the biogas digester's effluent and mixed it well with the temple's floral waste and cow dung before allowing it to decompose for 60 days at 40°C.

This led to outstanding growth characteristics being attained in terms of height, flowering period, and quantity of flowers produced as compared to the control sets when the prepared vermicompost was used for pot culture as a fertilizer in several flowering plants (which were not treated with vermicompost).

To handle floral waste, Shouche et al. (2011) used a variety of techniques, including composting and vermicomposting. Vermicompost was created using various proportions of dung and floral waste combinations. Numerous parameters, including temperature, pH and moisture content, first indicated minor periodic variations and were later shown to be stable.

- Development of microbial consortia for efficient floral waste from temple degradation. They extracted bacterial cultures from soil samples they had taken from the vicinity in and around the temples. The obtained flower waste was dried, combined with agar media, and then streaked with selected soil samples for isolation. The digestion of the waste was found to be improved by the microbial consortium, and the bio-manure consortium was found to have good quality without creating any environmental risks.
- It has been discovered that vermicompost made from various floral waste has a high nutrient quality and a massive number of microorganisms. The growth rate of the plants grown from these vermicomposts was found to be more effective than that of the control. Auxin, gibberellins, and other plant hormones and enzymes found in vermicompost are thought to stimulate plant development while discouraging plant diseases. Thus, these vermicompost produced higher plant yields as a result.
- Management of flower waste was done by using vermicomposting technology, which was then compared with kitchen waste and farm yard waste vermicompost. The physicochemical analysis of the flower waste vermicompost showed better results in comparison to both the other waste composts. Along with it, plant growth parameters were also studied for the above-mentioned vermicompost, which revealed that temple waste vermicompost should enhance the growth parameters of plants.
- The vermicompost made from floral waste is shown to perform very well when used as a substrate for vermicomposting when various parameters, including pH, temperature, moisture content, organic carbon, available phosphorus, etc., were analysed.
- Thoothukudi (in Tamil Nadu) is home to a large number of dry flower processing and exporting industries that work with flowers. The organic waste generated by these companies primarily consists of floral waste. Siluvai and Aneeshia (2014) worked on producing quality compost from this waste. They used the fungi *Ganoderma incidum, Pleurotus sapidus* and *Pleurot flabellatus* for their fungal cultures. The decomposition of waste and production of compost at a reasonable cost was found to be made possible by the *Pleurotus* species.
- Analysis was done on parameters such as temperature, pH, electrical conductivity, moisture content and solid sample. They determined the highest temperature on the fourth day of heap composting and demonstrated the effectiveness of composting as a ‘zero-waste’ method for handling organic waste like flowers.

16.2.1.1 Preparation of vermicompost through floral waste

16.2.1.1.1 Collection, segregation and shredding of waste (flowers)

The non-biodegradable portion of the floral waste collected from temples was manually sorted out, and the biodegradable waste, including garlands and flowers, was divided and shredded into small pieces.

16.2.1.2 Air drying/pre-composting

Spreading the separated floral waste over paper for 48 hours allowed it to air dry. After being air dried, the samples were pre-composted for 10 days to prepare them for the vermicomposting process, which benefits from its thermophilic nature to reduce weight and pathogens (Nair et al., 2006).

16.2.1.3 Selection of earthworm species

Eisenia foetida species of earthworm are frequently used for composting. It can survive in organic wastes with a variety of moisture contents and has good temperature tolerance.

16.2.1.4 Preparation of HDPE vermi beds

Portable HDPE Vermi Bed of dimensions 4 × 10 ft, was used which had an aluminium stand and green shade. It is a unique and latest technology concept for earthworm farming or vermiculture. It is lightweight, waterproof and U.V. Stabilised. Flexible, mobile, economical and easy to handle and install.

Portable HDPE Vermi Beds are unique with the latest technology concept for Earthworm Farming. Some of its features are as follows:

- Highly portable.
- Lightweight.
- Low-cost benefit.
- High mobility.
- Easy installation.
- Durable, long lasting, waterproof.
- Available in size 3600 × 1200 × 600 mm ($L \times W \times H$).
- U.V. stabilised.
- Flexible and economical.
- Easy to handle and install.

16.2.1.5 Selection of area

The area selected should have dimensions 6 × 15 ft, and must be made even and elevated, with a slope of 4″ to 6″ towards the drain so as to allow the liquid produced by earthworm to accumulate in the pit. Sharp stones should also be removed to prevent the bed from being damaged.

16.2.1.6 *Methodology*

- Flower waste and cow dung are mixed in equal quantities (5 kg each).
- Alternate two layers of 6″ each of farm waste, floral waste, cow dung are placed one over the other.
- About 200 earthworms will be introduced into partially digested material kept in a vermi bed and the moisture level is maintained at 60%.
- Watering is done untill the content heat is satisfactorily removed, which is done for around 2 days.
- After the 45th day, when the composting process was over, the worm's weight increased by 85%. The amount of finished vermicompost obtained is 2.57 kg.
- Verm wash was produced and collected in the container.
- Verm wash diluted with water @ 3% or 5% is sprayed on plants as anti fungous nutritious effective.

The processes of vermicomposting were carried out for a period of 45–60 days. The temperature and moisture content was maintained by sprinkling an adequate quantity of water at frequent intervals.

16.2.2 Essential oil extraction

The flowers offered to deities are available as temple flower waste, and it was discovered that roses flower offerings were up to 50% of this material which makes them suitable for essential oil extraction (Voon et al., 2011). Rose oil has almost 300 different compounds. The main flowers offered at Indian temples include rose, jasmine, marigolds, chrysanthemum, hyacinth, hibiscus and tuberose. When these flowers are thrown carelessly, then their disposal becomes problematic. The use of floral waste material by extracting dyes and essential oils from them is through proper usage of these floral offerings and decomposition (Perumal et al., 2012).

16.2.3 Extraction of dyes

Coloured pigments present in flowers give them a characteristic colour which attracts the eyes of the viewer. The pigments from coloured flowers are extracted to be further used for a variety of purposes (Vankar et al., 2009).

D. Jothi, an Ethiopian scientist, tried to extract natural dyes from African marigold flowers for textile colouring (2008). According to him, there are four steps involved in dying cotton and silk with marigold flowers: pre-treatment, extraction of the dyes from the flower, mordanting (fixing the dye with the fibre), and dying.

For extracting colour from the flower, the plant source was crushed, then dissolved in distilled water, and afterwards boiled for two hours. With the material-to-liquor ratio at 1:40, silk and cotton fabrics were washed in a solution comprising 0.5 g/L sodium carbonate and 2 g/L non-ionic detergent solution at 50°C for 25 minutes.

The used material was properly cleaned with tap water and allowed to air dry. Prior to mordanting, the scrubbed material was immersed in clean water for 30 minutes. The sample was moistened before being placed in a mordant solution heated to 80°C for 30 minutes. The mordanted material was then rinsed with water, pressed and dried.

The cotton and silk samples were dyed using dye extract at a 1:4–0 (material: liquor) ratio. The material was dyed and allowed to dry at room temperature; after that, it was washed with cold water and then soaked in a brine solution for dye fixing.

- The novel extraction method produced dyes with a higher yield and richer colour. Took 100 g. Dry, crushed flower and dispersed in ethanol (500 mL and to 50°C for 1.5 hours).
- After removal, ordinary filter paper were used to filter the extract. The filtrate was then collected, and the solvent was evaporated using a rotatory evaporator to recover its dryness. The cotton fibre was scoured for 4 hours in a solution containing 5 g/L of sodium carbonate and 3 g/L of non-ionic detergent before being rinsed and air dried at room temperature and then soaked for 30 minutes in clean water before dyeing.
- After scouring, cotton fibre was treated with a 2% tannic acid solution in a dipping condition for at least 4–5 hours, squeezed, dried and then treated with a 2% alum and 1% stannous chloride solution in a water bath at 400°C for an hour. The dyeing procedure took place for two hours at a temperature of 30 to 40°C. After that, the material was dipped in a saturated brine solution for 15 minutes, which serves as a dye fixer and then rinsed off thoroughly with running tap water and dried in the open.

Benefits of floral dye:

1. The flower dye is more readily available, less expensive, and neither poisonous nor allergic to human health.
2. These natural dyes were non-carcinogenic, and more significantly, environmentally beneficial.
3. India offers a huge amount of flowers and various breeds.

Drawbacks of floral dye:

1. If natural dyes are to be made commercially available, they must meet the same stringent specifications as synthetic dyes.
2. Fabrics dyed with natural dye fade more quickly than fabric dyed with synthetic dyes, and they also slightly fade when exposed to sunlight for longer than two hours.
3. Compound shades and multicolour fabrics production are not up to the mark with floral dye.

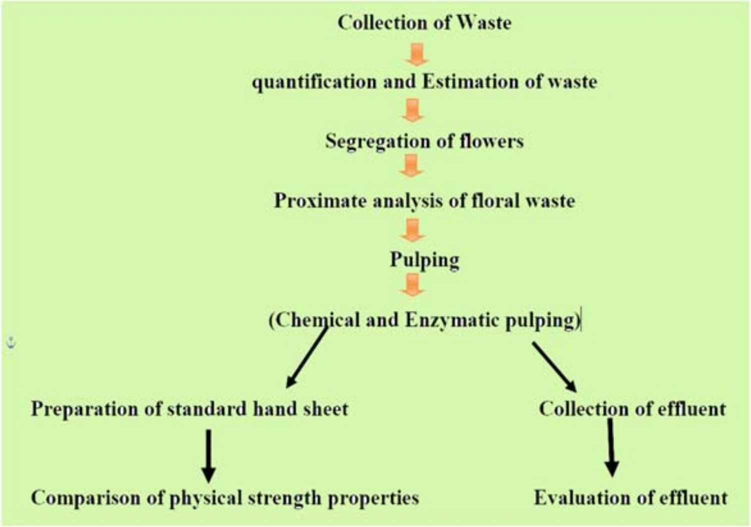
16.2.4 Medicinal uses

Some flowers from the temple flower trash, like marigolds and hibiscus Rosa sinensis, have medicinal qualities and can be used for that purpose; they are typically consumed as decoctions (Voon et al., 2011).

16.2.5 Conversion of floral waste into handmade paper

Prior to the invention of machinery that could mass-produce paper during the Industrial Revolution of the 18th century, paper was created by hand for thousands of years. An industry depending on forests is the paper industry. A significant source of concern is the decline in forest land. Handmade paper is a sustainable product for the environment because it is an environmentally friendly and clean product (Yeboah, 2011). Dhaked et al. (2003) described handmade paper as ‘A sheet of paper or board manufactured by hand’. 13,000 tonnes of handmade paper and board were produced in 1995, valued at Rs 250 million, which is 0.4% of the total paper production in India.

The manufacturing of pulp from non-wood resources has received a lot of interest recently due to the rise in demand for paper and board. In most cases, the raw materials used to make handmade paper include leftovers from the textile industry, various locally accessible bast fibres, and recycled secondary fibres. The majority of bast, leaf, cotton and linter fibres come from annual plants and have short renewable cycles. Utilizing non-polluting chemicals like lime, soda ash, caustic soda, oxalates, oxygen and peroxides during the pulping process minimises any environmental destruction. For the delignification operations, it is not necessary to use harmful toxins chemicals like alkali sulphide and sulphite, chlorine and chlorine compounds.



The study involves the following steps:

16.2.5.1 *Collection of waste*

The amount of waste generated varies from temple to temple depending on the special days of the deities, so waste from the chosen temples will be collected once a week or on certain days.

16.2.5.2 *Characterisation of waste*

The whole garbage will be divided into biodegradable and non-biodegradable waste and later further divided into degradable materials like flowers, cotton, food, incense sticks, etc.

16.2.5.3 *Segregation of flowers*

The non-biodegradable parts will be removed, and flowers will be segregated from the biodegradable waste.

16.2.5.4 *Proximate analysis of floral waste*

The floral waste will be then analysed for the parameters such as ash, cellulose content, lignin content, Cold and hot water solubility and alcohol benzene solubility. Analysis of these parameters will be done using standard TAPPI test methods.

16.2.5.5 Pulping

On the basis of pulping for hand sheet preparation, two experimental groups will be designed.

Group I: Chemical Pulping - Open hot digestion of the flower waste will be done with NaOH.

Group II: Enzymatic Pulping - The waste will be treated with different enzymes for the preparation of pulp.

Optimisation of the pulping process will be done in terms of enzyme and chemical doses and treatment and duration of the treatment/cooking time. Each group will be further subdivided into three experimental sets on the basis of doses and treatment conditions.

16.2.5.6 Preparation of sample hand sheet

Hand sheets will be prepared using the extracted pulp (Both enzymatically and chemically).

16.2.6 Herbal products

16.2.6.1 Incense sticks

The ecology suffers when incense is burned and flowers are offered to God. During religious festivals, incense is burned in homes and temples. Particulate particles (PM), gas by-products, and several other chemical compounds can all be present in incense smoke (fumes). The use of flower wastes in the creation of herbal incense sticks has been reported. Under a Project named Mission Sakshama initiated by CSIR – CIMAP, Lucknow, aimed at using waste flowers, the floral waste is converted to incense sticks (Table 16.1). On average 1500 agarbattis (incense) can be made from one kg of raw material (filling material such as flower powder and binding material) through this technology. Flowers like *genda* are used to make incense sticks. The method has been discussed below-

- Temple wastes are collected in separate dust bins. The flowers are then segregated and set out to dry.
- The dried flowers are powdered and mixed with binding powder and sawdust.

- The charcoal is mixed in the mixture to make it combustible.
- Prepare distilled water and add to the incense mixture slowly. Knead the incense dough slowly.
- This mixture is rolled over bamboo sticks to produce the final product.
- Binding materials used may be ‘Gum Arabic’ or ‘Makko’ (an incense powder material derived from tree bark).

16.2.6.2 *Herbal* gulal

In India, the use of colour at various events dates back as far as history. According to published accounts, Holi has traditionally been coloured with natural dye made from keshu, palash or *tesu* flowers, as well as turmeric.

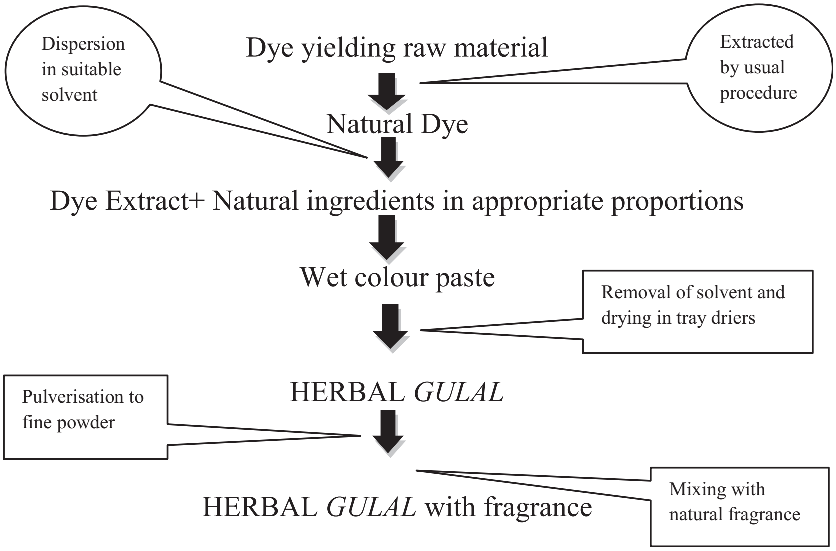
CSIR - NBRI has developed the process technology for the manufacturing of Herbal *Gulal* (dry colour powder) composition using natural dyes and natural materials in order to give a safe alternative to synthetic-based dry colours.

Around the world, dry colours are used for a variety of celebrations, dances, and home decorations. For instance, traditional Holi play involves a significant number of dry colours, and in the current climate, nearly all are synthetic dye-based colours. The market’s supply of dry colours (*Gulals*) is typically non-standard in terms of requirements and characteristics, making their quality unsuitable for direct skin and facial application. All of the blended dry colours (*Gulals*) for playing Holi are synthetic dye-based and contain substances like clay, sand, dolomite, chalk, starch, etc. Additionally, fried *Gulal* powders contain mica powder which seriously harm eyes. Some sources claim that the colours used for Holi include a sizable number of poisonous and dangerous substances.

The rapid colours used during Holi are created using chemicals and dyes intended for use in painting and dyeing. These hues are damaging to the skin and eyes. Holi colours have reportedly been prepared using a number of dangerous synthetic chemicals, according to findings from the Industrial Toxicology Research Centre in Lucknow. Auramine (yellow), Malachite (green), Rodamice B (bright reddish violet), and Methyl violet are a few of these (Violet colours).

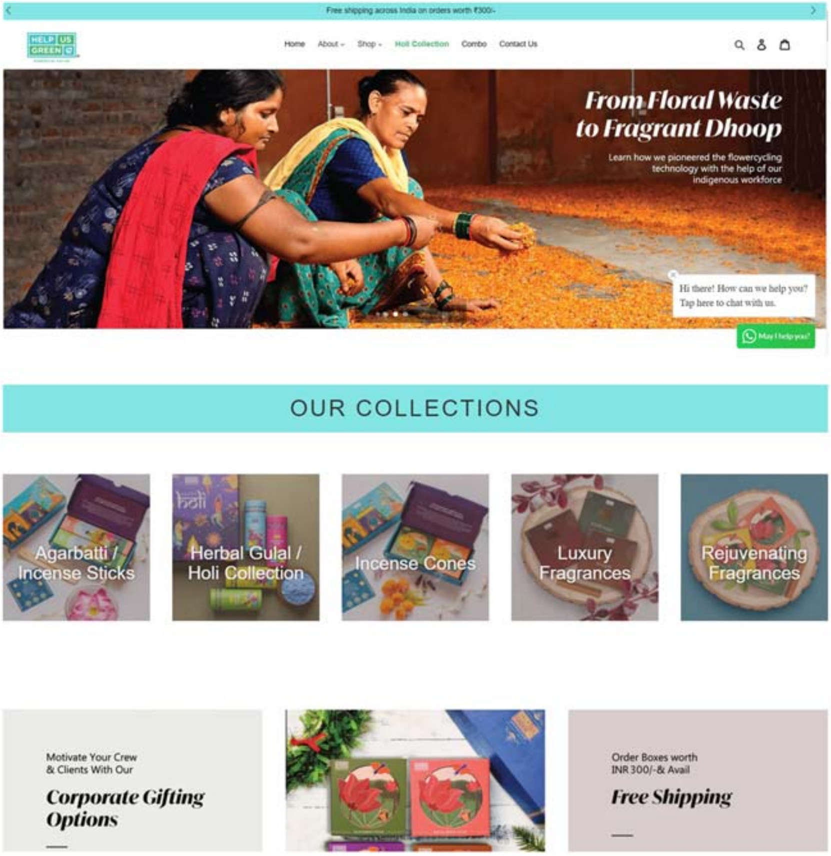
Some of the natural dyes used in the process are Turmeric, Annatto, Indigo, Beetroot, etc.

Process Technology for preparation of Herbal *Gulal*



16.2.7 Bioenergy gas

Anaerobic digestion biotechnology has drawn attention from all around the world because of the rising demand for energy and the need for economical environmental protection. Typically, fermentation biotechnologists use microorganisms or microbes as a technique to convert sugar into ethyl alcohol. Due to the severe energy issue facing the globe today, ethanol has been carefully considered to be the most acceptable fossil fuel energy source. In the anaerobic process, organic carbon is transformed into its most oxidised state (CO_2) and its most reduced state (HCO_3) by consecutive oxidations and reductions (CH_4). In the absence of oxygen, a vast variety of microorganisms accelerate the process. Anaerobic digestion has a number of benefits over conventional waste treatment techniques, including reduced sludge creation, low cost, great energy efficiency and an easy procedure. Additionally, it has a favourable effect on the environment because it combines waste stabilisation with the production of net fuel and permits the use of effluent as fertiliser. The biogas from floral waste can be used for electricity generation and as a fuel (Kumar and Swapanavahini, 2012).



Free shipping across India on orders worth ₹300/-

From Floral Waste to Fragrant Dhoop

Learn how we pioneered the flowercycling technology with the help of our indigenous workforce

Hi there! How can we help you? Tap here to chat with us.

Motivate Your Crew & Clients With Our

Corporate Gifting Options

Order Boxes worth INR 300/-& Avail

Free Shipping

16.3 Entrepreneurship development

16.3.1 Help us green NGO

Adopting a circular economy approach, the Help Us Green NGO in Uttar Pradesh, India, recycles floral waste into useful products: charcoal-free incense, organic compost and biodegradable packaging material. The founders of this green company, Ankit Agarwal and Karan Rastogi, have successfully trademarked the term ‘Flower cycling’ for their approach towards innovative technology.

16.4 Women empowerment

Women key holders of society, are directly responsible for the management and conservation of the waste that is being generated from the house and their surrounding area. Women can display her active participation at each step. This idea touched the economically weaker section of the society especially *Dalit* women, and was able to support them economically, which affects and helped in the upliftment of the society.

16.5 Conclusion

The analysis of various techniques for turning waste from temples into useful products, such as vermicompost, biogas, dyes, incense sticks, and handmade paper, suggested that temple waste could not only be disposed of safely and environmentally friendly but can also be used to create a variety of goods. This study not only suggests a different method for managing floral waste but also how it might be used as a resource to make usable products rather than being burned or dumped in the ground. It would also aid temples in bringing in more revenue. Utilizing floral waste will eventually benefit society since it would allow people to live in a cleaner and healthier environment. In order to have a clean environment and financial independence, awareness should be raised among temple administrators, pilgrims and waste-handling personnel to embrace vermicomposting on a big scale. The floral waste produced can also be used to make rose water, essence, natural colours, incense sticks, paper that has a mottled appearance, handmade paper and numerous ornamental items. This will lessen the strain on India's overburdened garbage disposal issues. The 'green temple concept' may prove useful in developing government waste management policies and in advocating a sustainable development strategy for temples.

References

Dhaked, R.K., Waghmare, C.K., Alam, S.I., Kamboj, D.V., Singh, L., 2003. Effect of propionate toxicity on methanogenesis of night soil at psychrophilic temperature. Bioresour. Technol. 87, 299–303.

Jadhav, A., Chitanand, M., Shete, H., 2013. Flower waste degradation using microbial consortium. J. Agric. Vet. Sci. 3, 1–4.

Kiran, D.A., Kavitha, H., 2020. Review on flower waste management and its conversion to value added products to conserve environment. Five days Virtual International Conference on Sustainability and New Paradigms in Civil Engineering. Organised by DSCE, Bangalore.

Kumar, M.S., Swapanavahini, K., 2012. Nutrient reduction and biogas production of rose residues by anaerobic digestion in a batch reactor. Int. J. Adv. Res. Sci. Technol. 1 (2), 125–129.

Mishra, N., 2013. Unholy Mess: Temple Waste: A Concern. Times of India. http://www.timesofindia.indiatimes.com.

Perumal, K., Sambanda, T., Savitha, J., 2012. Characterization of essential oil from offered temple flower Rosa damascena mill. Asian J. Exp. Biol. Sci. 3, 1–5.

Ravishankar, R., Raju, A.B., Abdul, B.M., Mohpatra, A.K., Kumar, M., 2014. Extraction of useful products from temple flower wastes. J. Chem. Eng. Res. 2 (1), 231–239.

Shouche, S., Pandey, A., Bhati, P., 2011. Study about the changes in physical parameters during vermicomposting of floral wastes. J. Environ. Res. 6 (1), 63–68.

Siluvai, C., Aneeshia, K., 2014. Fungal degradation of dry flower industrial waste and evaluation of dye level by HPLC. J. Microbiology Biotechnol. Res. 4 (3), 40–50.

Vankar, P.S., Sanker, R., Wijayapala, S., 2009. Utilization of temple waste flower- tagetus erecta for dyeing of cotton, wool, silk on industrial scale. J. Text. Appar. Technol. Manage 6 (1), 1–15.

Voon, H.C., Bhat, R., Rusul, G., 2011. Flower extracts and their essential oils as potential antimicrobial agents for food uses and pharmaceutical applications. Compr. Rev. Food Sci. Food Saf. 11 (1), 1–22. Available from: https://doi.org/10.1111/j.1541-4337.2011.00169.

Waghmode, M.S., Gunjal, A.B., Nawani, N.N., Patil, N.N., 2016. Management of Floral Waste by Conversion to Value-Added Products and Their Other Applications. Springer Science + Business Media, Dordrecht.

Yadav, I., Juneja, S.K., Chauhan, S., 2015. Temple waste utilization and management: a review. Int. J. Eng. Res. Technol. 2, 14–19.

Yeboah, R., 2011. Waste fabrics that can be used to produce handmade papers when combined with the paper mulberry bark. Report on Thesis Submitted to the School of Graduate Studies, Kwame Nkrumah University of Science and Technology, Kumasi.

Chapter 17

Agriwaste burning management through microbial intervention

Pradeep Semwal[1], **Basudev Majhi**[1,2], **Deen Dayal Pandey**[1], **Sankalp Misra**[1], **Shashank Kumar Mishra**[1] **and Puneet Singh Chauhan**[1]
[1]*Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India,* [2]*Academy of Scientific and Innovative Research (AcSIR), Ghaziabad, Uttar Pradesh, India*

17.1 Introduction

The growing and processing of raw agri products such as vegetables, fruits, poultry, meat, crops and dairy products generate agriwastes. They are the byproducts obtained during the processing of these agri products (El-Ramady et al., 2020). They are beneficial to humans but have low economic values compared to their cost of collection, transportation and processing for their utilization. Agriwaste can be liquids, solids, or slurries depending on the type and source of agri activities they arise from, these agriwastes can comprise food processing waste, animal waste such as manure and animal cadaver, crop waste, and hazardous and toxic agriwaste, such as herbicides, weedicides, pesticides, etc.

Based on the increasing demand for agri products, it is likely that the amount of agriwaste will also increase. These are wastes that have high carbon content. The highest amount of recorded agriwastes are generated from rice, maize, wheat/barley, cotton and sugarcane. All agriwastes are not hazardous, the ones that need to be eliminated are the wastes that are rich in toxic pollutants or radioactive products that can harm the environment and have a deleterious effect on human health. The useful Agriwastes need to be converted or transformed into useful economic products (De and Singh, 2021).

Microorganisms are the best possible options for maintaining the ecological balance along with sustainability. Thus, the introduction of plant growth-promoting bacteria (PGPR), is gaining attention among researchers. Different ecosystems when exposed to various pesticide residues, result in a huge threat to multiple useful environmental factors, plants and humans. Pesticides also degrade the plant beneficial aspects of rhizobacteria resulting

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00006-2

in lesser nodulation in leguminous plants, poor effect on root development of grasses, and decreased nitrogen accumulation in grains, and therefore, a better alternative to pesticides is required. PGPR provides better solutions to these drawbacks caused by pesticides by improving plant biomass production and lessening the detrimental phytotoxic effects. PGPR also helps in the degradation of harmful residues left over by pesticides and other contaminants, thereby improving the number of xenobiotics in the rhizosphere (Maji et al., 2020).

PGPR are bacterial groups that play a role in the promotion of plant growth by exhibiting beneficial properties for crop plants. They have some characteristics, such as colonisation in the host plant's rhizosphere and inside the root system. Some of these colonies enter the host's root by building up some important endophytic populations, which affect the plant positively. PGPR can also enhance the root surface area, which results in increased plant productivity as the nutrient uptake increases. The function of various organic molecules released by PGPR, such as Indole acetic acid, gibberellins and cytokinins, is essential for plants (Syed and Tollamadugu, 2019).

The mechanisms followed by PGPR in plant growth enhancement are reduction in plant disease, colonisation of plant roots, increase in Photo hormones via production of organic molecules, and abiotic factors, such as pH, temperature, soil nature, etc., also help PGPR to grow. PGPR also helps in the breakdown of pesticide residues, which increases the microflora production around plants resulting in sustainable growth (Mishra et al., 2019).

17.2 Agriwaste

The increase in the world's population has led to the demand for increased agri production, which in the last five decades has increased approximately three times. The importance of agri products cannot be overshadowed as the entire globe is somehow dependent on the same, but it also comes along with a negative impact on the environment, aquatic life and humans. Therefore, the wastes generated from the agri sector must be efficiently and effectively handled in a sustainable manner, which will benefit the reduction of GH gas emissions and their use as fossil fuels. It will also benefit in increasing employment, bioenergy production, green market production and their bioconversion to animal feed. Agriwastes can be generated from many sources, and a major proportion is obtained from pesticides, which include herbicides and insecticides. It is estimated that food production will fall globally to approximately 42% as pesticide usage constantly increases. Agriwastes influence human health, animal health and the environment negatively and the mismanagement of the same is a major concern. Many people who generate such waste, like farmers and households, do not know how to manage them or its negative impacts. Every year the number of agriwaste increases by the rate of 7.5%, and these wastes are either being dumped or burned, which results in air pollution, contamination of soil and also water pollution (Seglah et al., 2020).

Agriwaste is generated from unjustifiable use of farming methods and excessive chemical use in cultivation in the form of fertilisers and pesticides (Fig. 17.1). These majorly affect the rural environment particularly and the world environment generally. Expanding agri production has resulted in higher amounts of livestock waste, agro-industrial by-products and agri-crop residues. As the developing countries intensify farming systems, the rate of increase in agriwaste is supposed to rise. Among the total agriwaste produced per year, approximately 80% of organic waste is produced from total solid waste on any farm (Conly and Dae, 2019).

The types of waste generated can be classified based on the agri practice adapted and discussed in the following sections.

17.2.1 Waste from cultivation

Along with the crops, the environment also supports the generation and growth of weeds and insects. Thus creating a high demand for insecticides and weedicides to kill insects to protect the crops from diseases. This urge for chemical-based solutions often results in their excessive use by farmers. Also, after use, the containers of the chemical pesticides are usually discarded in water bodies and fields. According to Plant Protection Department (PPD), approximately 1.8% of the chemicals used in pesticides are left in

FIGURE 17.1 Types of waste generated from agri practices.

their packaging containers, which can cause severe environmental problems such as food poisoning, contaminated farm fields and deteriorated food hygiene. Many farmers use more pesticides for their cultivation than the appropriate amount required, which results in the degradation of soil health and deterioration in product quality. These excessive chemicals can evaporate in the air, causing pollution or entering water bodies by water runoffs or irrigation methods resulting in contamination and degradation of the aquatic environment (Adejumo and Adebiyi, 2020).

17.2.2 Waste from livestock

The waste from livestock production includes manures and slaughterhouse discards, wastewater from cage wash, animal baths, sanitisation of slaughterhouse and urinal discharges, and air pollutants like solid odours, hydrogen sulphide and methane. Being built around residential areas, the wastes produced are highly effective to the human population. Air pollution is caused by the smell originating from the digestion process of livestock wastes, animal urine and decaying organic matter of manure; the intensity of the smell depends upon the animal population, cage ventilation, temperature and humidity. The amounts of ammonia, methane and hydrogen sulphide depend on the stage of digestion, constituents of food, microbes and the condition health of animals. These untreated wastes generated from livestock are capable of generating GH gases, which negatively impact the environment, and fertility of the soil and cause water pollution. The proportion of water waste in livestock wastes is approximately 75%–95% of the total waste produced, and the rest includes organic matter, parasitic eggs, microbes and inorganic matter, which have the potential to spreading diseases and negatively affecting the environment (Adejumo and Adebiyi, 2020).

17.2.3 Waste from aquaculture

The expansion of aquaculture has increased the use of feeds to boost productivity. The primary determinant of the volume of trash produced in a system is the amount of feed consumed in it. Aquaculture produces a lot of waste, including metabolic waste, which can be suspended or dissolved. Approximately 30% of the feed consumed on a farm that is run efficiently will end up as solid waste. The ambient temperature affects feeding rates. Increased feeding due to a rise in temperature leads to more waste being produced. Water flow patterns in manufacturing facilities are crucial for waste management because they reduce the fragmentation of fish faeces and enable quick settling and concentration of settleable materials. This is important because it may swiftly capture a significant portion of nonfragmented faeces, which will significantly reduce the dissolved organic waste (Wang et al., 2021).

17.2.4 Waste from chemicals

In this context, chemical wastes refer to agri solid wastes, such as pesticide bottles or containers that are produced as a result of the usage of pesticides, insecticides and herbicides on farms and in retail settings. Many unskilled and untrained farmers in developing nations are still responsible for handling pesticides, insecticides and herbicides, which leads to abuse by these uninformed farmers. Some inexperienced farmers mishandle pesticide canisters, creating unpredictably dangerous environmental conditions. According to reports, roughly 2% of pesticides remain in the containers after use. Ignorant or untrained users may discard these pesticides into ponds or open fields, which can lead to water and environmental contamination, food poisoning, and the loss of many lives (Gontard et al., 2018).

17.2.5 Waste from households

Agri activities typically result in food for household consumption. The creation of agri solid wastes is typically a byproduct of family consumption of agri products. Some of these wastes are unavoidably produced. Banana and orange peels, for instance, are frequently disposed of as agri solid wastes in houses. Inadvertent production of agri solid wastes can, however, also occur from food spoilage. When restaurants are counted as kitchens, the amount of agri solid waste produced by kitchens increases (commercial kitchens). In 88%–94% of all kitchens, trash is agri solid waste (food waste) (Conly and Dae, 2019; Adejumo and Adebiyi, 2020).

17.2.6 Waste from food spoilage

With the right drying processes, the rotting of many agri products could be avoided. Farmers would have been better able to combat food spoilage and agri solid waste, improving food security and lessening the negative effects of agri solid waste on human health and the environment. Many farmers rely heavily on the erratic solar system to dry their harvest before it is stored, as well as on the outdated method of moisture monitoring, which is neither reliable nor useful. Aflatoxin infestation has reportedly occurred as a result of inadequate moisture content monitoring of grain before storage. Aspergillus flavus is the source of aflatoxin. Aflatoxin contamination of food and livestock feed can result in considerable annual crop losses worldwide. It is both a source of and a consequence of food spoilage. According to estimates, filamentous fungi cause mycotoxins to contaminate food and feed, which leads to the destruction of around 10% of the world's crop yield. According to reports, aflatoxins cause liver cancers, harm people's health in underdeveloped nations, and cause enormous economic losses. *Aspergillus flavus* produces the aflatoxins B1 and B2, which induce preharvest and postharvest crop infection.

One further significant source or cause of agriwaste is food spoilage. In addition to increasing waste due to deterioration, pest and insect infestation may also do so (Conly and Dae, 2019; Adejumo and Adebiyi, 2020)

17.3 Negative impact of agriwaste

It is impossible to overstate the impact of agri output on human health, climate change, animal health, and the environment. For instance, it has been argued that dramatically reducing GH gas emissions is necessary to stop the impending threat to the globe, its people and the planet by preventing a temperature increase of at least 35.6 °F on average. Approximately 37% and 65% of the world's methane and nitrous oxide emissions, which are more harmful than carbon dioxide, are attributed to animal production, respectively. Uncontrolled burning of agri solid wastes releases emissions that are relevant to the climate. The improper management of agri solid wastes has an impact on climate change, which, in turn, hinders the production of food. It is impossible to overstate the negative impacts of the careless disposal of agriwastes. The major impacts includethe uncontrolled burning of generated garbage that has implications for human health and the environment. Although these wastes can be recycled to improve soil fertility because they are rich in nutrients needed for sustainable agri production; indiscriminate dumping and burning of agri solid waste have led to pollution, a threat to human lives, as well as other environmental problems, calling for global attention.

The obstruction of waterways has been one of the main causes of flooding. Waterways are primarily hindered by human activity, such as building on them or clogging them with solid trash. When agri solid waste is dumped carelessly in an agri setting, streams may become blocked, which may cause flooding and the loss of life and property.

Growing agri activity has been connected to an increase in the human population, which, in turn, causes an increase in the production of agri solid waste. Approximately 7.5 billion people live on the planet at this time, and a sizeable number of them still lack access to enough food consumption. The negative repercussions of food insecurity are extensive and include anything from poor health to sluggish educational and employment growth. By 2030, one of the crucial 17 Global Sustainable Goals (GSG) is to eradicate hunger, establish food security, enhance nutrition and advance sustainable agriculture. Unfortunately, there are still around 821 million hungry people on the planet, 10 years before the deadline for achieving this goal. It has been stated that the main cause of food insecurity is not that we do not have enough to eat. It has been stated that agri solid wastes, particularly food waste, are the primary cause of food insecurity rather than a lack of food production.

Africa and Asia are the two continents with the fastest population growth in the world, and coincidentally, these two continents also have the highest rates of food insecurity and ineffective waste management. According to estimates, one-third of the food we produce each year is lost or wasted, at a cost of roughly $1 trillion each year. Wastage happens more frequently in the production and supply chain in developing countries than it does on the plate in wealthy countries. To increase food security by increasing the production of animal protein, agri solid wastes can be repurposed as nontraditional feed ingredients (Bhuvaneshwari et al., 2019).

17.4 Burning of agriwaste

Crop residue burning causes a variety of environmental issues. Burning crop residue has several harmful effects, including the release of GH gases that contribute to global warming, elevated levels of particulate matter (PM) and smog that pose health risks, the loss of agri lands' biodiversity, and a decline in soil fertility. Burning crop residue increases the number of air pollutants such as CO_2, CO, NH_3, NO_x, SO_x, nonmethane hydrocarbon (NMHC), volatile organic compounds (VOCs), semivolatile organic compounds (SVOCs) and PM by a large amount. This explains why organic carbon, nitrogen, and other nutrients that would have been kept in the soil have been lost. The burning of 98.4 Mt of agri residue led to the release of about 8.57 Mt of CO, 141.15 Mt of CO_2, 0.037 Mt of SO_x, 0.23 Mt of NO_x, 0.12 Mt of NH_3, 1.46 Mt of NMVOC, 0.65 Mt of NMHC and 1.21 Mt of PM, with CO_2 accounting for 91.6% of the total emissions. The remaining 8.43% was made up of 66.6% CO, 2.2.0% NO, 5% NMHC and 11% NMVOC (Jain et al., 2014; Bhuvaneshwari et al., 2019).

In Asia, biomass from both human and natural sources is burned on an annual basis in amounts of about 730 Mt, with India accounting for 18% of that total. Crop burning contributes greatly to climate change and raises the number of particulates (PM) in the atmosphere. The release of fine black and brown carbon (primary and secondary), which alters light absorption, is one factor in global climate change. In general, PM in the air is divided into PM2.5 and PM10 categories based on their aerodynamic diameter and chemical makeup (PM2.5, or fine PM, has an aerodynamic diameter of less than 2.5 m, and PM10, or coarse PM, has an aerodynamic diameter greater than 10 m). Lightweight particles can move farther with the wind and can remain suspended in the air for longer. Because the particles are light and stay in the air for a longer period than heavier ones, they have a greater negative impact during bad weather (Dai et al., 2018).

Burning causes soil fertility to continuously decline in addition to the production of gases and aerosols. The heat produced by burning wastes increases soil temperature and decreases bacterial and fungal populations. At a depth of 10 mm, the residue burning raises the subsurface temperature to roughly 33.8°C–42.2°C, and the effects can last for up to 15 cm of the topsoil. Frequent burning destroys the soil's beneficial microflora, and fauna lowers the soil's potential for retaining nitrogen and carbon and loses a significant amount of organic matter. Crop burning significantly disrupts the soil's carbon-nitrogen balance (Lohan et al., 2017).

According to reports, burning a tonne of straw results in the loss of 1.2 kg of sulphur, 5.5 kg of nitrogen, 2.3 kg of P, 25 kg of potassium, and the full amount of organic carbon. Crop residue from various crops typically contains 80% nitrogen (N), 25% phosphorus (P), 50% sulphur (S) and 20% potassium (K). Crop residue can enrich the soil with C, N, P and K if it is left in the actual soil. According to India's National Ambient Air Quality Standard, the permitted threshold for PM2.5 is 40 g/m^3, although the WHO standard for acceptable levels of PM2.5 in the air is 10 g/m^3. Delhi, in the National Capital Region, however, recorded a mean value of 98 g/m^3, which is 10 times higher than the WHO limit and at least twice as high as the Indian standard. In the Punjab district of Patiala, the burning of rice residue was thought to contribute between 60 and 390 mg/m^3 of PM2.5 annually. The smoke, along with fog, dust and industrial pollutants, along with the arrival of milder weather in November, creates a thick haze. Lack of wind, which generally helps to disperse air pollution, makes the issue worse for several days. Lahore, New Delhi, Lucknow and Kanpur were among the major cities that had high pollution levels. The NASA Aqua satellite's Moderate Resolution Imaging Spectro- Radiometer (MODIS) took a natural-colour picture of haze and fog covering the northern states of India. Crop residue burning in Delhi produces 17 times as much PM as all other sources combined, including industry, burning of waste and vehicle emissions. As a result, the residue burning in India's northwest accounts for around 20% of the nation's organic and elemental carbon emissions from burning agriwaste. Burning rice and wheat straw will result in cumulative CO, CO_2, N_2O and NO_x emissions of 0.11, 2.306, 0.002 and 0.084 Mt, respectively (Dai et al., 2018; Koul et al., 2022).

With 17% of the world's population and agrarian history, India produces enormous amounts of food grains like wheat and rice for both internal use and export. The majority of crop residue from the several crops grown – including rice, wheat, and sugarcane – is burned. Since these crops offer significant returns on investment, it is very difficult for farmers to discover substitute crops that yield fewer crop residues. The lack of available workers, according to the Ministry of Agriculture, is to blame for a rise in agri residue burning on farms. The months of April–May and November–December saw the majority of crop residue burning, accounting for 80% of total burning. The crop patterns used to assure higher economic returns are ascribed to this, leaving little time between the cultivation of two successive crops. Even more extreme measures are taken by certain farmers, who plant three crops in a row with little time between harvest and replanting (Koul et al., 2022).

Of note, 149.25 MT of carbon dioxide (CO_2), over 9 MT of carbon monoxide (CO), 0.25 MT of sulphur oxides (SO_x), 1.28 MT of PM, and 0.07 MT of black carbon are among the estimated emissions from crop residue burning, according to a study. These are directly responsible for the melting of the Himalayan glaciers, the haze in Delhi, and environmental degradation. One centimetre of soil is heated by burning paddy straw, bringing the soil's temperature up to 33.8°C−42.2°C (Fig. 17.2).

The bacterial and fungal communities necessary for a rich soil are eliminated as a result. Burning crop debris harms both the organic quality and other microorganisms present in the soil's top layer. The wrath of 'enemy' bugs has increased as a result of the extinction of 'friendly' pests, making crops more susceptible to disease. The ability of the higher soil layers to dissolve has also decreased. In addition to organic carbon, a report claims that one tonne of stubble burning results in the loss of 5.5 kg of nitrogen, 2.3 kg of phosphorus, 25 kg of potassium and more than 1 kg of sulphur from the soil. Because of a rise in pollution, 84.5% of people had health issues. It was shown that 76.88% of respondents reported irritation in their eyes, 44.8% in their nose and 45.5% in their throat; 41.6% of respondents reported coughing or an increase in coughing and 18% reported wheezing. According to a different study by the Institute for Social and Economic Change in Bengaluru, Punjabi rural residents spend Rs 7.6 crore annually on medical care for illnesses brought on by burning stubble (Bhuvaneshwari et al., 2019).

FIGURE 17.2 Burning of agriwaste. Source*: Adapted from https://science.thewire.in/politics/government/stubble-burning-punjab-haryana-rice-harvesting-wheat-sowing-delhi-air-pollution/*.

17.5 Agriwaste usage

A quick use of the leftovers is required by agriwaste usage technology, or the residues must be stored in a way that prevents deterioration or renders them unusable for processing into the intended end product. These wastes can be used for a variety of different things (Lohan et al., 2018) as discussed below.

17.5.1 As fertilizers

The amount of input energy needed at the farm level is directly impacted by the use of animal manure as fertilizer. Chemical fertilizer might use manure to supply 19%, 38% and 61% of the nitrogen, P and potassium. Using large-scale confinement manures as fertilizer is, however, associated with significant energy costs for transport, distribution, the need for storage facilities, odour issues, and the potential for groundwater contamination. It is reported that high phosphorus levels in poultry manure have a positive impact on crop growth and yield. When used in conjunction with mineral phosphorus fertilizer for agri usage, it is also beneficial. Because manure improves the soil's physical state, water-holding capacity, and stability of the soil structure, it boosts soil fertility by enhancing nutrient retention capacity (or cation exchange capacity) (Modak et al., 2017).

17.5.2 In anaerobic digestion for biofuel production

Manure, in particular, can be used to produce methane gas from agriwaste. The gas works well for heating applications like operating a broiler, heating water, drying grains, etc. A two-step microbial fermentation is used in the anaerobic digestion of agriwaste to create methane-rich gas. The volatile solids are first broken down by acid-forming bacteria into organic acids, which are then used by methanogenic organisms to produce methane-rich gas. Methane makes up 50%−70% of the average gas produced, along with CO_2, 25%−45%, N_2, 0.5%−3%, H_2, 1%−10% and traces of H_2S. The heating value of the gas is in the range of 18−25 MJ/m^3, according to studies. The high initial costs and the methane gas's explosive characteristics are two of the digestion system's main drawbacks. The benefits, however, considerably exceed the aforementioned drawbacks. Large poultry, swine and dairy waste may be treated and disposed of with no smell thanks to anaerobic digestion. The waste is stabilised, the digesting sludge is largely odourless, and the original waste still has some fertilizer value. Anaerobic digestion is a method for converting agri solid wastes into green energy. The stability of anaerobic digestion may be hampered by the high protein and fat content of these wastes, as well as the lack of effective technology needed to dispose of biogas residues (Meegoda et al., 2018). Pretreatment methods like mechanical (sonication), chemical (acid or alkali), oxidative (ozone), biological (enzyme addition), thermal ad osmotic (freezing and sodium chloride treatment), and biological (enzyme addition) methods may, however, improve the physical and chemical properties of the wastes, enhancing their solubilisation of organic particles, sterilisation effect and promotion of their subsequent recycling (biogas production). A sustainable renewable energy source that tends to support rural and regional development, reduce CO_2 emissions, create job opportunities, and replace energy from nonrenewable fossil fuels with green energy, biofuel and bioenergy attracts many hopes despite the many obstacles facing its production. In some developing nations, enormous quantities of agri solid waste (rich in cellulose, hemicellulose, starch, lipids and proteins) that are burned in open fields or allowed to build up can be used to create biofuels. To increase the conversion of biomass to alternative energy sources or the production of biofuels, key stakeholders and political leaders, especially in developing nations, should collaborate with researchers. This is anticipated to boost energy output and lower the financial costs associated with waste disposal, in addition to reducing the health threat posed by open-field burning or dumping of agri solid wastes (Abad-Segura et al., 2019; Halisçelik and Soytas, 2019).

17.5.3 As adsorbents for heavy metal elimination

Heavy metal pollution caused by the excessive environmental release as a result of industrialisation and urbanisation is a major issue on a global scale. Heavy metal ions like copper, cadmium, mercury, zinc, chromium and lead ions do not degrade into safe byproducts, in contrast to organic contaminants, the majority of which are subject to biological breakdown. Because heavy metal ions are hazardous to a wide variety of biological forms, their presence raises serious concerns. Adsorption is a highly successful method for removing heavy metal from waste streams, as seen in studies on the treatment of effluent harbouring heavy metal, and activated carbon has been frequently used. In using the adsorption technique, agriwastes are a low-cost alternative for the treatment of effluents containing heavy metals. Researchers have looked into low-cost agriwaste for the removal of heavy metals from wastewater, including sugarcane bagasse, rice husk, sawdust, coconut husk, oil palm shell, neem bark, etc. (Abd-Elsalam et al., 2021).

17.5.4 For pyrolysis for biochar production

In pyrolysis systems, agriwaste is heated up to a temperature of 400°C–600°C in the absence of oxygen to vapourize a portion of the material, leaving a char behind. This is considered to be a higher technology procedure for the usage of agriwastes. Others are hydro-gasification and hydrolysis. They are used for the preparation of chemicals from agriwaste, as well as for energy recovery. Of particular interest to agriculture are the preparation of alcohols for fuel, ammonia for fertilizers, and glucose for food and feed. Pyrolysis of agriwaste yields oil, char and low heating value gas. The thermochemical process known as pyrolysis, which occurs at low temperatures in an oxygen-free atmosphere, produces biochar, a porous material with fine-grained carbon content. It is a mixture of varying amounts of carbon (C), hydrogen (H), oxygen (O), nitrogen (N), sulphur (S) and ash. The very porous characteristic of biochar, when added to soil, aids in better water retention and increased soil surface area. It primarily interacts with soil bacteria, plant roots and the soil matrix. It also aids in nutrient retention and triggers a variety of biogeochemical processes. Numerous studies have documented rising pH levels, rising earthworm populations, and declining fertilizer use. In particular, biochar is employed in many different chemical applications, as well as the water treatment, building, food, cosmetic, metallurgical and wastewater treatment industries (Enaime and Lübken, 2021).

17.5.5 As animal feed

The issue with animal feed in the majority of developing nations is the scarcity of protein sources, despite significant attempts being undertaken to discover substitute supplements. Crop residues are low in protein, carbohydrate, and fat and abundant in fibre. As a result, the conventional approach to raising livestock production – supplementing grass and pasture with grains and protein concentrate – might not be sufficient to fulfil future demands for meat protein. The use of protein and grains for the human diet will compete with the use of these ingredients for animal feed. Using leftovers to feed farmed animals can solve these issues. There have been several attempts to recycle agriwaste by feeding it to cattle as a cheap source of feed for producing animal-source protein. Mycomeat, a novel feed component, is likewise made from agri solid wastes. The wastes were used as a substrate, and a mixture of the substrates and the produced fungi (mushroom) was given to broiler chicks. Mycomeat fed some agri solid wastes to albino rats as an unconventional feed ingredient and suggested treating the wastes to improve the outcome. To feed grower pigs, a mixture of 40% cassava peel, 40% concentrate and 20% watermelon wastes can be employed. Instead of being carelessly wasted or burned, poultry feathers could be used for a variety of items (Valenti et al., 2020).

17.5.6 For direct combustion

One of the earliest biomass conversion techniques used by humans is the straightforward act of burning agriwaste for fuel. The quick chemical reaction (oxidation) of biomass and oxygen, the release of energy, and the simultaneous creation of the final oxidation products of organic matter – CO_2 and water − are all components of the complete combustion of agriwaste. If oxidation proceeds at a sufficient rate, the energy released often take the form of radiant and thermal energy; the amount released depends on the biomass's enthalpy of combustion. Agriwaste must be manufactured into solid form to be used effectively during the thermal conversion process. Typically, it is burned for purposes such as heating, cooking, making charcoal, producing steam, and producing mechanical and electric power. Combustion continues to be the most common method for converting agriwaste to energy or fuels, accounting for more than 95% of all biomass energy used today (Enaime and Lübken, 2021; Zhang et al., 2018).

17.5.7 For composting

Kitchen wastes, which are primarily agri solid waste from food waste, can be sterilised and used as animal feed, composted as fertilizer, or converted into bioenergy by anaerobic digestion. These wastes are good candidates for composting due to their high levels of nutrients and organic matter, but composting may be hampered by their high levels of salt, moisture, and oil. Crop residues, like animal manure and food waste, is a good raw material for compost because of its high organic content. The natural rotting or breakdown of organic waste by microorganisms under regulated conditions is known as composting. Compost, which is a rich source of organic matter, is crucial for maintaining soil fertility and promoting sustainable agri output. Composting the soil enhances its physical, chemical and biological qualities and can entirely replace the need for agri chemicals like fertilizer and insecticides. The advantages of compost-enriched soil include greater potential for increased yields and tolerance to outside forces, including drought, disease and toxicity. Because of increased soil microbial activity, these methods also aid in greater nutrient uptake and active nutrient cycling. Composting is mediated by different micro-organisms actuating in an aerobic environment. Organic biomass naturally contains bacteria, fungi, actinomycetes, algae and protozoa, which are also sometimes artificially added to expedite decomposition. When the organic matter of animal or plant origin is broken down into substances with shorter molecular chains in aerobic circumstances, this is considered to be biological maturity. More stable, hygienic, humus-rich compost is beneficial for crops, and recycling of soil organic matter is ultimately formed (Zhang et al., 2018; Aldaco et al., 2020; de Oliveira et al., 2021).

17.5.8 Production of silica

With an atomic weight of 28, silicon is the second most prevalent nonmetallic element in the crust of the earth and produces silica and silicates. Becasue of its affinity for oxygen, it is rarely discovered in its elemental condition. It has been described as a helpful trace element that is extensively present in meals. Atherosclerosis is less likely to form, and it improves the structural integrity of hair, skin, nails, immunity, bone mineralisation, and bone calcification. Silicon compounds are broken down into accessible forms of silicic acid (ortho, meta, di and tri-silicates) in the GIT in the presence of hydrochloric acid and other gastric fluids and are distributed into various body organs. Although dietary sources are low in silicon and may need to be supplemented in diets through other ways, silicon amount decreases with age and tends to be more abundant in plants than animal sources. About 75% of the plasma silicon is eliminated within a few hours of consumption because it does not bind with plasma proteins. Silica may be found in agri solid wastes. Using chemical, thermal, and microbiological processes, silica has been created from agri solid wastes, such as maize cobs, rice husks, bagasse and rice straw (Singh and Endley, 2020).

17.6 Agriwaste management

The management of agriwaste (AWM) for ecological agriculture and sustainable development has recently attracted the attention of policymakers. The traditional method of managing agriwaste has been to release it into the environment, either treated or untreated. To avoid contaminating the air, water and land resources and to prevent the spread of dangerous compounds, wastes need to be viewed as potential resources rather than undesirable and unwanted. Incentives and technologies that are used more effectively, a shift in mindset and attitudes, and improved methods of managing agriwaste are all necessary for this. The management of organic wastes, particularly animal manure, can have a substantial negative impact on the quality of the land, water and air. Staggered wastes act as a breeding ground for insects and a vector for disease. Acid rain is caused by the uncontrolled breakdown of organic wastes, which also releases offensive gases and causes ammonia to volatilise. There are growing worries about the following due to the intensification of animal production on a limited amount of land:

- Water quality due to higher nitrogen and phosphorous loadings.
- Pathogens and antimicrobial compounds in the manure.
- Air quality issues brought on by ammonia, methane and nitrous oxide emissions.
- Soil quality due to higher potassium and phosphorous loadings.

According to the definition given, an agriwaste management system (AWMS) is a ‘planned system in which all necessary components are installed and managed to control and use byproducts of agri production in a manner that sustains or enhances the quality of air, water, soil, plant and animal resources’. Such a system is created using a total systems approach, meaning it is made to handle all agri production waste and put it to use all year round. The primary factor affecting how agriwastes are handled is their total solids (TS) concentration. The climate, the kind of animal, how much water the animal consumes, and the type of feed, for instance, have an impact on the TS concentration in expelled manure. In the majority of systems, it is possible to predict or determine the waste's consistency. Beddings or other solid garbage can be added to the waste to increase its TS concentration; water can be added to decrease its TS concentration, and the waste can be protected from further water to stabilise its TS concentration. Because it impacts the overall volume of waste to be managed, the TS concentration is significant. The initial cost of the liquid handling equipment may, however, be more than that for solid waste systems. Liquid waste management systems are frequently simpler to automate and administer than those for solid wastes (Dauda et al., 2019).

There are six fundamental AWMS functionalities – production, collection, storage, processing, transmission and usage. The quantity and type of agriwaste produced determine production. If waste production is significant enough to cause resource concerns, management of the trash is necessary. The type, consistency, volume, location and timing of the waste created are all factors that go into a thorough examination of production. The act of collecting waste at its source or place of dumping is referred to as a collection. The AWMS plan needs to specify the collection strategy, where the collection points will be when they are collected, how much labour will be needed, what equipment or physical facilities will be required, and how much each component will cost to manage and install, and how the collection will affect the consistency of the waste. The trash is temporarily contained or held in place by the storage function. The control of the schedule and timing of system processes, such as the treatment and application or use of the waste, which could be impacted by the weather or interfered with by other operations, is provided by the storage facility of a waste management system.

The waste management system should specify the length of time that the waste must be stored, the volume of storage that must be kept there, the kind of storage facility that must be used, its location, size, and installation costs, as well as its management costs and the effect that storage will have on the consistency of the waste. Any process, including physical, biological and chemical treatment, is considered to be a form of treatment since it raises the waste's potential for positive use while reducing its potential for pollution or toxicity. It includes pretreatment activities like analysing the waste's characteristics before treatment, figuring out the waste's desired characteristics after treatment, choosing the type, estimated size, location and installation cost of the treatment facility, as well as the management cost of the treatment process. Depending on the TS content, transfer refers to the movement and transportation of the waste from the collection to the usage stage as a solid, liquid or slurry. Usage, which involves recycling reusable waste materials and reintroduction of nonreusable garbage into the ecosystem, is the process of putting waste to good use (Dauda et al., 2019; Sukul and Kumar, 2020).

By lowering the amount of garbage produced, reusing the waste products with straightforward treatments, and recycling the waste by using it as a resource to create the same or modified products, the notion of minimising waste minimises the quantity and negative impacts of waste formation. This is commonly known as the '3R' method. Some waste products can be used as raw materials to create new products or recycled versions of the same product. Reusing wastes multiple times equals harvesting fresh, identical or comparable products. This prevents the exploitation of new resources and lowers the production of waste. Overall, the 3Rs individually or collectively prevent the exploitation of new resources, increase the value of the resources that have already been mined, and most significantly, reduce waste production and its negative repercussions. The three R's (reduce, reuse, recycle) philosophy seeks to achieve effective waste minimisation through:

- Choosing to use objects carefully to decrease the quantity of waste generated.
- Repetition in the usage of things or parts of things that nevertheless have useful components.
- Using waste materials themselves as resources.

It is claimed that hierarchically employing the 3Rs will improve waste minimisation efficiency. The '3Rs', or reduce, reuse and recycle, are the waste management solutions that are categorised according to their desirability and are part of the waste hierarchy. The hierarchy of the 3Rs is intended to reflect their relative importance. The waste hierarchy has evolved over the past ten years, but the fundamental idea has remained the cornerstone of the majority of waste minimisation techniques. The waste hierarchy's goal is to get the most practical use out of things while producing the least amount of garbage possible. The 3R approach as noted, is conventionally expressed through a pyramid hierarchy in which an increase in environmental benefits of each approach is placed from bottom to top (Lewis et al., 2018) (Fig. 17.3)

17.7 Use of microbes in agriwaste management

A lot of agriwaste is produced as a result of the world's growing population driving up the demand for food production. The buildup of agriwaste causes the land to deteriorate, removing its nutritional value and posing environmental risks. Crop leftovers are frequently found in agriwaste and can serve as a great source of organic carbon and plant nutrients. Retaining crop remains after harvesting helps to stop soil erosion. Most of the agriwaste is, however, burned, wreaking environmental disasters because there are insufficient sustainable waste management procedures. Burning produces pollution and a loss of the organic content of the residues by emitting 8.77 Mt of CO, 141.15 Mt of CO_2, 0.23 Mt of NO and 0.12 Mt of NH_3. Environmental restoration has undergone a revolution as a result of the realisation that

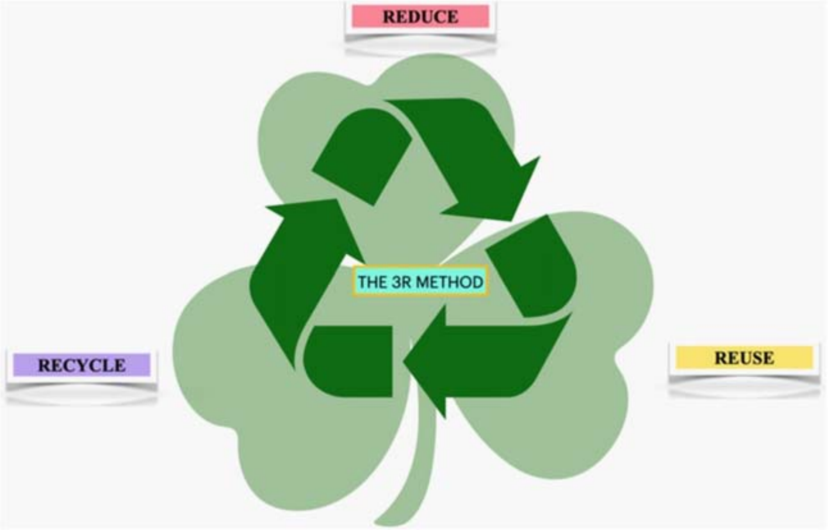


FIGURE 17.3 Sustainable practices for agriwaste management.

environmental damage poses a threat to human health. The agriculture sector can significantly profit from recycling trash, which will also improve the soil's fertility in the long term. Additionally, it can contribute to a healthier and cleaner environment by making wise use of all recyclable materials without labelling them as garbage. For a long time, microbes have been regarded as decomposers because they convert complex organic materials into simple inorganic materials that can be used to produce energy. Nitrogen fixation, phosphorus solubilisation, and cellulose degradation increase the nutritional content. Biological degradation is being employed more frequently in the treatment of all industrial, agri and organic waste management processes for both economic and ecological reasons.

Depending on the role that the organisms play in the decomposition process, the decomposers can be divided into two categories: primary and secondary. The organisms that induce fragmentation and comminution and do not require enzymes for decomposition are known as Secondary decomposers, whereas the bacteria that contain the necessary enzymes, such as cellulase, pectinase, and ligase, for breaking down agriwaste are known as Primary decomposers. Among the most well-known decomposers in existence, there are earthworms, bacteria, fungi, nematodes, and micro-arthropods. As they possess hydrolytic enzymes that break down crop leftovers such as cellulose, pectin, hemicellulose, and polyaromatic substances, fungi and bacteria play a vital role in decomposition. Crop residue lysis transforms organic waste into humus and its nutrients, which are subsequently stored in the soil and can be recycled by new crops grown during the following fallow period (Okur, 2022; Yu et al., 2019).

17.7.1 Degradation of cellulosic agriwaste

Cellulose, a polysaccharide produced by bacteria that prefer to thrive on cellulosic materials, can naturally break down cellulose, which is composed of hundreds of glucose units linked together by -1,4 bonds. Cellulase is known to be produced by both bacteria and fungi.

Fungi in cellulose degradation – Some fungus species, including Trichoderma and Aspergillus, are employed industrially to produce simple enzymes. The A. flavus has endoglucanase activity and breaks down carboxymethyl cellulose. Cellulosic waste from agriculture might decompose due to anaerobic or aerobic fungus. To dissolve cellulose, an enzyme called cellulase is secreted by the majority of fungi. They frequently cleave cellulose's 1,4-glycosidic bond. Endoglucanase, exocellobiohydrolase and glucosidase are three separate types of cellulase needed for full cellulose hydrolysis. Because of this, the cellulase system of fungi exhibits synergism (Bhagat and Kokitkar, 2021; Almeida et al., 2021)

Bacteria in cellulose degradation – It has been reported that certain strains from the phyla Firmicutes, Actinobacteria, Proteobacteria and Bacteroidetes are capable of breaking down cellulose. Some genes could potentially encode the cellulase, according to studies and studies of the bacterial genome. Further research revealed that aerobic cellulose breakdown was considerably more frequent than anaerobic degradation. Interaction between metabolically different bacteria, whose activities are controlled by environmental conditions, is necessary for anaerobic decomposition to occur. Under anaerobic conditions, cellulose is transformed into CH_4, CO_2, and H_2O (Syed and Tollamadugu, 2019).

17.7.2 Degradation of lignin

Coniferyl alcohol, p-coumaryl alcohol and sinapyl alcohol are complex organic polymers that make up the structural components of most plants. Given that lignin contains the majority of the planet's renewable carbon, lignin degradation is crucial for the carbon cycle.

Fungal decomposition – The majority of fungi that break down lignin are macro and micro fungi. White rot is the macro-fungus most well-known for its ability to break down a complex, resistant organic lignin into simpler molecules when grown in an aerobic environment. On the other hand, microfungi can break down lignin in facultative and aerobic conditions to produce humus and manure. Laccase, manganese peroxidase and lignin peroxidase are the enzymes produced by white rot, and they break down lignin to CO_2 in a process called enzymatic combustion. This procedure aids in the bioconversion of agriwaste into energy while also reducing pollution. As the lignin is broken down, cellulose is made usable (Chukwuma et al., 2020).

Bacterial decomposition – Bacteria can not only degrade lignin but can also degrade the small molecules that are formed during the degradation of lignin by fungi. Some strains are also known to mineralise and solubilise lignin. Bacteria use the same enzymes for lignin degradation as fungi and are known to be better than them as they have a wider tolerance for temperature and oxygen limitation (Abu-Omar et al., 2021).

17.7.3 Degradation of hemicellulose

Hemicellulose is a polysaccharide made up of several sugar molecules, including those with five and six carbons. Although infrequently, they may also contain L-sugars, they primarily contain D-pentose sugar. They hydrolyse into their parts, sugar and uronic acid when they are subjected to microbial deterioration. Hemicellulase is a highly well-known enzyme that produces hydrolysis.

The constituent sugars include xylose, arabinose, galactose, and mannose, which are further decomposed into organic acids, alcohol, carbon dioxide (CO_2), and water. Uronic acid is further hydrolysed into pentoses and carbon dioxide (CO_2). Both the fungal and bacterial populations are known to degrade hemicellulose and its different forms, including Xylan, Mannan, Galactan and Arabinan. Several enzymes work together to break down the primary carbohydrate, Xylan. When xylan is broken down, the endo-1,4-β-xylanase produces oligosaccharides, and 1,4-β-xylosidase processes the oligosaccharides to produce xylose (Abe et al., 2021).

17.7.4 Degradation of pectin

The plant cell wall contains the heteropolysaccharide pectin, which gives the plant cell structural effectiveness. Galactouronic acid makes up its structure. Microbes release pectinase to break down pectin, according to several studies. To fully break down, more auxiliary enzymes are required. Both of those auxiliary enzymes, polysaccharide lyases and pectin and glycoside hydrolases, are involved in the breakdown of carbohydrates (Chhetri et al., 2020; Ibrahim et al., 2021).

Agriwaste is rising together with the rise in global food demand. The resultant organic waste has a variety of outcomes depending on the location. In rural settings, trash is used as animal feed or composted to produce humic substances; but in urban areas, waste is either burnt or disposed of in landfills, which, over time, can have detrimental impacts on the ecosystem and groundwater. The theory of waste valorisation, which was recently developed, states that trash can be used to increase the value of a product by turning it into other resources, such as chemicals, fuels and energy that can have significant economic and environmental benefits. Recycling agri trash can both assure a secured food supply and sustainable food production.

Microorganisms are crucial to the breakdown of agriwaste and give the soil nutrients and a source of energy that can soon be recycled. Compared to incineration, this approach is more sustainable. Most issues related to the dangerous environmental effects of other methods of treating agriwaste can be reduced by using it.

17.8 Conclusion

The use of agrochemicals, greater arable land with modern irrigation systems, and mechanised farming methods have all led to the exponential rise in agricultural productivity and agricultural waste in many nations. The management of agricultural waste sustainably has grown to be a significant concern, particularly for emerging nations like India with rising populations, production rates, and economic growth. Crop residues are one type of agricultural waste that has presented unique issues because of its enormous volume and lack of management tools.

Agriculture has always been an important aspect of providing various resources to the existing human life; as huge as its production scale varies, so does the amount of agriwaste generated worldwide. It consists of waste produced by the chemical products used for increasing the productivity of plants in agricultural industries, aquaculture and households. These wastes are either burned or buried, resulting in the pollution of the environment – mainly air and water pollution, as it releases gases such as CO, CO_2, NH_3, SO_x, NO_x, etc., which are toxic to the environment, as well as the human and animal health. Low awareness related to these wastes among humans has also been an important reason for the mistreatment of agriwaste. The long-established uses of these wastes involve them being used as fodder for animals, as manure for the enhancement of soil health, along with their use in paper, textile and other agri-based industries. The byproducts obtained by the various agricultural practices, when treated in the right way, have the potential to be used in a sustainable manner.

Along with the pollution caused, a large amount of microbial community living in the soil is eradicated by the burning practices. Some of these microbial communities have symbiotic relationships with plants helping them in maintaining their productivity and keeping the soil nourished for longer generations. The microorganisms also have the potential to degrade the main constituents of the agriwaste, which are cellulose, hemicellulose, pectin, lignin and polyaromatic substances, by producing hydrolytic enzymes. Among the vast microbial diversity, PGPRs have gained keen interest as they have the potential to promote plant growth and can help in nitrogen fixation, phosphate solubilisation, and hormone production, and can also affect plant metabolism directly by improving the root development and plant enzymatic activity. By the synergistic effect, they can also help other microbes to benefit plants by either promoting the growth of the plant or by suppressing the pathogens that can harm them as the hydrolytic enzymes produced by the PGPR have the potential to breakdown the cell wall, DNA and proteins of the phytopathogen. This microbial implementation, therefore, lowers the use of chemicals such as fertilizers and pesticides, which also helps in decreasing the pollution from the environment.

Acknowledgments

The authors acknowledge the Director, CSIR National Botanical Research Institute for providing facilities and support during the study. This work is supported by the CSIR-Network (MLP0048 and MLP0049) and an in-house project (OLP109) funded by the Council of Scientific and Industrial Research, New Delhi, India.

References

Abad-Segura, E., Cortés-García, F.J., Belmonte-Ureña, L.J., 2019. The sustainable approach to corporate social responsibility: a global analysis and future trends. Sustainability 11 (19), 5382.

Abd-Elsalam, K.A., Periakaruppan, R., Rajeshkumar, S. (Eds.), 2021. Agri-waste and microbes for production of sustainable nanomaterials. Elsevier.

Abe, M.M., Branciforti, M.C., Brienzo, M., 2021. Biodegradation of hemicellulose-cellulose-starch-based bioplastics and microbial polyesters. Recycling 6 (1), 22.

Abu-Omar, M.M., Barta, K., Beckham, G.T., Luterbacher, J.S., Ralph, J., Rinaldi, R., et al., 2021. Guidelines for performing lignin-first biorefining. Energy Environ. Sci. 14 (1), 262−292.

Adejumo, I.O., Adebiyi, O.A., 2020. Agri solid wastes: causes, effects, and effective management. Strateg. Sustain. Solid. Waste Manag. 8.

Aldaco, R., Hoehn, D., Laso, J., Margallo, M., Ruiz-Salmón, J., Cristobal, J., et al., 2020. Food waste management during the COVID-19 outbreak: a holistic climate, economic and nutritional approach. Sci. Total Environ. 742, 140524.

Almeida, D.A., Horta, M.A.C., Ferreira Filho, J.A., Murad, N.F., de Souza, A.P., 2021. The synergistic actions of hydrolytic genes reveal the mechanism of *Trichoderma harzianum* for cellulose degradation. J. Biotechnol. 334, 1−10.

Bhagat, S.A., Kokitkar, S.S., 2021. Isolation and identification of bacteria with cellulose-degrading potential from soil and optimization of cellulase production. J. Appl. Biol. Biotechnol. 9 (6), 1−6.

Bhuvaneshwari, S., Hettiarachchi, H., Meegoda, J.N., 2019. Crop residue burning in India: policy challenges and potential solutions. Int. J. Env. Res. Public. Health .

Chhetri, R.K., Aryal, N., Kharel, S., Poudel, R.C., Pant, D., 2020. Agro-based industrial wastes as potent sources of alternative energy and organic fertilizers. Current developments in biotechnology and bioengineering. Elsevier, pp. 121−136.

Chukwuma, O.B., Rafatullah, M., Tajarudin, H.A., Ismail, N., 2020. Lignocellulolytic enzymes in biotechnological and industrial processes: a review. Sustainability 12 (18), 7282.

Dai, Y., Sun, Q., Wang, W., Lu, L., Liu, M., Li, J., et al., 2018. Utilizations of agriwaste as adsorbent for the removal of contaminants: a review. Chemosphere 211, 235−253.

Dauda, A.B., Ajadi, A., Tola-Fabunmi, A.S., Akinwole, A.O., 2019. Waste production in aquaculture: sources, components and managements in different culture systems. Aquaculture Fish. 4 (3), 81−88.

de Oliveira, M.M., Lago, A., Dal'Magro, G.P., 2021. Food loss and waste in the context of the circular economy: a systematic review. J. Clean. Prod. 294, 126284.

De, A., Singh, S., 2021. Analysis of fuzzy applications in the agri-supply chain: a literature review. J. Clean. Prod.

El-Ramady, H., El-Henawy, A., Amer, M., Omara, A.E.D., Elsakhawy, T., Elbasiouny, H., et al., 2020. Agricultural waste and its nano-management: mini review. Egypt. J. Soil. Sci. 60 (4), 349−364.

Enaime, G., Lübken, M., 2021. Agriwaste-based biochar for agronomic applications. Appl. Sci. 11 (19), 8914.

Gontard, N., Sonesson, U., Birkved, M., Majone, M., Bolzonella, D., Celli, A., et al., 2018. A research challenge vision regarding management of agriwaste in a circular bio-based economy. Crit. Rev. Environ. Sci. Technol. 48 (6), 614−654.

Halisçelik, E., Soytas, M.A., 2019. Sustainable development from millennium 2015 to sustainable development goals 2030. Sustain. Dev. 27 (4), 545−572.

Hansen, C.L., Yeol Cheong, D., 2019. Chapter 26—Agriwaste management in food processing. In: Kutz, Myer (Ed.), Handbook of Farm, Dairy and Food Machinery Engineering, third ed. Academic Press.

Ibrahim, H.H., Bilsborrow, P.E., Phan, A.N., 2021. Intensification of pre-treatment and fractionation of agri residues. Chem. Eng. Processing-Process Intensif. 159, 108231.

Jain, N., Bhatia, A., Pathak, H., 2014. Emission of air pollutants from crop residue burning in India. Aerosol Air Qual. Res. 14 (1), 422−430.

Koul, B., Yakoob, M., Shah, M.P., 2022. Agriwaste management strategies for environmental sustainability. Environ. Res. 206, 112285.

Lewis, K., Tzilivakis, J., Warner, D., & Green, A., 2018. Agri-environmental Management in Europe: Sustainable Challenges and Solutions-From Policy Interventions to Practical Farm Management. 5m Books Ltd.

Lohan, S.K., Jat, H.S., Yadav, A.K., Sidhu, H.S., Jat, M.L., Choudhary, M., et al., 2018. Burning issues of paddy residue management in north-west states of India. Renew. Sustain. Energy Rev. 81, 693−706.

Maji, S., Dwivedi, D.H., Singh, N., Kishor, S., Gond, M., 2020. Agriwaste: its impact on environment and management approaches. In: Bharagava, R. (Ed.), Emerging eco-friendly green technologies for wastewater treatment. Microorganisms for sustainability, vol. 18. Springer, Singapore.

Meegoda, J.N., Li, B., Patel, K., Wang, L.B., 2018. A review of the processes, parameters, and optimization of anaerobic digestion. Int. J. Environ. Res. Public Health 15 (10), 2224.

Mishra, P., Singh, P.P., Singh, S.K., Verma, H., 2019. Sustainable agriculture and benefits of organic farming to special emphasis on PGPR. Role of plant growth promoting microorganisms in sustainable agriculture and nanotechnology. Woodhead Publishing, pp. 75−87.

Modak, P., Pariatamby, A., Seadon, J., Bhada-Tata, P. and International Solid Waste Association, 2017. Asia waste management outlook.

Okur, N., 2022. Management of soil microorganisms in sustainable agri systems. Innovative Agri Pratices Soil, Plant. Environ. 89.

Seglah, P.A., Wang, Y., Wang, H., Bi, Y., Zhou, K., Wang, Y., et al., 2020. Crop straw utilization and field burning in Northern region of Ghana. J. Clean. Prod. 261, 121191.

Singh, S.P., Endley, N., 2020. Fabrication of nano-silica from agri residue and their application. Nanomaterials for agriculture and forestry applications. Elsevier, pp. 107−134.

Sukul, P., Kumar, K., 2020. Integrated crop residues management for a sustainable agriculture: only answer to the stuble burning—a review. Plant. Arch. 20 (2), 2660−2664.

Syed, S., Tollamadugu, N.P., 2019. Role of plant growth-promoting microorganisms as a tool for environmental sustainability. Recent developments in applied microbiology and biochemistry. Academic Press, pp. 209−222.

Valenti, F., Porto, S.M., Selvaggi, R., Pecorino, B., 2020. Co-digestion of by-products and agri residues: a bioeconomy perspective for a Mediterranean feedstock mixture. Sci. Total Environ. 700, 134440.

Wang, W., Wei, W., Gao, S., Chen, G., Yuan, J., Li, Y., 2021. Agri and aquaculture wastes as concrete components: a review. Front. Mater. 8, 762568.

Yu, Y.Y., Li, S.M., Qiu, J.P., Li, J.G., Luo, Y.M., Guo, J.H., 2019. The combination of agriwaste compost and biofertilizer improves yield and enhances the sustainability of a pepper field. J. Plant. Nutr. Soil. Sci. 182 (4), 560−569.

Zhang, L., Zhang, J., Zeng, G., Dong, H., Chen, Y., Huang, C., et al., 2018. Multivariate relationships between microbial communities and environmental variables during co-composting of sewage sludge and agriwaste in the presence of PVP-AgNPs. Bioresour. Technol. 261, 10−18.

Chapter 18

Biowaste fortification by plant growth promoting microorganisms

Tanya Singh[1,2], **Sankalp Misra**[1,2], **Srishti Kar**[1], **Vijaykant Dixit**[1],
Shashank Kumar Mishra[1,2] **and Puneet Singh Chauhan**[1,2]
[1]Microbial Technology Division, Council of Scientific and Industrial Research-National Botanical Research Institute (CSIR-NBRI), Lucknow, Uttar Pradesh, India, [2]Academy of Scientific and Innovative Research (AcSIR), Ghaziabad, Uttar Pradesh, India

18.1 Introduction

As the world is becoming more populous and people are shifting to cities, more products and services are being provided to civilians and investment and international trade are increasing. As a result, organisations are facing the challenge of dealing with an enormous amount of waste being generated. Large-scale organic waste from many sources, including municipal and urban, agriculture, food industry, etc., is among the major issues that need an effective solution. Bioconversion into value-added goods can help to solve this problem in a sustainable and eco-friendly manner. Biomass is expected to provide 10%−14% of world energy generation by 2050, thus reducing our dependency on fossil fuels in future (Ghisellini and Ulgiati, 2020). Wastes generated by the agricultural sector, industrial leftovers and urban pollution have some valuable organic ingredients present in them that can be transformed into valuable goods. Because of the large volume of organic waste generated each year (about 13*109 tonnes), its usage in future is likely to be beneficial (Mihai and Ingrao, 2018). Bio-valourisation of organic waste is being used to produce high-value products that are more useful than the traditional treatment (Fig. 18.1). While attempting to address the issue of environmental deterioration, this has increased the value of organic wastes. Because fossil energy is scarce, past research on biomass transformation focused mostly on strategies to generate different forms of bioenergy (Seo et al., 2013).

Recent Trends in Solid Waste Management. DOI: https://doi.org/10.1016/B978-0-443-15206-1.00014-1

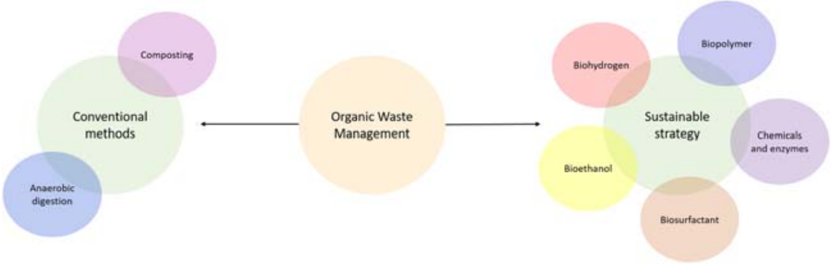


FIGURE 18.1 Conventional and Sustainable methods for management of organic waste.

Organic matter is an important component of soil because it helps with crop yield, sustainable agriculture, and soil fertility replenishment. It is a valuable source of organic material with numerous benefits. On the other hand, when organic waste is disposed of in landfills, it generates greenhouse gases and can cause annoyance and health hazards (Al-Khatib et al., 2010). Therefore, it is necessary to adopt efficient assembly, transportation and regulation methods to turn it into a soil conditioner (Liew et al., 2022). Carbohydrates, lipids, proteins, lignocellulose and other organic components make up the organic fractions, and because of this, a large range of high-value products can be generated economically using biotechnology. Plant growth-promoting microorganisms (PGPM) can digest bioorganic materials found in organic wastes to obtain the energy they require for growth and development. Organic waste can be turned into a variety of functional products using several methods, including biological processes like anaerobic digestion and other fermentation technologies (Cucina et al., 2021). Biopolymers, pharmaceutical preparations, biofuel, functional meals, bioactive chemicals, and a variety of other biotechnological applications can all benefit from the biotransformation of biowastes through PGPM (Mohan et al., 2019).

18.2 Types of organic waste

Organic waste has accumulated as a result of human, agricultural, industrial, and municipal activities. All wastes can be recycled back into the natural system. Organic waste now accounts for roughly 46% of all solid waste globally (Fig. 18.2) (Chavan et al., 2022). According to data from the Global Waste Management Reviewers, global organic waste output reached 2.01 billion tonnes in 2016. (Al-Khatib et al., 2010). Roughly 43% of worldwide trash is produced in Middle and East Asia and Europe. On the contrary, The North and Central East African regions contribute only a small amount, accounting for 15% of worldwide organic waste output (Asomani-Boateng, 2007). Organic waste can be produced by almost any biomass production stream, and the waste content mostly determines the origin of biomass.

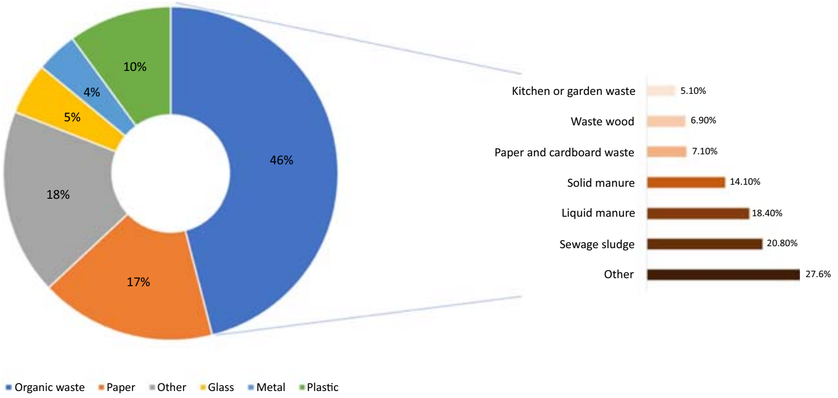


FIGURE 18.2 Global organic waste production.

Biodegradable waste is categorised according to the source of waste residue to ensure its efficient use.

18.2.1 Municipal solid waste

Municipal waste management is a crucial problem for municipalities all across the world. In many nations, the cheapest and most advantageous method of managing municipal solid waste is landfilling (Jeswani and Azapagic, 2016). Food waste, sewage and wastewater, and human waste are all included in municipal solid waste. Food waste management is critical as it has the potential for public health issues and environmental deterioration. The most effective strategy to manage food waste is to reduce it at the source. But, owing to the easy biodegradability and the rising proportion of eat-outs, the preventive approach is somewhat limiting and often impossible. The emphasis has shifted away from using these wastes as animal feeds and toward their use as value-added bioactive compounds. Production of bioethanol, enzymes, fibres, proteins and phenolic compounds are among the valuable compounds that scientists are trying to produce from organic waste (Tsouko et al., 2019). The combination of home effluent (kitchen and bathroom water) and water from business institutions, eateries, health centres, and schools are known as sewage sludge and wastewater. It has an unpleasant odour, agitation foam, slow settling, and a high solid and organic content. Commonly the sewage water from wastewater treatment is spread on open lands. But it is a known fact that untreated sewage sludge contains a variety of pollutants and toxins, which are (re)introduced into the surrounding through this practice, and as a result, have negative impacts on the ecosystem and human health. Faeces and urine generally comprise 30 g of C, 90 g of organic matter, 10–12 g of N, and 2–3 g of P and K daily. After the decomposition of excreta, it can be used as a fertilizer and soil conditioner, and to increase soil water holding capacity while also mitigating heavy metals and toxins. To fight a variety of harmful organisms found in excreta, such as bacteria, protozoans, viruses and helminths, effective management and sanitation are required (Polprasert and Koottatep, 2017).

18.2.2 Agricultural wastes

Seed production, horticulture, nursery and market gardens, dairy farming, and livestock rearing all produce significant amounts of agricultural waste. The majority of agricultural waste is burned or incinerated due to inadequate management and restricted access to disposal facilities (Polprasert and Koottatep, 2017). Crop leftovers are the most plentiful, inexpensive, and easily accessible organic waste with a high lignocellulose content for natural biodegradation. Wheat straw, pearl millet, sugarcane stover, sorghum stover, maize, rice straw, finger millet, leaves, chopped trees and yard trimmings are all included. The majority of crop waste in India is being used as animal feed, for home fuel and for animal bedding. When agro-waste decomposes, it enriches soil N-microbial biomass (Polprasert and Koottatep, 2017), and thus, can be used to improve soil health.

18.2.3 Agro-industrial waste

The agricultural industry is a producer of food, rice products, coconut products, beverages, tea, coffee, textiles, printing and paper products, tobacco, furniture, rubber and more. The main wastes of these industries are flour wastes, dye waste, husks and sawdust and rubber waste, which offer the possibility of recycling. India is ranked 1st in the production and consumption of sugar cane. Molasses used in linen bagasse, press sludge or filter cake are by-products of the sugar cane industry and distilleries. India produces 7.5 Mt molasses, 43.8 Mt bagasse, 8 Mt press sludge, and 7.4 Mt bagasse ash as by-products annually (Dotaniya et al., 2016). China, The United States of America, The Philippines and India are the top producers of fruit and vegetable processing waste, each producing 32, 15, 6.5 and 1.8 million tonnes of organic waste, respectively (Wadhwa and Bakshi, 2013). Pineapple (45%–55%), mango and citrus fruits (30%–50%), guava (10%), pomegranate (40%–50%), and grapes (20%) produce biowaste as a result of industrial processing (Banerjee et al., 2018).

Oil cakes and oil mill effluents are produced in great quantities, primarily during oil extraction and during the mill cleaning process. These usually contain cellulosic materials, fats, oils and greases, which can cause serious environmental problems if untreated wastewater is discharged into rivers. Oil cake can be used as a bacterial feed for producing commercially important enzymes in a cost-effective manner (Edwinoliver et al., 2010).

18.2.4 Animal and poultry waste

Urinary waste includes manure from cattle and poultry waste, solid manure or farmyard manure, and wastewater from feedlots, bedding, farms, silage juices, and other sources. Beef cattle, chickens, dairy cattle, broilers, ducks, swine, sheep and lambs, and horses are among the livestock that contributes to the production of organic waste. The content of waste varies according to the animals, including species, age, weight, size, water intake and feed, climatic circumstances and so on (Polprasert and Koottatep, 2017). Faeces, waste feeds, bedding materials, and feathers make up chicken litter. Every year, millions of tonnes of chicken litter are produced worldwide, most of which are recycled and applied to agricultural land as cheap organic fertilizers (Enticknap et al., 2006). Poultry waste can be a good source of keratin, calcium, lipid, essential minerals and proteins, but they are not being used in an economically efficient way. (Jayathilakan et al., 2012).

18.3 Bioconversion of waste to value-added products

Organic waste contains many important nutrients essential for the growth of microorganisms, and so they can be used as a substrate for the production of bio-products via microbes. Biowaste can be a source for essential components like carbohydrates, proteins and minerals that can be used by plant growth promoting microbes (PGPM) as biosynthetic precursors for the generation of diverse bioproducts. Several PGPMs have been tested for the production of valuable products, which are listed in Table 18.1.

18.3.1 Organic fertilizer

With an increasing demand for the production of more food, there is pressure on the agriculture sector to improve yield as the land is limited. So, disproportionate use of chemicals is being practised, which has led to a decline in soil health and disturbed the agroecosystem. An alternative to chemical fertilizers is organic manure, which can increase the yield of crop plants and can also improve overall soil health. The conversion of organic waste using decomposers and fortification of that matter with beneficial PGPM is gaining interest. Composting is mainly a three-phase process involving bacteria, fungi and actinomycetes, which act on the lignocellulosic material (Zeng et al., 2016). Aerobic composting produces a stabilized organic product, which is beneficial for plant health. Research is going on around the world to find good biodegraders that can decompose waste in an effective manner. Recently many microbes have been isolated and characterised from different agricultural fields, humus-rich sites and composts that show better decomposition ability and can catalyse biodegradation. (Eida et al., 2009). Some reported that biodegraders have been listed in Table 18.2. Fortification of the organic product with PGPM having N-fixing, P-solubilising, K-mobilising ability

TABLE 18.1 Biowaste conversion through plant growth promoting microorganisms.

PGPM	Organic waste source	Value added product	References
Aspergillus species	Winery wastewater	Lipase, Protease, Tannase	Bharathiraja et al. (2020)
Bacillus tequilensis	Vine shoot	Biosurfactant	Bharathiraja et al. (2020)
Enterococcus sp.	Sludge from cardboard industry	Polyhydroxyalkanoates	Yadav et al. (2020)
Lactobacillus bulgaricus	Whey	Single-cell protein	Panesar et al. (2016)
Rhyzopus orysae	Waste soybean oil	Biodiesel	Lee et al. (2013)
Klebsiella pneumoniae	Waste glycerol	1,3 Propanediol	Almeida et al. (2012)
Saccharomyces cerevisiae	Spruce bark	Bioethanol	Kemppainen et al. (2012)
Aspergillus terreus	Mild alkali-treated rice straw	Cellulases	Narra et al. (2012)
Aspergillus niger	Apple pomace biowaste	Citric acid	Dhillon et al. (2011)
Aspergillus niger	Orange peel	Polygalacturonases	Maller et al. (2011)
Rhodopseudomonas palustris	Sugar beet molasses	Biohydrogen	Özgür et al. (2010)

and biocontrol agents can further improve the effectiveness of the compost. The organic compost is rich in C content, and thus, makes the native PGPM more effective (Rashid et al., 2016). It also improves the interaction of microbes with root rhizosphere. The compost combined with PGPM can improve soil health through the bioremediation of pollutants (petroleum hydrocarbons) (Gomez and Sartaj, 2014).

18.3.2 Biohydrogen

Hydrogen is the most eco-friendly, non-polluting fuel because only water vapours are created when hydrogen is used. The biological methods for

TABLE 18.2 Microorganisms for enhanced decomposition of organic waste.

PGPM	Biodegradation activity	References
Bacillus sp.	Lignin degradation	Chandra et al. (2007)
Pseudomonas putida	Manganese peroxydases	Ahmad et al. (2010)
Bacillus sp.	Carboxymethylcellulose and xylan degradation	Tsegaye et al. (2019)
Pseudomonas aeruginosa	Manganese peroxydases	Bholay et al. (2012)
Bacillus amyloliquefaciens	Cellulase	Taha et al. (2015)
Paenibacillus sp.	Lignin degradation	Chandra et al. (2007)
Ochrobactrum oryzae	Laccase activity	Tsegaye et al. (2019)
Trichoderma reesei	Cellulase and hemicellulose	Nieves et al. (1997)
Trichoderma harzianum	Hemicellulose degradation	Jørgensen et al. (2003)
Aspergillus niger	Cellulase, xyalanase production	Romero et al. (2007)

producing biohydrogen can be divided into two categories: light-dependent (Biophotolysis, Photofermentation) and light-independent methods (via microbial electrolysis cell and dark fermentation (DF)) (Nabgan et al., 2021). Over the years, a system−hybrid method has also been examined. Yang et al. (2010) found that combining dark fermentation and photofermentation with a pretreated maize cob resulted in higher production of biohydrogen. The maximal biohydrogen yield via dark fermentation was 120.3 mL H_2/g corn cob. The cost of raw material poses a major issue for the production of biohydrogen by photofermentation or dark fermentation, which can be solved by using waste products. The choice of waste materials for biohydrogen generation is based on factors like biodegradability, availability, carbohydrate content and cost. Several waste materials, such as agricultural industry wastes containing starch, cheese whey, cellulose-containing wastes, simple sugars, or waste sludge from treatment plants, and food industry wastewaters, have been experimented on over the years as starters for the production of H_2. The synthesis of biohydrogen requires a larger number of reactors due to poorer yields. This issue can be addressed through the proper selection and use of suitable organisms or microbial consortiums and the optimisation of environmental settings, as well as improvements in light usage efficiency and the development of good photo-bioreactors (Tian et al., 2019).

18.3.3 Bioethanol

An energy crisis has emerged from a considerable rise in the use of fossil fuels and their exhaustion. As a result, there is an increased demand for cheaper methods to generate energy. Along with the production of hydrogen from biowaste, the production of ethanol and biodiesel has also dragged the attention of scientists. Hydrolysis, fermentation and ethanol recovery are the three basic processes in the production of ethanol from organic wastes. These three phases work as a chain of events that affect the subsequent stages. Hydrolysis is used to produce fermentable sugar, from which ethanol is obtained through fermentation. Several PGPMs, such as *Trichoderma reesei, Aspergillus niger*, *Saccharomyces cerevisiae* and *Wickerhamomyces anomalus*, can successfully use the diversity of fermentable sugars generated from the hydrolysis process due to advancement in genetic technology (Koçar and Civaş, 2013). Kuhad et al. (2011) have shown that the genetically altered *Pichia* strain can consume both hexose and pentose and ferment them simultaneously to improve conversion efficiency. A variety of biowaste and lignocellulosic materials can be used for the production of ethanol, with starch being commonly used as raw material for lab to pilot-scale production (Yang et al., 2014). The problem of minimal yield in the fermentation stage can be handled successfully by using genetically engineered bacterial isolates for enhanced sugar degradability. Pineapple leaves are a great bioethanol source material because they contain 60%–80% holocellulose. In a recent study, it was reported that under ideal conditions, the rate of bioethanol production was 7.12% v/v (Chintagunta et al., 2017). Out of the total ethyl alcohol production, ethanol fuel accounts for nearly 2/3 of the production, as it is a low-cost alternative to gasoline. (Yang et al., 2014). Existing research aims to reduce the number of processing stages and increase in production of bioethanol.

18.3.4 Enzymes and chemicals

Enzymes are defined as biological catalysts that play an important role in different facets of life and its processes. Plants, animals and microbes are all-natural sources of enzymes. Rising enzyme prices have, however, prompted researchers to focus on and investigate alternative ways of producing enzymes, as well as to improve existing methods for producing enzymes in a cost-effective and long-term manner. Microbial sources have been preferred over plant and animal sources due to technological advantages such as shorter fermentation times, economic feasibility and larger yield. Several bioprocesses are arising as a result of breakthroughs in biotechnology for the synthesis of enzymes from various industrial wastes. The agro-industry is one of the most important contributors to this. Lignocellulosic materials are the primary elements of such wastes (consisting of cellulose, hemicelluloses and lignin). Pre-treatments are frequently used in procedures. Using agro-wastes, bacteria and fungi could be employed to produce enzymes. On a large scale, pre-treatment that allows for high rates of saccharification could improve enzyme synthesis. Ravindran et al. (2018) examined the yield of four enzymes by *Bacillus* species when given different agro-waste as feed-stock. To determine the influence of various pre-treatment measures, such as microwave, acid, alkali, or ultrasound on enzyme synthesis, researchers used various treatments. Pre-treatment with chemicals resulted in a greater yield of enzymes like amylase and protease (Singh et al., 2016). *Aspergillus niger* was shown to have a threefold and a twofold increase in xylanase and cellulase activity, respectively. *Nesterenkonia* sp. strain F, a halotolerant bacterium, produced 97 U amylase/mL bagasse obtained from sweet sorghum plants, which were treated with Tween 80. Single-cell protein-containing important amino acids, such as threonine (7%), phenylalanine (8%) and methionine and lysine (6% each), were isolated from hemicellulosic hydrolysate (Lolasi et al., 2018). *A. niger* produced lipase from shea butter cake with a yield of 3.35 U/g ds (Salihu et al., 2016). Jatropha seed cake, when fermented with *Pseudomonas aeruginosa* yielded 1818 U protease/g ds and 625 U/g ds lipase (Lee et al., 2013). Several compounds have been discovered as byproducts of industrial bioprocesses, in addition to enzymes. Microbes can produce citric acid from starch and sucrose found in renewable resources, including fruit waste, such as citrus waste, banana peels, apple pomace and date debris, as well as beet and cane molasses. Citric acid was produced from date waste in the range of 0.67–0.76 g/g sugars and 0.193–0.64 g/g from orange peel by *Aspergillus niger* (Acourene and Ammouche, 2012; Hamdy, 2013). *Lactobacillus delbrueckii* fermented food waste rich in starch prepared with a mixture containing- amyloglucosidase, protease and amylase to generate lactic acid. *Lactobacillus bulgaricus* recycled unsterile food waste and produced 35.12 g/L of lactic acid when pH and temperature were regulated (Wang et al., 2010).

18.3.5 Biopolymer

Feed accounts for approximately 45% of the cost of microbially generated polyhydroxyalkanoates (PHAs). Process profitability can be improved by substituting feed with food and fruit waste. PHA output by a PGPM *Wautersia eutropha* under batch-fed conditions using wheat milling byproducts was 0.3 g PHB/g waste (Xu et al., 2010). Similarly, *Cupriavidus necator*, produced 0.38 g PHB/g glucose by digesting food waste under continuous culture conditions. Polyhydroxybutyrate (PHB) synthesis by *C. necator* from kitchen trash was reported with a productivity of 0.065 g/L/h and yield of 0.38 g/g under batch culture conditions. By using intermittent feeding, production was increased fourfold under batch-fed conditions (Omar et al., 2011). In a study, after the production of H_2, food waste was fermented and converted into PHA. P(3HB-3HV), a PHA copolymer containing 90% PHB, was found in unfermented waste; however, its concentration was up to 35% from fermented food waste in an anaerobic condition (Reddy and Mohan, 2012). Kumar et al. (2009) reported that pea shells could be an alternative feed for the production of PHAs. They found that *Bacillus* sp. could metabolise pea shells under batch culture to produce 1.95—3.37 g PHB/L. *Bacillus thuringiensis*, on the other hand, proved to be a productive bacterium, producing 2.07—3.0 g PHB/L. By combining the hydrolysates of pea shells and onion peels, PHA output by *B. thuringiensis* and *B. cereus* rose to 3.66-fold (Kumar et al., 2016). The regulation of PHA composition is possible by regulating fatty acid generation from potato peels, apple pomace, onion peels and pea shells using hydrolytic enzymes produced by *Proteus* sp. and *Bacillus* sp. (Kumar et al., 2016).

18.3.6 Biosurfactant

Biosurfactant are amphiphilic molecules produced by microbes that have diverse industrial applications. The food and agriculture waste are rich in lignocellulose, and thus, provide microbes with a nutrient-rich environment (Panadare, 2015). It has been demonstrated that this lignocellulosic biomass can be converted into products that can be used as supplements for the production of biosurfactants (Das et al., 2018). Different biowastes have been tested for their ability to create biosurfactants that include citrus, fried palm oil, yam residues and sugarbeet bagasse (Panadare, 2015). The substrate used for the growth of microbe determines the type of biosurfactant produced (Singh et al., 2019). The agro-waste is first hydrolysed to produce sugar that can then be used for the production of biosurfactants. *Penicillium citrinum* is reported to hydrolyse straw and hay (Mardawati et al., 2018). The sugar produced can be converted by *Achromobacter* sp. The crude extract of *Achromobacter* sp. BP(1)5 biosurfactant showed 27.22% emulsifying efficiency on kerosene (Mouafo et al., 2018). Glycolipopeptides and glycolipids were obtained when *Lactobacillus* bacteria were provided with sugar beet molasses (cellulose fibres) and glycerol as substrate.

18.4 Future perspective and challenges for sustainable waste management

The addition of waste residues and their poor management has resulted in the contamination of water, land and air. Therefore, biowaste management and the production of valuable products are gaining popularity in many countries. Biomass-based biofuel manufacturing is improved more than other bio-based goods such as organic acids, biofertilizers, enzymes and so on. Increased productivity and yield, as well as simplifying the biowaste transformation process, are the two most essential challenges that these biotechnology advancements must address. An effective technique to improve the bioconversion process is to use genetically altered strains that can use many precursors derived from bio-waste at the same time and convert them to valuable compounds (Blankschien et al., 2010). On a laboratory scale, the generation of organic acids and enzymes from waste biomass has been demonstrated, which is the first step toward pilot-scale bioconversion. Additionally, the development of effective processes and their improvement is required. For industrial purposes, product quality is a critical factor. Consistent quality and improved efficiency of goods need to be confirmed in a pilot-scale setting before they can be used commercially in the case of enzymes and antibiotics (Kuhad et al., 2011). Upscaling should concentrate on biowaste that is simple to convert and available in large quantities. The bioconversion technique requires careful optimisation based on a range of factors, including pre-treatment strategies, feedstock selection and by-product generation output, to ensure its market efficiency.

18.5 Conclusions

Developing sustainable solutions for managing organic waste is important for our society. The strategies should be economically beneficial and safe for the environment while being efficient in taking full advantage of the valuable material in the organic waste. Organic waste is a well-studied fuel source when converting to useful bioproducts due to its composition and inexpensive properties. Various uses have been identified to achieve sustainable recycling of organic waste, including the production of high-quality organic products, integration into industrial processes and energy recovery. The commercialisation of the products derived from biowaste is an important step yet to be taken as the research done till date is restricted to lab demonstrations. Bio-valourisation of waste and the production of high-value products will significantly promote a sustainable recycling economy. The main challenges for biowaste management include the handling and transportation of organic waste from different industries and the research and development of downstream recycling processes. Future work will include rationalisation and technology integration of high-value products to bridge the gap between technology and the market.

Acknowledgements

The authors acknowledge the Director, CSIR National Botanical Research Institute for providing facilities and support during the study. This work is supported by the CSIR-Network (MLP0048 and MLP0049) and an in-house project (OLP109) funded by the Council of Scientific and Industrial Research, New Delhi, India.

References

Acourene, S., Ammouche, A., 2012. Optimization of ethanol, citric acid, and α-amylase production from date wastes by strains of *Saccharomyces cerevisiae*, *Aspergillus niger*, and *Candida guilliermondii*. J. Ind. Microbiol. Biotechnol. 39 (5), 759–766.

Ahmad, M., Taylor, C.R., Pink, D., Burton, K., Eastwood, D., Bending, G.D., et al., 2010. Development of novel assays for lignin degradation: comparative analysis of bacterial and fungal lignin degraders. Mol. Biosyst. 6 (5), 815–821.

Al-Khatib, I.A., Monou, M., Zahra, A.S.F.A., Shaheen, H.Q., Kassinos, D., 2010. Solid waste characterization, quantification and management practices in developing countries. A case study: Nablus district–Palestine. J. Environ. Manag. 91 (5), 1131–1138.

Almeida, J.R., Fávaro, L.C., Quirino, B.F., 2012. Biodiesel biorefinery: opportunities and challenges for microbial production of fuels and chemicals from glycerol waste. Biotechnol. Biofuels 5 (1), 1–16.

Asomani-Boateng, R., 2007. Closing the loop: community-based organic solid waste recycling, urban gardening, and land use planning in Ghana, West Africa. J. Plan. Educ. Res. 27 (2), 132–145.

Banerjee, S., Ranganathan, V., Patti, A., Arora, A., 2018. Valorisation of pineapple wastes for food and therapeutic applications. Trends Food Sci. Technol. 82, 60–70.

Bharathiraja, B., Iyyappan, J., Jayamuthunagai, J., Kumar, R.P., Sirohi, R., Gnansounou, E., et al., 2020. Critical review on bioconversion of winery wastes into value-added products. Ind. Crop. Products 158, 112954.

Bholay, A.D., Borkhataria, B.V., Jadhav, P.U., Palekar, K.S., Dhalkari, M.V., Nalawade, P.M., 2012. Bacterial lignin peroxidase: a tool for biobleaching and biodegradation of industrial effluents. Univers. J. Environ. Res. Technol. 2, 1.

Blankschien, M.D., Clomburg, J.M., Gonzalez, R., 2010. Metabolic engineering of *Escherichia coli* for the production of succinate from glycerol. Metab. Eng. 12 (5), 409–419.

Chandra, R., Raj, A., Purohit, H.J., Kapley, A., 2007. Characterisation and optimisation of three potential aerobic bacterial strains for kraft lignin degradation from pulp paper waste. Chemosphere 67 (4), 839–846.

Chavan, S., Yadav, B., Atmakuri, A., Tyagi, R.D., Wong, J.W., Drogui, P., 2022. Bioconversion of organic wastes into value-added products: A review. Bioresour. Technol. 344, 126398.

Chintagunta, A.D., Ray, S., Banerjee, R., 2017. An integrated bioprocess for bioethanol and biomanure production from pineapple leaf waste. J. Clean. Prod. 165, 1508–1516.

Cucina, M., De Nisi, P., Trombino, L., Tambone, F., Adani, F., 2021. Degradation of bioplastics in organic waste by mesophilic anaerobic digestion, composting and soil incubation. Waste Manag. 134, 67–77.

Das, A.J., Ambust, S., Kumar, R., 2018. Development of biosurfactant based cosmetic formulation of toothpaste and exploring its efficacy. Adv. Ind. Biotechnol. 1 (5).

Dhillon, G.S., Brar, S.K., Verma, M., Tyagi, R.D., 2011. Utilization of different agro-industrial wastes for sustainable bioproduction of citric acid by *Aspergillus niger*. Biochem. Eng. J. 54 (2), 83–92.

Dotaniya, M.L., Datta, S.C., Biswas, D.R., Dotaniya, C.K., Meena, B.L., Rajendiran, S., et al., 2016. Use of sugarcane industrial by-products for improving sugarcane productivity and soil health. Int. J. Recycling Org. Waste Agric. 5 (3), 185–194.

Edwinoliver, N.G., Thirunavukarasu, K., Naidu, R.B., Gowthaman, M.K., Kambe, T.N., Kamini, N.R., 2010. Scale up of a novel tri-substrate fermentation for enhanced production of *Aspergillus niger* lipase for tallow hydrolysis. Bioresour. Technol. 101 (17), 6791–6796.

Eida, M.F., Nagaoka, T., Wasaki, J., Kouno, K., 2009. Isolation and characterization of cellulose-decomposing bacteria inhabiting sawdust and coffee residue composts. Microbes Environ. 1202170356-1202170356.

Enticknap, J.J., Nonogaki, H., Place, A.R., Hill, R.T., 2006. Microbial diversity associated with odor modification for production of fertilizers from chicken litter. Appl. Environ. Microbiol. 72 (6), 4105–4114.

Ghisellini, P., Ulgiati, S., 2020. Circular economy transition in Italy. Achievements, perspectives and constraints. J. Clean. Prod. 243, 118360.

Gomez, F., Sartaj, M., 2014. Optimization of field scale biopiles for bioremediation of petroleum hydrocarbon contaminated soil at low temperature conditions by response surface methodology (RSM). Int. Biodeterior. Biodegrad. 89, 103–109.

Hamdy, H.S., 2013. Citric acid production by *Aspergillus niger* grown on orange peel medium fortified with cane molasses. Ann. Microbiol. 63 (1), 267–278.

Jayathilakan, K., Sultana, K., Radhakrishna, K., Bawa, A.S., 2012. Utilization of byproducts and waste materials from meat, poultry and fish processing industries: a review. J. Food Sci. Technol. 49 (3), 278–293.

Jeswani, H.K., Azapagic, A., 2016. Assessing the environmental sustainability of energy recovery from municipal solid waste in the UK. Waste Manag. 50, 346–363.

Jørgensen, H., Eriksson, T., Börjesson, J., Tjerneld, F., Olsson, L., 2003. Purification and characterization of five cellulases and one xylanase from *Penicillium brasilianum* IBT 20888. Enzyme Microb. Technol. 32 (7), 851–861.

Kemppainen, K., Inkinen, J., Uusitalo, J., Nakari-Setälä, T., Siika-aho, M., 2012. Hot water extraction and steam explosion as pretreatments for ethanol production from spruce bark. Bioresour. Technol. 117, 131–139.

Koçar, G., Civaş, N., 2013. An overview of biofuels from energy crops: Current status and future prospects. Renew. Sustain. Energy Rev. 28, 900–916.

Kuhad, R.C., Gupta, R., Khasa, Y.P., Singh, A., Zhang, Y.H.P., 2011. Bioethanol production from pentose sugars: Current status and future prospects. Renew. Sustain. Energy Rev. 15 (9), 4950–4962.

Kumar, P., Ray, S., Kalia, V.C., 2016. Production of co-polymers of polyhydroxyalkanoates by regulating the hydrolysis of biowastes. Bioresour. Technol. 200, 413–419.

Kumar, T., Singh, M., Purohit, H.J., Kalia, V.C., 2009. Potential of *Bacillus* sp. to produce polyhydroxybutyrate from biowaste. J. Appl. Microbiol. 106 (6), 2017–2023.

Lee, J.H., Kim, S.B., Yoo, H.Y., Lee, J.H., Han, S.O., Park, C., et al., 2013. Co-immobilization of *Candida rugosa* and *Rhyzopus oryzae* lipases and biodiesel production. Korean J. Chem. Eng. 30 (6), 1335–1338.

Lee, J.H., Kim, S.B., Yoo, H.Y., Suh, Y.J., Kang, G.B., Jang, W.I., et al., 2013. Biodiesel production by enzymatic process using Jatropha oil and waste soybean oil. Biotechnol. Bioprocess Eng. 18 (4), 703–708.

Liew, C.S., Yunus, N.M., Chidi, B.S., Lam, M.K., Goh, P.S., Mohamad, M., et al., 2022. A review on recent disposal of hazardous sewage sludge via anaerobic digestion and novel composting. J. Hazard. Mater. 423, 126995.

Lolasi, F., Amiri, H., Asadollahi, M.A., Karimi, K., 2018. Using sweet sorghum bagasse for production of amylases required for its grain hydrolysis via a biorefinery platform. Ind. Crop. Products 125, 473–481.

Maller, A., Damásio, A.R.L., Silva, T.M.D., Jorge, J.A., Terenzi, H.F., Polizeli, M.D.L.T.D.M., 2011. Biotechnological potential of agro-industrial wastes as a carbon source to thermostable polygalacturonase production in *Aspergillus niveus*. Enzyme Res. 2011.

Mardawati, E., Andoyo, R., Syukra, K.A., Kresnowati, M.T. A.P., Bindar, Y., 2018, March. Production of xylitol from corn cob hydrolysate through acid and enzymatic hydrolysis by yeast. In IOP Conference Series: Earth and Environmental Science (Vol. 141, No. 1, p. 012019). IOP Publishing.

Mihai, F.C., Ingrao, C., 2018. Assessment of biowaste losses through unsound waste management practices in rural areas and the role of home composting. J. Clean. Prod. 172, 1631−1638.

Mohan, S.V., Dahiya, S., Amulya, K., Katakojwala, R., Vanitha, T.K., 2019. Can circular bioeconomy be fueled by waste biorefineries—a closer look. Bioresour. Technol. Rep. 7, 100277.

Mouafo, T.H., Mbawala, A., Ndjouenkeu, R., 2018. Effect of different carbon sources on biosurfactants' production by three strains of *Lactobacillus* spp. Biomed. Res. Int. 2018.

Nabgan, W., Abdullah, T.A.T., Nabgan, B., Jalil, A.A., Nordin, A.H., Ul-Hamid, A., et al., 2021. Catalytic biohydrogen production from organic waste materials: A literature review and bibliometric analysis. Int. J. Hydrog. Energy 46 (60), 30903−30925.

Narra, M., Dixit, G., Divecha, J., Madamwar, D., Shah, A.R., 2012. Production of cellulases by solid state fermentation with *Aspergillus terreus* and enzymatic hydrolysis of mild alkali-treated rice straw. Bioresour. Technol. 121, 355−361.

Nieves, R.A., Ehrman, C.I., Adney, W.S., Elander, R.T., Himmel, M.E., 1997. Survey and analysis of commercial cellulase preparations suitable for biomass conversion to ethanol. World J. Microbiol. Biotechnol. 14 (2), 301−304.

Omar, F.N., Rahman, A.A., Hafid, H.S., Mumtaz, T., Yee, P.L., Hassan, M.A., 2011. Utilization of kitchen waste for the production of green thermoplastic polyhydroxybutyrate (PHB) by *Cupriavidus necator* CCGUG 52238. Afr. J. Microbiol. Res. 5 (19), 2873−2879.

Özgür, E., Mars, A.E., Peksel, B., Louwerse, A., Yücel, M., Gündüz, U., et al., 2010. Biohydrogen production from beet molasses by sequential dark and photofermentation. Int. J. Hydrog. Energy 35 (2), 511−517.

Panadare, D.C., 2015. Applications of waste cooking oil other than biodiesel: a review. Iran. J. Chem. Eng. (IJChE) 12 (3), 55−76.

Panesar, P.S., Kaur, R., Singla, G., Sangwan, R.S., 2016. Bio-processing of agro-industrial wastes for production of food-grade enzymes: progress and prospects. Appl. Food Biotechnol. 3 (4), 208−227.

Polprasert, C., Koottatep, T., 2017. Organic Waste Recycling: Technology, Management and Sustainability. IWA Publishing.

Rashid, M.I., Mujawar, L.H., Shahzad, T., Almeelbi, T., Ismail, I.M., Oves, M., 2016. Bacteria and fungi can contribute to nutrients bioavailability and aggregate formation in degraded soils. Microbiol. Res. 183, 26−41.

Ravindran, R., Hassan, S.S., Williams, G.A., Jaiswal, A.K., 2018. A review on bioconversion of agro-industrial wastes to industrially important enzymes. Bioengineering 5 (4), 93.

Reddy, M.V., Mohan, S.V., 2012. Influence of aerobic and anoxic microenvironments on polyhydroxyalkanoates (PHA) production from food waste and acidogenic effluents using aerobic consortia. Bioresour. Technol. 103 (1), 313−321.

Romero, E., Speranza, M., García-Guinea, J., Martínez, Á.T., Martínez, M.J., 2007. An anamorph of the white-rot fungus *Bjerkandera adusta* capable of colonizing and degrading compact disc components. FEMS Microbiol. Lett. 275 (1), 122−129.

Salihu, A., Bala, M., Alam, M.Z., 2016. Lipase production by *Aspergillus niger* using sheanut cake: An optimization study. J. Taibah Univ. Sci. 10 (6), 850−859.

Seo, S.W., Yang, J., Min, B.E., Jang, S., Lim, J.H., Lim, H.G., et al., 2013. Synthetic biology: tools to design microbes for the production of chemicals and fuels. Biotechnol. Adv. 31 (6), 811−817.

Singh, R., Kumar, M., Mittal, A., Mehta, P.K., 2016. Microbial enzymes: industrial progress in 21st century. 3 Biotech. 6 (2), 1–15.

Singh, P., Patil, Y., Rale, V., 2019. Biosurfactant production: emerging trends and promising strategies. J. Appl. Microbiol. 126 (1), 2–13.

Taha, M., Shahsavari, E., Al-Hothaly, K., Mouradov, A., Smith, A.T., Ball, A.S., et al., 2015. Enhanced biological straw saccharification through coculturing of lignocellulose-degrading microorganisms. Appl. Biochem. Biotechnol. 175 (8), 3709–3728.

Tian, H., Li, J., Yan, M., Tong, Y.W., Wang, C.H., Wang, X., 2019. Organic waste to biohydrogen: a critical review from technological development and environmental impact analysis perspective. Appl. Energy 256, 113961.

Tsegaye, B., Balomajumder, C., Roy, P., 2019. Isolation and characterization of novel lignolytic, cellulolytic, and hemicellulolytic bacteria from wood-feeding termite *Cryptotermes brevis*. Int. Microbiol. 22 (1), 29–39.

Tsouko, E., Alexandri, M., Fernandes, K.V., Freire, D.M. G., Mallouchos, A., Koutinas, A.A., 2019. Extraction of phenolic compounds from palm oil processing residues and their application as antioxidants. Food Technol. Biotechnol. 57 (1), 29.

Wadhwa, M., Bakshi, M.P.S., 2013. Utilization of fruit and vegetable wastes as livestock feed and as substrates for generation of other value-added products. Rap. Publ. 4, 1–67.

Wang, M., Xu, Z.J., Qiu, T.L., Sun, X.H., Han, M.L., Wang, X.M., 2010.). Kinetics of lactic acid fermentation on food waste by *Lactobacillus bulgaricus*, Advanced Materials Research, Vol. 113. Trans Tech Publications Ltd., pp. 1235–1238.

Xu, Y., Wang, R.H., Koutinas, A.A., Webb, C., 2010. Microbial biodegradable plastic production from a wheat-based biorefining strategy. Process. Biochem. 45 (2), 153–163.

Yadav, B., Pandey, A., Kumar, L.R., Tyagi, R.D., 2020. Bioconversion of waste (water)/residues to bioplastics-A circular bioeconomy approach. Bioresour. Technol. 298, 122584.

Yang, H., Guo, L., Liu, F., 2010. Enhanced bio-hydrogen production from corncob by a two-step process: dark-and photo-fermentation. Bioresour. Technol. 101 (6), 2049–2052.

Yang, X., Lee, S.J., Yoo, H.Y., Choi, H.S., Park, C., Kim, S.W., 2014. Biorefinery of instant noodle waste to biofuels. Bioresour. Technol. 159, 17–23.

Zeng, Y., De Guardia, A., Dabert, P., 2016. Improving composting as a post-treatment of anaerobic digestate. Bioresour. Technol. 201, 293–303.

This page intentionally left blank

Index

Note: Page numbers followed by "*f*" and "*t*" refer to figures and tables, respectively.

Woodhead Publishing
Advances in Pollution Research

Recent Trends in Solid Waste Management

Recent Trends in Solid Waste Management presents comprehensive information on the recent advances in solid waste treatment and management processes. The book covers a wide range of topics related to solid waste treatment, disposal, and handling. Readers can also acquire up-to-date/background information about global annual solid waste generation and effective waste management strategies (recycle, reuse, and remediate). Furthermore, future study directions (open questions) are identified. This book assists both the academic and industrial communities by providing extensive information on waste separation procedures and new technology for solid waste treatment.

Key features

- Covers a wide range of topics related to solid waste treatment methods, including new treatment systems
- Provides a thorough overview of the processing and disposal of solid and hazardous waste generated during the COVID-19 pandemic
- Highlights innovative technologies that make recovering value-added materials easier and generate bioelectricity from solid waste

About the editors

Dr Balasubramani Ravindran, PhD is a professor (assistant) in the Department of Environmental Energy and Engineering, Kyonggi University, Suwon-Si, South Korea. His research focuses on solid waste treatment and wastewater generated from domestic and industries through aerobic and anaerobic fermentation. He has over 150 publications in international peer-reviewed journals, filed patents, edited books, and published book chapters. He serves as an academic editor, editorial board member, and guest editor in several international journals.

Dr Sanjay Kumar Gupta is a deputy technical officer at the Environmental Engineering, Civil Eng., Indian Institute of Technology in Delhi, India. Dr Gupta began his career in 1999 as a researcher at the CSIR-Indian Institute of Toxicology Research in Lucknow, India. Dr Gupta has published over 100 articles and 5 books. His scientific interests include ecotoxicological risk assessment, bioremediation and the application of algae technologies for wastewater treatment and biofuels.

Dr Sartaj Ahmad Bhat is working as a JSPS researcher at the River Basin Research Center, Gifu University, Japan. His research interests focus on the vermicomposting treatment of various solid wastes, especially for investigations on the fate and behavior of emerging pollutants during the biological treatment of organic wastes. He has published over 60 papers in peer-reviewed journals and edited over 13 books.

Dr Puneet Singh Chauhan is the Senior Principal Scientist and Head of the Microbial Technologies Division of CSIR-National Botanical Research Institute. His research interests include agricultural microbiology, plant-microbe interactions, and microbial ecology. He is on the editorial boards of several prestigious journals and has over 100 SCI research articles to his credit.

Dr Neha Tyagi is a lead consultant at Rebound Enviro Tech Pvt. Ltd. Meerut, India. Dr Tyagi is a gold medalist in MTech (Environmental Eng.) from the Malaviya National Institute of Technology (MNIT), Jaipur, India. She completed her PhD in 2016 in environmental sciences and engineering from the Indian Institute of Technology, Bombay, India. Dr Neha has published more than 18 papers in peer-reviewed international journals and conferences.

ISBN 978-0-443-15206-1
9 780443 152061

ELSEVIER
WP
WOODHEAD
PUBLISHING
An imprint of Elsevier
elsevier.com/books-and-journals