FROM THE GROUND UP: MEASUREMENT, REVIEW, AND EVALUATION OF PLASTIC WASTE MANAGEMENT AT VARYING LANDSCAPE SCALES

by

AMY LAUREN BROOKS

(Under the Direction of Jenna R. Jambeck)

ABSTRACT

Once considered a miracle material, plastic is now a rapidly accumulating environmental threat. Over 6.3 billion metric tons (MT) of plastic waste has been generated since the 1950s, but the global distribution of waste varies considerably. Given the transboundary nature of plastic pollution, multi-scale, integrated policy approaches are likely to be some of the most effective. To inform multi-scale actions, we need multi-scale science. Here, four topics related to plastic waste management are presented at varying scales and contexts. First, a global-scale assessment of the international plastic scrap trade provides an update and comparison of trade patterns following the implementation of the Chinese ban on imported plastic waste. An estimated 5.6 million MT of plastic scrap have been displaced as of 2019, with evidence that the ban may have reduced mismanaged plastic waste in China, while shifting plastic scrap to other developing economies, including Sub-Saharan Africa. Next, a regional-scale assessment of plastic waste management in the Latin America and Caribbean region determined that 7.15 million MT of mismanaged plastic waste was generated in the region in 2020. While upper-middle income countries in the South American subregion contribute most, high-income Caribbean countries have substantially higher rates of per capita mismanaged plastic than the rest of the region. Third, a basin-scale analysis of plastic waste management in the Ganges River basin provides an empirical extrapolation method for comparison of modeled estimates of plastic litter resulting in an estimated 9.8 billion plastic litter items and 245,000 metric tons of plastic litter lost to the basin environment in 2019. Finally, an experimental method for rapid data collection of community-scale anthropogenic debris is assessed and applied to ten communities in the Ganges River basin finding that remote communities have higher per capita litter densities, indicating that rural waste management strategies may reduce the quantities of plastic waste reaching the basin and river environment. Though ranging in context, approach, and scale, these studies taken together reveal patterns in urbanization, income, and rural vs. urban waste management that may inform strategies for preventing plastic pollution.

INDEX WORDS: PLASTIC WASTE, PLASTIC SCRAP TRADE, LATIN AMERICA AND THE CARIBBEAN, GANGES RIVER, PHOTOQUADRAT ANALYSIS

FROM THE GROUND UP: MEASUREMENT, REVIEW, AND EVALUATION OF PLASTIC WASTE MANAGEMENT AT VARYING LANDSCAPE SCALES

by

AMY LAUREN BROOKS

B.Ag. Animal Science, University of Georgia, 2013

B.S. Civil Engineering, University of Georgia, 2016

A Dissertation Submitted to the Graduate Faculty of The University of Georgia in Partial Fulfillment of the Requirements for the Degree

DOCTOR OF PHILOSOPHY

ATHENS, GEORGIA

2021

FROM THE GROUND UP: MEASUREMENT, REVIEW, AND EVALUATION OF PLASTIC WASTE MANAGEMENT AT VARYING LANDSCAPE SCALES

by

AMY LAUREN BROOKS

Major Professor: Jenna Jambeck
Committee: Jason Locklin
Lan Mu
Tim Townsend

Electronic Version Approved:

Ron Walcott
Vice Provost for Graduate Education and Dean of the Graduate School
The University of Georgia
August 2021

DEDICATION

To the many people I have had the privilege of meeting who were willing to share their stories and insights about picking up, picking through, reusing, revitalizing, refurbishing, restoring, and disposing of waste. Thank you for turning toward what others turn away and for seeing utility in the forgotten.

ACKNOWLEDGEMENTS

I have so much gratitude for all the support, encouragement, and friendship that brought me to this point. First and foremost, I extend my deepest appreciation to Dr. Jenna Jambeck for her continuous mentorship, guidance, and patience. I am so grateful for the incredible opportunities and experiences that I got to be a part of because of you. Thank you for being a role model of adaptability and thoughtfulness in research and for encouraging me to trust my intuition, to be kind to myself, and to always reach for new heights. I look forward to continued collaboration! I also extend my sincere thanks to Dr. Jason Locklin, Dr. Mu Lan, and Dr. Tim Townsend for their time and willingness to serve on my committee and for guiding this work with their valuable insight and feedback. Along with the University of Georgia, I thank the National Geographic Society and the Inter-American Development Bank for financial support and resources during my program.

The UGA College of Engineering and the Jambeck Research Group have provided a wonderful academic home over the years. I am so appreciative of the support and efforts of my friends, colleagues, and travel buddies in the Jambeck Research Group. In the last year, I had the great benefit of taking two of the most thought-provoking classes I had at UGA with Dr. John Schramski and Dr. Jo Walther, and I extend my gratitude to them for many inspiring conversations that have greatly influenced how I might like to shape my academic career going forward. Thanks also to my colleagues in the EETIncubator led by Dr. Nicki Sochacka for providing a fun and welcoming space for discussion of all things research and for expanding my interests to include engineering education. I extend my thanks and acknowledgement to Emily Barber and Ellen Jones for their hard work processing the photoquadrat data in Chapter Five. Lastly, I gratefully acknowledge the assistance of Margaret Sapp and Victoria Martinez. Thank you for always looking after all of us students!

Perhaps some of the most valuable outcomes of my PhD are the friendships that have survived through and emerged from it. Through the National Geographic Sea to Source Expedition, I had the great privilege of collaborating with a very long list of some amazing scientists that I'm proud to now call friends. Stateside, I thank Dr. Jacqueline Newbold, for over a decade of close friendship, adventure, music sharing, and rousing debates. You inspired me to go to grad school in the first place and I will be forever grateful for that! I also extend a huge thanks to my friend, Dr. Leeann Haskins, especially for answering a LOT of stats questions, but also for offering so much support, thoughtful problem solving, and friendship as I finished up my dissertation. Finally, I thank my dearest friend and closest thing I'll ever have to a sister, Katelyn Stallings, PE, for putting up with me nonstop, keeping me on my toes with your quick wit, checking calcs and writing, and for being my biggest cheerleader and closest confidant. They say that when the character of a person is uncertain, look to their friends. By that logic, I have a lot to be proud of.

I am especially thankful to my boyfriend, Will Huffine, who is the epitome of kindness, pragmatism, and loyalty. Thank you for making me so happy. It means so much to me that you looked after me and made sure I took breaks, got sleep, saw some sunshine, and stayed fed while I was working on this dissertation! Thanks also for putting up with my incessant curiosity, which often presented in the form of never-ending would-your-rather questions, and for including me in your plans. I can’t wait for all the fun we have ahead of us!

Finally, there’s the family you pick and then there’s the family you get! My brother, Matthew, was the first person who ever told me about plastic in the ocean after boated throughout while serving in the Marines. Thank you for your positive attitude and for future financial assistance if this thing doesn’t pay off! To my dad, thank you for instilling perseverance, the Ranger spirit, and a passion for learning. If it wasn’t for you, I might not have ever ended up studying engineering in the first place.

I would not have been able to accomplish this without the unwavering support and encouragement from my mom, Melissa. Without complaint or objection, you came to my aid so many times and in so many ways as I wandered through this academic journey. I cannot thank you enough for taking care of Jack while I was traveling, in the field, or defending my dissertation! Your company and help during the final push of my degree program made this far easier than it should have been. Thank you for all the good times, for still loving me even when I’m at my worst, and for reminding me to not take things too seriously! Thanks for everything, Mom!

Finally, although I spent a lot of time separated by highways, mountains, and oceans from my dog Jack, he gave my days a much-needed structure and balance. Thank you for your companionship and protection, and for all your hugs and staring contests and walks and for not getting tired of me telling you how cute you are.

Thank you all. My heart is so full of gratitude.

TABLE OF CONTENTS

LIST OF TABLES

Page

LIST OF FIGURES

Page

CHAPTER 1

INTRODUCTION

Plastics fall under a wide range of synthetic or semi-synthetic materials derived from petroleum, natural gas, or biobased substances and can be designed and formed for nearly any application to be rigid, flexible, waterproof, resistant to temperature changes, durable against physical or chemical interactions, and coated, dyed, or mixed with other additives such as flame retardants, antimicrobials, plasticizers, etc. Because of their low-density properties, plastics allow for light-weighting in certain industries such as packaging, which can reduce costs associated with transport and shipping of products. This relatively inexpensive and wide-reaching versatility has allowed plastic to become embedded in almost every aspect of human life. Between 1950 and 2017, a cumulative 8.3 billion metric tons (MT) of virgin plastic resin and fibers have been produced, representing a compound annual growth rate (CAGR) of 8.4% (Geyer et al. 2017). In 2019 alone, 368 million metric tons (MMT) of plastic were produced. Plastics are utilized in a multitude of different end products, but most plastics (39.9%) are produced for the packaging industry (Plastics Europe 2020). While the material itself provides many benefits to society in the form of medical supplies, building materials, and prolonged shelf life of foods, the consequences of the persistent nature of this material are beginning to be realized.

As of 2015, 30% of all plastics produced (2.5 billion MT) remain in use. In contrast, a cumulative 6.3 billion MT of plastic waste was generated as of 2015, which is expected to increase to 12 billion MT by 2050 under business-as-usual (BAU) production and management methods (Geyer et al. 2017). In 2016 alone, approximately 2.01 billion MT of general waste was created globally with 241 MMT (12%) being composed of plastic waste (Kaza et al. 2018). As plastics have experienced a rapid and large growth in production, there has also been a simultaneous increase in plastics in the waste stream. In the USA for example, plastic made up 0.4% of plastic in the municipal solid waste (MSW) stream in 1960, but as of 2014 accounts for 13% of the MSW stream (Ryan 2015, US EPA 2019). In other regions of the world, plastics have been introduced to the waste stream more recently. Plastics and rubbers composed only 0.60% of the MSW stream in India in 1996, but as of 2011, plastics composed 10.1% of the national waste stream, representing a 21% increase in the plastic composition per year in that time frame (Joshi et al. 2016).

Highly populated regions such East Asia and the Pacific (EAS) and Sub-Saharan Africa (SSF) generate higher volumes of waste, however, per capita waste generation rates are greater in high income countries (HIC) and regions like North America and Europe. HIC regions have the largest plastic waste compositions at 12.5%, nearly double that of low-income countries (LIC) whose average plastic composition is 6.6% (Kaza et al. 2018). As economic development and urbanization increases, material consumption and waste generation become less associated with gross domestic product (GDP). By this trend, higher income nations and regions are expected to see more gradual growth in waste generation quantities in the coming decades, while more rapid growth is predicted in low and middle income countries in SSF and South Asia (Kaza et al. 2018).

Globally, only 11.4% of all waste is managed via controlled or sanitary landfills, while 25.2% is treated via unspecified landfills that may lack environmental controls, and 33% is still managed via open dumping. The remainder is treated via technologies such as recycling, incineration, composting, and other advanced methods like anaerobic digestion (Kaza et al. 2018). Of the plastic waste that is generated, only 9% is recycled globally, 12% is incinerated, and the remainder is either deposited in landfills or the natural environment (Geyer et al. 2017). HIC countries have the highest rate of recycling (29%), followed by lower-middle income countries (6%), and upper-middle income and LIC countries (both 4%; Kaza et al. 2018). Data availability specific to the recycling of plastics is limited, but Europe has the highest rate of plastics recycling (30%), followed by China (25%; Geyer et al. 2017). The USA has had a consistent rate of 9% plastics recycling since 2012 (US EPA 2014, Geyer et al. 2017), however, the more recent data from the US EPA indicates that it the national plastic recycling rate may have decreased to 8.7% (US EPA 2020). Often, plastics recycling does not occur domestically, particularly in Europe and North America, with many HIC countries exporting plastic scrap to developing economies located in EAS that may lack waste management infrastructure and so fate for recycling is not a guarantee (Brooks et al. 2018).

Recycling is typically limited to plastic materials that make economic sense to process into feedstock for production of products with recycled content. Because of the expenses associated with recovering waste plastics (i.e., collecting, sorting, cleaning, processing, transporting, etc.), it is often more cost effective for plastics producers to use virgin plastic resin as opposed to recycled resin (Moss 2017). Even in the informal waste sector, waste pickers will opt to pick those plastics that retain their value as waste such as polyethylene terephthalate (PET) bottles or polypropylene (PP) goods, as these items are in higher demand for recycling since they have greater potential for providing cost-competitive materials (Moss 2017). Complicating the recycling industry further are fluctuations in global markets like the petroleum industry as well as changing shipping costs and commodity prices (Hoornweg and Bhada-Tata 2012).

Between 60 and 99 MMT of plastic waste was mismanaged and available to enter global waterways in 2015 alone (Lebreton and Andrady 2019). The ocean may act as the final sink for this waste, but given the complexity of quantifying the distribution, fate, and accumulation of plastic pollution, the true quantities entering natural ecosystems are yet unknown. Given the nature of human consumption and waste management, most plastic waste is likely is generated on land, with an estimated annual input of land-based mismanaged plastic into the ocean ranging from 4.8 to 12.7 MMT (Jambeck et al. 2015). While coastal populations drive this number, inland populations can contribute to plastic waste inputs as well, with river emissions of plastic debris to the ocean are estimated to be between 0.4 and 4 MMT annually (Lebreton et al. 2017, Schmidt et al. 2017, Mai et al. 2020). More recent global riverine estimates find that 1,000 rivers, mostly small urban rivers, contribute to 80% of global mismanaged plastic waste emissions to the sea. Aside from riverine inputs, the four primary sources of ocean plastic pollution include municipal, land-based agriculture, maritime and ocean-based aquaculture, and industrial sources (Jambeck et al. 2020), and a recent estimate incorporating land-based, ocean-based, riverine, waste exports, and microplastic inputs reports a total of 15 MMT of annual plastic leakage into the ocean (Forrest et al. 2019).

Given that most of plastic is used for packaging, which is often designed for single use and immediate disposal, much of plastic waste that is generated and leaked into the environment is related to packaging. Unfortunately, many polymers and composite materials are used for specific food requirements such as moisture control for shelf stability, however, these properties reduce the value of the packaging once it becomes waste, resulting in less incentive for uptake and proper disposal (Moss 2017). Plastic packaging for food, beverage, and tobacco items, which is often designed specifically for single use, has contributed to 61% of global beach litter (Schweitzer et al. 2018), and a multitude of published formal litter monitoring and citizen science studies have found that plastics often make up the largest portion of littered waste (Hardesty et al. 2017, Nelms et al. 2017, Shayne 2018). The International Coastal Cleanup, which publishes a yearly report on litter items documented during the annual cleanup event held in over 100 participating countries, reports nearly every year that the top ten of the most reported litter items are single-use plastics (Ocean Conservancy 2019, Ocean Conservancy 2020).

Plastics can reach the environment through a range of sources on land and sea (Law 2017, Jambeck et al. 2020), and one considerable threat to the marine environment includes abandoned, lost, and discarded fishing gear, with at least 640,000 MT estimated to be lost at sea annually (Macfadyen et al. 2009). Once plastic is in the natural environment, it can pose risks to ecosystems, wildlife, humans, and even the economy (Jambeck et al. 2020). In nature, plastics are known to breakdown and fragment due to sunlight and mechanical abrasion into smaller and smaller pieces categorized as macroplastics (> 20mm in diameter), mesoplastics (5-20mm), microplastics (<5mm), and nanoplastics (<1µm; Napper and Thompson 2020). Evidence of the presence of both macro- and microplastics in many different settings have been documented through a multitude of scientific research. Plastics have been found on beaches all over the world and even in remote and uninhabited shorelines (Browne et al. 2011, Lavers and Bond 2017, Serra-Gonçalves et al. 2019, van Sebille et al. 2019). They have been found in sea ice (Bergmann and Klages 2012), the deepest ocean trenches (Jamieson et al. 2019), and the tallest mountains including Mount Everest (Napper et al. 2020). There is evidence that plastic may deposit on shorelines via sea breeze (Allen et al. 2020), and recent studies found microplastics present in Arctic snow (Bergmann et al. 2019), as well as rain, where an estimated 1,000 MT of plastic is deposited in US protected areas ever year via atmospheric precipitation (Brahney et al. 2020). All major oceans have documented presence of floating plastic debris (Eriksen et al. 2014) and it has been found in the Great Lakes (Eriksen et al. 2013). Growing evidence from the latter half of the past decade show that freshwater ecosystems and rivers may be hotspots for plastic pollution (van Emmerik and Schwarz 2020), and with historical focus on the marine environment, new research is beginning to show evidence and concern with terrestrial plastic (Horton and Dixon 2018).

Given the growing body of evidence of plastics accumulating in the natural environment, it is inevitable that the material may threaten wildlife and human health. One study has found that there are over 700 documentations of marine life encounters with plastic debris (Gall and Thompson 2015). The biggest threats to marine animals include entanglement, ingestion, and chemical contamination, and it is estimated that 99% of all seabird species may ingest plastics by 2050 without improved waste management systems for the material (Wilcox et al. 2015). Another recent study shows that ingestion of small plastic particles is likely a substantial risk to survival of endangered and threatened sea turtles (White et al. 2018). While wildlife interactions are a significant global concern, there is also the potential for threats to human health and livelihood. With evidence of microplastics in products ranging from bottled drinking water (Mason et al. 2018), beer (Liebezeit and Liebezeit 2014), and cosmetics (Napper et al. 2015), it is no surprise that the average American ingests between 39-52 thousand particles of microplastics annually (Cox et al. 2019) and that microplastics have been found in human waste (Schwabl et al. 2019). Exposure to high concentrations of plastic particles have been shown to cause physical and chemical toxicities under laboratory settings, however, it has not yet been concluded that concentrations in nature have reached the threshold for toxicity (SAPEA and Academies 2019). Indirect effects of mismanaged plastic and microplastics to humans include a threat to the food supply from mismanaged plastic like lost and abandoned fishing gear which may cause 5-30% reductions in some fish stocks (Global Ghost Gear Initiative 2018). Mismanaged plastic can also have economic implications given that 95% of plastic packaging material value is lost to the economy after its initial use (Ellen MacArthur Foundation 2017), and an estimated environmental cost of each MT of marine plastic ranging from $3,300 to $33,000 (Beaumont et al. 2019).

Considering the vast applications of plastic, the concurring plastic waste generation and treatment strategies, and its accumulation and persistence in nature, there are many places along the lifespan of plastics for implementing strategies for reducing downstream leakage into the environment that can be led by industry, governments, citizen engagement efforts, and global waste management sectors (Worm et al. 2017). Concepts like the circular economy and green engineering can improve resource efficiency, reduce dependence on virgin plastic production, increase the uptake of plastics in the waste stream, and divert plastic waste from landfills (Anastas and Zimmerman 2003, World Economic Forum et al. 2016). Both global and national policies related to plastic waste have increased in the past five years (Karasik et al. 2020) and more than 60 countries have introduced bag bans or levies as of 2018 (UNEP 2018a, UNEP 2018b). Policy efforts such as material bans and economic incentives like deposit schemes have shown demonstrated effectiveness by reduced consumption of banned products (UNEP 2018b), increased uptake of recyclable material (Vig and Kraft 2019), and reduction in littered plastic material on seafloors and beaches (Maes et al. 2018, Schuyler et al. 2018).

Knowledge regarding sources, drivers, distribution, fates, and impacts of plastic waste is growing but research in this field is still considered to be in its infancy and there are still many research questions that remain, particularly as the plastic waste landscape is rapidly changing under current global challenges such as climate change, increasing natural disasters, and COVID-19. By 2030, annual emissions of mismanaged plastic waste to aquatic environments may reach 53 MMT per year (Borrelle et al. 2020). Further, even if all feasible actions were taken immediately, an estimated 710 MMT of plastic waste will still leak into the environment by 2040 (Lau et al. 2020). Advancing knowledge is one way to inform strategies for reducing plastic production, accelerating innovation, improving waste management infrastructure, and strengthening disposal systems.

This work explores four topics under the umbrella of plastic waste management with the primary objective of informing global, regional, basin-scale, and community-based understanding of plastic material management. Chapter Two provides an updated and expanded analysis of the international plastic scrap trade including an assessment the global impacts of the Chinese import ban on plastic waste, building on previously published work by the author (Brooks et al. 2018). Chapter Three aims to generate comprehensive knowledge around plastic waste management in an understudied world region, Latin America and the Caribbean. This work applies a plastic intervention framework proposed by Jambeck (2016) to collect, synthesize, model, and forecast relevant information for improving management of plastics in the region. Chapter Four compares modeled and empirical estimates of land-based plastic debris for the Ganges River basin. Chapter Four is an experimental application of a rapid collection of field data for measuring and monitoring land-based plastic waste in communities.

While each section of this dissertation can stand alone as a separate research objective, they all connect in the context of scale. Most research in environmental plastic pollution ranges from small scale empirical surveys to large scale global models, but there is growing recognition for purposeful selection of appropriate landscape scales for measuring and modeling the ‘plastic cycle’ to better inform strategies for intervention (Hoellein and Rochman 2021). Taken together, these chapters serve as a multifaceted inquiry into plastic waste management in different geospatial contexts to do so.

REFERENCES

Allen, S., D. Allen, K. Moss, G. Le Roux, V. R. Phoenix and J. E. Sonke (2020). "Examination of the ocean as a source for atmospheric microplastics." PLoS ONE **15**(5): 1-14.

Anastas, P. and J. Zimmerman (2003). "Design through the 12 Principals of Green Engineering." Environmental Science & Technology.

Beaumont, N. J., M. Aanesen, M. C. Austen, T. Börger, J. R. Clark, M. Cole, T. Hooper, P. K. Lindeque, C. Pascoe and K. J. Wyles (2019). "Global ecological, social and economic impacts of marine plastic." Marine Pollution Bulletin **142**: 189-195.

Bergmann, M. and M. Klages (2012). "Increase of litter at the Arctic deep-sea observatory HAUSGARTEN." Marine Pollution Bulletin **64**(12): 2734-2741.

Bergmann, M., S. Mützel, S. Primpke, M. B. Tekman, J. Trachsel and G. Gerdts (2019). "White and wonderful? Microplastics prevail in snow from the Alps to the Arctic." Science Advances **5**(8): eaax1157.

Borrelle, S. B., J. Ringma, K. L. Law, C. C. Monnahan, L. Lebreton, A. McGivern, E. Murphy, J. Jambeck, G. H. Leonard, M. A. Hilleary, M. Eriksen, H. P. Possingham, H. De Frond, L. R. Gerber, B. Polidoro, A. Tahir, M. Bernard, N. Mallos, M. Barnes and C. M. Rochman (2020). "Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution." Science **369**(6510): 1515.

Brahney, J., M. Hallerud, E. Heim, M. Hahnenberger and S. Sukumaran (2020). "Plastic rain in protected areas of the United States." Science **368**(6496): 1257.

Brooks, A. L., S. Wang and J. R. Jambeck (2018). "The Chinese import ban and its impact on global plastic waste trade." Science Advances **4**(6).

Browne, M. A., P. Crump, S. J. Niven, E. Teuten, A. Tonkin and T. Galloway (2011). "Accumulation of microplastic on shorelines worldwide: sources and sinks." Environ Sci Technol **45**.

Cox, K. D., G. A. Covernton, H. L. Davies, J. F. Dower, F. Juanes and S. E. Dudas (2019). "Human Consumption of Microplastics." Environmental Science & Technology **53**(12): 7068-7074.

Ellen MacArthur Foundation. (2017). "The New Plastics Economy: Catalyzing Action." Retrieved January 23, 2017, from https://www.ellenmacarthurfoundation.org/assets/downloads/New-Plastics-Economy_Catalysing-Action_13-1-17.pdf.

Eriksen, M., L. C. M. Lebreton, H. S. Carson, M. Thiel, C. J. Moore, J. C. Borerro, F. Galgani, P. G. Ryan and J. Reisser (2014). "Plastic Pollution in the World's Oceans: More than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea." PLOS ONE **9**(12): e111913.

Eriksen, M., S. Mason, S. Wilson, C. Box, A. Zellers and W. Edwards (2013). "Microplastic pollution in the surface waters of the Laurentian Great Lakes." Mar Pollut Bull **77**.

Forrest, A., L. Giacovazzi, S. Dunlop, J. Reisser, D. Tickler, A. Jamieson and J. J. Meeuwig (2019). "Eliminating Plastic Pollution: How a Voluntary Contribution From Industry Will Drive the Circular Plastics Economy." Frontiers in Marine Science **6**(627).

Gall, S. C. and R. C. Thompson (2015). "The impact of debris on marine life." Marine pollution bulletin **92**(1-2): 170-179.

Geyer, R., J. R. Jambeck and K. L. Law (2017). "Production, use, and fate of all plastics ever made." Science Advances **3**(7): e1700782.

Global Ghost Gear Initiative (2018). Annual Report 2018.

Hardesty, B. D., T. Lawson, T. Velde, M. Lansdell and C. Wilcox (2017). "Estimating quantities and sources of marine debris at a continental scale." Frontiers in Ecology and the Environment **15**(1): 18-25.

Hoellein, T. J. and C. M. Rochman (2021). "The “plastic cycle”: a watershed-scale model of plastic pools and fluxes." **19**(3): 176-183.

Hoornweg, D. and P. Bhada-Tata (2012). "What a waste: a global review of solid waste management." Urban development series knowledge papers **15**: 1-98.

Horton, A. A. and S. J. Dixon (2018). Microplastics: An introduction to environmental transport processes. **5**.

Jambeck, J. R. (2016). Written Testimony. Hearing: Marine Debris and Wildlife: Impacts, Sources and Solutions. Washington, DC, U.S. SENATE COMMITTEE ON ENVIRONMENT AND PUBLIC WORKS.

Jambeck, J. R., R. Geyer, C. Wilcox, T. R. Siegler, M. Perryman and A. Andrady (2015). "Plastic waste inputs from land into the ocean." Science **347**.

Jambeck, J. R., E. Moss, B. Dubey, Z. Arifin, L. Godfrey, B. D. Hardesty, I. G. Hendrawan, T. T. Hien, L. Junguo, M. Matlock, S. Pahl, K. Raubenheimer, M. Thiel, R. Thompson and L. C. Woodall (2020). Leveraging Multi-Target Strategies to Address Plastic Pollution in the Context of an Already Stressed Ocean. H. L. P. f. a. S. O. Economy. Washington, DC, World Resources Institute.

Jamieson, A. J., L. S. R. Brooks, W. D. K. Reid, S. B. Piertney, B. E. Narayanaswamy and T. D. Linley (2019). "Microplastics and synthetic particles ingested by deep-sea amphipods in six of the deepest marine ecosystems on Earth." **6**(2): 180667.

Joshi, R., S. Ahmed and C. Ng (2016). "Status and challenges of municipal solid waste management in India: A review." Cogent Environmental Science **2**: 1139434.

Karasik, R., T. Vegh, Z. Diana, J. Bering, J. Caldas, A. Pickle, D. Rittschof and J. Virdin. (2020). 20 Years of Government Responses to the Global Plastic Pollution Problem: The Plastics Policy Inventory. . Durham, NC, Duke University.

Kaza, S., L. Yao, P. Bhada-Tata and F. Van Woerden (2018). What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050. Urban Development Series. Washington, DC, World Bank.

Lau, W. W. Y., Y. Shiran, R. M. Bailey, E. Cook, M. R. Stuchtey, J. Koskella, C. A. Velis, L. Godfrey, J. Boucher, M. B. Murphy, R. C. Thompson, E. Jankowska, A. Castillo Castillo, T. D. Pilditch, B. Dixon, L. Koerselman, E. Kosior, E. Favoino, J. Gutberlet, S. Baulch, M. E. Atreya, D. Fischer, K. K. He, M. M. Petit, U. R. Sumaila, E. Neil, M. V. Bernhofen, K. Lawrence and J. E. Palardy (2020). "Evaluating scenarios toward zero plastic pollution." Science: eaba9475.

Lavers, J. L. and A. L. Bond (2017). "Exceptional and rapid accumulation of anthropogenic debris on one of the world’s most remote and pristine islands." Proceedings of the National Academy of Sciences **114**(23): 6052-6055.

Law, K. (2017). "Plastics in the Marine Environment." Annual Review of Marine Science **9**: 205-229.

Lebreton, L. and A. Andrady (2019). "Future scenarios of global plastic waste generation and disposal." Palgrave Communications **5**(1).

Lebreton, L. C. M., J. van der Zwet, J.-W. Damsteeg, B. Slat, A. Andrady and J. Reisser (2017). "River plastic emissions to the world's oceans." Nature Communications **8**: 15611.

Liebezeit, G. and E. Liebezeit (2014). "Synthetic particles as contaminants in German beers." Food Additives & Contaminants: Part A **31**(9): 1574-1578.

Macfadyen, G., T. Huntington and R. Cappell (2009). "Abandoned, lost or otherwise discarded fishing gear." FAO Fisheries and Aquaculture Technical Paper(No.523): 115 pp.

Maes, T., J. Barry, H. A. Leslie, A. D. Vethaak, E. E. M. Nicolaus, R. J. Law, B. P. Lyons, R. Martinez, B. Harley and J. E. Thain (2018). "Below the surface: Twenty-five years of seafloor litter monitoring in coastal seas of North West Europe (1992–2017)." Science of The Total Environment **630**: 790-798.

Mai, L., X.-F. Sun, L.-L. Xia, L.-J. Bao, L.-Y. Liu and E. Y. Zeng (2020). "Global Riverine Plastic Outflows." Environmental Science & Technology.

Mason, S. A., V. G. Welch and J. Neratko (2018). "Synthetic Polymer Contamination in Bottled Water." Frontiers in chemistry **6**: 407-407.

Moss, E., Eidson, A., and Jambeck J. (2017). Sea of Opportunity: Supply Chain Investment Opportunities to Address Marine Plastic Pollution. New York, New York, Encourage Capital on behalf of Vulcan, Inc.

Napper, I. E., A. Bakir, S. J. Rowland and R. C. Thompson (2015). "Characterisation, quantity and sorptive properties of microplastics extracted from cosmetics." Mar Pollut Bull.

Napper, I. E., B. F. R. Davies, H. Clifford, S. Elvin, H. J. Koldewey, P. A. Mayewski, K. R. Miner, M. Potocki, A. C. Elmore, A. P. Gajurel and R. C. Thompson (2020).

"Reaching New Heights in Plastic Pollution—Preliminary Findings of Microplastics on Mount Everest." One Earth **3**(5): 621-630.

Napper, I. E. and R. C. Thompson (2020). "Plastic Debris in the Marine Environment: History and Future Challenges." Global Challenges **4**(6): 1900081.

Nelms, S. E., C. Coombes, L. C. Foster, T. S. Galloway, B. J. Godley, P. K. Lindeque and M. J. Witt (2017). "Marine anthropogenic litter on British beaches: A 10-year nationwide assessment using citizen science data." Science of The Total Environment **579**: 1399-1409.

Ocean Conservancy (2019). The Beach and Beyond: International Coastal Cleanup 2019 Report.

Ocean Conservancy (2020). Together, We Are Team Ocean - 2020 Report. International Coastal Cleanup.

Plastics Europe (2020). "Plastics the Facts - 2020."

Ryan, P. (2015). A Brief History of Marine Litter Research. Marine Anthropogenic Litter. Bergmann et al. (eds.), Springer, New York, NY.

SAPEA and S. A. f. P. b. E. Academies (2019). A Scientific Perspective on Microplastics in Nature and Society. Berlin: SAPEA.

Schmidt, C., T. Krauth and S. Wagner (2017). "Export of Plastic Debris by Rivers into the Sea." Environmental Science & Technology **51**(21): 12246-12253.

Schuyler, Q., B. D. Hardesty, T. J. Lawson, K. Opie and C. Wilcox (2018). "Economic incentives reduce plastic inputs to the ocean." Marine Policy **96**: 250-255.

Schwabl, P., S. Köppel, P. Königshofer, T. Bucsics, M. Trauner, T. Reiberger and B. Liebmann (2019). "Detection of Various Microplastics in Human Stool." Annals of Internal Medicine **171**(7): 453-457.

Schweitzer, J.-P., S. Gionfra, M. Pantzar, D. Mottershead, E. Watkins, F. Petsinaris, P. ten Brink, E. Ptak, C. Lacey and C. Janssens (2018). Unwrapped: How throwaway plastics is failing to solve Europe's food waste problem (and what we need to do instead), Institute for European Environmental Policy (IEEP).

Serra-Gonçalves, C., J. L. Lavers and A. L. Bond (2019). "Global Review of Beach Debris Monitoring and Future Recommendations." Environmental Science & Technology **53**(21): 12158-12167.

Shayne, K. E. (2018). PARTICIPATORY SENSING MARINE DEBRIS: ESTIMATES FOR AMOUNTS, TYPES, AND DISTIBUTION USING MARINE DEBRIS TRAKCER DATA, University of Georgia.

UNEP (2018a). "Legal Limits on Single-Use Plastics and Microplastics: A Global Review of National Laws and Regulations."

UNEP (2018b). Single-Use Plastics: A Roadmap for Sustainability.

US EPA (2014). Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Tables and Figures for 2012. O. o. R. C. a. Recovery. Washington, DC.

US EPA (2019). Advancing Sustainable Materials Management: 2017 Fact Sheet Assessing Trends in Material Generation, Recycling, Composting, Combustion with Energy Recovery and Landfilling in the United States. Washington, DC.

US EPA (2020). Advancing Sustainable Materials Management: 2017 Fact Sheet Assessing Trends in Material Generation, Recycling, Composting, Combustion with Energy Recovery and Landfilling in the United States. Washington, DC.

van Emmerik, T. and A. Schwarz (2020). "Plastic debris in rivers." WIREs Water **7**(1): e1398.

van Sebille, E., P. Delandmeter, J. Schofield, B. D. Hardesty, J. Jones and A. Donnelly (2019). "Basin-scale sources and pathways of microplastic that ends up in the Galápagos Archipelago." Ocean Sci. **15**(5): 1341-1349.

Vig, N. J. and M. E. Kraft (2019). Environmental Policy: New Directions for the Twenty-First Century. Los Angeles, CQ Press.

White, E. M., S. Clark, C. A. Manire, B. Crawford, S. Wang, J. Locklin and B. W. Ritchie (2018). "Ingested Micronizing Plastic Particle Compositions and Size Distributions within Stranded Post-Hatchling Sea Turtles." Environmental Science & Technology **52**(18): 10307-10316.

Wilcox, C., E. Van Sebille and B. D. Hardesty (2015). "Threat of plastic pollution to seabirds is global, pervasive, and increasing." Proc Natl Acad Sci U S A **112**(38): 11899-11904.

World Economic Forum, Ellen MacArthur Foundation and M. Company. (2016). "The new plastics economy: rethinking the future of plastics." from https://www.ellenmacarthurfoundation.org/publications/the-new-plastics-economy-rethinking-the-future-of-plastics.

Worm, B., H. K. Lotze, I. Jubinville, C. Wilcox and J. Jambeck (2017). "Plastic as a Persistent Marine Pollutant." Annual Review of Environment and Resources **42**(1): 1-26.

CHAPTER 2

AN UPDATED ASSESSMENT OF THE INTERNATIONAL PLASTIC SCRAP TRADE AND THE GLOBAL IMPACT OF THE CHINESE IMPORT BAN[1]

[1] Amy L. Brooks & Jenna R. Jambeck. To be submitted to *Resources, Conservation, and Recycling.*

2.0 ABSTRACT

Despite the global plastic recycling rate of just 9%, the international plastic scrap trade has rapidly developed into a complex system for global management of plastic waste in only a few decades. However, the status quo was interrupted in 2018 when a Chinese ban waste imports was implemented, displacing the world's plastic scrap. While long-term impacts of the ban are uncertain, near term impacts may reveal emerging trends related to displaced plastic scrap resulting from the ban. Here, trade data from 1988 to 2019 is examined to provide an update to previously published research by Brooks et al. (2018) to provide insight regarding early impacts of the ban. The analysis revealed that annual imports and exports of plastic scrap have decreased 59% and 48%, respectively, between pre- and post-ban conditions. Given a nearly 100% enforcement rate of the ban, China's rank in global imports has reduced from 1st in the world in 2016 to 40th in 2019, having only imported 924 metric tons (MT) that year. As of 2019, 15.8 million metric tons (MMT) of plastic scrap have been displaced globally, with the largest percent regional increase in imports seen in Sub-Saharan Africa and the Middle East and North Africa. Early indications show a positive environmental impact in the context of global mismanaged plastic waste quantities that are attributable to plastic scrap trade, as an estimated 4.52 MMT of plastic scrap were inadequately managed in 2016, compared to only 921 MT in 2019. That said, these changes are only early indications of the ban, and recent amendments to the Basel Convention in place as of January 2021 now regulate the international trade of plastic scrap, which could result in compounding impacts to the global system. As such, continued monitoring of the issue will be needed to identify trends and enact early responses to potential environmental impacts.

2.1 INTRODUCTION

While only 9% of plastic scrap is recycled globally (Geyer et al. 2017), in just three decades, the global plastic recycling industry has become a complex and globalized waste management system. While national and sub-national recycling efforts vary from place to place (Kaza et al. 2018), recent attention has been turned to the international plastic scrap trade following reports of concerning amounts of plastic waste leaking into the environment (Jambeck et al. 2015, Lebreton et al. 2017). Of the potentially 15 million metric tons (MMT) of plastic waste that enters the ocean annually, an estimated 804,000 metric tons (MT) may be attributable to the plastic scrap trade.

Exports of plastic scrap have generally been led by high income countries (HIC) and members of the Organization for Economic Cooperative Development (OECD) in North America (NAC) and Europe and Central Asia (ECS; Brooks et al. 2018), despite many of these countries having highly developed or sufficient access to domestic waste management infrastructure. As a result, global mismanaged plastic waste quantities my not fully account for the contribution of developed countries and regions that rely on exporting. For example, one estimate from 2017 found that up to 7.3% of European post-consumer high density polyethylene (HDPE) destined for recycling may have been lost to the environment as a result of being exported abroad (Bishop et al. 2020). Similarly, in 2016, 88% of US exports were destined for countries in which there were known challenges with adequate management of waste (Brooks et al. 2018), however, the US has historically been characterized by high rates of waste collection (Kaza et al. 2018), and only a 2% rate of mismanagement driven by littering (Kaza et al. 2018). However, recent research has updated the US contribution to global quantities of mismanaged plastic waste to account for waste that is exported from the country such that an estimated 0.15-0.99 MMT of US plastic waste was exported and potentially mismanaged among importing countries in 2016, making the US the third highest generator of mismanaged plastic waste (Law 2020).

While wealthy NAC and ECS countries have contributed significantly to global trade of plastic scrap (Pacini et al. 2021), China, an upper-middle income country with the second largest gross domestic product (GDP) in the world, has also played a critical role in the international plastic scrap trade historically, having imported a cumulative 45% of all plastic scrap as of 2016 (Brooks et al. 2018). As the leading producer of plastics (Plastics Europe 2020), it has been suggested that imports of plastic waste into the country have been driven by China's domestic consumption of products containing recycled plastic (Huang et al. 2020). Regardless, one estimate determined that 10-13% of plastic waste by mass has required management in addition to domestically generated plastic waste (Brooks et al. 2018), and with an estimated 23% of waste in the country being inadequately managed (Law 2020), it is likely that plastic imports may be inadequately disposed.

In 2018, this enmeshed global recycling system experienced a major disruption when China, the leading importer of plastic scrap, implemented a permanent ban on imports of 24 waste materials including plastic scrap. While this policy was motivated by desire to advance sustainable domestic development and reduce reliance on foreign raw material (Qu et al. 2019), the ban may displace an estimated 111 MMT of plastic scrap by 2030 (Brooks et al. 2018). Early impacts of the ban suggest that domestic management of plastic waste in many exporting countries have seen significant positive and negative effects. Japan, for example, expects to see the advancement of thermal recycling of low-grade plastic scrap domestically due to the ban, and national policy recommendations include more comprehensive and accurate data collection on exports of plastic scrap routes to better inform domestic capabilities for collection and sorting of waste recycling (Morita and Hayashi 2018). In other cases where development of domestic recycling is not feasible or a priority, the constraining impact of the Chinese ban may now be motivation for the reduction of domestic use of plastic, improvement of waste collection systems, and redesigning of plastic packaging and products such that they retain more value in domestic recycling streams. For example, the European Union is working to standardize guidance for collection and sorting of waste as part of the European Strategy for Plastics in a Circular Economy and is developing the European Committee of Standardization to specifically promote minimum quality standards for sorted plastic waste and recycled plastics (European Commission 2018). In contrast, some preliminary reports show impacts from the ban in other regions may have exacerbated challenges with plastic waste management. For example, the ban has caused global recycling routes to shift to other countries in the Southeast Asian region are seeing that do not necessarily have adequate waste management infrastructure for the increased quantities of waste material are (He et al. 2018, Lim 2018, Wang et al. 2020), but there is likely no country or region in the world that has the capacity to absorb the plastic scrap displaced by the ban (Lim 2018, Huang et al. 2020).

While the global industry was certainly impacted by the restrictions, significant impacts have been documented in China. For example, with reduced imports of plastic scrap from foreign countries due to the ban, there is in turn increasing reliance on domestic plastic waste as feedstock for recycled plastic products and, in turn, may be leading to more efficient solid waste management in the country (Qu et al. 2019). However, the shift to more efficient systems comes with a cost, as some evidence indicates that China may have seen the largest value-added loss (78.4%) resulting from the import ban (Huang et al. 2020). Further, because of the advancement of international regulations, China has also had a spate of recycling and port facility removals from permissible facility lists (Raubenheimer and McIlgorm 2018), which could disproportionately impact members of the plastic recycling workforce who are of low socioeconomic status.

The impacts of the ban are wide-ranging and with the impact only having been in place for three years, widespread reactions and global effects on plastic waste management are still developing. Despite the relative infancy of the ban status, systematic monitoring may help to understand how impacts are developing and provide insight into what to expect in coming years. Some studies have assessed impacts of the ban; however, no studies have accounted for changes seen in 2019 in relation to the global pre-ban conditions. As such, the primary objective of this research is to provide a detailed assessment of the status of the international plastic scrap trade through 2019. This is accomplished through 1) expansion on methods used by Brooks et al. (2018) to provide an up-to-date summarization of the cumulative plastic scrap trade as of 2019, and 2) assessment of the impact of the Chinese ban on the global plastic scrap trade including differences in trade quantities, sources and destinations, displacement of plastic scrap by the ban, and potential environmental impacts. Thirty-one years of global trade data were quantitatively summarized over time, geospatially, and by economic groupings.

Differences between pre- and post-ban conditions are determined via comparison testing at various levels of aggregated trade data. Through these approaches, countries and groups that have had significant participation in the global plastic scrap trade and/or have been significantly impacted by the Chinese import ban are identified. The resulting findings establish the conditions of the plastic scrap trade as of 2019 and early impacts of the ban, which may tentatively inform future developments and long-term impacts.

2.2 MATERIALS AND METHODS

2.2.1 Data sources and preparation

Data for internationally traded commodities, including plastic scrap, were accessed via the UN Comtrade Database (https://comtrade.un.org/; accessed 20 February 2021). Data were aggregated across four categories of plastic scrap, denoted by the Harmonized Commodity Description and Coding Systems (HS; Table 2.1). The trade data include trade quantities, corresponding trade values in current USD, year of reporting, reporting country, territory, or economy (hereinafter referred to as 'country' or 'reporter' unless otherwise necessary for context), polymer, and trade flow (i.e., import and export flows). For analysis of specific countries, data were downloaded for individual reporters, which provided data for source and destination locations.

Table 2.1. Summary of HS codes for international plastic scrap commodities

Commodity	HS commodity code	Abbreviation
Waste, parings, and scraps of plastics of	*3915*	
Ethylene polymers	.10	PE
Styrene polymers	.20	PS
Vinyl chloride polymers	.30	PVC
Plastics, nes*	.90	Other plastics

* nes, Not elsewhere specified

The countries were sorted by World Bank region and lending groups to examine whether there were differences between world regions defined by the The World Bank (2020) as follows: East Asia and Pacific (EAS), Europe and Central Asia (ECS), Latin America and the Caribbean (LCN), Middle East and North Africa (MEA), North America (NAC), South Asia (SAS), and Sub-Saharan Africa (SSF). Similarly, data were collated by four economic classifications defined by the The World Bank (2020) as follows: low-income (LIC), lower-middle income (LMC), upper-middle income (UMC), and high-income (HIC). The World Bank classifications were chosen for alignment with previous research related to plastic waste management (Jambeck et al. 2015, Law 2020) and the plastic scrap trade (Brooks et al. 2018). Additionally, the original trade quantities were reported in kilograms, which were converted to metric tons for this analysis. Lastly, reporters were categorized by membership in the Organization for Economic Cooperative Development (OECD), most of which are wealthy countries that have experienced a decoupling of economic growth from waste generation and increasing rates of recycling and recovery of waste (OECD 2019).

2.2.2 Update on the trade of plastic scrap (1988 – 2019)

Temporal analysis

Data from 1988 to 2019 were compared from multiple perspectives including temporal (i.e., annual and cumulative quantities), geospatial (i.e., by country and World bank region) and by economic classifications (i.e., by World Bank income groups and OECD membership). For temporal analysis, data were aggregated by year so that annual trade quantities and values per metric ton could be plotted over time and assessed visually to determine stationarity of the time series. Regardless of the observed stationarity of the data, the distribution of annual quantities for each trade flow were visually assessed to determine normality via histograms and the data were summarized with descriptive statistics including the range, mean, and median. Cumulative trade quantities were calculated as of 2019 and similarly plotted over time to aid visualization of growth since 1988.

Spatial analysis

Reporters were ranked and major contributors were identified in terms of traded quantities of plastic scrap. Data were aggregated by the associated world regions and quantitatively summarized. Similarly, the regions were ranked by contribution to the global trade quantities. Differences between regions were compared using the Kruskal-Wallis (K-W) test, a rank-based, non-parametric analog to analysis of variance (ANOVA) and determines the equality of median variable distributions across two or more categorical and independent groups. In this case, the mass of imports and exports were compared over the nominal region categories. The null hypothesis of the K-W test states that the groups' medians are the same and was rejected for $p \leq .05$. Post hoc pairwise comparisons were conducted using the Dunn (1964) procedure with a Bonferroni correction to reduce propagation of Type I errors over multiple comparisons. All statistical tests were run in either Microsoft Excel or Stata 16.1. Maps were created in ArcMap 10.8 to aid in visualization of the results at the global scale.

Analysis by economic classification

Analysis of trade quantities across economic categories was conducted in a similar manner to that of the spatial analysis described above. Reporters were aggregated by World Bank income groups and OECD membership for separate analysis. The K-W test was similarly used to assess differences between economic groups, while differences between OECD members and non-members were determined by the Mann-Whitney U test, which is a rank-based, non-parametric alternative to the independent-samples t-test that compares median differences between two unrelated groups. The null hypothesis states that the distributions in each group are equal and was rejected for $p < .05$.

2.2.3 Impact of the ban (2016 vs. 2019)

Temporal, spatial, and economic comparisons

To examine the impact of the Chinese ban, trade data by reporter were isolated for 2016 and 2019 to represent pre- and post-ban differences in quantities traded, respectively. These years were selected for comparison because data reported for 2016 represented trade prior to the announcement of the ban 2017, which had a clear impact before the ban was implemented in 2018 (Brooks et al. 2018), while data from 2019 reflected the most recent year with presumably whole reported trade values. The distribution of annual trade quantities by country for each year were visually assessed to determine normality and descriptive statistics were generated. Here, temporal analysis primarily encompassed comparison of annual and cumulative quantities were compared between 2016 and 2019 and determination of total absolute and relative change of the total quantities that were imported and exported in each year.

Differences across countries, regions, income groups, and OECD membership were determined by calculating the absolute and relative changes seen in reported traded quantities as well as the proportion of total trade quantities between 2016 and 2019. The greatest increases and decreases in trade quantities between years were identified. Further, the top ten countries in terms of traded quantities were also compared between the countries to identify any changes in major contributors. Only countries that reported values of trade were included in the change calculations (i.e., blank entries were removed). Countries that reported zero metric tons were included since these were considered valid entries. Finally, Wilcoxon signed rank tests were used to assess differences between the two years based on quantities traded by each reporter, region, income group, and OECD membership classification. The Wilcoxon signed rank test is a rank-based, non-parametric test that determines the equality of distributions between two paired variables. The null hypothesis, which states that the distribution of the dependent variable is the same across the categorical variable, in this case defined by pre-ban (i.e., 2016) and post-ban (i.e., 2019), was rejected when $p < .05$. Maps were generated in ArcMap 10.8 to facilitate visualization of changes in quantities due to the ban.

Country briefs

Previous research has identified major historical contributors of the global plastic scrap trade (Brooks et al. 2018), which informed the development of country briefs to better quantify impacts the key historical participants of the global plastic scrap trade including impacts to Chinese imports and impacts to exports from USA, Japan, and Germany. Notably, Hong Kong has historically contributed some of the largest quantities traded plastic scrap, however, as a Special Administrative Region (SAR) of China, it has largely acted as an entry port to China, with at least 63% of Hong Kong imports in 2016 having been exported to mainland China (Brooks et al. 2018). For this reason, Hong Kong was not examined individually, but was included in the assessment of Chinese imports. In addition to the historical participants in the global plastic scrap trade, the results from the spatial analysis described in the previous section also informed the inclusion of briefs for three key countries of interest: Malaysia, Indonesia, and Vietnam, which have experienced increasing import quantities that are likely the result of the ban. The country briefs were informed by trade data for quantities of plastic scrap traded between each key country and its corresponding global partners in 2016 and 2019. Data were processed as described in Section 2.2.1 such that partner countries were classified by region, economic classification, and OECD membership. Differences in overall quantities and spatial and economic patterns were determined, and visualization of differences was supported through geospatial maps generated in ArcMap 10.8.

Displaced plastic waste

Brooks et al. (2018) previously estimated the quantity of plastic scrap that would be displaced by the Chinese import ban based on simple linear regression of cumulative Chinese imports. The annual quantities estimated by this model were compared to the actual reported values for each year since the ban (i.e., 2017, 2018, and 2019) to assess how the ban has progressed since being implemented. Brooks et al. (2018) also provided implementation scenarios for the ban based on five levels of implementation where 0% would represent no enforcement of the ban and 100% represented exhaustive enforcement of the ban. Within this range, 25%, 50%, and 75% implementation scenarios were provided, and the quantities imported by China were used to assess what level of enforcement is occurring as of 2019.

Displaced plastic waste was also evaluated for the top three historically exporting countries, excluding Hong Kong, to assess how much the ban may have displaced from these historical trade partners. Cumulative tonnage of Chinese imports for each of the three countries were plotted and based on methods used by Brooks et al. (2018) for estimating displaced plastic waste, simple linear regression was used to model the imports from 2007 – 2017 to forecast the cumulative quantity of plastic scrap that would have been imported into China from each country by 2030 had the ban not been implemented. Total displaced plastic waste as of 2030 was then calculated for each country by taking the theoretical cumulative values at year 2030 and subtracting the actual cumulative values that were imported as of 2019. Similarly, to estimate the current quantities of displaced plastic waste between 2018 and 2019, the actual reported imports from 2019 were subtracted from the forecasted quantities for 2019.

Impacts to global mismanaged plastic scrap

Jambeck et al. (2015) defined mismanaged plastic as the sum of quantities of littered and inadequately managed plastic waste. Inadequately managed waste encompasses waste disposal methods such as open dumping, burning, burying, and deposition in waterways. Quantities of inadequately managed plastic scrap were estimated for 2016 and 2019 to determine if the ban had any effect on the quantities of mismanaged plastic waste in key importing countries identified in the spatial comparison as well as regions and income groups. were sourced from Law (2020). For individual countries, the inadequately managed fraction was assigned directly. For assessment of changes in mismanaged plastic waste by region and income group, the inadequately managed fraction was determined based on paired combinations of region and income categories, such that the inadequately managed fraction was averaged across countries that were categorized by the region-income pairs. Then, the inadequate fraction was applied to each paired groupings' imported tonnage in 2016 and 2019. Finally, the absolute and relative difference in quantities of mismanaged plastic scrap were calculated between the years was determined. Percent change was calculated by taking the difference in the total inadequately managed quantities between the two years divided by the total inadequately managed quantity from 2016 for each region-income group.

2.3 RESULTS

2.3.1 Update on trade of plastic scrap (1988 – 2019)

Temporal analysis

For the temporal analysis, data were aggregated by country for 32 years. Visual assessment of histograms for annual quantities of globally traded plastic scrap indicated that the data was non-normally distributed. Imports ranged from 145,000 metric tons in 1988 to 16.5 MMT in 2010, with the median quantity of imports being 6.62 MMT (Table 2.2). Exports ranged from 170,000 metric tons in 1988 to 15.7 MMT in 2010, with the mean being 6.30 MMT (Table 2.2). Trade quantities peaked in the 2010s and have since returned to levels similar to traded quantities seen in the early 2000s (Figure 2.1). Imports ranged from \$76.7 million USD in 1988 to \$9.99 billion USD in 2011, with a median value of \$3.83 billion USD (Table 2.2). Trade value of exports ranged from \$72.4 million USD in 1988 to \$7.22 billion USD in 2011, with a median value of \$2.23 billion USD (Table 2.2). Taking trade value per metric ton, however, shows that the minimum value per USD of exports ranged from \$259 per metric ton in 2004 to \$509 USD per metric ton in 2013, with a median \$399 USD per metric ton (Table 2.2). Imports similarly ranged from \$257 in 2002 to \$646 in 2013, with a median value of \$428 USD per metric ton (Table 2.2).

Table 2.2. Summary statistics for annual quantities and value of trade plastic scrap from 1988 to 2019

Measure	*n*	*m*	*sd*	*mdn*	*min*[a]	*max*
Trade flow (MMT)						
Imports	32	8.17	5.94	6.62	0.145	16.5
Exports	32	7.44	5.53	6.30	0.170	15.7
Trade value (billion USD)						
Imports	32	3.83	3.43	2.40	0.077	9.99
Exports	32	3.03	2.47	2.23	0.072	7.22
USD per metric ton						
Imports	32	436	112	428	257	646
Exports	32	395	63.7	399	259	509

Visual assessment of the annual trade quantities and values per metric tons from 1988 to 2019 indicated that the trade quantities and values were non-stationary over time given the irregular annual values for traded plastic scrap (Figure 2.1). Stationarity is met when the average and spread of values grows at a constant rate over time, but when this condition is not met in time series data, it can be difficult to deriving meaningful conclusions or inferences, and so the quantitative summary visual interpretation of patterns in trade over time is provided here. As of 2019, a cumulative 238 MMT and 262 MMT have been exported and imported since 1988, respectively, which is equivalent to

$97.6 and $123 billion USD (current value). As shown in Figure 2.2, quantities and values of traded plastic scrap have begun to plateau as of 2019.

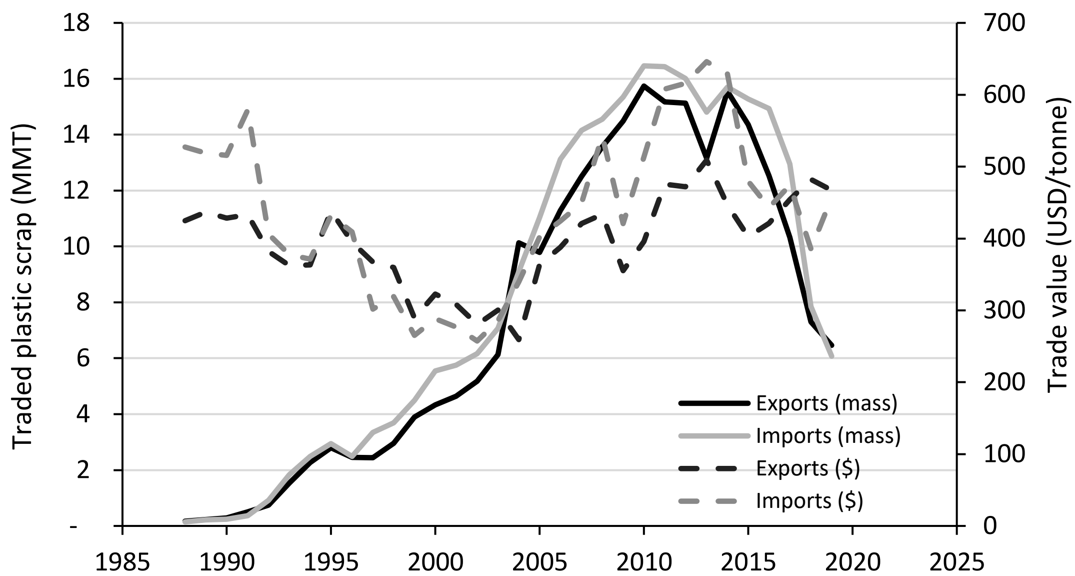

Figure 2.1. Annual trade quantities in MMT and trade value in USD per metric ton from 1988 to 2019.

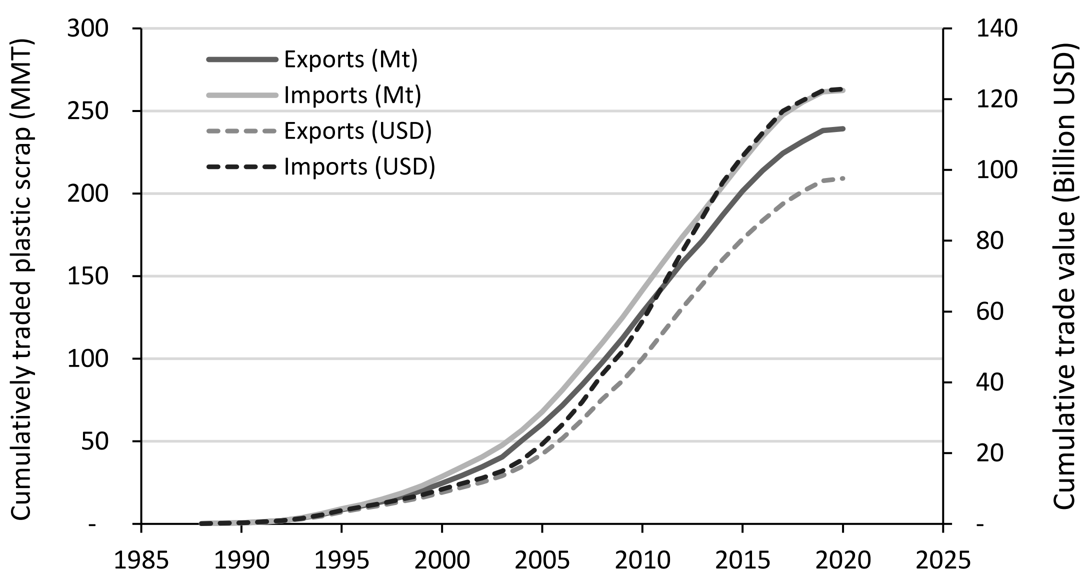

Figure 2.2. Cumulative trade quantities and values from 1988 to 2019.

Spatial analysis

A total of 194 countries and territories reported some combination of trade values from 1988 to 2019 and were included in the overall analysis. Excluding blank entries, 192 reported values for imports and 179 reported values for exports. Visual assessment determined that trade quantities by countries from 1988 to 2019 were non-normally distributed and extremely right skewed. Although parametric tests can be appropriate for non-normally distributed data with large sample sizes, non-parametric tests were chosen due to the severity of the skewing. Likewise, medians are reported as the primary measure of central tendency, however means are provided in Table 2.3.

Between 1988 and 2019, cumulative imports by country ranged from 0.01 metric tons in Bermuda to 112 MMT in China, with the median quantity of imports being 12,400 metric tons. Reported exports ranged from 0 metric tons reported by both Djibouti and Montserrat to 58.4 MMT in Hong Kong, with a median cumulative quantity of exports of 50,900 metric tons (Table 2.3).

Table 2.3. Summary statistics for quantity of traded plastic scrap by reporting country or territory from 1988 to 2019 in MMT

Measure	*n*	*m*	*sd*	*mdn*	*min*	*max*
Trade flow						
Imports	193	1.34	9.46	0.012	0.005	112
Exports	180	1.32	5.54	0.051	0.001	58.4

As of 2019, China has contributed to 43% of all imports, followed by Hong Kong (26%), and USA (3.5%). Hong Kong has contributed most (25%) to exports as of 2019, followed by USA and Japan (11%), and Germany (9.1%). The top ten countries by each trade flow have contributed to 86% and 77% of all imports and exports, respectively. Six of these, Hong Kong, USA, Netherlands, Germany, Belgium, and Canada, are represented as major contributors for both imports and exports. The combined contribution of the top ten reporters was higher than annual proportions in recent years (i.e., 2016 and 2019). If taken collectively, the EU-28 would rank 2nd in both cumulative imports and exports from 1988 to 2019, representing 33% and 16% of the total traded quantities, respectively.

Table 2.4 Top ten contributing countries and territories by proportion of trade quantities from 1988 to 2019

Rank	*Imports*		*Exports*	
	Reporter	Proportion (%)	Reporter	Proportion (%)
1	China	43	Hong Kong	25
2	Hong Kong	26	USA	11
3	USA	3.8	Japan	11
4	Netherlands	2.7	Germany	9.1
5	Germany	2.6	Mexico	4.8
6	Belgium	2.0	UK	4.2
7	Canada	1.6	Netherlands	3.6
8	Malaysia	1.5	France	3.6
9	India	1.4	Belgium	3.4
10	Italy	1.4	Canada	1.8
Total		86		77

Across the seven World Bank regions, imports ranged from 654,000 metric tons in SSF to 190 MMT in EAS (*mdn* = 3.84). Similarly, exports ranged from 647,000 metric tons in SSF to 105 MMT in EAS (*mdn* = 8.36). Taken together, the EAS, ECS, and NAC regions have contributed most to both historical imports and exports from 1988 to 2019 (91% and 84%, respectively). EAS has historically contributed most to both imports and exports (73% and 44%, respectively). This is followed by ECS which has contributed to 17% and 34% of imports and exports, respectively, and NAC, which has contributed 5.4% and 13%, respectively. EAS and ECS have together contributed to 90% and 78%, respectively. In contrast, SSF has contributed the least to both imports and exports at 0.3% for each.

The K-W test for both imports and exports showed that there were statistically significant differences between each region's quantities of traded plastic scrap between 1988 and 2019 ($\chi^2(6) = 35.618$, $p = .0001$ for imports, and $\chi^2(6) = 44.118$, $p = .0001$ for exports). Subsequently, post hoc pairwise comparisons revealed ECS significantly differed from EAS, LCN, and SSF for imported plastic scrap, and there were similar significant differences in exported plastic scrap quantities between ECS and LCN and between SSF and EAS, ECS, and MEA (Table 2.6).

Table 2.5 Ranking of regional contribution by proportion of trade quantities from 1988 to 2019

Rank	Imports		Exports	
	Region	Proportion (%)	Region	Proportion (%)
1	EAS	73	EAS	44
2	ECS	17	ECS	34
3	NAC	5.4	NAC	13
4	SAS	1.6	LCN	5.8
5	LCN	0.8	MEA	1.2
6	MEA	0.4	SAS	0.4
7	SSF	0.3	SSF	0.3

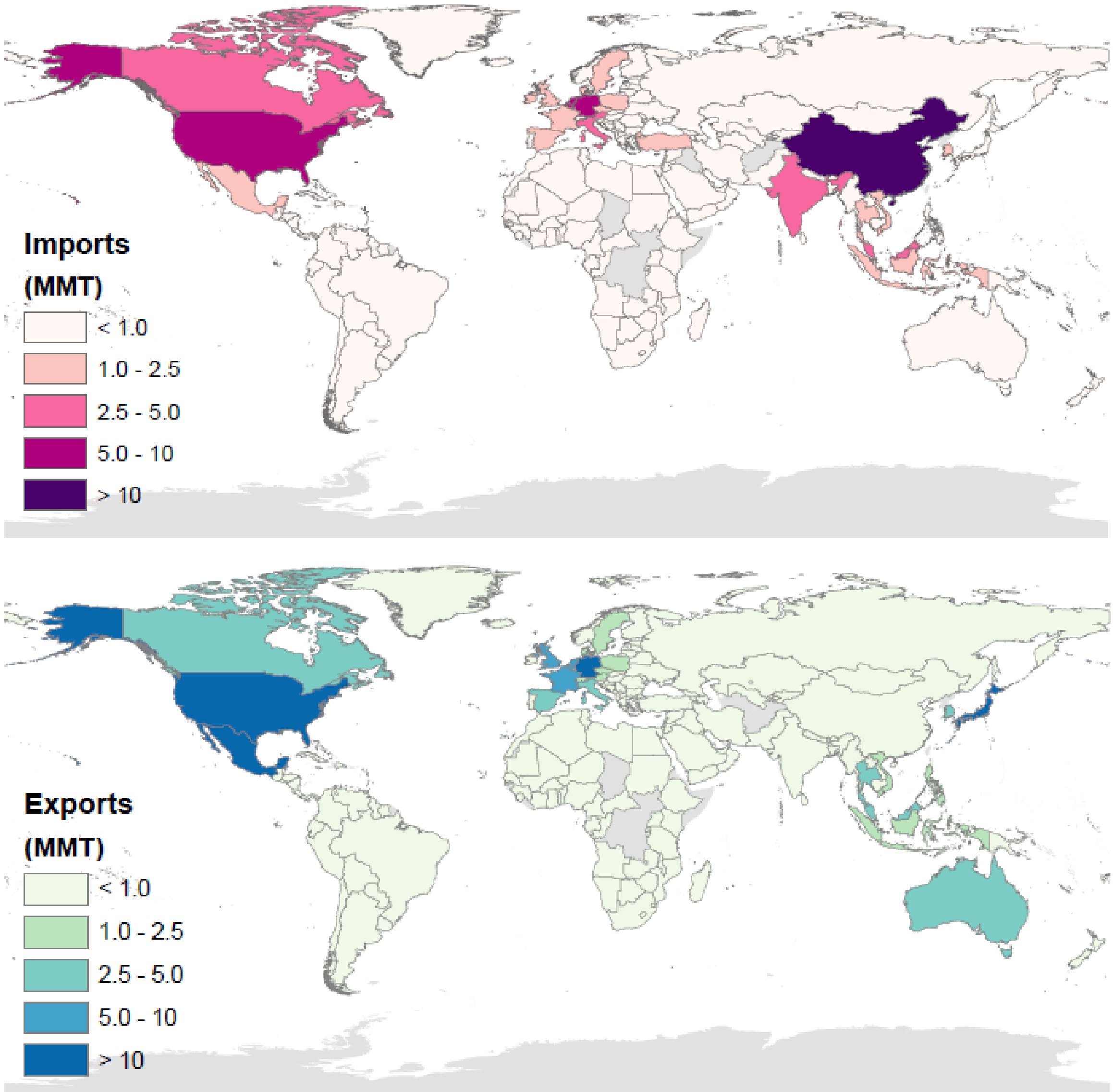

Figure 2.3. Cumulative imports and exports by country from 1988 to 2019

Table 2.6 Post hoc comparisons of regional median import and export trade quantities from 1988 to 2019 using Dunn's procedure. Median trade quantities reported in MMT.

Imports	*n*	*Mdn (MMT)*	EAS	ECS	LCN	MEA	NAC	SAS	SSF
EAS	31	0.003	1						
ECS	50	0.207	-3.00*	1					
LCN	38	0.005	1.40	4.76***	1				
MEA	20	0.032	-0.57	1.97	-1.82	1			
NAC	3	4.27	-1.15	-0.02	-1.72	-0.86	1		
SAS	7	0.041	-0.16	1.53	-0.99	0.22	0.92	1	
SSF	43	0.002	1.63***	5.14	0.20	2.02	1.81	1.11	1
Exports	*n*	*Mdn (MMT)*	EAS	ECS	LCN	MEA	NAC	SAS	SSF
EAS	26	0.084	1						
ECS	49	0.248	-1.37	1					
LCN	36	0.023	2.25	4.15***	1				
MEA	28	0.079	0.30	1.59	-1.76	1			
NAC	3	4.38	-0.42	0.13	-1.39	-0.56	1		
SAS	7	0.014	0.72	1.58	-0.66	0.49	0.81	1	
SSF	38	0.004	3.88**	6.11***	1.76	3.25*	2.07	1.66	1

$^{*}p < .05$, $^{**}p < .01$, $^{***}p < .001$

Analysis by economic classification

Across the four World Bank economic classifications, imports ranged from 215,000 metric tons in LIC to 122 MMT in UMC (*mdn* = 65.5), while exports ranged from 488,000 metric tons in LIC to 205 MMT in HIC (*mdn* = 14.7). HIC and UMC have contributed the most (95%) historical imports (49% and 47%, respectively), while exports have been heavily driven by HIC, which has contributed to 86% of all exports followed by UMC (9.1%). In contrast, the LMC and LIC economies have together contributed to 3.4% of both imports and exports (Table 2.7).

Table 2.7. Ranking of economic groups by contribution by proportion of trade quantities from 1988 to 2019

Rank	Region	Proportion (%)	
		Imports	Exports
1	HIC	49	86
2	UMC	47	9.1
3	LMC	3.3	3.2
4	LIC	0.1	0.2

To examine the differences between income groups, the K-W test for both imports and exports showed that there were statistically significant differences between each region's quantities of traded plastic scrap between 1988 and 2019 ($\chi^2(3) = 35.481$, p = .0001 for imports and $\chi^2(3) = 16.581$, p = .0009 for exports). Subsequently, post hoc pairwise comparisons revealed the HIC quantities of imports were significantly different from LMC and LIC. Among exports, the HIC group was found to be statistically different than all three other income groups, and UMC was significantly different from LIC (Table 2.8).

Table 2.8. Post hoc comparisons of economic classes by median import and export trade quantities from 1988 to 2019 using Dunn's procedure.

Imports	*n*	*Mdn (MMT)*	HIC	UMC	LMC	LIC
HIC	64	0.003	1			
UMC	49	0.207	2.31	1		
LMC	42	0.005	2.59^{*}	-0.31	1	
LIC	22	0.032	3.78^{***}	-1.86	-1.58	1
Exports	*n*	*Mdn (MMT)*	HIC	UMC	LMC	LIC
HIC	64	9.01	1			
UMC	49	0.059	3.36^{**}	1		
LMC	42	0.014	4.17^{***}	-0.91	1	
LIC	22	0.004	5.31^{***}	-2.63^{*}	-1.83	1

$^{*}p < .05$, $^{**}p < .01$, $^{***}p < .001$

Cumulatively, OECD members (*n* = 34) have imported and exported 60.4 MMT and 154 MMT, respectively between 1988 and 2019, which is equivalent to 23% and 65% of imports and exports, respectively. On the other hand, non-OECD member have imported and exported 201 MMT and 83.8 MMT of plastic scrap, respectively, in the same period (equivalent to 77%% and 35%). A Mann-Whitney U test indicated that differences between the OECD and non-OECD groups for cumulative imports were significant (z = -7.533; p < .001) and exports (z = -8.014; p < .001). Based on these values, OECD members have exported 1.8 times that of non-OECD members, while non-OECD members have imported 3.3 times that of OECD members. economic classifications.

2.3.2 Impact of the ban (2016 vs. 2019)

Temporal patterns

By 2016, a cumulative 214 MMT of plastic scrap had been exported globally, compared to 235 MMT having been imported globally. As of 2019, a cumulative 238 MMT of plastic scrap has been exported, compared to 262 MMT of imports. In 2016, 150 countries and territories reported quantities for traded plastic scrap, with all 150 reporting imports and 147 reporting exports. The number of reporting countries decreased in 2019, with 128 countries reporting trade values, and 127 and 117 reporting values for imports and exports, respectively. In 2016, 14.9 MMT were imported, compared to 6.08 MMT in 2019, equivalent to a 59% decrease and difference of 8.82 MMT. Similarly, 12.5 MMT and 6.46 MMT were exported in 2016 and 2019, respectively, which is equivalent to a 48% decrease, or a 6.04 MMT difference between the years.

Spatial analysis

In 2016, reported imports ranged from 0.02 metric tons in St. Kitts and Nevis and Aruba to 7.35 MMT in China, while exports ranged from 2.66 metric tons in the Bahamas to 2.82 MMT in Hong Kong. Excluding a report of 0 from Belize, 2019 reported imports ranged from 0.04 metric tons in Samoa to 7.35 MMT in Hong Kong. Exports in 2019 ranged from 0.02 in Samoa (similarly excluding a 0 entry from Slovakia) to 1.09 MMT in Germany. The mean quantities of both trade flows decreased between 2016 and 2019. However, the median value of trade quantities increased between 2016 and 2019 (Table 2.9). Wilcoxon signed rank tests between all countries that reported in 2016 and 2019 indicated statistically significant differences between exports (z =2.700, p = .0069) and imports (z = -2.703, p = .0069) between the years.

Table 2.9. Summary statistics for pre- and post-ban trade quantities across reporters in metric tons

Trade flow	year	*n*	*m*	*sd*	*mdn*	*min*	*max*
Imports							
	2016	150	99,600	642,000	1,560	0	7.35×10^6
	2019	127	47,800	106,000	3,880	0	607,000
Exports							
	2016	147	85,200	302,000	5,880	2.66	2.82×10^6
	2019	117	55,200	147,000	7,150	0	1.09×10^6

Between 2016 and 2019, 74 countries saw a decrease in imports, while 102 countries saw a decrease in exports. In contrast, increases in imports were seen in 83 countries, while 48 countries saw increases in exports. For countries that reported quantities in both 2016 and 2019, the largest increase in the quantity of imports was 178,000 metric tons in Vietnam, while the largest decrease was 7.35 MMT in China. Similarly, the largest increase in the quantity of exports was 0.28 MMT in the USA, while the largest decrease was 2.59 MMT in Hong Kong. Differences in imports and exports are visualized by country in Figure 2.4.

Between these years, the total contribution of the top ten reporters for imports decreased by nearly a quarter from 85% in 2016 to 59% in 2019, and the top ten reporters for exports also decreased from 72% in 2016 to 66% in 2019, indicating a considerable decrease in trade scrap since the ban. As expected, China shifted from the number one importer in 2016 to being ranked 78th in 2019, during which China contributed to < 0.1% of the total imports for that year following the implementation of the ban. If EU-28 countries are taken collectively, they would have been ranked 1st and 3rd in exports (42%) and imports (18%), respectively in 2016. Similarly, they would rank 1st in both exports and imports (54% and 46%, respectively) in 2019 with China having dropped from the top 10 (Table 2.10).

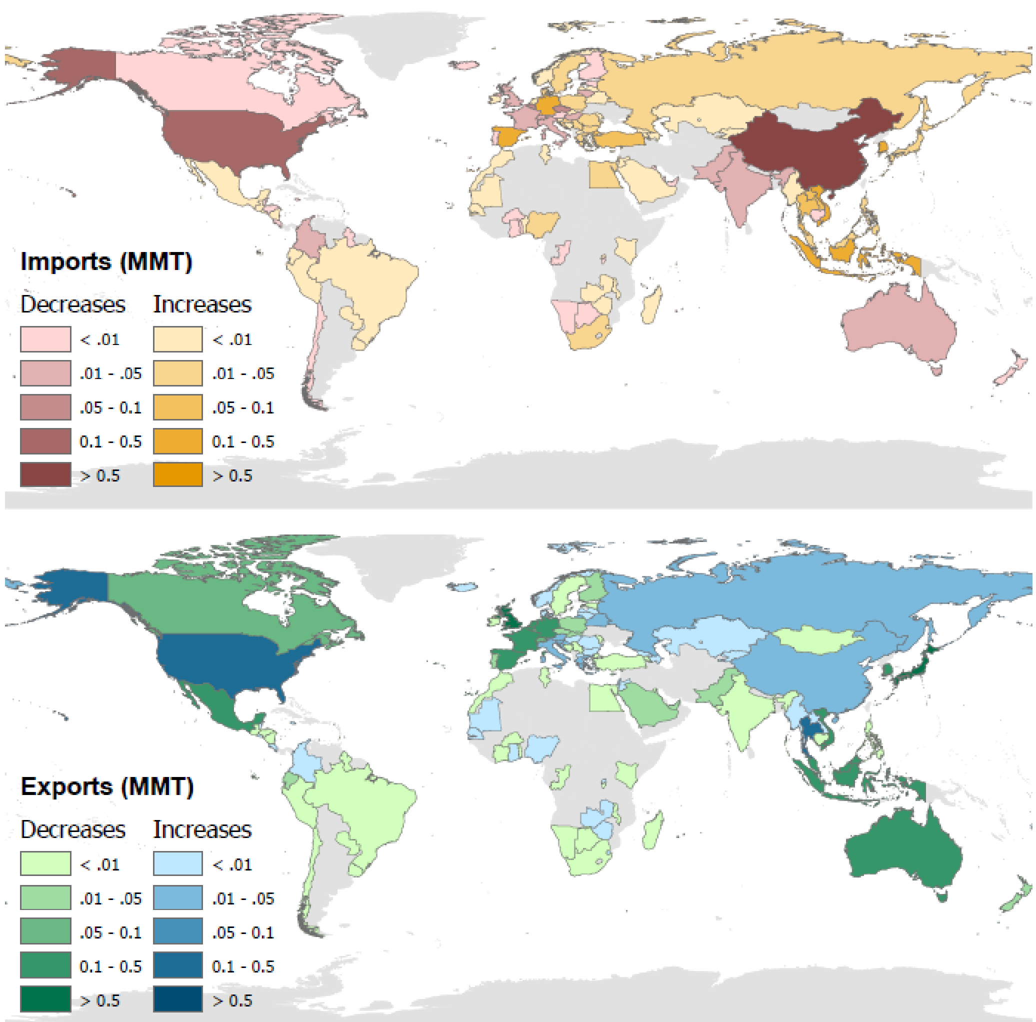

Figure 2.4. Differences in quantities of traded plastic scrap by country between 2016 (pre-ban) and 2019 (post-ban).

Table 2.10. Top ten importing and exporting countries/territories, world region, and income classification in 2016 and 2019.

	Year					
	2016 (pre-ban)			2019 (post-ban)		
Rank	Reporter	Region (Income)	Proportion (%)	Reporter	Region (Income)	Proportion (%)
Imports						
1	China	EAS (UMC)	49	Hong Kong	EAS (HIC)	10
2	Hong Kong	EAS (HIC)	19	Netherlands	ECS (HIC)	10
3	Netherlands	ECS (HIC)	3.9	Germany	ECS (HIC)	8.2
4	USA	NAC (HIC)	2.8	Other Asia	-	5.7
5	Germany	ECS (HIC)	2.3	Malaysia	EAS (UMC)	5.5
6	Belgium	ECS (HIC)	2.1	Vietnam	EAS (LMC)	4.6
7	Malaysia	EAS (UMC)	1.9	Indonesia	EAS (LMC)	4.1
8	Austria	ECS (HIC)	1.5	Turkey	ECS (UMC)	4.1
9	Other Asia	-	1.2	Belgium	ECS (HIC)	3.8
10	Canada	NAC (HIC)	1.1	Austria	ECS (HIC)	3.7
Total			85			59
Exports						
1	Hong Kong	EAS (HIC)	23	Germany	ECS (HIC)	17
2	Japan	EAS (HIC)	12	Japan	EAS (HIC)	14
3	Germany	ECS (HIC)	12	Netherlands	ECS (HIC)	6.0
4	UK	ECS (HIC)	6.4	France	ECS (HIC)	5.8
5	Netherlands	ECS (HIC)	3.9	USA	NAC (HIC)	4.9
6	France	ECS (HIC)	3.9	Belgium	ECS (HIC)	4.7
7	Belgium	ECS (HIC)	3.7	Mexico	LCN (UMC)	3.7
8	Mexico	LCN (UMC)	3.3	Hong Kong	EAS (HIC)	3.6
9	Spain	ECS (HIC)	2.5	Thailand	EAS (UMC)	3.5
10	Vietnam	EAS (LMC)	2.2	Italy	ECS (HIC)	3.2
Total			72			66

Historically, the EAS region has had the highest contribution in both imports and exports of plastic scrap. This was mirrored in 2016, when the region had the highest proportion of imports and exports across the regions (73% and 46%, respectively), however, this was not the case in 2019 when the region contributed the 2nd most in imports and exports (32% and 29%, respectively). In 2019, ECS took over as the largest contributor at 53% of imports and 56% of exports. All other regions contributed to around 5% or less in both years and both trade flows, with SSF contributing least to imports in both years (0.2% and 0.9%, respectively) and SAS contributing least to exports at 0.2% in both years (Table 2.11).

Table 2.11. Comparison between 2016 and 2019 by regional proportions of imports and exports of plastic scrap.

	Year	
	2016 (pre-ban)	2019 (post-ban)
Econ. class	Proportion (%)	Proportion (%)
Imports		
EAS	73	32
ECS	19	53
LCN	0.7	1.5
MEA	0.4	1.5
NAC	4.0	3.2
SAS	1.3	2.4
SSF	0.2	0.9
Exports		
EAS	46	29
ECS	44	56
LCN	4.5	5.1
MEA	1.7	1.7
NAC	2.0	7.1
SAS	0.2	0.2
SSF	0.4	0.4

Between 2016 and 2019, ECS, MEA, and SSF saw increases in imports, while LCN, SAS, NAC, and EAS saw decreases (Figure 2.5). All regions, saw decreases in exports except for NAC and SSF. The EAS region saw the largest difference in trade quantities between 2016 and 2019 in both imports and exports, which decreased 8.97 MMT and 3.91 MMT between 2016 and 2019 (equivalent to 82% and 68% reduction). The largest increase in imports was observed in ECS (0.33 MMT, 12%), and the largest increase in exports was seen in NAC (0.21 MMT, 82%). Otherwise, all other regions saw an overall decrease in exports between 2016 and 2019. Despite low quantities overall, the largest relative increase in imports was seen in SSF (59%), which has not historically been a major trade partner.

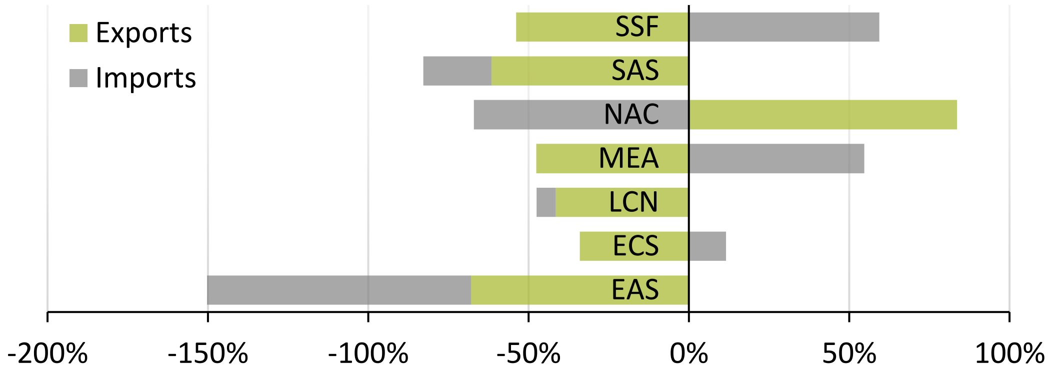

Figure 2.5. Percent difference in trade quantities by region between 2016 (pre-ban) and 2019 (post-ban).

Analysis by economic classification

In 2016, UMC had the highest proportion of imports (55%) and HIC had the highest proportion of exports (87%). In 2019 these HIC contributed most to both imports and exports (65% and 83%, respectively), and UMC's contribution reduced to 19%. The LIC region contributed the least in both trade flows and both years (< 1% for all).

Table 2.12 Proportions of trade quantities by economic classification cumulatively and annually in 2016 and 2019

	Year	
	2016 (pre-ban)	2019 (post-ban)
Econ. class	Proportion (%)	Proportion (%)
Imports		
HIC	42%	65%
UMC	55%	19%
LMC	3.2%	16%
LIC	0.1%	0.4%
Exports		
HIC	87%	83%
UMC	7.1%	12%
LMC	6.1%	5.6%
LIC	0.2%	0.1%

HIC and UMC saw the largest decreases in imports between 2016 and 2019, with the largest difference seen in UMC (-7.03 MMT, -87%). In contrast, LMC and LIC saw increases in imports, with the largest difference in LMC (434,000 metric tons, 91%). All income groups saw a reduction in exports between 2016 and 2019, with the smallest difference seen in LIC (-20,700 metric tons, 122%) and the largest seen in HIC (-5.40 MMT, -51%). Relative to 2016, the largest relative differences in both imports and exports were seen in LIC (122% and -72%, respectively). Finally, Wilcoxon signed rank tests showed that the differences in distributions of import quantities between 2016 and 2019 were statistically significant in the LMC (z = -2.324, p = .0190) and UMC groups (z = -2.805, p = .0041). Further, exports were significantly different in the HIC group.

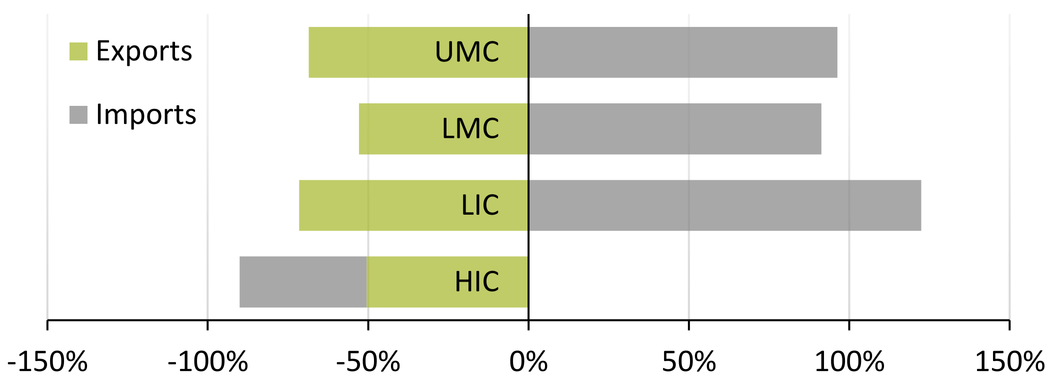

Figure 2.6. Percent difference in trade quantities by income group between 2016 (pre-ban) and 2019 (post-ban) group.

In 2016, non-OECD members contributed to 77% of imports, while non-OECD members contributed to 65% of exports. In 2019, OECD members contributed the most to both imports and exports (58% and 80%, respectively). Non-OECD member proportions of imports reduced to 42% in 2019. By absolute difference between 2016 and

2019, both OECD and non-OECD groups saw decreases in quantities of imports and exports, with the largest reductions in non-OECD regions, which decreased by 8.97 MMT and 3.08 MMT (equivalent to a relative decrease of 79% and 71%, respectively). OECD members saw far less impact between the years, with an overall .06 MMT decrease in imports and 2.88 MMT decrease in exports (equivalent to relative decreases of 2% and 36%, respectively). Wilcoxon signed rank tests indicated that the distributions in imports between 2016 and 2019 were significantly different for non-OECD members (z = -3.501, p =.0004), and not significantly different for OECD members (z = -0.453, p =.6606). In contrast, the distribution in exports were significantly different in OECD member countries (z = 2.966, p =.0023), but were not significantly different for non-OECD members (z =1.103, p = 0.2727).

Imports: China

China saw the largest absolute decrease in imports between 2016 and 2019. China imported 7.35 MMT from 116 reporting partners in 2016 with quantities ranging from 13 metric tons from ‘Areas, nes’ to 1.78 MMT from Hong Kong. In contrast, China imported only 924 metric tons from five trade partners, Japan, Germany, South Korea, Malaysia, and USA in 2019 (Figure 2.7), resulting in an almost 100% decrease in imports. Prior to the ban, in 2016, most (61%) imports came from EAS countries and 76% were from HIC countries, both of which were largely influenced by quantities imported from Hong Kong and Japan. Historically, China has traded the largest quantities of plastic with Hong Kong, USA, Japan, and Germany (Table 2.10). As an SAR of mainland China, Hong Kong has historically worked closely with mainland. While previous research reported that 63% of Hong Kong exports were destined for China in 2016, national level data for Hong Kong evaluated here determined that 99% of Hong Kong's exports were destined for China that year. Following the ban, Hong Kong exports shrank 92% and were instead exported to 25 other trade partners, primarily Thailand (39%), Vietnam (35%), and Malaysia (22%). See visualization of Hong Kong trading partners by quantity in Figure 2.8a.

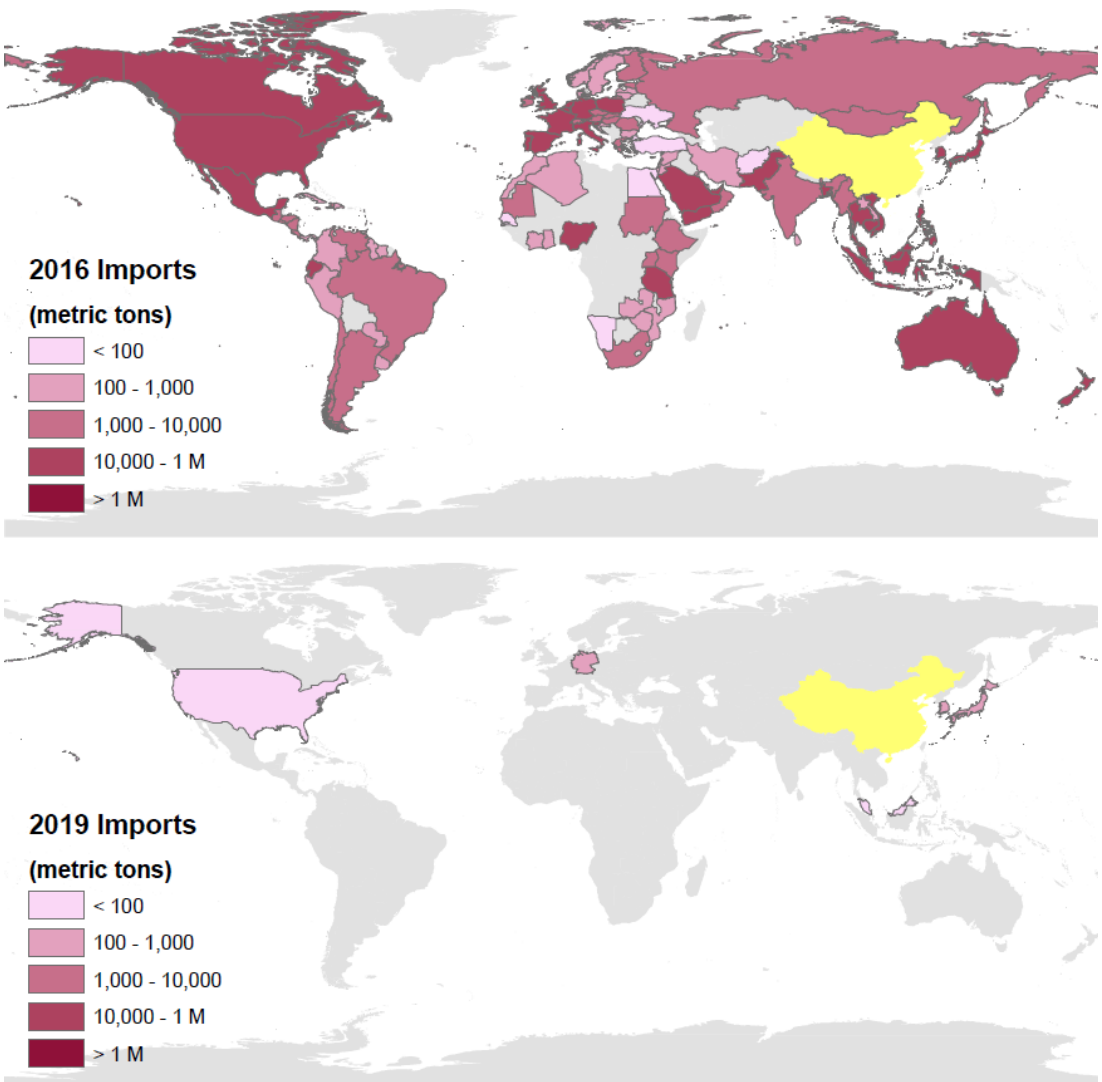

Figure 2.7. Map of Chinese imports in 2016 (top) and 2019 (bottom); Yellow polygon, China; Red polygons, source countries.

Exports: USA

In 2016, the USA exported 1.17 MMT of plastic scrap to 89 countries, ranging from 0.07 metric tons to South Africa to 0.48 MMT to China. In contrast, the USA exported 0.67 MMT of exports to 94 partners in 2019, down 43% since 2016. Exports ranged from 0.15 metric tons to Serbia to 0.15 MMT to Canada. Although the proportion sent to Canada increased by 9% between 2016 and 2019, the quantity of plastic scrap exported to Canada decreased by about 16,000 metric tons. In 2016, almost half of the exports from the USA were destined for UMC countries in the EAS region and 70% were destined for EAS countries, while in 2019, most went to HIC countries in the EAS. Decreased exports were seen at all income groups, with the largest absolute decrease being in the UMC group. See visualization of US trading partners by quantity in Figure 2.8b.

Exports: Japan

Between 2016 and 2019, Japan's exports decreased 41%, with 1.53 MMT having been exported in 2016 prior to the ban in place and 0.90 MMT following the ban in 2019. Japan's exports ranged from 1 metric ton bound for Switzerland to 0.80 MMT to China in 2016. In 2019, exports ranged from 4 metric tons for Austria to 0.26 MMT to Malaysia. The number of partners increased, with Japan having exported to 29 partners in 2016 and 48 partners in 2019. In 2016, 84% of Japan's exports were destined for China and Hong Kong, followed by 4.5% to Other Asia, nes, 4.3% to Vietnam, and 2.2% to Malaysia. exported to five countries, China, Hong Kong, Other Asia, nes, Vietnam, and Malaysia. Following the ban, 70% of Japan's exports went to four partners, Malaysia (29%), Other

Asia, nes (17%), Vietnam (13%), and Thailand (11%). With Malaysia being the number one recipient of plastic scrap from Japan, exports to the country increased 695% from 2016 to 2019. Most of Japan's exports have gone to EAS countries in both 2016 (95%) and 2019 (76%). By income, most of Japan's exports have gone to UMC partners (56% in 2016 and 43% in 2019). In 2016, exports to HIC countries were almost 7.5 times the quantities exported to LMC partners. As of 2019, however, these two groups saw almost the same quantity of plastic scrap from Japan, with LMC partners having increased from a 4.7% share of all of Japan's exports in 2016 to 21% in 2019. See visualization of Japan's trading partners by quantity in Figure 2.8c.

Exports: Germany

In 2016, Germany exported 1.46 MMT of plastic scrap internationally to 66 partner countries but saw a 25% reduction in exports as of 2019, with 1.09 MMT exported to 70 partners. Prior to the ban, Germany exported 39% of its plastic scrap to China and 14% to Hong Kong (53% combined). Following the ban, Germany sent 17% of its plastic scrap to Malaysia, which represented a 363% increase in exports there. Another 14% was exported to the Netherlands, and 8.5% were exported to Hong Kong. By region, Germany sent 58% of its scrap to EAS, followed by 37% to other countries in the ECS. These regions switched in terms of proportion of German exports following the ban, such that Germany exported 63% of its plastic scrap to ECS and 31% to EAS. With large quantities being exported to Hong Kong and other ECS countries before and after the ban, it is expected the Germany exports mostly to HIC countries, and the proportion increased from 51% in 2016 to 65% in 2019. See visualization of Germany's trade partners by quantity in Figure 2.8d.

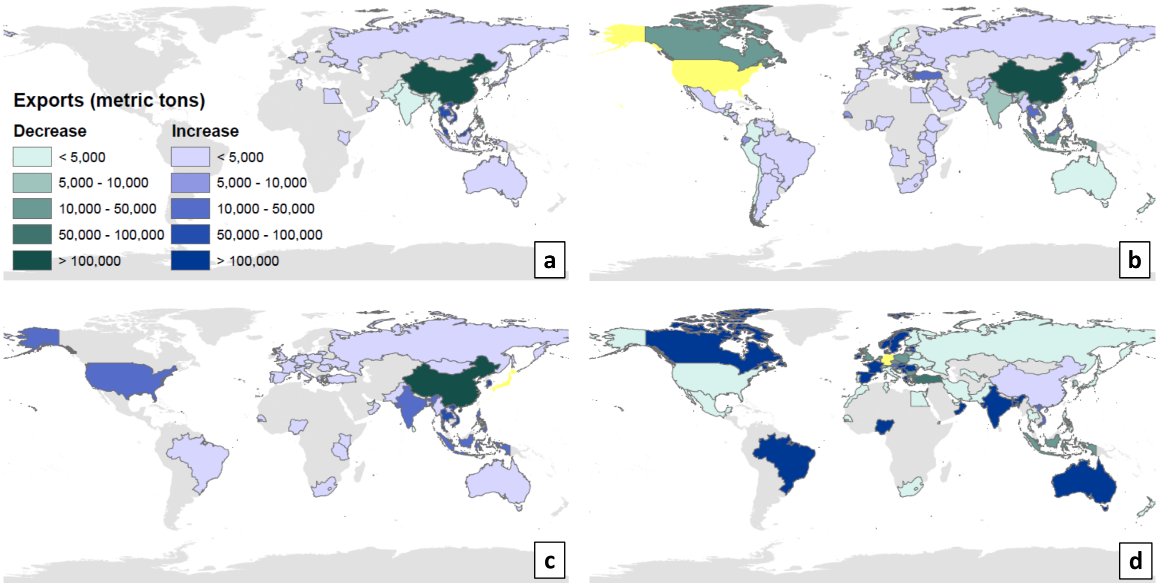

Figure 2.8. Map of major differences in exports between major exporting countries. A) Hong Kong, B) USA, C) Japan, D) Germany; Yellow polygons, exporting countries; Blue and green polygons, export destinations. Note that Hong Kong is too small in area to detect at this scale.

<u>Imports: Malaysia</u>

Malaysia imported 0.29 MMT of plastic scrap from 53 partners in 2016, with 24% sourced from the UK. With overall 16% growth in tonnage, Malaysia imported 0.33 MMT from 42 partners in 2019, with most sourced from the USA (18%), followed by Germany (14%), and Japan (12%). These three countries and the UK contributed to 55% of Malaysia's imports in 2019. Regionally, imports followed a similar pattern between 2016 and 2019, with about half coming from ECS countries, a third from EAS countries and a fifth from NAC. Imports into Malaysia have consistently been almost entirely sourced from HIC countries, representing 98% of imports in both 2016 and 2019.

Imports: Vietnam

Vietnam saw a 174% increase in imports between 2016 and 2019. It imported 0.10 MMT from 62 partners in 2016, and 0.28 MMT from 47 partners in 2019. The country's 2016 imports ranged from 2 metric tons from Uruguay to 0.03 MMT from Thailand. In 2019, Imports ranged from 5 metric tons from Pakistan to 0.09 MMT from Japan in 2019. By region, imports from EAS increased from 53% in 2016 to 77% in 2019, largely led by Hong Kong contributing to 17% in 2019, and Japan tripling its exports to Vietnam in 2019. The gap between proportions of imports from HIC countries and UMC countries was about 15 in 2016 and increased to a difference of 65 between the proportions in 2019. Similarly, trade with LMC countries increased from 3% in 2016 to 10% in 2019.

Imports: Indonesia

Indonesia imported 0.25 MMT from 43 partners in 2019, up 106% from 2016 imports, which totaled to 0.12 MMT from 27 partners. The 2016 imports ranged from 0.05 metric tons from Thailand to 50,300 from the USA, while in 2019 it ranged from 3.5 metric tons from Vietnam to 0.04 MMT from Marshall Islands, which was the second highest exporter of scrap to Indonesia in 2016. Indonesia's imports from the USA in 2016 reduced by 28% by mass in 2019. Most of Indonesia's imports are from the NAC and EAS regions in 2016, and ECS's proportion increased by 40 as of 2019. Imports have largely come from HIC countries both before and after the ban, however, the proportion of HIC imports increased in 2019, while UMC imports, which were largely influenced by the Marshall Islands, decreased in 2019.

Displaced plastic waste

Accounting for the small quantities of imported scrap in 2018 and 2019, a cumulative 15.78 MMT have been displaced by the ban as of 2019, representing 14% of the 111 MMT that were estimated to be displaced by 2030 by Brooks et al. (2018). Had the ban been fully implemented in 2018, a cumulative 15.81 MMT would have been displaced as of 2019, however, 52,500 metric tons of plastic scrap were still imported by China after the ban was in place, which represents a 99.7% implementation scenario.

Based on the analysis here, Chinese imports have dramatically decreased since the country implemented the ban and may provide insight as to what to expect in coming years (Figure 2.9). Simple linear regression predicted the cumulative quantities of plastic scrap that China would have imported from USA, Japan, and Germany as of 2030. All three regression models significantly predicted the cumulative quantities imported plastic scrap and model fit was 0.991, 0.994, and 0.999 for the USA, Japan, and Germany forecasts, respectively. In the last ten years of cumulative imports from three of the historical major exporters, USA, Japan, and Germany, China has displaced an estimated 1.70 MMT, 1.72 MMT, and 2.13 MMT, respectively. Altogether, the ban has displaced a total 5.56 MMT of plastic scrap from these three countries since the ban, equivalent to 35% of the 15.81 MMT of plastic scrap that is estimated to have been displaced since the implementation of the ban. By 2030, the ban will have displaced 10.2 MMT from the USA, 11.4 MMT from Japan, and 9.88 MMT from Germany, equal to a total of 31.5 MMT, which is 28% of the 111 MMT estimated to be globally displaced by the ban by 2030 (Brooks et al. 2018).

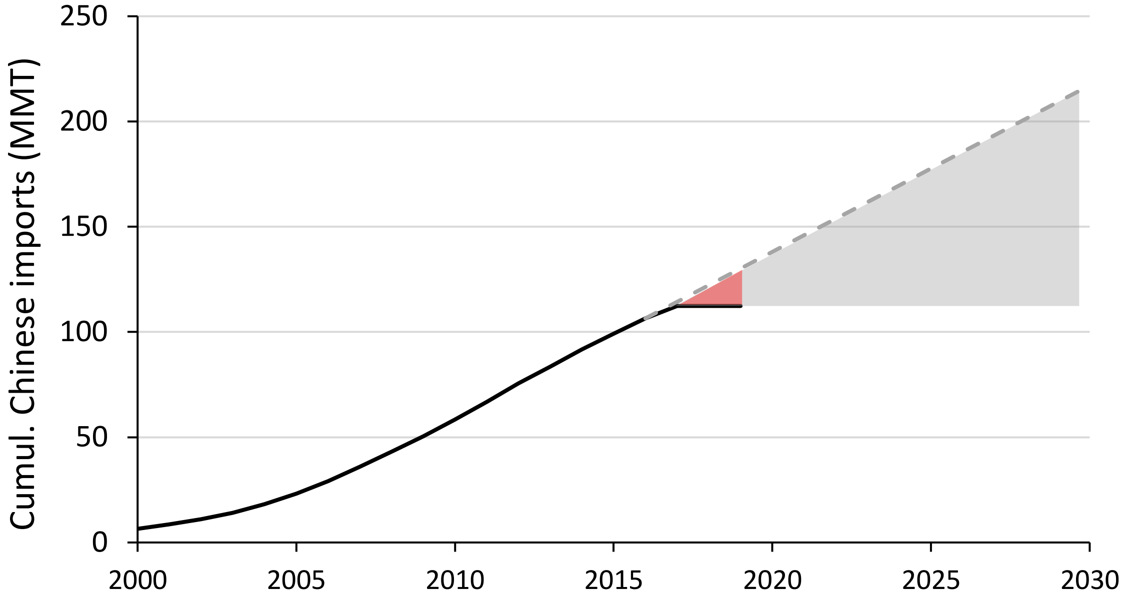

Figure 2.9 Observed Chinese imports (Mt) prior to the ban (2010-2018) and following the ban (2018-2019). (Gray, estimated displaced plastic waste by 2030; Red, displaced plastic waste as of 2019).

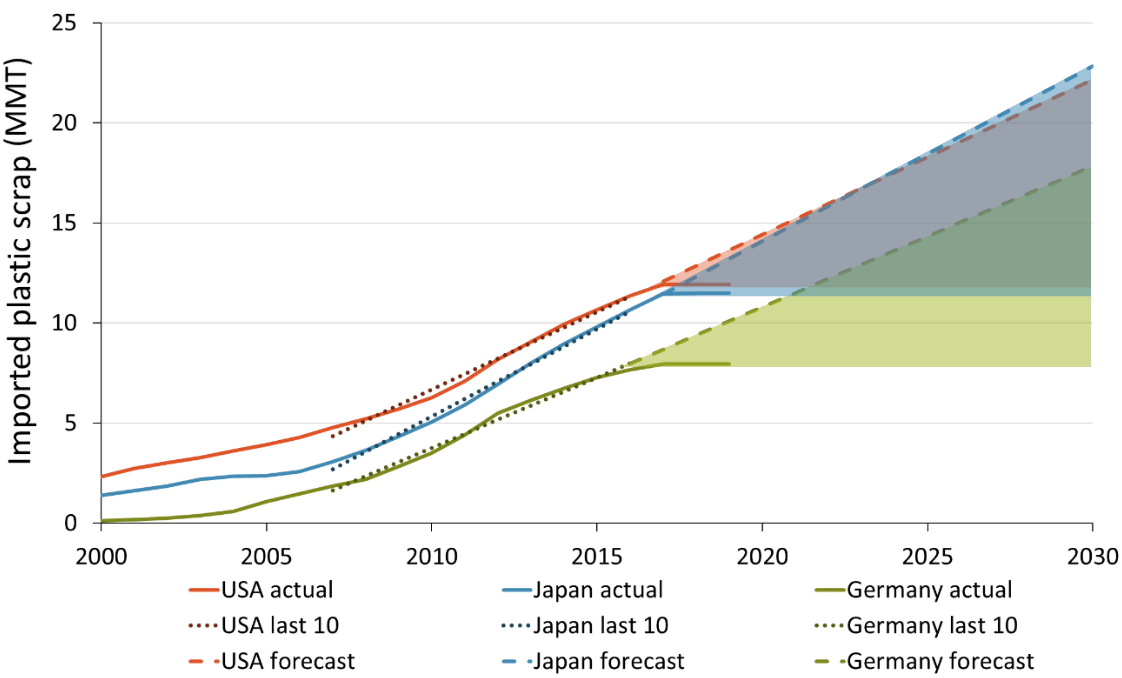

Figure 2.10. Actual (2000 – 2019) and forecasted (2018-2030) Chinese imports from USA, Japan, and Germany. (Last 10, linear regression 2007 – 2017 imports from each country.

Impact to global mismanaged plastic waste

Based on values reported by Law (2020), the fraction of waste inadequately managed in China was 23%. Based on this inadequate fraction, a total 1.71 MMT of imported plastic scrap were potentially inadequately managed in 2016, compared to only 215 MT in 2019. Taking the difference between the 2016 and 2019 value, a 1.7 MMT deficit of inadequately managed plastic scrap resulted from the ban, representing a near 100% decrease in mismanaged plastic waste related to plastic imports in China. Similarly, Malaysia, Vietnam, and Indonesia had reported inadequate fractions of 17.9%, 62%, and 59%, respectively (Law 2020). Together the three countries had a total of 185,000 metric tons and 379,000 metric tons of inadequately managed plastic scrap in 2016 and 2019, respectively. As a result, a surplus 0.19 MMT of inadequately managed plastic scrap resulted from the Chinese ban across these three importing countries, accounting for ~1.0% of the cumulative displaced plastic waste as of 2019. Individually, Malaysia saw a relatively small increase of 16%, compared to Vietnam and Indonesia which saw a 174% and 106% increase, respectively.

The fraction of inadequately managed waste by the aggregation of countries into paired region-income categories ranged from 0% in NAC-HIC to 94% in SSF-LIC. An estimated 4.53 MMT of plastic scrap may have been inadequately managed in 2016 prior to the ban, compared to only 921 MT in 2019. Based on mean proportions of inadequately managed waste from Law (2020) combined across regions and income groups, both ECS and SSF may have seen surplus quantities of inadequately managed plastic scrap following the ban by 52% and 4.7%, respectively. All other regions saw overall deficits in inadequately managed scrap, ranging from 33% to 63%. The LMC income group was the only group that saw an increase (21%) in inadequately managed plastic scrap in 2019. By region-income groups, the largest increase in inadequately managed plastic scrap was in LMC countries in ECS (121%), while the largest decrease was 100% in UMC countries in SAS (Table 2.13).

Table 2.13. Contingency table showing percent change in quantities of inadequately managed plastic scrap between 2016 and 2019. Values shown as ratio. Bold values represent highest and lowest values by region, income level, and combination.

	HIC	LIC	LMC	UMC	avg
Region					
EAS	-74	-	06	**-90**	**-53**
ECS	-22	-	**120**	57	**52**
LCN	-60	-	-3.6	-37	-33
MEA	-38	-	20	-95	-38
SAS	-	-	-26		-26
SSF	-	-59	11	62	4.7
NAC	-	-	-	-	-
Avg	-48	**-59**	**21**	-21	

2.4 DISCUSSION

2.4.1 Update on trade of plastic scrap (1988 – 2019)

Temporal analysis aided by visualized trade data in Figure 2.1 determined that annual values of trade in both mass and value were non-stationary over time. This irregularity was expected, considering the multitude of interconnected factors that may influence the quantity of trade in the past 30 years such as increasing globalization in the 1990s (Brooks et al. 2018), changing policies across various scales, and shifts in populations, consumption, production, and economies. While conclusions are somewhat tentative for evaluating change over time, the findings presented here provide insight into how the global picture of internationally traded scrap is changing.

Both raw quantities of imports and exports appear to have peaked twice in the 2010s and have been experiencing a steep decrease since the second peak in 2014. Trade quantities have not recovered or begun to increase as of 2019 and quantities are comparable to those traded in the early 2000s. Trade value per ton has fluctuated between about $200 and $600 USD since the 1980s and show a period between 1988 and the early 2000s when the commodity price of scrap was decoupled from the quantities traded. This period of decoupled growth in trade quantities paired with decreasing trade values of plastic waste is likely related to the rapid growth in the recycling industry between the 1980s and 2000s. For instance, the USA generation of plastic packaging waste increased 265% from 1980 to 2005, while the mass of recycled plastic packaging increased from 10,000 tons in 1980 to 1.25 million tons in 2005, equivalent to a 12,700% increase (US EPA 2018). During this period of advancement in recycling technology and consumer willingness to participate due to concerns about plastic pollution, globalization of trade was simultaneously growing. As such, prices of plastic scrap likely became coupled with the traded quantities once the international market was established and a sufficient supply of post-consumer plastics were available to recycle at rates abroad that were cheaper than those domestically.

Since the mid-2000s, trade values have followed similar annual fluctuations as the quantities with apparent valleys related to temporary import bans implemented in China in the 2010s (Tran et al. 2021). The past few years have seen a decrease in quantities of plastic scrap being traded internationally, most of which is likely due to preemptive reactions to the Chinese import ban, which was announced in mid-2017 and implemented in 2018 (Chinese Ministry of Environmental Protection 2017). Given China’s central role in the historical plastic scrap trade, the world has been experiencing resounding impacts, and given that this assessment is limited to the recent years since the ban, the full impacts of the ban are not yet fully understood.

Temporal patterns

At the temporal scale of the cumulative assessment, the effects of the ban were not significant to enough detect its impacts relative to the national, regional, and economic changes over the cumulative quantities of traded plastic scrap, though this was not the case when recent years were isolated in the second part of the analysis, which is discussed further in Section 2.4.2. Despite the small differences in overall trade, this review and update provided some new insight into differences between groups in the regional, income, and OECD categories.

Spatial analysis

The contribution among countries at the upper end of trade quantities did not see a significant shift in the cumulative values of traded plastic scrap. Compared with (Brooks et al. 2018), reductions in quantities since the ban were minimal overall, with 1-2% reductions in traded quantities. Previously identified major importers and exporters by Brooks et al. (2018) continue to trade the highest quantities and proportion of plastic scrap. Top historical exports stayed the same as of 2019, though small fluctuations occurred in the proportions that the countries contribute, but not yet anything indicative of a particular pattern. Imports, on the other hand, saw one new UMC-EAS country join the top ten ranks, with Malaysia ranking as the ninth overall importer of plastic scrap as of 2019, having contributed to 1.5% of cumulative imports (Table 2.4). This is initial indication of the geospatial shift such that scrap that would have been imported by China is now ending up in other proximal countries.

China had a detectable reduction in its cumulative contribution of plastics scrap imports having reduced its previous share from 45% by Brooks et al. (2018) to 43% of the world's plastic scrap. This change appears to propagate through the data such that the pattern slightly detectable across regions and income groups as well where the shift in the proportion of EAS imports is incorporated, and correspondingly complimented by slight increases in contributions of ECS and NAC.

The overall rank by proportion stayed the same for both regions and income groups as of 2019, however, incremental changes may have occurred such that ECS, NAC, MEA, and SSF saw slight increases in their proportion of imports. Similarly, ECS and SSF saw slight increases in their proportion of exports, while NAC and LCN saw slight decreases. By income, rankings have remained the same, with HIC groups still contributing to most trade by mass, however, HIC and UMC saw slight decreases in their proportion of exports, paired with slight increases in LMC and LIC, potentially indicating a shift in participating income groups. The LMC group's proportion of imports also saw an apparent increase, which may signal that more waste is being sent to these countries due to the displacement of plastic scrap from the Chinese ban. Similarly, OECD saw a slight increase in cumulative exports having contributed to 65% of all exports as of 2019 versus 64% as reported in Brooks et al. (2018).

Differences within the regions and income groups and between the OECD and non-OECD members were also determined as part of this study. Most differences were not significant between groups, however, the regions on the extreme ends tended to have significant differences in imports and exports given the substantial differences in median trade quantities. Although NAC had the highest median quantity in both exports and imports, with only three countries in the group, the distribution lacks statistical power, and so no significant results were detected for this region.

The differences observed in the SSF region were not surprising given the low participation of the region in terms of traded quantities. The significant difference detected between LCN and ECS, however, was surprising given the similarities in urbanization and waste management, warranting further investigation into what sets these regions apart. Differences are likely influenced by the significantly larger quantities of historical imports and exports from the ECS region compared to LCN. Notably, the variables associated with this study are complex and differences may be more or less pronounced depending how the data is 'sliced'. Future work may investigate alternative modeling approaches to account for this that are outside of the scope of this work.

Economically, HIC was significantly different from all income groups in terms of exports, and only significantly different from both LMC and LIC groups in imports. Despite having similar medians, the range of export quantities was much larger in HIC versus the other groups, however, the UMC and HIC had comparable ranges in imports which likely led to no observed statistical difference. For both trade flows, LIC groups were significantly different from UMC and LMC, which is not unexpected considering the difference in proportions of exports for each (*mdn* = 9.01 MMT in HIC versus *mdn* = 0.004 MMT in LIC). The UMC and LMC groups, however, did not significantly differ from each other despite UMC's median being 4.2 times higher than that of the LMC group, however, further inspection revealed that these two income groups contribute relatively similar quantities of exports, though UMC had significantly higher quantities of imports.

Given that imports quantities are higher in non-OECD countries and exports are higher in OECD countries, it is not surprising that significant differences were observed between the two groups, however, the tests still confirmed their differences in terms of distributions. The proportion of non-OECD imports is concerning, given that most population growth in the past 70 years has been in non-OECD countries, while simultaneously, most economic growth has occurred in OECD countries, which indicates that OECD drives global consumption, contributing to global inequality (Steffen et al. 2015). Future research may explore this relationship in greater depth, particularly in relation to the pollution haven hypothesis which posits those wealthy regions reduce environmental degradation at home by distributing production and pollution to industrializing regions (Cole 2004).

2.4.2 Impact of the ban (2016 vs. 2019)

Temporal patterns

Comparisons between traded quantities between 2016 and 2019 provided a clearer understanding of the impact of the ban compared to the cumulative review described in the previous section. Trade quantities were far less in 2019 than they were in 2016, and fewer countries reported participating in trade. Among those that did, more saw a decrease in exports than in imports, providing further evidence for the broad displacement of plastic scrap.

Spatial analysis

Some patterns are emerging as early responses to the ban. The USA saw the largest increase between 2016 and 2019, which was initially surprising, as it was assumed quantities of exports would have decreased due to displacement by the ban. However, the US reported a substantial reduction in exports in 2016. After a period of consistently exporting about 2 MMT annually, the quantities plummeted 98% to 0.035 MMT in 2016, most likely in in anticipation of the imminent ban on imports in China, on which the US had depended for so long. Since then, however, exports have been gradually increasing, which has contributed to the prominent increase in exports detected in this study.

Similarly, countries in Southeast Asia emerged as potential new recipients of large proportion of the world's waste, which has been observed in similar recent studies (Pacini et al. 2021). As predicted by Brooks et al. (2018), imports have shifted to countries that are proximal to China, with other countries in EAS such as Malaysia, Vietnam, and Indonesia remaining or appearing on the top ten countries list in 2019 (See Table 2.4 and Figure 2.3), however, these nations are formulating their own versions of regulations around importing, processing, and management plastic scrap (Pacini et al. 2021).

With dramatic reduction in Chinese imports, the EAS region also saw a decrease in 2019. Further, UMC and LMC groups saw significant changes in imports, with UMC seeing a decrease and LMC seeing an increase. while HIC saw a significant reduction in exports, despite only decreasing by 4% between the years. One unanticipated result was the large relative increase in imports in SSF. Further inspection of this group found that Nigeria had a large absolute increase in trade quantities between 2016 and 2019, nearly doubling their imports. By relative change, Madagascar, Senegal, and Rwanda had the three largest increases in exports relative to their 2016 values. There has been media recent media attention focused on US plastics industry investment into plastic recycling infrastructure in Africa, namely, Kenya, for potential continuation of US distribution of waste to other regions that have lagging waste management infrastructure and has the potential to add increased burden on already limited waste infrastructure (Tabuchi 2020).

The changes seen in UMC and LMC groups are likely attributable to the significant decrease in imports in China, particularly with some of the plastic waste displaced by China (UMC) to other countries that are UMC and LMC in the region. The LIC group had the largest relative increase in imports which is concerning given the connection between income and waste management (Jambeck et al. 2015, Kaza et al. 2018). Further inspection revealed that the same SSF countries driving the high relative change in the regional analysis, Madagascar, Senegal, and Rwanda, were driving the large relative increase in imports by LIC countries. Additionally, the largest absolute differences identified other LIC countries that may be experiencing increases in imports, including Yemen and Zimbabwe which have seen large increases in imports compared to 2016 (though still small relative to other regions and groups). Regarding OECD members, the reduction in quantities in each group was not unexpected given the implementation of the ban in Hong Kong and China which are both non-OECD members. Given that OECD members did not experience a significant increase in imports OECD members apparently continue to import at same levels, having only changed by 2% between each year. Considering that direct impacts were seen in middle income regions, there may be some early evidence that effects of the ban could be inadvertently contributing to global inequality, which may be a pertinent topic for future research.

Country briefs

China saw a drastic decrease in imports between 2016 and 2019 as visually evident in Figure 2.7. These changes were expected given the nature of the import ban, however, reactions in other regions and countries are somewhat more nuanced. All three of the major importers saw an increase in the number of trade partners in 2019 and appeared to shift away from concentrating all exports toward one country or region, which makes sense given the previous reliance on China and the fact that most other countries in the region do not have the capacity to take in and adequately manage equivalent quantities as that in China (Lim 2018, Huang et al. 2020). One observation that did not appear to have a clear explanation was that USA, Japan, and Germany all saw close to 40% increase in exports between 2016 and 2019. This similarity in growth may be due to chance or coincidence but may warrant further investigation.

Given the changing destinations, it stands that some middle-income countries in the EAS region have seen impact as importers, particularly Malaysia, Vietnam, and Indonesia, which were represented in the top ten list of importers in 2019 (Table 2.10). Malaysia was the only country in the group whose majority trade partners were not proximal to Malaysia geospatially as well as by income. In other words, all three of the exporting countries traded with other countries in their region who were of similar income groups (i.e., USA and Canada, Germany and Netherlands, Japan, and South Korea). Similarly, Vietnam and Indonesia traded significant quantities with similar nations in their region. Combined with Malaysia's position in the top ten importers, and the fact that all three of the major exporters discussed contributed the highest quantities of Malaysia's imports, it seems that Malaysia has the potential to continue managing high quantities of plastic scrap unless (Morita and Hayashi 2018). Despite some indications that Thailand would play a significant role in absorbing plastic scrap displaced by the Chinese ban, the country was largely absent or not prevalent in terms of imports of waste like other countries in the region. This is likely directly attributable to a restriction on imports of plastic waste in the country (Sasaki 2021).

Displaced Plastic Waste

One way that the ban may be evaluated is by the quantity of plastic imports that may have been displaced by the ban. While only two years of UN Comtrade data were evaluated since the implementation of the ban, the ban is clearly being enforced under a near 100% scenario. While the Chinese ban has likely displaced a cumulative 17 MMT of plastic scrap, it is evident that major exporters have been directly impacted by the ban. Reactions from major exporters seem to range from policy efforts targeting circular economy initiatives (EU Comission 2019), improvements in domestic waste management, and increased treatment through other methods like landfilling and incineration.

Mismanaged plastic scrap

While many unknowns remain, there may be some early positive outcomes of the ban. One recent study has suggested that there may be realized global benefits from the

Chinese import ban in the form of annual eco-cost savings of 2.35 billion euros and may have improved environmental indicators such as fine particulate matter formation, freshwater ecotoxicity, and water consumption (Wen et al. 2021). Rough calculations described here for different region and income combinations, as well as a handful of countries, shows that some countries and groups may experience deficits of inadequately managed plastic scrap, which could be a positive outcome of the ban. On the other hand, surplus quantities of plastic scrap may be occurring in countries and groups experiencing increases in imports such as Malaysia, Vietnam, and Indonesia, as well as the LIC group which saw the only increase in imports across the income groups. Given China's historic influence in global mismanaged plastic waste and marine inputs into the ocean, early indications here are that the import ban has had a potentially positive result for the country in the context of mismanaged plastic waste and marine debris. That said, this positive outcome for China is also the given the displacement of plastic scrap, however, other regions and groups have likely seen a different result.

2.4.3 Policy implications

Since the ban was implemented in 2018, there has been wide public coverage and momentum around the issue of plastic scrap trade and global pollution reduction, and as a result there have been growing international calls for redefining plastic scrap as a hazardous material due to its potential negative consequences in the environment and subsequent effects to economies. From a waste management perspective, the transboundary movement of plastic waste as a commodity is justification for global agreements that have been called for previously (Borrelle et al. 2017, Worm et al. 2017).

Global agreements on the plastics recycling industry will further shift the way plastic waste is regulated which will no doubt result in a new set of challenges (Woodring and Hyde 2019). Concepts such as Extended Producer Responsibility (EPR) and strict liability may prove useful for holding waste producers and exporters accountable for making sure that the material they ship is properly managed by any receiving entity.

Although international regulations and treaties can be difficult to enforce at the local scale (Bodansky 2010), they are effective at raising global awareness, drive consensus among the many countries, territories, and economic states in the world, and establishing standards for international interactions. In the context of the global plastic scrap trade, one relevant international treaty is the Basel Convention on the Control of Transboundary Movements of Hazardous Waste and Their Disposal, which was designed to minimize and regulate the transfer of hazardous waste from developed countries to less developed regions. The Convention also provides a framework for knowledge sharing and promoting the proper management of waste, including harmonization of technical standards and practices, which could help build capacity to properly manage plastic waste around the world.

Prior to the Chinese ban on imports, plastics were not considered "waste requiring special consideration" despite the known hazards to animals and the environment, so no regulations were in place to ensure proper management in destination countries or holding exporting parties accountable. An amendment to the Convention led by Norway in 2018 and accepted by the 14th Meeting of the Conference of Parties to the Basel Convention in 2019 includes the following amendments to the Convention which were made effective as of January 21st, 2021:

- Annex II Amendment: Addition of plastic scrap to the list of the categories of wastes requiring special consideration
- Annex VIII Amendment: Addition of plastic scrap to the list of waste material presumed to be hazardous and therefore subject to prior informed consent procedure that includes mixtures of plastic waste containing or contaminated to an extent that it exhibits characteristics such as ecotoxicity, delayed or chronic toxicity, poisonous, infectious, etc.
- Annex IX Amendment: Allows for the transboundary movement of plastics waste (cured resins, non-halogenated and fluorinated polymers) and mixtures of PE, PP, or PET provided they are free of contamination and are bound for recycling in an environmentally sound manner (Secretariat of the Basel Convention 2018)

While the Basel Convention may help to reduce the potential of hazardous trading of plastic waste and subsequent mismanagement, these international agreements alone do not adequately address the full lifecycle of plastic products because they can be difficult to enforce on the ground. While China ratified the Basel Convention in 1992, there has still been considerable environmental harm in the country due to imports of contaminated waste material, bringing into question the effectiveness of the Convention (Raubenheimer and McIlgorm 2018). The Basel Convention has 187 participating parties and 53 signatories; however, the USA is one of the only top exporting countries that has not ratified the Basel Convention, and claims independent authority to decide that an when lacking capacity for efficient disposal of waste, exporting is permitted if the importing country can dispose of waste in an economically and environmental efficient manner (He et al. 2018, Secretariat of the Basel Convention 2020a). Further, the USA has yet to sign the amendments for plastic waste included in the Basel Convention or notify the Conference of Parties regarding its intention for signing (Secretariat of the Basel Convention 2020b), suggesting that plastic waste could still be exported from the country under its own authority. However, most the USA's trade partners are signatories of the Basel and its amendments, which could limit the ability of the USA to export plastic scrap. Being one of the major historical participants of the global plastic scrap trade, this could have severe impacts to USA exports of plastic scrap. Given the substantial impacts that are likely to occur by the new Basel amendments, continued monitoring of the impacts of the Chinese import ban and recent international policies is imperative for comprehensive global management of plastic waste.

2.5 CONCLUSION

The purpose of this study was to provide an update on the international plastic scrap trade Brooks et al. (2018) and to expand on that work through assessment of the early effects of the ban on the global plastic scrap trade by country, region, economic classification, and OECD membership. Already, 15.8 MMT of plastic scrap has been displaced by the ban. Effects of the ban have been universal for participants of the plastic scrap trade evident reductions in plastic trade for both imports and exports. However, the different types of trade participants appear to be experiencing the initial outcomes of the ban differently, with high income countries seeing decreases in exports, but appearing In addition to reviewing effects of the ban across these categories, newly available trade data informed the estimate of displaced plastic waste finding that China has implemented the ban to nearly 100% enforcement, such that reaching 111 MMT of displaced plastic waste in 2030 may very well be in progress.

REFERENCES

Bishop, G., D. Styles and P. N. L. Lens (2020). "Recycling of European plastic is a pathway for plastic debris in the ocean." Environment International **142**: 105893.

Bodansky, D. (2010). The Art and Craft of International Environmental Law. Cambridge, Mass, Harvard University Press.

Borrelle, S. B., C. M. Rochman, M. Liboiron, A. L. Bond, A. Lusher, H. Bradshaw and J. F. Provencher (2017). "Opinion: Why we need an international agreement on marine plastic pollution." Proceedings of the National Academy of Sciences of the United States of America **114**(38): 9994-9997.

Brooks, A. L., S. Wang and J. R. Jambeck (2018). "The Chinese import ban and its impact on global plastic waste trade." Science Advances **4**(6).

Chinese Ministry of Environmental Protection (2017). Announcement of releasing the Catalogues of Imported Wastes Management. **no. 39**.

Cole, M. A. (2004). "Trade, the pollution haven hypothesis and the environmental Kuznets curve: examining the linkages." Ecological Economics **48**(1): 71-81.

Dunn, O. J. (1964). "Multiple Comparisons Using Rank Sums." Technometrics **6**(3): 241-252.

EU Comission (2019). Circular Economy: Commission welcomes European Parliament adoption of new rules on single–use plastics to reduce marine litter, European Commission.

European Commission (2018). ANNEXES to the Communication from the Commission to the European Parliament, the Council, the European Economic and Social

Committee and the Committee of the Regions: A European Strategy for Plastics in a Circular Economy. Brussels. **28**.

Geyer, R., J. R. Jambeck and K. L. Law (2017). "Production, use, and fate of all plastics ever made." Science Advances **3**(7): e1700782.

He, K., Q. Tan, L. Zheng and J. Li (2018). "Adapting to new policy environment - past pattern and future trend in us-sino waste plastic trade flow." International Journal of Sustainable Development & World Ecology **25**(8): 702-711.

Huang, Q., G. Chen, Y. Wang, S. Chen, L. Xu and R. Wang (2020). "Modelling the global impact of China's ban on plastic waste imports." Resources, Conservation and Recycling **154**: 104607.

Jambeck, J. R., R. Geyer, C. Wilcox, T. R. Siegler, M. Perryman and A. Andrady (2015). "Plastic waste inputs from land into the ocean." Science **347**.

Kaza, S., L. Yao, P. Bhada-Tata and F. Van Woerden (2018). What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050. Urban Development Series. Washington, DC, World Bank.

Law, K. L., Starr, N., Siegler, T., Jambeck, J., Mallos, N., Leonard, G. (2020). "The United States' contribution of plastic waste to land and ocean." Science Advances **6**(44).

Lebreton, L. C. M., J. van der Zwet, J.-W. Damsteeg, B. Slat, A. Andrady and J. Reisser (2017). "River plastic emissions to the world's oceans." Nature Communications **8**: 15611.

Lim, X. (2018). "Problem plastics." Chemistry & Industry **82**(8): 39-39.

Morita, Y. and S. Hayashi (2018). Proposals to Strengthen Japan's Domestic Measures and Regional Cooperation on Stable and Environmentally Sound Plastic Scrap Recycling
Response to China's Ban on Imports of Plastic Scrap, Institute for Global Environmental Strategies.

OECD (2019). Waste Management and the Circular Economy in Selected OECD Countries.

Pacini, H., G. Shi, A. Sanches-Pereira and A. C. D. S. Filho (2021). "Network analysis of international trade in plastic scrap." Sustainable Production and Consumption **27**: 203-216.

Plastics Europe (2020). "Plastics the Facts - 2020."

Qu, S., Y. Guo, Z. Ma, W.-Q. Chen, J. Liu, G. Liu, Y. Wang and M. Xu (2019). "Implications of China's foreign waste ban on the global circular economy." Resources, Conservation & Recycling **144**: 252-255.

Raubenheimer, K. and A. McIlgorm (2018). "Can the Basel and Stockholm Conventions provide a global framework to reduce the impact of marine plastic litter?" Marine Policy **96**: 285-290.

Sasaki, S. (2021). "The effects on Thailand of China's import restrictions on waste: measures and challenges related to the international recycling of waste plastic and e-waste." Journal of Material Cycles and Waste Management **23**(1): 77-83.

Secretariat of the Basel Convention (2018). BASEL CONVENTION: ON THE CONTROL OF TRANSBOUNDARY MOVEMENTS OF HAZARDOUS WASTES AND THEIR DISPOSAL. U. N. Environment, Secretariat of the Basel Convention.

Secretariat of the Basel Convention. (2020a). "Parties to the Basel Convention on the Control of Transboundary Movements of Hazardous Wastes and their Disposal." Status of Ratifications Retrieved 5 July 2020, from http://www.basel.int/Countries/StatusofRatifications/PartiesSignatories/tabid/4499/Default.aspx#US17.

Secretariat of the Basel Convention. (2020b). "Amendments to Annexes II, VIII and IX to the Basel Convention." Status of Ratifications, from http://www.basel.int/Countries/StatusofRatifications/PlasticWasteamendments/tabid/8377/Default.aspx.

Steffen, W., W. Broadgate, L. Deutsch, O. Gaffney and C. Ludwig (2015). "The trajectory of the Anthropocene: The Great Acceleration." **2**(1): 81-98.

Tabuchi, H., Corkery, M., Mureithi, C. (2020). Big Oil Is in Trouble. Its Plan: Flood Africa With Plastic. The New York Times. New York, NY.

The World Bank. (2020). "World Bank country and lending groups." from https://datahelpdesk.worldbank.org/knowledgebase/articles/906519-world-bank-country-and-lending-groups.

Tran, T., H. Goto and T. Matsuda (2021). "The Impact of China's Tightening Environmental Regulations on International Waste Trade and Logistics." Sustainability **13**(2).

US EPA (2018). S. Y. F. a. F. D. f. 1960-2018. Washington, DC.

Wang, C., L. Zhao, M. K. Lim, W.-Q. Chen and J. W. Sutherland (2020). "Structure of the global plastic waste trade network and the impact of China's import Ban." Resources, Conservation and Recycling **153**: 104591.

Wen, Z., Y. Xie, M. Chen and C. D. Dinga (2021). "China's plastic import ban increases prospects of environmental impact mitigation of plastic waste trade flow worldwide." Nature Communications **12**(1): 425.

Woodring, D. and I. Hyde (2019). "Prepare for Round Three of the Plastic Waste Trade War: After China closed its doors to scrap plastic imports in 2017 and Westerners scrambled to find other export targets, diplomats chew on a proposal to add mixed plastic waste to a list of materials that require the receipt of a country's informed acceptance of this type of trade under the Basel Convention." Plastics Engineering(6): 32.

Worm, B., H. K. Lotze, I. Jubinville, C. Wilcox and J. Jambeck (2017). "Plastic as a Persistent Marine Pollutant." Annual Review of Environment and Resources **42**(1): 1-26.

CHAPTER 3

PLASTIC WASTE MANAGEMENT IN THE LATIN AMERICAN AND CARIBBEAN REGION[2]

[2] Brooks, A.L., Mozo-Reyes, E., Jambeck, J.R. To be submitted to *Environmental Pollution*.

3.0 ABSTRACT

The global threat of environmental plastic pollution has been well-established, but improved understanding at smaller scales may improve understanding of the complex plastic cycle and help to illuminate specific strategies. The Latin American and Caribbean (LCN) has the highest regional fraction of plastic in the waste stream but the lowest regional recycling rate, relative to other world regions. However, the region has remained relatively understudied in the context of plastic waste management and environmental plastic pollution. To inform gaps in knowledge related to plastic waste management in the LCN, a plastic intervention framework proposed by Jambeck (2016) guided data synthesis, modeled estimates, and evaluation of plastic waste management in the region. Based on the findings, the LCN region generated 24.2 Mt of plastic waste in 2020, and 7.15 Mt of this were mismanaged within the region. Further, an estimated 793,00 metric tons may have entered the ocean from land-based sources. While upper-middle income countries in the region play a key role in terms of quantities of waste generation and consequent pollution, high income countries in the Caribbean appear to have substantially high levels of per capita mismanaged plastic waste that contribute to a disproportionate amount of plastic waste given the small populations, which is especially concerning given the area to coastline ratio in these countries. The present study provides baseline knowledge around plastic waste management to define regionally specific challenges and inform future efforts for efficient plastic waste management in the region. Further this research substantiates the need for continued efforts to mitigate mismanaged plastic waste and support coordinated actions toward reducing plastic consumption, improving plastic waste management, and preventing losses to the environment.

3.1 INTRODUCTION

Plastic pollution is a rapidly accumulating modern challenge with widespread impact felt around the world. Over 8.30 billion metric tons (MT) of plastic has been generated since the 1950s, and 369 million metric tons (MMT) were produced in 2019 alone, showing no signs of slowing down. The global status of plastic waste management is understandably complex given the transboundary nature of plastic pollution. Over 6.3 billion metric tons of plastic waste has been generated since production began, but where and how waste is generated can vary drastically from place to place within the global system. Waste generation is positively correlated with urbanization, and high-income countries with more developed economies typically produce more waste and have higher waste collection rates (Kaza et al. 2018). In turn, low- and middle-income countries and regions that have rapidly developing economies but lack developed waste management infrastructure see the highest amounts of mismanaged plastic waste generation and plastic inputs into the ocean (Jambeck et al. 2015, Lebreton and Andrady 2019).

Compared to other world regions, the Latin American and Caribbean (LCN) region has the third highest waste generation rate after North America (NAC) and Europe and Central Asia (ECA) and the highest regional proportion of plastic in the waste stream (12.4%; Kaza et al. 2018), which is a driver of higher levels of plastic leakage into the environment (Ryberg et al. 2019). That said, the region has a relatively high rate of waste collection (84%), however, waste management infrastructure is still developing in some countries in the region, with many cities in LCN lacking waste management infrastructure entirely. As such, the LCN has the lowest regional recycling rate in the world (4.5%; Kaza et al. 2018), and some reports indicate that some solid waste in the region is disposed of in open areas near or in rivers or washed away during storms (Malizia and Monmany-Garzia 2019). In 2015, the LCN generated 19 MMT of plastic waste, of which 7.9 MMT were unsoundly managed, making the region the third most generating region in the world (Lebreton and Andrady 2019). Despite the region experiencing challenges in plastic waste management, the region has been understudied in the global context of plastic waste management (Blettler et al. 2018, Malizia and Monmany-Garzia 2019).

Waste generation is positively correlated with urbanization, and high-income countries with more developed economies typically produce more waste and have higher waste collection rates (Kaza et al. 2018). The LCN region has particularly high rates of urbanization (~81%), similar to that of NAC, which has the highest regional urbanization rate, however, LCN's regional per capita waste generation rate (0.99 kg/day) is less than half that of North America's (2.20 kg/day; Kaza et al. 2018). Further, global analyses find that waste disposal methods improve as income levels increase and consumption becomes less coupled with GDP, however, the LCN region has lower fractions of inadequately managed plastic waste relative to income levels in the region (Jambeck et al. 2015, Kaza et al. 2018). Most countries in the LCN region are considered upper-middle income (UMC), where the highest growth in waste generation rates (related to increased consumerism) and plastic in the waste stream is occurring globally (Jambeck et al. 2015, Kaza et al. 2018). Lower- and middle-income countries that have rapidly developing economies but lack developed waste management infrastructure see the highest amounts of mismanaged plastic waste generation and plastic inputs into the ocean (Jambeck et al. 2015, Lebreton and Andrady 2019). In 2016, LCN had a population of 638 million people, and while there are modernized waste management systems in the region, these are largely based on income level (Kaza et al. 2018). Further, with nearly 120,000 km of coastline and high concentrations of the population located within 50 km of these coastlines (Juan 2001, Center for International Earth Science Information Network (CIESIN) Columbia University 2018), there may be a pronounced risk of plastic marine debris inputs from coastal populations in the LCN.

With growing populations and increasing urbanization in the region, economic growth without fully developed infrastructure could lead to increased leakage of plastic into the environment. To illuminate regional patterns in plastic waste management and pollution, this study follows a plastic intervention framework proposed by Jambeck (2016) providing syntheses of data relevant to plastic production, alternatives to plastic, plastic waste generation, plastic waste infrastructure, and leakage of plastic waste into the environment.

3.2 METHODS AND MATERIALS

3.2.1 Description of region

The present study encompasses the Latin America and Caribbean region as defined by the World Bank country groups (The World Bank 2020). For this study, each country was categorized by four income groups: high income (HIC), upper-middle income (UMC), lower-middle income (LMC), and low income (LIC) according to The World Bank (2020), and three sub-regions: Caribbean, Central America, and South America. Additionally, six economic states in the geographic region were included in the analysis which are territories of other nations: Anguilla and the Falkland Island (British territories) and French Guiana, Guadeloupe, Martinique, and Montserrat (French territories). All countries and territories included in the analysis are listed in Table 3.1 by their income classification.

As part of the modeling procedures, each country's total populations in 2020 were sourced from the Central Intelligence Agency (CIA) 2020 World Factbook. The related coastal populations for each country were determined using gridded world population density data at 1 km resolution from the Center for International Earth Science Information (CIESEN) Global Rural-Urban Mapping Project (GRUMP) for 2020 in ArcMap 10.8. The rasterized population density data were extracted to regions of each country that were within 50 km of the coastline and summed as to determine each country's coastal population.

Table 3.1 List of LCN countries by World Bank 2020 economic status
(Bold states represent sovereign territories)

Low Income (LIC)	Lower Middle Income (LMC)	Upper Middle Income (UMC)	High Income (HIC)
Haiti	Bolivia	Argentina	**Anguilla**
	El Salvador	Belize	Antigua and Barbuda
	Honduras	Brazil	Aruba
	Nicaragua	Colombia	Bahamas, The
		Costa Rica	Barbados
		Cuba	British Virgin Islands
		Dominica	Cayman Islands
		Dominican Republic	Chile
		Ecuador	Curacao
		French Guiana	**Falkland Islands**
		Grenada	**Martinique**
		Guatemala	Panama
		Guadeloupe	Puerto Rico
		Guyana	Sint Maarten (Dutch part)
		Jamaica	St. Kitts and Nevis
		Mexico	St. Martin (French part)
		Montserrat	Trinidad and Tobago
		Paraguay	Turks and Caicos Islands
		Peru	Uruguay
		St. Lucia	Virgin Islands (U.S.)
		St. Vincent and the Grenadines	
		Suriname	
		Venezuela, RB	

3.2.2 Plastic intervention framework

Plastic waste management is a complex system with from plastic production to plastic waste treatment or mismanagement. Previous research focused on regional analyses of plastic waste management has examined the material through the lens of the plastic intervention framework by Jambeck (2016; Figure 3.1). Contextual background and procedures for assessing LCN through each intervention point are described in the following sections.

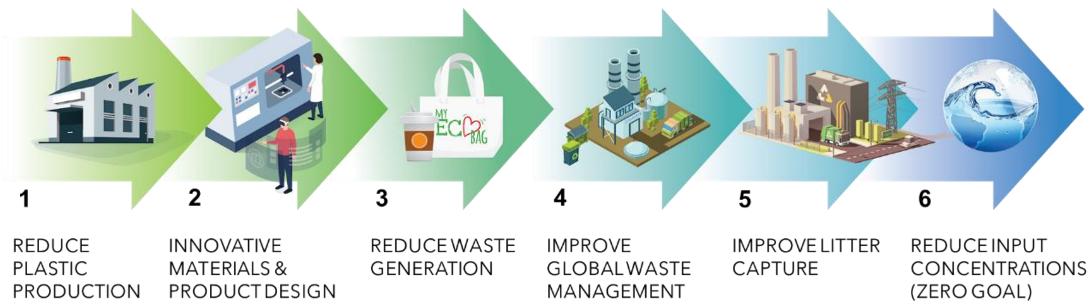

Figure 3.1. Plastic intervention framework (Jambeck 2016).

Plastic production, innovative materials, and product design

The first two intervention stages, plastic production and materials and product design, are considered together due to limited availability of industry data. Data regarding plastic production are largely inaccessible to the public and so, data regarding plastic production was limited to the region as a whole as provided in an annual report from Plastics Europe. The annual contribution of LCN plastic production were collected from reports available from 2005 to 2019 in terms of proportion of global production. Based on global quantities of plastic produced annually available in the same reports, the quantity of plastic produced by LCN each year were calculated along with the cumulative production of plastics in the region from 2005 to 2019 (Plastics Europe 2020).

Like virgin plastic production, data on bio-based plastic production is limited globally, however, European Bioplastics provides annual reports with proportions of production reported by world region, with the most current report being from 2019. These proportions were collated for the LCN region for comparison to other regions.

Plastic waste generation

Data for per capita waste generation rates and waste composition with respect to plastic were aggregated for the LCN region by country from, Kaza et al. (2018). To estimate plastic waste generation (PWG) within each country, national waste generation rates were converted to mass per year based on 2020 populations in both total and coastal regions of LCA and multiplied by the proportion of plastic reported for each country. For countries in which no plastic proportion was available values were supplemented by estimates from Law (2020).

Plastic waste management

Information regarding waste collection trends in the region as well as the proportion of various waste treatment methods used by each country were sourced from Kaza et al. (2018). Data on treatment methods were used to determine what proportion of each country's waste management methods are considered inadequate. Inadequate here refers to categories of waste treatment methods provided in (Kaza et al. 2018) which include open dumping, landfilling (controlled, sanitary, or unspecified), recycling, composting, anaerobic digestion, incineration, advanced thermal treatment, deposition in waterways, other methods (i.e., open burning or burying), and waste that is unaccounted for. Because of the natural discrepancy in data collection, methodologies, reporting, and inconsistent units, definitions, etc., data is only provided for treatment systems that have reliable sources.

Inadequate management included the fraction of national waste streams that were not treated in controlled settings (i.e., open dumps or waterway deposition), cannot be accounted for, and other methods (i.e., open burning or burying) that were detailed for each country. For the countries that reported a treatment value for the Unspecified Landfill category, the associated waste management literature for the country was reviewed and its inclusion in the inadequately managed fraction was decided on a case-by-case basis. Lastly, brief insight on the international trade of plastic scrap is provided based on methods from Brooks et al. (2018) and supplemented with updates from Chapter 2 (Brooks and Jambeck In preparation).

Quantities of mismanaged plastic waste (MMPW) were estimated based on modeling procedures developed by Jambeck et al. (2015) and were calculated for both total and coastal LCN populations for 2020. This model combines the proportion of inadequately managed waste and the proportion of littered plastic waste, for which a constant 2% value was used across all countries based on litter and waste reports from the US EPA and Keep America Beautiful, which provides one of the only estimates of the proportion of waste that is littered (Mid-Atlantic Solid Waste Consultants 2009, US EPA 2014).

Litter capture

Land-based inputs of plastic waste were estimated based on the procedure from (Jambeck et al. 2015). This method generates an estimated range of plastic marine inputs based on low, medium, and high input scenarios which are based on 15%, 25%, and 40% of MMPW, respectively. Plastic marine debris inputs were calculated for each LCN country with coastal populations for 2020. Plastic marine debris quantities for each country were then aggregated based on LCN sub-region and economic classifications. Characterizations of land-based plastic debris were generated based on results from the 2020 International Coastal Cleanup (Ocean Conservancy 2020), which reports values for the top ten most documented litter items in global cleanups, 100% of which are plastic.

3.2.3 Future scenarios

Plastic waste generation and mismanaged plastic waste quantities were projected to 2030 and 2050 for both total and coastal population in the LCN, and plastic marine debris quantities were estimated for the same year based on the procedures developed by Jambeck et al. (2015). Population sizes 2030 and 2050 were sourced from (Kaza et al. 2018) and based on the procedure from Jambeck et al. (2015), each country’s plastic composition for each country was increased annually by 0.19% to account for growth in plastic consumption by 2030 and 2050.

3.3. RESULTS

3.3.1 Descriptive statistics

A total of *N* = 48 countries were included in the analysis. Under World Bank economic classifications for 2020, only one LIC country was included in the analysis (i.e., Haiti), four LMC, 22 UMC, and 21 HIC, indicating that most countries in the LCN are UMC or HIC countries Table 3.1. LCN countries were further segregated into three sub-regions, Caribbean (*n* = 26), Central America (*n* = 8), and South America (*n* = 14). By joint distribution between subregion and economic classification, most countries in the analysis (33%; *n* = 16) were HIC and located in the Caribbean sub-region, followed by (21%; *n* = 10) UMC countries in South America, and (19%; *n* = 9) UMC countries in the Caribbean. The only LIC country was Haiti in the Caribbean subregion (Table 3.2).

Table 3.2. Contingency table showing count of LCN countries in each subregion and economic classification pairing

	Economic classification				
	HIC	*UMC*	*LMC*	*LI*	*Total*
Subregion					
Caribbean	16	9	0	1	26
Central America	1	4	3	0	8
South America	3	10	1	0	14
Total	20	23	4	1	48

The main independent variables for the analysis included population, per capita waste generation rates, waste stream composition with respect to plastic, and the fraction of waste that was managed inadequately as determined by waste treatment methods described by Kaza et al. (2018). Visual inspection of the distributions of each variable indicated that the distributions were generally extremely skewed right across the countries. While it is widely agreed that normality can be assumed for $N \geq 30$ (Dixon and Leach 1978), here, outliers heavily influence the mean, particularly among the outcome variables of the model such as plastic waste generation and mismanaged plastic waste. For this reason, median is reported as the measure of central tendency for the variables, however, means and standard deviations for each variable across the countries are provided for comparison when relevant.

This analysis explores plastic waste generation and management among entire country populations as well as those populations that are coastal (i.e., within 50 km of the coastline) and descriptive statistics are provided here for both (Table 3.3). The total population for the region was estimated to be 552 million, with the largest total population in Brazil (172 million, and smallest total population in the Falkland Islands (3,190), with the median population of LCN countries being 543,000 people. The South American subregion was estimated to contribute 65% of the region's total population, followed by Central America (28%), and the Caribbean (6.9%).

Coastal waste generation quantities were generally reflective of the proportion of each country's population that was considered coastal. With the two landlocked countries, Bolivia, and Paraguay, excluded, the total coastal population for the region was 205 million (equivalent to 37% of the region's total population), with the largest and smallest coastal populations in Brazil (64.2 million) and the Falkland Islands (3,190), respectively. More than half (n = 25) of the countries had equivalent total and coastal populations (i.e., 100% of the populations were considered coastal). Colombia and Guatemala had the smallest proportion of coastal populations at 17%. Based on visual inspection, the populations in LCN are generally concentrated along coastlines in LCN.

By region, the proportion of populations changed compared to the total populations. While the South American subregion still contributed most (63%), the Caribbean subregion contribution increased to 18% and Central America reduced to 19%.

Table 3.3. Summary statistics for total and coastal populations in LCN countries in MMT

Population	*n*	*min*	*max*	*mean*	*sd*	*median*
Total	48	.0032	172	11.5	29.9	0.543
Coastal	46	.0032	64.2	4.46	10.3	0.392

3.3.2 Plastic Intervention Framework

Plastic production and innovative product design

Compared to other world regions, LCN has historically contributed a relatively small amount to global plastic production, with an average of 4.4% of global production of plastic annually from 2015 to 2019. In this same period, LCN has produced a cumulative 192 MMT of virgin plastic resin, with a 2.6% CAGR. At this rate, LCN will generate an estimated 379 MMT of plastic resin by 2030 (Figure 3.2), which is just 3% greater than that which was produced globally in 2019 alone. Further, a recent assessment of plastic product exports from LCN shows that LCN exported 9.8 MMT of plastics in 2018, mostly (26%) in the form of final manufactured plastic products (2.5 MMT). Plastic packaging made up 0.7 MMT of plastic exports that same year (Barrowclough and Vivas Eugui 2021).

There are few examples of innovative product design in the LCN region, though many national and regional policies are targeting shifts toward circular economy initiatives (Schröder et al. 2020), which will likely include efforts for around development of more sustainable materials and product systems. By proportion, LCN contributes a greater fraction of global bio-plastic production than that of virgin plastic production in the region (Figure 3.3).

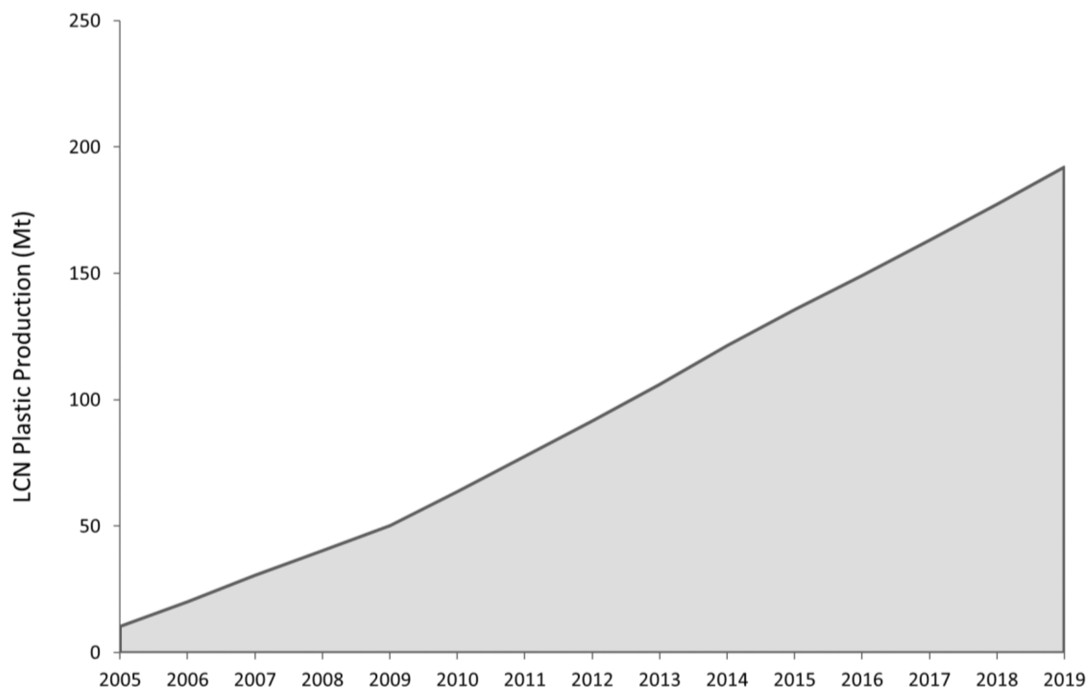

Figure 3.2. Cumulative growth in plastic production in LCN.

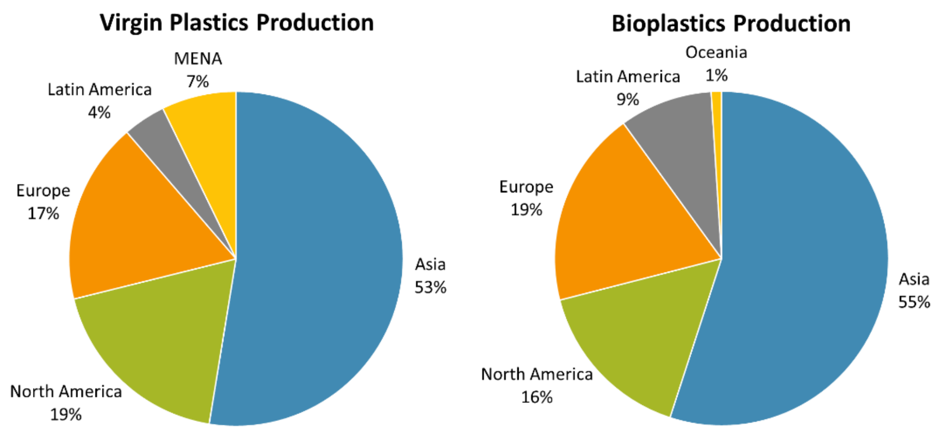

Figure 3.3. LCN plastic (left) and bioplastic (right) production relative to other regions.

Plastic waste generation

For the *N* = 48 countries included in this analysis, per capita waste generation rates ranged from 0.41 kg per person per day in Suriname (UMC; South America) to 4.46 kg per person per day in the US Virgin Islands (HIC; Caribbean), with the median per capita waste generation rate being 1.04 kg per person per day (*M* = 1.36, *SD* = 0.91). By sub-region, the Caribbean had the highest per capita waste generation rate, 1.74 kg/day, which was twice that of both the Central and South American sub-regions. By income, the HIC group had the highest per capita waste generation rate (2.07 kg/day) compared to 0.58 kg/day in the only LIC country, Haiti.

Most waste generated in LCN (52%) is food and green waste, followed by a large proportion of unspecified 'other' waste, and paper and cardboard (13%). Across the whole region, the proportion of plastic ranges from 6.4% in the British Virgin Islands (HIC; Caribbean) to 23% in St. Kitts and Nevis (HIC; Caribbean) with the median plastic proportion being 13% (*M* = 13, *SD* = 3.29). By sub-region, the Caribbean has the highest plastic fraction at 13.5% compared to 11.9% in the South American sub-region. The HIC group also had the highest plastic fraction (13.2%), while the LMC group had the lowest plastic fraction (11.6%).

When the waste generation rate and plastic fraction are considered together, the median per capita plastic waste generation rate is 0.13 kg/day, with the lowest being in Suriname (0.05 kg/day) and the highest being in the US Virgin Islands (0.58 kg/day). All countries with the top ten highest per capita plastic waste generation rates are HIC island states in the Caribbean, except for Saint Lucia which is a UMC country in the same sub-region. Across the Caribbean sub-region, the per capita plastic waste generation rate is 0.23 kg/day, followed by South America (0.18 kg/day), and Central America (0.11 kg/day). By income category, the HIC group has the highest per capita plastic waste generation rate (0.26 kg/day), while the LIC group has the lowest (0.07 kg/day).

Table 3.4. Summary statistics for waste generation variables for countries in LCN (WGR, waste generation rate; PWGR, plastic waste generation rate).

Variable	*unit*	*N*	*min*	*max*	*mean*	*sd*	*median*
WGR per person	kg/day	48	0.41	4.46	1.36	0.91	1.04
Plastic composition	%	48	6.35	23.4	13.0	3.29	12.6
PWGR per person	kg/day	48	0.05	0.58	0.17	0.11	0.13

The LCN generated an estimated 24.4 MMT of plastic waste in 2020. Of this mass, five countries contributed to 77%, namely, Brazil, Mexico, Argentina, Colombia, and Venezuela. Brazil, a UMC country in the South American subregion, was the largest generator of plastic waste at 8.86 MMT, representing 36% of the region’s total plastic waste generation (Figure 3.4). Conversely, Montserrat, a UMC country in the Caribbean subregion contributed the least, with only 273 tons (<0.01%) of plastic waste generated by the territory in 2020. By sub-region, the largest generator of plastic waste was the South American region, which generated 16.2 MMT of plastic waste in 2020, representing 66%. The smallest quantity was generated in the Caribbean region (1.7 MMT, equivalent to 6.7% of the region's total). By income category, the UMC group, generated 21.9 MMT in 2020, representing 89% of the whole region's plastic waste. In contrast, the LIC group generated the least (307,000 metric tons, equivalent to 1.3% of the region's total).

Table 3.5. Summary statistics solid waste generation (SWG) and plastic waste generation (PWG) for total and coastal regions of countries in LCN in metric tons.

Variable	*min*	*max*	*mean*	*sd*	*median*
Total (*N* = 48 for all)					
SWG	354	6.51×10^{7}	4.11×10^{6}	1.16×10^{7}	2.04×10^{5}
PWG	58.1	8.86×10^{6}	5.09×10^{5}	1.48×10^{6}	2.80×10^{4}
Coastal (*N* = 46 for all)					
SWG	354	2.44×10^{7}	1.61×10^{6}	3.88×10^{6}	1.45×10^{5}
PWG	58.1	3.31×10^{6}	1.99×10^{5}	5.15×10^{5}	2.21×10^{4}

When LCN sub-regions and income categories are considered together, the largest waste generation rate and plastic waste generation rate are seen in HIC countries in the Caribbean region, while the lowest of each are in UMC countries in South America. The UMC countries in South America generated the largest quantity of plastic waste (15.4 MMT), compared to the lowest combination generator, LMC countries in South America, which generated 207,000 metric tons in 2020.

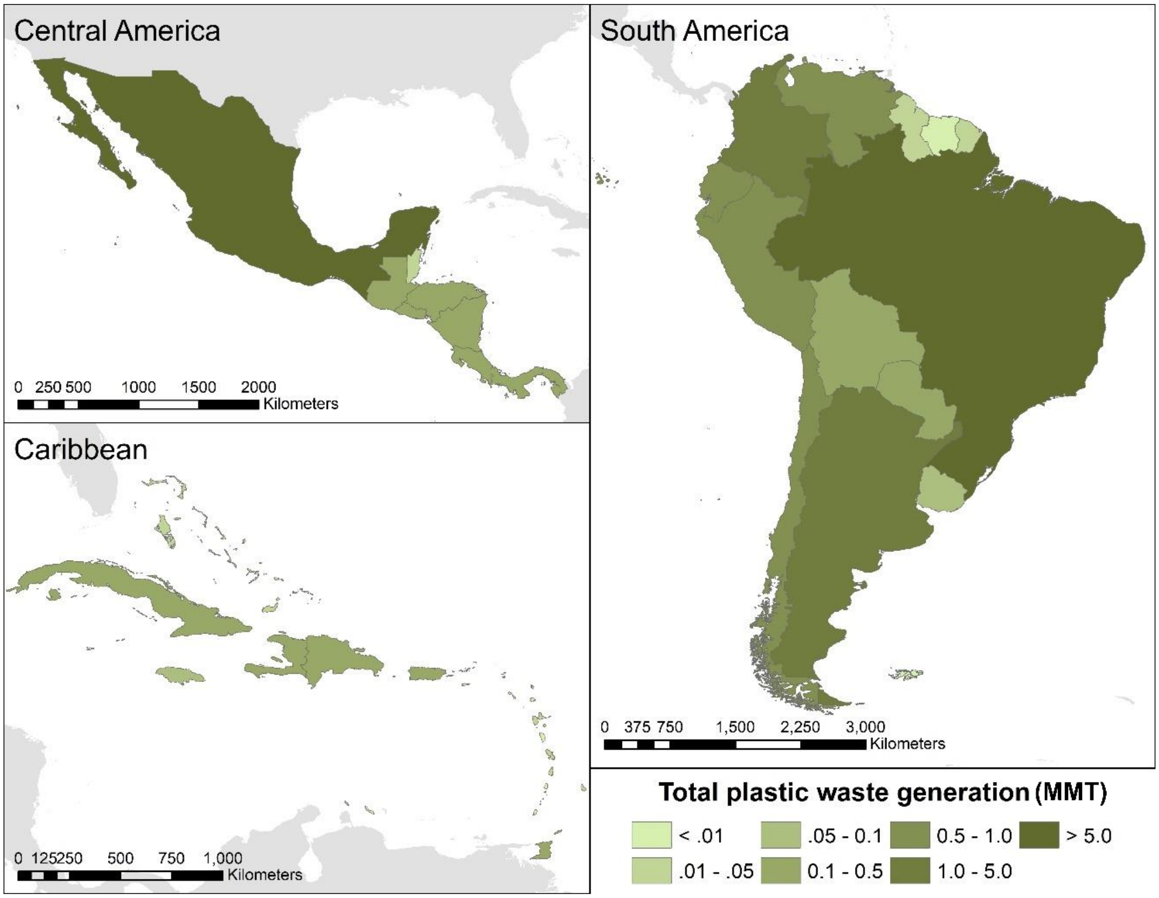

Figure 3.4. Total plastic waste generation by country.

Plastic waste management

Data on waste collection in and treatment mechanisms is somewhat limited for the LCN region, however, the region has a collection coverage rate of 84%, with most collection occurring in urban areas, and 30% of collection occurring in rural areas. From Kaza et al. (2018), nine LCN countries report rural waste collection and the units of reporting vary by coverage based on percentage of households, population, geographic area, and total waste. For those that do report, St. Vincent and the Grenadines reported the highest percentage of rural waste collection at 95% of rural households, followed by Ecuador (63%), and Peru (38%). These three countries also reported the highest coverage of urban households with Ecuador reporting 99% coverage of urban households, followed by Colombia and St Vincent and the Grenadines (97%), and Peru (95%).

For this study, adequate waste treatment methods included composting, controlled landfills, incineration, recycling, and sanitary landfills with gas collection. Few countries reported composting as a major treatment method, with all reported being 1% or less, despite the high levels of organic material in the waste stream. Similarly, only two countries reported values for incineration with Chile reporting 0.14% and British Virgin Islands reporting 80% for treatment of the waste stream. More countries (n = 18) reported values for recycling as a waste treatment ranging from 0.37% in Chile to 21% in the Cayman Islands. Sanitary landfills with landfill gas collection systems were reported by n = 16 countries and ranged from 0.1% in Dominican Republic to 89% in Colombia. HIC countries had the highest proportion of waste disposed in controlled landfills (59%), however, UMC countries had the highest proportion (45%) of waste managed via sanitary landfill systems. LMC countries had the highest recycling fraction at 12% compared to 9% in HIC and 6.7% in UMC countries.

Inadequate waste management included open dumping, unspecified landfills that did not have supporting documentation for treatment, waterways, and marine bodies, unaccounted for waste, and a catchall other category, which included burning, burying, or disposing in bodies of water. Across n = 21 countries that reported values, open dumping ranged from 4.0% in Colombia to 84% in Turks and Caicos, with the average for the region being 36%, although Kaza et al. (2018) reports a region-wide open dumping rate of 26.8%. Many countries (n = 23) reported unaccounted for waste, which ranged from

0.1% in El Salvador to 98% in Curacao. Both UMC and HIC had high values reported for unspecified landfills (81% and 67%, respectively).

Taken altogether, inadequately managed waste by country ranged from 1% in Barbados to 100% in Suriname, with the median being 32% (*m* = 37%, *sd* = 27; Table 3.6). By income group, the LIC (i.e., Haiti) saw the highest level of inadequately managed waste (90%), followed by LMC (47%), and UMC and HIC were tied at 35% each. By sub-region, South America saw the lowest proportion of inadequately managed waste (37%), while Central America saw the highest (43%). Of the plastic waste generated in the region, 6.66 MMT (27%) were inadequately managed, and 489,000 metric tons were littered. By country, the quantity of inadequately managed waste ranged from 38 metric tons in Antigua and Barbuda to 2.07 MMT in Brazil. By sub-region, South America had the highest quantity of inadequately managed waste at 3.84 MMT, equivalent to 58%. The UMC income group contributed to 84% of all inadequately managed waste, with most in the South American region. Table 3.6 summarizes the inadequately managed waste by percentage and quantities generated by both the total and coastal populations.

Table 3.6. Summary statistics for quantity of waste that is inadequately managed in LCN countries in metric tons.

Variable	*n*	*min*	*max*	*mean*	*sd*	*median*
Inad. (%)	48	1	100	36.8	27.2	32.1
Total	48	18.7	2.06×10^6	1.39×10^5	3.41×10^5	1.08×10^4
Coastal	46	18.7	7.69×10^5	6.50×10^4	1.33×10^5	6.98×10^3

Plastic leakage

A total of 7.15 MMT of plastic waste were mismanaged by the region in 2020, representing 29% of the total plastic waste generated in the region. Of, this 48% was in Brazil and Mexico (Figure 3.5). Montserrat had the lowest quantity of country-wide mismanaged plastic waste, having generated 93 metric tons (Table 3.7). Like the inadequately managed waste quantities, the UMC income group had the highest quantity of total mismanaged plastic waste, representing 84% of the region's total. Most (58%) of this was from South American countries. In contrast, the LIC group generated the smallest quantity (292,000 metric tons) across the income groups. Taking the sub-region and income groups together, HIC Central American countries had the smallest quantity of mismanaged plastic waste, contributing only 1% to the region's total.

When normalized by population, mismanaged plastic waste ranged from 0.001 kg/day per person in St. Vincent and the Grenadines to 0.118 kg/day in Turks and Caicos, with the median being 0.017 kg/day per person for the region (Table 3.7). Across income groups, per capita mismanaged plastic waste ranged from .014 kg/day in the LMC group to 0.039 kg/day in the HIC group, and by sub-region ranged from 0.017 kg/day in Central America to 0.033 kg/day in the Caribbean.

When the sub-region and income group were considered together, the highest per capita mismanaged plastic waste rate was 0.045 kg/day in HIC Caribbean countries, which was twice that of the next highest combination of the categories, the LIC-Caribbean group (represented solely by Haiti), which had a per capita mismanaged plastic waste rate of 0.025 kg/day. The combination with the lowest rate was the LMC group, which has a per capita mismanaged plastic waste rate of 0.012 kg/day.

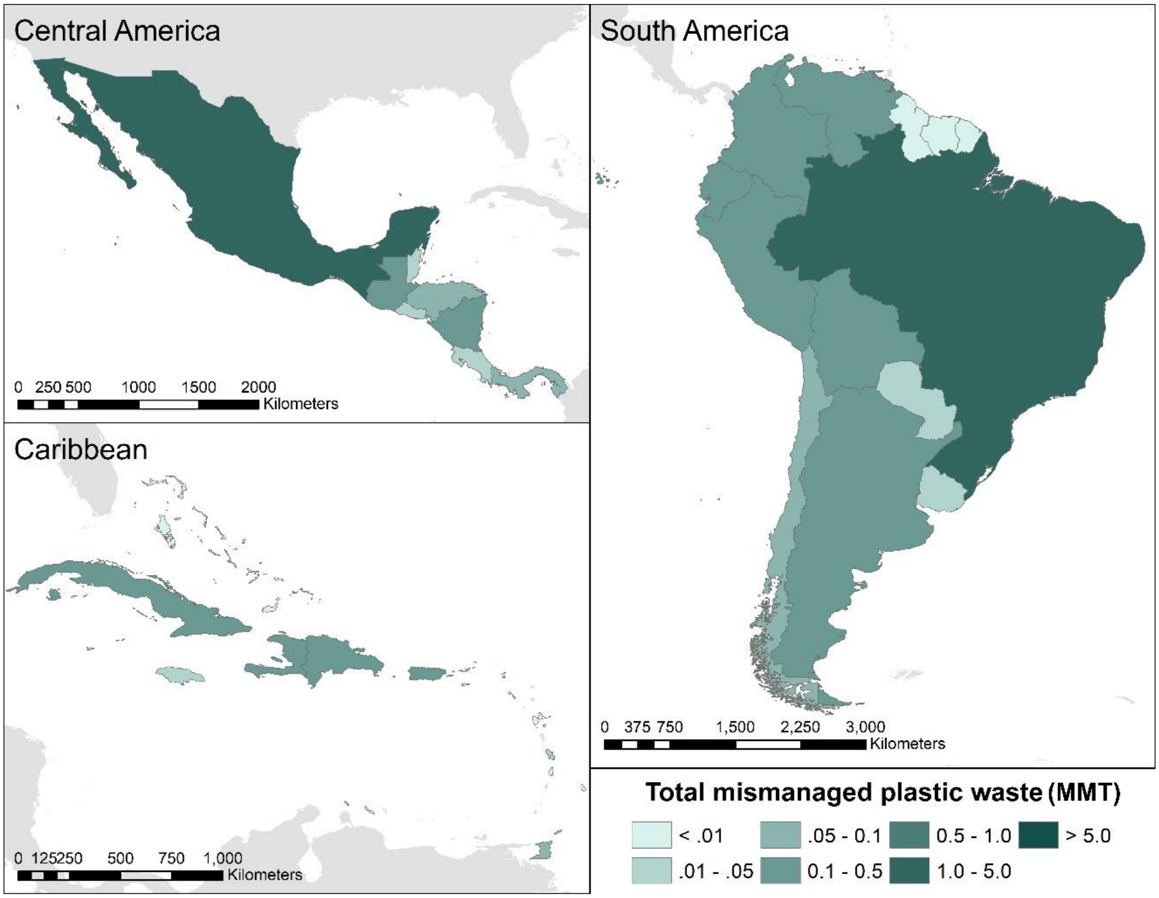

Figure 3.5. Total mismanaged plastic waste by country.

Table 3.7. Summary statistics for plastic leakage quantities in LCN countries.

Variable	*unit*	*min*	*max*	*mean*	*sd*	*median*
Total (*N* = 48)						
Littered PW	metric tons	5.5	177,000	10,200	29,400	560
MMPW	metric tons	93	2.23×10^6	149,000	366,000	11,300
MMPW per person	kg/day	0.001	0.118	0.025	0.026	0.017
Coastal (*N* = 46)						
Littered PW	metric tons	5.5	66,300	3,970	10,200	441
MMPW	metric tons	93	835,000	69,000	141,000	7,750
MMPW per person	kg/day	0.001	0.118	0.025	0.026	0.017
Plastic marine debris	metric tons	23.3	209,000	17,200	35,300	1,940

Land-based plastic marine debris

Jambeck et al. (2015) estimated global plastic inputs from land to the sea based on plastic waste generated, mismanaged, and proportions that had potential to enter the ocean based on populations within 50 km of the coastline. For the LCN, the coastal populations generated 9.14 MMT of plastic waste in 2020, which represents 37% of the region's total tonnage generated. Of this, 3.17 MMT were mismanaged, equivalent to 35% of the coastal plastic waste generated, and 13% of region's total plastic waste generated. Brazil contributed to 26% of the coastal mismanaged plastic waste, followed by the Dominican Republic (10%), and Haiti (8.6%). Six countries contributed less than 0.01% of the coastal mismanaged plastic waste: Montserrat, St. Vincent and the Grenadines, Antigua and Barbuda, Falkland Islands, Dominica, and Grenada. Figure 3.6 provides a visualization of the coastal mismanaged plastic waste across the LCN.

By sub-region, South America contributed to most (51%) of the region's coastal mismanaged plastic waste in 2020, having mismanaged 1.63 MMT along the sub-region's coastlines, followed by the Caribbean (33%), and Central America (16%). By income, the UMC group contributed to 75% of the region's coastal mismanaged plastic waste, having generated 2.39 MMT in 2020, followed by the HIC group (12%), LIC (8.6%), and LMC (4.3%). Across both the sub-region and income categories, UMC countries in South America saw the highest quantities of coastal mismanaged waste, while HIC countries in the South America saw the least among all the combinations of the two categories.

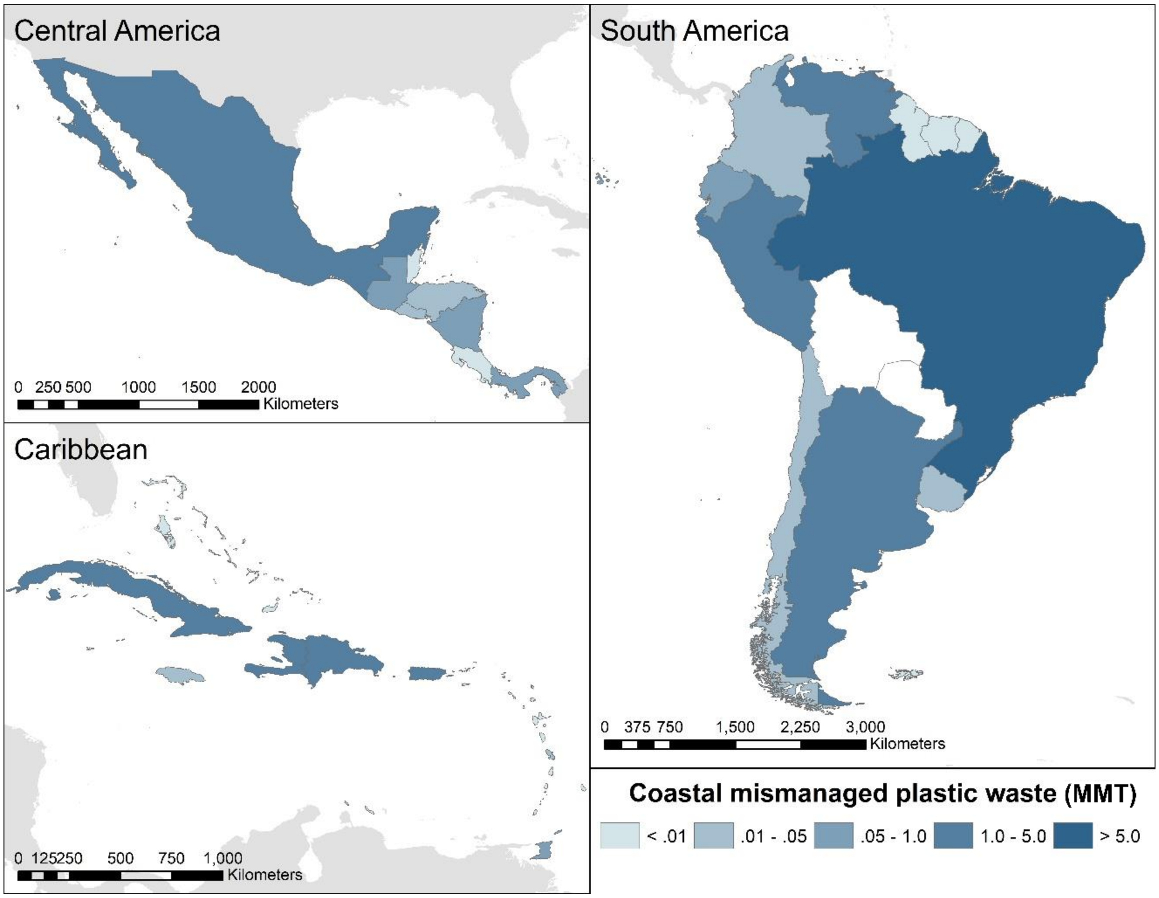

Figure 3.6. Coastal mismanaged plastic waste by country.

Based on the methods by Jambeck et al. (2015), between 476,000 and 1.27 MMT of plastic waste may have entered the ocean from populated regions of the LCN coasts, with the mid-estimate being 793,000 metric tons for 2020 (Table 3.8). The top ten countries together contributed 84% of this amount, while the bottom 29 countries contributed less than 1% individually, and collectively contributed to only 4.5% of the total plastic marine debris inputs. The South American sub-group had the highest quantity of land-based inputs of plastic marine debris at 408,000 metric tons, followed by Caribbean (259,000 metric tons), and Central America (126,000 metric tons). The UMC group had the highest quantity of plastic marine debris inputs across the income categories, while the lowest quantity was seen in the LMC group. In combination, the lowest quantity was observed in the HIC-South American countries, which contributed 1.4%, compared to the UMC-South American countries, which contributed 50%.

Table 3.8. Top ten countries by land-based plastic marine debris inputs.

Country	Econ. Class	Coastal pop. (x 10^6)	Plastic waste gen. rate (kg/ppd)	Fraction mism. (%)	Mism. plastic waste (metric tons x 10^3)	% of region total	Plastic marine debris (metric tons x 10^3)
Brazil	UMC	64.2	0.14	25.2	835	26	209
Dominican Rep.	UMC	8.86	0.11	93.7	330	10	82.4
Haiti	LIC	10.9	0.07	92.1	272	9	68.0
Peru	UMC	14.9	0.08	58.4	252	8	63.1
Mexico	UMC	21.2	0.13	23.0	227	7	56.6
Argentina	UMC	13.0	0.17	24.6	196	6	49.1
Venezuela	UMC	14.9	0.10	34.1	189	6	47.3
Puerto Rico	HIC	2.92	0.34	35.3	129	4	32.3
Cuba	UMC	8.54	0.06	61.8	125	4	31.2
Trinidad and Tobago	HIC	1.06	0.28	89.2	98.2	3	24.5
Top ten total		160			2,650		663
Region total		205			3,170		793

3.3.3 Future scenarios

Assuming BAU circumstances in which no improvements are made to waste management infrastructure, the LCN could generate 36.7 MMT of plastic waste in 2030 and 48.3 MMT in 2050, which would result in 7.15 MMT of mismanaged plastic waste in 2030 and 14.6 MMT in 2050. Along the coasts of LCN, land-based plastic marine debris may reach 1.18 MMT in 2030 and 1.43 MMT in 2050. At this rate, annual plastic waste generation, mismanaged plastic waste, and plastic marine debris inputs in the region will increase by 50% by 2030 and will have doubled by 2050. Finally, simple linear regression indicated that a cumulative 331 MMT could be generated by the region in 2030, followed by 1,170 MMT in 2050 (Table 3.9).

Growth in quantities of plastic waste that is expected to be generated and managed in the region does not vary geospatially, with the largest increases in terms of mass expected in Brazil, Mexico, Argentina, and Peru. However, by percent increase in growth, the US Virgin Islands, St. Kitts and Nevis, Antigua and Barbuda, and Cayman Islands—all high-income countries in the Caribbean subregion—may experience the largest growth in plastic waste generation, mismanaged plastic waste, and plastic marine debris in the coming decades.

Table 3.9. Annual and cumulative quantities of plastic waste generation (PWG), mismanaged plastic waste (MMPW) and plastic marine debris (PMD) forecasted for 2020, 2030, and 2050, and associated coefficient of determination for goodness of fit for each linear prediction.

Year	PWG (MMT)	MMPW (MMT)	PMD (MMT)
2020	24.4	7.15	0.79
2030	36.7	10.8	1.18
2050	48.3	14.6	1.43
Cumulative	1,170	349	36.3
R^2	.96	.97	.90

3.4 DISCUSSION

3.4.1 Plastic intervention framework

Plastic production

The LCN region is not a major producer of either fossil fuel derived or bio-based plastics relative to other regions. Although the region does export virgin plastics and products, only 2.9% of 336 MMT of global plastics exports in 2019 originated from the LCN (Barrowclough and Vivas Eugui 2021). Given the small role that the region plays in global plastic production, efforts for plastic material management may be more effectively focused toward improving the waste management system in the near term, which will be well aligned with regional efforts to shift toward the circular economy (Schröder et al. 2020). This is especially important considering that management of plastic alternatives has implications within the waste stream, and without proper education and infrastructure, have the potential to cause more damage than intended.

Plastic waste generation

While the median per capita waste generation rate determined here was much greater than the global average, it was comparable to rates seen in wealthy and urbanized regions like Europe and North America. Plastic waste generation appeared to be heavily influenced by high income countries in the Caribbean region, although the total quantities in this subregion were the smallest compared to South and Central America. Similarly, the plastic proportion in the region was driven by the high-income countries, in the Caribbean, which had the highest plastic fraction in the waste stream, as high as 23% in St. Kitts and Nevis, lending to the highest per capita plastic waste generation rate.

The estimated 24.4 MMT of plastic waste generated in the region in 2020 is comparable to previously published estimates. Lebreton and Andrady (2019) estimated that 19 MMT of plastic waste were generated in the LCN in 2015, a difference of 5.4 MMT than the estimate here and equivalent to 29% difference from Lebreton and Andrady's estimate for 2015 compared to the estimate reported here for 2020. The increase in values is not unreasonable assuming population growth after 2015 but may be derived from differences in values used for calculating waste generation. For example, both the present study and the Lebreton and Andrady (2019) work relied on national waste characteristics reported by the World Bank, however, the present study accessed more recently published data (Kaza et al. 2018) than Lebreton and Andrady (2019), which extracted waste values from Hoornweg and Bhada-Tata (2012).

Plastic waste management

Regionally, LCN ranks relatively high in terms of waste collection coverage after North America and Europe and Central Asia regions and has similar coverage as that seen in the Middle East and North Africa. However, of all regions, LCN has the smallest proportion of recycling (4.5%), compared to the global average of 14% (Kaza et al. 2018). Given high waste collection rates in the region, particularly in urban areas, compared to the rest of the world, the low rate is unexpected. That said, plastic recycling is growing in the region, and is more prominent in LMC countries.

Most plastic recycling in the region is targeted toward recycling polyethylene terephthalate (PET) and high-density polyethylene (HDPE; Schröder et al. 2020), however, only a small portion of municipal waste reaches recycling facilities since many waste streams are mixed and designated for landfills (Horodytska et al. 2019). Though not a major point of focus in this study, the LCN has not historically been a major contributor in the international plastic scrap trade as a region (Pacini et al. 2021), though Mexico was established as the 5th largest exporter in the world as of 2016 (Brooks et al. 2018). The relatively low recycling rates reported in the region, may contribute to the small quantities of exported scrap, and as such, the region likely does not significantly contribute to global mismanaged plastic scrap given low participation. Similarly, the region is not a major importer of plastic being the third lowest importer of plastic scrap (0.8% by mass) as of 2019 (as presented in Chapter 2).

While the region currently low rates of recycling, the LCN has the highest proportion of waste managed via landfill (69%), compared to the global average of 25%. South and Central America have a high proportion of landfill disposal which would indicate that much of the plastic waste generated in the region is disposed of via landfill storage, which drove the lowest sub-regional rate of inadequate management seen in Central American subregion. Despite the slightly higher levels of recycling in LMC countries, they still had high levels of inadequately managed waste given that many lacked advanced landfill systems, which is concerning given that 71% of microplastic presence in the region is attributed to inadequately managed plastic waste (Savino et al. 2018). Similarly, 11.52% of microplastics in the region have been attributed to littering which would suggest that mismanaged plastic waste could contribute around 82% of microplastics (Savino et al. 2018).

Plastic leakage

The estimated total mismanaged plastic waste for the region was 7.15 MMT, which aligned well with the Lebreton and Andrady (2019) model that previously estimated that LCN region generated 7.9 MMT of mismanaged plastic waste in 2020, which is slightly higher than the estimated quantity reported here. This is likely influenced by variation in values used for model parameters used to estimate the fraction of waste that was managed inadequately for each country. Lebreton and Andrady (2019) cited self-reported country-level values sourced from the Waste Atlas (2016), which could be evaluated against the values used here to supplement the findings of this analysis. The same study by Lebreton and Andrady (2019) estimated that the South

American sub-region generated 5.81 MMT in South America, and Brazil is the only country that was reported in the global top ten list of national mismanaged plastic waste generators in 2015, with an estimated 3.68 MMT after China, India, and Philippines. The mismanaged plastic waste estimates for LCN subregions here were higher than the ranges of mismanaged plastic waste for each sub-region estimated by the Lebreton and Andrady (2019) model, however, they were of similar magnitude and the totaled low estimate for the whole LCN region was only 0.21 metric tons greater than the estimate reported for the region here.

Table 3.10. Comparison between present study and estimates for 2020 by Lebreton and Andrady (2019) for LCN and sub-regions.

Region	Mismanaged plastic waste (Mt) in 2020				Distance from range	Comparison (present vs. L&A 2019)
	Lebreton and Andrady (2019)			Present study		
	Low	Mid	High			
Caribbean	0.56	0.73	0.87	1.08	.21	Greater than high
Central America	1.36	1.59	1.69	1.91	.22	Greater than high
South America	5.44	6.21	6.44	4.16	1.3	Less than low
LCN Total	7.36	8.53	9.00	7.15	.21	Less than low

Despite having the lowest rate of inadequately managed waste, South America generated the largest quantity of mismanaged plastic waste, and along with the total plastic waste generation, the total quantities were largely attributable to UMC countries in the region. Per capita rates of mismanaged plastic waste, however, revealed some unexpected results. The HIC countries in the Caribbean had the highest per capita mismanaged plastic waste rate in the region and were three times higher than the average rate for HIC countries based on values reported by Law (2020). Further, the only LIC-Caribbean country, Haiti, had the second highest rate across the income-region combinations. Globally, LIC countries typically have had the lowest per capita rates of mismanaged plastic waste (0.013 kg/day based on Law (2020)), but this could be driven by the high rate of inadequately managed waste in the country, 90%.

Recent research from Borrelle et al. (2020) estimated that plastic debris emissions from the LCN region would total to 3.09 MMT in 2020, which is nearly four times higher than the mid estimate presented here. However, their estimate is comprised of emissions into all aquatic ecosystems, which included major rivers, lakes, and oceans, while the estimate here only reflects emissions into the ocean from coastal populations. Data available from Borrelle et al. (2020) does not provide information regarding proportions of emissions that enter each type of aquatic ecosystem, so direct comparisons between marine emissions were not possible.

By subregion, South America plays a significant role in land-based plastic debris inputs into the ocean, despite having smaller proportions of plastic in the waste stream and lower per capita waste generation rates. However, the South America sub-region hosts the largest proportion of the LCN region's population, and so the cumulative waste generated across the number of people in the region drives the higher total quantities. The Caribbean sub-region contributed to the second largest input of land-based marine debris, but given the much smaller populations there, this is likely attributable to the substantial per capita mismanaged plastic waste rates. This is of particular concern given that many Caribbean countries have small land areas relative to their coastlines, which, when combined with high rates of precipitation, have been linked with higher quantities of plastic emissions to the ocean (Meijer et al. 2021).

It is well known that mismanaged plastic waste from communities can be transported to marine environments via river systems (Lebreton et al. 2017, Schmidt et al. 2017). Previous research indicated that only one river in the LCN region, the Amazon, is among the top ten most polluting rivers (Lebreton et al. 2017). More recently, Brazil, Guatemala, Haiti, Dominican Republic, and Venezuela were all identified as top 20 countries based on annual emissions of plastic into the ocean through riverine transport, with a total 1,962 rivers between these countries contributing to 100% of their emissions (Meijer et al. 2021).

Future scenarios

Based on the values estimated for 2030 and 2050, the model presented here estimates a quantity of mismanaged plastic waste that is reasonably compared to the prediction for 2025 by Jambeck et al. (2015), which estimated that the LCN region would generate 4.3 MMT of mismanaged plastic waste in 2025. In contrast, the present model estimates that only 793,000 metric tons of mismanaged plastic waste was generated in 2020, and grow to 1.18 MMT in 2030, which is only 27% of the quantity that was predicted for 2025 by Jambeck et al. (2015). However, the projected quantities for the total region (10.8 MMT in 2030 and 14.6 MMT in 2050), were closer to projections by Lebreton and Andrady (2019), which estimated that mismanaged plastic waste in the region would reach 11.6 MMT in 2040 and 14.1 in 2050, however, this still indicates slower growth predicted here than that predicted by Lebreton and Andrady (2019).

Particularly for the small island countries located in the Caribbean and countries with significant coastlines (e.g., Chile) relative to the land areas, future predictions of waste generation and management can inform planning strategies to prepare for size requirements, strategies for reduction, and diversion from waste storage methods (i.e., landfill diversion), which could become a serious challenge for small island countries with limited space for waste infrastructure and storage. Similarly, dispersion of mismanaged waste in these types of geographies presents a unique challenge as leaked items either concentrate on land, where it can cause severe blockage of stormwater and sewage systems causing flooding and raising the risk of mosquito-borne diseases (Chitotombe et al. 2014), or enter the ocean, where it can cause harm to wildlife and fragment into toxic size fragments (Wilcox et al. 2016).

3.4.2 Regional policy implications

Global policies targeting plastic pollution and improved management of plastic waste generally focus on plastic bags, single-use plastic items, and microbeads through a range of regulatory tools such as bans, limitations, and levies on certain items, as well as Extended Producer Responsibility (EPR) schemes, market-based instruments, and mandated recycling, which are carried out and enforced at various scales including the national, sub-national, and city levels.

Eleven LCN countries reported national policies in place or imminent (UNEP 2018b), most of which target plastic bag bans, though Ecuador's ban targets polystyrene. Similarly, in Antigua and Barbuda, in addition to a national plastic bag ban, there is also a ban on expanded polystyrene (often called "Styrofoam," although this is a brand name, hereafter called "EPS foam") food containers, utensils, and coolers. Most national-level plastic bag policies include a full ban in Antigua and Barbuda, partial ban in Chile, prohibition of polyethylene bags in commercial and retail businesses in Panama, and a specific ban on black polyethylene bags in Haiti. Two countries, Brazil, and Uruguay, have EPR schemes associated with plastic bag regulations (UNEP 2018a), and Uruguay and Colombia both have national levies in place targeting plastic bags.

Some bag regulations are accompanied by bans on other single-use plastic items. For example, the national ban on plastic bags in Belize is accompanied by a ban on EPS foam and plastic food utensils as well. Guyana and St. Vincent and the Grenadines have bans targeting EPS foam products as well. Information showing documented impacts from national bans are largely unavailable, however, the policy in Colombia has resulted in a 27% reduction in plastic bag use (United Nations Environment Program (UNEP) 2017). Costa Rica has passed comprehensive national policies focusing on banning a range of disposable plastic products with the goal of fully eliminating them by 2021 (UNEP 2018a). Lastly, Brazil is the first country in the region to propose legislation that would ban microbeads at the national level (UNEP 2018a). Further, five LCN countries, Antigua and Barbuda, the Bahamas, Bolivia, Paraguay, and Uruguay, include EPR measures like take-back schemes, deposit refunds, and waste collection guarantees as part of plastic bag regulations (UNEP 2018a).

In addition to national policies, sub-national policies implemented at the local level are in place in ten different locations throughout the region. Most local policies are plastic bag bans. Argentina and Argentina have documented sub-national/city level regulations, and the Dominican Republic regulates reuse and recycling of plastic bags at the municipal level (UNEP 2018a). Specifically, San Pedro La Laguna, Guatemala bans both plastic bags and EPS foam. Two of the largest cities in the region have widespread sub-national policies. One, Mexico City, has a combined ban and levy policy that requires retailers to charge a fee for plastic bags, and the policy further requires bags to be biodegradable. Similarly, Rio de Janeiro, Brazil has a levy in place that requires markets to both provide alternatives to plastic bags and take back bags for proper disposal, while also incentivizing the public to bring their own bags through shopping discounts and deposit schemes (Beveridge & Diamond P.C. 2009). Like the national policies, little published research information is available regarding the impact of these policies, although some success has been reported, though some research has indicated that regulatory incentives can be effective toward reducing plastic debris in the environment (Maes et al. 2018, Schuyler et al. 2018). The regulatory policy on plastic bags in Rio de Janeiro policy has reportedly resulted in a 24% annual reduction (Siqueira 2011). And, in Honduras, the plastic bag policy in the Bay Islands communities have seen complete elimination of plastic bags in Guanaja and an 80% and 50% reduction in use in Utila and Roatan, respectively (The Summit Foundation 2017).

Outside of direct bans and levies on specific single use plastic items, other promising forms of governance include deposit schemes (Vig and Kraft 2019) and extended producer responsibility (EPR), wherein manufacturers and producers are held legally responsible for the way in which their products are managed as waste. Policies targeted at producers of plastic products, such as economic tools like deposit schemes that allow consumers to return plastic items in return for a small financial incentive have been documented to be particularly successful for reducing mismanaged plastic waste (Schuyler et al. 2018). Globally, 17% of EPR policies target packaging, including paper and plastic (OECD 2016). Regionally, EPR schemes have been established in Chile, Mexico, Brazil, Argentina, and Colombia, with most of the programs focusing on electronic waste (OECD 2016). Under the 2005 Comprehensive Policy on Waste Management in Chile, a draft EPR framework was submitted in 2013 which aims to hold producers of priority products, including packaging, legally accountable for management systems for those products once they become waste. Under this law, PET bottles have a 12% collection rate achieved by the program in its early stage (OECD 2014). Similarly, in 2018, the Colombian Ministry of Environment issued a resolution for the management of packaging, including plastic through an EPR program, with a goal of requiring producers to include minimum recycled content in their products, which will increase by 2% annually (Sostenible 2018). Local EPR schemes include Guayaquil, Ecuador, where a deposit scheme is in place through the city's bus transit system in which riders can return plastic bottles for two cents each (Alcaldia de Guayaquil 2019).

At current, multiple international finance schemes are targeted toward the LCN region, with efforts from the World Bank, Inter-American Development Bank (IADB), focused on development of circular economy practices in waste management and recycling in Argentina, Belize, Bolivia, Brazil, Chile, Colombia, Suriname, Peru, and Uruguay. Similarly, the Global Environment Facility (GEF) has a specific focus on financing initiatives toward implementing circular economy concepts in plastic waste management, PET recycling, and waste valorization in Guyana, Peru, Suriname, and Uruguay (Schröder et al. 2020). Notably, all countries participating in these schemes are within the South American sub-region, except for Belize.

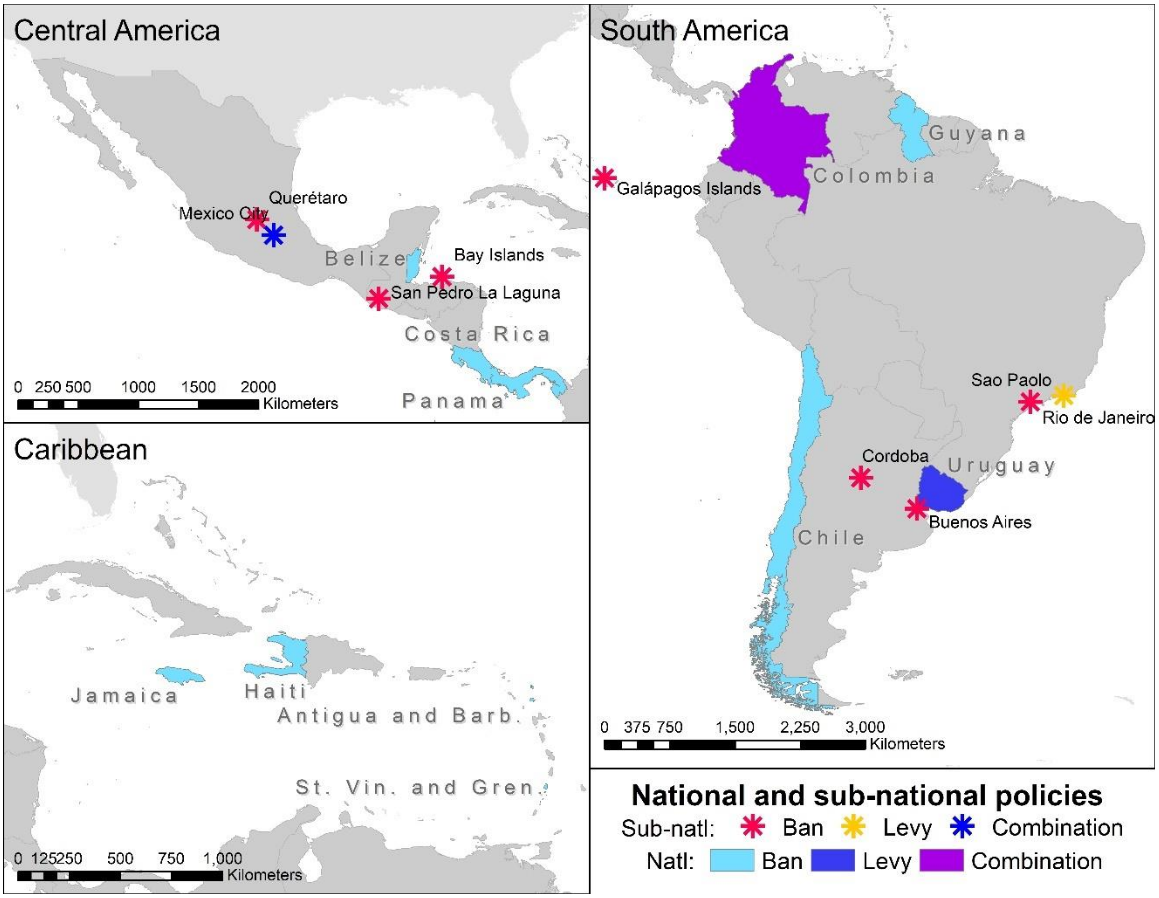

Figure 3.7. Map of policies targeted toward single-use plastic in LCN

3.5 CONCLUSIONS

At the global scale, the LCN region appears neutral, neither creating the highest or lowest quantities of mismanaged plastic waste and marine debris. This may inadvertently reduce attention toward the region which would support further advancements in infrastructure, research, and innovation. However, there are clearly reason for concern in the region, ranging from the large populations generating high quantities of plastic waste and large quantities of mismanaged plastic to the small island nations that have substantial rates of mismanaged plastic waste per person tied to high potential for leakage into the ocean given the ratio of coastlines to land areas. Further, some of the countries in the region have large GDPs relative to the rest of the world which may be contributing to the quantities of plastic waste being generated in countries like Brazil and Mexico. In contrast, some of the HIC Caribbean islands that have high levels of per capita waste generation rates have high GDP per capita, such as the US Virgin Islands. The relationship between national productivity, consumption, and mismanaged plastic waste may warrant further research. Additionally, although not a focus of the present analysis, the informal waste sector plays a significant role in countries in the LCN (Noel 2010, Botello-Álvarez et al. 2018), and more knowledge is needed to better understand the role that this group plays in regional management of plastic waste.

This work only considers plastics that are generated, managed, and leaked from municipal waste streams, though there are other sources of plastic in the environment not included in this analysis such as maritime sources like lost, discarded, and derelict fishing gear from both commercial and recreational fishing, aquaculture, and losses from shipping (Law 2017, Jambeck et al. 2020). Microplastics can enter the ocean via non-treated and treated wastewater discharges and from run-off from land where abrasion and exposure can cause plastic to fragment into smaller pieces (Ziajahromi et al. 2017). These are additive sources of plastic into the aquatic systems and ocean in the LCN region and regional research on microplastics is growing in the region (Kutralam-Muniasamy et al. 2020), despite remaining knowledge gaps. Maritime sources of plastic may warrant further investigation as this report focused primarily on municipal solid waste as a source of plastic input into the environment and waterways. Finally, the regional scale assessment of plastic waste management in LCN is still limited in that smaller scale variation and patterns are still not detectable at the regional scale. Likely there are subnational and local factors that play significant roles in the quantities of mismanaged plastic waste in the region, and research that builds on this may consider examining these sub-regions in more detail. Further, waste is inherently generated and managed at the community level. Given LCN's urban populations, plastic waste management in urban centers may reveal where there are disparities in the high urban collection rates, low segregation and recycling, and relationships with mismanaged plastic waste. For instance, Brazil's Sao Paolo was the fourth highest city in the world in estimates of mismanaged plastic waste estimates for 2015 (Lebreton and Andrady 2019). Ground-level, empirical research may help to continue filling gaps in knowledge around plastic waste management in the LCN region. Eventually data-rich, multi-scale models may provide the most in-depth comprehension of effective waste management strategies in cities, sub-regions, countries, and beyond.

REFERENCES

Sheavly SB (2005) Marine debris—an overview of a critical issue for our oceans. In: Proceedings, Sixth Meeting of the UN open-ended informal consultative processes on oceans and the law of the sea. http://www.un.org/Depts/los/consultative_process/documents/6_sheavly.pdf. Accessed 4 Aug 2015.

Alcaldia de Guayaquil. (2019). "En la Terminal Río Daule se inaugura máquina de reciclaje y trituración de botellas, el jueves 27 de junio. (In the Rio Daule terminal, the plastic bottles recycling machine is unveiled, on Thursday June 27)." Retrieved June 21, 2020, from https://guayaquil.gob.ec/Paginas/noticias/2019/junio/noticia5012.aspx.

Barrowclough, D. and D. Vivas Eugui (2021). Making Waves -- Plastic Production and Trade in Latin America and the Caribbean: The shift towards a Circular Economy. SELA/UNCTAD15. Virtual.

Beveridge & Diamond P.C. (2009). "Rio de Janeiro Imposes Retailer Responsibility for Plastic Shopping Bags." Latin American Region Environmental Quarterly (October), from http://www.bdlaw.com/news-697.html.

Blettler, M. C. M., E. Abrial, F. R. Khan, N. Sivri and L. A. Espinola (2018). "Freshwater plastic pollution: Recognizing research biases and identifying knowledge gaps." Water Research **143**: 416-424.

Borrelle, S. B., J. Ringma, K. L. Law, C. C. Monnahan, L. Lebreton, A. McGivern, E. Murphy, J. Jambeck, G. H. Leonard, M. A. Hilleary, M. Eriksen, H. P. Possingham, H. De Frond, L. R. Gerber, B. Polidoro, A. Tahir, M. Bernard, N. Mallos, M. Barnes and C. M. Rochman (2020). "Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution." Science **369**(6510): 1515.

Botello-Álvarez, J. E., P. Rivas-García, L. Fausto-Castro, A. Estrada-Baltazar and R. Gomez-Gonzalez (2018). "Informal collection, recycling and export of valuable waste as transcendent factor in the municipal solid waste management: A Latin-American reality." Journal of Cleaner Production **182**: 485-495.

Brooks, A. L. and J. R. Jambeck (In preparation). "Assessment of the international plastic scrap trade and the global impact of the Chinese import ban: An upate."

Brooks, A. L., S. Wang and J. R. Jambeck (2018). "The Chinese import ban and its impact on global plastic waste trade." Science Advances **4**(6).

Center for International Earth Science Information Network (CIESIN) Columbia University (2018). Gridded Population of the World, Version 4 (GPWv4): Population Density, Revision 11. Palisades, NY, NASA Socioeconomic Data and Applications Center (SEDAC).

Central Intelligence Agency (2020). The World Factbook. Washington, DC.

Chitotombe, J. W., S. J. I. J. o. D. Gukurume and Sustainability (2014). "The plastic bag 'ban'controversy in Zimbabwe: An analysis of policy issues and local responses." **3**(5): 1000-1012.

Dixon, C. J. and B. Leach (1978). Sampling methods for geographical research, Geo Abstracts.

EuropeanBioplastics (2019). Bioplastics: Facts and Figures.

Hoornweg, D. and P. Bhada-Tata (2012). What A Waste: A Global Review of Solid Waste Management. Urban Development. Washington, DC, World Bank. **15**.

Horodytska, O., A. Cabanes and A. Fullana (2019). Plastic Waste Management: Current Status and Weaknesses. Berlin, Heidelberg, Springer Berlin Heidelberg**:** 1-18.

Jambeck, J. R. (2016). Written Testimony. Hearing: Marine Debris and Wildlife: Impacts, Sources and Solutions. Washington, DC, U.S. SENATE COMMITTEE ON ENVIRONMENT AND PUBLIC WORKS.

Jambeck, J. R., R. Geyer, C. Wilcox, T. R. Siegler, M. Perryman and A. Andrady (2015). "Plastic waste inputs from land into the ocean." Science **347**.

Jambeck, J. R., E. Moss, B. Dubey, Z. Arifin, L. Godfrey, B. D. Hardesty, I. G. Hendrawan, T. T. Hien, L. Junguo, M. Matlock, S. Pahl, K. Raubenheimer, M. Thiel, R. Thompson and L. C. Woodall (2020). Leveraging Multi-Target Strategies to Address Plastic Pollution in the Context of an Already Stressed Ocean. H. L. P. f. a. S. O. Economy. Washington, DC, World Resources Institute.

Juan, M. B. (2001). "The Coasts of Latin America at the End of the Century." Journal of Coastal Research **17**(4): 885-899.

Kaza, S., L. Yao, P. Bhada-Tata and F. Van Woerden (2018). What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050. Urban Development Series. Washington, DC, World Bank.

Kutralam-Muniasamy, G., F. Pérez-Guevara, I. Elizalde-Martínez and V. C. Shruti (2020). "Review of current trends, advances and analytical challenges for microplastics contamination in Latin America." Environmental Pollution **267**: 115463.

Law, K. (2017). "Plastics in the Marine Environment." Annual Review of Marine Science **9**: 205-229.

Law, K. L., Starr, N., Siegler, T., Jambeck, J., Mallos, N., Leonard, G. (2020). "The United States' contribution of plastic waste to land and ocean." Science Advances **6**(44).

Lebreton, L. and A. Andrady (2019). "Future scenarios of global plastic waste generation and disposal." Palgrave Communications **5**(1).

Lebreton, L. C. M., J. van der Zwet, J.-W. Damsteeg, B. Slat, A. Andrady and J. Reisser (2017). "River plastic emissions to the world's oceans." Nature Communications **8**: 15611.

Maes, T., J. Barry, H. A. Leslie, A. D. Vethaak, E. E. M. Nicolaus, R. J. Law, B. P. Lyons, R. Martinez, B. Harley and J. E. Thain (2018). "Below the surface: Twenty-five years of seafloor litter monitoring in coastal seas of North West Europe (1992–2017)." Science of The Total Environment **630**: 790-798.

Malizia, A. and A. C. Monmany-Garzia (2019). "Terrestrial ecologists should stop ignoring plastic pollution in the Anthropocene time." Science of The Total Environment **668**: 1025-1029.

Meijer, L. J. J., T. van Emmerik, R. van der Ent, C. Schmidt and L. Lebreton (2021). "More than 1000 rivers account for 80% of global riverine plastic emissions into the ocean." Science Advances **7**(18): eaaz5803.

Mid-Atlantic Solid Waste Consultants (2009). 2009 National Visible Litter Survey and Litter Cost Study. Washington, DC, Keep America Beautiful.

Noel, C. (2010). "Solid waste workers and livelihood strategies in Greater Port-au-Prince, Haiti." Waste Management **30**(6): 1138-1148.

Ocean Conservancy (2020). Together, We Are Team Ocean - 2020 Report. International Coastal Cleanup.

OECD (2014). What have we learned about Extended Producer Responsibility in the past decade? Case Study: Chile, Organization for Economic Cooperative Development.

OECD (2016). Extended Producer Responsibility.

Pacini, H., G. Shi, A. Sanches-Pereira and A. C. D. S. Filho (2021). "Network analysis of international trade in plastic scrap." Sustainable Production and Consumption **27**: 203-216.

Plastics Europe (2020). "Plastics the Facts - 2020."

Ryberg, M. W., M. Z. Hauschild, F. Wang, S. Averous-Monnery and A. Laurent (2019). "Global environmental losses of plastics across their value chains." Resources, Conservation & Recycling **151**.

Savino, A., G. Solorzano, C. Quispe and M. J. U. N. E. P. Correal, Argentina (2018). "Waste management outlook for Latin America and the Caribbean." **9**.

Schmidt, C., T. Krauth and S. Wagner (2017). "Export of Plastic Debris by Rivers into the Sea." Environmental Science & Technology **51**(21): 12246-12253.

Schröder, P., M. Albaladejo, P. A. Ribas, M. MacEwen and J. J. C. H. Tilkanen (2020). "The circular economy in Latin America and the Caribbean."

Schuyler, Q., B. Hardesty, T. J. Lawson, K. Opie and C. Wilcox (2018). Economic incentives reduce plastic inputs to the ocean.

Schuyler, Q., B. D. Hardesty, T. J. Lawson, K. Opie and C. Wilcox (2018). "Economic incentives reduce plastic inputs to the ocean." Marine Policy **96**: 250-255.

Siqueira, L. (2011). Plastic bag bans in Brazil spark debate. ICIS News. **July 11**.

Sostenible, M. d. A. y. D. (2018). Por la cual se reglamenta la gestion ambiental de los residuos de envases y empaques de papel, carton, plastico, vidrio, metal y se toman ostras determinaciones.

The Summit Foundation. (2017). "Eliminating Plastic Pollution on the Mesoamerican Reef." from http://www.summitfdn.org/mesoamerican-reef/eliminating-plastic-pollution-on-the-mesoamericanreef/.

The World Bank. (2020). "World Bank country and lending groups." from https://datahelpdesk.worldbank.org/knowledgebase/articles/906519-world-bank-country-and-lending-groups.

UNEP (2018a). "Legal Limits on Single-Use Plastics and Microplastics: A Global Review of National Laws and Regulations."

UNEP (2018b). Single-Use Plastics: A Roadmap for Sustainability.

United Nations Environment Program (UNEP). (2017). "Farewell plastics carrier bag!", from https://oceanconference.un.org/commitments/?id=16526.

US EPA (2014). Municipal Solid Waste Generation, Recycling, and Disposal in the United States: Tables and Figures for 2012. O. o. R. C. a. Recovery. Washington, DC.

Vig, N. J. and M. E. Kraft (2019). Environmental Policy: New Directions for the Twenty-First Century. Los Angeles, CQ Press.

Waste Atlas. (2016). "Waste Atlas." D-Waste Retrieved October 25, 2020, from http://www.atlas.d-waste.com/.

Wilcox, C., N. J. Mallos, G. H. Leonard, A. Rodriguez and B. D. Hardesty (2016). "Using expert elicitation to estimate the impacts of plastic pollution on marine wildlife." Marine Policy **65**: 107-114.

Ziajahromi, S., P. A. Neale, L. Rintoul and F. D. L. Leusch (2017). "Wastewater treatment plants as a pathway for microplastics: Development of a new approach to sample wastewater-based microplastics." Water Research **112**: 93-99.

CHAPTER 4

ASSESSMENT OF LAND-BASED PLASTIC WASTE IN THE GANGES RIVER BASIN USING EMPIRICAL EXTRAPOLATION AND QUANTITATIVE MODELING PROCEDURES [3]

[3]Brooks, A.L., Youngblood, K.M., Nishat, B., Patel, S., Sharma, E., Singh, K., Verma, G., Maddalene, T., Dubey, B.K., Jambeck, J.R. *To be submitted to Environmental Pollution*

4.0 ABSTRACT

In recent years, modeled estimates of plastic waste generation, management, and pollution have provided key outlooks and implications at the regional and international scale. Past estimates have been built on aggregated data over multiple scales, which can impact the outcome of analyses over large areas. Model calibration based on empirical data is needed at meaningful scales like that of river basin to better understand how plastics cycle through the socio-ecological systems. Here, field collected land-based litter data are extrapolated to the Ganges River basin for comparison with a model approach to support quantification of plastic waste generation and potential mismanagement throughout the basin. Between 39,200 and 392,000 metric tons of plastic waste was littered in the basin in 2019 based on both estimation approaches. The modeled estimate fell near the mean within the range, indicating agreement between the approaches. Despite the high concentration of people in many of the basin's urban centers, areas with relatively smaller populations saw the largest cumulative total of plastic litter. By the modeled estimate, 12.2 million metric tons (MMT) of plastic waste were generated throughout the basin and a corresponding 5.27 MMT of plastic waste were mismanaged. The modeled estimate highlighted the large contribution from India to these quantities, which were driven by the country's substantial spatial coverage and population occupying the basin. Based on these results, advancement of efforts for plastic waste collection and disposal outside of large urban centers, especially in India, may support the reduction of littered plastic waste. To better examine field calibration of common modeling procedures, future work should adapt the approach used here to other basins and spatial scales.

4.1 INTRODUCTION

The tie between plastic marine debris and land-based waste management practices has been well established in recent years (Jambeck et al. 2015, Lebreton and Andrady 2019, Jambeck et al. 2020), with rivers providing a key mode of transport for plastic waste from inland communities to the sea (Lebreton et al. 2017, Schmidt et al. 2017, Mai et al. 2020, Meijer et al. 2021). Research on environmental plastic pollution is rapidly evolving, but has thus far focused primarily on sources, sinks, and fate of plastics with respect to aquatic ecosystems and particularly the marine environment (Law 2017, Lebreton et al. 2019). However, plastics have been documented in upstream ecosystems including freshwater and terrestrial environments (de Souza Machado et al. 2018, van Emmerik and Schwarz 2020), and more research is needed to better understand the full plastic cycle (Blettler et al. 2018, Chae and An 2018, Bucci et al. 2019, Hoellein and Rochman 2021).

A review published in 2018 found that among the freshwater plastic pollution studies that existed at the time, most focused on microplastics and their impact on aquatic freshwater organisms (Blettler et al. 2018). This review also reported that 69% of freshwater plastic debris studies at the time had been conducted in comparatively high-income, industrialized countries (Gasperi et al. 2014, Jang et al. 2014, Lechner et al. 2014, McCormick et al. 2014, Morritt et al. 2014, Sadri and Thompson 2014, Cowger et al. 2019, Kataoka et al. 2019, Tramoy et al. 2019, Vriend et al. 2020), despite relatively lower levels of mismanaged waste (Blettler and Wantzen 2019). However, some more recent studies have investigated plastic pollution in freshwater systems such as the Saigon River (Lahens et al. 2018, van Emmerik et al. 2018, van Emmerik et al. 2019c), a multiriver system Jakarta (van Emmerik et al. 2019a), the Klang River in Malaysia (Geraeds et al. 2019), and multiple rivers in South Africa (Moss et al. 2021), where rapidly growing populations and economies are contributing to challenges with establishing effective plastic waste management systems.

More recently, a review of freshwater plastic research from 2019 specified that mismanaged household plastic waste is a severely under-evaluated topic within freshwater plastic pollution research (Blettler and Wantzen 2019). Household waste generation and municipal waste management are land-based activities, and so, in the context of freshwater plastic pollution, it stands that evaluation of plastic waste beyond the riparian zone is needed to better understand the link between mismanaged plastic waste, riverine plastic pollution, and the potential transport to the sea. Estimates of the quantities of land-based plastic waste and subsequent inputs into aquatic ecosystems (Lebreton et al. 2017, Lebreton and Andrady 2019) have largely centered on seminal work by Jambeck et al. (2015), which assessed global plastic waste generation and mismanagement by coastal populations. Large-scale global and regional models can deliver highly impactful findings that inform global challenges (Borer et al. 2014). In contrast, smaller scale environmental models can provide empirical insight and greater access to influential social and economic factors. As knowledge of plastic pollution sources, sinks, and fate have expanded, so too has the recognition that complex interconnected social, ecological, and economical systems inform integrated management of plastic waste, and the question of scale has emerged.

It has been asserted recently that river catchments may be a powerful landscape unit for assessing the complex systems through which plastic pollution cycles (Windsor et al. 2019, Hoellein and Rochman 2021). Existing basin-scale assessments have incorporated common methods to integrate land-based plastic waste models with empirical data (Hoffman and Hittinger 2017, Cowger et al. 2019, Tramoy et al. 2019). Watershed analyses such as these can provide foundations for better understanding of site-specific issues within the water-ecosystem-economy nexus (Cheng et al. 2014) and aid in integrated basin management (Li et al. 2018). Here, the Ganges River basin (GRB) is examined as a landscape unit in which modeled and empirical estimates of total plastic litter are generated.

A recent study by Meijer et al. (2021) indicated that India has the second largest contribution of river-based plastic emissions, and the Ganges River is estimated to deposit 105-172 thousand tons of plastic debris in the ocean annually, making it the 2nd most polluting river system in the world (Lebreton et al. 2017). The Ganges River system (known as the Ganga in India and the Padma and Meghna in Bangladesh, herein referred to as the Ganges) is one of the largest basins in the world, spanning four nations and 2,500 kilometers in length. The basin supports a substantially large number of people and various ecosystems and industries (Sarkar et al. 2019) and is a major cultural and religious icon in the region and throughout the world. With an estimated population of 720 million people by 2025, concerns regarding water quality and supply are rapidly growing (Hosterman et al. 2012) and contributing to this is the growing challenge with plastic waste management (Chattopadhyay et al. 2009).

Combined with rapid economic growth and lagging waste management infrastructure, developing regions see higher quantities of mismanaged plastic waste mismanaged despite smaller rates of consumption of plastic compared to developed regions (Kaza et al. 2018). Once mismanaged plastic waste reaches the river upstream, it potentially can travel considerable distances to downstream locations and beyond, as empirically demonstrated by a recent study along the Ganges (Duncan et al. 2020). Despite known challenges with plastic pollution, this basin has been overlooked in scientific research, particularly in the portion of the basin located in Bangladesh (Chowdhury et al. 2021). A recent study has documented microplastic presence in the river's sediment (Sarkar et al. 2019), and Napper et al. (2021) estimated that the Ganges may discharge 1-3 billion microplastic pieces to the sea daily. Research on macroplastic presence in the river is missing entirely, and while fishing activities have been documented as a significant source of plastic in the Ganges (Nelms et al. 2021), little is known about land-based plastic waste generation and management throughout the basin.

Plastic pollution in the GRB is very likely contributing to the environmental, social, and economic vulnerabilities that the river is facing and will continue to face if not addressed. To empirically support the modeling approach, land-based plastic debris data from Youngblood et al. (2021) were extrapolated to the basin and compared to the modeled estimate of littered plastic waste. Further, quantities of land-based plastic waste generation and management were estimated using established methods previously published by Jambeck et al. 2015. At this scale, estimated quantities of waste provide baseline measurements for comparison to support monitoring, managing, and advancing knowledge of plastic pollution over time given the changes expected in coming years.

4.2 MATERIALS AND METHODS

4.2.1 Framework for extrapolation of empirical litter data

Land-based plastic litter is an under-investigated source of plastic debris in riparian ecosystems. Most research on riverine plastic debris has focused on detection of and distribution of floating plastic debris using a range of monitoring methods such as active sampling (van Emmerik et al. 2019b), passive sampling (Tramoy et al. 2019), visual observations (van Emmerik et al. 2019c). More recently, advanced technical approaches have been used to track geospatially tagged litter items in river systems (Ivar do Sul et al. 2014, Tramoy et al. 2019, Duncan et al. 2020). What little research that has been conducted on land has often targeted riverbank documentation of plastic debris using citizen collected data (Rech et al. 2015, Barrows et al. 2018, Kiessling et al. 2019).

Human activity may play a key role in the presence of litter in the riparian ecosystem. Based upon citizen science data in river basins in Iowa, USA, Cowger et al. (2019) found that the presence of anthropogenic litter was correlated with population as well as developed and dense roadway land use areas. In contrast, population density has been shown to be a weak predictor of land-based debris quantities (Kawecki and Nowack 2020, Schuyler et al. 2021). As it stands, there are contradicting conclusions regarding the influence of population and plastic debris, but these differences could be the result of analyses that are sensitive to the location in which they were focused. Factors influencing the presence of plastic pollution remain an important topic of science investigation in the field. Here, litter density data is provided by Youngblood et al. (2021) for empirically estimating the total quantity of litter in the GRB. Notably, Youngblood et al. (2021) found that higher litter densities were observed in less populated areas of the GRB, and so the extrapolation reported here is dependent upon this specific relationship to population observed in the basin.

Based on the conceptual framework, the parameters for the extrapolation reflect empirically derived values of litter densities in items/m^2 determined from land-based litter surveys conducted in ten communities throughout the GRB as part of the National Geographic Sea to Source (S2S) Ganges Expedition between May and December 2019. Survey methods are described in Youngblood et al. (2021). Parameters included for the extrapolation of empirical data to estimate the basin-wide mass of plastic litter included three population-based litter densities, composition of plastic in field samples of litter, approximate count to mass conversion factors, and a range of turnover periods to estimate total litter quantities within a year. Litter densities and composition and the count to mass conversion factors were sourced from Youngblood et al. (2021), which reported litter densities as items/km^2 per 1,000 people and plastic composition across three population categories: low (100 - 2,000 people), mid (2,000 – 10,000 people), and high (>10,000 people).

Rasterized population data were sourced from the Oak Ridge National Laboratory (ORNL). The Landscan population dataset for 2019 was separated into the low, mid, and high population categories using ArcMap 10.8 to determine each category's total population and area (Table 4.1). Youngblood et al. (2021), which documented concentrated debris along roads, sidewalks, and gutters, which were estimated to cover 1% of the communities surveyed. To reflect the accumulation of debris along roads, sidewalks, and gutters, only 1% of the total areas of the low, mid, and high population categories were considered for the extrapolation. Using the total populations and areas across the three population categories and the spatial coverage assumption, the litter densities were converted to total item counts and multiplied by the fraction of items that were plastic.

While modeled estimates tend to report values in mass units (Jambeck et al. 2015), litter surveys are often reported as item counts and densities (Nelms et al. 2017, Nelms et al. 2020, Ocean Conservancy 2020, Youngblood et al. 2021). No direct conversion factors currently exist that allow for seamless translation between litter counts and masses, given the wide variety of product consumption and waste management across geographies and cultures. However, masses of commonly observed items from the field surveys by Youngblood et al. (2021) provided an approximated range to apply for the extrapolation. Here, an assumed 2-5 grams per item was applied to estimate the total mass of items in the basin based on the estimated total item count.

To account for accumulation of land-based debris over time, a range of turnover rates were applied to the mass estimate. Seasonal influences on plastic pollution have primarily focused on the effect seasonality has on riverine discharge of plastic debris. Modeled estimates by Lebreton et al. (2017) found that the riverine plastic emissions from the Ganges to the ocean peak in August at 44,500 MT per month as opposed to <150 MT per month in December and March, mostly driven by rainfall runoff in the monsoon season. In contrast, van Emmerik et al. (2019c) found the highest quantities of plastic transport from the Saigon River were during the dry season in December, with July and August emissions being the lowest. Regardless, knowledge around seasonality and accumulation of land-based litter limited, however, though it is likely that monsoons play a key role in deposition of plastic debris in the river. This is supported by

Youngblood et al. (2021), which did not find evidence of significant differences between quantities of land-based plastic debris in pre- and post-monsoon season surveys, suggesting that accumulated debris likely experiences displacement during the rainy season followed by rapid re-accumulation after the season ends. Given the remaining uncertainty around turnover of land-based plastic debris, a range of turnover rates was applied to the mass estimate to approximate the quantities that would result from items displacing and re-accumulating two to eight times per year. The low turnover rate is based on a one-time displacement from the monsoon season, while the high rate reflects turnover one time per month, excluding the dry months (December through March) identified by Lebreton et al. (2017). Figure 4.1 provides a visual overview of the procedure inputs and outputs.

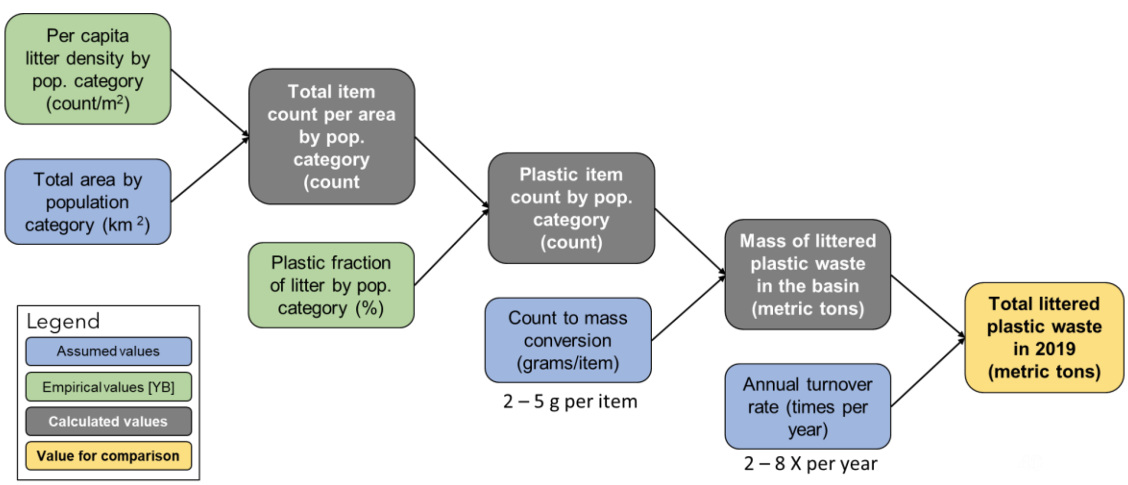

Figure 4.1. Visualization of extrapolation procedure to estimate basin-wide mass of littered plastic waste.

4.2.2. Quantitative model framework

One of the seminal modeling approaches for land-based plastic waste inputs into the ocean was developed by Jambeck et al. (2015) and has been used in various applications and a range of geospatial scales (Jambeck et al. 2017, Lebreton and Andrady 2019, Brooks et al. 2020, Law 2020). The approach has also been used in similar research to examine plastic waste inputs into river and lake basins (Hoffman and Hittinger 2017, Lebreton et al. 2017, Tramoy et al. 2019). The model parameters include population within a given area, per capita waste generation rates, composition of plastic in the waste stream, and waste management methods that reflect the amount of waste that is littered or inadequately managed. The resulting output provides mass estimates of plastic waste generation, littered plastic waste, and inadequately managed, the last two of which represent mismanaged plastic waste when combined.

Here, population data for the basin was sourced from the ORNL Landscan population dataset for 2019 for agreement with the extrapolated estimate. This rasterized global dataset was extracted to the basin using ArcMap 10.8 and population counts for each country within the basin were calculated for use as inputs into the mismanaged plastic waste model. Per capita waste generation rate, plastic composition, and waste management approaches were sourced from a global dataset provided by the World Bank (Kaza et al. 2018) unless data were missing, or more current information was available.

The total plastic waste generated in the basin was estimated by first converting the per capita waste generation rate for each country to an annual total based on corresponding population within each basin country. Then the proportion of plastic in each waste stream was applied to the total quantity. From here, the total mass of littered plastic waste in the basin was estimated by taking 2% of the total plastic waste generated based on the constant litter fraction used in the Jambeck et al. (2015) model. The procedure is visualized within the dashed portion of Figure 4.2. Finally, the total mismanaged plastic waste in MT was estimated by taking the fraction of waste that is inadequately managed in each country within the basin using the Jambeck et al. (2015) to provide estimated quantities in the basin (Figure 4.2). The data sources for inadequate fractions for each country are summarized in Table 4.3

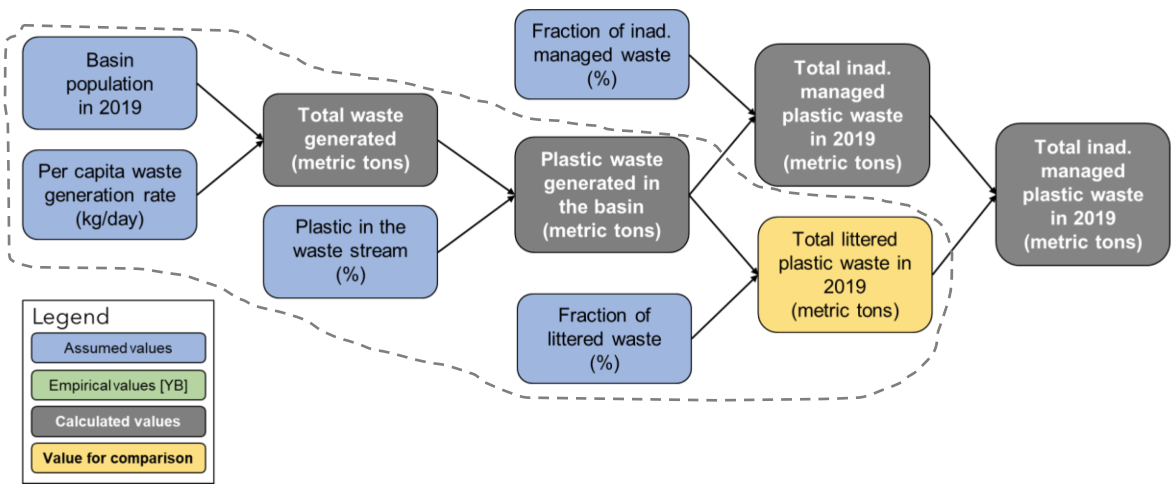

Figure 4.2. Visualization of the quantitative modeling procedure for estimating littered plastic waste, plastic waste generation, and mismanaged plastic waste. Dashed line defines the scope of the total plastic litter estimate.

4.3 RESULTS

4.3.1 Extrapolation of empirical litter counts to the GRB

Based on extrapolation of field-collected land-based debris data, an estimated 39,200-392,000 MT (m = 216,000) of plastic waste were littered in the GRB in 2019. This quantity equates to an average per capita litter rate of 0.0009 kg/day across the total inhabited basin population, which was determined to cover 580,000 km^2 within the basin. The low population group had the largest geospatial coverage of the basin at 530,000 km^2, equivalent to 91% of the populated regions of the basin. Compared to the low population group, the mid and high population categories were spatially concentrated and contributed to 7.7% and 0.9% coverage of the basin's inhabited area Figure 4.3. Given the proportion of spatial coverage, the low population group also had the largest total population when taken cumulatively, equating to 324 million, or 49% of the basin population, followed by the mid-level population group (34%) and the high population group (17%). The highest cumulative count and mass of plastic litter was found in the low population area with a total of 9.67 billion items. By extrapolation, the item count equates to an estimated 213,000 MT of littered plastic waste, which represents 98.6% of the basin's plastic litter by mass. The mid and high population categories accounted for 1.4% and 0.6% of the total plastic litter mass. Table 4.1 summarizes these findings.

Table 4.1. Empirically derived estimate of littered plastic waste in the GRB in 2019.

Pop. group	Pop. range	Plastic litter density (item/m^2)[a,b]	Basin-wide pop., millions[c]	Area in basin (km^2)[d]	Basin-wide plastic item count x 10^9	Plastic litter in GRB in 2019 (MT)	
						Low	High
Low	100-2,000	5.64	324	5300	48.3	38,700	387,000
Mid	2,000-10,000	1.34	227	448	0.685	548	5,480
High	>10,000	0.14	11.2	51.1	0.00407	3	33
Total			663	5,800	49.0	39,200	392,000

[a] Values provided by Youngblood et. al (In preparation)
[b] Per 1,000 people; Values based on plastic compositions of 88%, 85%, and 75% in low, mid, and high population categories
[c] Estimated in ArcMap 10.8
[d] Represented by approximately 1% coverage of surveyed areas in Youngblood et. al (In preparation)

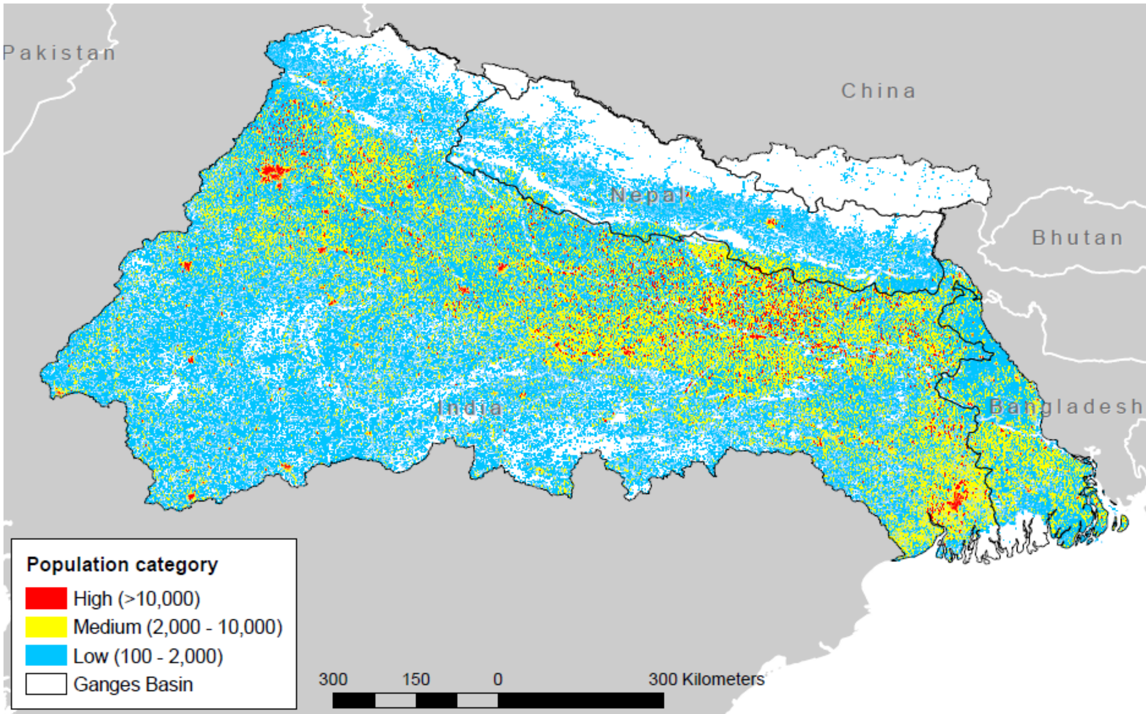

Figure 4.3. Areas within the GRB by population category defined by Youngblood et al. (2021).

4.3.2 Modeled estimates of littered and mismanaged plastic waste

Given the population and waste characteristics of each country within the basin, an estimated 12.2 MMT of plastic waste was generated within the GRB in 2019, with India contributing to 95% (11.7 MMT) of the basin's plastic waste generation. Based on the assumed litter fraction of 2%, the model estimated 245,000 MT of plastic waste were littered in the basin in 2019, representing a 12% error compared to the empirically derived estimate and a difference of 21,900 MT. When normalized by population, this quantity equates to an estimated 0.0004 kg of plastic waste is littered per capita per day (Table 4.2). An estimated 682 million people inhabited the Ganges basin in 2019, with 87% of the population located in India, followed by Bangladesh (8.9%), Nepal (4.6%), and China (<1%). Per capita waste generation rates among the basin countries ranged from 0.17 kg/day in Nepal to 0.57 kg/day in India (Kaza et al. 2018), with an average per capita waste generation rate of 0.36 kg/day (*SD* = 0.17) across the basin. The plastic composition within GRB countries ranged from 4.7% in Bangladesh to 15% in Nepal, with an average of 9.8% plastic (*SD* = 4.3) for the whole basin.

Table 4.2. Littered plastic waste for 2019 in countries within the GRB

Country	Basin population 2019, millions	World region[a]	Econ. status[a]	Per capita waste generation rate (kg/day)[b]	Plastic in waste stream (%)[b]	Plastic waste generation 2019 (MMT)	Plastic waste littered (MT)[c]
India	590	SAS	LMC	0.57	9.5	11.7	233,000
Bangladesh	60.6	SAS	LMC	0.28	4.7	0.289	5,780
Nepal	31.3	SAS	LMC	0.17	15	0.294	5,870
China	0.129	EAS	UMC	0.43	9.8	0.002	40
Total	682			0.36	9.8	12.2	245,000

[a] The World Bank (2020)
[b] Kaza et al. (2018); Value for India's plastic fraction from (Law 2020).
[c] Based on procedure by Jambeck et al. (2015) with 2% constant litter rate

Using the modeling procedure described here, an estimated 5.27 MMT of plastic waste was mismanaged in the basin in 2019. Across the basin countries, an average of 47% (*SD* = 29) of waste in the basin was managed inadequately through disposal methods such as open dumping, burning, burying, or deposition in waterways. The remainder of plastic waste generated within the basin was likely treated by advanced waste treatment methods such as sanitary or controlled landfills, recycling, or incineration. Nepal had the highest proportion of inadequately managed waste (90%), with most waste (60%) considered unaccounted for or deposited in unspecified sanitary landfills that face challenges relating to protests, poor management, and deficient equipment (Asian Development Bank 2013, Kaza et al. 2018). China had the lowest proportion of inadequate waste management at 23% based on Law (2020).

Taking the combined littered and inadequately managed waste fractions together, the proportion of waste that was mismanaged by country ranged from 25% in China to 92% in Nepal, with the mean fraction of mismanaged waste being 49% (*SD* = 29). Of the total mismanaged plastic waste, 93% was generated in India, followed by Nepal (5.1%), Bangladesh (2.0%), and China (<0.1%). When normalized by population for direct comparison, the per capita mismanaged plastic waste rate ranged from 0.005 kg/day in Bangladesh to 0.024 kg/day in Nepal with an average of 0.015 kg/day (*SD* = 0.009). Table 4.3 provides a summary of the inadequately and mismanaged plastic waste in the basin.

Table 4.3. Quantities of inadequately managed and mismanaged plastic waste in the GRB in 2019

Country	Basin pop. in 2019, x 10^6	Per capita waste generation rate (kg/day) [a]	Plastic fraction (%)[a]	Inad. managed waste (%)[b]	Inad. managed plastic waste (MT)[c]	Per capita MMPW (kg /day) [c]	MMPW 2019 (MT) [c]
India	590	0.57	9.5	40	4,66 x 10^6	0.023	4.90 x 10^6
Bangladesh	60.6	0.28	4.7	35	101,000	0.005	107,000
Nepal	31.3	0.17	15	90	264,000	0.024	270,000
China	0.129	0.43	9.8	23	460	0.011	500
Total	682	0.36	9.5	47	5.03 x 10^6	0.015	5.27 x 10^6

[a] Kaza et al. (2018)
[b] Values for each country as follows: India, NPC (2020); Bangladesh, Nishat (2020); Nepal, Kaza et al. (2018); China, Law (2020)
[c] Calculated values

4.4 DISCUSSION

The resulting quantities of littered plastic in the GRB estimated by the field and modeled approaches were comparable, and the modeled approach fell within the range estimated by the empirical extrapolation. Further, when compared to the mean plastic litter quantity from the empirical extrapolation, the modeled estimate was of similar magnitude, suggesting some agreement between the approaches. Though comparable in terms of estimated quantities, each estimate had its own set of implications related to the input parameters and resulting outcomes.

Substantial differences between low and high population categories emerged in the empirical extrapolation, which resulted in large quantities of plastic waste estimated to be littered in areas with smaller population when taken cumulatively. Although there may be fewer people (i.e., 100-2,000 people) in these areas and smaller overall quantities of plastic waste generated, these areas have larger spatial coverage of the basin compared to the high and mid population categories (Figure 4.3), which results in compounding amounts of plastic waste when considered altogether. Notably, the low population areas were considered low in relation to the extreme populations concentrated in urban centers in the basin. For instance, a mid-size metropolitan city in the USA such as Atlanta, Georgia, has an average population density of 1,490 people per km^2 while Delhi, India has a current population density of 11,300 people per km^2. If waste collection is lacking in the low population areas, then littering may be a significant source of mismanaged plastic waste in these areas. Smaller communities may lack sufficient waste streams to support the scalable development of waste management infrastructure, particularly for synthetic products like plastics. In developing regions, this may be further compounded by limited financial capital. As a result, access to services, in this case, adequate waste management infrastructure, may depend on physical proximity to urban centers (Cattaneo et al. 2021).

In contrast, highly populated areas had the lowest estimated quantities of both littered plastic count and mass. This is likely due to highly concentrated populations that together have significantly less spatial coverage (Figure 4.3). This does not imply, however, that smaller quantities of plastic waste are generated in urban areas with high populations, but that there may be less plastic littered in these areas given the increased likelihood that waste is collected in urban areas to prevent severe accumulation in densely populated areas (Sharma and Jain 2019). This trend has been documented in India, where waste collection efficiency decreases with the size of urban centers and is particularly lacking in rural areas (Sharma and Jain 2019).

In comparing extrapolated and modeled estimates of plastic litter, the methodological procedures used in this study led to some limitations. Both approaches relied on limited assumptions that would benefit from verification and aggregated data over a large area. The empirical estimate was based upon litter data from collected from ten communities in the basin which were all proximal to the main river channel and may not fully capture the composition and abundance of littered plastic debris throughout the region. Future empirical data collection could benefit from broader spatial coverage and expansion of data collection related to other potential influences on debris quantities such as socioeconomic, land use, or hydrological characteristics. Similarly, the modeled estimate was informed by national level statistics on waste characteristics and expert knowledge of local partners. Although these are considered best available data, they may suffer from methodological variation or differences in definitions, reporting, and units, and so may not fully reflect the systems of waste generation, composition, collection, and treatment throughout the basin. Future work might provide empirical data regarding household waste generation and composition to further inform upstream waste management methods relative to that documented as litter.

No methodologically equivalent estimate exists for comparison of the modeled waste values reported here, although the estimated total plastic waste generation in the basin was similar in magnitude to other estimates in the region. For instance, India's National Productivity Council (NPC) estimated that 9.4 MMT of plastic waste are generated in the country annually NPC (2020), and with 36% of the India's population located in the GRB, the NPC estimate would equate to 3.36 MMT of plastic waste generated in the basin annually, which is just a third of the 11.7 MMT estimated for the Indian portion of the GRB in 2019 here. Further, taking 2% of the estimated 3.36 MMT would result in roughly 67,200 MT of plastic litter in the basin based on the NPC estimate, which is lower than the 245,000 metric ton estimate, but falls within the estimated range determined by the empirical extrapolation. However, the methodology used for the NPC study is unclear which limits interpretation of the difference in quantities between the two estimates. Lebreton and Andrady (2019) also estimated that 21.8 MMT of plastic waste were generated in all of India in 2020, which is more than twice the NPC estimate for the country. Based on the fraction of India's population in the basin, the Lebreton and Andrady (2019) estimate equates to 7.85 MMT of plastic waste generated in the basin in 2020, which falls between the values estimated by NPC and the current study. This would also equate to 157,000 MT of litter, which is less than the modeled estimate of 245,000 but falls within the extrapolated estimate. If all four of the GRB countries are considered, the Lebreton and Andrady (2019) estimate for each country's total plastic waste management equates to 8.68 MMT of plastic waste generated and 173,000 MT of plastic littered in the basin, which may indicate the estimation here is inflated. However, more recently, Law (2020) estimated that India generated 26.3 MMT of plastic waste in 2016, which would equate to 9.45 MMT, only 2.25 MMT less than that estimated here for 2019.

The estimated quantity of mismanaged plastic waste in the basin is comparable to previous research as well. Lebreton and Andrady (2019) estimated that India generated 18.5 MMT of mismanaged plastic waste in 2020, which would equate to 6.67 MMT of mismanaged plastic waste based on the proportion of the country's population in the basin. This is comparable to the estimated 5.27 MMT estimated in the present study for 2019, however, the fraction of mismanaged plastic waste assumed for the Lebreton and Andrady (2019) study (85%) was higher than the value used for India in this study. Law (2020) estimated that 20.8 MMT of plastic waste was mismanaged in India in 2016, which would equate to 7.49 MMT of mismanaged plastic waste in the GRB. This estimate is 2.22 MMT higher than the mismanaged plastic waste estimate for 2019 presented here, but the inadequately managed fraction was also assumed to be greater in the Law (2020) study (i.e., 77%).

Specific to the Ganges River, Lebreton et al. (2017) report an estimated 1.49 MMT of mismanaged plastic waste transported from land to the sea annually. This estimate accounts for accumulation of mismanaged plastic waste at dams throughout the river (65%), which, if accounted for in estimating basin-wide mismanaged plastic waste would result in approximately 4.25 MMT in the basin in 2017, which is comparable to the 5.27 MMT estimated here for 2019. Lastly, the only estimate limited to the extent of the GRB by Schmidt et al. (2017) finds that 3.02 MMT of mismanaged plastic waste were generated in the basin using the same modeling procedure used in the present study, however, their estimate was for 2010. Still, with growth in population, plastic consumption patterns, and waste disposal methods, it is possible that their estimate projected to 2019 could align closely with the estimated quantity here.

As demonstrated by the modeled estimate, India plays a significant role in the plastic waste generation and disposal in the basin, which is expected given that the country takes up the most spatial coverage of the basin, has the largest national population in the basin (87%), the highest per capita waste generation rate, and second highest plastic fraction out of the four basin countries. India is expected to experience rapid growth in coming years, and under business-as-usual conditions, may take over as the largest generator of mismanaged plastic waste in 2013. Further, the country could generate an estimated 46.3 MMT of mismanaged plastic waste annually by 2060 (Borrelle et al. 2020), which is nearly four times that estimated to have been generated in 2019. As such, it is imperative that efforts to drastically curtail the quantities of mismanaged plastic waste in the region are developed and enforced in the country, particularly in areas where there are smaller waste streams and lacking dedicated waste management infrastructure.

The presence of plastic litter in built and natural environments is challenge shared throughout the world and will require coordinated action to reduce plastic waste generation, improve waste management, and prevent plastic waste from leaking into the environment (Borrelle et al. 2020, Lau et al. 2020). Multi-scale approaches that incorporate local, sub-national, and national approaches may be the most effective at addressing the problem (Vince and Hardesty 2017). Some government-supported attempts at the national level have targeted improving rural waste management in India in recent years. For example, the Swachh Bharat Mission implemented in 2014 by the

Indian government allocated funds of ₹1.0 million INR for rural sanitation development, and some of this is specifically aimed toward rural waste management infrastructure. Further, the Government of India’s Department of Water Resources, River Development, & Ganga Rejuvenation registered the National Mission for Clean Ganga (NMCG) in 2011 with the aim of restoration, ecological flow maintenance, and comprehensive management of the Ganges River.

Globally, bans on certain single use plastic packaging items have been shown to effectively reduce the prevalence of them in the environment (Maes et al. 2018, Schuyler et al. 2018). Bangladesh implemented the world’s first ban on plastic bags in 2002, however, the effectiveness has been uncertain due to lacking enforcement and accessible options for alternatives (UNEP 2018b). In 2018, India committed to eliminating single-use plastics by 2022 through bans, economic incentives, recycling requirements, and provisions for Extended Producer Responsibility. Both statewide and municipal policies have targeted bans on manufacture, supply, storage, and use of plastics, with some policies having strict penalties for non-compliance (UNEP 2018a). Past efforts for regulatory prohibitions on plastic bags in India suffered from insufficient enforcement (Gupta 2011), and so the ambitious goals of the current efforts will benefit from coordinated effort at monitoring and enforcement. Empirical surveys that support modeling efforts like that here, can be especially useful for documenting impact of these types of policies and financing efforts for environmental management of the Ganges River.

4.5 CONCLUSION

Ecological and environmental models are used extensively for estimating the presence of plastic pollution in the natural environment, and these models are inherently difficult to validate with empirical evidence which can require resource intensive collection methods. Methods for measuring environmental plastic pollution have so far focused on surveying freshwater and marine environments and have successfully informed calibration of modeled estimates in various domains. But given the urgent need for improving global plastic consumption and disposal method that are mostly land-based, it is vital that methods for reliable measuring and monitoring of land-based plastic debris need to be established and agreed upon. Here, a method for extrapolating empirical land-based litter data provided methods for calibration of quantitative methods commonly used for estimating land-based plastic waste management and inputs of mismanaged debris to the natural environment.

REFERENCES

Asian Development Bank (2013). Solid waste management in Nepal: Current status and policy recommendations. Mandaluyong City, Philippines, Asian Development Bank.

Barrows, A. P., K. S. Christiansen, E. T. Bode and T. J. Hoellein (2018). "A watershed-scale, citizen science approach to quantifying microplastic concentration in a mixed land-use river." Water research **147**: 382-392.

Blettler, M. and K. Wantzen (2019). "Threats Underestimated in Freshwater Plastic Pollution: Mini-Review." Water, Air, & Soil Pollution **230**.

Blettler, M. C. M., E. Abrial, F. R. Khan, N. Sivri and L. A. Espinola (2018). "Freshwater plastic pollution: Recognizing research biases and identifying knowledge gaps." Water Research **143**: 416-424.

Borer, E. T., W. S. Harpole, P. B. Adler, E. M. Lind, J. L. Orrock, E. W. Seabloom and M. D. Smith (2014). "Finding generality in ecology: a model for globally distributed experiments." **5**(1): 65-73.

Borrelle, S. B., J. Ringma, K. L. Law, C. C. Monnahan, L. Lebreton, A. McGivern, E. Murphy, J. Jambeck, G. H. Leonard, M. A. Hilleary, M. Eriksen, H. P. Possingham, H. De Frond, L. R. Gerber, B. Polidoro, A. Tahir, M. Bernard, N. Mallos, M. Barnes and C. M. Rochman (2020). "Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution." Science **369**(6510): 1515.

Brooks, A. L., J. R. Jambeck and E. Mozo-Reyes (2020). Plastic Waste Management and Leakage in Latin America and the Caribbean Inter-American Development Bank.

Bucci, K., M. Tulio and C. M. Rochman (2019). "What is known and unknown about the effects of plastic pollution: A meta-analysis and systematic review." Ecological Applications **n/a**(n/a).

Cattaneo, A., A. Nelson and T. McMenomy (2021). "Global mapping of urban–rural catchment areas reveals unequal access to services." Proceedings of the National Academy of Sciences **118**(2): e2011990118.

Chae, Y. and Y.-J. An (2018). "Current research trends on plastic pollution and ecological impacts on the soil ecosystem: A review." Environmental Pollution **240**: 387-395.

Chattopadhyay, S., A. Dutta and S. Ray (2009). "Municipal solid waste management in Kolkata, India – A review." Waste Management **29**(4): 1449-1458.

Cheng, G., X. Li, W. Zhao, Z. Xu, Q. Feng, S. Xiao and H. Xiao (2014). "Integrated study of the water–ecosystem–economy in the Heihe River Basin." National Science Review **1**(3): 413-428.

Chowdhury, G. W., H. J. Koldewey, E. Duncan, I. E. Napper, M. N. H. Niloy, S. E. Nelms, S. Sarker, S. Bhola and B. Nishat (2021). "Plastic pollution in aquatic systems in Bangladesh: A review of current knowledge." Science of The Total Environment **761**: 143285.

Cowger, W., A. B. Gray and R. C. Schultz (2019). "Anthropogenic litter cleanups in Iowa riparian areas reveal the importance of near-stream and watershed scale land use." Environmental Pollution **250**: 981-989.

de Souza Machado, A. A., W. Kloas, C. Zarfl, S. Hempel and M. C. Rillig (2018). "Microplastics as an emerging threat to terrestrial ecosystems." **24**(4): 1405-1416.

Duncan, E. M., A. Davies, A. Brooks, G. W. Chowdhury, B. J. Godley, J. Jambeck, T. Maddalene, I. Napper, S. E. Nelms, C. Rackstraw and H. Koldewey (2020). "Message in a bottle: Open source technology to track the movement of plastic pollution." PLOS ONE **15**(12): e0242459.

Gasperi, J., R. Dris, T. Bonin, V. Rocher and B. Tassin (2014). "Assessment of floating plastic debris in surface water along the Seine River." Environmental Pollution **195**: 163-166.

Geraeds, M., T. van Emmerik, R. de Vries and S. M. bin Ab Razak (2019). "Riverine Plastic Litter Monitoring Using Unmanned Aerial Vehicles (UAVs)." Remote Sensing **11**(17).

Gupta, K. (2011). Consumer responses to incentives to reduce plastic bag use: Evidence from a field experiment in urban India, SAMDEE.

Hoellein, T. J. and C. M. Rochman (2021). "The “plastic cycle”: a watershed-scale model of plastic pools and fluxes." **19**(3): 176-183.

Hoffman, M. J. and E. Hittinger (2017). "Inventory and transport of plastic debris in the Laurentian Great Lakes." Marine Pollution Bulletin **115**(1): 273-281.

Hosterman, H. R., P. G. McCornick, E. J. Kistin, B. Sharma and B. Luna (2012). "Freshwater, climate change and adaptation in the Ganges River Basin." Water Policy **14**(1): 67-79.

Ivar do Sul, J. A., M. F. Costa, J. S. Silva-Cavalcanti and M. C. B. Araújo (2014). "Plastic debris retention and exportation by a mangrove forest patch." Marine Pollution Bulletin **78**(1): 252-257.

Jambeck, J., B. D. Hardesty, A. L. Brooks, T. Friend, K. Teleki, J. Fabres, Y. Beaudoin, A. Bamba, J. Francis, A. J. Ribbink, T. Baleta, H. Bouwman, J. Knox and C. Wilcox (2017). "Challenges and emerging solutions to the land-based plastic waste issue in Africa." Marine Policy.

Jambeck, J. R., R. Geyer, C. Wilcox, T. R. Siegler, M. Perryman and A. Andrady (2015). "Plastic waste inputs from land into the ocean." Science **347**.

Jambeck, J. R., E. Moss, B. Dubey, Z. Arifin, L. Godfrey, B. D. Hardesty, I. G. Hendrawan, T. T. Hien, L. Junguo, M. Matlock, S. Pahl, K. Raubenheimer, M. Thiel, R. Thompson and L. C. Woodall (2020). Leveraging Multi-Target Strategies to Address Plastic Pollution in the Context of an Already Stressed Ocean. H. L. P. f. a. S. O. Economy. Washington, DC, World Resources Institute.

Jang, S. W., D. H. Kim, K. T. Seong, Y. H. Chung and H. J. Yoon (2014). "Analysis of floating debris behaviour in the Nakdong River basin of the southern Korean peninsula using satellite location tracking buoys." Mar Pollut Bull **88**(1-2): 275-283.

Kataoka, T., Y. Nihei, K. Kudou and H. Hinata (2019). "Assessment of the sources and inflow processes of microplastics in the river environments of Japan." Environmental Pollution **244**: 958-965.

Kawecki, D. and B. Nowack (2020). "A proxy-based approach to predict spatially resolved emissions of macro- and microplastic to the environment." Science of The Total Environment **748**: 141137.

Kaza, S., L. Yao, P. Bhada-Tata and F. Van Woerden (2018). What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050. Urban Development Series. Washington, DC, World Bank.

Kiessling, T., K. Knickmeier, K. Kruse, D. Brennecke, A. Nauendorf and M. Thiel (2019). "Plastic Pirates sample litter at rivers in Germany – Riverside litter and litter sources estimated by schoolchildren." Environmental Pollution **245**: 545-557.

Lahens, L., E. Strady, T.-C. Kieu-Le, R. Dris, K. Boukerma, E. Rinnert, J. Gasperi and B. Tassin (2018). "Macroplastic and microplastic contamination assessment of a tropical river (Saigon River, Vietnam) transversed by a developing megacity." Environmental Pollution **236**: 661-671.

Lau, W. W. Y., Y. Shiran, R. M. Bailey, E. Cook, M. R. Stuchtey, J. Koskella, C. A. Velis, L. Godfrey, J. Boucher, M. B. Murphy, R. C. Thompson, E. Jankowska, A. Castillo Castillo, T. D. Pilditch, B. Dixon, L. Koerselman, E. Kosior, E. Favoino, J. Gutberlet, S. Baulch, M. E. Atreya, D. Fischer, K. K. He, M. M. Petit, U. R.

Sumaila, E. Neil, M. V. Bernhofen, K. Lawrence and J. E. Palardy (2020). "Evaluating scenarios toward zero plastic pollution." Science: eaba9475.

Law, K. (2017). "Plastics in the Marine Environment." Annual Review of Marine Science **9**: 205-229.

Law, K. L., Starr, N., Siegler, T., Jambeck, J., Mallos, N., Leonard, G. (2020). "The United States' contribution of plastic waste to land and ocean." Science Advances **6**(44).

Lebreton, L. and A. Andrady (2019). "Future scenarios of global plastic waste generation and disposal." Palgrave Communications **5**(1).

Lebreton, L., M. Egger and B. Slat (2019). "A global mass budget for positively buoyant macroplastic debris in the ocean." Scientific Reports **9**(1): 12922.

Lebreton, L. C. M., J. van der Zwet, J.-W. Damsteeg, B. Slat, A. Andrady and J. Reisser (2017). "River plastic emissions to the world's oceans." Nature Communications **8**: 15611.

Lechner, A., H. Keckeis, F. Lumesberger-Loisl, B. Zens, R. Krusch and M. Tritthart (2014). "The Danube so colourful: a potpourri of plastic litter outnumbers fish larvae in Europe's second largest river." Environ Pollut **188**.

Li, X., G. Cheng, H. Lin, X. Cai, M. Fang, Y. Ge, X. Hu, M. Chen and W. Li (2018). "Watershed System Model: The Essentials to Model Complex Human-Nature System at the River Basin Scale." Journal of Geophysical Research: Atmospheres **123**(6): 3019-3034.

Maes, T., J. Barry, H. A. Leslie, A. D. Vethaak, E. E. M. Nicolaus, R. J. Law, B. P. Lyons, R. Martinez, B. Harley and J. E. Thain (2018). "Below the surface: Twenty-five years of seafloor litter monitoring in coastal seas of North West Europe (1992–2017)." Science of The Total Environment **630**: 790-798.

Mai, L., X.-F. Sun, L.-L. Xia, L.-J. Bao, L.-Y. Liu and E. Y. Zeng (2020). "Global Riverine Plastic Outflows." Environmental Science & Technology.

McCormick, A., T. J. Hoellein, S. A. Mason, J. Schluep and J. J. Kelly (2014). "Microplastic is an abundant and distinct microbial habitat in an urban river." Environ Sci Technol **48**.

Meijer, L. J. J., T. van Emmerik, R. van der Ent, C. Schmidt and L. Lebreton (2021). "More than 1000 rivers account for 80% of global riverine plastic emissions into the ocean." Science Advances **7**(18): eaaz5803.

Morritt, D., P. V. Stefanoudis, D. Pearce, O. A. Crimmen and P. F. Clark (2014). "Plastic in the Thames: a river runs through it." Marine Pollution Bulletin **78**(1-2): 196-200.

Moss, K., D. Allen, D. González-Fernández and S. Allen (2021). "Filling in the knowledge gap: Observing MacroPlastic litter in South Africa's rivers." Marine Pollution Bulletin **162**: 111876.

Napper, I. E., A. Baroth, A. C. Barrett, S. Bhola, G. W. Chowdhury, B. F. R. Davies, E. M. Duncan, S. Kumar, S. E. Nelms, M. N. Hasan Niloy, B. Nishat, T. Maddalene, R. C. Thompson and H. Koldewey (2021). "The abundance and characteristics of microplastics in surface water in the transboundary Ganges River." Environmental Pollution **274**: 116348.

Nelms, S. E., C. Coombes, L. C. Foster, T. S. Galloway, B. J. Godley, P. K. Lindeque and M. J. Witt (2017). "Marine anthropogenic litter on British beaches: A 10-year nationwide assessment using citizen science data." Science of The Total Environment **579**: 1399-1409.

Nelms, S. E., E. M. Duncan, S. Patel, R. Badola, S. Bhola, S. Chakma, G. W. Chowdhury, B. J. Godley, A. B. Haque, J. A. Johnson, H. Khatoon, S. Kumar, I. E. Napper, M. N. H. Niloy, T. Akter, S. Badola, A. Dev, S. Rawat, D. Santillo, S. Sarker, E. Sharma and H. Koldewey (2021). "Riverine plastic pollution from fisheries: Insights from the Ganges River system." Science of The Total Environment **756**: 143305.

Nelms, S. E., L. Eyles, B. J. Godley, P. B. Richardson, H. Selley, J.-L. Solandt and M. J. Witt (2020). "Investigating the distribution and regional occurrence of anthropogenic litter in English marine protected areas using 25 years of citizen-science beach clean data." Environmental Pollution **263**: 114365.

Nishat, B. N. (2020).

NPC (2020). Literature Survey and Desk Review. Promotion of Counter Measures Against Marine Plastic Litter in Southeast Asia and India, Government of India, Ministry of Commerce and Industry, Department for Promotion of Industry & Internal Trade, National Productivity Council.

Ocean Conservancy (2020). Together, We Are Team Ocean - 2020 Report. International Coastal Cleanup.

Rech, S., V. Macaya-Caquilpán, J. Pantoja, M. Rivadeneira, C. Campodónico and M. Thiel (2015). "Sampling of riverine litter with citizen scientists - findings and recommendations." Environmental Monitoring & Assessment **187**(6): 1-18.

Sadri, S. S. and R. C. Thompson (2014). "On the quantity and composition of floating plastic debris entering and leaving the Tamar Estuary, Southwest England." Marine Pollution Bulletin **81**(1): 55-60.

Sarkar, D. J., S. Das Sarkar, B. K. Das, R. K. Manna, B. K. Behera and S. Samanta (2019). Spatial distribution of meso and microplastics in the sediments of river Ganga at eastern India. **694**.

Schmidt, C., T. Krauth and S. Wagner (2017). "Export of Plastic Debris by Rivers into the Sea." Environmental Science & Technology **51**(21): 12246-12253.

Schuyler, Q., B. D. Hardesty, T. J. Lawson, K. Opie and C. Wilcox (2018). "Economic incentives reduce plastic inputs to the ocean." Marine Policy **96**: 250-255.

Schuyler, Q., C. Wilcox, T. J. Lawson, R. R. M. K. P. Ranatunga, C.-S. Hu, Global Plastics Project Partners and B. D. Hardesty (2021). "Human Population Density is a Poor Predictor of Debris in the Environment." **9**(133).

Sharma, K. and S. Jain (2019). "Overview of Municipal Solid Waste Generation, Composition, and Management in India." Journal of Environmental Engineering (United States) **145**.

The World Bank. (2020). "World Bank country and lending groups." from https://datahelpdesk.worldbank.org/knowledgebase/articles/906519-world-bank-country-and-lending-groups.

Tramoy, R., J. Gasperi, R. Dris, L. Colasse, C. Fisson, S. Sananes, V. Rocher and B. Tassin (2019). "Assessment of the Plastic Inputs From the Seine Basin to the Sea Using Statistical and Field Approaches." **6**(151).

UNEP (2018b). "Legal Limits on Single-Use Plastics and Microplastics: A Global Review of National Laws and Regulations."

UNEP (2018a). Single-Use Plastics: A Roadmap for Sustainability.

van Emmerik, T., T.-C. Kieu-Le, M. Loozen, K. van Oeveren, E. Strady, X.-T. Bui, M. Egger, J. Gasperi, L. Lebreton, P.-D. Nguyen, A. Schwarz, B. Slat and B. Tassin (2018). "A Methodology to Characterize Riverine Macroplastic Emission Into the Ocean." **5**(372).

van Emmerik, T., M. Loozen, K. van Oeveren, F. Buschman and G. Prinsen (2019a). "Riverine plastic emission from Jakarta into the ocean." Environmental Research Letters **14**(8): 084033.

van Emmerik, T., T. Romain, C. Caroline van, A. Soline, T. Robin, T. Bruno and G. Johnny (2019b). "Seine Plastic Debris Transport Tenfolded During Increased River Discharge." Frontiers in Marine Science **6**.

van Emmerik, T. and A. Schwarz (2020). "Plastic debris in rivers." WIREs Water **7**(1): e1398.

van Emmerik, T., E. Strady, T.-C. Kieu-Le, L. Nguyen and N. Gratiot (2019c). "Seasonality of riverine macroplastic transport." Scientific Reports **9**(1): 13549.

Vince, J. and B. D. Hardesty (2017). "Plastic pollution challenges in marine and coastal environments: from local to global governance." Restoration Ecology **25**(1): 123-128.

Vriend, P., C. van Calcar, M. Kooi, H. Landman, R. Pikaar and T. van Emmerik (2020). "Rapid Assessment of Floating Macroplastic Transport in the Rhine." **7**(10).

Windsor, F. M., I. Durance, A. A. Horton, R. C. Thompson, C. R. Tyler and S. J. Ormerod (2019). "A catchment-scale perspective of plastic pollution." Global Change Biology **25**(4): 1207-1221.

Youngblood, K., A. L. Brooks, G. Verma, A. Singh, T. Zakir, M. N. Sultana, N. Das, B. Dubey and J. R. Jambeck (2021). "The characterization of Litter in Communities along the Ganges River." Manuscript submitted for publication.

CHAPTER 5

AN EXPERIMENTAL ASSESSMENT OF A RAPID FIELD SURVEY METHOD FOR LAND-BASED ANTHROPOGENIC POLLUTION IN COMMUNITIES USING PHOTOQUADRAT ANALYSIS[4]

[4] Brooks, A.L., Das, N., Singh, K.A. Verma, G., Youngblood, K.Y., Barber, E.C., Jones Duncan, E.M., Maddalene, T., Napper, I.M., Nelms, S.E., E., Koldeway, H. Jambeck, J.R. To be submitted to *Environmental Pollution*.

5.0 ABSTRACT

Measurement and documentation of quantities, distribution, and composition of land-based plastic pollution is key informing evidence-based solutions aimed toward the prevention of mismanaged plastic waste from reaching aquatic ecosystems, however, few studies have addressed community-based debris. Using rapid field data collection and photoquadrat analysis procedures, the presence and composition of anthropogenic debris was documented in ten communities in throughout the Ganges River basin. Two rounds of surveys were conducted to reflect pre- and post-monsoon conditions. The surveys recorded a total of 9,520 items of debris, of which plastic comprised 58% of items by count. Much of the plastic documented was associated with food packaging and tobacco, however, plastic fragments were the most documented form (36%). Debris density was positively related to highly populated, urban survey sites, less populated areas had variable levels of litter density, warranting further investigation into the relationship between population, urbanization, and anthropogenic debris. When normalized by population, however, the smallest and most remote sites included in the study had the largest per capita debris densities, suggesting that distance to urban centers may negatively impact access to waste collection and treatment. Although the results of the photoquadrat analysis are comparable to other similar studies, preliminary analysis of the reliability of the method was determined to be inconclusive, and so the results are taken tentatively while the method is further refined. Nevertheless, the experimental approach presented here contributes to development of practical and rapid field approaches for empirical data collection of land-based anthropogenic debris.

5.1 INTRODUCTION

Detection of anthropogenic litter and debris in both urban and natural land-based environments can provide data-driven solutions targeted towards waste reduction, waste management infrastructure, education and awareness programs for proper waste disposal, or policy instruments such as bans or levies for problematic litter items. To date, most research related to plastic pollution has been focused on marine-based debris, leaving inland freshwater and terrestrial environments relatively unexplored (Blettler et al. 2018), although momentum in these areas is growing. Further, most research has focused on microplastics (i.e., plastic items that are less than 5 mm in size), rather than macroplastic, which is a primary source for both plastic marine debris and microplastics (van Emmerik and Schwarz 2020). Macroplastics can enter the environment through several pathways including littering, industry, storms, and inadequate waste management (Lechthaler et al. 2020). Further, quantities are exacerbated by short product lifespans for packaging plastics (Geyer et al. 2017), which make up the largest sector of global plastic production (Plastics Europe 2020).

Plastics such as those used in packaging suffer from low material value which reduces efficiency in collection and treatment of post-consumer items once they are disposed (Moss 2017). The fate of anthropogenic debris that leaks from waste management systems is relatively unknown and difficult to measure, given the susceptibility of plastic waste transport due to varying spatial and temporal conditions, (Thompson et al. 2004, Koelmans et al. 2017, Ryberg et al. 2019). Recent research in microplastic sampling methods has highlighted the need for rigorous and standardized quality assurance and quality control (QA/QC) in marine and freshwater systems (Hung et al. 2021), however, this need is not limited to microplastics research. Robust methods and QA/QC procedures are essential for macroplastics research but methods for doing so are not yet fully developed. While methods for detecting floating plastic debris in freshwater ecosystems have been published (Ryan et al. 2009, van Emmerik et al. 2018), terrestrial surveying methods in both natural and built environments have not yet converged despite the well-established need for better understanding the fate of mismanaged land-based plastic debris from human activities (Hoellein and Rochman 2021).

‘Bottom up’, ground-level surveying is inherently challenging because of the need for adequate sampling over time in each place and survey methods for large areas are generally selected based on optimization between physically accessing sufficient data for answering the research question and the feasibility of implementing the data collection in terms of time and financial investments relating to travel, equipment, training, and personnel (Meentemeyer 1989). Existing methods for land-based surveying have ranged from citizen science-based tools (Jambeck and Johnsen 2015), remote sensing (Topouzelis et al. 2019), and advanced image technology (Biermann et al. 2020), however, these approaches have varying advantages and disadvantages due to cost, reliability of data collection, and coverage. Crowdsourced data collection like that used in citizen science initiatives capitalize on the power of many participants (van Emmerik et al. 2020), and volunteer- and citizen scientist-collected data approaches have successfully improved understanding of the abundance, distribution, and composition of anthropogenic litter (Hidalgo-Ruz and Thiel 2013, Rech et al. 2015, Nelms et al. 2020). Further, citizen-science approaches simultaneously benefit those who participate through public engagement and raising awareness on environmental and conservation issues among participants (Wyles et al. 2016), generate large sets of useful data (Hidalgo-Ruz and Thiel 2015), and often remove and properly dispose of mismanaged waste material that would otherwise remain in the environment resulting in improved aesthetics of public places (Nelms et al. 2017). However, citizen science-based methods can suffer in reliability due to varying procedures, levels of training, and detailed record keeping that is needed for producing robust research results (Hidalgo-Ruz and Thiel 2015).

Remote sensing via satellite imagery or imagery collected with unmanned aerial systems (UAS) can provide high quality, standardized datasets over large areas, and are particularly useful for surveying locations that are difficult to reach. Technologies for remote object detection and advanced image analysis for environmental plastic pollution are rapidly developing because of the ability for efficient and ample data collection (Biermann et al. 2020, Tasseron et al. [Preprint]), and there is growing recognition of the potential for detection of plastics using satellite imagery (Goddijn-Murphy and Dufaur 2018, Maximenko et al. 2019, Biermann et al. 2020). Anthropogenic objects can be detected in aquatic ecosystems by differentiating spectral signatures of water, vegetation, and plastics (Garaba et al. 2018, Goddijn-Murphy and Dufaur 2018, Hafeez et al. 2018, Martínez-Vicente et al. 2019, Topouzelis et al. 2019), however the ability to distinguish plastics from other objects is challenged in land-based settings, particularly in the built environment, where there are many more material objects with complex spectral signatures, including in-use plastics.

Although technological advances in surveying tools and imagery analysis improve efficiency and reach for data collection over large areas, these techniques are not necessarily feasible for widespread application due to accessibility challenges associated with equipment and training costs and sophisticated imagery processing that are otherwise not necessary for field observations.

Until advanced technologies are more universally accessible, reliable methods for surveying environmental plastic pollution will rely on optimization of costs, feasibility, and reliability for continued expansion of knowledge in the field. Common ecological sampling techniques such as line and box transect surveys, quadrat analysis, or a combination of them, have been shown to reduce bias in field estimates especially when adapted for field conditions (Fanini and Lowry 2016). Quadrat analysis has been used to document microplastics in beach surveys (Fok and Cheung 2015) and more recently, analysis of land-based transect surveys of anthropogenic debris resulted in high correlations between plastic pollution and variables like land use, infrastructure, socioeconomics, and hyper-local site characteristics such as human presence, vegetation, and site type (Schuyler et al. 2021). Bridging the gap between advanced imagery analysis and traditional field surveying methods, photoquadrat analysis, which combines digital imagery collection with quadrats in the field, allows for rapid data collection, laboratory-based processing, and supports reaching sampling size requirements in applications that require rapid data collection (Parravicini et al. 2009, Molloy et al. 2013). While beach and riverbank settings may not require rapid field surveying approaches for documenting the abundance, distribution, and fate of anthropogenic debris, surveys conducted in urban or built environments can require safety considerations (Schuyler et al. 2021), which could be mitigated via rapid survey methods such as the photoquadrat approach. Here, an exploratory approach for rapid surveying of land-based anthropogenic pollution using photoquadrat data collection is presented. The goals of this research are to 1) demonstrate application of the photoquadrat method across ten land-based communities located in the Ganges River basin, and 2) evaluate spatial and temporal patterns in abundance, density, and composition across the communities based on the method.

5.2 MATERIALS AND METHODS

5.2.1 Description of study site

Photoquadrat data were collected for ten different communities throughout the Ganges River basin during the National Geographic Sea to Source (S2S) Ganges Expedition between May and December 2019. The ten sites were chosen based on proximity to the main river channel (i.e., all sites were within 3 km of the channel) and for achieving spatial coverage of a range of communities of varying population sizes across multiple states and districts within India and Bangladesh (Table 5.1; Figure 5.1). Pre-monsoon data collection occurred in May-June 2019 and post-monsoon data collection occurred in October-December 2019 and collection of photoquadrat data generally occurred over one to three days at each site depending on field conditions.

Table 5.1. Summary of sampling sites.

Site no.	Site name	Country	State/district	Latitude	Longitude
01	Char Fasson	Bangladesh	Bhola	22.17853	90.7101
02	Chandpur	Bangladesh	Chittagong	23.23209	90.66307
03	Rajbari	Bangladesh	Dhaka	23.76437	89.64752
04	Sahibganj	India	Jharkand	25.23909	87.64171
05	Patna	India	Bihar	25.59409	85.13756
06	Varanasi	India	Uttar Pradesh	25.31764	82.97391
07	Kannauj	India	Uttar Pradesh	28.24288	78.35879
08	Anupshahar	India	Uttar Pradesh	28.24288	78.35879
09	Rishikesh	India	Uttarakhand	30.08692	78.26761
10	Harsil	India	Uttarakhand	31.0383	78.73

Figure 5.1. Field site locations.

5.2.2 Procedure for selecting field survey sites

In situ sampling locations were selected using a stratified random sampling procedure in ArcMap 10.8. First, community-wide sampling areas were limited to inland regions that were within 5 km of the riverbank in the direction of the community and within 2.5 km up- and downstream of the approximate center of the community. Sampling areas were further refined based on raster population count data at 1x1 km resolution from the Oak Ridge National Laboratory (ORNL) for 2017, which was the most up to date dataset at time of sampling site selection. The population count data were intersected with the community-wide sampling areas and separated into quintiles so that gridded areas containing the top fifth of the community population were isolated. Using the NOAA Biogeography Branch's Sampling Design Tool for ArcMap, three target areas were randomly selected from the top fifth populated areas of the community, providing three 1 x 1 km areas in which field sampling would occur. Given that transects were only 100 meters in length, the 1 x 1 km areas were partitioned so that three 200 x 200 m sampling locations were randomly selected using the NOAA Sampling Design Tool. The 200 x 200 m size was chosen to allow for flexibility in the field to support access to potential survey areas and adequate safety once the surveying team was at the site. Figure 5.2 provides an example of the target areas and sampling locations generated for one site number four (i.e., Sahibganj).

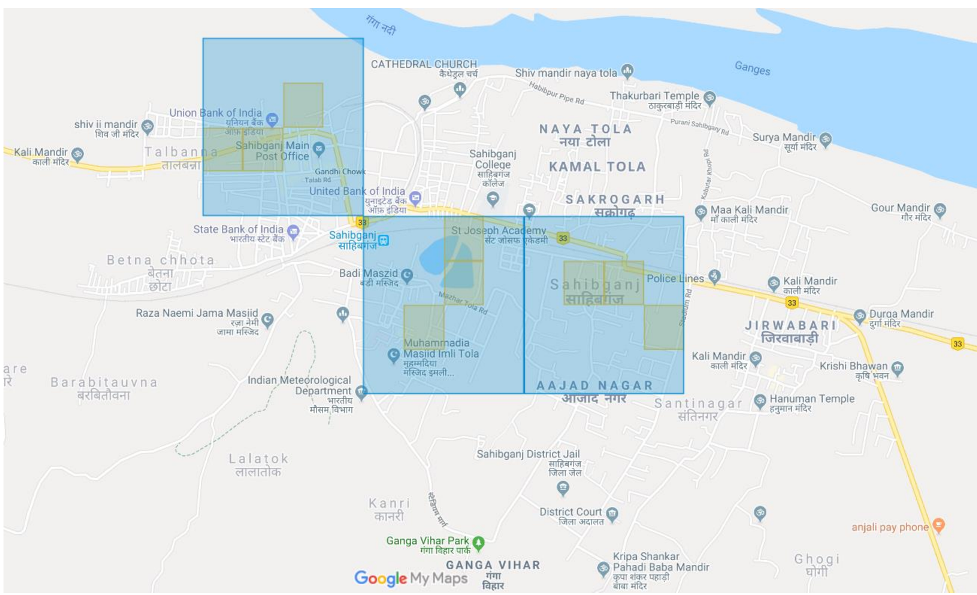

Figure 5.2. Sample target areas and surveying locations in Sahibganj, India; Blue squares, three 1 x 1 km target areas from the top fifth of the population within the community; Yellow squares, nine 200 x 200 m survey locations.

5.2.3 Photoquadrat survey procedure

Field surveys were conducted over 100 meter linear transects that followed pathways and sidewalks adjacent to roads and gutters, providing a safe path for surveyors among the built environment and pedestrian and vehicle traffic, however, this resulted in surveys that were not perfectly straight lines. A stainless-steel 0.5 x 0.5 m quadrat was placed every five meters, starting at 0 and ending at the 100 meters. When the quadrat was in place and stable on the ground, a digital photograph was taken with an Olympus Tough TG-5s camera which is capable of recording latitude, longitude, altitude, and vertical and horizontal positioning (see example in Figure 5.3). Spatial coordinates were recorded in the World Geodetic System (WGS) 1984 geographic coordinate system.

Figure 5.3. Example photoquadrat image.

5.2.4 Photoquadrat image processing

Image processing was undertaken as a collaborative effort between two research assistants based at the University of Georgia in the USA and the University of Plymouth in the UK. Each coder processed images from five communities for both pre- and post-monsoon datasets using ENVI by Harris Geospatial. Debris items were visually detected within the photoquadrat and coded using a shared debris list based on the National Geographic S2S Expedition list in the open access Marine Debris Tracker (MDT) mobile application, which was separately used to conduct surveys in the same transect locations as those presented here. Procedures for the MDT surveys are described in Youngblood et al. (2021).

To facilitate debris identification and differentiation between items that were similar in shape and appearance (e.g., sachets for tobacco packaging, food wrappers, and personal care items), the coders were trained by the author and team members who had relevant field experience with identifying commonly observed products and packaging that were unique to the region. When items were unable to be identified, but were clearly anthropogenic (i.e., foreign to the natural environment or synthetic), they were classified as ‘Unidentifiable Objects’. The documented items were further categorized and aggregated by material type (e.g., metal, glass, plastic) and sub-material type (e.g., tobacco products, food packaging, personal care).

5.2.5 Data analysis

Descriptive statistics

The range, mean, and median item count per quadrat were determined for each transect. Hierarchical litter densities were determined by taking the mean and median item count per quadrat across each transect, then both the mean and medians were averaged for each of the three samples within each site, then averaged by site, and then finally the pre- and post-monsoon data were averaged.

Temporal analysis

Pre-monsoon surveys were conducted between May and June, while post-monsoon surveys were conducted from late October to December (Table 5.2). Visual assessment was used to determine normality of count data for both pre- and post-monsoon. Seasonal influence on litter abundance and corresponding density was examined by aggregating the pre- and post-monsoon data for a subset of fifteen transects. The distribution of the densities across the transects were compared using the Wilcoxon signed rank test for paired scenarios of non-normally distributed data to detect differences between the pre- and post-monsoon conditions.

Table 5.2. Pre- and post-monsoon survey dates in 2019 by site.

Site	Survey dates (2019)	
	Pre-monsoon	Post-monsoon
Bhola	05/14 - 05/16	10/28 - 10/30
Chandpur	05/19 - 05/20	11/01 - 11/03
Rajbari	05/23 - 05/25	11/06 - 11/08
Sahibganj	05/28 - 05/30	11/12 - 11/14
Patna	06/03 - 06/05	11/17 - 11/19
Varanasi	06/07 - 06/09	11/21 - 11/23
Kannauj	06/12 - 06/14	11/26 - 11/28
Anupshahar	06/16 - 06/18	11/30 - 12/02
Rishikesh	06/21 - 06/23	12/05 - 12/07
Harsil	06/25 - 06/27	Not accessible

Spatial analysis

Mean debris densities and average median debris densities were determined for each site, each country, and across all expedition sites combined. Further, proportions of material type, material sub-categories, and item types were calculated for each transect and the average proportion is reported for each site and aggregated by country.

Geospatial patterns in the distributions of item counts, densities, and proportion of plastic items were examined visually across the sites using ArcMap 10.8. Debris densities for each site were then normalized by mean populations identified using the ORNL Landscan dataset for 2019 at each survey location at the time of sampling to facilitate comparisons between the sites and geospatial patterns. The resulting per capita debris densities were plotted against population to visually examine the relationship between the two variables.

5.2.6 Interrater reliability

Interrater reliability (IRR) is a common approach used in qualitative research to assess agreement between multiple coders, however, there are not widely agreed upon guidelines for performing the assessment (O'Connor and Joffe 2020). Previous research using quadrat analysis have commonly used the Cohen's kappa statistic as a measure of IRR (Beijbom et al. 2015, Griffin et al. 2017). Cohen's coefficient of agreement, κ, assesses agreeability between two independent judges deemed individually competent to make qualitative observations over categories on a nominal scale that are both mutually exclusive and exhaustive (Cohen 1960). Cohen's kappa is expressed as:

$$\kappa = \frac{p_0 - p_e}{1 - p_e} \tag{5.1}$$

where,

p_0 = proportion of data units in which the judges agree

p_e = proportion of units for which agreement is expected by chance

$p_0 - p_e$ = proportion of units in which beyond-chance agreement occurred

$1 - p_e$ = proportion of units for which disagreement is predicted between the coders

The calculated Cohen's kappa ranges from -1 to 1, where 1 is perfect agreement, 0 is perfect random agreement, and -1 is perfect disagreement. While the test provides an indication of agreeability between coders, interpretation of the kappa statistic within this range is not necessarily absolute since it is applied to qualitative analysis based on human observations (Davey et al. 2010). Landis and Koch (1977) presented one suggested benchmark for interpretation of the kappa statistics where $\kappa < 0$ represents poor strength of agreement and $\kappa = 0.91 – 1.00$ represents almost perfect agreement. Values that fall between 0 and 0.91 indicate slight, fair, moderate, or substantial strength in agreement. A more conservative interpretation provided by Krippendorff (2018) recommends that for $\kappa < .67$ conclusions should be discounted while $\kappa > 0.8$ indicates definite conclusions can be made. Values in between this range suggest that conclusions should be made tentatively. Both interpretations are reported for the IRR assessment presented here.

For the present study, Cohen's kappa was determined based on a subset of photoquadrat images that were coded by both research assistants from pre-monsoon images from Site 2 (Chandpur) using agreement matrices (example matrix provided in Table 5.3). Counts were first based on absolute presence or absence (i.e., binary categorization) of general debris items in photoquadrat images where presence of items was represented by 1 and absence was represented by 0. Images were further evaluated based on the absolute presence or absence of specific item types such as cigarettes, food wrappers, plastic bags, etc.

Table 5.3. Example agreement matrix from pre-monsoon survey in Chandpur.

	Coder A		
	Absent	Present	*Total*

Coder B	Absent	41	4	45
	Present	6	18	24

5.2.7 Comparison to complementary surveying method

The photoquadrat data collection procedure was performed in parallel with field survey methods that used the MDT mobile application which are described in detail in (Youngblood et al. 2021). In short, the MDT method generated continuous count data in the form of geolocated points through the same 100-meter transects surveyed with the photoquadrat method described here. Unlike the photoquadrat, which was bound by the 0.5 x 0.5 m quadrat, MDT data covered a 1-meter width visually approximated by reporters in the field. The MDT data were converted to a litter density for each transect and aggregated by site for comparison with the photoquadrat data. The resulting densities from the MDT and photoquadrat surveys were first compared between a subset of 18 transects from six sites. Direct comparisons were made between the transects based the mean density and proportion of materials, sub-materials, and items. The distribution of the densities determined for each transect by the two methods were further compared using the Wilcoxon signed rank test, which is a non-parametric test of comparisons for paired non-normally distributed data.

In addition to the direct transect comparisons, the litter densities, overall plastic proportion, and the proportion of five items are compared based on the items that were most recorded by each survey method for each site. Youngblood et al. (2021) reported litter densities normalized by 1,000 people for the MDT based on three population brackets: low (100 – 2,000 people), mid (2,000 – 10,000), and high (>10,000). For comparison, the photoquadrat debris densities were similarly normalized and categorized into the low, mid, and high population categories. Further the proportions of plastic in each population category were compared. Lastly, the counts of top five items of the MDT and photoquadrat data were contrasted side-by-side for the overall datasets.

5.3 RESULTS

5.3.1 Descriptive statistics

In total of 2,309 photoquadrat images were included in this study. Most photoquadrats were reported to have at least one item present, however, 24% of photoquadrats had zero counts. Most (21%) had counts of five items or less, and visual assessment of the distribution of count data across the all photoquadrats followed a non-normal, right-skewed pattern Figure 5.4. Samples from pre- and post-monsoon samples and individual cities followed this same distribution.

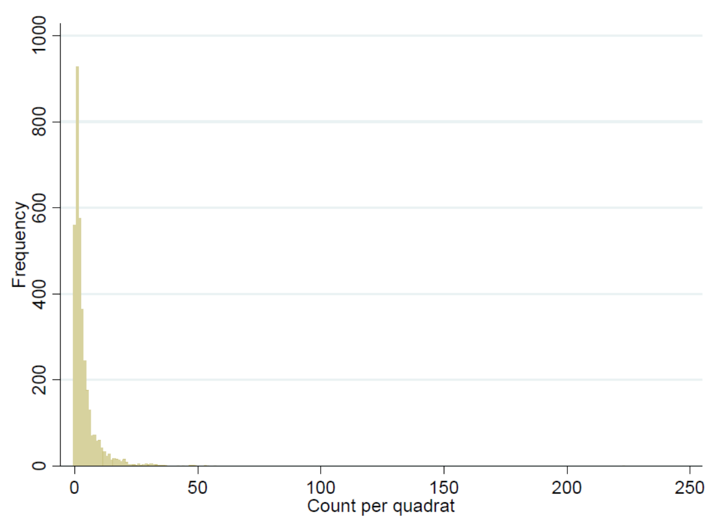

Figure 5.4. Frequency distribution of photoquadrat item counts.

Anthropogenic litter was found in 100% of the transects, with a total of 9,520 items documented across a cumulative 11,100 m^2 of transect areas and a cumulative 577

m^2 of quadrat areas (equivalent to 5.2% coverage of the total transect areas surveyed). Total count of items/quadrat ranged from 0 to 223 items in one photoquadrat from pre-monsoon Patna, however, this quadrat image was considered an outlier ($z = 29$), and upon visual inspection of the image, it was removed because the large number of items appeared to be individual paper pieces that were scattered after being hole-punched from full sheets of paper. This was not seen in any other location, and so the data point was removed as an outlier. After removal, the maximum count in a single quadrat was 57 in Rajbari. Across all transects, mean item counts ranged from 0.14 items in Anupshahar to 16 items in Chandpur, with an overall mean item count of 4.1 ($sd = 3.1$; average $mdn = 3.05$).

The most frequently identified material type was plastic, which accounted for 58% of all items, followed by other (19%), and paper (15%). The least identified materials were e-waste (0.1%), fishing gear (0.5%), and C&D materials (1%). The most frequently identified sub-material types were plastic fragments which made up 36% of all items, followed by other (19%), and paper (15%). By item type, film plastic fragments were the most frequently identified item, representing 29% of all items, followed by unknown objects (18%), and paper (15%). The least found item types were other glass, tires, and cotton buds, each of which represented 0.01% of all items. Under the plastic material category, 5,527 items were identified, with the most frequently recorded sub-material category being plastic fragments (62% of all plastic items), food plastic (20%), and tobacco products (13%). By plastic item type, film plastic fragments were the most frequently recorded item ($n = 2,790$), representing 50% of all plastic items, followed by hard plastic fragments (9.2%), and plastic food wrappers (7.5%).

Table 5.4. Top five material types, top five sub-material types, and top ten most common item types by count and proportion.

Category	*n*	Proportion of total item count (%)
Material type		
Plastic	5527	58
Other	1831	19
Paper	1444	15
Cloth	225	2.4
Metal	119	1.3
Sub-material type		
Plastic fragment	3440	36
Other	1831	19
Paper	1444	15
Food plastic	1082	11
Tobacco products	711	7.5
Item type		
Film plastic fragment	2790	29
Unknown object	1753	18
Paper	1390	15
Hard plastic fragment	509	5.3
Plastic food wrapper	412	4.3
Cigarette	401	4.2
Tobacco sachet	278	2.9
Plastic bag	217	2.3
Fabric piece	157	1.6
Plastic string	148	1.6
Total		9,520

5.3.2. Temporal analyses

During the pre-monsoon expedition, nine surveys were conducted at each site resulting in 1,756 photoquadrat images from the pre-monsoon surveys with a total count of 7,610 identified items. Only three surveys were conducted at each site during the post-monsoon field expedition, resulting in 553 photoquadrat images from the post-monsoon surveys and a total item count of 2,133 items identified. The mean pre-monsoon litter density was 0.62 items/m^2 (sd = 0.22), compared to the post-monsoon litter density which was 0.55 items/m^2 (sd = 0.26). A Wilcoxon rank signed test did not indicate a statistically significant difference between the distributions of transect counts in the pre- and post-monsoon surveys (z = -1.595, p = .111), which is visually supported by box plots shown in Figure 5.5.

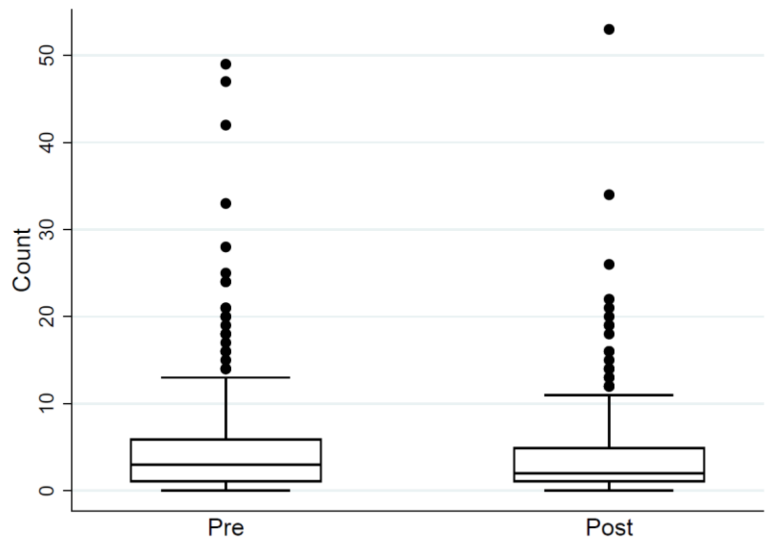

Figure 5.5. Box plot of pre- and post-monsoon debris count data for transects (n =15).

5.3.3 Spatial analyses

Three sites were Bangladesh, while seven were in India (Figure 5.1), and so most (n = 81) transects were performed in India compared to 30 transects in Bangladesh. As such, most items were recorded in surveys within India. Standardizing the counts by area to generate a litter density allowed for direct comparisons between locations. The overall litter densities were similar between the two countries, with a resulting average median debris density of 0.59 items/m^2 in Bangladesh and 0.57 items/m^2 in India (Table 5.5).

Table 5.5. Range, mean, and median items/meter squared in each site.

Site no.	Site name	Country	# Transects (pre/post)	Range of counts (mean count)	Mean debris items (item/m^2)	Avg. median value debris items/m^2
01	Bhola	Bangladesh	6 (3/3)	0 – 31(4.1)	0.83	0.72
02	Chandpur	Bangladesh	12 (9/3)	0 – 37 (5.2)	1.03	0.70
03	Rajbari	Bangladesh	12 (9/3)	0 – 57 (3.0)	0.61	0.36
04	Sahibganj	India	12 (9/3)	0 – 47 (3.3)	0.66	0.43
05	Patna	India	12 (9/3)	0 – 54 (6.5)	1.31	0.92
06	Varanasi	India	12 (9/3)	0 – 32 (3.5)	0.69	0.52
07	Kannauj	India	12 (9/3)	0 – 53 (4.4)	0.87	0.61
08	Anupshahar	India	12 (9/3)	0 – 23 (2.2)	0.45	0.33
09	Rishikesh	India	12 (9/3)	0 – 30 (5.1)	1.01	0.83
10	Harsil	India	9 (9/0)	0 – 87 (2.2)	0.43	0.33
		Bangladesh	30 (21/9)	0 – 57 (4.1)	0.82	0.59
		India	81(63/18)	0 – 54 (3.9)	0.77	0.57
ALL			111 (84/27)	0 – 57 (3.9)	0.79	0.58

By material type, the plastic made up the largest proportion in both countries in similar amounts, with plastic making up 60% and 57% of items in Bangladesh and India, respectively. By sub-material, plastic fragments were most frequently recorded in both countries and with similar proportions observed (34% and 37% in Bangladesh and India, respectively). Indian sites had a higher proportion of plastic fragments and food plastic, while sites in Bangladesh had a higher proportion of tobacco products and the ‘other’ plastic category. Although not among the top five sub-materials, Bangladesh had a slightly larger proportion of fishing gear than India (0.9% and 0.4%, respectively).

By item type, film plastic fragments, unknown objects, and paper made up the largest proportions of items in both countries. Taken together these items made up 65% and 54% of all items in India and Bangladesh, respectively. Unknown objects and paper items both had similar proportions between the countries, but film plastic fragments represented a larger proportion of items in India (32%) compared to 22% of all items in Bangladesh (Table 5.6). Following the top three items, the countries’ debris items and corresponding proportions deviated. Hard plastic fragments were the fourth most frequent item in India but were the fifth most frequent item in Bangladesh, however, hard plastic fragments made up a larger proportion of the total count in Bangladesh (6.6% compared to 4.3% in India). Cigarettes were the fourth most frequent item in Bangladesh, making up 10% of the total item count, while cigarettes were the eighth most frequent item, accounting for only 1.9% of all items in India. Compared to cigarettes, tobacco sachets were the sixth highest item in India, making up 4% of items in India, but this item was not among the top ten items in Bangladesh. (Table 5.6).

Based on visual observation, the distribution of debris densities and plastic proportions across the ten sites did not have any discernable geospatial patterns (Figure 5.6), however, the largest debris densities were spread across the basin, but corresponded with highly populated sites (i.e., Patna, Chandpur, and Rishikesh), while smallest debris densities were observed in sites that were the least populated at the survey locations (i.e., Harsil, Anupshahar, and Rajbari). By site, the average median debris items ranged from 0.33 items/m^2 in both Anupshahar and Harsil to 0.92 items/m^2 in Patna, with the average median litter debris density across all sites being 0.58 items/m^2 (*sd* = 0.21; Table 5.5)

Table 5.6. Comparison of top ten items in India and Bangladesh by proportion of total plastic items

India			Bangladesh		
Item	*n*	Proportion (%)	Item	*n*	Proportion (%)
Film Plastic Fragment	2,632	32	Film Plastic Fragment	1,045	22
Unknown Object	1,571	18	Unknown Object	738	19
Paper	1,312	15	Paper	517	13
Hard Plastic Fragment	347	4.3	Cigarette	371	10
Plastic Food Wrapper	331	4.0	Hard Plastic Fragment	286	8.1
Tobacco Sachet	296	4.0	Plastic Food Wrapper	181	5.2
Plastic Bag	268	2.3	Foam Plastic Fragment	141	3.3
Cigarette	160	1.9	Plastic Bag	156	2.3
Other Plastic	143	1.5	Fabric piece	117	2.2
Fabric Piece	140	1.5	Plastic String	115	1.9
Total	6,928		*Total*	3,633	

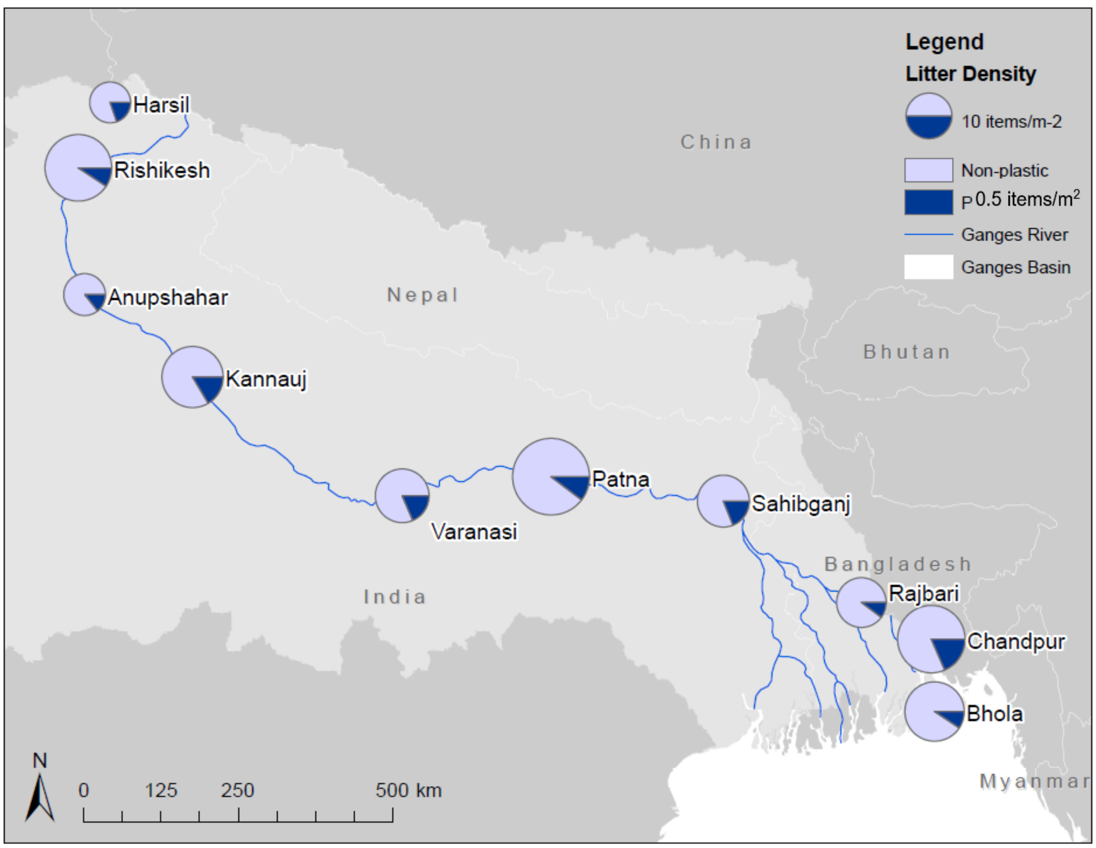

Figure 5.6. Average median litter density and proportion of plastic items by site.

By material, plastic had the highest proportion by material type for all sites except Bhola, Rishikesh, and Rajbari which mostly consisted of the ‘other’ material category (47% and 45%, respectively). The proportion of plastic did not correspond with population size, with the highest proportions in seen in Sahibganj, Harsil, and Chandpur, which ranged in population size. Plastic proportions across the sites ranged from 37% in Bhola to 76% in Sahibganj (Figure 5.7). Although fishing gear represented a small proportion of the surveyed items, the highest proportion of fishing gear related debris was found in Chandpur (1.1%), which was the second most downstream site (Figure 5.1).

Among the plastic sub-material categories, plastic fragments had the highest proportion across all sites, and ranged from 55% of all plastic items in Chandpur to 73% of plastic items in Anupshahar. with a mean proportion of 61% (Figure 5.8). Food plastics were the second most prominent item, which ranged from 16% in Anupshahar to 22% in Bhola. Lastly, tobacco products were the third most prominent plastic sub-material and ranged from 5.5% in Kannauj to 24% in Rajbari.

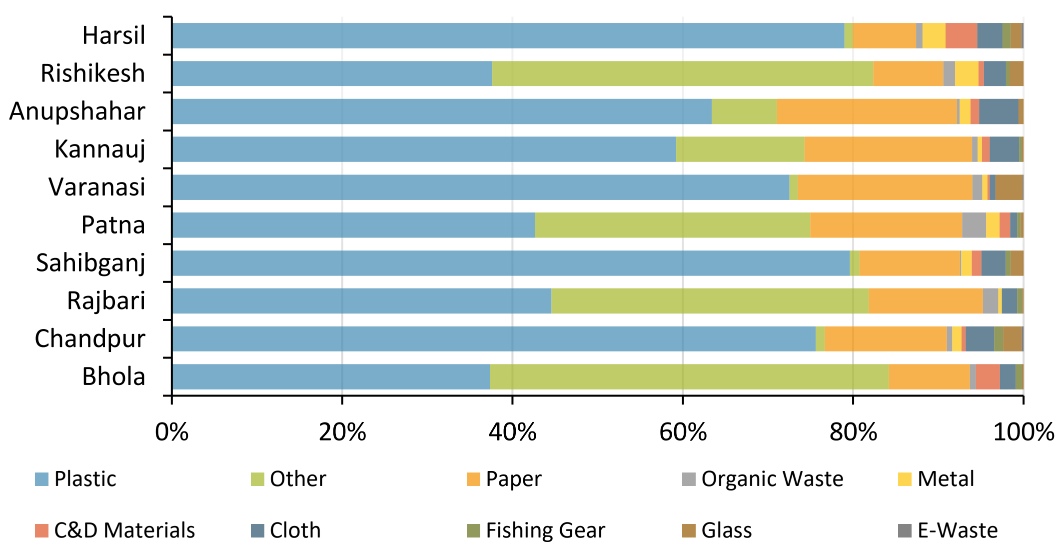

Figure 5.7. Proportion of materials by site.

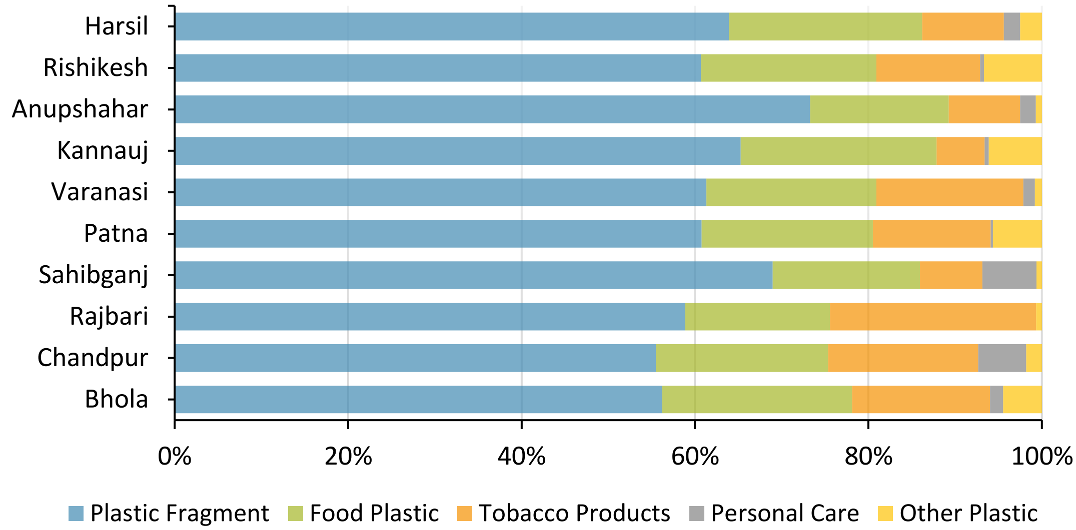

Figure 5.8. Proportions of plastic sub-categories by site.

When accounting for population, the distribution of mean count and debris density changed considerably across the sites. The mean population across all transect locations was 14,000 at the time of surveying based on ORNL Landscan population count data for 2019. Mean populations by site ranged from 669 people in Harsil to 45,900 in Patna. The three sites in Bangladesh had a higher mean population than the seven in India (17,300 and 12,500 people, respectively) although the range and spread of population counts were smaller in Bangladesh. Normalizing the debris density by the mean population for each community resulted in a mean per capita debris density of 1.9×10^{-4} items/m^2, with the highest per capita debris density in Bhola (5.7×10^{-4} items/m^2) and the lowest in Varanasi and Chandpur (1.9×10^{-5}) items/m^2; Figure 5.9).

The relationship between the mean population and mean debris density across each site followed a quadratic fit, such that the minimum debris density could theoretically occur in populations around 16,000 and increase as population either increases or decreases (Figure 5.10). That said, the sites with larger populations had almost uniform per capita debris densities, revealing a linear relationship between population and debris density among the more populated sites (i.e., Patna, Chandpur, Varanasi, Rajbari). Raw debris density values and per capita debris densities varied among the less populated sites despite similarity in population sizes. For instance, Anupshahar and Rishikesh had similar populations at the survey sites, however, the debris density was 2.5 times higher in Rishikesh. As a result, the relationship between debris density and population in the less populated sites was less predictable than in higher sites.

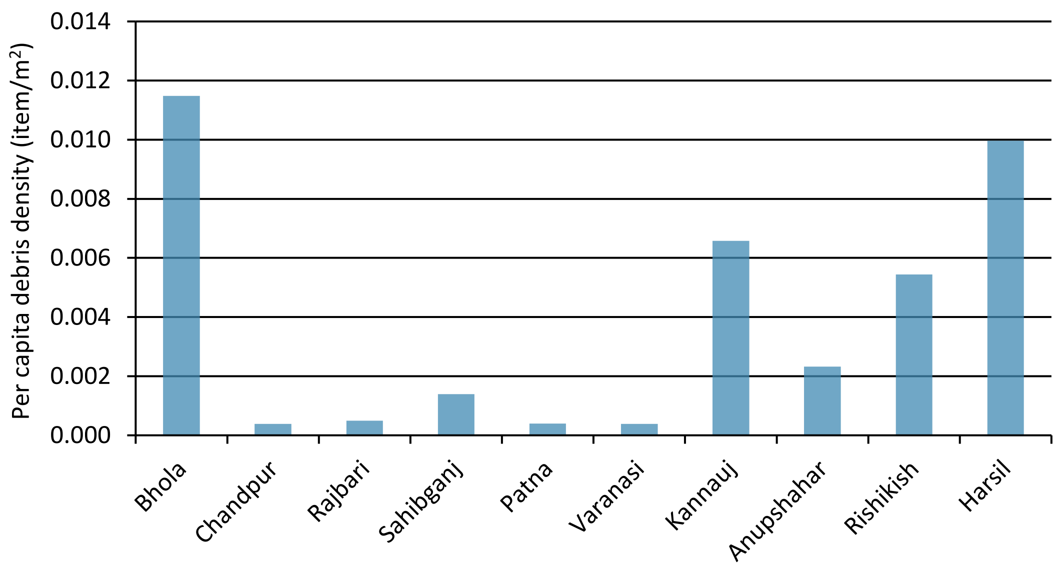

Figure 5.9. Per capita debris density by site.

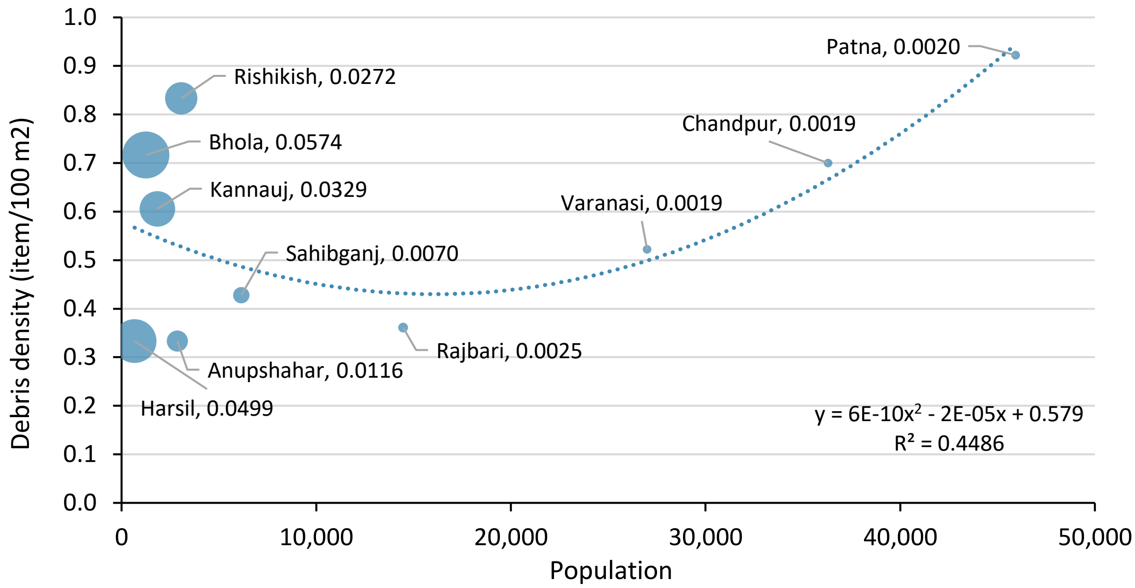

Figure 5.10. Bubble scatterplot showing the relationship between population and debris density across sites. Bubble size represents per capita debris density per 100 m^2.

5.3.4 Interrater reliability

Cohen's kappa for overall presence or absence of debris in the photoquadrat data for each transect ranged from 0.68 to 0.70, with a mean κ of 0.73, indicating substantial agreement based on the interpretation suggested by Landis and Koch (1977), and tentatively conclusive based on the interpretation by Krippendorff (2018; Table 5.7). When comparing the counts of specific items between coders, overall agreement was reduced. Cohen's kappa for the item-based assessment ranged from -0.39 to 0.22, with a mean κ of -0.13, indicating disagreement (Landis and Koch 1977) and discounted conclusions (Krippendorff 2018). Agreement was only observed in transect C and the combined transects (Table 5.7).

Table 5.7. Results of Cohen's kappa test for agreement; κ > 0 = agreement and κ < 0 = disagreement.

Transect	Cohen's κ	Interpretation: Strength of agreement/disagreement (Landis and Koch 1977)	Interpretation: Reliability of conclusions (Krippendorff 2018)
Agreement on presence of any debris			
A	.70	Substantial	Tentative
B	.68	Substantial	Tentative
C	.83	Almost perfect	Definite
Mean	.73	Substantial	Tentative
Combined	.67	Substantial	Tentative
Agreement on presence of specific debris items			
A	-.39	Fair disagreement	Discounted
B	-.23	Fair disagreement	Discounted
C	.22	Fair agreement	Discounted
Mean	-.13	Fair disagreement	Discounted
Combined	.16	Fair agreement	Discounted

5.3.5 Comparison to MDT data

The photoquadrat transects assessed for comparison to the MDT findings had a mean debris density of 0.71 items/m^2 compared to the mean debris density being 6.2 items/m^2 by the MDT method. Densities did not appear to align when compared between transects. By site, photoquadrat densities ranged from 0.23 items/m^2 Sahibganj to 1.3 items/m^2 in Patna, while the MDT densities ranged from 2.9 items/m^2 in Kannauj to 8.5 items/m^2 in Sahibganj. Direct comparisons of litter densities between transects resulted in percent differences ranging from -67% in Patna to -96% in Rishikesh, with a mean percent difference of 87% (Table 5.8). The mean difference between the two method densities ranged from –2.1 items/m^2 in Kannauj to -12 items/m^2 in Rishikesh, with a mean difference of -5.5 items/m^2. All densities determined by the MDT analysis were higher than that of the photoquadrat densities (Figure 5.8). A Wilcoxon rank signed test indicated a statistically significant difference between the distributions of transect counts in the photoquadrat and MDT surveys (z = 3.724, p = .0002).

Table 5.8. Comparison of item density people by transect (PQ, photoquadrat; MDT, Marine Debris Tracker).

Site	Transect ID	Item density (items/m^2)		Diff.	% diff.
		PQ	MDT		
Anupshahar	anup1	0.57	7.5	-6.9	-92
	anup2	0.40	8.0	-7.6	-95
	anup3	0.53	6.2	-5.7	-91
Kannauj	kann1	0.80	2.9	-2.1	-72
	kann2	0.73	4.1	-3.3	-82
	kann3	0.60	7.3	-6.7	-92
Patna	patn1	0.87	8.0	-7.1	-89
	patn2	1.3	5.6	-4.3	-76
	patn3	1.2	3.6	-2.4	-67
Rishikesh	rish1	0.47	12	-12	-96
	rish2	0.47	5.3	-4.8	-91
	rish3	1.1	6.0	-4.9	-82
Sahibganj	sahi1	0.23	5.1	-4.9	-95
	sahi2	0.53	8.5	-7.9	-94
	sahi3	0.80	5.0	-4.2	-84
Varanasi	vara1	0.60	3.6	-3.0	-83
	vara2	1.1	6.3	-5.2	-82
	vara3	0.40	6.0	-5.6	-93
	mean	0.71	6.2	-5.5	-87
	sd	0.31	2.2	2.4	8.5

When the methods were compared by normalized densities across the three population brackets identified by Youngblood et al. (2021), the mean percent difference was 86%, with the largest difference observed in the high population group and the smallest difference observed in the low population group. The mean absolute difference between each methods' debris densities across the population categories was 1.53 items/m^2, with the largest absolute difference seen between the low population group and the smallest seen in the high population group (Table 5.9). Despite the differences seen between the densities from each method, the densities across the population categories followed a similar pattern wherein the high population group had the lowest per capita litter density, and the low population group had the highest. However, photoquadrat method did have a larger difference between the densities in each population group compared to the MDT densities.

Table 5.9. Comparison of item density per 1,000 people by population category (PQ, photoquadrat; MDT, Marine Debris Tracker)

Pop. category	Pop. range	Item density (items/m^2) per 1,000 people		Diff.	% diff.
		PQ	MDT		
Low	100 - 2,000	0.47	6.43	-5.96	93
Mid	2,000 - 10,000	0.15	1.59	-1.44	90
High	> 10,000	0.02	0.19	-0.17	89
mean		0.21	2.74	-2.52	91

By material type, the plastic proportions from the photoquadrat method were consistently less than that of the MDT results. The largest absolute and percent difference between the plastic proportions from each method was observed in the high population group, while the smallest absolute and percent difference as observed in the low population group. The two methods did not have similar patterns across the population categories like that observed with the debris densities. The proportion of plastic was nearly uniform across the population categories for the photoquadrat data, while the MDT proportion of plastic ranged from 75% in the high population category to 88% in the low population category.

Table 5.10. Comparison of plastic proportion by population category (PQ, photoquadrat; MDT, Marine Debris Tracker)

Pop. category	Pop. range	Plastic		Diff.	% diff.
		PQ	MDT		
Low	100 - 2,000	59	88	-29	-33
Mid	2,000 - 10,000	60	85	-24	-29
High	> 10,000	59	75	-16	-22
mean		59	82	-23	-28

Plastic fragments accounted for the largest proportion by item type for the low, mid, and high population sites based on the photoquadrat method (Figure 5.11). This finding did not align with the results of the MDT transect method, which was mostly comprised of tobacco products and food wrapper across the low and medium population sites and other plastic items in the high population sites.

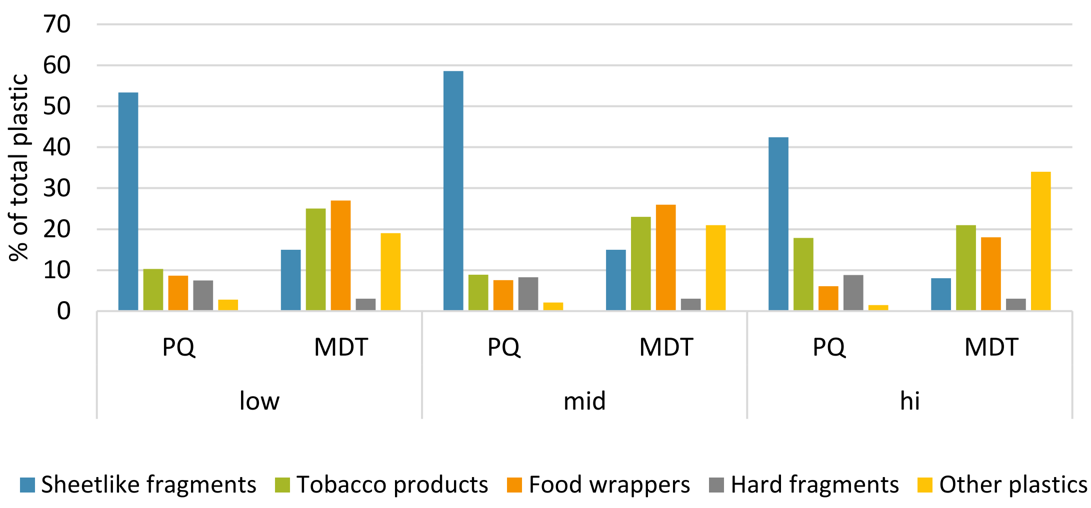

Figure 5.11. Column chart comparing top plastic item types by proportion for each population category. (PQ, photoquadrat; MDT, Marine Debris Tracker; low, mid, and high refer to the population brackets).

5.4 DISCUSSION

5.4.1 Descriptive statistics

Across all sites, plastic had the highest proportion by material among the photoquadrat samples. Most plastics are produced for packaging (Plastics Europe 2020) and designed for rapid disposal resulting in a short product lifetime (Geyer et al. 2017). A wide array of plastic packaging formats is tailored for storage, shelf-stability, and marketing the delivery of specific food, beverage, and tobacco products to consumers, and with low production and transportation costs associated with plastics, plastic packaging improves consumer accessibility to products. However, extreme variation in plastic packaging formats results in difficulty managing and disposing post-consumer plastic packaging waste, leading to reduced value retention, reduced incentivization for collection and treatment, and increased incidence of loss from waste management systems (Moss 2017). Food and tobacco packaging were the fourth and fifth most identified sub-material type and together comprised 19% of all items, however, the proportion of plastic fragments was nearly three times that (58%). The prevalence of plastic fragment among the identified items is consistent with other recent land-based surveys in both built environments and beaches, in which plastic fragments were the predominant portion of littered anthropogenic debris as well (Heo et al. 2013, Jayasiri et al. 2013, Hardesty et al. 2017, Nelms et al. 2017, Weideman et al. 2020, de Ramos et al. 2021, Giles et al. 2021). Because fragments are generally secondary plastics that result from weathering and abrasion of larger items (Andrady 2011), higher counts may be expected due to the physical transformation of one item to many. Fragments can be hazardous when ingested by wildlife due obstruction of the gut (Kühn et al. 2015), which can result in rapid onset of stress and mortality (Bergmann et al. 2015). In the region surveyed for this study, this may be a particular risk as many animals (i.e., livestock and canines) were observed scavenging among litter and accumulated waste at the time of photoquadrat data collection in the field. Dairy and meat can contribute to dietary exposure of harmful chemicals and micro- and nano-plastics through trophic transfer, which may have implications for human health (Smith et al. 2018).

Among the fragments, the most common type was film plastic fragments. Film fragments may be associated with plastic bags used in retail or food packaging, which have been banned in multiple countries and cities throughout the world (UNEP 2018a, UNEP 2018b). Bangladesh was the first country to ban polyethylene plastic bags in 2002, however poor enforcement and lack of alternative options for retailers and consumers to use has resulted in the ban being ineffective (UNEP 2018b). India has previously released regulatory efforts for banning some plastic packaging, supports establishment of infrastructure and extended producer responsibility schemes (Karasik et al. 2020). However, a 2009 effort to ban plastic bags in Delhi failed due to lacking public awareness and poor enforcement of the policy and its corresponding penalties (Gupta 2011). More recently, India also announced a national phase out on single-use plastic items by July 2022, which would include a ban on non-woven plastic bags, and prohibit manufacturing, import, distribution, and sale of items such as plastic utensils, cigarette packets, and PVC (Ministry of Environment Forest and Climate Change 2021). Previous research has empirically documented the effect of policies and levies targeting specific plastic items in aquatic and beach settings (Maes et al. 2018, Schuyler et al. 2018). The land-based debris surveying methods like that presented here could support monitoring presence of banned or regulated plastic items in the built and urban environments where many of these items are sold, consumed, and disposed.

5.4.2 Temporal analysis

There was no indication that the pre- and post-monsoon debris densities differed based on the photoquadrat data, which would suggest that land-based debris items documented in the pre-monsoon surveys were flushed during the rainy season and then rapidly reaccumulated to similar levels at the time of post-monsoon surveying. Anthropogenic debris in stormwater catchments can discharge into receiving bodies of water (Weideman et al. 2020), and there is increasing evidence linking monsoon rains to increased plastic debris discharged from rivers (van Emmerik et al. 2019a, van Emmerik et al. 2019b, van Emmerik et al. 2019c, Vriend et al. 2020, Meijer et al. 2021). Past research has also estimated that plastic discharge from the Ganges peaks in August at 44,500 metric tons per month due to the monsoon season and slows to 150 metric tons per month in December, the dry season (Lebreton et al. 2017). Further, recent land-based surveys of beaches on the western coast of India found higher deposition of shoreline debris during monsoon season (Sulochanan et al. 2019). While the similarity between pre- and post-monsoon quantities of litter was not surprising, future research could benefit from increasing temporal coverage by increasing the number of sampling events per year to better detect variations in time. For instance, broader temporal data could reveal seasonal differences that are unique to the region, such as the religious significance of the river, which is host to numerous large religious festivals, rituals, and events given that there is some evidence that debris increases along shorelines in India during events such as these (Jayasiri et al. 2013). Notably, both pre- and post-monsoon surveys were conducted over multi-week field expeditions, and so there could be undetected differences in items recorded in sites depending on when they were measured during each expedition period. Monsoons season typically spans June to September (Lebreton et al. 2017), which could have influenced presence of anthropogenic debris particularly during the pre-monsoon surveys. Expansion of this research may include assessment of climate and hydrologic influences in that period to better account for impacts the monsoons may have had.

5.4.3 Spatial patterns

India and Bangladesh had comparable litter densities, similar proportions of plastics, sub-materials, and item types. One notable difference was found in the composition of tobacco packing, which suggests variation in accessibility to and consumption of different tobacco products between the countries. This is a key insight for the design and development of effective policies, such as the phase out of cigarette packaging in India described previously. The photoquadrat analysis revealed higher quantities of plastic sachets used for loose tobacco in India, suggesting that regulations limited to cigarette packaging may be ineffective for the more commonly littered sachets. Documentation of products specific to communities is beneficial for examining the distribution of product types across communities to inform effective waste management strategies and policies (Rochman et al. 2016). In this case, policies targeting tobacco sachets may be more effective in India for example.

Across the sites, the average median litter densities were comparable to previously published land-based debris densities reported in previous studies. Although no surveys have reported debris abundance among non-coastal (i.e., inland) sites in the region for comparison, one recent study assessing stormwater discharge of land-based debris in urban catchment systems in Cape Town, South Africa, determined an average land-based debris density of 0.026 items/m^2 (Weideman et al. 2020), which equates to 4.5% of the average median debris density reported here. This difference could be driven by local socioeconomic characteristics, as the South African densities were reflective of high- and middle-income catchment communities. More recently, land-based methods using similar transect methods in various urban and built environments found a median density range of 0.04-0.58 items/m^2 and average median of 0.24 items/m^2 across seven urban cities in developing countries, equivalent to 50% of the average median density reported here. India and Bangladesh were not represented in the study, however, the densities reported are comparable in magnitude, although the densities found in the Ganges River basin communities were mostly higher than seven cities that were surveyed (Schuyler et al. 2021). Although the debris densities observed in other studies are of the same magnitude to that reported here, differences in values could be driven by methodological variation in survey methods used, which supports that improved standardization of methods would allow for more reliable comparison between surveys.

As described earlier, empirical evidence documenting the quantities and composition of anthropogenic debris provide utility in informing effective policy and efforts to prevent leakage into the environment. Variation in composition was observed in between Bangladesh and India, but this variation was also seen at the community-scale.

For example, the site with the highest proportion of plastic, Sahibganj, also had the highest count and proportion of plastic bottles across the sites. Plastic bottles, generally made of polyethylene terephthalate (PET), retain their value as post-consumer waste compared to other polymers and formats of plastic packaging and are more frequently collected by the informal sector (Moss 2017). The relatively high proportion of PET bottles in Sahibganj could very well be due to chance, and so repeated surveys combined with deeper investigation into the waste management systems there could provide more insight into the reasons for the difference. Similarly, a recent study that conducted riverbank surveys along the Ganges River found a mean density of 0.013 items/m^2 of abandoned, lost, or derelict fishing gear (ALDFG). In that study, Chandpur had the highest mean debris density of ALDFG items (0.076 items/m^2; Nelms et al. 2021). In the present study, Chandpur also had the highest proportion of fishing gear related items of all sites, and all three sites in Bangladesh had higher debris densities than that in the upstream sites in India, which aligned with Nelms et al. (2021), who determined that Bangladesh had higher densities of ALDFG compared to India.

The lack of geospatial differences in densities observed throughout the basin suggests that waste from upstream communities likely are not depositing in communities downstream. This could provide preliminary evidence that rather than transporting to other land-based locations downstream, debris items either remain in the river, are transported, and deposited in marine environments, or are captured in dams and collected for management, rather than depositing in other communities. Recent research empirically modeled the transport of plastic bottles in the Ganges River, though it was not conclusive as to whether that land-based accumulation of bottles in downstream communities was common (Duncan et al. 2020), suggesting that downstream communities may not be a major sink for debris in the river. To provide further knowledge related to the downstream deposition of debris, future research should examine the difference in quantities between inland communities and riverbank debris as well as variation in up- and downstream debris.

The communities with the highest per capita debris densities—Bhola and Harsil—were at the extreme ends of the extent of the expedition sites, with Bhola being the first site surveyed in southern Bangladesh, and Harsil being the last site surveyed in the Himalayan Mountains in northern India (Figure 5.1). These two communities had the smallest mean populations among the photoquadrat survey locations and were the most remote of the ten sites. Smaller communities that are distant from urban centers may lack physical access to waste collection and treatment of waste, resulting in higher per capita debris like that in Harsil and Bhola (Figure 5.9). Proximity to urban centers can influence access to services and opportunities (Cattaneo et al. 2021), which may support urban access to waste management infrastructure and potentially result in lower per capita debris densities.

It is well established that urbanization is correlated with industrialization, and the link between industrialization and environmental degradation has been frequently demonstrated via the environmental Kuznets curve (EKC), which describes a theoretical curvilinear relationship between per capita income and environmental degradation that can be used to predict inequality. The EKC shows that environmental degradation is lower in pre-industrialized societies due to low consumption, but it increases as societies experience industrialization and growth in productivity. After peaking during industrialization, environmental degradation then decreases as income per capita increases during the post-industrialization phase of development (Kuznets 2019). Barnes (2019) demonstrated that mismanaged plastic waste per capita follows the EKC, where plastic pollution increases as societies industrialize and decreases post-industrialization based on per capita income and scientific advancements. While per capita income is not examined as an indicator of debris density for the present study, the relationship between the populations in each surveyed site and their per capita debris densities provide evidence that the urban centers (e.g., Patna, Varanasi, Chandpur), which are generally more industrialized, may see lower per capita debris densities.

Under the EKC hypothesis, rural sites that are not urbanized would have lower per capita debris density as well, however, this condition is not met in the present study as the sites furthest from urban centers (i.e., Harsil and Bhola) had the highest per capita debris densities. This finding may be explained by the ubiquity of plastic packaging in goods that are consumed in both urban and rural locations alike. Although the smaller populations in rural communities may generate smaller overall quantities of plastic waste, because of the accessibility of products packaged in plastic, rural consumption of plastic could be like urban plastic consumption, which could contribute to the higher per capita debris density observed in rural areas if their plastic consumption patterns have not aligned with growth in industrialization or increasing per capita income in more urban areas. This phenomenon may be evident in societies described by Gollin et al. (2016) as ‘consumption cities’, which experience urbanization through reliance on importing goods without having resources to offer tradable services. Despite urbanization and available economy of scale for advanced infrastructure systems, these consumption cities often have poor living conditions. Prior research has shown that anthropogenic debris is negatively related to income (Marais et al. 2004, Schuyler et al. 2018). This complex relationship between economic growth, urbanization, and mismanaged plastic waste warrants further investigation beyond the scope of this study, but the results presented here provide a basis for doing so given the implications for policy development and strategies needed for management plastic waste and prevention of leakage into the environment.

5.4.4 Interrater reliability

IRR was not conclusive for the present study based on the methods for assessment of IRR that were used, given that the results varied depending on the level of data that were assessed (i.e., substantial agreement based on presence of general items versus disagreement based on presence of specific items). Because of these differences, it cannot be concluded that agreeability was found between the coders, and so results from the photoquadrat data are taken cautiously. Despite this drawback, Cohen's kappa statistic is limited as it does not account for the weight of count data and is instead calculated based solely on the presence or absence of items in a photoquadrat, but not the total number of items. In other words, if coder A identifies ten items in a photoquadrat and coder B identifies only three items in the same photoquadrat, agreement is met based on both coders reporting the presence of items rather than the quantity of items reported.

Additionally, at the time of writing, only three transects were available from the full data set that were processed by both coders, which is likely inadequate coverage of the data to reliably determine the consistency of agreement between the coders ability to recognize debris items. Further processing of overlapping photoquadrat data has been completed by the raters and will be incorporated into expansion of this work. For the present IRR analysis, variation between the coders could have been influenced by the site, which ranged in number of item counts, the conditions in which processing was conducted (e.g., distracting surroundings), level of fatigue, and experience with processing and identifying item types specific to the sites surveyed. Future research will supplement the existing IRR assessment by increasing the number of transects that have been processed by both coders.

5.4.5 Comparison to MDT

Based on the comparison between photoquadrat and MDT methods used for surveying the ten sites along the Ganges River, there was inconclusive agreement in debris densities. Item counts and corresponding densities based on the photoquadrat method described here were much lower than that of the MDT method, and the sites that saw the highest and lowest densities according to the photoquadrat method did not align with that in the MDT, and in fact Sahibganj had the lowest transect density of the photoquadrat data and the highest transect density in the MDT data. Accounting for population did not substantially improve agreement between the densities determined by each method, although debris density increased as population decreased in both methods. The photoquadrat surveys did result in litter densities similar to other studies on land-based debris, as reported in Section 5.4.1, but the larger values determined by the MDT method were within the range of debris densities reported by Schuyler et al. (2021) for seven urban cities in developing countries, which ranged from 0 to 52.4 items/m^2, the mean and median litter densities reported by Schuyler et al. (2021) were more similar to that determined by the photoquadrat analysis.

Given the differences in the methods procedures, it is not surprising that differences in the resulting counts and densities were observed. For example, the proportions of plastic determined by the photoquadrat analysis were lower than the MDT method. Similarly, the plastic proportion decreased as the population increased for MDT, but the proportion remained within 1 for the population categories among the photoquadrat data. Similarly, proportions in item types were quite different between the methods, with the photoquadrat method finding greater quantities of plastic fragments, as opposed to the MDT methods which found greater quantities of food wrappers and tobacco products. While the exact cause in variation is difficult to pinpoint, the smaller plastic compositions seen in the photoquadrat analysis could be influenced by high quantities of unidentifiable objects, which were likely easier to identify in the field with the MDT methodology. Further, given the inconclusive test for agreeability between the photoquadrat coders, it is plausible that coders inaccurately identified items, suggesting that increasing QA/QC measures and improved training on item identification could minimize inaccuracies.

Although the photoquadrat and MDT surveys were performed in the same place and at the same time, the differences observed between the litter densities from the two methods are likely attributable to considerable differences in data collection and processing methods. In theory, the photoquadrat collection method allows for standardized data collection by limiting counts of items to a defined area with known dimensions (i.e., the 0.5 x 0.5 m quadrat). Efforts were made to avoid purposeful placement of the quadrat over debris, but the linear nature of the transects and the adherence to pathways, sidewalks, and gutters, meant that photoquadrat images were potentially concentrated along areas where debris tends to accumulate, as evidenced by Schuyler et al. (2021). As such, the item counts observed in each quadrat could be inflated. In comparison, the MDT method standardized the item count over the full 100 m^2 transect, the length of which was measured with a distance wheel while the width was visually approximated by the technician that was surveying. Given the field approximation, there is potential for over- or underestimating the transect width which could impact the final item count. Further, the 1-meter width of the MDT transect likely resulted in greater coverage of debris-free areas within the transect that could influence the resulting debris density.

That said, each method has advantages for certain applications. The photoquadrat method allows for quick retrieval of digital images, which can support broad surveying over large areas if needed or rapid surveying in unfavorable climates or conditions. Digital images are then brought back to a computer-based setting for processing. In contrast, the MDT method relies on field identification of items, which is facilitated by the ability to interact with items three-dimensions. Additionally, the MDT app is designed to record time and geospatial coordinates of every item. That said, in certain situations, field collection can induce physical and mental fatigue, particularly in extreme temperatures or high-density areas which were often reflective of the sites included in this study. Digital processing of images in a laboratory or office settings means that climate can be controlled, distractions can be limited, and more time can be devoted to visual observation of the defined area for precise item detection. Regardless of the method, repetitive tasks like detecting items in multiple images at a time or documented hundreds of items of litter can be subject to fatigue, which can impact reliability of both methods.

One significant advantage of recording debris items via the photoquadrat method compared to field identification is the ability to revisit the raw image dataset for further analyses or reassessment. Further, open access debris data such as that available through the MDT mobile application are limited to the individual recorder of each item in the space and time in which the record was generated. The photoquadrat data, on the other hand, allows for multiple observations and identification of items via multiple coders. Further, with image analysis technologies advancing rapidly, photoquadrat data taken today could be stored and observed later once technologies are available to do so reliably. Eventually, trained artificial intelligence will likely be employed for debris detection given the profound efficiency and reduction in error it provides compared to human efforts. As such, detection of environmental plastic pollution will likely rely on imagery, whether remotely sensed or directly collected in the field.

5.5 CONCLUSION

Reliable methods for measuring and monitoring land-based anthropogenic debris are needed for evaluating a, particularly in the built environment or urban settings. Given the challenges with conducting ground-level surveys in higher trafficked locations, rapid survey methods may provide an option for efficient collection of standardized debris data. Here, the ecological field methods were adapted to explore photoquadrat analysis as a potential field method for surveying anthropogenic debris in community-based settings.

Through this experiment, litter densities and proportions of items were determined, and a positive relationship between high populations and debris density was revealed, while low population densities had varying degrees of litter abundance. Like many other land-based surveys conducted mostly in natural environments or urban beaches, plastic fragments were the most recorded type of plastic debris observed. However, given the discrepancy in agreement established in processed photoquadrat data, and differences observed in comparison to other methods, further refinement of data collection, processing, and analysis and examination of the data may provide more conclusive evidence as to the reliability of the method. With rapid advancements in technology, detecting plastic debris in the built environment may be conducted through sophisticated imagery analysis.

REFERENCES

Andrady, A. L. (2011). "Microplastics in the marine environment." Mar Pollut Bull **62**.

Barnes, S. J. (2019). "Understanding plastics pollution: The role of economic development and technological research." Environmental Pollution **249**: 812-821.

Beijbom, O., P. J. Edmunds, C. Roelfsema, J. Smith, D. I. Kline, B. P. Neal, M. J. Dunlap, V. Moriarty, T.-Y. Fan, C.-J. Tan, S. Chan, T. Treibitz, A. Gamst, B. G. Mitchell and D. Kriegman (2015). "Towards Automated Annotation of Benthic Survey Images: Variability of Human Experts and Operational Modes of Automation." PLOS ONE **10**(7): e0130312.

Bergmann, M., L. Gutow and M. Klages (2015). Marine anthropogenic litter, Springer Nature.

Biermann, L., D. Clewley, V. Martinez-Vicente and K. Topouzelis (2020). "Finding Plastic Patches in Coastal Waters using Optical Satellite Data." Scientific Reports **10**(1): 5364.

Blettler, M. C. M., E. Abrial, F. R. Khan, N. Sivri and L. A. Espinola (2018). "Freshwater plastic pollution: Recognizing research biases and identifying knowledge gaps." Water Research **143**: 416-424.

Cattaneo, A., A. Nelson and T. McMenomy (2021). "Global mapping of urban–rural catchment areas reveals unequal access to services." Proceedings of the National Academy of Sciences **118**(2): e2011990118.

Cohen, J. (1960). "A Coefficient of Agreement for Nominal Scales." Educational and Psychological Measurement **20**(1): 37-46.

Davey, J. W., P. C. Gugiu and C. L. J. J. o. M. E. Coryn (2010). "Quantitative methods for estimating the reliability of qualitative data." **6**(13): 140-162.

de Ramos, B., M. V. Alencar, F. L. Rodrigues, A. L. d. F. Lacerda and M. C. Proietti (2021). "Spatio-temporal characterization of litter at a touristic sandy beach in South Brazil." Environmental Pollution **280**: 116927.

Duncan, E. M., A. Davies, A. Brooks, G. W. Chowdhury, B. J. Godley, J. Jambeck, T. Maddalene, I. Napper, S. E. Nelms, C. Rackstraw and H. Koldewey (2020). "Message in a bottle: Open source technology to track the movement of plastic pollution." PLOS ONE **15**(12): e0242459.

Fanini, L. and J. K. Lowry (2016). "Comparing methods used in estimating biodiversity on sandy beaches: Pitfall vs. quadrat sampling." Ecological Indicators **60**: 358-366.

Fok, L. and P. K. Cheung (2015). "Hong Kong at the Pearl River Estuary: A hotspot of microplastic pollution." Marine Pollution Bulletin **99**(1): 112-118.

Garaba, S. P., J. Aitken, B. Slat, H. M. Dierssen, L. Lebreton, O. Zielinski and J. Reisser (2018). "Sensing Ocean Plastics with an Airborne Hyperspectral Shortwave Infrared Imager." Environmental Science & Technology **52**(20): 11699-11707.

Geyer, R., J. R. Jambeck and K. L. Law (2017). "Production, use, and fate of all plastics ever made." Science Advances **3**(7): e1700782.

Giles, R. K., C. A. T. Nguyen, T. T. Y. Hồ, C. V. Nguyễn, N. T. Ngô and C. M. Rochman (2021). "Source-Specific Patterns of Marine Debris and Associated

Ecological Impacts in the Red River Estuary of Xuan Thuy National Park, Vietnam." **9**(162).

Goddijn-Murphy, L. and J. Dufaur (2018). "Proof of concept for a model of light reflectance of plastics floating on natural waters." Marine Pollution Bulletin **135**: 1145-1157.

Gollin, D., R. Jedwab and D. Vollrath (2016). "Urbanization with and without industrialization." Journal of Economic Growth **21**(1): 35-70.

Griffin, K. J., L. H. Hedge, M. González-Rivero, O. I. Hoegh-Guldberg and E. L. Johnston (2017). "An evaluation of semi-automated methods for collecting ecosystem-level data in temperate marine systems." Ecology and Evolution **7**(13): 4640-4650.

Gupta, K. (2011). Consumer responses to incentives to reduce plastic bag use: Evidence from a field experiment in urban India, SAMDEE.

Hafeez, S., M. Wong, S. Abbas, J. Nichol and C. Kwok (2018). Detection and Monitoring of Marine Pollution Using Remote Sensing Technologies.

Hardesty, B. D., T. Lawson, T. Velde, M. Lansdell and C. Wilcox (2017). "Estimating quantities and sources of marine debris at a continental scale." Frontiers in Ecology and the Environment **15**(1): 18-25.

Heo, N. W., S. H. Hong, G. M. Han, S. Hong, J. Lee, Y. K. Song, M. Jang and W. J. Shim (2013). "Distribution of small plastic debris in cross-section and high strandline on Heungnam beach, South Korea." Ocean Science Journal **48**(2): 225-233.

Hidalgo-Ruz, V. and M. Thiel (2013). "Distribution and abundance of small plastic debris on beaches in the SE Pacific (Chile): a study supported by a citizen science project." Mar Environ Res **87-88**: 12-18.

Hidalgo-Ruz, V. and M. Thiel (2015). The Contribution of Citizen Scientists to the Monitoring of Marine Litter. Marine Anthropogenic Litter. M. Bergmann, L. Gutow and M. Klages. Cham, Springer International Publishing**:** 429-447.

Hoellein, T. J. and C. M. Rochman (2021). "The "plastic cycle": a watershed-scale model of plastic pools and fluxes." **19**(3): 176-183.

Hung, C., N. Klasios, X. Zhu, M. Sedlak, R. Sutton and C. M. Rochman (2021). "Methods Matter: Methods for Sampling Microplastic and Other Anthropogenic Particles and Their Implications for Monitoring and Ecological Risk Assessment." Integrated Environmental Assessment and Management **17**(1): 282-291.

Jambeck, J. R. and K. Johnsen (2015). "Citizen-Based Litter and Marine Debris Data Collection and Mapping." Computing in Science & Engineering **17**(4): 20-26.

Jayasiri, H., C. Purushothaman and A. Vennila (2013). "Quantitative analysis of plastic debris on recreational beaches in Mumbai, India." Marine pollution bulletin **77**(1-2): 107-112.

Karasik, R., T. Vegh, Z. Diana, J. Bering, J. Caldas, A. Pickle, D. Rittschof and J. Virdin. (2020). 20 Years of Government Responses to the Global Plastic Pollution Problem: The Plastics Policy Inventory. . Durham, NC, Duke University.

Koelmans, A. A., M. Kooi, K. L. Law and E. van Sebille (2017). "All is not lost: deriving a top-down mass budget of plastic at sea." Environmental Research Letters **12**(11): 114028.

Krippendorff, K. (2018). Content analysis: An introduction to its methodology, Sage publications.

Kühn, S., E. L. B. Rebolledo and J. A. J. M. a. l. van Franeker (2015). "Deleterious effects of litter on marine life." 75-116.

Kuznets, S. (2019). Economic growth and income inequality, Routledge.

Landis, J. R. and G. G. Koch (1977). "The Measurement of Observer Agreement for Categorical Data." Biometrics **33**(1): 159-174.

Lebreton, L. C. M., J. van der Zwet, J.-W. Damsteeg, B. Slat, A. Andrady and J. Reisser (2017). "River plastic emissions to the world's oceans." Nature Communications **8**: 15611.

Lechthaler, S., K. Waldschläger, G. Stauch and H. Schüttrumpf (2020). "The Way of Macroplastic through the Environment." **7**(10): 73.

Maes, T., J. Barry, H. A. Leslie, A. D. Vethaak, E. E. M. Nicolaus, R. J. Law, B. P. Lyons, R. Martinez, B. Harley and J. E. Thain (2018). "Below the surface: Twenty-five years of seafloor litter monitoring in coastal seas of North West Europe (1992–2017)." Science of The Total Environment **630**: 790-798.

Marais, M., N. Armitage and C. Wise (2004). "The measurement and reduction of urban litter entering stormwater drainage systems: paper 1-quantifying the problem using the city of Cape Town as a case study." Water Sa **30**(4): 469-482.

Martínez-Vicente, V., R. J. Clark, P. Corradi, S. Aliani, M. Arias, M. Bochow, G. Bonnery, M. Cole, A. Cózar, R. Donnelly, F. Echevarría, F. Galgani, P. S. Garaba, L. Goddijn-Murphy, L. Lebreton, A. H. Leslie, K. P. Lindeque, N. Maximenko, F.-R. Martin-Lauzer, D. Moller, P. Murphy, L. Palombi, V. Raimondi, J. Reisser, L. Romero, G. H. S. Simis, S. Sterckx, C. R. Thompson, N. K. Topouzelis, E. van Sebille, M. J. Veiga and D. A. Vethaak (2019). "Measuring Marine Plastic Debris from Space: Initial Assessment of Observation Requirements." Remote Sensing **11**(20).

Maximenko, N., P. Corradi, K. L. Law, E. Van Sebille, S. P. Garaba, R. S. Lampitt, F. Galgani, V. Martinez-Vicente, L. Goddijn-Murphy, J. M. Veiga, R. C. Thompson, C. Maes, D. Moller, C. R. Löscher, A. M. Addamo, M. R. Lamson, L. R. Centurioni, N. R. Posth, R. Lumpkin, M. Vinci, A. M. Martins, C. D. Pieper, A. Isobe, G. Hanke, M. Edwards, I. P. Chubarenko, E. Rodriguez, S. Aliani, M. Arias, G. P. Asner, A. Brosich, J. T. Carlton, Y. Chao, A.-M. Cook, A. B. Cundy, T. S. Galloway, A. Giorgetti, G. J. Goni, Y. Guichoux, L. E. Haram, B. D. Hardesty, N. Holdsworth, L. Lebreton, H. A. Leslie, I. Macadam-Somer, T. Mace, M. Manuel, R. Marsh, E. Martinez, D. J. Mayor, M. Le Moigne, M. E. Molina Jack, M. C. Mowlem, R. W. Obbard, K. Pabortsava, B. Robberson, A.-E. Rotaru, G. M. Ruiz, M. T. Spedicato, M. Thiel, A. Turra and C. Wilcox (2019). "Toward the Integrated Marine Debris Observing System." **6**(447).

Meentemeyer, V. (1989). "Geographical perspectives of space, time, and scale." Landscape Ecology **3**(3): 163-173.

Meijer, L. J. J., T. van Emmerik, R. van der Ent, C. Schmidt and L. Lebreton (2021). "More than 1000 rivers account for 80% of global riverine plastic emissions into the ocean." Science Advances **7**(18): eaaz5803.

Ministry of Environment Forest and Climate Change (2021). Draft Notification. New Delhi, India, Controller of Publications.

Molloy, P. P., M. Evanson, A. C. Nellas, J. L. Rist, J. E. Marcus, H. J. Koldewey and A. C. J. Vincent (2013). "How much sampling does it take to detect trends in coral-reef habitat using photoquadrat surveys?" Aquatic Conservation: Marine and Freshwater Ecosystems **23**(6): 820-837.

Moss, E., Eidson, A., and Jambeck J. (2017). Sea of Opportunity: Supply Chain Investment Opportunities to Address Marine Plastic Pollution. New York, New York, Encourage Capital on behalf of Vulcan, Inc.

Nelms, S. E., C. Coombes, L. C. Foster, T. S. Galloway, B. J. Godley, P. K. Lindeque and M. J. Witt (2017). "Marine anthropogenic litter on British beaches: A 10-year nationwide assessment using citizen science data." Science of The Total Environment **579**: 1399-1409.

Nelms, S. E., E. M. Duncan, S. Patel, R. Badola, S. Bhola, S. Chakma, G. W. Chowdhury, B. J. Godley, A. B. Haque, J. A. Johnson, H. Khatoon, S. Kumar, I. E. Napper, M. N. H. Niloy, T. Akter, S. Badola, A. Dev, S. Rawat, D. Santillo, S. Sarker, E. Sharma and H. Koldewey (2021). "Riverine plastic pollution from fisheries: Insights from the Ganges River system." Science of The Total Environment **756**: 143305.

Nelms, S. E., L. Eyles, B. J. Godley, P. B. Richardson, H. Selley, J.-L. Solandt and M. J. Witt (2020). "Investigating the distribution and regional occurrence of anthropogenic litter in English marine protected areas using 25 years of citizen-science beach clean data." Environmental Pollution **263**: 114365.

O'Connor, C. and H. Joffe (2020). "Intercoder Reliability in Qualitative Research: Debates and Practical Guidelines." International Journal of Qualitative Methods **19**: 1609406919899220.

Parravicini, V., C. Morri, G. Ciribilli, M. Montefalcone, G. Albertelli and C. N. Bianchi (2009). "Size matters more than method: Visual quadrats vs photography in measuring human impact on Mediterranean rocky reef communities." Estuarine, Coastal and Shelf Science **81**(3): 359-367.

Plastics Europe (2020). "Plastics the Facts - 2020."

Rech, S., V. Macaya-Caquilpán, J. Pantoja, M. Rivadeneira, C. Campodónico and M. Thiel (2015). "Sampling of riverine litter with citizen scientists - findings and recommendations." Environmental Monitoring & Assessment **187**(6): 1-18.

Rochman, C. M., A.-M. Cook and A. A. Koelmans (2016). "Plastic debris and policy: Using current scientific understanding to invoke positive change." **35**(7): 1617-1626.

Ryan, P. G., C. J. Moore, J. A. Franeker and C. L. Moloney (2009). "Monitoring the abundance of plastic debris in the marine environment." Philos Trans R Soc Lond B **364**.

Ryberg, M. W., M. Z. Hauschild, F. Wang, S. Averous-Monnery and A. Laurent (2019). "Global environmental losses of plastics across their value chains." Resources, Conservation & Recycling **151**.

Schuyler, Q., B. Hardesty, T. J. Lawson, K. Opie and C. Wilcox (2018). Economic incentives reduce plastic inputs to the ocean.

Schuyler, Q., C. Wilcox, T. J. Lawson, R. R. M. K. P. Ranatunga, C.-S. Hu, Global Plastics Project Partners and B. D. Hardesty (2021). "Human Population Density is a Poor Predictor of Debris in the Environment." **9**(133).

Smith, M., D. C. Love, C. M. Rochman and R. A. Neff (2018). "Microplastics in seafood and the implications for human health." Current environmental health reports **5**(3): 375-386.

Sulochanan, B., S. Veena, L. Ratheesh, S. Padua, P. Rohit, P. Kaladharan and V. Kripa (2019). "Temporal and spatial variability of beach litter in Mangaluru, India." Marine Pollution Bulletin **149**: 110541.

Tasseron, P., T. van Emmerik, J. Peller, L. Schreyeres and L. Biermann ([Preprint]). "Advancing floating macroplastic detection from space using hyperspectral imagery." Earth ArXiv.

Thompson, R. C., Y. Olsen, R. P. Mitchell, A. Davis, S. J. Rowland and A. W. John (2004). "Lost at sea: where is all the plastic?" Science **304**.

Topouzelis, K., A. Papakonstantinou and S. P. Garaba (2019). "Detection of floating plastics from satellite and unmanned aerial systems (Plastic Litter Project 2018)." International Journal of Applied Earth Observation and Geoinformation **79**: 175-183.

UNEP (2018a). "Legal Limits on Single-Use Plastics and Microplastics: A Global Review of National Laws and Regulations."

UNEP (2018b). Single-Use Plastics: A Roadmap for Sustainability.

van Emmerik, T., T.-C. Kieu-Le, M. Loozen, K. van Oeveren, E. Strady, X.-T. Bui, M. Egger, J. Gasperi, L. Lebreton, P.-D. Nguyen, A. Schwarz, B. Slat and B. Tassin (2018). "A Methodology to Characterize Riverine Macroplastic Emission Into the Ocean." **5**(372).

van Emmerik, T., M. Loozen, K. van Oeveren, F. Buschman and G. Prinsen (2019a). "Riverine plastic emission from Jakarta into the ocean." Environmental Research Letters **14**(8): 084033.

van Emmerik, T., T. Romain, C. Caroline van, A. Soline, T. Robin, T. Bruno and G. Johnny (2019b). "Seine Plastic Debris Transport Tenfolded During Increased River Discharge." Frontiers in Marine Science **6**.

van Emmerik, T. and A. Schwarz (2020). "Plastic debris in rivers." WIREs Water **7**(1): e1398.

van Emmerik, T., J. Seibert, B. Strobl, S. Etter, T. den Oudendammer, M. Rutten, M. S. bin Ab Razak and I. van Meerveld (2020). "Crowd-Based Observations of Riverine Macroplastic Pollution." **8**(298).

van Emmerik, T., E. Strady, T.-C. Kieu-Le, L. Nguyen and N. Gratiot (2019). "Seasonality of riverine macroplastic transport." Scientific Reports **9**(1): 13549.

Vriend, P., C. van Calcar, M. Kooi, H. Landman, R. Pikaar and T. van Emmerik (2020). "Rapid Assessment of Floating Macroplastic Transport in the Rhine." **7**(10).

Weideman, E. A., V. Perold, G. Arnold and P. G. Ryan (2020). "Quantifying changes in litter loads in urban stormwater run-off from Cape Town, South Africa, over the last two decades." Science of the Total Environment **724**.

Wyles, K. J., S. Pahl, M. Holland and R. C. Thompson (2016). "Can Beach Cleans Do More Than Clean-Up Litter? Comparing Beach Cleans to Other Coastal Activities." Environment and Behavior **49**(5): 509-535.

Youngblood, K., A. L. Brooks, G. Verma, A. Singh, T. Zakir, M. N. Sultana, N. Das, B. Dubey and J. R. Jambeck (2021). "The characterization of Litter in Communities along the Ganges River." Manuscript submitted for publication.

CHAPTER 6

CONCLUSIONS

The objective of this dissertation was to investigate patterns in plastic waste management at various scales ranging from the global plastic scrap trade to ground-level measurement of the presence, abundance, and density of litter in communities. To do so, international trade data, national waste management figures, and field-documented debris data were collected, processed, and examined with quantitative modeling and statistical tests of comparisons. Though the context varied between the chapters, each analysis provided new information and perspectives around each topic, and given the various scales (global, regional, basin, and local), patterns and themes emerged across the topics.

Chapter Two provided an update to the global plastic scrap trade as of 2019 and a comparison of trade characteristics by country, region, and income between the years immediately prior to and after the Chinese ban on waste imports, which has had resounding impacts felt across the world since it was implemented in 2018. Early effects of the ban provided by this analysis suggest countries that the ban has been enacted at a nearly 100% scenario, wherein minimal and gradually decreasing quantities of plastic scrap have been imported since 2018. Conversely, 15.2 million metric tons of plastic scrap have already been displaced by the ban such that Malaysia, Vietnam, and Indonesia have seen growth in imports, while high income countries in North America and Europe have been distributing plastic scrap exports across more importers. Though Sub-Saharan Africa (SSF) does not appear to be participating in the global plastic scrap trade in large quantities, some low income (LIC) countries in the region have seen significantly large relative increases in imports. With potential impacts resulting from the recent amendments to the Basel Convention that now regulates international trade of plastic scrap, continued monitoring of global trade trends will support identification of early trends and anticipation of possible environmental impacts that may result. Given the richness of the trade data, future work may build deeper understanding of this complex globalized system, through examining the data from different perspectives that reflect populations, openness to trade, and the influence of diverse national geographies, cultures, and governments in the context of global inequality.

Chapter Three examined patterns of plastic waste management in the Latin America and Caribbean region. Compared to other regions, LCN is relatively neutral in terms of plastic waste generation and management, and so it has garnished little attention compared to other regions that have seemingly greater and more urgent plastic waste management challenges. The LCN region is diverse with widely varying geographies, cultures, populations, and economies that lead to differences in how plastic is consumed and managed across LCN countries and subregions. To address the lack of focused research in the region, LCN was isolated and considered based on sub-regions, income classification, and to better understand within-region patterns, rather than that between-regions. This study found that LCN generated 24.2 Mt of plastic waste in 2020, and 7.15 Mt of this were mismanaged within the region. Further, an estimated 793,00 metric tons may have entered the ocean from land-based sources. While upper-middle income countries in the region play a key role in terms of quantities of waste generation and consequent pollution, high income countries in the Caribbean appear to have substantially high levels of per capita mismanaged plastic waste that contribute to a disproportionate amount of plastic waste given the small populations, which is especially concerning given the area to coastline ratio in these countries.

Chapter Four and Chapter Five tied focused on the Ganges River Basin. Chapter Four compared estimates of basin-wide quantities of plastic litter using empirical extrapolation and modeling procedures. The relative agreement between the empirically derived and modeled estimate resulted in a range of 39,200 to 392,000 metric tons of plastic waste being littered in the basin in 2019. Building on the littered plastic waste quantities, the modeled estimate found that 12.2 million metric tons (MMT) of plastic waste were generated throughout the basin and a corresponding 5.27 MMT of plastic waste were mismanaged. Chapters Three and Chapter Four highlight the need for empirical calibration of modeled estimates.

A rapid field method for collection of empirical data on land-based anthropogenic debris was presented in Chapter Five, finding that photoquadrat analysis may provide utility in the context of plastic pollution. Across ten sites and two field expeditions, 9,520 debris items were identified through the method, of which plastic waste the most prevalent material. Further, the most frequently documented item type was plastic fragments, which aligned well with similar published research. Although the preliminary reliability of the method was inconclusive based on interrater responsibility and comparison to other field methods conducted in parallel to the photoquadrat analysis, the experimental approach presented may contribute to development of practical and rapid field approaches for empirical data collection of land-based anthropogenic debris. Further, knowledge around land-based debris in communities, where waste is inherently managed, can inform community-specific profiles of plastic debris quantities and composition, which can inform local efforts to reduce incidence of environmental plastic pollution and empirically monitor the effectiveness of tools used to do so.

While the chapters presented here are loosely connected to each other within the overarching topic of plastic waste management and pollution, they are woven together by the influence that landscape scales play in each chapter in relation to estimating and evaluating the pressing global environmental challenge of plastic material management in these contexts. Scale is comprised of resolution as an observational unit and extent as the size of a study area, both of which must be defined by the researcher and is often a choice between maximizing species richness at small scales or optimizing data a large scale (Sheppard and McMaster 2004, Caro and Girling 2010). Scale is associative of species richness, latitude, altitude, and productivity (Caro and Girling 2010). In conservation fields, it is widely agreed that effects of habitat disturbance should not be evaluated at a single scale, and it stands to gain that it is beneficial to assess environmental issues and concerns related to conservation, such as plastic pollution, at multiple scales as well.

Patterns at the global or regional scale can help to identify geographic areas that need attention in a particular biological, conservation, political, etc., context (Vanewright et al. 1991). However, there is no simple rule for selecting the "proper" scale for attention, which results in optimizing for the application and balancing tradeoffs that may occur at various scales. As spatial components are more fully integrated into research agendas in environmental sciences, problems associated with spatial scale are more likely to be encountered (Meentemeyer 1989). At coarser scales, the overarching constraining variables are more easily defined, however, they can lack the detail and complexity associated with spatial scale hierarchies in nested systems like waste. This may be especially true for issues of plastic waste management and pollution where the problem with scale may occur. For example, adequately surveying ground level abundance and distribution of plastic litter at a given place and time can require extraordinary effort, which can conversely result in map scales approximating the spatial scale definition rather than defining the scale as it relates to the phenomenon of leaked plastic waste.

In environmental and ecological analyses, homogeneity at large-scales can be reasonably considered but potentially misleading as other scales may reveal wide ranging diversity and increased heterogeneity (Meentemeyer and Box 1987). Windsor et al. (2019) suggests that assessments of heterogeneity related to plastic pollution are needed at a range of spatial scales, from local patch dynamics at very fine scales, to comparisons between entire habitats and ecosystems. In conservation and ecology disciplines, trends and associations between taxa can be apparent at one scale but not detected in another, and some studies show that congruency between different metrics can be differentially affected by scale (Caro and Girling 2010). Likely, the abundance and distribution of plastics are no different, where at once they may display a particular pattern of distribution at a fine scale like that seen in Chapter Five, but as resolution decreases, local influences are dampened and large-scale influences like geopolitics, population density, or land use may be accentuated.

Ambiguity and chaos are inherent in ecosystems, but hierarchy theory posits that functional parameters, frequencies, and rates of activities can be more useful for understanding and defining integrated scales than physical structures (Allen and Starr 2017). The modern waste management system is inherently hierarchical, and every participant confronts a series of interrelated problems related to consumption and disposal of waste which occurs on different temporal and spatial scales. In the context of waste management, planning and modeling often involve global, regional, and even settlement-level demands; however, at the household scale, walking time and distance to waste infrastructure, or even products and packaging, can be prominent concerns, thereby replacing group and regional aggregation variables with the individual or family unit. Further, mismanaged waste may provide a form of accelerated feedback in the form of signals that we receive from our surroundings that we both derive meaning from and assign meaning to.

In the context of the international trade of plastic scrap, globally traded commodities experience transport as a function of both space and place. The transboundary movement of a unit of plastic scrap is at once absolute (i.e., it is transported over Euclidian distances from one physical space to another) and relative (i.e., a high-income nation exports it to a low-income nation), resulting in both physical trends and conceptual structures that may be evaluated at regional national and regional boundaries, given the limited national scale provided in publicly available data. As a result, geographical and national level economic inferences can be made, but at the global scale little can be said about the realities of what is happening on the ground when waste from one location is exported to another for management. By designating spaces for waste, society is freed of unwanted, useless, and valueless materials. In this sense, spatial analysis might inform the physical movement of waste in space, but when relative meaning is considered the transport of plastic scrap can take on more complexity in the global scale. What may be the simple movement of waste from a recycle bin in the USA could lead to transport of scrap over considerable distances to small informal operation in Southeast Asia. However, these physical distances are minor in comparison to the geopolitical and socioeconomic distances between the plastic scrap source and destination, which may not be sufficiently captured by the broad scale and coarse evaluations presented here.

Despite spanning vastly different scales, Chapters Two through Four are all arguably large-scale approaches to measuring and evaluating plastic waste management at global, regional, and basin scales. In both Chapters Two and Three, granular influences such as physical geography or spatial population distribution are relatively unaccounted for, and the complex influences of culture and socioeconomic are hardly realized. Waste characteristics aggregated at the national level drove these analyses, which cannot detect small scale differences such as urbanization, income, culture, and socioeconomic effects on the consumption of plastic good and corresponding waste generation and disposal. Regional scale assessments like that in Chapter Three can be helpful for assessing countries that are geospatially close or characteristically similar. However, da Silva C de Oliveira (2017), argues that using the word 'region' as a catch all term for nations that are similar or grouped for the sake of reducing complexity may undermine the relevance of regional scale analyses particularly in terms of regional policies through which regions can act as a designated space for political projects and action. In Chapter Four, empirically derived quantities of plastic litter begin to confirm estimates generated by common modeling approaches, however, singular, basin-wide values like the estimated 216,000 MT of plastic litter in the Ganges Basin serve to highlight the urgency of the problem. Only in Chapter Five, does the variation between communities begin to emerge, highlighting the differences in plastic waste management in urban and rural communities despite likely having similar access and consumption to plastic products. Yet, still individual behavior and household decisions are still undetected at this scale.

Plastic pollution is a rapidly accumulating environmental threat impacting wildlife, ecosystem, and human health. Over 8.3 billion metric tons of plastic waste have been generated since 1950 (Geyer et al. 2017), and between 4.7 and 15 million metric tons enter the ocean annually (Jambeck et al. 2015, Forrest et al. 2019). While the world collectively generated 2.01 billion MT of MSW waste in 2016, where and how waste is generated can vary drastically from place to place within the global system. Given the transboundary nature of plastic pollution, the global state of the plastic waste management is complex and interconnected. However, waste is inherently managed at the local, community scale. Given the multi-scale impacts of plastic pollution, multi-scale, integrated policy approaches are likely to be some of the most effective (Vince and Hardesty 2017).

As the world moves toward increased urbanization and development in the coming decades, larger populations will be able to consume more, thereby generating more waste, and decoupling waste from economic growth. While this can be seen as a victory in terms of large populations gaining wider access to goods and services, there are vast tradeoffs associated with rapid economic growth that is unmatched with waste management infrastructure, resulting in higher levels of mismanaged waste (Jambeck et al. 2015, Lebreton and Andrady 2019). Research in the field of plastic pollution and marine debris is in its relative infancy, and its impacts on people at the global, regional, and local scale are not yet fully understand (GESAMP 2015). Many studies focusing on microplastic abundance and animal ingestion in the marine environment have been conducted at small scales, but these are often disparate, lack scalability (Beaumont et al. 2019), and do not yet address large data gaps, particularly relating to freshwater macroplastics (Blettler et al. 2018). Evaluation of plastic pollution is largely lacking from the landscape ecology perspective and only a few analyses exploring the effects of marine pollution on ecosystem services (GESAMP 2015, Beaumont et al. 2019).

That said, deep understanding of the many factors that influence the presence, abundance, source, fate, and impacts of plastic pollution across multiple scales is needed. In this sense, global assessments of plastic waste are useful for identifying regional hot spots or geographies of interest, such as effects of ocean current on the transport of plastic marine debris to certain locations (Cózar et al. 2017), the global abundance and mass of floating plastic (Eriksen et al. 2014), or regions or countries that exhibit high levels of mismanaged plastic waste (Jambeck et al. 2015). Regional and national trends can be useful for identifying trends in economic development, waste management infrastructure, and policies. At finer scales, ground level surveys can inform decision making with empirical data, and influences of behaviors, culture, local geography, and individual waste practices can be more easily evaluated. Future work should examine multi-scale integration for improved understanding of the complexities driving modern plastic production and consumption to aid in development of both technical and social strategies for reducing plastic waste generation, advancing waste management, and preventing plastic waste from reaching the environment.

REFERENCES

Allen, T. F. and T. B. Starr (2017). Hierarchy, University of Chicago Press.

Beaumont, N. J., M. Aanesen, M. C. Austen, T. Börger, J. R. Clark, M. Cole, T. Hooper, P. K. Lindeque, C. Pascoe and K. J. Wyles (2019). "Global ecological, social and economic impacts of marine plastic." Marine Pollution Bulletin **142**: 189-195.

Blettler, M. C. M., E. Abrial, F. R. Khan, N. Sivri and L. A. Espinola (2018). "Freshwater plastic pollution: Recognizing research biases and identifying knowledge gaps." Water Research **143**: 416-424.

Bodansky, D. (2010). The Art and Craft of International Environmental Law. Cambridge, Mass, Harvard University Press.

Brooks, A. L., S. Wang and J. R. Jambeck (2018). "The Chinese import ban and its impact on global plastic waste trade." Science Advances **4**(6).

Caro, T. and S. Girling (2010). "Conservation by Proxy: Indicator, Umbrella, Keystone, Flagship, and Other Surrogate Species." Bibliovault OAI Repository, the University of Chicago Press.

Cózar, A., E. Martí, C. M. Duarte, J. García-de-Lomas, E. van Sebille, T. J. Ballatore, V. M. Eguíluz, J. I. González-Gordillo, M. L. Pedrotti, F. Echevarría, R. Troublè and X. Irigoien (2017). "The Arctic Ocean as a dead end for floating plastics in the North Atlantic branch of the Thermohaline Circulation." **3**(4).

da Silva C de Oliveira, J. (2017). "The Place of the Region in IR." Contexto Internacional **39**(1): 97-115.

Eriksen, M., L. C. M. Lebreton, H. S. Carson, M. Thiel, C. J. Moore, J. C. Borerro, F. Galgani, P. G. Ryan and J. Reisser (2014). "Plastic Pollution in the World's Oceans: More than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea." PLOS ONE **9**(12): e111913.

Forrest, A., L. Giacovazzi, S. Dunlop, J. Reisser, D. Tickler, A. Jamieson and J. J. Meeuwig (2019). "Eliminating Plastic Pollution: How a Voluntary Contribution From Industry Will Drive the Circular Plastics Economy." Frontiers in Marine Science **6**(627).

GESAMP (2015). Sources, fate and effects of microplastics in the marine environment: a global assessment. Reports and Studies 90. London: IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection.

Geyer, R., J. R. Jambeck and K. L. Law (2017). "Production, use, and fate of all plastics ever made." Science Advances **3**(7): e1700782.

Gold, M., Mika, K., Horowitz, C., Herzog, M., & Leitner, L. (2013). Stemming the Tide of Plastic Marine Litter: A Global Action Agenda, UCLA: Emmett Institute on Climate Change and the Environment.

Jambeck, J., B. D. Hardesty, A. L. Brooks, T. Friend, K. Teleki, J. Fabres, Y. Beaudoin, A. Bamba, J. Francis, A. J. Ribbink, T. Baleta, H. Bouwman, J. Knox and C. Wilcox (2017). "Challenges and emerging solutions to the land-based plastic waste issue in Africa." Marine Policy.

Jambeck, J. R., R. Geyer, C. Wilcox, T. R. Siegler, M. Perryman and A. Andrady (2015). "Plastic waste inputs from land into the ocean." Science **347**.

Jelinski, D. E. and J. Wu (1996). "The modifiable areal unit problem and implications for landscape ecology." Landscape Ecology **11**(3): 129-140.

Karasik, R., T. Vegh, Z. Diana, J. Bering, J. Caldas, A. Pickle, D. Rittschof and J. Virdin. (2020). 20 Years of Government Responses to the Global Plastic Pollution Problem: The Plastics Policy Inventory. . Durham, NC, Duke University.

Kaza, S., L. Yao, P. Bhada-Tata and F. Van Woerden (2018). What a Waste 2.0: A Global Snapshot of Solid Waste Management to 2050. Urban Development Series. Washington, DC, World Bank.

Law, K. L., Starr, N., Siegler, T., Jambeck, J., Mallos, N., Leonard, G. (2020). "The United States' contribution of plastic waste to land and ocean." Science Advances **6**(44).

Lebreton, L. and A. Andrady (2019). "Future scenarios of global plastic waste generation and disposal." Palgrave Communications **5**(1).

Meentemeyer, V. (1989). "Geographical perspectives of space, time, and scale." Landscape Ecology **3**(3): 163-173.

Meentemeyer, V. and E. O. Box (1987). Scale Effects in Landscape Studies. Landscape Heterogeneity and Disturbance. M. G. Turner. New York, NY, Springer New York**:** 15-34.

Openshaw, S. (1981). "The modifiable areal unit problem." Quantitative geography: A British view: 60-69.

Sheppard, E. S. and R. B. McMaster (2004). Scale and geographic inquiry. [electronic resource] : nature, society, and method, Blackwell Pub.

Stöfen-O'Brien, A., Naji, A., Brooks, A.L., . Jambeck, J.R, Khan, F. (In Preparation). "Plastic marine debris in the Gulf Cooperation Council region: challenges, opportunities and recommendations ".

Tramoy, R., J. Gasperi, R. Dris, L. Colasse, C. Fisson, S. Sananes, V. Rocher and B. Tassin (2019). "Assessment of the Plastic Inputs From the Seine Basin to the Sea Using Statistical and Field Approaches." **6**(151).

Tuan, Y.-F. (2001). Space and Place : the perspective of experience. Minneapolis, University of Minnesota Press.

Vanewright, R. I., C. J. Humphries and P. H. Williams (1991). WHAT TO PROTECT - SYSTEMATICS AND THE AGONY OF CHOICE. **55:** 235-254.

Vince, J. and B. D. Hardesty (2017). "Plastic pollution challenges in marine and coastal environments: from local to global governance." Restoration Ecology **25**(1): 123-128.

Windsor, F. M., I. Durance, A. A. Horton, R. C. Thompson, C. R. Tyler and S. J. Ormerod (2019). "A catchment-scale perspective of plastic pollution." Global Change Biology **25**(4): 1207-1221.

Savino, A., G. Solorzano, C. Quispe and M. J. U. N. E. P. Correal, Argentina (2018). "Waste management outlook for Latin America and the Caribbean." **9**.