INTO THE SMOKE:
A RESEARCH ON HOUSEHOLD AIR POLLUTION AND CLIMATE IMPACTS OF BIOMASS COOKSTOVES IN SENEGAL

Tesis doctoral

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Resumen

Más de la mitad de la población mundial depende de los combustibles sólidos (leña, carbón vegetal, residuos agrícolas y animales) para cocinar. Estos combustibles son quemados en el interior de los hogares en cocinas poco eficientes, convirtiendo los hogares en entornos muy peligrosos para la salud.

Cada año, la contaminación del aire interior provoca la muerte prematura de 4,3 millones de personas, un número mayor que la suma de las muertes producidas por el paludismo, el sida y la tuberculosis. Las mujeres, y los niños que a menudo las acompañan, inhalan estas sustancias durante horas, ya que ellas son las responsables de preparar la comida para sus familias. Además, ellas son las principales responsables de recolectar el combustible para cocinar, poniendo en peligro su seguridad y reduciendo de manera importante su tiempo disponible para la educación, el descanso y las actividades generadoras de ingresos.

Pero el problema no se queda en casa. La quema de combustibles sólidos a nivel doméstico supone el 12% de las emisiones mundiales de partículas finas (PM$_{2.5}$) y el 25% de las emisiones mundiales de carbono negro (BC, por sus siglas en inglés) o carbono elemental (EC), que se considera el segundo mayor contribuyente al calentamiento global, después del CO$_2$.

Afortunadamente, la comunidad mundial está intensificando sus esfuerzos para garantizar el acceso a una energía limpia y segura para cocinar en los hogares, y se considera ya un aspecto esencial para poder avanzar en los objetivos planteados en la Agenda 2030 para el Desarrollo Sostenible.

Existe una amplia gama de soluciones de cocinado limpios y eficientes, cada una de ellas con una determinada eficiencia, coste, ventajas y desafíos en cuanto a la satisfacción de las necesidades de las personas usuarias. Aunque estas soluciones de cocinado puedan parecer simples, la naturaleza estocástica de la combustión de la biomasa, combinada con las variaciones naturales durante su uso diario, la amplia variedad de tecnologías y combustibles disponibles, dificultan la evaluación de las mismas, especialmente de algunos parámetros o contaminantes, como el BC.

Hasta la fecha, se han realizado pocos estudios para cuantificar las emisiones de BC de la quema de biomasa a nivel residencial, en parte debido a que los métodos analíticos disponibles son relativamente costosos y complejos. Además, el efecto de calentamiento del BC es bastante incierto, puesto que se emite siempre junto con el carbono orgánico (OC), tradicionalmente asociado a efectos de enfriamiento.

Por otra parte, la necesidad de estudios para cuantificar los impactos relacionados con la quema de combustibles para cocinar no es homogénea en todo el mundo.
Tradicionalmente, la mayoría de los estudios se han centrado en Asia, y muy pocos estudios se han llevado a cabo en otras regiones del mundo, como en África subsahariana.

En este contexto, esta tesis pretende contribuir al conocimiento existente sobre los impactos en el clima y la calidad del aire interior producidos por el cocinado con biomasa a nivel residencial, con especial énfasis en las emisiones de BC y OC, y centrándose en la región de África Occidental, donde reside un tercio de la población total de África subsahariana y donde el 75% de la gente todavía utiliza madera y carbón para cocinar.

El Capítulo 3 presenta una intercomparación de tres métodos analíticos para estimar las emisiones de BC procedentes de los sistemas de cocinado. Dos métodos relativamente baratos y fáciles de usar, un reflectómetro (Difussion Systems Ltd.) y un sistema de análisis a través de teléfono móvil (Nexleaf Analytics), fueron validados con un dispositivo más preciso, el Analizador OC-EC (Sunset Laboratory).

Los resultados muestran una buena concordancia entre las dos técnicas de bajo coste, lo que supone una importante ayuda para superar las actuales barreras metodológicas para la determinación de emisiones de BC procedentes de cocinas en lugares con recursos limitados. Asimismo, la facilidad de uso del sistema basado en el teléfono móvil puede facilitar la participación de las usuarias de las cocinas en los procesos de recopilación de datos, aumentando así la cantidad y la frecuencia de la recolección de información.

En el capítulo 4 se presenta una cuantificación de emisiones de aerosoles carbonosos (EC y OC) de cuatro tipos de cocina ampliamente utilizados en Senegal y países vecinos: i) Tres piedras; ii) cocina tipo rocket; iii) cocina mejorada básica de cerámica; iv) gasificador. Para todas las cocinas analizadas, los Factores de Emisión (FE) (g/MJ) de EC y OC se vieron significativamente afectados por la especie forestal, desatando la necesidad de tener en cuenta el tipo de combustible al presentar los FE de los sistemas de cocinado. En cuanto al efecto del tipo de cocina, la cocina rocket dio lugar a los FE de EC más altos, mientras que las otras tres cocinas mostraron valores muy similares. En lo que se refiere a emisiones totales por test, el gasificador mostró los valores más bajos, y la rocket, los más elevados.

Asimismo, la cocina tradicional y la rocket fueron analizadas en una aldea rural de Senegal, con el fin de obtener datos de emisiones de EC y OC, y entender cuán representativos son los estudios estandarizados de laboratorio de las prácticas de cocinado reales de África Occidental. Al igual que en los resultados de laboratorio, la estufa tipo rocket mostró un FE y emisiones totales de EC más alto que la cocina tradicional. Además, el FE y emisiones totales de OC fueron más bajos con la cocina rocket, por lo que las emisiones de aerosoles carbonosos de este tipo de cocinas
producirían un efecto neto de calentamiento positivo, en comparación con la cocina tradicional.

El capítulo 5 presenta un análisis de concentración de PM$_{2.5}$, partículas ultrafinas, BC y monóxido de carbono (CO) en el interior de las viviendas en la misma aldea rural de Senegal. Los resultados confirman que la contaminación del aire en los hogares en esta zona se deben principalmente a las actividades de cocinado. Además, la instalación de la cocina tipo rocket contribuyó a una reducción significativa PM$_{2.5}$, partículas ultrafinas y CO, pero incrementó la concentración de BC, lo cual es coherente con los resultados de FE presentados en el capítulo 4.

Los resultados también evidencian que, además de un cambio en la fuente de emisión (es decir, la cocina y/o el combustible), las estrategias enfocadas en la mejora de la calidad del aire doméstico deben enfocarse en las prácticas de ventilación y el uso de los materiales de construcción más adecuados.

Según los resultados presentados en los capítulos 4 y 5, la implementación de estufas tipo rocket tendría un efecto positivo con respecto a la reducción de la contaminación interior, pero efectos más inciertos sobre el clima. Esto demuestra que los efectos en el clima y en salud de las soluciones de cocinado no siempre están alineados y destaca que ambas dimensiones deben ser consideradas en las decisiones tecnológicas. No obstante, en el estudio no se incluyen otros contaminantes emitidos y factores importantes, como la renovabilidad de la biomasa empleada para cocinar. Por lo tanto, los resultados presentados no significan que el uso de cocinas tipo rocket no tenga beneficios medioambientales, ya que el panorama completo es mucho más complicado e incierto.

La información presentada en esta tesis ayuda a tener una imagen más precisa de los impactos producidos por la quema de biomasa para cocinar a nivel residencial en la región de África Occidental. Además, puede ser útil para los implementadores y tomadores de decisiones de la región a la hora de seleccionar las soluciones tecnológicas más apropiadas en función de los impactos esperados, y para que los diseñadores optimicen los co-beneficios para el medio ambiente y la salud de las soluciones de cocinado.

Se necesitan más estudios de laboratorio y de terreno que analicen diferentes sistemas de cocinado, cubriendo otras áreas de África Occidental y que también estudien el efecto de la estacionalidad sobre las emisiones, con el fin de obtener datos más representativos para su uso en los inventarios de emisiones y modelos climáticos.
Abstract

Over half of the world’s population still rely on solid fuels, like fuelwood, charcoal, coal, agricultural residues and animal wastes for cooking. These fuels are burnt inside homes in inefficient stoves, turning the home into one of the most health damaging environments.

Each year, 4.3 million people prematurely die because of the air pollution inside their homes, a number above the deaths produced by malaria, aids and tuberculosis together. Women, and children who are often with them, inhale the smoke for hours, as they are responsible for preparing meals for their families. Moreover, they are the primary responsible for gathering fuel for cooking, facing safety risks and suffering significant constraints on their available time for education, rest and activities for income generation.

And the problem does not stay at home. Residential solid fuel burning accounts for up to 12% of global ambient fine particulate matter (PM$_{2.5}$) emissions and 25% of global emissions of black carbon (BC) or elemental carbon (EC), which is considered the second biggest climate forcer after CO$_2$.

Fortunately, the global community is intensifying its efforts to expand and accelerate access to clean household energy for cooking, since addressing this issue is essential to make progress toward the Sustainable Development Agenda.

A broad range of stove and fuel solutions have been developed, each one with differentiated efficiencies, costs, distribution models and challenges in term of meeting user needs. While household’ stoves often look simple in appearance, the stochastic nature of biomass combustion, combined with natural variations in daily use, the wide variety of fuel-stoves combinations available and their scattered location, make the stove evaluation challenging, especially for some parameters or pollutants, as the BC.

To date, quantitative data of BC emissions estimates from residential biomass burning for cooking is scarce, in part due to the relatively costly and complex analytical methods available. Moreover, the warming effect of BC is highly uncertain, as it is always co-emitted with organic carbon (OC), which has been traditionally linked to cooling effects in the atmosphere.

On the other hand, the need of studies to quantify coking-related impacts is not homogeneous across the globe. Traditionally, most studies have been focused in Asia with far few studies in other regions of the world.
In this context, this thesis aims to contribute to the existing knowledge about climate and indoor air quality impacts of biomass cookstoves, with a particular emphasis on BC and OC emissions, and focusing in the West African region.

Chapter 3 presents an intercomparison of analytical methods to estimate BC emissions from cookstoves. The performance of two relatively low-cost and easy-to-use methods, a cell-phone based system and smoke stain reflectometer, was validated against a more accurate device, the Sunset OCEC Analyzer (TOT).

A good agreement was observed between the two low cost techniques and the reference system for the aerosol types and concentrations assessed. This validation helps to overcome current methodological barriers to determine BC emissions from cookstoves in resource-constrained locations. Moreover, the easy use of the cell-phone based system may allow engaging cookstove users in the data collection process, increasing the amount and frequency of data collection and helping to raise public awareness about environmental and health issues related to cookstoves.

Chapter 4 presents a carbonaceous aerosols (EC and OC) emissions characterization of a set of stove-fuel combinations widely used in Senegal and neighbouring countries: i) Three-stone; ii) rocket stove; iii) Basic improved ceramic stove); and iv) natural draft gasifier.

For all the stoves tested, EC and OC Emission Factors (EFs) (g/MJ) were significantly affected by the wood species, highlighting the need of taking into account the fuel type when reporting cookstove carbonaceous aerosol EFs. The highest average EC EF per WBT was found with the rocket stove, while EC EF’s were fairly uniform across the other two improved cookstoves and the three stones. Considering EC and OC total emissions per test (Water Boiling Test), the gasifier induced to the smallest total emissions and the rocket stoves to the highest.

To understand how real cooking practices in this region influenced EC and OC emissions and how representative are the standardized studies of actual cooking practices in the region, the three stones and the rocket stove were tested in a rural village of Senegal. Similarly to laboratory results, rocket stove showed the highest EC EF and EC total emissions per cooking task. Moreover, OC emissions were reduced with the rocket stove, so carbonaceous emissions of this stove type would produce a net positive warming effect when compared the traditional stove.

Chapter 5 presents a real-world assessment of household concentrations of fine particulate matter, ultrafine particles, BC and carbon monoxide conducted in the same rural village. Results confirmed that HAP in this area is mainly due to cooking activities and showed that the installation of the rocket stove contributed to a significant reduction of fine and ultrafine particulate matter and CO concentrations, but increased indoor BC concentrations. Findings also evidence that, in addition to a switch in the
emission source (i.e. cookstove and/or fuel), successful strategies focused on the improvement of household air quality should improve ventilation practices and appropriate construction materials.

According to results presented in chapters 4 and 5, rocket stoves would have a positive effect with regard to HAP reduction, but more uncertain effects on climate. This proves that the climate and health-relevant properties of stoves do not always scale together and highlights that both dimensions should be considered in technological decisions. Notwithstanding, this research work is focused carbonaceous aerosol emissions, but it does not include climate impacts from other pollutants emitted or factors such as biomass renewability, so results do not mean that rocket stove types do not have climate benefits, as the full picture is much more complicated and uncertain.

The information presented in this thesis helps to have a more accurate picture of the climate and health impacts of residential burning of biofuels for cooking in the West African region. Moreover, it gives useful insights for the selection of the more appropriate technologies and it may be useful for stove designers to optimize environmental and health co-benefits of cooking solutions.

Further laboratory and field studies of different cooking technologies and fuels are needed, covering different stove-fuel combinations, in other areas of West Africa, and also studying the seasonality effect on emissions to have more representative EF data to be used in emission inventories and climate prediction models.
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<tr>
<td>ACCES</td>
<td>African Clean Cooking Energy Solution Initiative</td>
</tr>
<tr>
<td>ANOVA</td>
<td>Analysis of variance</td>
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<tr>
<td>BC</td>
<td>Black carbon</td>
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<tr>
<td>CCAC</td>
<td>Climate and Clean Air Coalition to Reduce Short-Lived Climate Pollutants</td>
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<tr>
<td>CDM</td>
<td>Clean Development Mechanism</td>
</tr>
<tr>
<td>CH₄</td>
<td>Methane</td>
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<td>CO</td>
<td>Carbon Monoxide</td>
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<td>EC</td>
<td>Elemental carbon</td>
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<td>ECOWAS</td>
<td>Economic Community of West African States</td>
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<td>ECREEE</td>
<td>ECOWAS Centre for Renewable Energy and Energy Efficiency</td>
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<td>Global Warming Potential</td>
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<td>Laboratory emission monitoring system</td>
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<td>Low and middle income</td>
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<td>Liquefied petroleum gas</td>
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<td>Products of incomplete combustion</td>
</tr>
<tr>
<td>PM</td>
<td>Particulate matter</td>
</tr>
<tr>
<td>PM₁₀</td>
<td>Particulate matter 10 micrometers or less in diameter</td>
</tr>
<tr>
<td>PM₁₀</td>
<td>Particulate matter 10 micrometers or less in diameter</td>
</tr>
<tr>
<td>Acronym</td>
<td>Definition</td>
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<td>---------</td>
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<tr>
<td>PM$_{2.5}$</td>
<td>Particulate matter 2.5 micrometers or less in diameter</td>
</tr>
<tr>
<td>SDG’s</td>
<td>Sustainable Development Goals</td>
</tr>
<tr>
<td>SE4all</td>
<td>Sustainable Energy for All</td>
</tr>
<tr>
<td>SLCP’s</td>
<td>Short Lived Climate Pollutants</td>
</tr>
<tr>
<td>TOT</td>
<td>Thermo-optical transmission</td>
</tr>
<tr>
<td>UFP</td>
<td>Ultrafine particles</td>
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<tr>
<td>UNEP</td>
<td>United Nations Environment Programme</td>
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<tr>
<td>UNFCCC</td>
<td>United Nations Framework Convention on Climate Change</td>
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<td>UNICEF</td>
<td>United Nations Children’s Fund</td>
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<tr>
<td>VOC’s</td>
<td>Volatile organic compounds</td>
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<tr>
<td>WACCA</td>
<td>West Africa Clean Cooking Alliance</td>
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<tr>
<td>WB</td>
<td>The World Bank</td>
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<td>WBT</td>
<td>Water Boiling Test</td>
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<td>WHO</td>
<td>World Health Organization</td>
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1. Introduction

Over half of the world’s population still rely on polluting and inefficient energy systems to meet their daily cooking needs. This has important social, economic and environmental consequences. Women are disproportionately affected.
1.1. Household energy for cooking. Global situation and impacts

Nowadays, around 3100 million people rely on solid fuels, like fuelwood, charcoal, coal, agricultural residues and animal wastes, as the primary source of energy to satisfy their basic energy needs (WHO, 2016). These fuels are most usually burnt inside homes either in open fires or poorly efficient traditional stoves, which lead to important consequences both on families’ living conditions and the environment.

The African Region, the South-East Asia Region, the Western Pacific Region and some Central American countries have the highest proportions of households primarily using solid fuels for cooking (Figure 1), with significant differences between urban and rural areas. Over 20% of urban households rely primarily on polluting fuels and technologies, while the ratio in rural areas ascends to an 80% (WHO, 2016).

Solid fuels are primarily burned in poorly ventilated spaces with inefficient stoves or open fires, causing high levels of household air pollution (HAP), the single most important environmental health risk factor worldwide (WHO, 2016).

![Figure 1. Global polluting fuel use in 2014. Source: (WHO, 2016)](image)

Indeed, HAP causes the premature death of 4.3 million people each year, a number above the deaths produced by malaria and tuberculosis together (WHO, 2014a) and also leads to many negative health impacts, including low birth weight and stillbirths, tuberculosis, asthma, respiratory infections, cancers, chronic headaches, cataract, and burns (Bruce et al., 2014; Ezzati and World Health Organization, 2004; WHO, 2016). Women are particularly at high risk of disease from exposure to HAP, owing to their greater involvement in daily cooking and other domestic activities, together with their children, who are under their care most of the day (Batliwala and Reddy, 2003; Desai et al., 2004; Torres-Duque et al., 2008) (WHO, 2016).
But the problem does not stay at home. Burning solid fuels in inefficient devices at household level is a major source of ambient air pollution and releases high emissions of some important contributors to global climate change, such as carbon dioxide (CO\textsubscript{2}), methane (CH\textsubscript{4}), black carbon (BC), and other short-lived climate pollutants (SLCP’s). Recent studies estimate that residential solid fuel burning accounts for up to 25% of global BC emissions (Bond et al., 2013) and that 12% of global ambient fine particulate matter (PM\textsubscript{2.5}) emissions come from cooking with solid fuels (Smith et al., 2014).

Furthermore, inefficient combustion for cooking purposes also leads to high levels of fuel consumption. About one third of global wood fuel harvesting is unsustainable, meaning that the rate of depletion of renewable biomass outstrips the rate of regrowth, resulting in a net addition of heat-trapping carbon to the atmosphere (Bailis et al., 2015).

Women and children, mostly girls, are the primary responsible of gathering fuel for cooking in most low and middle income (LMI) countries (Shailaja, 2000; Wickramasinghe, 2003). During this activity, they face safety risks, such as physical and sexual attacks, in addition to suffer significant constraints on their available time for education, rest and activities for income generation (WHO, 2016).

These facts lead to the conclusion that the global community must intensify in an urgent and inescapable manner its efforts to expand and accelerate access to clean household energy (WHO, 2016). Fortunately, today a growing consensus exits that expanding access to clean household energy for cooking (together with heating and lighting) is essential to make progress toward several Sustainable Development Goals (SDG’s), and presents an enormous opportunity to leverage the synergies currently offered by initiatives that encompass energy, gender, health and climate change (Mazorra, 2017; WHO, 2016).

Integrating clean cooking into development agendas can catalyse impacts to end energy poverty (SDG 7), to reduce global mortality and improving overall wellbeing (SDG 3), to promote gender equality and women’s empowerment (SDG 5) and SDG 11 and 13, regarding action to fight against climate change and to make cities and human settlements inclusive, safe, resilient and sustainable (Figure 2).

Figure 2. Household energy for cooking and SDG’s linkages. Source: WHO, 2016
1.2. Overview of clean and improved cooking solutions and related challenges

There is no “one size fits all” when it comes to designing an approach to address the issue of household energy for cooking. A broad range of stove and fuel solutions have been developed in order to make the shift from polluting cooking options to clean and improved ones, each one with differentiated efficiencies, costs, distribution models and challenges in term of meeting user needs.

Some of these solutions imply a switch to other type of fuels different from the traditionally used biomass, usually known as clean fuels, as they produce very low or any indoor air pollution (IAP) in homes (Putti et al., 2015):

**LPG stoves:** In this stoves, liquefied petroleum gas (LPG) from a canister burns very cleanly (from an indoor air pollution perspective) and efficiently. A typical LPG cooking system is made up of a steel cylinder filled with LPG, a pressure controller, a tube connecting the cylinder to the pressure controller and the burner, and finally the burner itself. The burner can consist of one or more cooking tops. LPG stoves are very convenient for users as they heat up quickly and temperature can be precisely controlled. However, LPG is a fossil fuel, with a measurable carbon footprint.

**Solar cookstoves:** Generate heat by directly capturing sun’s solar thermal energy. The fuel is available at no cost, and, as no smoke is produced, solar cookers do not produce any health or impact associated with cooking. However, these stoves can take more time to cook, and the efficiency depends of the strength of the sun at any given time. Moreover, it is very difficult to adapt to some meal times (i.e. it is not possible to cook at night or breakfast very early in the morning).

**Alcohol cookstoves:** Ethanol and methanol, pressurized or non-pressurized, typically achieve high combustion rates, and are therefore very efficient. They emit very low levels of carbon monoxide (CO), volatile organic compounds (VOC’s) and BC. They can also produce a carbon footprint. Moreover, the fuel is not widely available and could be costly.

**Biogas stoves:** Domestic biogas digesters convert organic wastes (animal manure, human waste, and crop residues) into biogas that includes methane, which is then effectively used in simple gas stoves. Biogas typically produces little household air pollution with no additional greenhouse gases (GHG) emitted to the atmosphere. However, cultural barriers and maintenance costs difficult the adoption of biogas stoves in many regions of the world.

**Electric cookstoves:** This stoves convert electrical energy into heat. Its use is limited to areas with electricity access, excluding most rural communities. Electric stoves do not
produce emissions at household levels, although environmental impact depends on the energy source.

During recent years, an increasing access to low cost LPG, electric stoves and other clean fuel and technologies across Asian, African, and Latin American population, has been produced, primarily between the middle-class in urban areas. However, previous experiences shown that most of the time there is no direct transition to these clean modern fuels. The stacking model, which means the parallel use of multiple types of fuel and technologies in a single household, offers a much more realistic picture than the idea of a fixed “energy ladder”, whereby choice of fuel graduates from solid to non-solid fuels as households get richer (IEA, 2014; Masera et al., 2000; Ruiz-Mercado et al., 2011; Ruiz-Mercado and Masera, 2015). Indeed, even after gaining access to LPG or electricity, traditional stoves may be kept in use for heating, to keep mosquitoes away, to cook certain types of food, to prepare food for their animals, etc.

Moreover, modern cooking appliances, such as LPG or electrical cookers, would be out of reach in the short-medium term for a large proportion of LMI countries population, while biomass would continue to be the fuel that people can most easily access and afford (Foell et al., 2011).

Therefore, gaining access to clean cooking facilities encompasses not only switching to alternative fuels, but also access to improved biomass cookstoves (fired with fuelwood, charcoal or pellets) that are more efficient, safer, and more environmentally sustainable than the traditional facilities, and still using solid biomass (IEA, 2014). A wide range of biomass stoves have been developed and the most common are as follows:

**Rocket stoves:** They consist in a L-shaped combustion chamber where the flow of hot gases is directed closer to the pot. Production of rocket stoves range from centrally produced products to locally produced artisanal products, which leads to high variations in the emission reductions and efficiency achieved. The rocket stove design is the most commonly type of improved cookstove used in sub-Saharan Africa.

**Plancha cookstoves:** They are designed to enclose the fire and to exhaust the emissions from combustion though a chimney. They are specialized for typical gastronomical practices in Central in South America. As in the case of rocket stoves, their performance varies greatly depending on design and conditions.

**Natural and forced draft gasifiers:** They convert biomass into a combustible gas using a two-stage combustion process: firstly, the biomass fuel is heated and converted into a combustible gas in the pre-combustion (or gasification) chamber; then the gas is completely oxidized in the second-stage combustion chamber, resulting in very few emissions (Roth, 2011). Forced draft gasifiers have a fan to blow air into the combustion chamber, resulting in a more complete combustion of the biomass fuel.
Performance of these stoves can be improved by using processed biomass fuels, such as pellets or briquettes.

**Improved charcoal stoves:** Typically consist of ceramic liner in a metal cladding, providing a higher efficiency, a hotter flame and an improved combustion when compared to traditional all metal charcoal stoves. Although performance varies a lot through design and production characteristics, in general, charcoal stoves emit very little particulate matter (PM), but considerable CO.

Improved and clean cooking solutions presented above, both using alternative and biomass fuels, have the potential to bring important and interconnected benefits (Figure 3), which are highly dependent on the fuel-technology choice, as well as the type of use and maintenance during daily cooking activities.

![Figure 3. Potential benefits related to the sustained use of improved and clean cooking solutions. Source: Adapted from Mazorra, 2017](image)

Being conscious of the potential benefits achieved through the use of clean and improved cooking solutions, a wide variety of actors (governments, corporations, foundations, civil society, individuals...) have been involved in their promotion since the 1980’s in many regions of the world. However, even though international and national policies, programmes and interventions in the last 40 years have advanced solutions, they have failed to substantially alter long-term trends (WHO, 2016).

This is strongly linked with the fact that interventions have traditionally focused in the installation of stoves, while overlooking a lot of other key factors, such as having a deep knowledge of user needs and preferences, the complexity linked to behaviour...
changes, or the highly importance of gender roles, inequalities and relative power in household decision-making.

As main responsible for household energy provision and cooking activities, women are not only the mainly impacted by problems of traditional used of solid fuels, but essential in influencing from switching to cleaner cooking options (Austin and Mejia, 2017; Ryan, 2014). However, in patriarchal households, women’s first-hand experiences of indoor air pollution often carry little sway and their preferences for healthier cooking options may be trumped by unilateral influence of some men over house-hold energy decisions (Ryan, 2014; WHO, 2016).

Other barriers to the wider uptake of improved solutions for cooking are related with the lack of awareness of health hazards and environmental degradation, the unaffordability of the proposed cooking solutions, the lack of supply and repair services, as well as religious and cultural believes (Puzzolo, 2013).

Moreover, recent studies showed that many improved biomass cookstoves on the market today, though a great improvement regarding traditional cooking, still produce enough PM$_{2.5}$ and BC emissions to be considered a health and environmental hazard (Grieshop et al., 2011; Jetter et al., 2012; Kar et al., 2012; Mazorra, 2017; Smith et al., 2014; Sota et al., 2014).

During the last years, great efforts have been made to understand and overcome the varied and complex barriers to increase the adoption and sustained use of improved and clean cooking solutions, with the key support of international initiatives like the Global Alliance for Clean Cookstoves (GACC), the Climate and Clean Air Coalition (CCCA), the World Health Organization (WHO), the United Nations Children’s Fund (UNICEF), the Sustainable Development Goals (SDG’s), or the Sustainable Energy for All initiative (SE4all).

Together, these organizations are working in developing innovative technological and finance solutions, increasing the cross-sector cooperation, promoting interdisciplinary research, expanding the knowledge base of impacts of traditional cooking, demonstrating the cost-effectiveness of clean cooking interventions, developing the market sector of clean and improved stove and fuels, monitoring and evaluating the adoption and effectiveness of the solutions promoted, searching for solutions to address gendered barriers to the adoption of cooking solutions, enabling policy and regulatory environment, and promoting the creation of more gender-responsive policies.
1.3. Research opportunities

Strengthening the base of evidence of the impacts related to the use of pollutant cooking systems and demonstrating the magnitude of the health, environmental, and socio-economic benefits of clean cookstoves and fuels is a critical priority that will help drive investment into solving this issue (Anenberg et al., 2013; Mazorra, 2017).

Hundreds, if not thousands, of stove related innovations have been implemented around the world (Simon et al., 2014), but not all provide the same benefits. While these stoves often look simple in appearance, the inherent stochastic nature of biomass combustion, combined with natural variations in daily use and the wide variety of fuel-stoves combinations available, make stoves evaluation challenging (L’Orange et al., 2012).

In order to provide transparency regarding the performance (i.e. efficiency in fuel use, emissions, safety, etc.) and subsequent potential benefits of different cooking solutions, standard certification procedures and performance benchmarks are being developed (Anenberg et al., 2013).

Putting these new guidelines into practice require increasing the evaluation of stoves and fuels performance under both laboratory and various real-world conditions. Pollutants different than the traditionally measured PM$_{2.5}$ or CO, such as ultrafine particles (UFP), BC and other co-pollutants, which leads to important health and climate impacts (CCAC, 2014; Kinney, 2015; WHO, 2016)) should be further explored (Anenberg et al., 2013).

Moreover, the need of studies to quantify coking-related impacts is not homogeneous across the globe. Traditionally, most studies have been focused in Asia (mainly India and China), with far few studies in other regions in the world. In the West African region, for example, where more than 730 million people rely on biomass for cooking (Saho and Reiss, 2013), very little research has been conducted to measure the impact of improved and clean cooking solutions.

This issue is particularity significant, as the impacts of available household cooking options can vary sharply from one region to another (Grieshop et al., 2011). In the case of BC and other SLCP’s, the impacts on climate and ambient air quality are highly spatially variable owing to several factors, and thus global-scale assessments may not properly represent the consequences on national and regional scale (Lacey et al., 2017).
1.4. Research aim and overview

This thesis aims to contribute to the existing knowledge about climate and indoor air quality impacts of biomass cookstoves, with a particular emphasis on BC and co-pollutant emissions, and focusing in the West African region. To that end, this dissertation addresses several specific research goals, which are presented in the following chapters.

Chapter 2 frames the context of the research. It develops the key concepts for understanding the importance of IAP and BC and co-pollutant emissions from cookstoves, and provides an overview of the situation of household clean energy access in sub-Saharan and Western Africa.

Chapter 3 presents an inter-comparison of analytical methods to estimate BC emissions from cookstoves. The performance of two relatively low-cost and easy-to-use methods is validated against a more accurate device, with two different aerosols types and different filtrates substrates. This validation helps to overcome the current existing methodological barriers to determine BC emissions from cookstoves in resource-constrained locations.

Chapter 4 presents a characterization, both in field and laboratory conditions, of BC and organic carbon (OC) emissions from a set of stove-fuel combinations used in Senegal and neighbouring countries. This novel information helps to have a more accurate picture of the climate impacts of residential burning of biofuels for cooking in the West African region.

Chapter 5 sets out the first study developed in Senegal to provide a field assessment of household concentrations of PM$_{2.5}$, UFP, BC and CO from the combustion of fuelwood for cooking. The findings add to the small number of quantitative studies on cook-smoke exposures in West-Africa and establish the bases for interventions to reduce IAP and corresponding adverse health impacts in this region.

Chapter 6 shows the primary contributions of the research work, and Chapter 7 concludes this thesis by outlining the limitations of the study and future research areas.
1.5. Personal motivation

My interest on cooking energy started in January 2013 in the region of Casamance (Senegal), where I had the chance to participate in an initiative of improved stoves implementation. When I first visited those rural communities, I felt really impressed by the scale and severity of problems related to the daily and essential action of cooking: enormous levels of smoke and heat inside cooking areas, where women spent around 5-6 hours per day, cost and time demanded for gathering fuel, etc.

When I came back to Spain, I decided to start a PhD on this issue to contribute with solutions to that critical situation, which is still minimally discussed at the energy and environment arenas in comparison with other advanced energy technologies, commercial fuels, and large-scale power plants.

Another motivation to conduct this research was to make this problem visible, because, despite of its severity, it still unknown for many people. The use of polluting technology to cover household energy needs exacerbates gender inequality it’s a big impediment to the personal development of many people and also and impediment for humanity to aspire to an equitable and sustainable development. This situation can and must change.

Lastly, I chose this research because it was an opportunity to look at the human dimension of energy use and environmental change, which is, surprisingly, not very commonly studied in engineering studies.
2. Research context
2.1. Cookstoves and household air pollution

Under ideal conditions, the combustion of any fuel produces only useful heat, CO$_2$ and water. But household combustion is far from ideal, rather always incomplete and often quite inefficient (WHO, 2016). During burning of solid fuels for cooking activities, products of incomplete combustion (PIC’s), such as PM, CO, poly-aromatic hydrocarbons (PAH’s), VOC’s, BC and many other substances are emitted. All of them are toxic to human beings in different ways and to varying degrees, turning the home into one of the most health damaging environments (WHO, 2016).

In dwellings with poor ventilation, 24-hour levels of PM$_{2.5}$ can reach 2,500 μg/m$^3$, i.e. 100 times the levels recommended as safe by WHO guidelines (WHO, 2006), and peaks during cooking may be as high as 20,000 μg/m$^3$ (see chapter 5). Exposure to IAP causes 4.3 million premature deaths each year and many adverse health outcomes, such as low birth weight and stillbirths, tuberculosis, asthma, respiratory infections, cancers, chronic headaches, cataract, and burns (Bruce et al., 2014; Ezzati and World Health Organization, 2004; WHO, 2016)

Gender roles in the household are major determinants of relative health risks to IAP (WHO, 2015). Women and children are at a particularly high risk of diseases, as they are the ones who spend most time in and around the home (WHO, 2016). IAP is the second largest environmental risk of non-communicable diseases in women of LMI countries, and it is the case of over half of childhood pneumonia deaths (the largest cause of death in children under 5 years) (WHO and EPA, 2016).

Moreover, solid cook fuel is sufficiently polluting and widespread to appreciably affect ambient (outdoor) air pollution (Smith et al., 2014). Indeed, recent studies estimated that IAP from cooking accounts for up to 12% of global ambient PM$_{2.5}$ pollution (Smith et al., 2014). Moreover, in some countries as India and China, as much as 30% of ambient air pollution is caused by household emissions (Zhang and Kotani, 2012).

In view of this situation, in 2014 the WHO published the “Guidelines for indoor air quality: household fuel combustion”, specifying targets for emissions rates for CO and PM$_{2.5}$ from household fuels and energy devices. These guidelines were developed with a pragmatic approach providing both interim targets (for 60% of homes meeting the targets) and aspirational targets (for 90% of homes meeting the targets). Moreover, the document provides guidance on the policy for a transition to a sustained adoption of clean fuels and technologies and emphasizes the importance of addressing all main household energy end uses for health benefits (WHO, 2014b).

These guidelines are also serving as the basis for an ongoing process to develop international cookstove standards by the GACC in collaboration with the International Organization for Standardization (ISO). The ISO process is still ongoing, but it has
already developed a framework for rating stoves from a baseline (Tier 0) to an aspirational situation (Tier 4), considering four indicators: fuel efficiency, emissions (total and indoor) and safety (GACC, 2012). PM$_{2.5}$ and CO emissions rates that define Tier 4 were determined based on the WHO Guidelines, while other Tier boundaries for Indoor Emissions were defined to be progressively lower relative to Tier 4.

![Figure 4](image)

**Figure 4. Emission rates by stove type and ISO Tiers classification. Source: Adapted from Putti et al., 2015**

Figure 4 shows that liquid and gas fuels such as LPG, propane, biogas, and alcohol, offer very low PM$_{2.5}$ and CO emission rates. Well performing gasifiers are the biomass stoves providing the lowest emissions of both PM$_{2.5}$ and CO. Improved stoves using biomass fuels, such as the rocket stove, may perform better in terms of emissions over the baseline of a traditional stove, but it may not produce low enough emissions at point-of-use to result in meaningful health benefits (WHO, 2016). Charcoal stoves typically have lower PM$_{2.5}$ emissions than rocket or traditional stoves, but can create dangerous levels of CO.

At this point, the challenge is to select cooking solutions that are affordable and acceptable to households; and, at the same time, sufficiently clean and able to effectively displace traditional fires to achieve dramatic emission reductions (Martin et al., 2014). Given that the alternative of providing use of clean fuels such as LPG may not be realistic for a big portion of population in LMI countries in the short-medium term (Martin et al., 2014), improved and clean biomass-burning stoves can be an important step towards cleaning up indoor air (WHO, 2016, p. 20).

In addition to emissions, a variety of factors affect pollutant concentrations levels in a closed space: ventilation, spatial characteristics, emission from external sources, emissions that can re-infiltrate, combination of different stoves, etc. (WHO and EPA,
Therefore, strategies and interventions to reduce IAP should focus on reducing the emission source as much as possible through a switch to clean and improved cooking solutions, while promoting adequate ventilation practices (chimneys, kitchen materials, windows, etc.) and ensuring a correct and sustained use of technologies (WHO, 2014b).

The key to any of these strategies is to develop a monitoring and evaluation system that documents stove use, emissions, indoor air concentration of pollutants and exposure levels in and around the household (Martin et al., 2014). Moreover, additional parameters different than PM$_{2.5}$ and CO, such as particle composition, number of particles, and surface area, should also be further explored (Anenberg et al., 2013).

### 2.2. Carbonaceous aerosol emissions from cookstoves

Black carbon (BC), or Elemental Carbon (EC), is a carbonaceous aerosol formed from the incomplete combustion of biomass and fossil fuels. It is a SLCP, with days to weeks of lifetime in the atmosphere, and considered a dangerous air pollutant and the second biggest climate forcer after CO$_2$ (Jacobson, 2000).

BC refers to the light-absorbing carbon, measured by change in light transmittance or reflection, whereas EC is measured by thermal evolution under high-temperature oxidation (Venkataraman et al., 2005). Even if EC and BC are not exactly equivalent (Petzold et al., 2013), both terms are often used interchangeably (Bond and Bergstrom, 2006; Chen et al., 2005; Li et al., 2009; Shen et al., 2010; Venkataraman et al., 2005).

BC contributes to climate change in a variety of ways: when airborne, it absorbs solar radiation, stimulating earth warming. When deposited on ice and snow, BC decreases the surface albedo, making them less reflective and more light absorbent. In this way, it accelerates the melting effect, especially in highly vulnerable environments as the Arctic and in glaciated regions like the Himalayas (Li et al., 2016).

Furthermore, BC causes impacts on environmental conditions at local and regional scale, such as visibility reductions and changes in rainfall patterns, which can be substantially larger than global impacts (Ramanathan and Carmichael, 2008).

Each region of the world has a unique mix of natural and pollution carbonaceous aerosol sources. In many regions, such as North America, Latin America, and Europe, diesel combustion for transportation is the dominant contributor to BC (Bond, 2004), while 60–80% of total BC emissions within Asia and Africa comes from residential burning of biofuels, mainly for cooking purposes (Bond et al., 2013) (see Figure 5).

Nowadays, global inventories indicate that residential sources are responsible for over 25% of BC emissions worldwide (Bond et al., 2013), although important regional
variations in emissions are expected in the coming decades, with decreases of up to 50% in North America and Europe due to mitigation measures in the transport sector, and significant increases in Asia and Africa (CCAC, 2014). By 2030, BC from traditional bioenergy use in Asia and Africa is expected to contribute to close to half of all global BC emissions (UNEP and WMO, 2011).

![Figure 5. Main BC sources by region and sector (2005). Source: adapted from (CCAC, 2014)](image)

BC is always co-emitted with organic carbon (OC), the other major component of carbonaceous aerosols, which has been traditionally linked to light scattering properties with consequent cooling effects in the atmosphere (Chung, 2002).

However, OC has recently been found to be a source of light absorption in the atmosphere due to the presence of brown carbon, which strongly absorbs radiation in the ultraviolet wavelengths (Bond et al., 2007; Chung et al., 2012; Feng et al., 2013; Saleh et al., 2014). The ratio of BC to OC, the portion of brown carbon and other co-pollutants with cooling effects (such as sulphate aerosols) depend on the source of emissions and determines the net warming or net cooling effect (CCAC, 2014).

In addition to climate effect, BC is a powerful air pollutant with detrimental impacts on public health. Recent studies have reported significant associations between long-term exposure to black carbon and all-cause and cardiopulmonary mortality, cardiovascular and respiratory diseases (Janssen et al., 2012, 2011).

Therefore, addressing BC emissions would improve the chances to remain global temperature rise below the maximum target of two degrees Celsius set under the international climate negotiations (UNEP and WMO, 2011), while providing significant
health benefits. Moreover, short BC lifetimes mean that climate benefits would be achieved quickly after mitigation (CCAC, 2014; WHO, 2016).

In recent years there has been an increase in studies and institutions addressing BC’s impacts. An example of this is the creation of the Climate and Clean Air Coalition to Reduce Short-Lived Climate Pollutants (CCAC), a voluntary partnership of governments, intergovernmental organizations, businesses, scientific institutions and civil society organizations committed to reduce SLCP’s, hosted by the United Nations Environment Programme (UNEP) and the WHO. The CCAC has proposed sixteen cost-effective control measures to reduce BC and other SLCP’s (CCAC, 2014; WHO, 2016), one of them being the replacement of traditional cooking with modern fuels and clean-burning biomass stoves.

Another remarkable initiative is the development of a new methodology for the quantification of BC and co-emitted species from cookstove projects, promoted by the Gold Standard Foundation and a group of cookstove organizations (The Gold Standard, 2015). These pollutants are not covered by the existing methodologies used for Clean Development Mechanism (CDM) projects and other mechanisms established under the United Nations Framework Convention on Climate Change (UNFCCC), which can result in incorrect judgments about the relative benefits of clean and improved stoves solutions (Johnson et al., 2008a).

The new methodology will award a certification, which should not be confused with carbon offsets. Organizations can not compensate emissions of GHG, like CO\textsubscript{2}, by reducing BC elsewhere, since BC impacts are highly localized depending on the region, season, and local weather. Meanwhile, inclusion of BC co-benefits in cookstoves projects may provide an opportunity to attract new additional funding for further scaling up of these activities (Hamrick, 2015).

However, despite the ongoing efforts to increase capacity for testing BC and other SLCPs, most biomass stoves technology worldwide have not been tested or proved to reduce BC yet (Jetter, 2015). The complexity of studying aerosol emissions from cooking is related to the wide distribution and remote location of the sources (Johnson et al., 2008b), the complicated effect of OC (Feng et al., 2013), the relatively complex BC and OC measurement methods (de la Sota et al., 2017; Ramanathan et al., 2011b), and because EC/BC and OC emissions range widely between different cooking technologies and biofuels (Grieshop et al., 2011; Guofeng et al., 2012; MacCarty et al., 2008b; Venkataraman et al., 2005)

As a consequence, further studies should be conducted to provide concrete scientific data allowing estimation of the net climate effect of different improved and clean cooking solutions in different regions worldwide.
2.3. Household energy for cooking in sub-Saharan and Western Africa

In many parts of the world, the use of biomass as energy source for cooking has generally peaked or will do so in the near future (Sander et al., 2011a). In sub-Saharan Africa, however, biomass is the main fuel used for cooking for 80% of the population (i.e. 700 million people) and will likely be for 880 million by 2020 (IEA, 2014).

Electricity and LPG are used exclusively for cooking by less than 10% in most African countries, less than 1% of households cook with kerosene and the vast majority of countries do not use coal at all (IEA, 2016; Rysankova et al., 2014; WHO, 2016).

When considering rural areas, almost 95% of the population use fuelwood and charcoal, and just 2% and 3% of households report use of electricity and gas as their primary cooking fuel (WHO, 2016). In urban areas there is also an important share of biomass consumption, much as charcoal, combined with LPG, natural gas, and electricity.

Around 7.5 million Tonnes (Mt) of PM$_{2.5}$ are emitted annually in Africa, of which almost three-quarters is from the indoor burning of biomass (IEA, 2016). The inefficient combustion of solid biomass in traditional stoves makes the residential sector in Africa the major emitter of PM$_{2.5}$ worldwide, accounting for more than 15% of all energy-related PM$_{2.5}$ emitted today (IEA, 2016).

In sub-Saharan Africa, IAP exposure is the single greatest health risk for women and girls (WHO, 2016), causing around half a million of premature deaths annually (IEA, 2016), half of them being children under five (Dherani et al., 2008). If no action is taken, by 2030, 870,000 people will die annually from diseases linked to solid fuel cooking (Rysankova et al., 2014).

More than 300 Mt of wood are consumed each year in Sub-Saharan Africa for cooking, including 130-180 Mt of wood for charcoal production, mostly conducted with traditional technologies with low wood-to-charcoal conversion efficiencies (Liyama, 2013; Rysankova et al., 2014). Recent data confirmed that woodfuel use in certain parts of sub-Saharan Africa, Eritrea, western Ethiopia, Kenya, Uganda, Rwanda and Burundi and some parts of West Africa, is unsustainable (Bailis et al., 2015). In some African countries, women spend as many as five hours per day gathering fuelwood for cooking (OECD/IEA, 2014).

Solid fuel use and charcoal production in sub-Saharan Africa generates 120–380 Mt CO$_2$-eq of Kyoto Protocol greenhouse gases (0.4–1.2% of global CO$_2$ emissions) and up to 600 Mt CO$_2$-eq when PM is included (Lambe et al., 2015b). Moreover, the change in rainfall patterns in sub-Saharan Africa has been associated with the increasing burning of biomass for cooking and land clearing (Ichoku et al., 2016; Ramanahatan, 2007).
According to IEA projections for 2040 (Figure 6) (OECD/IEA, 2014), the mix of fuels used by household for cooking energy is relatively inelastic. This is in part due to the rapid population growth expected in this region of the world, which can outstrip the moderate incremental progress in increasing access to clean, modern energy systems. Moreover, a transition to cleaner cooking fuels and appliances is not straightforward, as people who have access to fuels, such as LPG, natural gas, biogas or electricity, may also continue using solid biomass due to cultural or affordability reasons (fuel stacking) (OECD/IEA, 2014). Indeed, historical trends over the last 25 years show that population relying on traditional use of solid biomass has tracked population growth fairly closely, despite increasing incomes (Rysankova et al., 2014).

Meanwhile, one of the major changes within IEA projections is not captured in Figure 6, because it involves not a fuel switch but a change in the way that solid biomass is used (OECD/IEA, 2014). Access to more efficient stoves is growing rapidly across sub-Saharan Africa. For example, penetration rate of rocket stoves and similar improved stoves doubled from 4 million in 2011 to 8 million by the end of 2013 and advanced biomass cookstoves and clean renewable cooking alternatives such as biogas, solar and liquid biofuels cumulatively reached 1.3 million African families by late 2013 (Rysankova et al., 2014). Many improved cookstove businesses are already operating, and their numbers are growing as cookstove technologies advance and innovations in end-user finance make stoves more affordable (Lambe et al., 2015b).

But, despite these progress, the overall penetration of clean cookstoves and fuels in sub-Saharan Africa is still low, with only one in six households using clean cooking energy for most cooking needs (Lambe et al., 2015b), in partly due to the lack of finance resources in African households. Many cookstove programme implementers have seen carbon finance as an interesting revenue option to overcome the finance barriers in sub-Saharan Africa. Indeed, in 2013, 43% of new funding was attributed to

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**Figure 6. Household energy consumption for cooking by fuel in Sub-saharan Africa. Source: IEA, 2014**
carbon offset sales ($45 million) or carbon fund investments ($3.6 million) and the sale of carbon offsets drove the distribution of 1 million cookstoves (GACC, 2014; Lambe et al., 2015b).

In addition to economic revenues, one of the most recognized benefits of carbon finance is that, as it requires rigorous monitoring and evaluation, it helps to follow up the cookstove use, and enhance quality assurance on the production line (Lambe et al., 2015b). However, demand for carbon credits is currently minimal, so cookstove projects have to develop sustainable business models or other alternative source of revenue to support their operations (Lambe et al., 2015a).

Within the Economic Community of West African States (ECOWAS), which represents approximately one third of sub-Saharan Africa’s total population (Auth and Musolino, 2014), around 75% of population still uses wood and charcoal for cooking, often in inefficient stoves (Saho and Reiss, 2013). It is estimated that quite minor shares of the population (e.g. Sierra Leone (10%), Senegal (16%) and the Gambia (20%)) are using improved biomass cookstoves and less than 25% of inhabitants rely on electricity, kerosene and LPG for cooking (Auth and Musolino, 2014). As a consequence, IAP affects 257.8 million people and cause the prematurely death of 174,000 people each year (Auth and Musolino, 2014).

The fact that West Africa is one of the regions with a largest expected population increase (OECD/IEA, 2014) magnifies common challenges existing in other sub-Saharan African countries to achieve modern energy access for cooking. But, at the same time, West Africa is experiencing the most rapid economic growth of sub-Saharan regions (OECD/IEA, 2014), bringing potential opportunities for the development of clean cooking sector.

Indeed, a wide variety of initiatives in the region are promoting access to cooking solutions. In 2012, the ECOWAS Centre for Renewable Energy and Energy Efficiency...
(ECREEE) initiated a regional Cooking Energy initiative called West African Clean Cooking Alliance (WACCA) with the objective of promoting the adoption of clean and efficient cooking devices for all ECOWAS households, through policy and regulatory design, capacity building, and technology promotion. Other initiatives in the region include the World Bank-led African Clean Cooking Energy Solution Initiative (ACCES), which aims to promote market-based, large scale dissemination of clean cooking solutions in sub-Saharan Africa.
3. **Inter-comparison of methods to measure black carbon from cookstoves**
3.1. Background

As discussed in previous chapters, emissions of BC resulting from residential burning of biofuels is a climate and health global concern (Arora and Jain, 2015), and its quantification is critical to understand and evaluate the effectiveness of BC mitigation actions, such as the introduction of cleaner and more efficient cooking technologies.

However, BC emissions from cookstoves have been less characterized than those from other common sources, such as diesel engines (Johnson et al., 2008a). In recent years, the number of studies measuring biofuel combustion emissions increased (Bhattacharya et al., 2002; Fan and Zhang, 2001; MacCarty et al., 2008b; Zhang et al., 2000) but only a limited number of them analyzed BC cookstove emissions using optical and thermal-optical methods (Johnson et al., 2008a; Patange et al., 2015; Ramanathan et al., 2011a; Rehman et al., 2011; Venkataraman et al., 2005).

The scarcity of studies on biofuel emissions from cookstoves and, more specifically, on BC emissions, is due to the complexity related to measuring emissions from such a heterogeneous and diverse activity (MacCarty et al., 2008b). In general, pollutant measurements (including BC) in rural and resource-constrained areas are limited by the lack of adequate instrumentation, power supply and reproducible environmental conditions. Other causes are the wide distribution and remote location of these stoves, as well as the relatively invasive and complex assessment methods available (Johnson et al., 2008a). Very often, BC measurement techniques pose a challenge in terms of their high upfront cost and level of technical expertise required for operating them (Rehman et al., 2011).

In this chapter, the performance of two low-cost and easy to use methods for BC estimation were assessed: reflectometry, and a cell-phone-based system (in general, any camera-based system). These methods have the potential to be used for studies in resource-constrained locations due to their low cost, portability and low technical requirements. Moreover, their easy use can open up possibilities of involving cookstove users in the data collection process, which would result in awareness raising on this topic (Nelms et al., 2016). Although there are several examples of citizen science projects (Castell et al., 2015; Kaufman et al., 2014), this approach has rarely been used in cookstoves studies. The increased use of mobile phones can bridge the gap in resource constrained locations, where other infrastructure is lacking.

The performance of the lower-cost methods was assessed by comparison with the European reference method for EC quantification on filter samples (CEN TC264-WG35), thermo-optical transmission (TOT) analysis (Cavalli et al., 2010) which, due to its relative complexity and cost, is in general not applicable for field work in situations with limited means. The results obtained with the three techniques were compared for different aerosol types and loads, as well as for different filter substrates.
The aim was to assess the performance of the reflectometer and cellphone optical analysis when compared to the thermal-optical transmission (TOT) analyzer. If proved to be comparable within a given uncertainty range, these methods could be recommended as lower-cost tools for estimation of BC from cookstove emissions in resource-constrained locations.

3.2. Methods

3.2.1. Sample collection

Aerosol samples were collected in two locations, Dakar (Senegal) and Barcelona (Spain). Samples were collected to obtain a range of BC and EC loads (from low to high) and to assess the influence of different aerosol types on measured BC and EC concentrations. This enables a more robust comparison of the three techniques (Ahmed et al., 2009). Samples were collected on quartz fibre filters (Pallflex, tissuesquartz, 2500 QAT-UP) and glass fibre filters (Hi-Q environmental products, FPAE-102, 0.3 μm aerodynamic equivalent diameter).

In Dakar, samples were collected at the Research and Studies Centre on Renewable Energies of the University of Dakar (CERER, UCAD) using a Laboratory Emissions Monitoring System (LEMS), developed by the Aprovecho Research Centre (ARC) (Figure 8). This system was designed to sample cookstove emissions after dilution. In this way, the aerosol concentrations sampled may be considered as a good approximation of indoor emissions in a typical rural household in Senegal during the cooking hours. In Barcelona, on the other hand, aerosol concentrations were sampled in outdoor air at an urban background air quality monitoring station located in the vicinity of traffic emissions (IDAEA-CSIC Station).

Both types of samples were representative of markedly different aerosol types: biomass burning aerosols for indoor air quality in households using solid fuels as primary source of energy in Senegal, and urban aerosol dominated mainly by traffic emissions, characteristic of ambient air in a typical European city, Barcelona.

In the LEMS in Senegal, the stove was placed under a hood structure to collect exhaust flows produced during the combustion. The LEMS was equipped to measure CO, CO₂, and PM_{2.5} in real-time. It had also a gravimetric system to measure PM_{2.5} using filter-based sampling, which consists of a vacuum pump that draws a sample through a sample line and a critical orifice at a steady flow of 16.7 L/min. A cyclone particle separator was used so that PM_{2.5} was collected on a filter while the pump was switched on.
Filters for PM collection were regularly changed as they were quickly saturated with PM$_{2.5}$. To solve this, the ARC designed a modification of the gravimetric system by adding a second sample train in parallel with the gravimetric (PM$_{2.5}$) sample train, with a lower flow rating of 3 L/min (Figure 9). This separate BC sample train allowed a simultaneous collection of PM$_{2.5}$ and BC samples, as well as the use of different types of filter media.

The Water Boiling Test (WBT) protocol (version 4.2.3) was applied. It consists of a standardized test, commonly used in the laboratory, which is a simplified simulation of the cooking process intending to measure how efficiently a stove uses fuel to heat water in a cooking pot and the emissions produced while cooking (EPA et al., 2014). In this test, 5 litres of water are brought to boil (cold start phase) and then simmered for 45 min (simmer phase). Although WBT has recognized limitations to reproduce actual random conditions in the field (Johnson et al., 2008a; Roden et al., 2006; Shen et al.,
2013), its repeatability allows to study the effect of design on the performance of cookstoves, as well as the emission process influencing factors, mechanisms, kinetics, etc. (Arora and Jain, 2015; Chen et al., 2012; Shen et al., 2013).

In Barcelona, ambient air PM$_{2.5}$ samples were collected by means of a MCV CAV-A (MCV S.A.) high-volume sampler (30m$^3$/h). This type of sampler is certified to be equivalent to the EU reference for PM$_{2.5}$ monitoring. The sampler was located at the IDAEA-CSIC reference station (41°23′14″ N, 02°06′56″E), and samples were collected over 24h periods.

In total, 102 quartz fibre filters and 81 glass fibre filters were collected in Senegal, and 222 quartz fibre filters were collected in Barcelona. Some of these samples were removed because BC deposition was below or above analytical limits of detection. Finally, 73 quartz fibre filters collected in Senegal and 76 in Barcelona were analysed with the three analytical methods, and 52 glass fibre filters sampled in Senegal were analysed with the cell-phone system and the reflectometer, given that glass fibre filters are not recommended for analysis by TOT. Glass fibre filters are generally preferred to quartz ones in resources-constrained areas due to their lower cost.

3.2.2. Determination of BC and EC on filters

There are different methods to determine the concentration of BC (EC) based on the chemical, physical and light absorption properties of the particles (Petzold et al., 2013). The optical methods quantify the light absorbing component of the particles, and are based on the fact that BC absorbs light due to its colour. The thermal-optical methods determine the quantity EC based on its stability at high temperatures in an inert atmosphere. In this work, we used two optical filter-based techniques (reflectometry and cell-phone based system) and one thermal-optical transmission (TOT) filter-based technique, all of them offline. The latter determines EC concentrations, whereas the former two estimate BC concentrations.

**Cell-phone based system (Nexleaf).** It is an optical technique in which a photograph of the filter is captured with a cell-phone or a camera and transmitted to a server where an algorithm compares the image (the colour of the filter) with a calibrated scale (Figure 10). The BC load is estimated by measuring its reflectance in the red wavelength, based on the “blackness” of the photograph.
This system is easy to use, cheap and rapid and its performance was already assessed by its developers (Patange et al., 2015; Ramanathan et al., 2011a). According to the manufacturer specifications, the measurement range is 5-25 µg BC/cm$^2$, with an accuracy of 20% (Ramanathan et al., 2011a). It is important that the filter colours are not in the limits of the reference scale (too white or too black), and photos of the filters and optical reference cards need to be taken under relatively uniform lighting.

Within a citizen science approach, cookstove users may use the camera on their mobile phones to capture and transmit the images containing the filter and colour chart to the server, to automatically compute the daily BC concentrations for each household. This would only be possible if comparable sampling methods and durations are used, in order to allow for comparison of BC loadings (µg/cm$^2$) between different users.

**Smoke stain reflectometer (Model 43D, Diffusion Systems Ltd.).** The reflectometer is a non-destructive and portable system, commonly used to determine BC concentrations (Begum et al., 2007; Biswas et al., 2003; Downward et al., 2015). It has already been used in several cookstove BC emission studies (Huboyo et al., 2009; Johnson et al., 2008a). A high-performance LED with maximum emission at 650 nm shines on the filter, and the reflected light is measured by a photo-sensitive element (Safo-Adu et al., 2014).

Then, the electrical response is amplified to produce a reflectance reading (RR). The darker the filters, the lower the amount of reflected light, in such a way that low RR values correspond to high BC loads and vice-versa. The reflectometer reads on a scale of 0 (black) to 100 (white), although RR<10 and >90 suffer from major uncertainty (Adeti, P.J, 2012).

**Sunset Laboratory Carbon Analyzer.** This instrument determines the elemental carbon (EC) content on quartz fibre filters by using a thermal-optical method. Carbonaceous aerosols are thermally desorbed from the filter substrate under an inert helium atmosphere followed by an oxidizing atmosphere, using controlled heating ramps. The carbon contained in the sample is detected as CH$_4$ by a flame-ionization detector.
(Petzold et al., 2013) while OC and EC carbon are discriminated by monitoring the optical transmittance through the filter sample. In this work, the EUSAAR2 thermal protocol was used (Cavalli et al., 2010). Even though differences have been reported with the use of different temperature protocols (EUSAAR2, NIOSH, IMPROVE) (Cavalli et al., 2010; Chow et al., 2004) these differences are systematic and may be corrected when comparing results from studies using different protocols for the same types of aerosols.

In this work, the thermal-optical analysis was considered as the reference system for comparison with the two lower-cost optical methods described above. This technique was recently established as the reference EU method for OC and EC quantification on filter samples by the CEN standardization working group WG35 (CENTC264-WG35).

3.2.3. Method limitations

A number of limitations must be taken into account in this assessment:

a) Regarding the optical methods, the extent of filter loading can influence particle, thus biasing the results (EPA, 2012).

b) When using the reflectometer, the device is in direct contact with the filter surface. This may lead to potential mass loss and/or cross contamination (Yan et al., 2011). The use of a teflon ring is recommended to protect the filter and mitigate potential impacts of contamination.

c) The dependence of aerosol absorption on different wavelengths. The reflectometer works in the visible spectrum, and therefore BC measurements may be affected by interference with other light absorbing components. In the cell-phone system, BC load is estimated by measuring its reflectance in the red wavelength, where interference of other light absorbing components, like brown carbon, should be small or within the instrument’s uncertainty range (Ramanathan et al., 2011a).

d) For all three methods, results are influenced by the chemical composition and emissions sources of the aerosol, filter loading and uniformity of the particle deposit (Watson et al., 2005). BC/EC measurements from biomass smoke may be more strongly affected than diesel engine samples, in part because of their higher levels of inorganic components and brown carbon (Novakov and Corrigan, 1995).

e) The methods assessed quantify BC (EC) concentrations, but do not measure organic carbon (OC). This should be considered a limitation from the point of view of climate studies, where both OC and BC (EC) data are needed to obtain a reasonable characterisation of aerosols warming and cooling implications.
3.2.4. Technical requirements and features of the methods

In addition to the performance of the methods regarding their comparability with the reference, data quality and uncertainty of the measurements, the selection of a specific method for measuring BC (EC) emissions from cookstoves should also consider technical requirements. Table 1 includes the main parameters to be evaluated. (*Prices don’t take into account costs of sampling equipment; the only include costs of the equipment for filter analysis).

<table>
<thead>
<tr>
<th>Method</th>
<th>Robustness</th>
<th>Difficulty of use</th>
<th>Requires a computer for operation</th>
<th>May operate on battery</th>
<th>Types of filter applicable</th>
<th>Accuracy (according to manufacturers)</th>
<th>Price*</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cell-phone based system (Nexleaf)</td>
<td>High</td>
<td>Low</td>
<td>Yes or a smartphone, both with internet access</td>
<td>Yes</td>
<td>Paper, Teflon, glass fibre, quartz fibre</td>
<td>Low-Medium (20% agreement with thermos-optical method)(Ramanathan et al., 2011a)</td>
<td>Card with reference scale (free) 2 euro/filter analyzed Cell-phone costs</td>
</tr>
<tr>
<td>Smoke Stain Reflectometer (Model 43D, Diffusion Systems Ltd.)</td>
<td>Medium</td>
<td>Medium</td>
<td>No, device readings are displayed in the screen</td>
<td>No. Power requirements: 110-230VAC 50/60HZ</td>
<td>Paper (recommended by the manufactured ), Teflon, glass fibre and quartz fibre filters</td>
<td>Medium-High 5%</td>
<td>&lt;4.000 euros (approx.)</td>
</tr>
<tr>
<td>Sunset Laboratory Carbon Analyser</td>
<td>Medium-Low</td>
<td>Medium-High</td>
<td>Yes</td>
<td>No. Power requirements: 110-230VAC 50/60HZ</td>
<td>Quartz fibre filters</td>
<td>High (±0,1 µg/cm² for EC)</td>
<td>&gt;40.000 euros (approx.)</td>
</tr>
</tbody>
</table>

Table 1. Technical requirements and features of the methods evaluated during the intercomparison. Source: own elaboration
3.2.5. Data analysis

The statistical program STATGRAPHICS (StatPoint Technologies, Inc., 2014) was used for data analysis. The relationship between the analytical methods was established using the coefficient of determination ($R^2$), and an analysis of variance (ANOVA) was used to compare regression lines from the two types of aerosols. Further, the Wilcoxon signed-rank test (Woolson, 2008) a non-parametric test procedure for the analysis of matched-pair data, was conducted in order to assess the differences between results from the cell-phone and the TOT system.

3.3. Results and discussion

3.3.1. BC analysis by the cell-phone system compared with EC by thermo-optical analysis

Figure 11 presents results for BC load using quartz fibre filters with two different types of aerosols: biomass aerosols from cookstove emissions, and urban aerosols. For both types of aerosols, a high degree of correlation between the cell-phone and TOT methods may be observed ($R^2>0.80$).

For biomass aerosol samples, linear regression resulted in a slope of 1.35 with a correlation coefficient ($R^2$) equal to 0.84 ($p<0.05$), after exclusion of two outliers. In the case of urban aerosols, the correlation was slightly higher ($R^2=0.88$), with a regression line slope=0.71 ($p<0.05$). The systematic error of the methods and the experimental
error may explain the intercept of the regression curve, although the value is very close to zero. These results show that, for each different type of aerosol, BC concentrations measured with the optical cell-phone method are well correlated with the reference EC concentration range shown in Figure 11 (2-20 µg/cm²).

This good correlation would suggest that the optical cell-phone method could be considered an option for the estimation of BC concentrations in cookstove studies. It is essential to note that an initial calibration of the method with a reference instrument and using EC from local aerosol samples as reference would be an absolute prerequisite for this. Any study attempting to use these methods must previously calibrate them using the same type of aerosol (under the same real-world conditions) as will be targeted.

ANOVA results showed that slopes obtained from cookstove and urban aerosols were statistically different, with a confidence level of 95%. The different slopes obtained result from differences in the physical and chemical composition of the aerosols studied, mainly their mass absorption cross-section (MAC) (Zanatta et al., 2016). The MAC is defined as the light absorption coefficient ($\sigma_{ap}$) divided by elemental carbon mass concentration ($m_{EC}$), and expressed in units of m²/g (Zanatta et al., 2016). Different types of aerosols are characterised by different MAC values, especially when dealing with the combustion of different fuels (Ahmed et al., 2009; Watson et al., 2005). In particular, wood burning emits large amounts of organic gases along with BC aerosols during smoldering (Bond et al., 2007; Petzold et al., 2013; WHO and Scovronik, 2015). The coating of non-absorbing organic (OC) layers on BC particles enhances the absorption (increases the MAC) (Zanatta et al., 2016), resulting in higher BC loads by the optical method (Ahmed et al., 2009).

Furthermore, light-absorbing organic matter, also known as brown carbon (Andreae and Gelencsér, 2006), may have influenced BC measurements. As a reference, in the present study urban aerosol samples had OC loadings between 5.6 and 22.9 µg/cm², and an average OC/EC of 2.45 ($\pm$1.14) (for a representative subset of samples). For biomass aerosols, OC filters loadings were 3.8 - 53.7, and the average OC/EC was 2.61 ($\pm$1.78). Thus, OC/EC ratios were similar for both types of aerosols in this study, suggesting that the relative OC load did not explain the differences in response to optical methods for the urban and biomass aerosols. However, one very relevant limitation is that a number of the biomass aerosol filters were not preserved cold during transport and until analysis, which may have accounted for probable OC losses.

Studies have shown that biomass aerosols have higher light absorption than traffic-generated particles (Salako, 2012), while others had shown an opposite trend (Jeong et al., 2004). The discrepancies could be due to the complex interaction of physical aerosol properties and the filter matrix (Andreae and Gelencsér, 2006).
Another factor possibly affecting the different slopes obtained in Figure 11 could be particle size, given that the cookstove emissions were fresh particles directly emitted by the source, whereas the urban particles underwent a certain ageing during transport from the source (vehicular traffic) to the urban background station. However, further studies would be necessary to evaluate the potential influence of particle size.

To assess the statistical differences between both methods as a function of the aerosol load, the Wilcoxon signed-rank test, a non-parametric statistical hypothesis test recommended for small sample sizes, was applied. Differences were quantified by calculating the absolute relative differences between the cell-phone and the TOT results, for each data point and then averaged (in absolute value) across samples. An analysis of the differences between the TOT and the cell-phone systems at different loadings was carried out in order to understand the potential influence of the filter loading on the optical methods response (Weingartner et al., 2003).

The Wilcoxon signed-rank test indicated that results from both methodologies were, with 95% significance level, slightly different except for urban aerosols with low EC load. Table 2 shows that these differences were not large, and that they varied with aerosol load: larger similarities were observed at higher aerosol loads (with decreasing relative differences, from 43% to 36% in cookstove aerosols, with increasing particle load for biomass aerosols, Table 2). The only exception to this was observed for urban aerosols at low concentrations (non-statistically significant differences). This result highlights the overall need to take into account the aerosol load, as well as the aerosol type, when calibrating the cell-phone method.

Filter loading produces two effects on the optical methods response. On the one hand, the mass loading increases the absorption coefficient (Presser et al., 2014). At the same time, shadowing effects may occur on the filter surface with heavier filter mass loading, resulting in an apparent reduced optical path length through the filter.

<table>
<thead>
<tr>
<th>EC (BC)</th>
<th>Wilcoxon test (p&lt;0.05)</th>
<th>Absolute value relative differences</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Biomass cookstove aerosol</td>
<td>Urban aerosol</td>
</tr>
<tr>
<td>1 to 5 (N=13)</td>
<td>Statistically significantly differences (Z=3.146)</td>
<td>Non statistically significantly differences (Z= 1.470)</td>
</tr>
<tr>
<td>5 to 10 (N=40)</td>
<td>Statistically significantly differences (Z=4.631)</td>
<td>Statistically significantly differences (Z= 5.520)</td>
</tr>
<tr>
<td>10 to 20 (N=20)</td>
<td>Statistically significantly differences (Z=3.864)</td>
<td>Statistically significantly differences (Z=2.097)</td>
</tr>
</tbody>
</table>

Table 2. Wilcoxon test results and absolute value relative difference between results from the Nexleaf system and TOT, as a function of aerosol type and EC load. N =number of samples, Z=test statistic. Source: own elaboration
(Presser et al., 2014; Weingartner et al., 2003) and leading to the underestimation of the measured optical signals with high filter loads.

In general, in this kind of studies (e.g., cookstove studies) measured concentrations remain within a given (high) concentration range. Therefore carrying out the calibrations proposed in this work, tailored to the specific aerosol under study and at the concentrations expected, should be feasible. In our study, because the differences are relatively small (between 36-43%) one single correction equation is proposed.

3.3.2. Reflectance analysis by smoke stain reflectometer compared with EC by thermo-optical analysis

Data are already available in the literature correlating reflectance measurements with EC values (Adeti, P.J, 2012). However, due to the variability in carbonaceous aerosol levels and composition associated with different sources (Hinds, 1999), including fuel-stove combinations, local calibration for the specific type of aerosol under study of reflectance-based EC measurements is still necessary (Johnson et al., 2008a).

By regressing the EC load (µg/cm²) against the reflectance values from the reflectometer two calculation equations were obtained (p<0.05). After following a regression analysis, the best regression model to describe the relationship between RR values and EC was found to be a reciprocal lineal model. The equation to calculate EC from RR values determined with the reflectometer in the case of urban aerosol studied is \(1/RR = 0.0034EC+0.0116\) (\(R^2=0.9\)) and, in the case of biomass cookstove aerosol studied, \(1/RR= 0.0081EC-0.006\) (\(R^2=0.9\)). The results in Figure 12 show a clearly low dispersion of the data points, indicating a high accuracy in the estimation of EC using RR values.
3.3.3. Analysis of filter substrate on BC and EC quantification

To evaluate the influence of the filter substrate on the performance of the optical methods evaluated in this study, 16 paired filters (quartz and glass fibre) were identically loaded using the double-filter system of the LEMS (see Methods section). The BC load for each type of filter substrate was then analysed using reflectometry and cell-phone methods, the only two methods which are able to analyse both quartz and glass fibre filters. The comparison between filter substrates is especially relevant due to the significantly lower cost of glass fibre filters when compared to quartz fibre ones, in addition to their lower dependence on ambient humidity (for mass determination purposes by gravimetry). As a result, glass fibre filters are much more appropriate for resource-constrained sampling locations.

Figure 12. EC load (µg/cm²) determined by TOT plotted against reflectance readings from the smoke stain reflectometer. Source: own elaboration

Figure 13. Comparison of BC concentrations determined by the reflectometer (right) and Nexleaf system (left) on quartz filters and co-located glass fiber filters. Source: own elaboration
Figure 13 shows that optical BC measurements were moderately affected by the filter substrate. For identically-charged filters, reflectance values on glass fibre filters were 7% lower than for quartz fibre filters, with a significantly low dispersion of the data ($R^2=0.99$). Regarding the cell-phone system results, the influence of the filter substrate tended to be larger and with an opposite trend, with BC concentrations on glass fibre filters being approximately 12% higher than those determined on quartz fibre filters. Data dispersion, even being still low, was higher ($R^2=0.94$) than by reflectometry.

There is visual evidence that filter characteristics are different. Glass fibre filters are matted, while quartz fibre ones are fibrous (Figure 15 and Figure 14). Also, they have different pore size. Previous studies have explained that fibrous filters allow particles to become partly or completely embedded in the filter (Bond et al., 1999). The penetration of particles into the filters causes multiple scattering within the filter (Davy et al., 2017; Presser et al., 2014) and may cause filters being more reflective for a given aerosol loading than if particles were retained on their surface (Edwards et al., 1983). This occurs on matted filters, where particles appear to be distributed discretely over the uneven topography of the matted surface (Presser et al., 2014).

Although a more extensive empirical study could be useful to confirm the influence of filter substrate, results suggest that, in addition to taking into account the BC source (fuel and emission process), when calibrating optical methods it is also necessary to consider the filter substrate. ¡Error! No se encuentra el origen de la referencia. summarizes calculation and correction equations presented in Figure 11, Figure 12, and Figure 13.

An external verification of the correlations summarized in ¡Error! No se encuentra el origen de la referencia. would be recommended. In general, calculation and subsequent testing of these correction coefficients against a reference system is recommended for each specific study, given that each specific population is affected by EC and BC emissions from emission processes, which determine the type of aerosol
generated (Bond, 2004) and thus the correction coefficients. Statistical representativeness should also be taken into account.

<table>
<thead>
<tr>
<th>Correction equation (TOT vs Optical method with quartz fibre filters)</th>
<th>Cell-phone system</th>
<th>Smoke stain reflectometer</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Biomass cookstove aerosol</strong></td>
<td>BC = 1.3504 EC + 0.1897 R² = 0.8356</td>
<td>BC = 0.7066 EC + 0.594 R² = 0.8812</td>
</tr>
<tr>
<td><strong>Urban aerosol</strong></td>
<td></td>
<td>1/RR = 0.0081 EC - 0.006 R² = 0.9216</td>
</tr>
<tr>
<td><strong>Biomass cookstove aerosol</strong></td>
<td></td>
<td>1/RR = 0.0034 EC + 0.0116 R² = 0.9293</td>
</tr>
<tr>
<td><strong>Urban aerosol</strong></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Filter material adjustment (quartz-glass fibre)</th>
<th>Cell-phone system</th>
<th>Smoke stain reflectometer</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>BC_{glass} = 1.1177 BC_{quartz} + 0.3463 R² = 0.9541</strong></td>
<td></td>
<td><strong>RR_{glass} = 0.9346 RR_{quartz} - 1.1783 R² = 0.9894</strong></td>
</tr>
</tbody>
</table>

Table 3. Summary of correction equations obtained as a function of analytical method, aerosol and filter type. BC and EC are expressed in μg/cm²; RR=Reflectance readings in %. Source: own elaboration.
4. Quantification of carbonaceous aerosol emissions from cookstoves in Senegal
4.1. Background

As previously discussed in section 2.2, cooking-related aerosols have been much less studied than those of other common sources (Roden et al., 2006; Soneja et al., 2015), leading to an important underestimation of climate impacts from traditional burning of biomass and a lack of knowledge of emission effects due to the switch from traditional to improved cookstoves (Grieshop et al., 2011; Lee and Chandler, 2013).

Some studies determining BC (EC) and OC EFs from cooking technologies are available for the Asian Region (Arora and Jain, 2015; Chen et al., 2012; Guofeng et al., 2012; Li et al., 2009; Shen et al., 2013, 2010; Venkataraman et al., 2005; Zhang et al., 2000); and some contributions have been made from Latin America (Johnson et al., 2008b; Roden et al., 2009, 2006). However, only one study has performed in sub-Saharan Africa (Malawi) (Wathore et al., 2017) and, to our knowledge, there are no studies regarding the characterization of aerosol emissions from cooking in the West African Region.

This chapter presents a laboratory characterization of aerosol emissions from four stove-biofuel combinations widely used in Senegal, where 83% percent of households depend on biomass fuels to cover their daily cooking energy needs (WB, 2013). Two of these stove-fuel combinations were, in addition, tested in a rural village of Senegal under uncontrolled conditions to understand how real cooking practices in this region influence carbonaceous aerosol emissions. Finally, total carbonaceous aerosol emissions and the net climate effect from fuelwood burning at household level in Senegal and West Africa were estimated.

A better knowledge of aerosol emissions from residential biofuels in West Africa should contribute to reduce the uncertainties on emission inventories and climate prediction models at regional level, and should derive in a more representative and robust estimation of the climatic impacts of this source. Moreover, this information should help improved cookstove initiatives to be considered within the national and regional SLCP’s Reduction Strategies, for addressing both near-term and long-term climate change impact and, simultaneously, promoting health and social benefits (Shoemaker et al., 2013).

4.2. Methods

4.2.1. Laboratory sampling

Laboratory tests were conducted at the Centre de Études et Recherches sur les Énergies Renouvelables (CERER) of the University of Dakar, using a LEMS. Three replicates of WBT were conducted to test each cooking system. Both LEMS and WBT were already explained in detail in section 3.2.1.
4.2.2. Field sampling

The study was conducted in March 2016 in Bibane, a rural village with approximately 650 inhabitants (74 families) in the commune of Niakhar, in the region of Fatick (Senegal) (Figure 16). Typical home structures are made of earth bricks and thatched roof. The kitchen is physically separated from the bedroom area, and very poorly ventilated (Figure 16). This type of household structure is very common in rural areas of Senegal and other Sub-Saharan West African countries (Ochieng et al., 2013).

![Figure 16. Typical kitchen in Bibane, rural village in the region of Fatick, Senegal. Source: own elaboration.](image)

In Bibane, like most rural sub-Saharan villages, women are responsible for cooking, as well as for the rest of the household chores. Cooking is among the most time-consuming of women’s responsibilities (Wodon and Blackden, 2006). They spend much of their time near the stove and performing other tasks related to food preparation, such as pounding the millet by hand, fetching water, etc. They usually cook inside the kitchen, to protect the food and fire from animals, wind and children, and to be protected from the sun. Fuelwood is scarce in Bibane, so its collection requires long journeys to the forest. As a result, fuel used to cook is heterogeneous, with a high percentage of families frequently using crop residues and dung.

All of the families in the village used traditional three-stone fires before improved cook stoves were introduced in 2012, within the framework of a program to optimize the use of forest resources in the region. In total, 2,500 improved cookstoves (known as Noflaye Jegg) were distributed in the region, 50 of them in Bibane. However, the research team discovered that the majority of the improved cookstoves were in disuse, completely or partially damaged, and only 15 out of 50 were in a good or fair state, and considered adequate to be included in the study.

Before starting measurements, a communal meeting was conducted to explain the study purpose and activities, where women provided oral consent to enrol in the study. Finally, the 15 households with improved cookstoves were included, together with other 14 using the traditional three stone fire.
Emissions testing was done using a Portable Emission monitoring system (PEMS), with the same design of the LEMS (section 3.2.1), but with a portable hood made from fireproof fabric. The portable hood was placed before starting the meal preparation directly over the centre of the combustion zone (see Figure 17) at 50-80 cm above the cooking surface, to allow cookers to have an acceptable distance for the stove use. Background concentrations were sampled and then subtracted from the emission sample concentrations to account for background contributions to emissions samples.

![Portable Emission Monitoring System (PEMS), designed by the Aprovecho Research Centre. Source: own elaboration.](image)

Emission sampling was started once the pot was put over the cookstove and start time was recorded, including the ignition phase emissions. At the end of the food preparation stage, emissions sampling was terminated, cooked food weighed and end time recorded using a digital scale of 0–30 kg range with 1 g resolution. Fuelwood was measured before and after cooking sessions, as well as the charcoal formed. The goal was to minimize interference with normal cooking practices, so the food prepared during the tests was freely chosen by families.

An important limitation of the PEMS sampling in remote areas where lack of electrical power. We connected it to a diesel generator, placed far away enough to not disturb women while cooking and to avoid emissions interferences.

### 4.2.3. Stoves and fuels

Four stoves were selected between the most commonly used in rural Senegal, each one representing a method of fuelwood combustion: i) Three-stone (open burning...
traditional stove); ii) Noflaye Jegg (rocket combustion chamber-type locally produced); iii) Jambaar bois (basic improved ceramic stove, widely distributed in the Dakar region by the German cooperation PERACOD program); iv) Prime Square Stove (a top-lit updraft gasifier, TLUD, with a natural draft, with no fan). Photos of the stoves are shown in Figure 18.

Each stove was analysed in the laboratory using two wood species very commonly used in Senegal as residential fuel for cooking: Casuarina Equisetifolia (common named Filao) and Cordyla Pinnata (common named Dimb) (Ndiaye et al., 2015).

Both were split in regular pieces of 15x5x3cm. Moisture content in wet basis for dimb and filao was 8% and 8.8%, respectively, and calorific value was 21.79 MJ/Kg and 19.0 MJ/Kg, respectively.

Three stones and Noflaye Jegg were the stove types analysed in the field. Three stones is still the most commonly used biomass stove in Senegal, and Noflaye Jegg stove has been widely implemented in the Casamance Region, (25,000 units between 2012 and 2016, (Sota et al., 2014) and in Fatick (2500 units in 2012). Rocket stove design is the most commonly used type of improved cookstove in sub-Saharan Africa (Lambe et al., 2015b).

As fuel used in Bibane was quite heterogeneous due to fuelwood scarcity, the same wood was distributed to every family participating in the study. The test performance with uniform wood type allowed to reduce variability between households, and thus the sample size needed to have statistically significant results, and to focus the study in the effect of stove type on carbonaceous aerosol emissions. Wood used during field testing (dimb) was purchased to a local vendor locally, split in bigger and more irregular pieces than those used in the laboratory, and distributed to households. Purchased dimb had a moisture content of 5% on a wet basis and 21.69 MJ/Kg of calorific value.
4.2.4. EC and OC analysis

After sampling, particle-loaded filters were packed with aluminium foil and stored in a freezer until analysis. Gravimetric measurements of glass fibre filters were conducted using a high precision digital balance at the CERER, and OC/EC analyses on quartz filters were conducted on a Sunset Laboratory Carbon Analyser using the EUSAAR2 Protocol (see section 3.2.2.) in the Institute of Environmental Assessment and Water Research (IDÆA-CSIC) in Barcelona. The Sunset Laboratory carbon Analyser was chosen between the methods presented in chapter 2 because it is the most accurate method (see Table 1), and because it is the only one which can analyse OC content. For this reason, results presented in this chapter are termed as EC.

4.2.5. Data Analysis

Kruskal-Wallis one-way analysis of variance was used to identify significant differences of OC and EC EFs with different cookstoves, fuels, and during different WBT phase. Wilcoxon t-test was conducted in order to identify pairs with statistically significant difference in EF values. Non parametric tests were used because observations were not normally distributed, due to the relative small sample size and the presence of outliers (mainly in results from the field study).

The inter quartile range (IQR), difference between the 75th percentile and the 25th percentile, was used to determine the test-to-test variability. This measure is robust against outliers and non-normal data. All statistical analyse were conducted at a significance level of 0.05 and performed using IBM SPSS Statistics Base software and STATGRAPHICS (StatPoint Technologies, Inc., 2014).

4.3. Results and discussion

4.3.1. Laboratory estimation of EC and OC emission using the WBT

Figure 19 presents EC EFs (in grams per MJ, for comparison on the same scale), total fuel consumption per WBT, and EC total emissions per WBT completed.

Results on Figure 19a show that the type of stove had an effect on EC EFs. The highest average EC EF per WBT was found with the Noflaye Jegg for both fuels (dimb and filao), while EC EFs were fairly uniform across the other two improved cookstoves and the three stones.

Differences of EC EF’s between stoves are related to the nature of combustion (Arora and Jain, 2015; MacCarty et al., 2007a), which is, in turn, affected by a number of factors (e.g. air-fuel ratio, burn rate, combustion temperature, combustion efficiency, thermal efficiency, residence time in the combustion chamber, flame turbulence, etc.) (MacCarty et al., 2008b; Venkataraman et al., 2005).
In this study, the CO/CO$_2$ ratio was used as a benchmark for stove combustion efficiency (Guofeng et al., 2012; Li et al., 2009) (Table 4). The Noflaye Jegg performed with the lowest CO/CO$_2$ ratios (3.4±1.1% for dimb and 5.3±0.1% for filao), indicating efficient combustion, and the highest EC emissions (0.18±0.06 g/MJ and 0.06±0.01 g/MJ) for dimb and filao, respectively.

<table>
<thead>
<tr>
<th>CO/CO$_2$ ratio (%)</th>
<th>Noflaye Jegg</th>
<th>Three stones</th>
<th>Jambaar bois</th>
<th>Gasifier</th>
</tr>
</thead>
<tbody>
<tr>
<td>Dimb</td>
<td>3.4±1.1%</td>
<td>8.7±0.5%</td>
<td>7.8±2.4%</td>
<td>3.6±0.6%</td>
</tr>
<tr>
<td>Filao</td>
<td>5.3±0.1%</td>
<td>8.3±1.8%</td>
<td>6.7±1.9 %</td>
<td>4.0±0.4%</td>
</tr>
</tbody>
</table>

Table 4. CO/CO$_2$ ratio (expressed in %) for the stove-fuel combinations analysed in the laboratory. Source: own elaboration

Rocket stoves have a strong draft which enhances air flow through the fire, improving the combustion efficiency and resulting in higher combustion temperatures (Li et al., 2009; MacCarty et al., 2008b), which typically results in higher EC emissions (Bond, 2004; Cachier et al., 1996; MacCarty et al., 2008a; Rau, 1989).
The three stones showed the highest CO/CO\(_2\) ratios (8.7±0.5\% with dimb and 8.3±1.8\% with filao), indicating inefficient combustion, and low values of EC EF (0.09±0.01 g/MJ for dimb and 0.04±0.01 g/MJ for filao).

Jambaar bois showed very similar values to the three stones fire, both in combustion efficiency and EC EFs values. CO/CO\(_2\) ratio was 7.8±2.4\% and 6.7±1.9 \%; and EC EFs were 0.09±0.01 g/MJ and 0.05±0.01 g/MJ, with dimb and filao, respectively. Previous studies have also found EFs for basic ceramic stove similar to traditional stoves (Wathore et al., 2017). Table 6 summarises EC and OC EFs determined in previous laboratory and field studies.

The gasifier showed the lowest CO/CO\(_2\) ratio (i.e. high efficiency), 3.6±0.6\% and 4.0±0.4\%, with dimb and filao, respectively; and low EC EF’s: 0.09±0.02 g/MJ when burning dimb (equal to EF of jambaar bois and three stones), and 0.05±0.01 g/MJ with filao. This is coherent with previous findings showing the lowest EF for gasifiers in comparison with traditional and other basic improved stoves (Arora and Jain, 2015; MacCarty et al., 2008b; Wathore et al., 2017), (Table 6).

In addition to the cookstove technology, the biofuel characteristics also determine combustion conditions, and thus, aerosol emissions (Li et al., 2009; Venkataraman et al., 2005; Venkataraman and Rao, 2001). In this study, EC EF’s of all the stoves tested showed higher EC EF’s when burning dimb over filao (64.2\%, 51.7\%, 49.7\% and 56.7\% for the Noflaye Jegg, Jambaar bois, gasifier and three-stones, respectively), although not statistically significant at 95\% confidence level.

A number of characteristics of fuel wood could affect pollutant emissions: i) the geometrical shape and size of the fuel; ii) the density (Atiku et al., 2016; Mitchell et al., 2016); iii) the lignin content (Atiku et al., 2016); (v) the heat value and (vi) the moisture (Shen et al., 2010), among others.

Dimb and filao used in this study had similar moisture fraction (8\% and 8.8\%, on a wet basis) and calorific value (21.79 MJ/Kg and 19.0 MJ/Kg), and both were split in stick of 15x3x4 cm. Typical lignin content is around 26\% for casuarina esquetiofila (Filao) (Mahmood, 1993), and 40\% for cordyla pinnata (dimb) (Samba, 2001). Previous studies show that lignin content promotes elemental carbon formation (Atiku et al., 2016; Tillman, 1987), so this may partially explain the higher EC EF’s of wood dimb.

Results suggest that emission factors of EC are dependent on the type of fuel burned, but further studies could study the effect of physical and chemical characteristics of filao and dimb in EC EF’s, which cannot be explained at this stage.

Figure 19b presents EC EFs for the stove-fuel combinations by WBT phase, except in the case of the gasifier, where only the EF of the whole WBT is presented, because its cylindrical combustion chamber cannot be emptied during the burning cycle. EC EFs
were found to be higher during the cold start phase for every stove-fuel combination analysed. This can be explained because cold start phase is characterized by strong flames and significant amounts of elemental carbon (Atiku et al., 2016; Roden et al., 2009), especially shortly after fire ignition (Roden et al., 2006), whereas the simmer phase is characterized by smaller flames, with lower emission of EC particles (Roden et al., 2006).

EC EFs were 20%, 10%, 29% higher for the cold start phase with regard to simmer, for the dimb with three stone, jambaar bois and noflaye jegg, respectively; and 25%, 20% and 14% higher for filao with three stone, Jambaar Bois and Noflaye Jegg, though not statistically significant (p>0.05). Previous studies also found that changes in combustion conditions during the cooking cycle induced variation in EC EFs (Arora and Jain, 2015; Jetter et al., 2012; Roden et al., 2006).

In terms of total EC emission per WBT completed, Figure 19d shows that results followed almost the same trend as in the case of average EC EFs per WBT. Noflaye Jegg showed the highest EC EF and did not show a reduction in fuel wood consumption per task completed with respect to the three stones fire (Figure 19c). Therefore, this type of stove has the highest EC emission per WBT (3.75 g/WBT with dimb and 1.90 g/WBT with filao). On the other side, the gasifier showed the highest fuel savings and low EC EFs, so this type of stove induced to the smallest total EC emissions per WBT (1.35 g/WBT with dimb and 0.57 g/WBT with filao). An intermediate situation was found for the three stones and the Jambaar bois, which show very similar results in terms of EC total emissions (three stones, 2.00 g/WBT with dimb and 0.67 g/WBT with filao and Jambaar bois (1.94 g/WBT with dimb and 0.89 g/WBT with filao).

Figure 20 presents OC EFs for three stones+filao, jambaar bois+filao, gasifier+filao and gasifier+dimb. OC EFs for the rest of stove-fuel combinations are not presented due to problems in filter conservations between WBT and EC/OC analysis.

When burning wood filao, the average OC EF for the whole WBT was found to be the lowest for the gasifier (0.08±0.01 g OC/MJ), followed by three stones (0.18±0.03 g OC/MJ) and then Jambaar bois (0.21±0.08 g OC/MJ). Low OC and EC emissions of gasifier are explained because this technology emits very low levels of particulate matter when compared to other biomass stoves (MacCarty et al., 2008b). As already observed in previous studies, in the case of the three-stone fire and jambaar bois, a large bed of charcoal under the flaming fuel resulted in smoldering conditions, with high OC particles emissions (MacCarty et al., 2007a). Overall, the EF OC results appear to be consistent with prior studies (Table 6).
Phases that were associated with higher OC emissions were different depending on the cookstove tested, as occurred in previous studies (Arora and Jain, 2015). For the traditional stove, OC EF was found to be higher during simmer (0.19±0.05 g OC/MJ) than during cold start (0.16±0.03 g OC/MJ), whereas Jambaar bois showed highest OC EF during cold start (0.22±0.09 g OC/MJ) when compared to simmer phase (0.19±0.05 g OC/MJ), although none of these differences between WBT phases were statistically significant (Wilcoxon, p<0.05).

The higher OC emissions during simmer phase can be attributed to high-carbon content and large surface area per unit mass of charcoal, that results in the formation of organic compounds (Akagi et al., 2010). On the other hand, higher OC emissions during flaming conditions (cold start phase) with the Jambaar Bois, could be attributed to low flame temperatures due to heat losses through the cookstove walls (Arora and Jain, 2015).

Regarding the influence of fuelwood species on OC EFs, the gasifier showed higher OC EFs when burning dimb (0.46±0.05 g OC/MJ) than with filao (0.08±0.01 g OC/MJ). Higher OC emissions with dimb might be related with the ash content of this fuelwood specie (Guofeng et al., 2012), but further chemical composition analysis should be
done to verify this assumption Therefore, gasifier results would suggest that OC EF’s are dependent on the type of fuel burned, although emission tests with more types of stove should be performed.

Figure 20d presents results of OC/EC ratio, considered an environmentally relevant property with good predictive capabilities, even for fuels in which brown carbon absorption is significant (Pokhrel et al., 2015). Overall, all stove fuel combinations showed OC/EC ratios higher than one, indicating the dominance of OC and suggesting that the smoldering combustion was dominant during the WBT.

Figures 19d and 20c show that OC and EC total emissions had a relatively large degree of test-to-test variability, and boxes overlapped considerably. For this reason, differences in OC and EC EFs across stove types, phase of WBT and type of fuel were not found to be statistically significant in most cases (p>0.05).

In terms of EC total emissions, the highest variability was observed for Noflaye Jegg (IQR=2.8 with dimb and IQR= 1.03 with filao). This is because flaming conditions are more present in the cooking cycle for this stove, characterized by more frequent peaks of emissions, which significantly affect total EC. For OC total emissions, Jambaar bois showed the highest IQR, equal to 5.8.

In this study three replicates of WBT were conducted because it was commonly used in the literature (Wang et al., 2014) and because of time and resource limitations. However, number of replicates is linked to the stove-fuel combination being tested and sometimes three are not enough to ensure confidence in the results (Wang et al., 2014). These findings suggest that in forthcoming emissions testing of the stoves included in this study, a larger number of WBT replicates, especially for the higher emitters such as the rocket stove, should be analysed.

Table 5 summarizes EC and OC emission factors, EC/OC ratios and total emissions per WBT from the stove-fuel combinations tested under laboratory.
<table>
<thead>
<tr>
<th>Stove type</th>
<th>Fuel type</th>
<th>EC (g/MJ)</th>
<th>EC (g/kg)</th>
<th>OC (g/MJ)</th>
<th>OC (g/kg)</th>
<th>OC (g/L of water)</th>
<th>OC/EC</th>
<th>Fuel consumed per WBT</th>
<th>Total BC emitted per WBT</th>
<th>Total OC emitted per WBT</th>
</tr>
</thead>
<tbody>
<tr>
<td>Three stones</td>
<td>Dimb</td>
<td>0.09±0.01</td>
<td>1.98±0.17</td>
<td>0.20±0.01</td>
<td>No data</td>
<td>No data</td>
<td>No data</td>
<td>1.00±0.02</td>
<td>2.00±0.20</td>
<td>No data</td>
</tr>
<tr>
<td>Filao</td>
<td>0.04±0.01</td>
<td>0.75±0.20</td>
<td>0.07±0.02</td>
<td>0.18±0.03</td>
<td>3.33±0.61</td>
<td>0.29±0.05</td>
<td>4.67±0.85</td>
<td>0.89±0.02</td>
<td>0.67±0.16</td>
<td>2.92±0.46</td>
</tr>
<tr>
<td>Jambaar bois</td>
<td>Dimb</td>
<td>0.09±0.01</td>
<td>2.05±0.16</td>
<td>0.19±0.02</td>
<td>No data</td>
<td>No data</td>
<td>No data</td>
<td>0.94±0.10</td>
<td>1.94±0.34</td>
<td>No data</td>
</tr>
<tr>
<td>Filao</td>
<td>0.05±0.01</td>
<td>0.87±0.23</td>
<td>0.08±0.02</td>
<td>0.21±0.08</td>
<td>4.04±1.50</td>
<td>0.42±0.28</td>
<td>4.61±0.40</td>
<td>0.98±0.09</td>
<td>0.89±0.49</td>
<td>4.22±2.80</td>
</tr>
<tr>
<td>Noflaye Jegg</td>
<td>Dimb</td>
<td>0.18±0.06</td>
<td>3.93±1.20</td>
<td>0.36±0.11</td>
<td>No data</td>
<td>No data</td>
<td>No data</td>
<td>0.96±0.06</td>
<td>3.75±1.37</td>
<td>No data</td>
</tr>
<tr>
<td>Filao</td>
<td>0.06±0.01</td>
<td>1.23±0.20</td>
<td>0.19±0.02</td>
<td>No data</td>
<td>No data</td>
<td>No data</td>
<td>No data</td>
<td>1.63±0.12</td>
<td>1.90±0.31</td>
<td>No data</td>
</tr>
<tr>
<td>Gasifier</td>
<td>Dimb</td>
<td>0.09±0.02</td>
<td>1.95±0.52</td>
<td>0.27±0.07</td>
<td>0.46±0.05</td>
<td>9.91±1.11</td>
<td>1.38±0.15</td>
<td>5.40±1.71</td>
<td>0.70±0.02</td>
<td>1.35±0.35</td>
</tr>
<tr>
<td>Filao</td>
<td>0.05±0.01</td>
<td>0.86±0.18</td>
<td>0.11±0.02</td>
<td>0.08±0.01</td>
<td>1.49±0.28</td>
<td>0.20±0.04</td>
<td>1.76±0.24</td>
<td>0.67±0.01</td>
<td>0.57±0.12</td>
<td>0.63±0.48</td>
</tr>
</tbody>
</table>

Table 5. Summary of EC and OC emission factors, EC/OC ratios and total emissions per WBT from the stove-fuel combinations tested in the laboratory
4.3.2. Field estimation of EC and OC emissions

Emission tests were conducted in 29 households in the rural village of Bibane: 15 with the Noflaye Jegg and 14 with the three-stone fire. Most families cooked inside the kitchen space during the tests (separated from the rest of the house and very poorly ventilated), although few of them cooked outdoors.

All families prepared typical meals of rural households in the region. They consisted of rice with vegetables, fish, or meat for lunch, whereas for dinner water was boiled to steam cous-cous and a sauce was prepared. Time of preparation ranged from 38 to 111 minutes (average=75 minutes), with no significant differences between lunch and dinner and between the type of stove used. Likewise, no differences were found in the quantity of fuel used per kg of food prepared between the three stones and the Noflaye Jegg (0.6 ±0.4 kg of fuel/kg food), consistent with previous laboratory results.

As fuelwood type was the same for every household, field results focused on the analysis of the influence of stove type on EC, OC and PM emissions. PM EFs (Figure 21a) and OC EFs (Figure 21c) were 30.3% and 35.7% lower for Noflaye Jegg when compared to the three stones, although this difference was not statistically significant (p>0.05). On the other hand, EC EFs (Figure 21b), increased a 75% (p<0.05) with Noflaye Jegg in comparison with the three stones stove. The ratio OC/EC (Figure 21d) was 1.77±0.56 for the three stones, indicating the dominance of OC, which is caused by the smoldering combustion (Arora et al., 2013). OC/EC ratio < 1 (0.67±0.28) for the Noflaye Jegg is showing the dominance of flaming conditions, typical in the rocket stoves (MacCarty et al., 2008a)
Therefore, while the Noflaye Jegg reduces PM EF, the lower OC/EC ratio implies that carbonaceous aerosol emissions with this stove are more warming (from a climate perspective) when compared to the three stones, which could lessen some of the impact of reducing overall aerosol emissions (Johnson et al., 2008b). However, the comparison of both stoves should be done in terms of total EC and OC emissions, taking into account their fuel-efficiency (section 4.3.4). Overall, EC and OC EFs obtained agree well with previous studies (Table 6), although many differences were observed due to differences on stove type, fuel parameters, operation conditions, etc. Nonetheless, most publications present EF in g/kg instead of g/MJ, which prevents comparison between different fuel types at same level. Table 7 presents a summary of EF’s, ratios and total emissions from in-field emission testing for each household included in the study.
<table>
<thead>
<tr>
<th>Type of Study</th>
<th>Country</th>
<th>Type of stove and fuel</th>
<th>EC EF (g/Kg)</th>
<th>OC EF (g/Kg)</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>Field UCT and laboratory WBT</td>
<td>Senegal</td>
<td>Fuel: wood (Cordyla Pinnata and Casuarina Equisetifolia)</td>
<td></td>
<td></td>
<td>This study</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Stoves:</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>- Three stones (lab WBT)</td>
<td>(a) 1.98±0.17,</td>
<td>(a) 3.33±0.61</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>- Rocket stove (lab WBT)</td>
<td>b 0.75±0.20</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>- Ceramic improved stove (lab WBT)</td>
<td>(b) 3.93±1.20,</td>
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Table 6. Comparison of Measured EC and OC EF for fuelwood combustion with previous studies. Source: own elaboration.
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<td>10T</td>
<td>Three stones</td>
<td>Sauce</td>
<td>Dimb</td>
<td>95,6</td>
<td>2,2</td>
<td>4,1</td>
<td>0,5</td>
<td>0,30</td>
<td>6,55</td>
<td>0,08</td>
<td>1,64</td>
<td>0,14</td>
<td>2,96</td>
<td>1,81</td>
<td>0,25</td>
<td>0,45</td>
</tr>
<tr>
<td>11T</td>
<td>Three stones</td>
<td>Rice</td>
<td>Dimb</td>
<td>64,3</td>
<td>2,0</td>
<td>3,6</td>
<td>0,5</td>
<td>0,40</td>
<td>8,66</td>
<td>0,04</td>
<td>0,96</td>
<td>0,13</td>
<td>2,73</td>
<td>2,83</td>
<td>0,11</td>
<td>0,31</td>
</tr>
<tr>
<td>12T</td>
<td>Three stones</td>
<td>Rice</td>
<td>Dimb</td>
<td>65,8</td>
<td>1,6</td>
<td>8,7</td>
<td>0,2</td>
<td>0,28</td>
<td>6,09</td>
<td>0,08</td>
<td>1,80</td>
<td>0,13</td>
<td>2,77</td>
<td>1,53</td>
<td>0,30</td>
<td>0,45</td>
</tr>
<tr>
<td>13T</td>
<td>Three stones</td>
<td>Sauce</td>
<td>Dimb</td>
<td>101,1</td>
<td>4,1</td>
<td>2,1</td>
<td>1,9</td>
<td>0,46</td>
<td>9,96</td>
<td>0,11</td>
<td>2,38</td>
<td>0,21</td>
<td>4,58</td>
<td>1,92</td>
<td>0,24</td>
<td>0,46</td>
</tr>
<tr>
<td>14T</td>
<td>Three stones</td>
<td>Couscous</td>
<td>Dimb</td>
<td>156,2</td>
<td>5,5</td>
<td>12,2</td>
<td>0,11</td>
<td>2,47</td>
<td>0,03</td>
<td>0,72</td>
<td>0,04</td>
<td>0,94</td>
<td>1,31</td>
<td>0,29</td>
<td>0,38</td>
<td></td>
</tr>
</tbody>
</table>

Table 7. Summary of EC, OC and PM Emission Factors, ratios and total emissions from in-field emission testing for each household included in the study. Source: own elaboration
### 4.3.3. Laboratory vs. field results

EC EFs from WBT were compared with results obtained under real world conditions for the three stones and the Noflaye Jegg stoves when burning dimb (Figure 22. Boxplots of EC EF (g/MJ) for each type of stove and test. Source: own elaboration.).

EC EFs estimated at laboratory scale with the WBT were 18.0% higher than EF determined during daily cooking activities for the Noflaye Jegg, and 14.5% higher in the case of the three stones (not significant, p-value>0.05). Previous studies also found differences between results from laboratory tests and those obtained during uncontrolled field conditions (Arora and Jain, 2015; Chen et al., 2012; Johnson et al., 2008a; Roden et al., 2009, 2006; Shen et al., 2013), although differences found in these studies were higher than in this study. Further studies with a higher number of laboratory and field repetitions would be necessary to obtain a more representative comparison.

Differences between laboratory and field results can be explained by several factors. Firstly, the fire management behaviour in the laboratory was essentially normalized; cooking fire was constantly tended, and fuel was added more frequently, whereas the process observed in the field was rather random, including some low combustion efficiency situations. On the other hand, wood pieces used in the laboratory had regular shape and dimensions, while in the field they were less uniform in shape and with larger dimensions. Smaller wood has been found to yield lower PM emissions and a higher BC fraction emissions (Bond, 2004) (Li et al., 2009). Finally, minor fugitive emissions may have escaped through the fabric hood during field testing.

Differences were lower when comparing EF from field tests with those obtained during the simmer phase of WBT: 3.8% for the three stones and 4.2% for the Noflaye Jegg (p>0.05) (Table 8).
<table>
<thead>
<tr>
<th>Stove-fuel</th>
<th>WBT</th>
<th>Field test</th>
<th>% difference*</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Cold start</td>
<td>Simmer</td>
<td>Average WBT</td>
</tr>
<tr>
<td>Three stone-dimb</td>
<td>0.10±0.01</td>
<td>0.08±0.01</td>
<td>0.09±0.01</td>
</tr>
<tr>
<td>Noflaye Jegg-dimb</td>
<td>0.21±0.07</td>
<td>0.15±0.04</td>
<td>0.18±0.06</td>
</tr>
</tbody>
</table>

Table 8. EC EFs and % difference between laboratory and field data sets. *differences were not significant (p>0.05). Source: own elaboration

This is coherent with field observations during testing activities, which showed that during meal preparations (rice and cous-cous), cookers maintained a constant and low-medium flame during almost all the time, except when frying vegetables, fish or meat for the first few minutes, which needed stronger flames.

Considering that, although user practices are highly individual, they may have regional averages governed by common customs (Chen et al., 2012), simmer phase of WBT is likely quite illustrative of EC emissions in homes of rural villages of Senegal. Further field measurements are needed as a reality check for each stove-testing and region.

4.3.4. Estimation of total emissions and net climate effect of cooking-related carbonaceous aerosol in Senegal

Total annual EC and OC emission per year of use of the Noflaye Jegg and the three stones were calculated with EF values and data of daily consumption (9.9±5.1 kg/day for the Noflaye Jegg and 9.6±5.5 Kg/day and for the three stones), both determined during the field study. As in this region cooking activities occur most often in indoor environments, it was necessary to take into account the fraction of emissions released to ambient air (exfiltration rate), which is dependent on the ventilation and particle deposition rates in kitchens (Soneja et al., 2015). Exfiltration rate for typical household characteristics of this study was not available, so total emissions have been determined using two values of exfiltration rate from previous studies (26% and 80%) (Soneja et al., 2015 and Venkataraman et al., 2005, respectively). Annual EC and OC emissions per stove were then factored by the Global Warming Potential (GWP) recommended by the IPCC, 900 for EC (Bond et al., 2013) and -46 for OC (Bond et al., 2011), for a time horizon of 100 years. Figure 23a presents the net climatic effect estimated with regard to EC and OC emissions for the three stones and Noflaye Jegg (in tons of CO2-eq per year).
Across a 100-year horizon, Noflaye Jegg showed an overall increase of 88% global warming potential in comparison to the traditional stove. In consequence, while the use of Noflaye Jegg has clear benefits for improving indoor air quality (Sota et al., 2014 and chapter 5 of this thesis), results suggest that this type of rocket stove, under typical cooking practices and fuelwood used Senegal, leads to negative climate effects with regard to carbonaceous aerosol emissions.

The fact that rocket and other types of natural draft stoves increase EC EFs is not new (Arora and Jain, 2015; MacCarty et al., 2008a). The unexpected point is that Noflaye Jegg stove did not provide fuel saving benefits, neither in the laboratory study, nor in the field. For this reason, this work emphasizes the importance of taking into account total EC and OC emissions in addition to EFs.

Besides this, total annual emissions of EC and OC from residential burning of fuel wood in Senegal were estimated using the average value of EC and OC EFs determined in this study (1.85 g EC/kg and 3.97 g OC/kg) and the total annual firewood used in Senegalese households (450 ktoe/year, for the period 2000-2005) (Dafraallah, 2009).

As previously discussed, EC and OC EFs vary significantly across different types of stove-fuel combinations. However, average EFs from this study could be considered as representative, as they were estimated from a set of biomass cookstoves and wood fuels widely used in Senegal. Fuel consumption is also highly dependent on type of stove used, but data on the fractions of different stoves used in Senegal are unavailable, so the total annually firewood used in Senegalese households was the best information available.

Total emissions of EC and OC from residential fuelwood combustion in Senegal were estimated to be 421.1±213.0 and 561.4±411.4 t/year, respectively, with 26% of exfiltration rate, and 1295.6±655.3 t/year and 1727.5±1265.8 t/year with 80% of

Figure 23. Figure 23a: 100 years Global Warming Potential (from Elemental carbon and Organic Carbon emitted) per year of use of Noflaye Jeg stove and Three stones; Figure23b: 100 years Global Warming Potential (from Elemental carbon and Organic Carbon emitted) per year from household fuelwood cookstoves in Senegal. Results are presented for a rate exfiltration of 26% and 80%. Source: own elaboration.
exfiltration rate (Table 9). Factoring by GWP$\text{100}$ values, net effect of EC and OC emissions from household fuelwood consumption in Senegal were 353.1 kton of CO$_2$-eq with 26% of exfiltration rate and 1086.6 ton of CO$_2$-eq with 80% of exfiltration (Figure 23b).

Using the 5th percentile of the EC and OC EFs data set and 26% of exfiltration rate, the result was 173.4 ton of CO$_2$-eq, and, when using the 95th percentile of EC and OC EFs and 80% of exfiltration rate, calculations show a result of 2031.9 ton of CO$_2$-eq. This range (173.4-2031.9 ton of CO$_2$-eq) constitutes a rough estimation of the net effect on climate of carbonaceous aerosols emitted into the atmosphere from household use of fuelwood for cooking in Senegal.

It is important to highlight that a complete assessment of the global warming potential should include other species co-emitted from residential cookstove use, such as SO$_2$, NO$_x$, CO, CO$_2$, CH$_4$, N$_2$O (Bhattacharya et al., 2002; Roden et al., 2006; Smith et al., 2000). However, the objective at this stage was to assess the net climate effect of cooking-related carbonaceous aerosol emissions in Senegal, which is still uncertain.

Total EC and OC emissions from fuelwood consumption in Sub-Saharan and West Africa were also calculated with the same EF and exfiltration rates used for Senegal, and using data on personal consumption of energy for cooking in Sub-Saharan and West Africa (IEA, 2006; OECD/IEA, 2014). Results presented in Table 9, together with results of EC and OC emissions from other regions of the world where residential biofuel combustion is prevalent, show that carbonaceous emissions from cooking in Sub-Saharan Africa are relevant and should be taken into account within national and regional climate mitigation strategies.

<table>
<thead>
<tr>
<th>Base year</th>
<th>Fuel Type</th>
<th>Region</th>
<th>EC emissions (Gg/year)</th>
<th>OC emission (Gg/year)</th>
<th>Study</th>
</tr>
</thead>
<tbody>
<tr>
<td>2000-05</td>
<td>Fuelwood</td>
<td>Senegal</td>
<td>26%: 0.4 (0.2-0.8)</td>
<td>80%: 1.3 (0.6-2.4)</td>
<td>This study</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>26%: 0.6 (0.2-1.1)</td>
<td>80%: 1.7 (0.6-3.3)</td>
<td></td>
</tr>
<tr>
<td>2005</td>
<td>Biomass$^c$</td>
<td>sub-Saharan Africa</td>
<td>26%: 190.6 (92.0-357.1)</td>
<td>80%: 586.5 (283.1-1098.8)</td>
<td>This study</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>26%: 254.1 (94.2-488.6)</td>
<td>80%: 782.0 (289.9-1503.3)</td>
<td></td>
</tr>
<tr>
<td>2012</td>
<td>Fuelwood$^d$</td>
<td>West Africa</td>
<td>26%: 118.8 (51.3-229.9)</td>
<td>80%: 365.6 (157.8-707.4)</td>
<td>This study</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>26%: 254.9 (104.0-542.7)</td>
<td>80%: 784.5 (320.1-1669.7)</td>
<td></td>
</tr>
<tr>
<td>2007</td>
<td>Fuelwood</td>
<td>Rural China</td>
<td>92</td>
<td>75.7</td>
<td>(Guofeng et al., 2012)</td>
</tr>
<tr>
<td>2000</td>
<td>Fuelwood</td>
<td>China</td>
<td>109</td>
<td>547.8</td>
<td>(Cao et al., 2006)</td>
</tr>
<tr>
<td>2000-01</td>
<td>Fuelwood</td>
<td>India</td>
<td>*165 (50-530)</td>
<td>No data</td>
<td>(Habib et al., 2004)</td>
</tr>
<tr>
<td>2000</td>
<td>Fossil fuels, biofuels, open biomass burning and urban waste</td>
<td>Global</td>
<td>7500 (2000-29,000)</td>
<td>47,000 (18,000-180,000)</td>
<td>(Bond et al., 2013)</td>
</tr>
</tbody>
</table>

Table 9. BC and OC emissions in different regions of the world determined in this study and in previous publications. Source: own elaboration. $^a$ % of total emissions released to the atmosphere (exfiltration rate); $^b$ Average, 5$^{th}$ percentile and 95$^{th}$ percentile (in parentheses); $^c$ fuelwood, charcoal, agricultural waste and animal dung; $^d$ Includes fuelwood directly consumed by households and fuelwood to produce charcoal; $^e$ Central value and uncertainty range at 95%.
5. Indoor air pollution from biomass cookstoves in Senegal
5.1. Background

In Senegal, IAP accounts for 4.8% of the national burden of disease (WHO, 2007), and cause the prematurely death of 6,300 people every year (WHO, 2009). Although an important increase of LPG use at household level in urban areas occurred in 2009, LPG subsidies were eliminated and consumers returned in large numbers to wood-based biomass for cooking (Sander et al., 2011b). Nowadays, biomass accounts for 83% of the demand for domestic fuel use in rural areas and 58% in urban areas (WB, 2013).

The international donor community and national governments increased efforts since the 1980’s to disseminate cleaner and more efficient burning stove designs in Senegal, as in many LMI countries. A number of studies have analysed impacts on household air pollution and personal exposure from cooking activities in Asia (Balakrishnan et al., 2015; Bartington et al., 2017; Chengappa et al., 2007; Chowdhury et al., 2013; Dasgupta et al., 2006; Dutta et al., 2007; Gao et al., 2009; Hu et al., 2014; Kar et al., 2012; Siddiqui et al., 2009), Latin America (Albalak et al., 2001; Armendáriz et al., 2008; Commodore et al., 2013; Fitzgerald et al., 2012; Naeher et al., 2000; Northcross et al., 2010; Park and Lee, 2003; Riojas-Rodríguez et al., 2011; Smith et al., 2010; Zuk et al., 2007) and Africa (Ezzati et al., 2000; Ochieng et al., 2017, 2013; Pennise et al., 2009). However, almost no studies exist evaluating indoor air quality impacts from biomass combustion with traditional stoves and indoor air quality improvements derived from the use of improved cookstoves in Senegal.

This study provides a comprehensive real-world assessment of household pollutant concentrations from the combustion of biomass fuels for cooking in Senegal. It focuses on the quantification and characterization of BC, PM, UFP and CO indoor concentrations from burning wood fuel during cooking activities using traditional and improved cookstoves in a Senegalese rural village, which may be considered representative of rural areas in the country.

UFP refers to the particles with an aerodynamic diameter less than 0.1 µm (Pope and Dockery, 2006), characterized by low mass and large surface area. Their concentrations are not accurately reflected by particle mass concentrations, so they are monitored in terms of particle number concentrations (Sahu et al., 2011). In addition, the lung-deposited surface area (LDSA) concentration is increasingly emphasized as relevant parameter to estimate health effects of UFPs in urban areas (Ntziachrystos et al., 2007; Reche et al., 2015). Recent studies show that most particles generated from biomass combustion are typically between 0.05 µm and 0.2 µm (Tiwari et al., 2014). However, to date, research on this parameter with regard to cookstove emissions is still relatively scarce (Hosgood et al., 2012; Leavey et al., 2015; Patel et al., 2016; Sahu et al., 2011), especially when compared to studies focused on CO or PM mass.
Moreover, the study identifies parameters such as type of wood and household characteristics (i.e. construction materials, ventilation and kitchen size), which influence indoor air pollutant concentrations, in addition to the type of stove.

5.2. Methods

5.2.1. Household selection and study design

A cross-sectional study (Edwards et al., 2007) was conducted to evaluate the comparative impact of improved and traditional stoves on indoor air pollutant concentrations (PM$_{2.5}$, UFP, BC and CO) in the rural village of Bibane (described in detail in section 4.2.2), during the same period in which the quantification study of carbonaceous aerosols emissions was conducted. Even though the air quality monitors were not worn by women, it is assumed that indoor air pollutant concentrations may be considered as a proxy for personal exposure during cooking hours due to the small size of the kitchens and their low ventilation rates.

The sample size advisable in cross-sectional sample designs to evaluate changes in indoor air pollution due to improved stoves depends on the detectable difference in means and the covariance (COV) of measurements. For improved cookstoves without a chimney, it is realistic to expect a difference of 60%, and COV of 0.7. Therefore, the advisable sample size per stove type for this study was estimated to be 22 (Edwards et al., 2007).

The largest possible sample size of improved cookstoves in Bibane was 15 (number of improved stoves in good or fair good state and available for the study), so the same wood specie (dimb) was distributed to all families to reduce the sample size advisable, as it was done for the quantification study of carbonaceous aerosol emissions (chapter 4).

A total of 22 households was selected, 12 with the traditional stove and 10 with the improved rocket stove. The 15 improved stoves available were not included in the study due to time and resource constraints. Information on general kitchen characteristics was annotated, and the daily fuel wood was measured using a digital scale of 0–30 kg range with 1 g resolution. General features of study kitchens are summarized in Table 10, while and Table 11 and Table 12 provide individual information of each household.
<table>
<thead>
<tr>
<th>Kitchen characteristics</th>
<th>N</th>
</tr>
</thead>
<tbody>
<tr>
<td>Stove type</td>
<td></td>
</tr>
<tr>
<td>Improved</td>
<td>10</td>
</tr>
<tr>
<td>Traditional</td>
<td>12</td>
</tr>
<tr>
<td>Wall materials</td>
<td></td>
</tr>
<tr>
<td>Earth bricks</td>
<td>16</td>
</tr>
<tr>
<td>Tin</td>
<td>1</td>
</tr>
<tr>
<td>Thatched</td>
<td>1</td>
</tr>
<tr>
<td>Missing</td>
<td>4</td>
</tr>
<tr>
<td>Roof materials</td>
<td></td>
</tr>
<tr>
<td>Thatched</td>
<td>17</td>
</tr>
<tr>
<td>Tin</td>
<td>1</td>
</tr>
<tr>
<td>Missing data</td>
<td>4</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Kitchen characteristics</th>
<th>Mean±sd</th>
</tr>
</thead>
<tbody>
<tr>
<td>Kitchen volume (m³)</td>
<td>22.0±9.3</td>
</tr>
<tr>
<td>Free area of inlet/outlet opening (windows+doors+holes) (m²)</td>
<td>2.1±1.6</td>
</tr>
<tr>
<td>Fuel consumption (kg)</td>
<td>9.8±5.2</td>
</tr>
</tbody>
</table>

Table 10. Summary of participant kitchen characteristics. Source: own elaboration
<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>nº of windows and holes</td>
<td>0</td>
<td>6</td>
<td>1</td>
<td>1</td>
<td>2</td>
<td>1</td>
<td>3</td>
<td>1</td>
<td>0</td>
<td>3</td>
</tr>
<tr>
<td>nº of doors</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
</tr>
<tr>
<td>Area of windows and holes m²</td>
<td>0</td>
<td>0.082</td>
<td>0.24</td>
<td>0.2</td>
<td>0.072</td>
<td>0.35</td>
<td>0.86</td>
<td>0.58</td>
<td>0</td>
<td>0.32</td>
</tr>
<tr>
<td>Area of doors m²</td>
<td>1.65</td>
<td>1.20</td>
<td>1.30</td>
<td>1.26</td>
<td>0.97</td>
<td>2.16</td>
<td>2.026</td>
<td>1.65</td>
<td>0.86</td>
<td>1.21</td>
</tr>
<tr>
<td>free area of inlet opening (windows+holes+doors) m²</td>
<td>1.65</td>
<td>1.29</td>
<td>1.54</td>
<td>1.46</td>
<td>1.039</td>
<td>2.51</td>
<td>2.89</td>
<td>2.23</td>
<td>0.86</td>
<td>1.54</td>
</tr>
<tr>
<td>Kitchen volume (m³)</td>
<td>23.43</td>
<td>16.37</td>
<td>18.28</td>
<td>15.19</td>
<td>21.77</td>
<td>20.38</td>
<td>19.23</td>
<td>48.70</td>
<td>21.70</td>
<td>13.91</td>
</tr>
<tr>
<td>Walls material</td>
<td>earth bricks</td>
<td>earth bricks</td>
<td>earth bricks</td>
<td>earth bricks</td>
<td>thatched</td>
<td>earth bricks</td>
<td>earth bricks</td>
<td>tin</td>
<td>earth bricks</td>
<td></td>
</tr>
<tr>
<td>Roof material</td>
<td>thatched</td>
<td>thatched</td>
<td>thatched</td>
<td>thatched</td>
<td>thatched</td>
<td>thatched</td>
<td>thatched</td>
<td>tin</td>
<td>thatched</td>
<td>thatched</td>
</tr>
<tr>
<td>Household population (fraction of standard adult child of 0-14 years=0.5)</td>
<td>9.50</td>
<td>4.50</td>
<td>9.50</td>
<td>8</td>
<td>7</td>
<td>17</td>
<td>6.50</td>
<td>8.50</td>
<td>16</td>
<td></td>
</tr>
<tr>
<td>Wood consumption (kg/day)</td>
<td>5.40</td>
<td>4.60</td>
<td>11.80</td>
<td>20</td>
<td>15.80</td>
<td>6.40</td>
<td>8.27</td>
<td>7.40</td>
<td>10</td>
<td></td>
</tr>
</tbody>
</table>

Table 11. Participant kitchen characteristics. Improved cookstoves. Source: own elaboration
<table>
<thead>
<tr>
<th>Household code</th>
<th>BI-01-T</th>
<th>BI-02-T</th>
<th>BI-03-T</th>
<th>BI-04-T</th>
<th>BI-05-T</th>
<th>BI-06-T</th>
<th>BI-07-T</th>
<th>BI-08-T</th>
<th>BI-09-T</th>
<th>BI-10-T</th>
<th>BI-11-T</th>
<th>BI-12-T</th>
</tr>
</thead>
<tbody>
<tr>
<td>nº of windows and holes</td>
<td>4</td>
<td>2</td>
<td>3</td>
<td>1</td>
<td>1</td>
<td>2</td>
<td>no data</td>
<td>1</td>
<td>no data</td>
<td>no data</td>
<td>no data</td>
<td>3</td>
</tr>
<tr>
<td>nº of doors</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>2</td>
<td>no data</td>
<td>1</td>
<td>no data</td>
<td>no data</td>
<td>no data</td>
<td>1</td>
</tr>
<tr>
<td>Area of windows and holes m²</td>
<td>0.15</td>
<td>4.30</td>
<td>0.098</td>
<td>5.21</td>
<td>0.046</td>
<td>0.11</td>
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<td>0.038</td>
<td>no data</td>
<td>no data</td>
<td>no data</td>
<td>0.21</td>
</tr>
<tr>
<td>Area of doors m²</td>
<td>1.28</td>
<td>1.22</td>
<td>0.87</td>
<td>1.50</td>
<td>1.09</td>
<td>2.67</td>
<td>no data</td>
<td>1.43</td>
<td>no data</td>
<td>no data</td>
<td>no data</td>
<td>1.024</td>
</tr>
<tr>
<td>free area of inlet opening (windows+holes+doors) m²</td>
<td>1.43</td>
<td>5.52</td>
<td>0.97</td>
<td>6.70</td>
<td>1.14</td>
<td>2.78</td>
<td>no data</td>
<td>1.47</td>
<td>no data</td>
<td>no data</td>
<td>no data</td>
<td>1.23</td>
</tr>
<tr>
<td>Kitchen volume (m³)</td>
<td>18.99</td>
<td>30.29</td>
<td>14.49</td>
<td>21.79</td>
<td>10.44</td>
<td>39.28</td>
<td>no data</td>
<td>18.92</td>
<td>no data</td>
<td>no data</td>
<td>no data</td>
<td>17.32</td>
</tr>
<tr>
<td>Walls material</td>
<td>earth bricks</td>
<td>earth bricks</td>
<td>earth bricks</td>
<td>earth bricks</td>
<td>earth bricks</td>
<td>no data</td>
<td>earth bricks</td>
<td>no data</td>
<td>no data</td>
<td>no data</td>
<td>no data</td>
<td>earth bricks</td>
</tr>
<tr>
<td>Roof material</td>
<td>thatched</td>
<td>thatched</td>
<td>thatched</td>
<td>thatched</td>
<td>thatched</td>
<td>thatched</td>
<td>no data</td>
<td>thatched</td>
<td>no data</td>
<td>no data</td>
<td>no data</td>
<td>thatched</td>
</tr>
<tr>
<td>Household population (fraction of standard adult child of 0-14 years=0.5)</td>
<td>4.50</td>
<td>7</td>
<td>4</td>
<td>13</td>
<td>19</td>
<td>9</td>
<td>4.5</td>
<td>7</td>
<td>5</td>
<td>4.5</td>
<td>5.5</td>
<td>9</td>
</tr>
<tr>
<td>Wood consumption (kg/day)</td>
<td>9.32</td>
<td>4.66</td>
<td>1.24</td>
<td>13.50</td>
<td>20</td>
<td>15.80</td>
<td>15.54</td>
<td>9.33</td>
<td>6.86</td>
<td>4.3</td>
<td>6.68</td>
<td>8.49</td>
</tr>
</tbody>
</table>

Table 12. Participant kitchen characteristics. Traditional cookstoves. Source: own elaboration.
5.2.2. Indoor Air Pollution monitoring

Indoor air pollutant (PM$_{2.5}$, UFP, BC, CO and LDSA) concentrations were monitored in all kitchens using the same instrumentation. The instrumentation deployed was a function of its portability, autonomy, range of concentration, budget and sensitivity. One household with a traditional stove and one with improved stove were always monitored simultaneously in order to account for meteorological variability.

**Dusttrak (TSI Inc., Shoreview, MN, Model DRX).** Aerosol particulate monitor which monitors real time particle concentrations (PM$_1$, PM$_{2.5}$, PM$_{10}$) using a laser photometer with a reading range of 0–100 mg/m$^3$ and a resolution of 0.001 mg/m$^3$. It was installed in 21 households during the main cooking periods (lunch and dinner) and during part of non-cooking periods (several minutes before and after cooking). DustTrak was used for studies in similar contexts (Chowdhury et al., 2013; Ochieng et al., 2017) and its performance has been already assessed (Chowdhury et al., 2007; Rivas et al., 2017; Viana et al., 2015).

**DiSCmini (Matter Aerosol).** Portable monitor (Testo (Fierz et al., 2011) determining particle number concentration and mean diameter in the range 10-700 nm, connected to an impactor with a cutoff at 700 nm to prevent interference with coarse particles. Anti-static tubing was used during all intercomparison exercises (Asbach et al., 2017; Todea et al., 2016; Viana et al., 2015). The DiSCmini compares reasonably well with reference instruments under laboratory and field settings in urban and industrial environments (Koehler and Peters, 2015; Viana et al., 2015), with an uncertainty of 30% in terms of particle number concentration. However, to this author's knowledge, no field study of indoor air pollution from stoves used this device. It was installed in 6 households, during the main cooking periods and part of non-cooking periods. This instrument suffered frequent failures during the study due to the fact that UFP concentrations monitored were frequently outside its monitoring range.

**MicroAeth AE-51 (AethLabs).** Portable black carbon monitor with no cyclone at the inlet. The flow rate of AE-51 was set at the minimum, i.e. 50 mL/min, but due to very high BC loading in cookstove emissions, the filter strip had to be changed with a high frequency (every 5 minutes approximately) to prevent saturation (Rehman et al., 2011). This fact complicated the sampling procedure and resulted in data losses, and therefore the use of a diluter is advised for future studies. Previous studies have used this device for indoor BC monitoring studies from cooking activities (Kar et al., 2012; Rehman et al., 2011). Microaeth was installed in 15 households, during the main cooking periods and during part of non-cooking periods. The uncertainty of this instrument was quantified for outdoor air stations as 10% (Viana et al., 2015).

**Indoor Air Pollution Meter 5000 Series (IAP meter; Aprovecho Research Center, OR, USA).** Red light scattering photometer to measure PM$_{2.5}$ concentration, with a reading
range of 0-60 mg/m$^3$ and a resolution of 0.025 mg/m$^3$. It also includes an electrochemical sensor to measure CO concentration. It was installed in 22 households, during 24-hour periods. This device has already been used in certain indoor air pollution studies from cookstoves, (Grabow et al., 2013; Sota et al., 2014) but its uncertainty has not been reported in the scientific literature.

Monitoring devices were deployed according to standard protocols, 1 m away from the emission source and 1.45 m above the ground to represent breathing position of cooks, on the side of the stove opposite to open doors and windows (Chengappa et al., 2007). Inlets were hung approximately 10 cm apart from each other. The sampling interval was set to 1 minute for all instruments.

Devices were not installed in the same number of households because: i) some of them failed throughout the campaign (e.g. DisCMini), ii) there were insufficient filter tickets available, due to the high replacement frequency (Microaeth) and iii) some battery failed (DustTrak). Differences in the sampling period duration were due to the devices’ autonomy. While IAP meters had autonomy enough to measure during 24h periods, DustTrak, DiSCmini and Microaeth had not. Therefore, the latter were deployed during the main cooking periods (lunch and dinner), and during part of the background (non-cooking) period. Table 13 and Table 14 describe the devices installed in each household.
### Table 13. Devices installed per household. Traditional cookstove. Source: own elaboration

<table>
<thead>
<tr>
<th>Household code</th>
<th>BI-01-T</th>
<th>BI-02-T</th>
<th>BI-03-TT</th>
<th>BI-04-T</th>
<th>BI-05-T</th>
<th>BI-06-T</th>
<th>BI-07-T</th>
<th>BI-08-T</th>
<th>BI-09-T</th>
<th>BI-10-T</th>
<th>BI-11-T</th>
<th>BI-12-T</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>IAP meter</strong></td>
<td>5014</td>
<td>5014</td>
<td>5014</td>
<td>5014</td>
<td>5025</td>
<td>5025</td>
<td>5025</td>
<td>5025</td>
<td>5014</td>
<td>5014</td>
<td>5014</td>
<td>5014</td>
</tr>
<tr>
<td><strong>DustTrak</strong></td>
<td>2</td>
<td>NO</td>
<td>3</td>
<td>3</td>
<td>3</td>
<td>3</td>
<td>2</td>
<td>3</td>
<td>NO</td>
<td>3</td>
<td>2</td>
<td>2</td>
</tr>
<tr>
<td><strong>Microaeth</strong></td>
<td>5</td>
<td>787</td>
<td>5</td>
<td>5</td>
<td>ACS</td>
<td>5</td>
<td>5</td>
<td>5</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
</tr>
<tr>
<td><strong>Discmini</strong></td>
<td>MDS</td>
<td>IMPACT</td>
<td>IMPACT</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
</tr>
</tbody>
</table>

### Table 14. Devices installed per household. Improved cookstove. Source: own elaboration

<table>
<thead>
<tr>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
<th></th>
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<tbody>
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<td><strong>IAP meter code</strong></td>
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<td>5025</td>
<td>5025</td>
<td>5025</td>
<td>5025</td>
<td>5014</td>
<td>5014</td>
<td>5014</td>
<td>5025</td>
<td>5025</td>
</tr>
<tr>
<td><strong>DustTrak code</strong></td>
<td>3</td>
<td>3</td>
<td>2</td>
<td>2</td>
<td>3</td>
<td>3</td>
<td>2</td>
<td>2</td>
<td>2</td>
<td>3</td>
</tr>
<tr>
<td><strong>Microaeth code</strong></td>
<td>787</td>
<td>5</td>
<td>ACS and 787</td>
<td>787</td>
<td>ACS</td>
<td>ACS</td>
<td>5</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
</tr>
<tr>
<td><strong>Discmini code</strong></td>
<td>IMPACT</td>
<td>MDS</td>
<td>MDS</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
<td>NO</td>
</tr>
</tbody>
</table>
5.2.3. Instrument quality control

Microaeth, Discmini, and DustTrak monitors were previously intercompared for a 24 h period with their stationary counterparts (Multi-angle absorption photometer (MAAP), condensation particle counter (CPC) and optical particle counter (OPC), respectively) (Viana et al., 2015) in an urban air quality monitoring station in Barcelona (Spain). Detailed results are presented in Figure 24, Figure 25, Figure 26 and Figure 27. Comparisons were carried out for quality control purposes, i.e. correction equations were not applied, given the differences between the test (urban) and field (cookstove) in aerosol load and composition. In addition, the first day of the campaign all of the monitors were collocated in a chosen kitchen for field validation, evaluating inter-instrument variability, sensitivity, and consistency (Chowdhury et al., 2013). Details results are presented in Figure 28, Figure 29 and Figure 30. DustTrak was used for the field-validation of the IAP meter, as explained in section 5.3.4.

![Figure 24. Laboratory comparison. Relationship between DustTrak monitors (code: DustTrak 2) and OPC (GRIMM) concentrations (µg/m³) for the period 12/02/2016-26/02/2016 at the Barcelona (BCN) site. Source: own elaboration](image1)

![Figure 25. Laboratory comparison. Relationship between DustTrak monitors (code: DustTrak 3) and OPC (GRIMM) concentrations (µg/m³) for the period 12/02/2016-26/02/2016 at the Barcelona (BCN) site. Source: own elaboration](image2)
Figure 26. Laboratory comparison. Relationship between Microaeth (AE51_787, AE51_ACS and AE51_5) and MAAP concentrations (µg/m³) obtained for the period 12/02/2016-26/02/2016 at the BCN site. Source: own elaboration.

Figure 27. Laboratory comparison. Relationship between DISCmini (MDS, MD impact) and CPC concentrations (pt/m³) obtained for the period 12/02/2016-26/02/2016 at the BCN site. Source: own elaboration.

Figure 28. Field intercomparison. Relationship of DustTrak monitors (DST2 and DST3) concentrations obtained at the household BI-01-A. Source: own elaboration.
Figure 29. Field intercomparison. Relationship of Microaeths monitors (AE51_S, AE51_ACS and AE51_787) concentrations obtained at the household BI-01-A. Source: own elaboration.

Figure 30. Field intercomparison. Relationship of Discmini monitors (Impact and MDS) concentrations obtained at the household BI-01-A. Source: own elaboration.
5.2.4. Data Analysis

Concentration levels of the different pollutants with the two types of stoves were processed and arithmetic means (AM), standard deviations (SD), medians, and range were calculated. The Kruskal–Wallis test was used to test for differences in pollutants concentration between stove types. This test was selected because indoor air pollution data from this study doesn’t fit a normal distribution.

In addition, a linear mixed-effects model was constructed to assess the effect of factors at household-level on indoor air pollution. This model includes fixed and random effects. The fixed effects were: stove type, kitchen volume, free area of inlet/outlet opening and the kitchen construction materials. In addition, the random effects characterize variation due to individual differences. In this study, households were considered as random effect (Hu et al., 2014).

The formula that describes the model is: \( y_{it} = \beta_0 + \beta_1 x_1 + \beta_2 x_2 \ldots + \beta_n x_n + u_i + \varepsilon_{it} \)  

where \( y_{it} \) represents the pollutant concentration value modelled for household \( i \) at day \( t \); \( \beta_0 \) represents the intercept value; \( \beta_1 \) through to \( \beta_n \) represent the fixed effect variable coefficients for variables \( x_1 \) through \( x_n \); \( u_i \) are random effects at household level, assumed normally distributed with mean 0 and variance \( s_u^2 \), and \( \varepsilon_{it} \) are the model residuals assumed normally distributed with mean 0 and variance \( s_u^2 \) (Rivas et al., 2015). Data analysis was performed using IBM SPSS Statistics Base and Python software.
5.3. Results and discussion

5.3.1. Indoor air pollutant concentrations during cooking activities

Figure 31 shows a representative example of particulate and gaseous indoor concentrations for a household using the rocket stove (BI-02-A), covering the two main daily cooking activities (lunch and dinner) and a portion of non-cooking period. This example was selected based on data availability for all the parameters under study: particle number concentration, mass (PM$_{10}$, PM$_{2.5}$, PM$_{1}$) and mean diameter, as well as BC and CO concentrations. LDSA concentrations were also available (not shown graphically, for clarity).

The pollutant time series shown in Figure 31, point to cooking as the main emission source of indoor air pollution and personal exposure (especially for women and children). During cooking periods, number concentrations (N) and particle mass (PM$_{2.5}$) reached up to $3 \times 10^6$ #/cm$^3$ and 2000 µg/m$^3$, respectively. LDSA and CO concentrations climb to 4800 µm$^2$/cm$^3$ and 105 ppm, respectively. As previously found in past studies using solid fuel in biomass stoves, results presented in Figure 31 support the evidence that particulate emissions (PM, N and BC) and CO are co-emitted (Chengappa et al., 2007; Naeher et al., 2000; Northcross et al., 2010).
In the example shown in Figure 31, during non-cooking activities, average indoor CO and BC concentrations were close to zero, 1.28 ppm and 1.33 μg/m³ respectively. Another study conducted in rural villages in Northwest Bangladesh also showed very low indoor CO concentrations (0.2 ppm) during non-cooking activities (Chowdhury, 2012), whereas slightly higher BC concentrations (3.7 μg/m³ ±0.9,) were monitored in villages of northern India surrounded by traffic (Kar et al., 2012).

Average indoor background concentrations were PM$_{10}$=159.9 μg/m³, PM$_{2.5}$=99.1μg/m³ and PM$_{1}$=88.5 μg/m³, similar to other studies in rural areas in India (PM$_{2.5}$ from 45 to 207 μg/m³) (Sambandam et al., 2015). High PM$_{10}$/PM$_{2.5}$ ratios (1.6 on average) were monitored during non-cooking periods due to indoor and outdoor dust, which was re-suspended by wind, movement of inhabitants, and animals in the kitchen, among other causes. In the present study, during cooking periods PM$_{10}$/PM$_{2.5}$ ratios were on average 1.1. This highlights that particles are coarser during non-cooking periods and the important contribution of fine particles to indoor air during cooking periods.

Regarding UFPs, mean background N concentrations during non-cooking hours were 5,568 #/cm$^3$, 99% lower than during cooking periods. These results are lower than those reported by Chowdhury et al. (2012) in Bangladesh (Chowdhury, 2012), where mean UFP concentrations during non-cooking hours was 15,000 ± 7,200#/cm$^3$. This may be due to differences in ventilation rates and/or lack of enough time between cooking periods to ventilate the room. Finally, in this example, LDSA mean concentration during non-cooking periods was 50.5 μm$^2$/cm$^3$, similar to rural background sites measurements found in other studies (Casino, Italy, 69 μm$^2$/cm$^3$) (Buonanno et al., 2012).

Although ambient air quality data was not monitored simultaneously with indoor air pollution, contributions to household pollution from other combustion sources such as transport or industry were not considered significant in Bibane. Households were sufficiently dispersed so that neighbourhood cooking activities did not affect other households, and no significant additional combustion sources (e.g., forest fires) were recorded during the study period. A study conducted in rural villages of Western Africa showed that 74–87% of total PM$_{2.5}$ mass concentrations in cooking areas was from biomass burning particles, together with a small percent of crustal material and soil dust, from the Sahara desert (Zhou et al., 2014).

Figure 32 shows, for the same representative household (BI-02-A), the detailed time-series of pollutant concentrations during a single combustion episode (lunch preparation). The aim of this analysis was to assess emission patterns for different pollutants as a function of the cooking cycle.
Fire was lit at 11.31h am (marked with a red star in Figure 32) and a peak of all pollutants (particles and CO) was observed immediately after, showing the fast impact of combustion on indoor concentrations (10,000 µg/m$^3$ for PM$_{2.5}$, 1,000 µg/m$^3$ for BC, 2.5·$10^6$/cm$^3$, 4,800 µm$^2$/cm$^3$ for LDSA, 30 ppm for CO). Mean particle diameter decreased with increasing particle number concentrations (from 62 nm during non-cooking to 51 during cooking hours), although it did not show such a fast evolution, probably linked to instrumental limitations (particle diameter measurements were close to the instrument’s detection limit). In this study, the high emissions during the initial few minutes of the combustion phase were due to the use of straw as accelerant to start the fire. Other authors have reported similar results with the use of ignition products (Arora et al., 2013; Roden et al., 2006).

After the initial increase in concentrations (recorded for all pollutants), PM concentrations decreased (down to 2,000 µg/m$^3$), while the impact of emissions on indoor air were more persistent over time with regard to particle number (concentrations remaining at 1.5·$10^6$/cm$^3$). This has implications with regard to population (mainly women and children) exposure to UFPs (Ko et al., 2000). Conversely, PM mass concentrations progressively accumulated and increased over time, reaching at the end of the cooking period similar concentrations to those monitored during its initial phase (8,000 µg/m$^3$).

As shown in Figure 32, PM concentrations showed a larger variability throughout the combustion period than other parameters such as UFP or CO concentrations. Pollutant
peaks during cooking occurred when fuel was added or moved, due to activities such as placing or removing the cooking pot on the fire, stirring the food, or as a result of external factors (e.g., changing wind speed). The high variability in mean particle diameter during cooking periods is also noteworthy, as it indicates that different types of emissions were generated during the different types of combustion stages (influenced by fuel feeding, strength of the flame, ventilation, etc.).

As expected, \( \text{PM}_{10} \), \( \text{PM}_{2.5} \) and \( \text{PM}_1 \) concentrations were highly similar during the cooking period, indicating that biomass combustion was dominated by fine and ultrafine particles. Indeed, other works analysed particle size distribution during biomass burning showing that \( \text{PM}_1 \) was the main contributor to \( \text{PM}_{2.5} \) and \( \text{PM}_{10} \) fractions (Arora et al., 2013; Park and Lee, 2003).

Although fire was completely extinguished at 13.03h, PM and BC concentrations began to decrease toward the end of the food preparation period (12.52 h), when the cooker stopped feeding the stove and used the residual heat to finish the cooking. PM and BC levels returned to concentrations closer to those prior to the cooking period (117 µg/m\(^3\) and 1.2 µg/m\(^3\) for BC) around 10 minutes after the fire was extinguished. CO and N concentrations increased during the final period of cooking, when using residual heat. Once the fire was extinguished, N showed a similar trend to PM and BC, while concentrations of CO showed a gradual decrease over 25 minutes. The same behaviour for CO was found in previous studies (Chowdhury, 2012), and it is indicative of the remaining charcoal smouldering once the fire was extinguished.

5.3.2. Indoor air pollutant concentrations during cooking periods as a function of the type of stove

Table 15 presents the descriptive statistics for pollutant concentrations during cooking periods in the studied households, as a function of the type of stove used: traditional or improved. In total, 8 households were available with traditional stoves, and 9 for improved ones. However, not all pollutants were available for both types of households (irrespective of the use of traditional or improved cookstoves) given that instrumental failures were frequent due to the high concentrations monitored. The instrument which was most affected by technical failures was the DiscMini, and thus particle number, LDSA concentrations and mean particle diameter were only available for 3 improved cookstove and 3 traditional cookstove households. Results shown refer only to cooking periods. \( \text{PM}_1 \), \( \text{PM}_{2.5} \) and \( \text{PM}_{10} \) values during combustion showed less than 10% of difference for both types of stoves pointing to a predominance of fine particles. For clarity reasons, \( \text{PM}_{2.5} \) was selected for presentation in Table 15 as an indicator for \( \text{PM}_{10} \) and \( \text{PM}_1 \) mass fractions, as it is most commonly used in indoor air pollution studies and international standards.
During cooking hours, results show that PM, UFP, LDSA and CO concentrations in homes using improved stoves were statistically lower than in those with traditional stoves. Conversely, BC concentrations were higher in improved cookstove households. All of these differences were statistically significant according to the Kruskal-Wallis test (p<0.05).

Based on this analysis, reductions (or increases) in pollutant concentrations during cooking as a function of the stove were pollutant-dependent. The largest reductions due to improved cookstove implementation were observed for PM$_{2.5}$, with a 75.4% reduction in median concentrations. CO concentrations decreased by 54.3%, followed by particle number concentrations (30.0%) and LDSA (14.2%). As a result, the use of improved cookstoves in the households of this study (between 3 and 9, depending on the pollutant) showed clear and statistically significant improvements for indoor air quality with regard to particle mass (PM$_{2.5}$), particle number, LDSA and CO concentrations.

Other studies also found CO and PM$_{2.5}$ concentration reductions with the rocket stove, when compared to a traditional three stone fire: De la Sota et al. found 36% and 33% of 24-hr mean percent change for PM$_{2.5}$ and CO respectively in rural Senegal (Sota et al., 2014); and median percent reductions in 24-h PM$_{2.5}$ and CO concentrations ranged from 2 to 71% and 10–66% respectively in southern India (Sambandam et al., 2015).

Even though this is a positive result in relative terms, it should be noted that, in absolute terms, median concentrations during cooking periods remained high even when improved cookstoves were used: 1,780 µg/m$^3$ for PM$_{2.5}$, 1.5*10$^6$/cm$^3$ for UFPs, 4,014.2 μm$^2$/cm$^3$ for LDSA, and 34.75 ppm for CO.

Following an opposite behaviour, median BC concentrations increased by 36.1% (also statistically significant), from 497.7 µg/m$^3$ with traditional stoves to 677.6 µg/m$^3$ with improved cookstoves. This result highlights the complexity linked to the estimation of indoor air pollution impact of cookstove interventions, as improved stoves do not perform similarly for different pollutants (Arora and Jain, 2015; Jetter and Kariher, 2009; Kar et al., 2012; MacCarty et al., 2010, 2008b; Roden et al., 2009).

These findings are consistent with results presented in chapter 4, showing that the Noflaye Jegg rocket stove emitted more BC (EC) than the traditional stove. The consequence of this is that negative health effects from BC exposure can still be expected after installation of improved stoves, in addition to climate warming effects (Baumgartner et al., 2014).
<table>
<thead>
<tr>
<th>Type of stove</th>
<th>PM$_{2.5}$(µg/m$^3$)</th>
<th>Particle Number Concentration (N) (pt/cm$^3$)</th>
<th>Mean diameter (nm)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>n</td>
<td>Am ± sd</td>
<td>Median</td>
</tr>
<tr>
<td>Improved cookstove</td>
<td>9</td>
<td>4,621.9±8,387.1</td>
<td>1,780</td>
</tr>
<tr>
<td>Traditional cookstove</td>
<td>8</td>
<td>10,563.9±10,328.5</td>
<td>7,240</td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Type of stove</th>
<th>Lung Deposited Surface Area (LDSA) (µm$^2$ / cm$^3$)</th>
<th>Black Carbon (BC) (µg/m$^3$)</th>
<th>Carbon monoxide (CO) (ppm)</th>
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</thead>
<tbody>
<tr>
<td></td>
<td>n</td>
<td>Am ± sd</td>
<td>Median</td>
</tr>
<tr>
<td>Improved cookstove</td>
<td>3</td>
<td>4,201.3±1,962.6</td>
<td>4,014.2</td>
</tr>
<tr>
<td>Traditional cookstove</td>
<td>3</td>
<td>5,085.9±2,980.1</td>
<td>4,680.2</td>
</tr>
</tbody>
</table>

Table 15. Descriptive statistics of indoor air pollutants concentrations by stove type. n=number of households sampled. Am=arithmetic mean. Sd= standard deviation. *significant variation (p<0.05; Kruskal-Wallis tests). % of change in median compared to the traditional stove. Source: own elaboration
Finally, mean particle diameter increased from 42 to 50 nm (20.7%, Table 15) when rocket cookstoves were used. Previous studies showed a shift toward finer particles with some types of improved stoves (Arora et al., 2013; Just et al., 2013). For example, Just et al., 2013, found a mean particle diameter of 35 nm with the rocket stove and 61 nm with the three stone fire. However, such studies were conducted under controlled conditions and may not represent field conditions. Moreover, particle size distributions are significantly affected by type of cookstoves and fuels (Just et al., 2013; Tiwari et al., 2014). Finally, higher LDSA values for three stone can be attributed to larger emissions of ultrafine particles (Sahu et al., 2011).

With regard to previous studies, although several authors assessed indoor air pollution from cookstoves use in the field (Armendáriz et al., 2010, 2008; Chengappa et al., 2007; Chowdhury et al., 2013; Hu et al., 2014; MacCarty et al., 2007b; Ochieng et al., 2013; Park and Lee, 2003; Ruth et al., 2014; Sambandam et al., 2015; Siddiqui et al., 2009; Sota et al., 2014; Zuk et al., 2007), most of them present 24h or 48h average data, which do not reflect acute exposure levels observed during cooking periods. There are few field studies differentiating between cooking and non-cooking periods, whose results are summarized in Table 16, together with those obtained in this work. LDSA values are not included since, to this author's knowledge, none of the field studies that evaluated this parameter (Hosgood et al., 2012; Sahu et al., 2011) showed indoor concentration values separately for both cooking and non-cooking periods.
## Pollutant concentration during cooking activities

<table>
<thead>
<tr>
<th></th>
<th>This study</th>
<th>Other studies</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>CO (ppm)</strong></td>
<td><strong>Traditional:</strong></td>
<td><strong>Balakrishnan et al., 2015, India</strong></td>
</tr>
<tr>
<td></td>
<td>Median: 34.8</td>
<td><strong>Traditional:</strong></td>
</tr>
<tr>
<td></td>
<td>Mean (SD): 41.3 (26.8)</td>
<td>Median: 57.6</td>
</tr>
<tr>
<td></td>
<td><strong>Rocket stove:</strong></td>
<td>Mean (SD): 56.2 (13.8)</td>
</tr>
<tr>
<td></td>
<td>Median: 15.9</td>
<td><strong>Forced-draft advanced cookstove:</strong></td>
</tr>
<tr>
<td></td>
<td>Mean (SD): 23.8 (26.5)</td>
<td>Median: 20</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mean (SD): 21.5 (4.8)</td>
</tr>
<tr>
<td><strong>PM$_{2.5}$ (µg/m$^3$)</strong></td>
<td><strong>Traditional:</strong></td>
<td><strong>Balakrishnan et al., 2015, India</strong></td>
</tr>
<tr>
<td></td>
<td>Median: 7,240</td>
<td><strong>Traditional:</strong></td>
</tr>
<tr>
<td></td>
<td>Mean (SD): 10,563.9 (10,328.5)</td>
<td>Median: 1,451</td>
</tr>
<tr>
<td></td>
<td><strong>Rocket stove:</strong></td>
<td>Mean (SD): 1,991 (2060)</td>
</tr>
<tr>
<td></td>
<td>Median: 1,780</td>
<td><strong>Forced-draft advanced combustion stove:</strong></td>
</tr>
<tr>
<td></td>
<td>Mean (SD): 4,621.9 (8,387.1)</td>
<td>Median: 1,046</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Mean (SD): 1,813 (2,686)</td>
</tr>
<tr>
<td><strong>N (pt/cm$^3$)</strong></td>
<td><strong>Traditional:</strong></td>
<td><strong>Zhang et al., 2012, China</strong></td>
</tr>
<tr>
<td></td>
<td>Median: 2,184,800</td>
<td><strong>Traditional:</strong></td>
</tr>
<tr>
<td></td>
<td>Mean(SD): 2,576,060 (1,913,410)</td>
<td>Mean: 290,000</td>
</tr>
<tr>
<td></td>
<td><strong>Rocket stove:</strong></td>
<td>(Chowdhury, 2012), Bangladesh</td>
</tr>
<tr>
<td></td>
<td>Median: 1,529,080</td>
<td><strong>Chimney biomass improved stove:</strong></td>
</tr>
<tr>
<td></td>
<td>Mean(SD): 1,713,620±1,079,780</td>
<td>Mean (SD): 75,000 (31,000)</td>
</tr>
<tr>
<td><strong>Mean diameter (nm)</strong></td>
<td><strong>Traditional:</strong></td>
<td><strong>Zhang et al., 2012, China</strong></td>
</tr>
<tr>
<td></td>
<td>Median: 38.1</td>
<td><strong>Traditional:</strong></td>
</tr>
<tr>
<td></td>
<td>Mean (SD): 41.8 (24.5)</td>
<td>Mean (start combustion-10min): 40-50</td>
</tr>
<tr>
<td></td>
<td><strong>Rocket stove:</strong></td>
<td>Mean (10-15 min): 63</td>
</tr>
<tr>
<td></td>
<td>Median: 46</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Mean (SD): 50.4 (18.3)</td>
<td></td>
</tr>
<tr>
<td><strong>BC (µg/m$^3$)</strong></td>
<td><strong>Traditional:</strong></td>
<td><strong>Kar et al., 2012, India</strong></td>
</tr>
<tr>
<td></td>
<td>Median: 497.7</td>
<td><strong>Traditional:</strong></td>
</tr>
<tr>
<td></td>
<td>Mean (SD): 767.9 (874.3)</td>
<td>Mean (SD): 127.6 (103.5)</td>
</tr>
<tr>
<td></td>
<td><strong>Rocket stove:</strong></td>
<td><strong>Natural draft gasifier:</strong></td>
</tr>
<tr>
<td></td>
<td>Median: 677.6</td>
<td>Mean (SD): 75.5 (59.4)</td>
</tr>
<tr>
<td></td>
<td>Mean (SD): 1,042.8 (1,213.5)</td>
<td></td>
</tr>
</tbody>
</table>

Table 16. Summary of pollutant concentration during cooking activities reported for previous studies and within this study. Source: own elaboration

Table 16 shows that results were generally higher in comparison with other field studies, with the exception of CO and mean particle diameter, which are indeed comparable. This significant variation might be explained by the differences between studies regarding fuels and stoves characteristics, variations in cooking habits, stove location, ventilation, etc. (Just et al., 2013; Siddiqui et al., 2009).
Figure 33. Box-plots of pollutants concentrations for three-stone fire and the rocket stove. The boxplot presents minimum, first quartile, median, (middle dark line), third quartile and maximum values. Source: own elaboration.
Moreover, the range of pollutant concentrations during cooking periods is also highly variable within stove types (Commodore et al., 2013). Figure 33 presents box-plots of pollutant concentrations for the three-stone fire and the rocket stoves.

The large variability observed in Figure 33 within stove type can be attributed to a myriad of factors (e.g., variability in cookstove use and time, activity patterns, weather conditions, household room configuration and ventilation, cooking behaviours, etc.) (Clark et al., 2013). It highlights that averaging concentrations across households or experiments is a major limitation when addressing cookstove studies, as opposed to other types of air quality studies (e.g., ambient air). As shown in Figure 33, the mean/median values should not be considered fully representative of the datasets, and other metrics of data variability (e.g., standard deviation, interquartile range) are essential to understand the data distribution.

5.3.3. Effect of household-level determinants on indoor air pollution

A linear mixed effect modelling was performed to identify factors at household-level that might have a significant role in determining PM$_{2.5}$ concentrations during cooking periods (Hu et al., 2014).

Factors considered for inclusion as fixed effects were: stove type (traditional, improved), walls material (earth bricks, thatched, tin), roof material (thatched, tin), kitchen volume (m$^3$), free area of inlet/outlet opening (m$^2$), fuel consumption (continuous, kg/day) and elapsed time (minutes). Elapsed time was included as a continuous variable to examine temporal changes in concentrations (Albalak et al., 2001). Fuel characteristics were not considered as a factor, since the same type of wood was distributed between participants during the sampling period.

Other factors, such as type of meal prepared, meteorological conditions or cooker behaviours were not measured, so their influences were included in the model as random effect at household level, with a scalar (variance component) covariance structure (Hu et al., 2014).

Table 17 provides estimates of individual parameters ($\beta$), as well as their standard errors and confidence intervals. The variables selected for inclusion in the final model were those which significantly impacted PM$_{2.5}$ measurements ($p<0.05$) and contributed to the lowest Akaike information criterion (a measure of the relative quality of statistical models for a given set of data) score (Hu et al., 2014). To improve normality and variance homogeneity, log-transformed concentrations were used.
<table>
<thead>
<tr>
<th>Parameter</th>
<th>Coefficient (β) (log-µg/m$^3$)</th>
<th>Standard error</th>
<th>95% confidence interval</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>Type of stove</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Improved</td>
<td>3.61</td>
<td>0.28</td>
<td>3.06; 4.16</td>
</tr>
<tr>
<td>Traditional</td>
<td>3.89</td>
<td>0.27</td>
<td>3.36; 4.42</td>
</tr>
<tr>
<td><strong>Roof material</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Thatched</td>
<td>-0.69</td>
<td>0.16</td>
<td>-1.0034; -0.37</td>
</tr>
<tr>
<td>Tin</td>
<td>0*</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td><strong>Wall material</strong></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Earth bricks</td>
<td>0.49</td>
<td>0.047</td>
<td>0.40; 0.58</td>
</tr>
<tr>
<td>Thatched</td>
<td>1.02</td>
<td>0.076</td>
<td>0.87; 1.17</td>
</tr>
<tr>
<td>Tin</td>
<td>0*</td>
<td>0</td>
<td></td>
</tr>
<tr>
<td><strong>Kitchen volume (m$^3$)</strong></td>
<td>-0.019</td>
<td>0.0054</td>
<td>-0.029; 0.0081</td>
</tr>
<tr>
<td><strong>Free area of inlet opening (m$^2$)</strong></td>
<td>-0.12</td>
<td>0.015</td>
<td>-0.15; -0.092</td>
</tr>
<tr>
<td><strong>Daily Fuel consumption (kg/day)</strong></td>
<td>0.040</td>
<td>0.0030</td>
<td>0.034; 0.045</td>
</tr>
<tr>
<td><strong>Variation explained between households (%)</strong></td>
<td>35</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

Table 17. Linear Mixed Effect Modelling of log-transformed PM$_{2.5}$. *This parameter is set to zero because it is the reference level for the model Source: own elaboration

The estimate of the “Elapsed time” parameter was not significant (p>0.05), and therefore it was removed from the model. Coefficients indicate an increase in PM$_{2.5}$ concentration (in log-µg/m$^3$) for traditional cookstoves with respect to improved cookstoves.

For the roof material variable, PM$_{2.5}$ concentration decreased with thatch roofing with respect to tin, which is the reference format (β=0). This is due to the fact that thatched roofs enhance ventilation and further decrease indoor air pollutants compared to homes with metal roofs, also observed in previous studies (Ruth et al., 2014). Regarding wall materials, thatched walls resulted in the highest predicted PM$_{2.5}$. However, as only one house had tin walls (see Table 17), this material was not considered representative and therefore more samples should be included to analyse its effect on indoor air pollutant concentrations. Results evidenced that kitchen volume has a significant negative association with PM$_{2.5}$, with an approximately 1.9% decrease in log-PM$_{2.5}$ concentrations for every unit increase in volume. This is consistent with other studies (Albalak et al., 2001).

Finally, the free area of inlet/outlet opening (sum of areas of windows, doors and other holes in the kitchen) (Iyengar, 2015) was correlated with PM$_{2.5}$ concentrations, with a 12% decrease in log-PM$_{2.5}$ concentrations for every unit increase in free area of inlet opening. Results are coherent with other studies showing that particulate levels can significantly decrease by the existence of opening structures (Baumgartner et al., 2011; Bruce et al., 2004; Dasgupta et al., 2006; Park and Lee, 2003; Ruth et al., 2014).
As shown in Table 17, the linear mixed effect model explained 35% of the variation of indoor PM$_{2.5}$ concentration. The remaining variation may reasonably be due to imperfect specification of variables, such as cooking time, other household combustion sources, etc. (Pokhrel et al., 2015). It should be noted that this study aimed at evaluating the association of PM$_{2.5}$ concentrations with factors at household level rather than obtaining a predictive model.

PM$_{2.5}$ was selected to perform the linear mixed effect model because it is the pollutant analysed in a greater number of households (8 traditional and 9 improved). Since the other pollutants are simultaneously produced during biomass combustion (Chengappa et al., 2007), factors at household level could be assumed to affect in a similar way, although additional experiments would be necessary to confirm this hypothesis.

5.3.4. Comparison between IAP meter and DustTrak units

As previously explained, IAP meter units were co-located with DustTrak monitors. The aim of this comparison was to assess the performance of this lower cost instrument (de la Sota et al., 2017). It is relevant to present the correlation between these two methods because although the IAP meter has been used in a number of studies (Gorritty et al., 2011; Grabow et al., 2013; Sota et al., 2014), there are no publications covering field assessments of its performance compared to other commercial light-scattering measuring instruments.

Figure 34 shows the correlation of one DustTrak (code 3) with one IAP meter (code 5025) during measurements for the same household previously presented in Figure 31 and Figure 32, including both cooking and non-cooking periods to cover the entire range of typical daily variation of household PM$_{2.5}$ concentrations.

![Figure 34. Correlation of DustTrak response with IAP meter for PM2.5 concentrations (µg/m$^3$) at 1 min resolution. Source: own elaboration](image-url)
The response of the IAP meter showed a relatively good linearity with the DustTrak on a minute by minute basis ($R^2 = 0.73$). However, the correlation equations showed a large inter-variability across the co-location tests, and in certain households there was no such a good fit. Table 18 shows the results for all the co-location tests. $R^2$ values ranged between 0.12 and 0.87 in all of the households where IAP meters and DustTraks were co-located. In total, a 66.7% of the households (10 households) exhibited $R^2$ values higher than 0.75, while in the remaining 5 cases the dispersion of the data was considered high ($R^2 < 0.75$).

<table>
<thead>
<tr>
<th>Household code. Improved cookstove</th>
<th>BI-01-I (DST 3 vs IAP meter 5025)</th>
<th>BI-02-I (DST 3 vs IAP meter 5025)</th>
<th>BI-03-I (DST 2 vs IAP meter 5025)</th>
<th>BI-05-I (DST 3 vs IAP meter 5014)</th>
<th>BI-06-I (DST 3 vs IAP meter 5014)</th>
<th>BI-07-I (DST 2 vs IAP meter 5014)</th>
<th>BI-08-I (DST 2 vs IAP meter 5014)</th>
<th>BI-09-I (DST 2 vs IAP meter 5025)</th>
</tr>
</thead>
<tbody>
<tr>
<td>DustTrak vs IAP meter</td>
<td>$y = 0.25x + 8.15$ $R^2 = 0.8143$</td>
<td>$y = 0.18x + 10.71$ $R^2 = 0.73$</td>
<td>$y = 0.21x - 4.01$ $R^2 = 0.77$</td>
<td>$y = 0.11x - 39.71$ $R^2 = 0.78$</td>
<td>$y = 0.32x + 39.11$ $R^2 = 0.63$</td>
<td>$y = 0.31x + 47.97$ $R^2 = 0.87$</td>
<td>$y = 0.22x + 9.89$ $R^2 = 0.56$</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>Household code. Traditional cookstove</th>
<th>BI-01-T (DST 2 vs IAP meter 5014)</th>
<th>BI-03-T (DST 3 vs IAP meter 5014)</th>
<th>BI-05-T (DST 3 vs IAP meter 5014)</th>
<th>BI-06-T (DST 3 vs IAP meter 5025)</th>
<th>BI-07-T (DST 2 vs IAP meter 5025)</th>
<th>BI-11-T (DST 2 vs IAP meter 5014)</th>
<th>BI-12-T (DST2 vs IAP meter 5014)</th>
</tr>
</thead>
<tbody>
<tr>
<td>$y = 0.21x + 673.74$ $R^2 = 0.12$</td>
<td>$y = 0.22x - 6.14$ $R^2 = 0.79$</td>
<td>$y = 0.20x + 13.52$ $R^2 = 0.70$</td>
<td>$y = 0.12x - 91.17$ $R^2 = 0.80$</td>
<td>$y = 0.15x + 57.47$ $R^2 = 0.82$</td>
<td>$y = 0.16x + 88.42$ $R^2 = 0.78$</td>
<td>$y = 0.17x - 111.90$ $R^2 = 0.89$</td>
<td></td>
</tr>
</tbody>
</table>

Table 18. Field intercomparison. Relationship of IAP meter (5014 and 5025) and DustTrak (2 and 3) concentrations. DustTrak vs IAP meter

This variability in the responses of the IAP meter with regard to the DustTrak (considered here to be the internal reference) may be explained by different reasons: i) the fact that they are different instruments operating on different principles (different laser characteristics); ii) DustTrak and IAP meter were not placed at the exact same point in the kitchen (the inlets of the devices were hung approximately 10-20 cm apart from each other, so it may occur that devices were not equally affected by factors such as ventilation; iii) DustTrak zero was set with an ambient free of pollution (using a zero filter), whereas the IAP meter background was PM$_{2.5}$ from ambient air. Therefore, results from IAP meter presented in this study should only be interpreted as relative measures, since background concentrations were not considered in the reported PM2.5 values; and to estimate the additional pollution directly caused by cooking, it would be necessary to measure background concentrations in the area; and iv) Two different IAP meters were used during the study. Although both IAP meters belong to the 5000 series, previous studies showed that using inexpensive off-the-shelf smoke detector technology may suffer from substantial intra-unit variability (Chowdhury et al., 2007). This result indicates the need of individual instrument calibrations, which is also necessary when using other more costly air pollution monitors (Chowdhury, 2012; de la Sota et al., 2017; Viana et al., 2015).
6. Conclusions
In recent years, addressing cookstove black carbon emissions has aroused interest as a mean to mitigate this potent short-lived forcing agent (Grieshop et al., 2011). Among other advantages, the inclusion of black carbon (BC) co-benefits in cookstoves projects may provide an opportunity to attract new additional funding for further scaling up of these activities (Hamrick, 2015). However, most stoves implemented worldwide have not yet been tested with regard to their BC emissions (Jetter, 2015). Moreover, new improved and clean cooking solutions (including both fuels and stoves) are being developed continuously around the globe, each one with very different health, social, environment, and economic related potential impacts, which in turn can sharply vary from region to region.

This thesis furthers the body of knowledge in the field of climate impacts as a result of cooking with traditional biomass stoves and the role played by alternative cooking solutions, with a particular emphasis on black carbon and co-pollutants emissions, and focusing in Western Africa. The findings of this study also add some insights to the small number of current quantitative studies on indoor air pollution from cooking in this region.

In chapter 3, two low cost optical techniques to determine black carbon emissions, a cell-phone based system and a reflectometer, were tested against a more sophisticated reference system, based on thermo-optical analysis. Results show that, for the aerosol types and concentrations tested, the lower-cost methods may constitute good alternatives for the estimation of black carbon concentrations on filter substrates in cookstove studies under resource-constrained conditions. This finding can help to leverage important methodological barriers, as black carbon measurement techniques usually pose a challenge in terms of their high upfront cost and level of technical expertise required for their operation (Ramanathan et al., 2011).

Prior to the application of either of the low cost methods, locally-determined correction coefficients must be calculated by comparison with a reference method based on a statistically significant number of samples for each specific study. The calibration should be carried out using filters collected in the field and under the same conditions expected in the field, given that laboratory conditions will not yield the same type of emissions. Results also highlight the need to take into account the aerosol load on the filter and the filter substrate when calibrating lower cost methods.

The easy use of the cell-phone system opens up options for its application in science projects which actively involve citizens in scientific endeavours, also known as citizen science. This methodology could be useful for the process of data collection, which would increase the amount and frequency of data being collected, reducing the time and cost of sampling. Moreover, this would rise awareness regarding the environmental and health issues related to cookstoves and may lead to positive behavioural changes (Wyles et al., 2016).
Chapter 4 provided original cooking-related carbonaceous (elemental and organic carbon, EC and OC) aerosols emissions estimates in Senegal, where previous information was not available. Four biomass stoves, each one representing a type of combustion, were tested in the laboratory using the standardized water boiling test and filter samples analysed with a thermal-optical method.

The highest average elemental carbon emission factor (EF) (g/MJ) was found for the rocket stove. This was also found in previous studies, and can be explained by the fact that this type of stove has a strong draft which typically results in higher flame temperatures and elemental carbon emissions. What is remarkable is the fact that the rocket stove analysed in this study did not show a reduction in fuel wood consumption per task completed (i.e., Water Boiling Test, WBT) with respect to the three stones fire. As a result, this type of stove was the one with the highest EC total emissions. On the other side, the gasifier showed the highest fuel savings and very low elemental and organic carbon emission factors, so this type of stove had the smallest total emissions per WBT. Similarly to other studies, the basic improved ceramic stove showed similar results to the three stones fire, both in emission factors and in fuel consumed per WBT.

Results also showed that emission factors of EC and OC were also dependent on the wood specie burned, but further analysis are needed to study the effect of physical and chemical characteristics on carbonaceous aerosol emissions.

Likewise, the three stones and the rocket stove were tested under uncontrolled conditions in a rural village of Senegal, to understand how real cooking practices in this region influenced EC and OC emissions. The rocket stove had significantly lower PM$_{2.5}$ EF when compared with the three stones, but, similarly to laboratory results, it showed higher black EC EFs and did not reduced the daily fuelwood consumption. Moreover, OC, which has been associated with cooling properties, was reduced with the rocket stove. Therefore, carbonaceous emissions of the rocket stove produces a net positive warming effect in comparison with the traditional stove.

Differences between laboratory and field results were found, although less than expected based on previous studies. Field results were found to be more similar to results from the simmer phase of the WBT. This is coherent with field observations during testing activities, which showed that during meal preparations, cookers maintained a constant and low-medium flame almost all the time. This finding could be very useful to understand how representative are the standardized studies of actual cooking practices in Senegal, and other West African countries.

Finally, total EC and OC emissions and the net climate effect from fuelwood burning at household level in Senegal and West Africa were estimated. Quantifying the contribution that traditional and alternative cookstoves play in the emissions of
carbonaceous aerosol provides more accurate data that may contribute to reduce uncertainties of emission inventories and climate prediction models at regional level. This is especially important in the case of black carbon, as its climate impacts are highly dependent on the place where it is emitted (Hamrick, 2015).

This information should give additional impetus for improved and clean cookstove dissemination actions to be considered by governments and decision makers in West African countries in the Nationally Appropriate Mitigation Actions (NAMAs) and other funds and strategies targeted to climate change mitigation.

Chapter 5 presented the first real-world assessment of household concentrations of PM$_{2.5}$, ultrafine particles, black carbon and carbon monoxide from the combustion of fuelwood for cooking in Senegal. Results confirmed that household air pollution in this area is mainly due to cooking activities and showed that the installation of the rocket stove contributed to a significant reduction of total fine and ultrafine particulate number and CO concentration with respect to traditional stoves, but increased indoor BC concentrations. This increase of indoor BC concentrations with the rocket stove despite the fact that total fine particle concentration decrease is coherent with results presented in chapter 4.

The study also identified factors at household-level with a significant role in determining pollutant concentrations during cooking periods. Thatched roofs were found to conduct to lower IAP levels, when compared to homes with metal roofs. Higher values of the free area of inlet/outlet opening (i.e. sum of areas of windows, doors and other holes in the kitchen) also conducted to a reduction of indoor air pollution. Finally, though to a lesser extent, kitchen volume was also found to affect indoor concentration pollutant levels. In this region kitchens are generally very poorly ventilated for a number of cultural and lack of awareness reasons, and this is the first study in Senegal to investigate in detail how and to what extent different household and ventilation characteristics finally affects IAP.

Findings evidence that, in addition to a switch in the emission source (i.e. cookstove and/or fuel), successful strategies focused on the improvement of household air quality in Senegal (and other populations of Sub-Saharan Africa) should improve ventilation practices, appropriate construction materials and education on the importance of having a healthy cooking environment.

According to results presented in chapters 4 and 5, rocket stoves would have a positive effect with regard to IAP reduction, but more uncertain effects on climate, as they emit more warming (BC) particles. This proves that the climate and health-relevant properties of stoves do not always scale together and highlights that both dimensions should be considered in technological decisions (Grieshop et al., 2011).
It is important to note that the observation of no change in fuel use with the rocket stove was not found in other studies. For example, (Mazorra, 2017) found substantial fuel savings with the same type of rocket stove in other West-African regions, and this most likely would imply a reduction of total BC emissions. Moreover, this research work is focused on black and organic carbon emissions from biomass stoves, which have an important effect on climate change at both regional and global levels, but it does not include climate impacts from other pollutants emitted (such as methane, carbon monoxide, etc.) or factors affecting forestry, such as biomass renewability. Therefore, results presented in this work do not mean that rocket stove use does not have climate benefits, as the full picture is much more complicated and uncertain (see Section 7. Limitations and further research). Rocket stoves, if properly designed, used and maintained, might produce substantial benefits, such as reduce cooking time, decrease Products of Incomplete Combustion (PIC's) production, lower fuel wood consumption or reduce danger of burns, as previously found in other studies. Therefore, the types of improved biomass stoves found in this study are not the end point, but the first step towards the clean cooking.

On this path towards clean and sustainable cooking, findings from this research may be useful in the development of effective technologies promoting both climate and health co-benefits and may give useful insights for the selection of more appropriate technological solutions in each specific context.

However, it should be noted that an effective technology design by itself is not a solution (Ezzati et al., 2000; Grieshop et al., 2011). Indeed, the high dropout rate found in the rural village of Senegal were this research study was conducted, together with the numerous failed cookstoves initiatives around the world, confirm that adoption and sustained use of stoves by families is plausibly the most important and challenging consideration.

In addition to provide technical advances, improved stoves have to cover most of the desirable stove characteristics desired by users, and this is not an easy task. For instance, gasifiers may fail to cover the household’s preferences and needs as they are often top-loading only, meaning that the fuel must be fed into the chamber from above; they require much more frequent re-loading of fuel during cooking sessions; fan gasifiers may need electricity, which is not available in many regions, and fuelwood pieces need to be quite small, adding extra work to the cooking-related activities. Moreover, gasifiers are usually significantly more expensive when compared to other basic biomass improved stoves.

Therefore, in sub-Saharan Africa, where LPG and other modern fuel cooking appliances would be out of reach for the majority of population in the medium term, efforts should be focused on the design of gasifiers and other types of advanced biomass stoves more affordable and suitable for real cooking practices. Moreover, it is
necessary to continue working to improve the performance (e.g. reduce black carbon emissions, fuel consumption, etc.) of already available biomass stoves, such as the rocket stove.

Finally, once technical and usability requirements are achieved, the key point is to enable an appropriate environment that assures a long-term use and maintenance of the cooking solutions. If not achieved, all the efforts made to value the potential health and climate benefits of improved kitchens, such as this thesis, will be futile.

The keys to adoption and sustained use are so complex and broad that they are being the object of numerous research studies, and are beyond the scope of this thesis. However, one of the most important that has been traditionally forgotten should be stressed: If women cannot make independent choices about household resource use, public policies and other efforts may not be able to promote the cooking technology adoption (Miller and Mobarak, 2013). A broader social understanding and actions to fight gender inequalities should accompany every effort of clean cooking promotion.
7. Limitations and further research
While this study provides valuable insights in the field of climate and indoor air pollution impacts from biomass stoves, a number of limitations should be considered in interpreting the results. Moreover, several new questions and research needs emerged during this work, that would need to be covered in further studies. Both limitations and further research are outlined below:

• Given that both laboratory and field testing of stoves are rather costly and time and resource intensive, this research faced some sample size limitations. In the laboratory, three replicates of WBT were conducted. However, findings suggested that in forthcoming emissions testing it would be desirable to include more replicates, especially for the more emitting stoves, as the rocket stove types. In field studies, the sample size was rather limited and the variability of results between households was considerable. Further studies should include, to the possible extent within available resources, a larger sample size to provide more statistically significant insights. At this point, it would be very useful to study innovative approaches of data collection in a way that more information was collected without a high increase of resources.

• With regards to methodological limitations, due to very high levels of cookstove emissions, the filter strip of the portable BC monitor (MicroAeth AE-51, AethLabs) had to be changed with a high frequency to prevent saturation, complicating the sampling procedure and resulting in data losses. The monitor used to determine particle number concentration and mean diameter (DiSCmini, Matter Aerosol) failed throughout the campaign, also due to the high levels of emissions. Therefore, a use of diluters is recommended for future studies. Although PM$_{2.5}$ concentrations were performed with two devices based in laser photometers previously used in indoor air pollution studies from cookstoves, further studies should preferably include an in-situ gravimetric calibration of these devices.

• EFs determined in the laboratory vary significantly across type of fuel-stove combinations tested. Moreover, the field test focused on a single village and only one type of improved stove, so great caution must be taken before extrapolating the results to other locations or cooking solutions used in Senegal and other West African countries. More laboratory and field studies of different cooking technologies and fuels are needed, covering different areas of the region and studying the seasonality effect on emissions. This would derive in more representative EFs data to be used in emission inventories and climate prediction models at regional level.

• Further studies to estimate the impacts of cooking activities in West Africa should include the impact of other health and climate relevant pollutants (e.g. brown carbon, methane, carbon monoxide, carbon dioxide, polycyclic aromatic hydrocarbons (PAHs), etc.), as well as other climate-relevant aspects, such as fuelwood
renewability. These studies should be especially focused on biomass-based advanced technologies and fuels, such as gasifiers and pellets, because they will probably be the more accessible and affordable clean cooking solutions in sub-Saharan Africa, and to date very few impact measurements have been carried out on these stoves and fuels.

- Future research on cooking testing should involve a larger system, including technology, fuel, users, cooking environment and the surrounding ambient air. In this regard, studies should include personal exposure monitoring and consider the stove-use behaviours (i.e. cooking tasks, work schedules, etc.) Cooking environment characteristics, such as ventilation, construction materials or stove orientations, should also be systematically studied to better understand the role they play in stove emissions production, efficiency, safety parameters and Indoor air pollution. It would also be very interesting to analyse the contribution of household air pollution to outdoor (ambient) air pollution, which remains very poorly characterized.
8. References


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