CONSIDERING THE ‘EFFORT FACTOR’ IN FISHERIES: A METHODOLOGY FOR RECONSTRUCTING GLOBAL FISHING EFFORT AND CO_2 EMISSIONS, 1950 - 2010

by

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Abstract

Whether or not fisheries are sustainable not only affects ocean health, but also human health; a large portion of the population depends on marine ecosystems for food, livelihoods and social values. Our understanding of how fishing impacts the environment is lacking and under the threats of global climate change, the extent to which the ocean can continue to provide goods and services is questionable. Chapter 1 introduces some critical knowledge gaps in fisheries and problems with how marine resources have been managed in the past.

Chapter 2 describes a methodology that can be used to quantify and reconstruct historical fishing effort to create a global fishing effort database. Historically fisheries management has not given adequate consideration to the 'effort factor', potentially resulting in the mismanagement of marine resources. The methodology was applied to the Exclusive Economic Zones of 9 maritime countries, and preliminary results suggest that, although fishing effort appears to be stabilizing, catch per unit of effort is decreasing.

Chapter 3 uses the fishing effort calculated in Chapter 2 to estimate the CO_2 emissions from fishing over time. Fishing is not perceived as an important contributor to greenhouse gas emissions and climate change, despite using fishing vessels that rely on the combustion of fossil fuels (Wilson 1999). As in Chapter 2, the methodology was applied to 9 EEZs. It was found that the CO_2 per unit of catch (CO_2PUC; tonnes) increased, despite increases in fuel efficiency, and the industrial sector emitted 3 times more CO_2PUC than the small-scale sector in 2010. It was estimated that fishing contributes approximately between 2.8 – 5.2% to global CO_2 emissions annually. The final chapter, Chapter 4, discusses the preliminary results of the 9 sample EEZs within the context of the sustainability of fisheries and what it means to be sustainable.

Preface

All research, compilation, analyses and writing for this thesis were completed by me in close collaboration with the *Sea Around Us* project. My primary supervisor Prof. Daniel Pauly, along with the rest of my supervisory committee, Prof. Rashid Sumaila and Dr. William Cheung provided key contributions to the project design, methodology and analyses. Chapter 2, which focuses on fishing effort, improved upon the *Sea Around Us* project's global fishing effort database previously developed by Gelchu and Pauly (2007) and Anticamara *et al.* (2011). Valuable catch data was also supplied, and used in Chapters 2 and 3, by the *Sea Around Us* project.

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List of Abbreviations and Acronyms

CPUE	Catch per unit of effort
MSY	Maximum sustainable yield
MEY	Maximum economic yield
FAO	Food and Agriculture Organization of the United Nations
RFMO(s)	Regional Fisheries Management Organization(s)
ICCAT	International Commission for the Conservation of Atlantic Tuna
EROI	Energy return on investment
kW	Kilowatts
GT	Gross tonnage
GHG	Greenhouse gas
PPM	Parts per million
GDP	Gross domestic product
MCAA	Monte Carlo chain analyses
EEZ(s)	Exclusive economic zone(s)
CO_2PUC	Carbon dioxide emitted per unit of catch (in tonnes)
MSC	Marine Stewardship Council
LCA	Life-cycle assessment

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Chapter 1: Introduction

1.1 Identifying knowledge gaps

Whether or not anthropogenic global warming is occurring is no longer a real topic of debate. What is more difficult to discern are the main contributors to greenhouse gas (GHG) emissions, whether or not mitigation is an appropriate call to action and how the effects of climate change can be accounted for in management objectives. In climate change research, like in many other fields, the focus has been on terrestrial ecosystems, probably a consequence of our terrestrial bias (Bostrom 2002). Although humans live on land, the majority of the population live within 50 km of a coast and 40% of the population depend on marine resources as a significant source of protein (FAO 2010). Marine resources are consequently adversely affected by a number of human activities, arguably the most pressing of which is overfishing.

Overfishing is defined as “the harvesting of a fish population at a rate greater than which the population can replenish itself through reproduction and growth” (Rosenberg 2003). Overfishing is well established as a primary contributor to the collapse of marine ecosystems (Jackson *et al.* 2001; Pauly *et al.* 2002; Worm *et al.* 2006). While the solution seems simple – reduce the amount of fishing – policies aimed at reducing fishing pressure have been largely unsuccessful (Rosenberg 2003; Pascoe and Greboval 2005; Rosenberg *et al.* 2006). The reasons for this are many, but it is suffice to say that the common-property nature of fisheries, technological improvements, and poor incentive structures of fisheries policies contribute substantially to the problem (Sumaila *et al.* 2008; Sumaila 2010).

Fishing effort itself is not easily defined, nor is it easily measured. As utilized in this study, fishing effort is simply the product of fishing power, or capacity, and time spent fishing. There are a number of factors which can affect fishing power ranging from technical (for example, engine size) to operational measures such as the skilled knowledge of the boat's skipper (Wilson 1999) which makes collecting accurate fishing effort data challenging. In addition, fishing effort is not always required to report and when it is, it is often aggregated into a composite indicator such as catch per unit of effort, or CPUE (Squires 1987). Traditionally, fisheries managers use models such as Maximum Sustainable Yield (MSY) or Maximum Economic Yield (MEY) that emphasize consumption-based indicators like catch: if catches (or in the case of MEY, profits) are increasing or stable, then the stock is healthy and likely to be characterised as sustainable (Beddington *et al.* 2007).

The current status of fishing effort datasets varies considerably between countries and organisations. The European Vessel Registry is likely the most well-kept of effort-related data that are publicly available. This registry only contains vessel data on European Union member countries, and is very limited prior to 1970. The Food and Agriculture Organization (FAO) asks its member countries to submit effort data. However the quality of the data that are supplied differs greatly between member countries, and although the database was initiated in the 1950s, the bulk of the data are post-1970. Some Regional Fisheries Management Organizations (RFMOs), such as the International Commission for the Conservation of Atlantic Tunas (ICCAT), have good effort data, but they are limited to only the past one to two decades and are, again, only for very specific fisheries at specific geographic location. Additionally some countries now maintain fishing effort data, but they are not usually standardised between fisheries, and are often not publicly available. More recently, the *Sea Around Us* attempted to aggregate many of these datasets into one global dataset extending back to 1950 (Gelchu and Pauly 2007; Anticamara *et al.* 2011) and it is this database that provided the foundation from which the present study calculated fishing effort and is discussed in Chapter 2. While it appears that there are many fishing effort datasets available, they lack consistency, detail, coverage and accessibility. Consequently, in reality, there is very little documenting of how many boats are actually operating, what the characteristics of fishing vessels are, their operation time and where they are currently operating, let alone historically.

A lack of understanding of fishing effort has also resulted in a poor grasp of how fishing contributes to climate change. Most studies aimed at assessing fishing's carbon footprint are life cycle assessments of specific stocks or fisheries (for examples, see Tyedmers 2001; Kitts *et al.* 2008; Ziegler *et al.* 2011; Vazquez-Rowe *et al.* 2014) and tend to use energy return on investment (EROI) as the primary indicator for energy profiling; in order for fishing to be sustainable, the energy gained from fisheries (edible protein for example) must be more than the energy invested (in the case of fisheries, this is normally measured as fuel consumed); EROI compares energy released by fuel with that gained by human consumption and attempts to estimate the efficiency of a system in terms of calories. Calories, defined by the amount of energy it takes to raise 1 gram of water 1 °c at standard pressure, provides a common unit of measurement between fuel burned and edible energy returned in fisheries. The problem, however, is that calories burned in fuel cannot be metabolized by humans, thus measurements of EROI are difficult to interpret. It also assumes that fish are equal in caloric content[1], which is not the case.

[1] In general, 1 gram of fat equals 9 calories, and 1 gram of carbohydrates or protein equals 4 calories to the human body. The amount of protein and fat content in any one fish varies, and assuming caloric homogeneity between fish will likely not provide an accurate estimate of the efficiency of fisheries. Alternatively, it would make more sense to estimate system efficiency in dollars using the cost of fuel and the value of catch. This method would account for the heterogeneous nature of landed catches.

At the time of writing this thesis, there was only one global study known to the author that had attempted to estimate the EROI of global fisheries (Tyedmers *et al.* 2005). Tyedmers *et al.* (2005) reported that in the year 2000 global fisheries expended 12.5 times the fuel energy than they provided in edible protein energy and that fishing boats released approximately 134 million tonnes of CO_2 into the atmosphere at an average rate of 1.7 t of CO_2 per tonne of live-weight landed product. Similar to the majority of fisheries studies, the global assessment by Tyedmers *et al.* (2005) used methods rooted in consumption statistics, i.e., how much fuel would need to be burned in order to catch what was officially landed. The accuracy of this approach, as pointed out by its authors, thus depends on the accuracy of reported landings, which are widely perceived as vastly underreported (Pauly 1998). Furthermore, the study was a snapshot in time and provided no information on how trends in EROI or CO_2 emissions have changed over time.

Around the same time the study conducted by Tyedmers *et al.* (2005) was published, another study reconstructing the fuel consumption of ships and their subsequent CO_2 emissions since 1925 became available (Endresen *et al.* 2007). The authors used fuel sales data, again a consumption-based indicator, to estimate the emissions produced by the world's ships since 1925, which included fishing vessels greater than 100 gross tonnes (GT). Endresen *et al.* (2007) did not report a separate CO_2 emissions for the fishing sector, but their methods estimated that fishing vessels consumed roughly 10% of total marine fuel sales based on the number of boats and engine power reported by the FAO in 1998. It is likely that their results underestimate fuel consumption, and subsequent CO_2 estimates for the global fishing fleet, as they only accounted for vessels greater than 100 GT, which excludes an estimated 1.3 million fishing vessels (Endresen *et al.* 2007). Also, as mentioned above, the number of boats included in the FAO vessel registry is likely to be underestimated. Furthermore, the study used GT as an indicator of boat size, and which the authors assumed that fuel consumption was directly proportional to GT. For example, they assumed that a boat of 100 GT consumed half as much fuel as a boat of 200 GT. However, the relationship between GT and power was found by Anticamara *et al.* (2011) to be exponential and therefore, using the example above, a boat of 200 GT was found to be much more than twice the power of a boat of 100 GT. Assuming a linearly proportional relationship between GT and fuel consumption could result in a substantial underestimate of CO_2 emissions and also masks potential trends in how different fleets contribute to GHG emissions.

The future of global fisheries is unknown; what was once described as an 'inexhaustible' resource (Huxley 1882) is now widely perceived as declining, and doing so increasingly fast (Watson *et al.* 2012). Despite declining catches in many regions of the world, per capita consumption of fish is increasing and, to add insult to injury, some of the areas where seafood consumption is highest have significant food security issues (Kent 1997; Pauly *et al.* 2005; Garcia and Rosenberg 2010). While scientists have spent considerable effort, especially in the last two decades, to understand the essential ecosystem processes that need to be maintained in order to continue supporting an increasing demand for marine resources, climate change threatens to alter the operational underpinnings of ecosystems in ways that cannot easily be predicted or managed for (Cheung *et al.* 2012). Thus, before it is possible to understand how to mitigate and/or manage the effects of climate change, it is first necessary to examine the factors contributing to GHG emissions.

GHG emissions, primarily from the combustion of fuel, have been found to be the main driver behind climate change (Raupach *et al.* 2007). As such, there has recently been much inquiry into which industrial sectors are large contributors and how their carbon emissions can be reduced (for example: McMichael *et al.* 2007; Fuglestvedt *et al.* 2008) At present, fishing is an industry that has been largely overlooked despite being one of the most energy-intensive food production methods in the world today, relying almost completely upon fossil fuels (Wilson 1999). The contribution of fisheries to global GHG emissions has largely been unstudied, but a preliminary analysis suggested that industrial fishing contributes approximately 1% to global GHG emissions (Tyedmers *et al.* 2005). It is the primary focus of this thesis to develop a methodology, based on the work previously done by Anticamara *et al.* (2011) for the *Sea Around Us*, for estimating global fishing effort from 1950 – 2010, followed by a methodology for converting effort into CO_2 emissions to estimate the contribution of the world's fisheries to GHG emissions.

1.2 Measuring growth and managing natural resources: the problem with consumption-based statistics

Traditionally, environmental impact discussions focus on the idea that consumption has a direct relationship with ecological impact and therefore by reducing consumption, the effect on the environment is also reduced (Davidson *et al.* 2014). For example, CO_2 emissions from driving are considered to be a major contributor to GHG emissions in the United States (EPA 2014b); as a result there are many initiatives aimed at individuals to drive less, carpool or take public transit. While this approach may be effective at reducing an individual's carbon footprint, a decreasing trend in the amount of fuel sold in a given year (or some other metric used as a proxy to estimate CO_2 emissions from the transportation sector), does not necessarily equate to a reduced ecological impact. The reason is because, although consumption drives ecological impacts, environmental effects are a *function of effort*; the inputs required to produce commodities has a stronger relationship with ecological impact than does the volume of commodities produced and subsequently consumed (Davidson *et al.* 2014).

Historically, advances in technology have masked the importance of the “Effort Factor” (Davidson *et al.* 2014), which describes the direct relationship between effort and ecological impact. This factor affects us through the efficiency of natural resource exploitation and their accessibility. Probably more developed in the literature is the concept that changes in technological efficiency effect how the environment is impacted, generally in a diminishing manner. Thus, technological advances in efficiency and in the development of new technologies are often described as “silver bullets” (Ehrlich 1975; Waggoner and Ausubel 2002; Vira and Adams 2009). In other words, advances in technology will enable reduced environmental impact without compromising the resource value. And as in the past, it should also accommodate increased demand based on a growing human population and increased standards of living. The silver bullet approach allows us to have it all: decreased environmental impact, and the ability to supply a growing population with a better quality of life for less effort.

Relying on the human capacity to adapt through technological efficiency appears to be one option for managing global climate change and some recent studies have suggested that society already possess the technology capable of meeting the globe’s energy demands for the next 50 years and provides a stabilizing CO_2 scenario at 500 ppm (Pacala and Socolow 2004). There are also researchers who are slightly less optimistic, but suggest that society is on the “brink” of having sufficient technology to stabilize CO_2 emissions, but lacks the economic drive to invest and act on climate change (Hoffert *et al.* 2002). Much of the optimism surrounding advances in technology efficiency as an effective way to manage ecological impacts is rooted in the belief that society has avoided catastrophe in the past via technological adaptation, for example, by preventing the explosion of the ‘Population Bomb’ as originally described by Ehrlich in 1968 (1975).

Ehrlich (1975) predicted that the 1970s and 1980s would be characterized by mass starvation, an increase in the world death rate and that mitigation programs would be unsuccessful mostly due to unhindered population growth, and the inability to feed the world's growing population. The 1980s came and went without world famine and per capita food production has actually increased by about 41% since the 1960s with much of the success being attributed to technological efficiency and innovation (Lam 2011). A report conducted by the Population Studies Centre in the United States in 2011 suggests that there are a number of demographic reasons, paired with technological innovation, that resulted in the non-explosion of the 'Population Bomb', and posed that there is no imminent concern for society's ability to avoid famine and ecological catastrophe, (Lam 2011). In keeping with consumption-based statistics, the report's results are based on metrics of food production and completely ignore production effort as an important measure of sustainability.

The 'Green Revolution', characterized by technological improvements in the agricultural industry, did increase agricultural yields in the 1960s and 1970s, but since the 1980s efficiency gains have stabilized (Davidson *et al.* 2014). Green revolution technology required high water and chemical inputs, consequently degrading the quality of land and ultimately having a negative impact on agriculture yields (Kendall and Pimentel 1994; Pingali 2012). Although production of food per capita has continued to rise as pointed out in the Population Studies Centre report (Lam 2011), there has been both a decrease in yield per unit effort and an expansion onto land that was not previously considered to be viable for agriculture (a change in accessibility); these metrics are not captured by consumption – based indices and subsequently has helped cultivate a culture of optimism surrounding the potential for technology to continue to provide solutions to dwindling natural resources.

As in agriculture, the effect that changes in technology have had in fisheries is not often considered and up until recently, technological innovation placed no emphasis on reducing the environmental impacts from fishing but instead focused on increasing efficiency (Kennelly and Broadhurst 2002). Therefore to determine whether or not catches are sustainable, it is critical to know how efficiency has changed the way fishing operates. In other words are the comparisons of global catch statistics from year to year actually comparable? An important component to answering whether or not global catch statistics are comparable between now and the past is knowing the extent to which fisheries has expanded spatially. It has been documented that the global expansion of fisheries is more or less complete; fishing occurs in all of the world's oceans, in every marine ecosystem and at every depth (Morato *et al.* 2006; Halpern *et al.* 2008; Watson *et al.* 2012). However, to answer the question whether or not global landings data are directly comparable from year to year, the answer is no, i.e., not without accounting for changes in effort. A recent study published in 2012 that attempted to account for changes in effort via expansion concluded that while catches are presently stagnating at best, catch per unit of effort is rapidly declining (Watson *et al.* 2012). The results suggest a very different outlook on the sustainability of global fisheries than reported global catch data.

In addition to efficiency, technology has masked the Effort Factor through changes in accessibility. For example, in many coastal areas, subsistence fishers collect invertebrates by gleaning in the intertidal region and was only able to expand sub-tidally with the introduction of snorkel or dive gear (Hender *et al.* 2001). It should be noted that there is a complex relationship between technology, accessibility and economic viability: what may be technologically accessible is not necessarily economically viable at an industrial scale, and may therefore not be implemented. As natural resources become scarcer and prices rise, what was not previously accessible for economic reasons, however, may become viable for exploitation (Murphy and Hall 2011; Davidson *et al.* 2014) . Rather than investing and adjusting into less environmentally impactful energy sources, industry tends to spatially expand into higher cost but traditional energy options (Davidson *et al.* 2014), the financials of which are either subsidised and or absorbed by the general public by increased consumer prices. In contrast, the environmental cost may be "borrowed" from future generations (Azar and Holmberg 1995).

Accessibility, driven by advances in technology, of fisheries resources has changed dramatically over the last 150 years, for example the transition to steam powered and eventually diesel powered engines allowed access to fishing grounds further offshore that were not previously accessible by sail- or rowboat (Kennelly and Broadhurst 2002). Similarly, it has been suggested that fisheries have expanded bathymetrically, i.e., into deeper water, with the advent of new technology and in response to declining stocks inshore (Morato *et al.* 2006). Consequently, global marine fisheries landings peaked in the 1980s at about 80 million tonnes and have been relatively stable since (FAO 2012), suggesting that catches are 'sustainable'. It is with this information that many policy decisions are made whilst the change in where fishing is occurring and the effect of effort are not adequately considered. Many fisheries attempt to calculate CPUE providing at least some effort data, but it is often very fishery specific and does not capture the effects of technological changes in accessibility. Changes in accessibility in fisheries are difficult to pinpoint because effort in fisheries is generally measured by gear type, thus limiting comparisons within a single gear type. Also changes in accessibility are likely to occur when a fishery changes from one gear to another, for example, from spearfishing for lobster to trap fishing for lobster.

Advances in technology have masked the importance of the Effort Factor in sustainability science through increased efficiency and accessibility and has resulted in potentially an unfounded optimism regarding the future of Earth's natural resources, many of which are threatened by climate change. Contributing to the undue optimism is the idea that technology in the past has provided the tools to avoid environmental catastrophes such as water and food shortages. It could be argued that those who suggest that humans have avoided ecological catastrophes through advances in technology are likely to be suffering from the shifting baseline syndrome and do not recognize that there has already been significant losses globally in the form of extinction, ecosystem state changes and reductions in biodiversity (Myers *et al.* 2000; Pimm and Raven 2000; Stork 2009). The shifting baseline syndrome simply states that each generation of scientists accepts the ecosystem state at the beginning of their careers as normal, and measures all change during their careers from this state, subsequently ignoring substantial changes that happened in the past (Pauly 1995). This is well documented in fisheries (e.g., Baum and Myers 2004; Saenz-Arroyo *et al.* 2005; Pinnegar and Engelhard 2008). The concept has since been documented in a wide number of fields and can be appropriately applied to our understanding of the potential that technological efficiency holds in mitigating the effects of global climate change; technology is inaccurately considered to be effective at mitigating environmental impacts because most resource managers measure changes on a relatively short timescale, and use inappropriate indicators such as human consumption to understand sustainability. However, humans have been altering ecosystems for much longer than the last 160 years, the effects of which are almost always ignored in conservation management.

The industrial revolution and the development of advanced technology relied on a culture of "new frontiers" in a time when natural resources were relatively abundant (Davidson *et al.*

2014). Therefore it is possible that the power of technology will decrease as natural resources become scarcer. There is potential for technology to act on an innovation frontier, in other words, developing new technologies that produce more from a given set of inputs (Lam 2011), rather than those technologies which aim to exploit new resources. Innovation frontiers do provide some hope, but they come with an unsettling amount of uncertainty; if there can be anything that society has learned from the past 100 years is that there are no benefits without costs. In addition, research designed to shed light on how ecological impacts can be reversed suggest that the development and growth of a sector causing environmental damage are not likely to be proportionate to subsequent efforts to curb or reverse growth (York 2012).

CO_2 emissions demonstrate well the asymmetric relationship between growth and impact. Economic growth is highly correlated with increases in oil consumption and thus, GHG emissions (Murphy and Hall 2011). It is often assumed that declines in economic growth, usually measured in Gross Domestic Product (GDP) per capita, also imply a proportional decline in CO_2 emissions (York 2012). A study done by York (2012) found that assuming CO_2 emissions declines in proportion with GDP per capita is erroneous, and demonstrated that in years where GDP per capita shrinks, CO_2 emissions do not decline in equal proportion. The author suggests that this is probably due to the fact that economic growth produces durable goods which are not removed by economic decline and therefore continue to contribute to CO_2 emissions even after growth is curtailed (York 2012). Asymmetrical relationships present a challenge in the effort to combat climate change, especially from within a consumption-based paradigm, as it further supports the hypothesis that reducing consumption or reversing growth are not necessarily going to have the desired mitigation achievements.

Consumption-based analyses, those that focus on the volume of goods produced and subsequently consumed, continue to be applied in sustainability science despite not providing an accurate picture of the state of natural resources. Such approaches have remained common mostly because changes in technological efficiency and accessibility, along with the shifting baseline syndrome have masked environmental impacts and created a culture of optimism surrounding the potential for technological advances to continue to promise increased benefits at reduced costs. More recently, some studies have highlighted the importance of effort when evaluating true ecological impacts, the results of which suggest that the relationship between effort and ecological impact is more causal than that of consumption indicating that effort indices are likely to provide better metrics for evaluating ecological change and sustainability. It is on this premise that the thesis presented here focuses on developing a method for analyzing GHG emissions of the world's fishing fleets based on effort data as opposed to previous and traditional studies which focus on reported catch data.

1.3 Research objectives

Overall, the project aimed to undertake an evidence-based analysis using the best available information and analytical approaches to develop a methodology for estimating global fishing effort and subsequently GHG emissions from 1950 - 2010. The results will enhance our understanding of how fishing effort has changed over time, the industry's contribution to climate change as well as provide insight into how the industry may reduce its carbon footprint.

Previous studies of fishing effort focus mostly on specific fisheries (Wiviott and Mathews ; Tyedmers 2001; Thrane 2004; Schau *et al.* 2009a). The results of fishing effort studies generally conclude that fishing effort has increased over time, but CPUE has decreased, therefore fisheries are, in many places, suffering from overcapacity and low profitability (Beare and

McKenzie 2006; FAO 2006; Sumaila *et al.* 2008; Sumaila *et al.* 2012) . The methodology developed for Chapter 2 to estimate global fishing effort, in this thesis, is predicted to yield similar results in fishing effort and CPUE trends, while maintaining fleet characteristics such as sector and gear that have not been previously included in global studies.

Chapter 3 aims to develop a methodology for estimating the GHG emissions of the world's fishing fleets since 1950 by converting the fishing effort estimated in Chapter 2 into CO_2 emissions. As previously described, the carbon contributions of fishing have been predicted to compose approximately 1% of global GHG emissions and 1.7 tonnes of CO_2 emitted per tonne of catch (Tyedmers *et al.* 2005). The methodology developed in this thesis is independent of catch.

Hypotheses:

1) Fishing effort globally has increased since 1950 faster than catches hence CPUE will have decreased;
2) CO_2 emissions per unit of catch from the small-scale will be less than that of the industrial sector;
3) CO_2 emissions from the global fishing fleet contribute at least 1% of the world's total GHG emissions annually.

The above hypotheses were tested using data from 9 EEZs. Projections of global trends were tentatively estimated. The ultimate goal of the project was to demonstrate that the methodologies presented could be used to estimate fishing effort as well as CO_2 emissions, and will be applied to all fishing countries in the near future as a part of the *Sea Around Us*'s work. At which time it will be necessary to re-visit the results discussed in this thesis.

Chapter 2: Developing a methodology for calculating global fishing effort 1950 - 2010

2.1 Historical development and the globalization of fisheries

It is widely accepted by scientists that significant human impacts on the marine environment began with the Industrial Revolution, but the reality is that humans have been overexploiting local marine resources for at least 125,000 years (Richter *et al.* 2008). Pre-Industrial Revolution human impacts on marine ecosystems are often ignored partly because of the small-scale nature of early fisheries – the effects of which are often considered to be negligible despite numerous contemporary studies suggesting otherwise (Jackson *et al.* 2001; Chuenpagdee *et al.* 2006; Zeller *et al.* 2007b; Govender 2013) – and partly because of the shifting baseline syndrome (Pauly 1995). That is not to say that the Industrial Revolution did not substantially alter fishing practices; it signifies the beginning of the globalization of fisheries.

2.1.1 Fishing for food

The first record of tool use in fisheries is currently dated at 90,000 years ago and it was hypothesized that people in Africa had already developed a complex subsistence based fishery using harpoons (Yellen *et al.* 1995). Many of the first fisheries were likely to be beach-based and resembled what is now referred to as subsistence fishing, that is, fishing to provide food for one's family (and not for sale)/ Subsistence fishing is still practiced in many regions of the world and is a crucial source of protein in areas where alternatives are either few and/or expensive (FAO 2006). What characterizes a subsistence fishery, such as fishing objectives, cultural significance, and governance frameworks has probably changed the least relative to other fishing sectors since the Industrial Revolution, and in part contributes to the persistence of the idea that small-scale fishers have little impact on marine ecosystems.

The earliest evidence for a sea-faring fishing vessel was found in Kuwait and was dated to ~7000 years ago (Carter 2002). The evidence of fishing vessels suggests a potential shift in not only what marine resources were accessible, but also in the purpose of fishing. The presence of a vessel signifies an investment in fishing, one which would need to be justified, and thus has been hypothesized as an indication that fishers were aiming to catch in excess of what was required to feed their own families (Carter 2002). Fishing for local sale or trade is now known as 'artisanal fishing' and, not unlike subsistence fishing, plays an important role in local economies, especially in developing countries (Chuenpagdee *et al.* 2006). Subsistence and artisanal fishing, along with recreational fishing, are usually undifferentiated in fisheries studies and grouped under the umbrella term of small-scale fisheries (Zeller *et al.* 2014). Although these sectors are extremely diverse, it is often justified to consider them as one sector because they are unified by similar social, structural and governance characteristics and while this may be true, it ignores technological descriptors such as boat size, or gear type (Mills *et al.* 2011), that affects the relationship between stakeholders and the environment.

Two thousand years ago, fishing technology had diversified substantially and included angling, i.e., the use of hooks and lines, castnets, dragnets, traps and tridents (Kennelly and Broadhurst 2002). In the 14th century, the first beam trawl, then called a 'wondyrchoun', was documented and incidentally is the same time when the potential impacts changes in fishing technology may have on the environment first appeared:

> *The great and long iron of the wondyrchoun runs so heavily and hardly over the ground when fishing that it destroys the flowers of the land below the water, and*

> *also the spat of oysters, mussels and other fish upon which the great fish were accustomed to be fed and nourished. By which instrument in many places the fishermen take such quantity of small fish that they know not what to do with them, to the great damage of the Commons of the Realm and the destruction of the fisheries – Edward III* (as cited in Kennelly and Broadhurst 2002)

Despite the concerns and warnings about the use of destructive fishing techniques in Europe, fishing technology continued to increase in efficiency (Kennelly and Broadhurst 2002), even though fishers were limited to near-shore areas. In general, fisheries operations up until the mid-1800s, were spatially limited primarily for two reasons: speed, as boats were powered by either sail or rowing and the inability to keep catches from spoiling due to a lack of refrigeration. Unable to sell beyond relatively local markets, fishers operating in economies that may have otherwise been conducive to expansion were unable to. As such, up until the Industrial Revolution, fisheries remained largely artisanal in nature with a focus on providing food for their communities.

2.1.2 The Industrial Revolution, globalization and fishing for consumers

At the time of the Industrial Revolution, driven by the opening of new markets and opportunities presented with technological innovation and prosperity, fishing fleets in Europe and North America expanded greatly (Kennelly and Broadhurst 2002; Engelhard 2009). At the turn of the 20th century, sail-powered vessels could not compete with vessels powered by steam engines. For example, it was estimated that a typical steam otter trawler caught about 10-20 times more cod per hour than the contemporary sailing beam trawler (Engelhard 2009). It was at this time that the industrial fishing sector began to emerge and the development of off-shore fisheries became an economically feasible endeavour.

By the early 1900s vessels powered by steam engines dominated the fishing fleet; however, their prevalence would be relatively short-lived. The World Wars, although offering fish stocks some respite from the previous century's exploitation, catalysed the development of the diesel powered engine and refrigeration technology (Engelhard 2009; Thurstan *et al.* 2010). The first record of 'refrigeration' in fisheries actually occurred in 1797 when some fishing boats used natural ice to help preserve catches but it was not until nearly 100 years later that mechanical refrigeration first appeared onboard fishing vessels (Dellacasa 1987). Technologically advanced refrigeration systems first began appearing aboard tuna seiners in the 1930s and by the 1960s, was an integral part of the world's industrial fishing sector (Wang and Wang 2005). The role that refrigeration played in the expansion of fisheries cannot be overstated. After about 12 hours, if a fish is not frozen, the quality of the product declines significantly and can spoil within a few days (Johnston *et al.* 1994); therefore without refrigeration, the product had to be delivered to consumers within a few days. Refrigeration, and increasingly fast transport systems, enabled fishers to access international markets so much so that today there remains no portion of the ocean not impacted by fisheries (Halpern *et al.* 2008; Anticamara *et al.* 2011). Consequently, artisanal fishers were increasingly marginalized and are often outcompeted even in local markets, as smaller boats do not have the capacity for deep freeze technologies that requires a minimum horsepower well in excess of that available on artisanal vessels (Wang and Wang 2005; Pollnac 2007).

Following the Second World War, diesel engines were widely adopted in the industrial fishing sectors of many countries. The diesel engine was popular because it was safer, more reliable and cheaper to operate than its steam-driven counterparts (Fisk 2010). And the fishing sector's dependence on oil began. At the time oil was relatively inexpensive, but now the cost of fuel can account for up to 60% of total costs and thus threatening the economic viability of many of the world's fisheries (Sumaila *et al.* 2007; Allison *et al.* 2009). Economic growth is highly correlated with increases in oil consumption, and in the era of peak oil, the consequences of declining fossil fuel supplies on the economy is, at the very least, concerning (Murphy and Hall 2011). The effects of increasing fuel costs has driven research aimed at reducing fuel consumption, but many studies are finding that basic fishing effort data are hard to come by, thus limiting studies to small time periods and to very specific fisheries.

The globalization of the world's food supply has distanced the individual consumer from their food supply and the external costs of food production, such as environmental impacts, are not accounted for in the prices consumers pay. Globalization has reduced the importance of distance, and facilitates connections between otherwise disparate networks (Rood and Schechter 2007). Industrial fisheries target consumers in international markets often operating in conflict with small-scale fisheries that target local markets. The perception that small-scale fishers have negligible impacts on marine ecosystems fuels this conflict creating an unhelpful dichotomy between the "little guys" versus the "big guys" when reality suggests that there is a continuum connecting the two (Pollnac 2007). Although it is difficult not to make comparisons between industrial and artisanal fishers, and the present study does so tentatively, there is a need for fisheries research to investigate finer scale comparisons such as between different types of artisanal fishers (a broad category in itself) or between different gear types that are utilized by both sectors. Such research studies will provide a more comprehensive understanding of the impacts of both sectors and help to smooth out the "little guy" versus "big guy" argument.

The industrialization, and subsequent expansion of the world's fisheries, has created a complex network of relationships between stakeholders. The Industrial Revolution, although it by no means marks the beginning of human impacts on marine resources, does signify a shift towards the globalization of fisheries. In order to understand the future of fisheries, it is important to know its past developmental trends, a current knowledge gap in fisheries science. Shortly after World War II, the technology required for the global expansion of fisheries was widely implemented, and thus provides a good starting point for reconstructing global fishing effort. The following sections outline, in detail, the methodology proposed for constructing a global fishing effort database from 1950 - 2010 as well as preliminary results of changes in global fishing effort.

2.2 Methods

2.2.1 Introduction to the *Sea Around Us* project's global fishing effort database

The original *Sea Around Us* fishing effort database was first constructed primarily by aggregating data from the FAO by Gelchu and Pauly (2007) for the time period 1970 – 1995. This first iteration focused heavily on spatial data in the form of port-based fishing effort and was heavily weighted towards industrial fisheries; the authors used primarily Gross Tons (GT) reported to the FAO to estimate horsepower of the global fishing fleet. While this database provided a starting point for estimating global fishing effort, it lacked in representation of both small-scale fisheries and developing countries.

The second iteration of the *Sea Around Us* fishing effort database was completed in 2011 by Anticamara *et al.* (2011) and attempted to calculate global fishing effort for the time period 1950 – 2010. The authors used the FAO data previously aggregated by Gelchu and Pauly (2007), conducted a literature search in an attempt to find data for countries not covered previously, and reported fishing effort in kW rather than horsepower. There were many exclusive economic zones (EEZs), however, for which data was not found, and thus effort from EEZs with "similar" catch profiles acted as “surrogates” for data-poor EEZs. Although more comprehensive than the previous version, the database was limited in its applications beyond its original intended purpose (to map the spatial expansion of global fisheries) because of the assumed similarity between EEZs and the concept of surrogacy. Thus, to calculate the effort of the world’s fishing fleets, it became necessary to create a framework that would enable users with different research aims to utilize the *Sea Around Us* global fishing effort data in a meaningful way.

2.2.2 Challenges and solutions: deconstruction of the global fishing effort database

When examined closely, the fishing effort database by Anticamara *et al.* (2011), referred to in this section simply as the ‘database’, was found to have a number of challenges that limited its research applications:

1. The predictor variables and equations used to calculate kW are inconsistent;
2. The length classes of boats, in some cases, were not realistic;
3. Sectors were not included;
4. Surrogate EEZ data were used for EEZs without data;
5. There was a lack of transparency in sources and no data quality indicators;
6. Indiscriminate extrapolations were performed both forwards and backwards.

In order to reconstruct the database in such a way that allowed for the data to be transparent and applied widely to a number of research questions, the aforementioned limitations first needed to be addressed. The following is a detailed description of each of the above points and how the methodology proposed provides an appropriate solution.

2.2.2.1 Predictor variables and equations used to calculate kW

In the final version of the database it was not clear what predictor variable had been used to estimate kW for individual fleets (represented by a row) in the database. By looking at previous, working versions of Anticamara *et al.* (2011), it became apparent that a combination of variables had been used to predict kW depending on what data were available: GT (Gross Tons), GRT (Gross Registered Tonnes) or Length (m). GT and GRT are unit less measures of a vessel's carrying capacity. GT provides an index for the volume of the entire ship's enclosed spaces whereas GRT provides a measure of the internal volume of a ship and there is no reliable conversion between the two (Pascoe and Greboval 2003). In other words, it is not advisable to convert between GT and GRT and vice versa, especially when this can be avoided (see below).

Anticamara *et al.* (2011) developed two equations from data in the Europa Vessel Registry to estimate kW using GT and Length (m) (Table 1). Despite having length data for 99% of the raw data entries in the database, the majority of the effort was calculated by using GT to estimate kW. A large portion of the raw data was reported in GRT and thus had to be first converted to GT, a controversial conversion that has little scientific support (see paragraph above). Anticamara *et al.* (2011) used a conversion equation where GT = 1.85*GRT that was obtained from a website with a url that is now broken (http://www.iim.csic.es). Furthermore, GT and GRT are usually reported for the total fleet, and not for individual boats; it was thus necessary to estimate individual GT by assuming that all the vessels contribute equally to the total GRT prior to calculating kW. Instead, Anticamara *et al.* (2011) calculated kW using the total converted fleet GT for the majority of the vessels in the database. Since the equation used to estimate kW is exponential, using total GT to solve is incorrect. For instance, to calculate the total kW of three hypothetical boats using the GT Equation in Table 1 and the method that Anticamara *et al.* (2011) used looks like this:

Method 1:

Number of boats (n) = 3 and Total GT = 150

$$kW = 11.26 * (GT^{0.71})$$

$$kW = 11.26 * (150^{0.71})$$

$$kW = 395$$

The correct way to solve the equation, given that the number of boats is available (and it was), would be to estimate GT for the individual boats by dividing the total GT by the number of boats, followed by solving the kW equation using individual boat GT and then multiplying that number by the total number of boats. For instance:

Method 2:

Total GT = 150, n = 3 and GT = 50

$$kW = n*(11.26*(GT^{0.71}))$$

$$kW = 3*(11.26*(50^{0.71}))$$

$$kW = 543$$

Method 2 makes the assumption that each vessel contributes power equally and the size range of vessels included within the total GT determines whether or not this assumption is ill-fitting for a particular fleet. For example, if the size range is small, then the impact of assuming equal GT for all vessels in the data on the estimation of kW is also small. In order to use Method 2, there would have to be accurate data on GT as well as significant coverage. As mentioned above, GT was not the most common variable included in the raw data; however length is, and GT is a metric that has only recently been calculated. Additionally, GT can only be calculated for decked vessels (recall: GT provides a measurement of the *internal* volume of a vessel) and thus is not a good predictor variable for the majority of the world's small-scale fishing fleet. Anticamara *et al.* (2011) recognized that for smaller vessels the low accuracy of using GT to estimate kW often resulted in impossible values and thus the authors tried to "fix" GT calculations for small vessels. GT is not a relevant variable for small vessels, and the FAO does not calculate GT for vessels less than 24.9 m (Pascoe and Greboval 2003). It is for these reasons that using GT as a predictor variable for kW is not a good choice and instead the methodology proposed in this thesis suggests Length (m) as a more appropriate indicator variable for power.

Length (m) was present in 99% of the raw data, and was expressed in ranges. In addition to developing an equation to estimate kW using GT, Anticamara *et al.* (2011) also used the same data to determine an equation to estimate kW using Length (m). Length is an ideal candidate for a predictor variable because it is the most commonly reported vessel variable, it is static over time for an individual vessel and is applicable to all different kinds of vessels regardless of sector or gear. It is also unlikely that in the future, it will not be reported as it is used in a number of other vessel descriptor calculations, for example, GRT. The methods proposed and applied in this thesis use Length (m) to calculate fishing capacity for specific fleets.

Table 1 Equations used to estimate fishing vessel capacity (Anticamara *et al.* 2011)

Predictor Variable	**Equation**	**Sample Size**	**R^2**
GT	kW = 11.26*(GT^0.71)	1,158	0.9
Length (m)	kW = 0.436*(L^2.021)	1,158	0.9

2.2.2.2 Assigning length (m) to vessels

The database contained 12 different length classes (Table 2), with the largest length class, 12, characterizing vessels that were "4000 + metres". The largest fishing vessel to date is the Atlantic Dawn and it only measures about 145 m in length (Anon 2003). The upper length classes included in the database are unrealistic; however, the database did have vessels in these categories. An effort was made to locate the references for these data but they were listed as originating with the FAO and no original spreadsheets from the FAO were found. I sent a request to the FAO's fishing vessel database department asking for the vessel data by country, length and gear and upon receipt of the data, I found that the largest boat length category used by the FAO was greater than 24 m and thus did not provide any information regarding the presence or absence of very large vessels. As a result, it was not possible to generate any kind of rule-based criteria to allocate the vessels in larger, unrealistic length categories into a category which better represented the length of the vessel. In order to solve this problem, all vessels that fell above Length Class 6 must be considered on an individual basis and it must be determined whether or not these data points should be excluded or re-allocated to a different length class based on adequate review of the literature for a specific EEZ.

Table 2 Length ranges (m) and their subsequent class codes as used in Anticamara *et al.* (2011).

Length range (m)	Length class code	Length range (m)	Length class code
1 - 7.9	1	150 -249.9	7
8 - 15.9	2	250 - 499.9	8
16 - 24.9	3	500 - 999.9	9
25 - 49.9	4	1000 - 1999.9	10
50 - 99.9	5	2000 - 3999.9	11
100 - 149.9	6*	4000 +	12

* Proposed maximum length class to be included in the database.

2.2.2.3 Assigning sector

The database did not include sector as a delimiting fleet characteristic. Excluding sector does not really affect the quality of the data or change the total effort exerted, but it does limit the comparisons which can be made between different vessels. Sectors vary in their characteristics, for example, management approaches differ widely between recreational and commercial fisheries, yet all sectors extract from the same pool of resources – the ocean. It is therefore beneficial to have some concept of how the different sectors contribute to total fishing effort. In the updated version of the database, efforts were made to assign sectors to the data that had previously been collected by Anticamara *et al.* (2011) based on length. Unless evidence to suggest otherwise was found, vessels in the first two length class categories of Table 2 were considered artisanal based on the assumption that artisanal vessels are generally small, and all those vessels above length class 2 were considered to be industrial based on the assumption that industrial vessels are generally large. The database by Anticamara *et al.* (2011) was unlikely to contain data for vessels in either of the two other sectors, subsistence and recreational. When searching for updated data and adding to the database (see section 2.2.3), fishing effort belonging to subsistence and/or the recreational sectors were added wherever possible.

2.2.2.4 Problems with surrogate EEZ data

One of the main goals of constructing a database of global fishing effort was be independent of catch data. For approximately 60 EEZs, i.e., 20% of the EEZs with marine fisheries, Anticamara *et al.* (2011) found effort data too poor to generate time series and subsequently performed a cluster analysis using catch data to determine which EEZs with effort data were most similar to those without effort data. The effort data for the most similar EEZs defined by catch, was then used in lieu of any effort data in data-poor EEZs. These methods were likely justifiable at the time because despite using surrogate data for roughly 20% of the EEZS in the world, the total catches of these EEZs constitute a small portion of the total world catch (~1-2%).

Many of the EEZs with surrogated data are small island developing states. It is probable that there is considerable small-scale fishing occurring within these EEZs, and statistics are unlikely to be reported. Although the methods used by Anticamara *et al.* (2011) are justifiable, they assume that reported catches adequately represent fishing effort, and subsequently homogenizes small-scale fishing effort between countries as well as ignores the unique historical development of fisheries between them. In addition, for these EEZs, fishing effort is no longer independent of catch. Therefore, it is proposed that the surrogate data no longer be included and that the data replaced by researching non-traditional sources, and possibly estimating fishing effort from variables other than catch, for example the reported number of fishers employed in the industry, in the event that no direct fishing effort data are found (see section 2.2.3).

2.2.2.5 Lack of transparency and no indication of data quality

One of the most critical components of the scientific method is replication, an important pre-requisite of which is transparency. The original database by Anticamara *et al.* (2011) did not record references by row and it was difficult to track down individual reference files. As such it was not possible to check data, know whether or not data was already included in the database nor was there any quantitative way to estimate the quality of the data. Every attempt was made to locate any sources that were used by Anticamara *et al.* (2011), but the success rate was low. It became necessary to then check each time series generated by Anticamara *et al.* (2011) against what could be found in the literature (both primary and grey sources). If there was evidence to confirm the presence of a time series, it was accepted into the database and referenced using a coding system whereby each reference was assigned a unique code, and a digital copy of the reference was kept. If a time series could not be validated, it was excluded.

In addition to including references within the framework of the database, it was necessary to take into account the quality of the references for each variable used to estimate kW. The three variables used to estimate fishing effort in the database are length (in metres; used to estimate kW/boat), number of boats and the number of days spent at sea (Table 3). In order to obtain an overall data quality score for each row in the database, an equally weighted average was calculated using the score for each of the aforementioned variables. Additionally a data quality score was also assigned to gear. This was done because it is thought that gear significantly affects fishing effort, especially with regards to fuel consumption (Gulbrandsen 2012). Understanding how gear affects fishing effort, fuel consumption and the sustainability of a fishery is a considerable knowledge gap that hopefully will be filled in the future. Including a data quality score for gear enables future users to consider how gear affects fishing effort while still being able to estimate data quality.

Table 3 Data quality scores for the variables used to estimate fishing effort

Quality	Score	Length	Number of Boats	Number Fishing Days
Worst	1	Estimate based on regional patterns	Estimate based on regional patterns	Estimate based on regional patterns
Satisfactory	2	Interpolated, carried forwards or backwards, estimate	Interpolated, carried forwards or backwards, estimate	Interpolated, carried forwards or backwards, estimate
Good	3	Estimated from a variable, e.g., GT, Grey literature	Estimated from a variable, e.g., number of fishermen, grey literature	Estimate developed by Anticamara *et al.* (2011)
Best	4	Reported raw data	Reported raw data	Reported raw data

2.2.2.6 Indiscriminate extrapolation backwards and forwards

The revised database, like the original built by Anticamara *et al.* (2011), aimed to reconstruct world fishing effort from 1950 – 2010, but that does not mean that all the time series within the database span the entire time period. The database by Anticamara *et al.* (2011) indiscriminately extrapolated all time series back to 1950 and carried them forwards to 2010, even if there were just a single data point. Aside from reporting effort where there wasn't any, this practice will obscure changing effort patterns. For example, if there is a reduced number of fishing vessels later on in the time period as a result of decommissioning policies for older vessels, but still an increase in overall effort due to fishers' investments in larger vessels, the magnitude of change will be masked. The methods used by Anticamara *et al.* (2011) provide high coverage for using less data, but at a cost of decreased detail and masking patterns in fishing effort, especially early on in the time period when raw effort data can be difficult to find.

In order to find a suitable start date for a time series it was necessary to review when each sector began, and if possible find information on when different gears were introduced. For the

EEZs where the *Sea Around Us* had completed a catch reconstruction, the start date for individual sectors for catch were used as tentative start dates for the effort time series. However since the *Sea Around Us* catch database does not consider gear, it was necessary to conduct a literature review for each EEZ to confirm whether or not there were different start dates for gear types within a sector. For instance, if an EEZ was found to have effort data for both industrial trawlers and industrial purse seiners it was necessary to confirm whether or not these two fisheries began at the same time or if one preceded the other. The same process was conducted for the other sectors.

Overall the challenges presented in the database were primarily a result of assessing global fishing effort without having to reconstruct fishing effort individually for each EEZ. The results of Anticamara *et al.* (2011) provided a good foundation for beginning to asses changing patterns in global fishing effort overtime. However, it was unable to provide the detail that would arise from reconstructing fishing effort for each EEZ individually – a task that would be extremely difficult and time consuming for any one person. The methods for reconstructing global fishing effort suggested and applied to nine EEZ**s** in this thesis provide a middle ground between the broad-scale, rule based methods used previously and reconstructing effort individually for each EEZ by using the data collected by Anticamara *et al.* (2011), but improving it with data specific to fishing effort trends in individual EEZs.

2.2.3 Worth repeating: methods for reconstructing global fishing effort

In order to implement the changes outlined above and generate a database of global fishing effort, three steps were taken using the raw data originally collected by Anticamara *et al.* (2011): (1) calculating kW for individual fleets; (2) generating the time series; and (3) performing an individual EEZ literature search. The details of each step are described below.

Step 1: Calculating kW for individual fleets

Fleet, as defined in this study, is a set of vessels from a specific EEZ that share the same sector, utilize the same gear and are similar in size. In effect, these vessels can be described as having equal capacity or fishing power, and in the present study, is measured in kilowatts (kW). The kilowatt is a unit of power and is different than measuring kWh which is a measure of energy[2]. In this study is a measurement of fuel expended or contained within something over a given unit of time, such as kW per hour, or in the case of the present study, kW per day. As such, in order to estimate the fishing effort of a fleet, *FE*, it is necessary to multiply fishing power (the kW) of the fleet by the number of days spent fishing (*DF*).

$$FE = kW*DF$$

Where *kW* is equal to[3]:

$$kW = 0.436*(L\text{^}2.021)$$

where *L* is the length (m) and was determined for a fleet by calculating the midpoint of a given length class category and days fished (DF) is equal to the number of days fished by a given fleet:

$$DF = N*CF$$

where *N* is the number of days fished and *CF* is a continent factor. The number of days fished was originally compiled in the first iteration of the fishing effort database designed by Gelchu and Pauly (2007). The continent factor was later applied by Anticamara *et al.* (2011) in order to

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[2] Energy is the capacity to do work, and can take many forms. For example, the calorie, joule, and kilowatt are all units of energy. In contrast, power refers to the rate at which work is done. For example, a watt is the number of joules per second; kilowatt- hour is also a measure of power, and is equal to 1000 watts running for 1 hour.

[3] The exponent to estimate kW from length is less than three, and is close to 2. This is because longer vessels are energetically favourable, i.e., the energey cost per unit of volume or weight to move a vessel through water diminishes with length because longer vessels tend to have a lower Reynold's number.

account for differences in time spent fishing based on geographical patterns (see Appendix A for complete details on the number of days fished used in the database).

In addition to power and the number of days fished, a major component required to calculate fishing effort is the number of boats operating in the fleet (n). Up until this point, the equation for calculating fishing effort is very similar to that of Anticamara *et al.* (2011), but in order to improve fishing effort estimates even further, two additional factors were added: a motorisation factor (*M*) and a technological coefficient (*T*). Motorisation was not particularly relevant in the previous versions of the fishing effort database as most of the fishing effort accounted for was industrialized and therefore assumed to be 100% motorised. The present study aimed to include small-scale fisheries, which are less likely to be motorised especially early on in the time period and therefore it affects how much fishing can be done in a given unit of time and area. In the present estimation, the default motorisation value was 1 (the fleet was 100% motorised), unless alternative information was found to indicate that the fleet did not become motorised until sometime after 1950.

Similarly, previous versions did not consider the change in technological efficiency overtime. Changes in technology have increased dramatically how efficient fishing boats are in the last 60 years. However, the extent to which it has affected fisheries is not well known, but it has been estimated that changes in technology have increased vessel capacity by approximately 2% per year since 1950 (Pauly and Palomares 2010). Fishing effort studies usually use static indicators such as GT or in this case length, and thus, a boat with a length of 10 m in 1950 will still have the same length in 2010 and are thus equal to one another. Studies assembled by Pauly and Palomares (2010) suggest otherwise and it was thus important to include a technological coefficient to better estimate fishing effort.

The full equation used to estimate fishing effort was:

$$FE = kW*DF*n*M*T$$

Step 1, as described above, of assembling the database is characterised by calculating fishing effort, but also included assigning vessels to specific fleets, and each fleet was assigned a unique case number, which is composed of EEZ code, sector, gear and length. Once Step 1 was completed it was possible to see the raw data anchor points for each fleet in the database.

Step 2: Generating the time series

Upon completion of Step 1, there were raw data anchor points for fleets in approximately 200 EEZs. Linear interpolation was used to generate estimates of the number of boats data for the years between two raw data points. The biggest challenge in this step was to determine the start and end years for each fleet. The *Sea Around Us* had in its catch database sector start dates for approximately 60 of the 200 EEZs at the time and these dates were used to determine when industrial and artisanal fishing began and/or ended in these EEZs. There were some instances where raw data points existed outside the bounds of what was contained in the *Sea Around Us* database and when this occurred the time series used the date suggested by the data collected Anticamara *et al.* (2011) but was flagged for checking. For the EEZs which did not yet have sector information in the *Sea Around Us* catch database, a literature search was conducted to estimate the time at which the different sectors began.

To interpolate between the time series start year and the first raw data point, the number of boats for each fleet at the beginning of the time series needed to be estimated. For the industrial sector the first year in the time series was assumed to have 1 boat, unless information was found otherwise (Step 3). To bring industrial time series forwards to present day, the last known raw data point was carried forwards. For the artisanal sector, the number of boats was carried backwards to the beginning of the time series from the earliest available raw data point. To estimate current artisanal fishing, the last known raw data point was carried forwards. These methods are similar to those used by Anticamara *et al.* (2011) and would result in underestimating changes in fishing effort throughout the time period. Therefore it was necessary to conduct research on individual EEZs and alter the time series accordingly.

Step 3: Individual EEZ literature search

Step 3 was designed to consider the validity of each of the time series that were generated by the previous two steps. Specifically, the following questions for each EEZ needed to be answered:

1) Do the start and end dates for each fleet that were developed in Step 2 based on sector also make sense given the gear type of the fleet? For example: Do references suggest that the gears in the industrial sector were implemented at the same time? Are any of the gears obsolete or are missing entirely? Does artisanal fishing still occur?
2) Does the number of boats used at the tail ends of the time series reflect what is reported in the literature? For example, can the default value of 1 vessel for the industrial sector be better estimated?
3) Are there time series with fishing vessels greater than 150 m? Can these vessels be reallocated to a more reasonable length class or should they be eliminated from the database?
4) What fishing effort is not currently represented in the database? Make an effort to add time series which were previously not included; and
5) Include references for each of these changes, and alter data quality scores accordingly.

Although the first two steps are complete for all EEZs for which there were raw data collected by Anticamara *et al.* (2011), the final step has been conducted on approximately twenty EEZs, nine of which are reported in this thesis. Only a subset of the world’s fishing fleets was completed in order to assess the effectiveness of the methodology, and to report on whether or not including these changes contribute to the understanding of global fishing effort.

2.2.4 Measuring data quality

Understanding the quality of data is an important component of scientific research. The methods used in this particular study involved different types of data sources for each of the variables used to calculate fishing effort. Consequently there are different sources of error not only between variables, but also within each variable. Under these circumstances, and in line with practices used by *Sea Around Us* project on the global catch database, I used a non – parametric approach that enabled different sources of error within each variable to be evaluated based on the quality and robustness of the data.

Each variable in each row of data was given a data quality score depending on the reference used to estimate the value for that variable in a given year and fleet (Table 3). This method ensured that for each variable, the data quality score was independent of the other variables. An equally weighted average was taken using the data quality scores for each of the three variables to estimate the data quality score for fishing effort for each year and fleet within an EEZ. The fishing effort quality scores for each fleet in an EEZ were then averaged so that each year was designated a data quality score between 1 and 4 for total fishing effort. Each score was assigned a percentage value; a score of 4 indicated that the data quality was high and that the estimate is likely to be within 10% of the true value, whereas a score of 1 suggested that the true value is likely to be within 40% of that estimated (Table 4). The percentage value associated with each score was assigned arbitrarily, but their structure was based on a criteria used by Intergovernmental Panel on Climate Change to assess data quality (Mastrandrea *et al.* 2010).

Table 4 Criteria and scores used to calculate fishing estimate certainty

Score		-%	+%	Corresponding IPCC criteria*
4	Very high	10	10	High agreement & robust evidence
3	High	20	20	High agreement & medium evidence **or** medium agreement & robust evidence
2	Low	30	30	High agreement & limited evidence **or** medium agreement & medium evidence **or** medium agreement & robust evidence.
1	Very low	40	40	Less than high agreement & less than robust evidence

*****Mastrandrea *et al.* (2010)

2.2.5 Calculating catch per unit of effort (CPUE)

For each of the EEZs included, CPUE was calculated using the reconstructed catches reported by the *Sea Around Us.* Catches are reported by sector, and it became evident that there were some discrepancies between fishing effort and estimate reconstructed catches, i.e., effort reported where there was no catch estimated. In order to ensure that the CPUE calculated used data that could be compared, the CPUE was only calculated for the sectors where both catch and fishing effort overlapped. To calculate CPUE, catches were first converted from tonnes to kilograms and then divided by fishing effort to estimate CPUE in kg ($kW\cdot day^{-1}$).

2.3 Results

Overall, total fishing effort in the 9 countries considered here was over 20 times higher in 2010 than it was in 1950 (Figure 1). Fishing effort increased more rapidly in the first half of the time series, and appears to have remained relatively stable since the early 2000s (Figure 1). Industrial fishing is the primary driver behind total fishing effort trends and was responsible for about 72% of total fishing effort in 2010; this composition does not differ substantially from industrial fishing levels in 1950 where the industrial sector made up about 70% of total fishing effort. The gear type with the highest fishing effort throughout the study period was trawling, accounting for 60% of total fishing effort in 2010. In contrast, the trawl fleet accounted for 40% of total fishing effort in 1950.

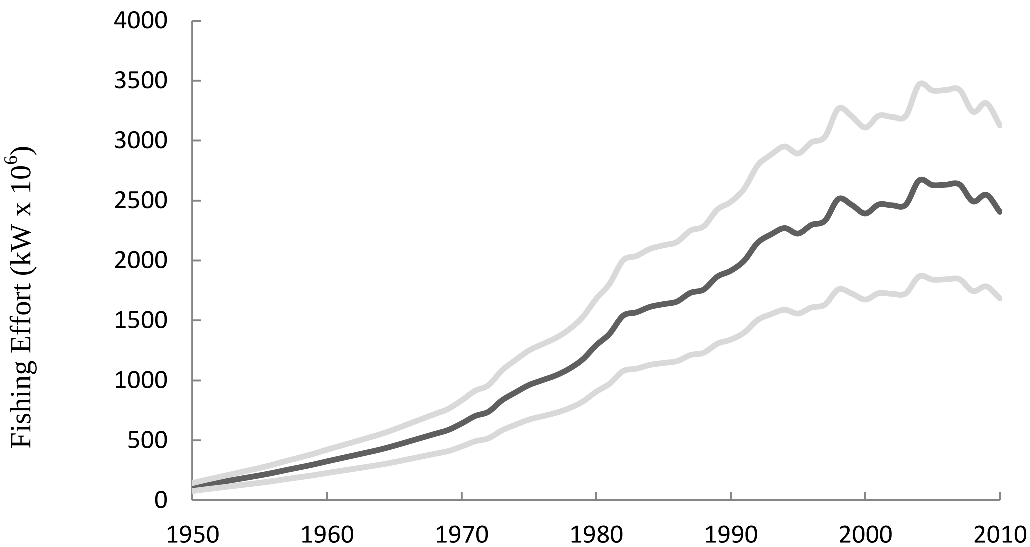

Figure 1 Total fishing effort from 1950 - 2010 as indicated for Algeria, American Samoa, Australia, Bahamas, Bahrain, Bangladesh, Myanmar, Cambodia and Panama. The grey lines correspond to the uncertainty of the fishing effort estimate.

2.3.1 Fishing effort by EEZ

<u>Algeria</u>

Fishing effort exerted by Algeria in 2010 has more than tripled since 1950 (Figure 2) with the vast majority of fishing effort a result of the industrial sector. The two primary gears used in the industrial sector were trawlers and purse seiners. For most of the time period, 1960 – 2000, the majority of fishing effort was exerted by trawlers. However, fishing effort from purse seiners have been increasing rapidly since 1990 and now contribute approximately the same amount of fishing effort as trawlers.

American Samoa

Fishing effort by American Samoa gradually increased up until the mid-1980s when it remained relatively unchanged until the mid-1990s (Figure 2). Starting in the mid-1990s, fishing effort increased again but much more quickly, doubling in about 5 years and is probably a result of industrial longlining for tuna. Since the early 2000s fishing effort has steadily declined (Figure 2) but fishing effort is still over 20 times what it was in 1950. There are three sectors included in the database: artisanal, subsistence and industrial. However, trends in fishing effort are largely driven by changes occurring in the industrial sector which accounts nearly 60% of total fishing effort in 2010. The industrial sector did not begin until the early 1980s; pre-1980 subsistence fishing was the dominant sector accounting for approximately 77% of total fishing effort in 1982. In 2010 the dominant gear type employed was longlining in the industrial sector which was accountable for about 62% of total fishing effort in 2010.

Australia

Australia exhibited a steep increase in fishing effort over a 20 year period from 1970 – 1990, but has since remained relatively stable (Figure 2). All of the vessels for Australia were labelled as industrial; this was done because the characteristics of the fisheries and technological capabilities of the smaller vessels in Australia are more like the industrial sector than they are artisanal. The trend is largely driven by a single, large industrial trawl fleet that accounted for 77% of total fishing effort in 2010, and which began operating in 1965.

Bahamas

Fishing effort has steadily increased since 1950, and was approximately 2.5 times greater in 2010 than it was in 1950 (Figure 2). The trends in fishing effort are largely driven by the artisanal sector, with the industrial sector only markedly contributing to fishing effort post 1980. Still it contributes only a quarter of the total fishing effort in 2010. Specific gear data for fisheries in Bahamas was difficult to ascertain, as most fisheries are multi-gear. Fishing for spiny lobster using casitas (a type of trap) is thought to be the main gear type used but the gear is recorded as “Not Known” or “Multipurpose” due to the multi-species nature of artisanal fisheries in this region.

Bahrain

Fishing effort in Bahrain steadily increased between 1950 and 1990 where fishing effort was about 6 times more in 1990 than 1950 levels (Figure 2). Post- 1990 fishing effort shows no discernible increasing or decreasing trend, but appears to be less stable (Figure 2). Similarly to the Bahamas, fishing effort trends in Bahrain are largely driven by the artisanal fishing sector. A small amount of industrial fishing occurred between 1966 and 1988, but there is no evidence that could be found at the time of this writing that an industrial sector existed outside this time period. Trawl boats are largely driving the increasing trend, whereas the marked increase in fishing effort observed in the early 1990s is due to a decrease in the number of trap fishers and an increase in the number of trawlers.

Bangladesh

Bangladesh’s fishing effort in 2010 is approximately 11 times what it was in 1950, steadily increasing throughout the time period (Figure 2). In the early 1980s, what appears to be a sudden increase and then decrease in fishing effort is a result of changes in a single, large vessel fleet; first an increase of vessels by 10, followed by a halving of the number of vessels in the following 2 years. Other than for these few years in the early 1980s, increases in fishing effort are largely driven by the artisanal sector, which accounted for about 79% of total fishing effort. Industrial fishing did not begin until 1970, when it increased until the early 1990s and has since remained relatively stable. The gear type which exerted the most fishing effort were gillnets, which accounted for approximately 38% of total fishing effort in 2010. This trend is true for most of the study period, apart from in the early 1980s when there were a several very large trawl boats operating.

<u>Cambodia</u>

Fishing effort in Cambodia has steadily increased throughout the study time period, with the exception of the Khmer Rouge Regime where fishing effort stagnated. In 2010 fishing effort was 12 times what it was in 1950 (Figure 2). Industrial fishing did not start until the early 1970s, but even then, the driving sector was the artisanal sector; the artisanal sector contributed 10 times more effort than the industrial sector did in 2010. The gear breakdown is more varied in Cambodia, but in 2010, the dominant gear used in terms of fishing effort were trawlers, accounting for about 25% of total fishing effort.

<u>Myanmar</u>

Fishing effort by Myanmar steadily increased between 1950 and 1998 when it decline and has since remained relatively stable (Figure 2). Industrial fishing began in 1955 and is the driving sector behind fishing effort trends throughout the study period and in 2010 accounted for about 75% of total fishing effort. Despite a reduction in fishing effort in the late 1990s, fishing effort in 2010 was 125 times what it was in 1950. Trawlers exert the highest amount of fishing effort for most of the time period, and in 2010 accounted for 45% of total fishing effort.

<u>Panama</u>

Panama’s fishing effort steadily increased between 1950 and the early 1990s when it began to stabilize (Figure 2). The final year included in the study time period suggests that Panama’s fishing effort might be declining (Figure 2). Despite the potential decline, fishing effort in 2010 was 58 times 1950 levels, the majority of which occurs on the Pacific coast. The industrial fishing sector is the primary driver behind fishing effort trends, with the artisanal sector only responsible for approximately 18% of total fishing effort in 2010. Trawling is responsible for the vast majority of fishing effort throughout the time period and in 2010 accounted for 77% of total fishing effort.

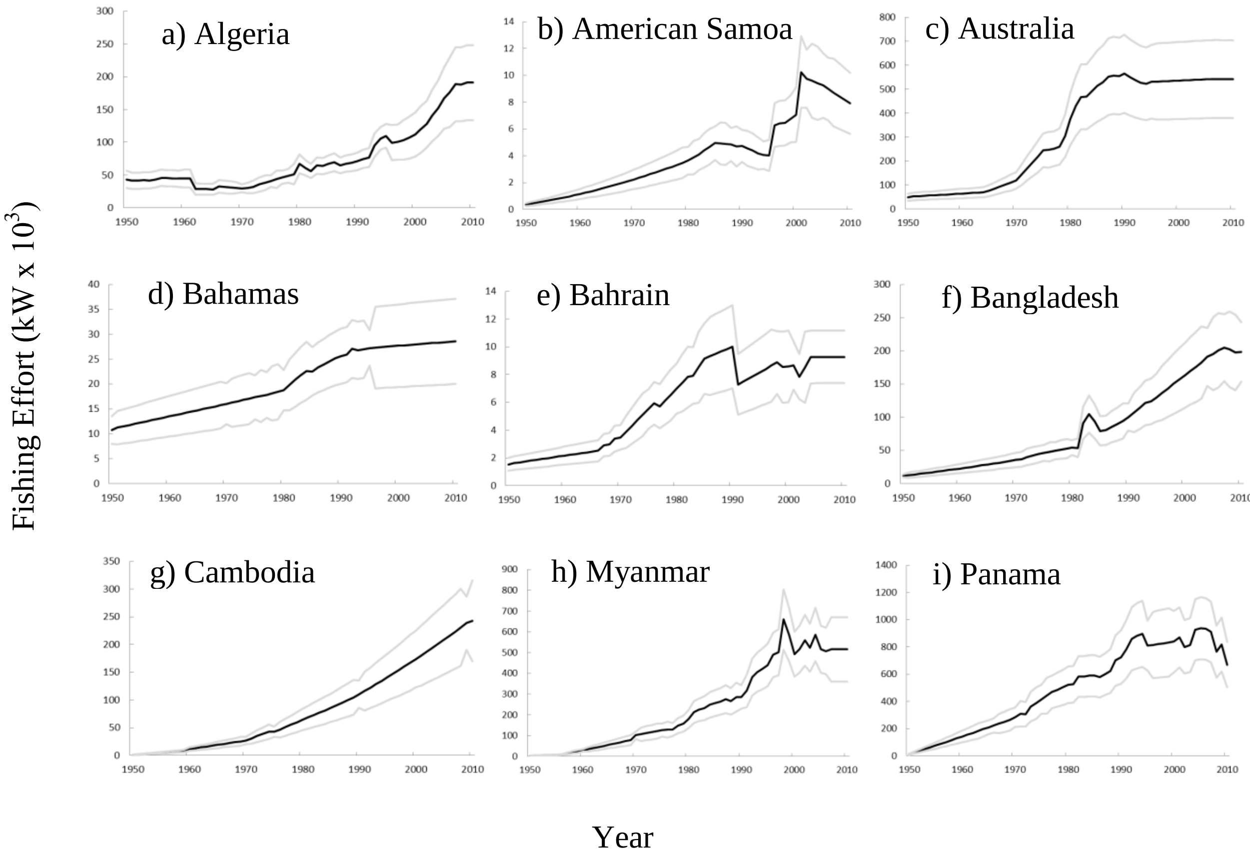

Figure 2 Fishing effort for each of the individual EEZs included in the analysis. The grey lines represent the annual average uncertainty of the variables used to estimate fishing effort.

2.3.2 CPUE by EEZ

CPUE was found to be declining for 7 out of the 9 EEZs included; Panama's CPUE fluctuated greatly throughout the study period and there is no real discernible trend, but it appears that it may be declining (Figure 3; Table 4). Similarly, Algeria's CPUE was relatively stable throughout the time period, but may be about to decline (Figure 3). In the EEZ with the greatest change, CPUE is over 100 times less in 2010 than it was in 1950; this occurred in American Samoa (Table 4). The EEZ which changed the least over the study period was Bahamas and is also the only EEZ where the CPUE increased (Table 4; Figure 3). Whether or not the industrial fishing sector or the small-scale sector dominated fishing effort did not seem to affect CPUE; the number of EEZs where CPUE declined and industrial fishing was the dominant sector in terms of effort was 5, and the number of EEZs where CPUE declined and small-scale fishing was the dominant sector in terms of effort was 4 (Table 4).

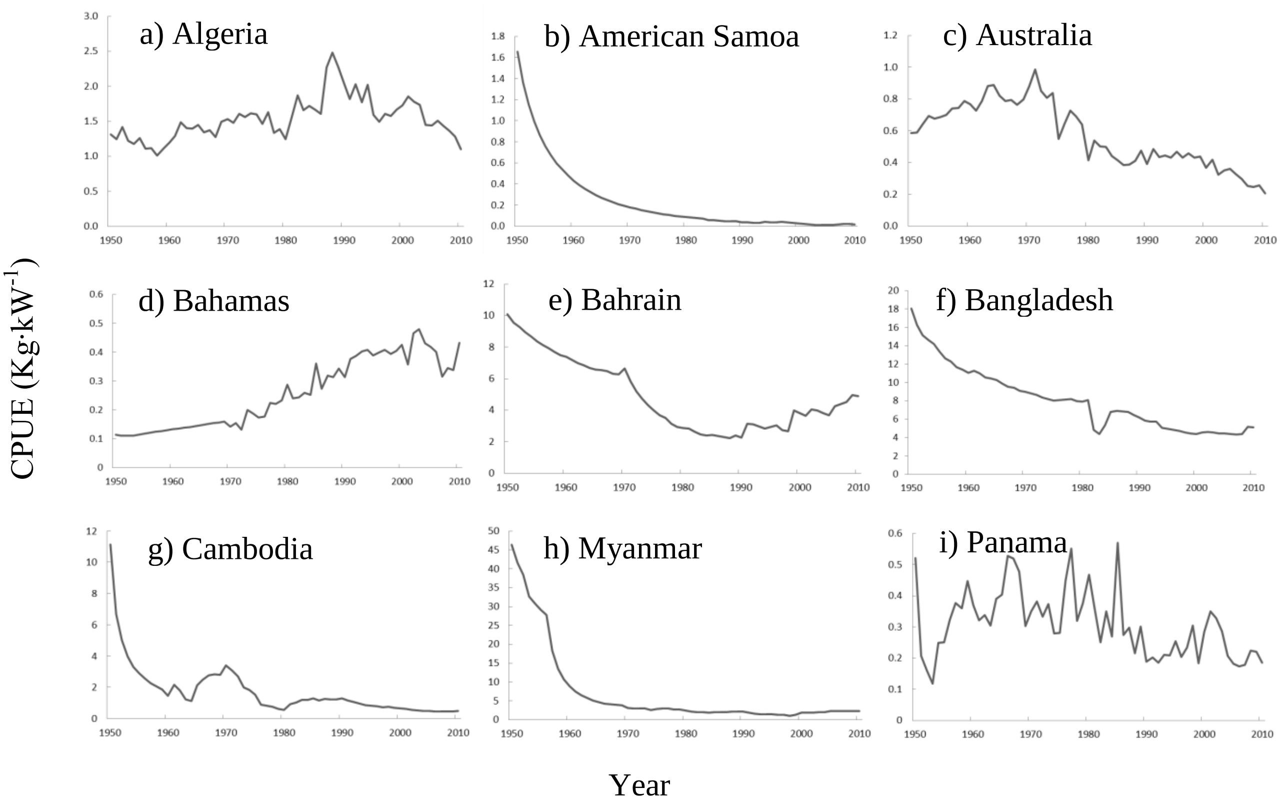

Figure 3 CPUE for each of the EEZs from 1950 - 2010.

Table 5 Changes in CPUE between 1950 and 2010 for EEZ included in the analysis

EEZ	CPUE 1950	CPUE 2010	Ratio	Direction of change	Sector*
Algeria	1.31	1.10	1.19	Negative	Industrial
American Samoa	1.66	0.02	108.59	Negative	Industrial
Australia	1.02	0.27	3.78	Negative	Industrial
Bahamas	0.11	0.43	0.26	Positive	Small-Scale
Bahrain	10.08	4.89	2.06	Negative	Small-Scale
Bangladesh	18.04	5.12	3.53	Negative	Small-Scale
Cambodia	46.36	2.24	20.74	Negative	Industrial
Myanmar	11.12	0.50	22.46	Negative	Small-Scale
Panama	0.52	0.18	2.82	Negative	Industrial

*is the sector responsible for the majority of total fishing effort during the study period

2.4 Discussion

The globalisation of fisheries over the last 150 years represents a critical change in the interaction between human societies and the ocean. Historically the world's oceans have been considered an infinite, open resource; this notion has contributed to today's present problem of overfishing. It is widely understood now that marine resources are not limitless, and that they need to be managed accordingly. But the problem of what is sustainable is a difficult one and traditional management plans rooted in consumption-based statistics assume incorrectly that reductions in consumption equal reduced environmental impacts (Davidson *et al.* 2014). While it is important to know how much of a resource is being used, it does not provide adequate detail on how exploitation affects the environment. Rather, the sustainability of a natural resource depends on the effort factor: the relationship between how much energy is expended to obtain said resource and the energy gained once extracted. Thus it was the purpose of this study to develop a methodology that could be used to estimate global fishing effort from 1950 – 2010 to determine fine-scale changes over time, a notable knowledge gap in fisheries science.

There are three main conclusions that can be drawn from the preliminary results of the 9 EEZs included in this chapter:

1. Fishing effort has increased over the study time period and may have reached its peak in the early to mid-1990s and has since stabilized;
2. CPUE for the most part is declining, and is substantially lower today than it was previously;
3. The impacts of sector on changes of CPUE appear to be minimal.

Curbing increases in fishing effort has been a recent objective in many of the world's major fishing countries (FAO 2003b). The preliminary results suggest that total fishing effort has been stable for about a decade suggesting that recent efforts, although not yet causing a decline in fishing effort, may at least be preventing fishing effort from continuing to rise. Although this is a possibility, given the observed individual trends of the EEZs included in the preliminary analysis, stabilization of total fishing effort is likely more a result of the distribution of fishing effort between EEZs rather than an indication of a true stable trend.

When examined individually total fishing effort was found to be increasing in the majority of EEZs. There were only three EEZs, Panama, Australia and Myanmar, that demonstrated a stabilizing trend in the final decade of the time period considered here. In contrast to the plans and policies of many of the world's largest fishing countries, such as those in Europe, many of the smaller fishing EEZs are continuing to develop their fisheries, despite indications of already declining stocks, for example, American Samoa (Zeller *et al.* 2006). Although artisanal fisheries contribute substantially less than the industrial sector to total fishing effort when looking at all EEZs combined, the effect of sector on the sustainability of fisheries appears to be the same once CPUE is considered.

CPUE is a relative measure of energy expended for energy returned and provides an indication of the sustainability of a natural resource. A stable CPUE, one where the amount of effort to gain 1 unit of the resources remains the same over time, suggests a sustainable level of exploitation. An increasing CPUE suggests that the resource productivity and/or standing stock is increasing and vice versa. An increasing CPUE is usually interpreted as a sign that the stock can be further exploited while a decreasing CPUE is indicative of overexploitation and unsustainable resource use. Using CPUE as a measure of sustainability is not always accurate, i.e., when fishing mortality and fishing effort are not proportional; in other words using CPUE as a measure of sustainability for any density-dependent stocks may result in overexploitation even if CPUE appears to be stable. This is known as hypo- or hyperstability. Hyperstability occurs when the density of a stock remains the same, but the area that the stock occupies is shrinking. A hyperstable CPUE is usually related to stock abundance at the local scale but not at the regional scale (Rose and Kuka 2011; Ulman *et al.* 2014). Hypostability is the opposite whereby the biomass of a stock is increasing but CPUE is not and would occur if the range is expanding. The extent to which either hypo- or hyperstability affect the CPUE results of this study are thought to be minimal. This is because the CPUE estimated is for the entire fishery of an EEZ, and thus is less dependent on any one stock that might be exhibiting density-dependent changes. Unless there was some factor, i.e., climate change, that was affecting the ranges of all stocks in the same way, hypo- and hyperstability are not likely actors in the overall declining CPUE trends.

It is possible that the spatial expansion of fisheries may result in some hyperstability in CPUE trends, particularly for Bahamas, as it was the only EEZ with an increasing CPUE. It is unlikely that the remaining EEZs are hyperstable due to spatial expansion of fisheries because their CPUE trends are declining. It is possible that spatial expansion might be masking the degree of the decline. In which case the effect of fishing on marine ecosystems and the sustainability of fisheries as they are is even less than what has been estimated in this analysis.

Often overexploitation is associated with large-scale, commercial operations, especially susceptible are common-property resources like fisheries, when an inappropriate economic rent (future value) has been applied to the population, leading to a situation that encourages overexploitation (Clark 1973). Current literature encourages community-based management, where the local population who have a vested interest in the long-term state of resources, are better equipped to sustainably manage local resources (Ruddle 1998; Blaikie 2006). As such it is often suggested that artisanal fishers, who by definition are local and small-scale, are more responsible and are likely to better manage the fisheries (Pomeroy 1996; Allison and Ellis 2001; Gelcich *et al.* 2005). It was thus an unexpected result that those EEZs where the majority of fishing effort was artisanal to have declining CPUEs to the same extent of those EEZs where industrial fishing is the dominant sector for fishing effort.

There are other reasons besides poor fisheries practices by artisanal fishers that might explain why EEZs where artisanal fishing effort dominates have equally declining CPUEs; one of those is illegal fishing. Illegal fishing is unauthorized fishing by foreign vessels in an EEZ, or by unauthorized domestic vessels within an EEZ and in 2011 was estimated to equal to approximately 20% of the world catch (Agnew *et al.* 2009). The distribution of illegal fishing is not even, and those EEZs most at risk are developing countries (Agnew *et al.* 2009). The EEZs dominated by artisanal fishing included in the preliminary analysis are, however, not developing with the exception of Bangladesh. Bangladesh does have a documented problem of illegal foreign fishing (see e.g., Karim 2014) supporting the hypothesis that illegal fishing may be the reason behind a declining CPUE of EEZs whose fisheries are dominated by local artisanal fishers.

Additional to illegal fishing, it is possible that a declining CPUE in EEZs dominated by artisanal fishing effort is an effect of *legal* foreign fishing where one EEZ permits fishing from other countries within its waters. The effort database does not capture the spatial distribution of fishing effort and is a limitation of the data; the effort estimated for a country is not necessarily applied within its own EEZ. As such the legal, industrial fishing conducted by a country within another country's EEZ potentially may be causing stress on local fish stocks and thus decreasing artisanal CPUE. The conflict between foreign industrial vessels and local artisanal fishers is well documented, especially in West Africa where the presence of foreign industrial fleets from Europe have severely impacted the stock status of marine resources and the socioeconomic conditions of local communities (Alder and Sumaila 2004). Similar to illegal fishing, the EEZs that are dominated by artisanal fishers included in the preliminary analysis do not appear to have excessive foreign access permits; Bahrain does not allow foreign fishing (FAO 2003a) and Bangladesh issues permits to around 75 foreign trawlers annually (Karim 2014). Myanmar, whose fisheries are dominated by industrial fishing effort, recently issued a statement that they will no longer be issuing foreign fishing permits due to fears of declining fish stocks (Aung 2014).

The above two explanations regarding declining CPUEs in EEZs dominated by artisanal fishing effort - the presence of illegal and legal foreign fishing – support the idea that industrial fishing practices tend to drive overexploitation of fish stocks. It does seem a probable explanation, and is consilient with the current fisheries management view of artisanal fishers are better stewards of local marine resources. Historically however, humans have been overexploiting and mismanaging marine resources including cases were the only fishers were small-scale (for examples see: Jackson 1997; Richter *et al.* 2008). In reality, the onset of industrial fishing has probably exacerbated already overfished ecosystems that were not identified as such due to the shifting baseline syndrome.

It is difficult to draw conclusions on the sustainability of fisheries based on the analysis presented in this thesis due to the small sample size. The EEZs included represent a wide range of fishing countries. However, they were only responsible for approximately 2% of the total world catch in 2010. It is the aim of the *Sea Around Us* to complete the database for all fishing countries, but it is beyond the scope of this thesis to do so. Once the database is completed, the trends that were illuminated in the preliminary analysis should be re-examined.

Chapter 3: Developing a methodology for estimating CO_2 emissions of the world's fishing fleet 1950 – 2010

3.1 Introduction

Since 1880, the mean global temperature has risen by approximately 1°C as a result of anthropogenic carbon dioxide emissions (IPCC 2013), which is now affecting the geophysical systems that underpin all life on Earth. The most recent results presented by the Intergovernmental Panel on Climate Change (IPCC) suggest that continued warming of another 1°C in the next century is likely, and perhaps an even greater increase in global mean temperature may occur should CO_2 emissions continue unabated (IPCC 2013). The last 20 years of climate change research has advanced quickly and become increasingly complex. However, the idea of humans adding to the naturally occurring greenhouse effect occurring in the Earth's atmosphere was first discussed over 100 years ago (Arrhenius 1896).

3.1.1 The evolution of climate change science

The hypothesis that humans could potentially alter the global climate through the emissions of certain gases, particularly carbon dioxide, had already been proposed by the 1900s (Kellogg 1987). But with little data and techniques available to test the potential impact of CO_2 emissions the idea was not paid much attention at all until about 60 years later when Charles Keeling and his associates, began researching and measuring the levels of CO_2 in the atmosphere (Keeling *et al.* 1976). At this time, Keeling *et al.* (1976) established that the concentration of CO_2 in the atmosphere was increasing on a global scale, but the effects were not yet known. Hypotheses included both a global warming scenario and a global cooling scenario, and scientists were split between the two 'camps'; those who considered global cooling an option were unsure of the role of other GHG, such as chlorofluorocarbons (CFCs), and it was possible that aerosols because of a light-scattering effect might result in a cooling of the Earth's surface (Kellogg 1987). Studies have since concluded that aerosols do not have a cooling effect, and, in fact contribute to global warming (Kellogg 1980).

Throughout the 1980s, studies predicting global warming began to increase and include information on temperature changes on a geological timescale (for example Berner *et al.* 1980; Oeschger *et al.* 1982). Following research published in the early 1980s, governments from around the world began to take an interest in the idea of global warming and the IPCC was established; the panel was charged with the mandate to determine whether or not global warming was occurring, and if it were, to determine the impacts and potential mitigation strategies. In 1990, the first IPCC report was released concluding that the globe was getting warmer and it would likely continue to do so. However, it was not until the second report, released in 1995, that the IPCC suggested that it was very likely that global warming was a result of human activity on the planet (IPCC 1995).

By the mid-1990s, the scientific community was reaching a consensus: the Earth was getting warmer and it was doing so because of humans. Research expanded to begin looking at other effects of global warming such as extinction risk of various animal and plant species, rising sea levels and ocean acidification. It became increasingly clear that the effects of increased GHG emissions in the atmosphere extended far beyond an increase in global mean temperature and that mitigation measures to reduce global warming are uncomfortably linked with a prosperous economy; the primary driver behind CO_2 emissions is the combustion of fossil fuels (Wilson 1999) and economic growth over the past 40 years is highly correlated with increases in oil consumption (Murphy and Hall 2011). Consequently it is assumed by many that reducing CO_2 emissions is costly and signifies reduced economic prosperity.

The necessary immediate action that is required to mitigate climate change has many industries evaluating which scenarios yield decreases in GHG emissions for the lowest cost. As a result there have been a lot of studies conducted to see which sectors contribute the most to climate change. For example, the agricultural sector in the United States contributed approximately 10% of total USA GHG emissions in 2012 (EPA 2014c). The FAO reports that the agriculture sector is responsible for approximately 1/3 of the world's total GHG emissions (FAO 2008). These figures have spurred the initiation of organizations such as the Clean Development Mechanism that earns carbon credits for eligible projects (often farmers) in developing countries which can then be sold to large companies exceeding their carbon allowance in industrialized nations. But keeping track of carbon has proven to be difficult; early results of carbon trading schemes are disappointing. No net decrease in CO_2 emissions can be detected as yet, and the carbon market remains highly volatile (Spash 2010).

The world's fleet of merchant ships over 100 GT is estimated to be about ~45,000 vessels (as cited in Dalsoren *et al.* 2009). The international marine shipping industry, a large boat-based sector similar to industrial fishing vessels, is estimated to contribute approximately 3% to global CO_2 emissions (Dalsoren *et al.* 2009; Psaraftis and Kontovas 2009). There are an estimated 1.3 million decked fishing vessels (as cited in Endresen *et al.* 2007); these vessels are substantially smaller than merchant ships, but their sheer number may result in a similar contribution to GHG emissions. One report has suggested that fishing has a larger climate impact than long-distance transports (Ziegler 2009). Another study, which was previously discussed in detail in Chapter 1, suggested that fishing contributed approximately 1% to total GHG emissions, but that it had a much smaller carbon footprint than other sources of protein such as chicken or beef (Tyedmers *et al.* 2005).

There is a widespread appreciation for the need to reduce GHG emissions and a number of large industrial sectors are now required to reduce their emissions. Fisheries have not yet been evaluated on a large scale, and are thus not incorporated into reduction plans. Fisheries, as an economic sector, are not perceived as a significant contributor to CO_2 emissions to date. However, preliminary analyses suggest that fishing contributes approximately 1-3% of annual global CO_2 emissions. Thus, the time to evaluate CO_2 emissions from fishing and subsequently better understand the sustainability of fish as food is now.

3.1.2 Fisheries and climate change

The discussion regarding the relationship between fisheries and climate change has focused on how fisheries will be affected by climate change, not vice versa. Research indicates that global warming will decrease primary production, redistribute fisheries resources, and decrease biodiversity as well as increase economic impacts and vulnerability of already vulnerable nations, mostly tropical island countries (e.g. Allison *et al.* 2009; Cheung *et al.* 2009; Chassot *et al.* 2010; Cheung *et al.* 2010; Pauly *et al.* 2014). Given that future warming is inevitable and that fisheries are likely to be severely impacted, mitigation research should be a management priority. Subsequently, it is equally important to understand not only how fisheries are impacted by global warming, but how the industry contributes to the problem as well.

The two most comprehensive studies estimating CO_2 from fisheries suggest that fishing contributes significantly to global GHG emissions and were already discussed in Chapter 1. To recap, the study done by Tyedmers *et al.* (2005) provided a snapshot of CO_2 emissions associated with how much catch was reported in official statistics and the second study discussed estimated CO_2 emissions over a substantial time period, but the focus was not on fishing boats; the study investigated all vessels greater than a certain size, some of which happened to be fishing boats (Endresen *et al.* 2007). There have been smaller studies that focus on specific fisheries tracking fuel and energy use. For example, a study done on Norwegian fisheries was able to compare fuel use coefficients between different gear types and found that trawlers were the least efficient, while purse seiners were the most fuel efficient per tonnage caught (Schau *et al.* 2009b). Similar results were found in the North Atlantic (Tyedmers 2001) and Sweden (Ziegler and Hansson 2003).

The above studies used methods that relied on consumption-based indicators; the input parameters were often fuel used (acquired through sales statistics) and reported catches. In some cases, fuel use was reported by the fisher through surveys, a method that might be slightly more accurate at least for the specific fishery than data acquired through sales statistics (for example Ziegler and Hansson 2003). Consequently the results differ greatly from study to study. For example, the study conducted by Tyedmers *et al.* (2005) found that the global average was 1.7 tonnes of CO_2 was emitted per tonne of live weight catch whereas the life-cycle assessment conducted by Ziegler and Hansson (2003) estimated that there was only 2.7 kg of CO_2 emitted per tonne of reported catch. The Ziegler and Hansson (2003) study looked specifically at the Swedish cod fishery and given that the authors estimated the carbon footprint for the entire production chain, and that the fleet is a high-powered, industrialized fleet, their estimate is surprisingly low.

Current studies are focusing on the life-cycle assessment (LCA) approach to determine the environmental footprint of food systems, including fisheries. The LCA approach attempts to include both direct and indirect environmental effects of food procurement, processing and delivery. One of the advantages of the LCA method is that one can see how different steps contribute individually to the impact; consequently a sort of cost-benefit analyses can be done to evaluate where efforts can have the greatest impact on reducing GHG emissions. LCA studies conducted thus far on fisheries have found that the direct combustion of fuel during fishing is the greatest contributor to GHG emissions from fishing (Daw *et al.* 2009; Sund 2009; Winther *et al.* 2009). LCA studies suggest that the carbon footprint of marine capture fisheries is smaller than other proteins such as beef and chicken, but there is still room for improvement. Additionally comparing the environmental impact of fishing (hunting wild animals) and farming (cultivation of livestock) is not necessarily informative (Sonesson *et al.* 2010).

The importance of understanding the carbon footprint of one's food lies in sustainability; there are currently 7 billion people on the planet and the population has not yet stabilized. Feeding a growing population is becoming challenging based on current resource use, and climate change is a source of uncertainty. At the writing of this thesis, the focus on fisheries and climate change has largely been centred on how climate change will affect fisheries, and not on how fisheries contribute to climate change. The studies that have been conducted do suggest fishing as an important GHG emitter, but have thus far used mostly consumption-based statistics such as fuel sales or reported catches to estimate CO_2 emissions. Such statistics are assumed to be indicative of environmental impact, but are in reality often ill-reported and unrepresentative; it is better to use effort as the relationship between effort and environmental impact is more causal (Davidson *et al.* 2014). The following section outlines in detail the methodology proposed to estimate how fishing has contributed to GHG emissions since 1950 by using fishing effort estimated in the Chapter two.

3.2 Methods

3.2.1 Fishing effort, fuel consumption and GHG emissions

GHG emissions for marine vessels are correlated with fuel consumption and it is possible to estimate fuel consumption from fishing effort (Latoree and Cardella 2008). The fishing effort calculated in Chapter 2 used nominal fishing power, estimated from length, multiplied by the number of fishing vessels, days spent fishing and a technological coefficient to calculate effective effort: the energy expended over a given unit of time ($kW \cdot days^{-1}$). Estimating CO_2 emissions can be done in a similar fashion using the nominal fishing power to first estimate fuel consumed per vessel in a given fleet; this number can subsequently be multiplied by an emissions factor to yield the effective CO_2 emissions from a given fleet.

3.2.2 Estimating nominal fuel consumption

The amount of fuel consumed by a vessel depends on many factors, both operational and technological (Wilson 1999; Gulbrandsen 2012). Additionally, many vessels continue to operate with engines that are decades old (Thomas *et al.* 2009; Abernethy *et al.* 2010) and implementation of fuel saving technologies likely varies depending on the country fishing, and sector. Still it is important to consider the potential changes in engine efficiency over time as this will affect the amount of CO_2 emitted. In order to convert fishing power to fuel consumption and take into account changes in fuel efficiency overtime, two variables are required: Specific Fuel Consumption Rate (SFR) and a Fuel Coefficient (FC).

The fuel consumption rate depends largely on the type of engine employed and the fuel used. Fishing vessels are usually equipped with a diesel engine (either a 2- or 4-stroke; common to industrial sector) or a spark-ignited Otto cycle engine, a.k.a., gasoline engine, which can also be 2- or 4-stroke; used most common in small-scale fisheries (Corbett and Koehler 2003). Stroke refers to the movement of the piston in the engine, and in general a 4-stroke engine is more fuel efficient. However, a 2-stroke engine is cheaper to manufacture and much lighter; the smaller weight of the 2-stroke engine can increase overall fuel economy because a lighter fishing vessel consumes less fuel. The industrial sector in this thesis was assumed to be equipped with a marine diesel engine because the studies that have thus far estimated GHG emissions from fishing have used marine diesel fuel consumption rates. In particular, a study published by Corbett and Koehler (2003) indicated that over 70% of fishing vessels are equipped with diesel engines; the study only looked at what this thesis defines as industrial fishing vessels. Corbett and Koehler (2003) also found that the overwhelming majority of diesel engines in the fishing fleet were 4-stroke. A literature search was conducted and the SFR used for industrial fishing was 0.2 kg of fuel burned per kW hour (Table 5). It should also be noted that there are different kinds of diesel oil used throughout the industry that subsequently also affect fuel consumption. These can mainly be broken into three categories: distillate, intermediate and residual (EPA 1999). Industry usually refers to the intermediate category simply as 'marine diesel" and during the literature search, marine diesel was the most common type of fuel referred to.

The above paragraph describes the SFR for industrial fishing vessels only. Artisanal engine characteristics differ from industrial fishing vessels; they are generally smaller, older and are often gasoline propelled as opposed to diesel. As a result, small-scale fishing vessels tend to be less fuel efficient but emit less carbon per unit of fuel burned. It was difficult to find specific SFR data for artisanal and subsistence fisheries as the majority of studies focus only on fuel economy in industrial fisheries. As a result the SFR used in this thesis for artisanal vessels is actually derived from recreational fisheries even though recreational fisheries differ than artisanal and subsistence fishing in many ways; The engine characteristics are, however, similar, i.e., gasoline propelled out-board engines of either the 2-stroke or 4-stroke design. A study by Winther (2007) suggested that a 4-stroke gas engine of the early 2000s has a SFR of 0.4 kg/kWh and a 2-stroke gas engine has a SFR of 0.35 kg/kWh (Table 5). Given that there was little detail on the breakdown of the engine size on the world's small-scale artisanal fleet, and the difference between the two engine classes is relatively small, the latter SFR was used in this study in order to remain conservative.

Table 6 References for Specific Fuel Consumption (SFR) and carbon dioxide emissions factors

Sector	SFR	Emissions factor	Study year(s)	Fuel type	Reference
Industrial	215 g/kWh	3.179 (t CO_2/t of fuel)	2004	Marine diesel	Dalsoren *et al.* (2009)
Industrial	212 g/kWh	3.170 (t CO_2/t of fuel)	2004	Marine diesel	Latoree and Cardella (2008)
Industrial	200 – 240 g/kWh	Not stated	1925 – 2002	Marine diesel	Endresen *et al.* (2007)
Industrial	Not stated	3.17 (t CO_2/t of fuel)	1999	Marine diesel	Ziegler and Hansson (2003)
Industrial	170 – 260 g/kWh	Not stated	1990 -2005	Marine diesel	Winther (2007)
Small-scale	Not stated	2.33 kg CO_2/L of fuel	2014	Petroleum	EPA (2014a)
Small-scale	Not stated	2.4 kg CO_2/ L of fuel	2014	Unleaded gas	CarbonFund (2014)
Small-scale	Not stated	3.0058 t CO_2/ t of fuel	2013	Petroleum	CarbonTrust (2013)
Small-scale	7 L/h	Not stated	2010	Petroleum	Gulbrandsen (2012)
Small-scale	0.4 kg/kWh*	Not stated	1990 – 2005	Petroleum	Winther (2007)
Small-scale	0.35kg/kWh**	Not stated	1990 – 2005	Petroleum	Winther (2007)

* 4-stroke gas engine common of recreational fishing boats;
** 2-stroke gas engine common of recreational fishing boats.

The SFR for industrial and artisanal vessels represents the fuel consumption at a given point in time, and engine efficiency has changed dramatically with advances in new technology since the 1950s. In order to account for changes in SFR over the time series a fuel coefficient (FC) was developed. Using time series fuel consumption data from Yanmar engines (1960 – 2001; Figure 4), an international builder and supplier of marine engines established in 1912 (www.yanmarmarine.eu), the SFR from 1950 – 2010 was estimated using linear interpolation between raw data points; the SFR values for 1950 – 1959 were the same as 1960, and those from 2002 – 2010 were also assumed to be constant at the 2001 rate. The year 2000 was set as the benchmark, i.e., the FC for the year 2000 = 1. The FC was then calculated by dividing the SFR for a given year by the SFR for the year 2000 to yield the change in SFR over time. Appendix B presents the original data in detail. The results are corroborated by a report done for Denmark fisheries that yielded similar changes in SFR over time (Winther 2007). The same FC was used for all sectors.

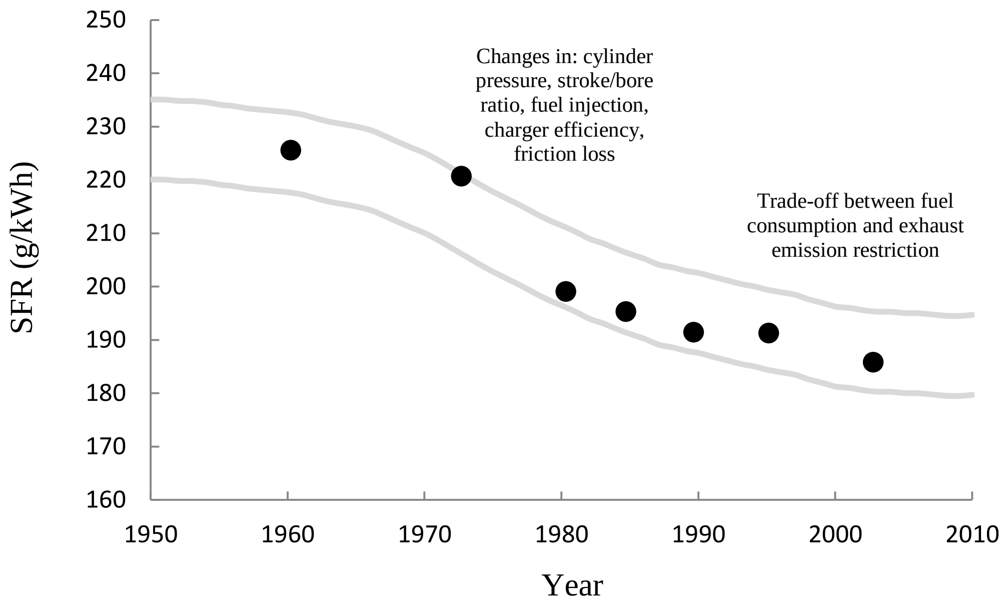

Figure 4 Changes in Specific Fuel Rate (SFR), both the upper and lower boundaries (represented by grey lines) and data points (represented by black dots) overtime using Yanmar engine data. Text describes technological changes accounting for changes in SFR (note: the y-scale does not start at origin).

The final factor that needed to be included to calculate nominal fishing fuel consumption was how many hours per day the engines are being run (H). This varies greatly between fisheries, not only regionally but also between sectors and gear types. For example many of the industrial fisheries operate 24 hours a day (Winther *et al.* 2009), whereas artisanal fishers are likely to return home at the end of the day. Another example that would affect engine run time would be gear; some gears such as trawling require the use of the main engine during fishing, whereas other gears do not. It was not possible to evaluate the hours fished individually for each fleet, and in concordance with previous fuel consumption fisheries studies, an estimate of mean hours fished was applied dependent on sector. Previous literature has dealt only with industrial scale fishing and have assumed a hours fished rate of 24 hours a day (for example: Tyedmers 2001, 2004; Dalsoren *et al.* 2009; Thomas *et al.* 2009; Winther *et al.* 2009). The number of hours spent fishing in the small-scale sector has not been well-documented. It was assumed that the artisanal as well as the subsistence sector spent 12 hours per day fishing, and the recreational sector spends 8 hours per day fishing, with engines running.

To calculate the nominal fishing fuel consumption the following equation was used:

$$\text{Nominal FC} = \text{FE*SFR*FC}$$

FE is the fishing effort per vessel (kW); SFR is the specific fuel consumption rate; FC is the fuel coefficient. The above equation represents how much capacity is present in the world's fishing fleet, but is not a realistic calculation of how much fuel is consumed. The above equation must then be multiplied by a number of factors to better represent effective fuel consumption.

3.2.3 Calculating effective fuel consumption

To change annual nominal fuel consumption from fishing (described above) to effective fuel consumption, three factors need to be considered: the number of vessels (n), motorisation (M) and the number of days fished per year (DF). These three factors were included in the construction of the fishing effort database (discussed in Chapter 2) and were used to estimate effective fishing effort. The primary difference between effective fishing effort and effective fuel consumption is the absence of a technological coefficient in the latter. Advances in technology that increase fishing capacity were included in the fishing effort calculation but were not included in the calculation of fuel consumption; the technological coefficient is relevant only to fishing effort whereas the fuel coefficient is relevant to only fuel consumption.

The equation to calculate effective fuel consumption is thus:

$$\text{Effective FC} = \text{Nominal FC}*n*M*H*DF$$

Nominal FC refers to nominal fishing effort; n is the number of vessels operating in the fleet; M is what percentage of the fleet is motorised; H is the number of hours the engine is running per day and DF is the number of days spent fishing each year. The number of hours fished varied depending on sector; the industrial sector was estimated to run 24 hours per day, whereas the artisanal and subsistence sector was estimated to operate 12 hours per day. The recreational sector was estimated to operate 8 hours per day. The industrial sector was estimated to operate 24 hours a day primarily for two reasons, the first is that it is difficult to start large engines. Therefore once they are running, shutting them down is unlikely to happen until the vessel is docked. Secondly, fishing may not be happening at all times, but when a vessel is not fishing, it is likely to be cruising to the next fishing grounds. Additionally, the operation of auxialliary engines and the necessary refrigeration required of distant water fleets increases the amount of fuel consumed. In comparison, small-scale fishers return home at the end of each work day, and do not have refrigeration capabilities.

The overall goal was to develop a methodology to estimate CO_2 emissions from fishing effort by first estimating fuel consumed by fishing vessels from fishing effort. Unlike fishing effort, fuel consumed is correlated with GHG emissions and by including an emissions factor in the calculation CO_2 emissions for individual fishing fleets can be estimated

3.2.4 Estimating CO_2 emissions

Fuel types differ in their GHG emissions; usually less refined and heavier petroleum types generate higher emissions than those fuels that have been distilled and refined. When fuel consumption was calculated from fishing effort, two different fuel types were used depending on sector: the industrial sector fuel type was marine diesel (intermediate distillate) and the small-scale sectors fuel type used was gasoline. The majority of the research available on emissions factors are centred on road-based transportation fuel types, although jet fuel and shipping transportation emissions factors have also been considered (e.g., DeLuchi 1991; Lakshmanan and Han 1997; Endresen and Sorgard 2003). In road-based studies, diesel fuel usually has a lower carbon footprint than its gasoline engine counterparts, mostly because automobile diesel engines have better fuel economy, but produce more CO_2 per litre of fuel burned (www.world-petroleum.org). The same was found to be true in this study as well; the emissions factor for marine diesel used was 3.17 tonnes of CO_2 per tonne of fuel burned and the emissions factor for gasoline used was 3.006 tonnes of CO_2 per tonne of fuel burned (Table 6). CO_2 emissions per year and fleet of the EEZs studied here were thus calculated by multiplying the effective fishing effort by the appropriate emissions factor.

The majority of the uncertainty in the calculation of CO_2 emissions arises from the estimate of fishing effort. Therefore the same data quality values that were estimated for fishing effort were used to indicate uncertainty surrounding CO_2 emissions. See Chapter 2.2.4 for method details

3.2.5 Estimating CO_2PUC and global emissions

Similar to catch per unit effort, CPUE, it is valuable to understand how CO_2 emissions have changed over time per unit of catch. CO_2PUC was calculated for individual EEZs as well as for all EEZs combined and was calculated by dividing the respective total CO_2 emissions by the respective catch as estimated by the *Sea Around Us*. The same calculation was also conducted for both industrial and small-scale fisheries. These values were then used to tentatively estimate global GHG emissions from fisheries for 2010 by multiplying the CO_2PUC for industrial fisheries by the industrial catch, as well as multiplying the CO_2PUC for small-scale fisheries and summing the two products to get total CO_2 emissions:

$$\text{Global } CO_2 = (CO_2PUC_{industrial} * Catch_{industrial}) + (CO_2PUC_{small\text{-}scale} * Catch_{small\text{-}scale})$$

The above calculation was performed twice using two different global catch estimates: the FAO reported catch and the reconstructed catch by the *Sea Around Us*. The value for CO_2PUC (for both industrial and small-scale) used in the equation was the average CO_2PUC from 2000 – 2010.

3.3 Results

3.3.1 CO_2 emissions

Overall the amount of CO_2 emissions from fishing in 2010 for the 9 EEZs considered in the preliminary analysis is 16 times higher than it was in 1950. Approximately 33 million tonnes of CO_2 from these 9 EEZs was emitted into the atmosphere in 2010 as a direct result of fishing. The trend lines for CO_2 emissions mirror those of fishing effort reported in Chapter 2; this is because estimated CO_2 is dependent on fishing effort. Trawling, by virtue of dominating the fishing effort sector, is thus the greatest CO_2 emitter throughout the study time period. Similarly the industrial sector in 2010 accounted for approximately 75% of total CO_2 emissions.

3.3.2 CO_2 emissions by EEZ

Algeria

The CO_2 emitted by Algeria's fishing fleet was 2.8 million tonnes in 2010, triple the amount in 1950 (Figure 5). The vast majority of emissions originate from the industrial sector despite the small-scale sectors composing 62% of the fleet in terms of vessel numbers in 2010.

American Samoa

Unlike most of the other EEZs, the CO_2 emissions estimated for American Samoa in 2010 are not at or near peak CO_2 emissions estimated in the study time period, but they are still 24 times what they were in 1950 (Figure 5). In 2010, the CO_2 emissions from fishing were approximately 97 thousand tonnes, compared to about 4 thousand tonnes in 1950. Pre-1980 the majority of CO_2 emissions were a result of small-scale fishing, but that quickly changed in the early 1990s; despite the small-scale sector comprising 97% of the fleet in terms of vessel numbers, the industrial sector is responsible for 68% of CO_2 emitted in 2010.

Australia

The CO_2 emissions in 2010 were approaching 34 million tonnes, a figure that is 16 times more than the CO_2 emissions in 1950 (Figure 5). All the fishing that was included in the fishing effort database was labelled industrial given the technological capacity of the fleet, but it can be noted that less than 1% of the fleet (measured by number of vessels) is accountable for 46% of the total CO2 emitted in 2010; these vessels are very large with a length of 125 m.

Bahamas

The CO_2 emissions in 2010 approached 420 thousand tonnes in 2010, 2.5 times what they were in 1950 (Figure 5). As noted in Chapter 2, industrial fishing is the primary driver of fleet trends post-1980 and despite only composing 3% of the fleet measured by vessel numbers, the industrial fisheries contributed a third of the CO_2 emitted in 2010.

Bahrain

The CO_2 emitted from Bahrain's fisheries in 2010 was 134 thousand tonnes and in 1950 it was only 23.5 thousand tonnes, a nearly 6 fold increase throughout the study time period (Figure 5). Bahrain is relatively unique amongst the EEZs included in the preliminary analysis because it has very little industrial fishing. And even in years where industrial fishing is present (1968 – 1988) the vast majority of CO_2 emissions still arose from the small-scale sector.

Bangladesh

In 2010 the CO_2 emitted from Bangladesh's fisheries was estimated to be 2.5 million tonnes and has increased 14 times since 1950 when the estimated CO_2 emissions was approximately 177 thousand tonnes (Figure 5). Similar to Bahrain the trends in CO_2 emissions are driven by artisanal fisheries however; in 2010 the industrial fisheries composed less than 1% of the number of fishing vessels, but a quarter of the CO_2 emissions that year came from industrial fishing activities.

Cambodia

In 2010 the CO_2 emissions in Cambodia were estimated to be 3.1 million tonnes, and have grown from approximately 16 thousand tonnes since 1950 (Figure 5); the 2010 value is 1919 times the 1950 estimate for CO_2 emissions and is largely driven by continually growth in the artisanal sector (not including the regime of the Khmer Rouge where fishing apparently stagnated). Unlike many of the other EEZs, industrial fishing sector only contributed 9% to overall emissions in 2010, but it comprise only a very small proportion of the total fleet as measured by number of vessels, less than 1%

Myanmar

In 2010 fisheries operated by Myanmar emitted 5.6 million tonnes of CO_2 compared to only 62 thousand tonnes in 1950 (Figure 5). The industrial sector contributes the greatest to annual CO_2 emissions compared to the small-scale sector, approximately 72%, even though the number of vessels operating in the industrial sector comprises only 6% of the fleet.

Panama

The CO_2 emissions in 2010 were estimated at 9.5 million tonnes, a figure that is over 40 times what was estimated in 1950, a mere 219 thousand tonnes (Figure 5). 86% of 2010 emissions were a result of the industrial fishing fleet, but this fleet comprises only 7% of the total fleet as measured by number of vessels.

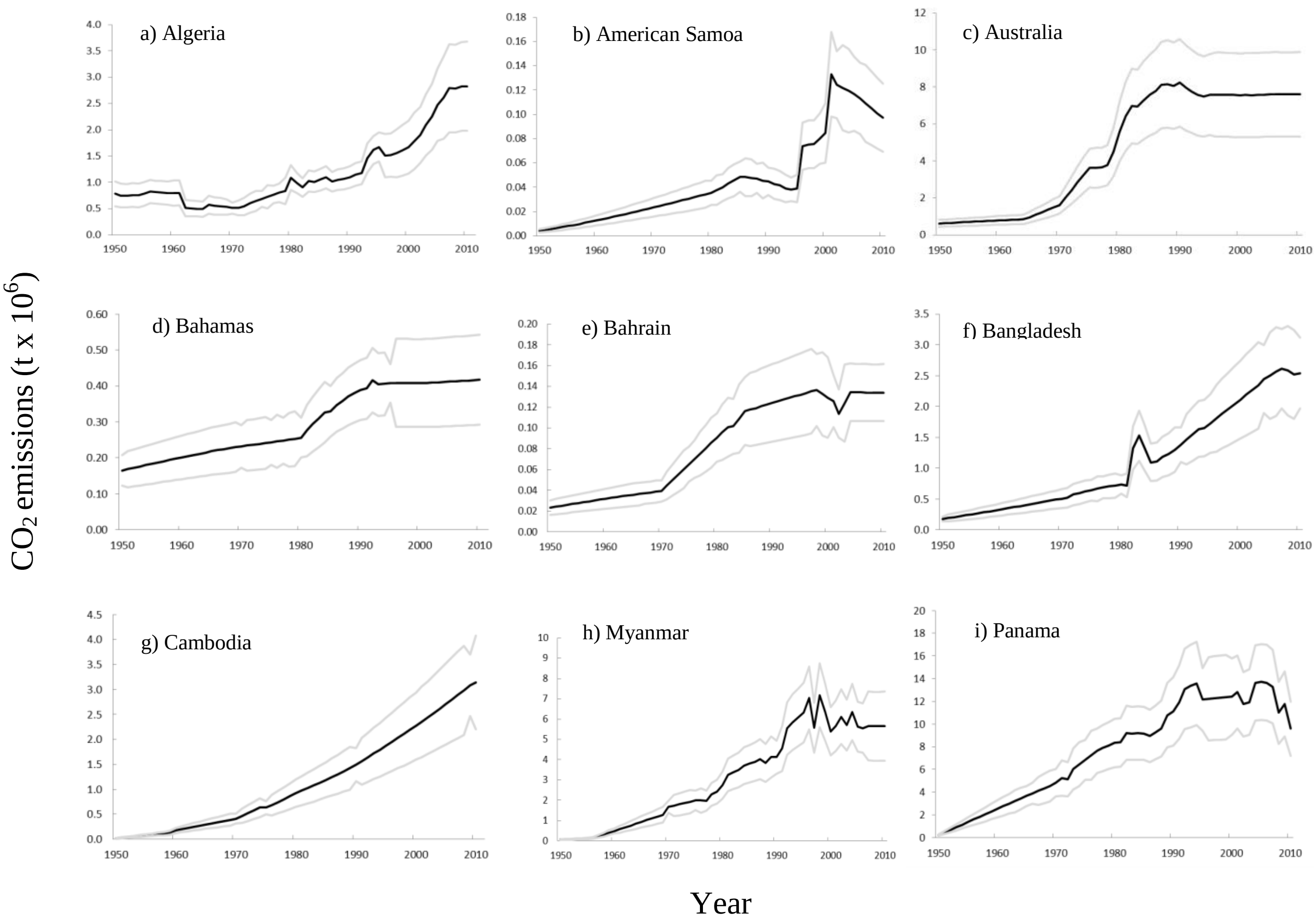

Figure 5 Annual carbon dioxide emissions (million tonnes) for each of the EEZs included in the analysis. The grey lines correspond to the average annual uncertainty of the variables used to estimate fishing effort, and are the same as those in Chapter 2

3.3.3 CO_2PUC by EEZ

For the majority of EEZs examined in this thesis, CO_2PUC has increased since 1950; the exceptions are Algeria, Panama and Bahamas (Figure 6). Algeria's CO_2PUC declined for the majority of the time, but has been increasing since the late 1990s and in 2010 has roughly the same CO_2PUC it had in 1950 of approximately 14 tonnes (Figure 6). In contrast Panama had a very high CO_2PUC early on in the study period, it declined quickly in the late 1950s and has since exhibited annual variation with some indication in more recent years that it might be increasing (Figure 6). Panama, although relatively stable, has the highest CO_2PUC with only Australia approaching similar values in more recent years. Bahamas is the only identifiable, long-term decreasing trend. But the change is small; a difference of 0.1 tonnes between 1950 and 2010 (Figure 6). The remaining 6 EEZs demonstrate increasing trends over the study period with Cambodia and Myanmar increasing the most, 25 times and 5 times respectively

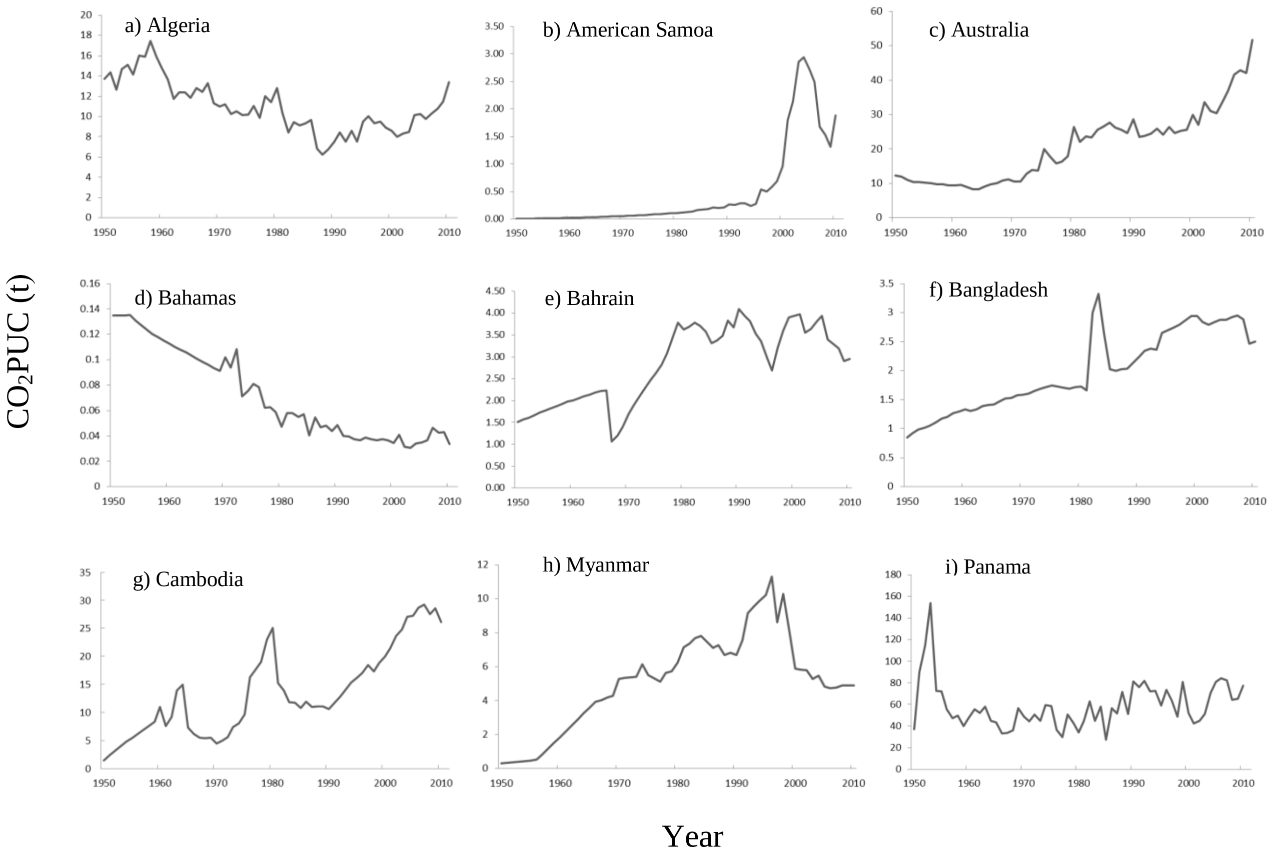

Figure 6 Annual carbon dioxide emissions per tonne of catch (in tonnes) for each of the EEZs from 1950 - 2010

In 1950 the CO_2 emissions per tonne of catch (CO_2PUC) was just less than 4 tonnes for the 9 EEZs included in the preliminary analysis. CO_2PUC steadily increased until about the year 1995, when CO_2PUC was approximately 14.5 tonnes. Since the late 1990s CO_2PUC appears to be declining and was approximately 9.5 tonnes (for industrial and small-scale fisheries combined) in 2010. The declining trend of CO_2PUC since the late 1990s is being driven mostly by the industrial fishing sector; when CO_2PUC is plotted for the separate sectors (industrial and small-scale) CO_2PUC for the small scale sector is increasing throughout the entire study period

(Figure 7). Despite the increasing CO_2PUC for small-scale fisheries, they still contribute substantially less CO_2PUC; in 2010 the small-scale sector had a CO_2PUC of approximately 5.5 whereas that of industrial fisheries was over 3 times as much, with a CO_2PUC of approximately 16.7 (Figure 7).

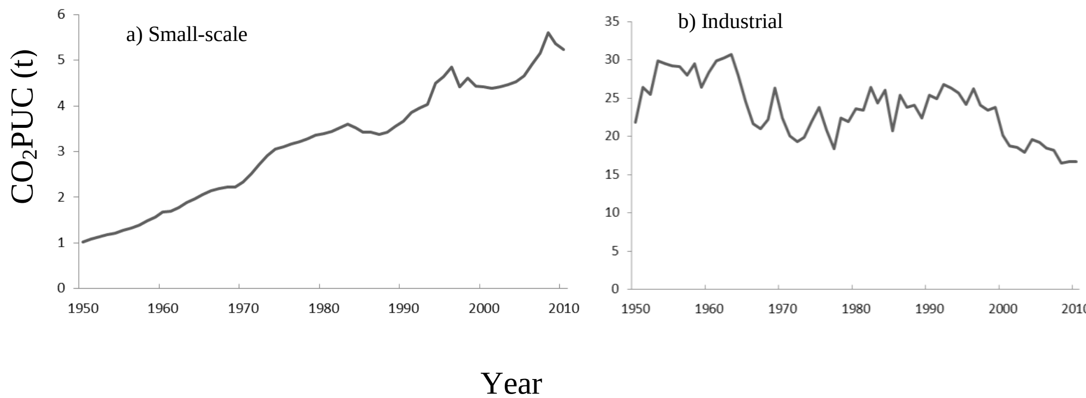

Figure 7 Annual carbon dioxide emissions per tonne of catch (tonnes) for both the small-scale and industrial sector from 1950 - 2010

3.3.4 Global CO_2 emissions

The global CO_2 emissions, regardless of which catch statistics used, was estimated to be over 1 billion tonnes in 2010. When the *Sea Around Us* reconstructed catch was used to estimate emissions, the number was 1.78 billion tonnes of CO_2 (+/- 30% for data uncertainty) emitted from global fisheries in 2010. In comparison when the FAO reported catch was used in the calculation, CO_2 emissions from fishing were estimated to be 1.18 billion tonnes (+/- 30% for data uncertainty) in 2010.

3.4 Discussion

Currently GHG emissions are tracking the high-risk scenarios predicted by the IPCC; in 2013 approximately 36 billion tonnes of CO_2 was released into the atmosphere at a rate of 2.3% more than the previous year suggesting that CO_2 per capita is increasing (IPCC 2013). If it is our goal to reduce the effects of increasing GHG on the global climate, then it is absolutely imperative that the international community come together to reduce CO_2 emissions. The ability to do so depend on a great many factors such as willingness, co-operation, economic viability, etc., but one of the most fundamental factors is the ability to identify key contributors to emissions and where reduction plans will have the greatest impact. Tracking CO_2 source emissions is difficult because there are few scenarios where CO_2 emissions can be measured directly and it is usually necessary to use CO_2 equivalent data to estimate GHG emissions. Therefore CO_2 emission estimates are only as good as the accuracy of the chosen indicator variable.

Both chapters 1 and 2 have discussed the relationship between resource use and environmental impact. A strong argument has been made for the idea that how much of a resource is consumed is not indicative of the environmental impact as a result of resource extraction. However, consumption-based indicators are still being used to estimate environmental impacts, and GHG emissions are no exception. It was the purpose of this study to develop a methodology that could be used to estimate the CO_2 emissions from fishing for the years 1950 - 2010 on a global scale using fishing effort to determine the extent of which fishing contributes to annual GHG emissions, a notable knowledge gap in fisheries science. There are three main conclusions that can be drawn from the preliminary results based on the 9 EEZs included in this chapter:

1. CO_2 emissions has both increased over the study time period and are much higher than previously published estimates;
2. Tonnes of CO_2 emitted per tonne of catch (CO_2PUC) has increased throughout the study time period and are also much higher than previously published estimates;

3. The CO_2PUC of the industrial sector has changed less than the CO_2PUC of the small-sector and may be declining. CO_2PUC of small-scale fisheries continues to increase, but is substantially lower than that of industrial fishing.

Previous estimates of CO_2 emissions from fishing indicated that the industry contributes to approximately 1% of global emissions (Tyedmers *et al.* 2005; Endresen *et al.* 2007). In comparison the results of the preliminary analysis suggest that this value is closer to 4 % (2.8 – 5.2% using 30% data certainty), between 1.18 – 1.78 billion tonnes of CO_2 in 2010. The difference between the estimate from this study and that of previous studies is substantial, and is likely attributed to the methods used. The study done by Tyedmers *et al.* (2005) modelled fisheries fuel consumption by summing the products of catches by species and a species-specific fuel estimate. This approach makes two major assumptions: that different species requires different amounts of fuel, and that the reported catch is representative of resource exploitation. The former assumption is likely to be true: some species are more energy intensive to catch; however, this is probably better represented by gear and how the species-specific fuel estimate was calculated was not clear. Gears are used to target specific species, and there is agreement that different gear types are more fuel intensive than others, for example, trawling (Wiviott and Mathews ; Gulbrandsen 2012). Without knowing what data went into the calculation, it is not possible to comment on how this factor may contribute to the differences between the estimate put forth by Tyedmers *et al.* (2005) and the one calculated in this analysis.

The second assumption, that marine catches are representative of resource extraction, is unlikely to be true and instead, reported catches are only a subset of what is actually caught (Pauly 1998, 2007). It would thus be expected that the CO_2 estimated using the FAO catch data be similar to that of Tyedmers *et al.* (2005) but it is not (1% vs. 3.2%). The difference between the two estimates might be a result of the factors that were originally used to model fuel consumption. Fuel consumption modelling by Tyedmers *et al.* (2005) was dependent on catch statistics that are established to be too low; studies that have attempted to reconstruct and report 'true' marine fisheries catches often find that actual catches are 2 -3 times what is reported in national and international statistics (for example: Zeller *et al.* 2007a; Abudaya *et al.* 2013; Tesfamichael and Pauly 2013; Pauly *et al.* 2014; Ulman *et al.* 2014; Zeller *et al.* 2014). The CO_2 emissions from fishing as estimated by Tyedmers *et al.* (2005) is thus likely to be underestimated. In contrast, the methods proposed in this thesis predicted fuel consumption independent of catch and therefore less likely to be influenced by its poor reporting status.

The CO_2 emissions estimated in this preliminary analysis is closer to that of the international shipping fleet estimate of 3% (Dalsoren *et al.* 2009). The study conducted by Dalsoren *et al.* (2009) used similar methods as proposed in this thesis to estimate CO_2 emissions in that they used vessel effort (kW) to estimate fuel consumption taken directly from Lloyd`s Fleet Registry Database; in cases where vessel effort was missing they too, used length to estimate vessel kW. Dalsoren *et al.* (2009) include $CO_2$ emissions from fishing in their estimate as some vessels in Lloyd`s Fleet Registry Database are labelled as fishing vessels. Lloyd`s Fleet Registry Database only includes vessels over 100 GT, and documents about 23,000 fishing vessels (as cited in Dalsoren *et al.* 2009). Fishing vessels greater than 100 GT comprise 1% of the world's fishing fleet (FAO 2005) and therefore the contribution of fisheries to CO_2 emissions as estimated by Dalsoren *et al.* (2009) is probably a gross underestimate.

Similar to the results of the total CO_2 emission, CO_2PUC both increased over the study time period and was much higher than previously published estimates; Tyedmers *et al.* (2005) reported that on average 1.7 tonnes of CO_2 were emitted for every tonne of catch in 2004. In comparison the CO_2PUC was 9.5 tonnes in 2010 for all sectors combined, 5.5 times more than that estimated by Tyedmers *et al.* (2005). It should also be noted that the 1.7 tonnes put forth by Tyedmers *et al.* (2005) is for industrial scale fishers; the true comparison is thus between 1.7 tonnes and 16 tonnes, the value that was estimated for industrial fisheries alone. The discrepancy between the two estimates can thus only partly be explained by coverage. The lack of independence between catch and CO_2 (recall: catch statistics were used to estimate species-specific fuel consumption) means that the estimate put forth by Tyedmers *et al.* (2005) is an average of the species-specific fuel consumption, and not a ratio of CO_2 to catch like the one presented here. Because CO_2 and catch were not independent of one another it was not possible for Tyedmers *et al.* (2005) to do so thereby limiting the result to an average, one that is probably substantially less than reality.

The majority of EEZs included in this analysis were on an increasing CO_2PUC trend, and as with fishing effort, the industrial sector is responsible for the majority of CO_2 emissions. When CO_2PUC for the sectors are examined separately, the result is that the industrial fishing sector is declining but the small-scale sector is actually increasing. Although the small-scale sector contributes proportionally less to annual CO_2 emissions, the results suggest that it may be contributing to overall increasing GHG emission trends. The growing contributions of the small-scale sector are probably a result of a lack of management; small-scale fisheries are often ignored, their impacts considered negligible and their role in communities downplayed. In contrast, many EEZs are implementing management protocols for fishing vessels aimed at reducing fuel consumption and increasing efficiency. The results of the preliminary analysis suggest that these efforts are having an impact; the CO_2PUC of the industrial sector is on a declining trend since the 1990s and is lower today than it was in 1950, but this trend may not continue once more countries are included in the database. Overall, the industrial sector still emits far more per tonne of catch than the artisanal sector and efforts to reduce industrial fishing would have a greater impact on annual CO_2 emissions.

In comparison to other protein choices fish is often cited as being more sustainable and having a smaller carbon footprint (Ziegler and Hansson 2003; Tyedmers *et al.* 2005; Vazquez-Rowe *et al.* 2014). The results of the preliminary analysis conducted for this thesis suggest that this may not be the case; regardless of which CO_2PUC value used (9.5 tonnes for all EEZs and sectors included in the analysis, or 18 tonnes for industrial fishing only) the tonnes of CO_2 emitted per tonne of product landed is still less than that of beef, but is actually higher than chicken. The FAO reported that for 1 tonne of beef, 22 tonnes of CO_2 are emitted (Gerber *et al.* 2013). A meta-analysis found that LCA studies had results ranging from 9 – 129 tonnes of CO_2 per tonne of beef (Nijdam *et al.* 2012). The same analysis found that chicken ranges from 2 – 6 tonnes of CO_2 emitted per tonne of poultry produced (Nijdam *et al.* 2012). Another meta-analysis conducted by de Vries and de Boer (2010) yielded the same result. In comparison, fish ranges from 4 – 18 tonnes depending on which sector it comes from. The review conducted by Nijdam *et al.* (2012) included fish protein in their analysis and found that it ranged from 1 – 18 tonnes of CO_2 per tonne of fish landed; the result of the preliminary analysis falls in the middle of this range perhaps suggesting that the methodology used may accurately estimate CO_2 emissions and that whether or not the fish is caught by industrial or small-scale fisheries makes a substantial difference.

It should be noted that the CO_2 emissions estimated here only represent those produced by the main engine. Often fishing vessels, especially those in the industrial sector, use auxiliary engines to power gear, refrigeration and cabin amenities (lights, ventilation, etc.). Auxiliary engines are thought to be roughly equal to 10% of main engine fuel consumption, but this varies depending on vessel type (Endresen and Sorgard 2003; Endresen *et al.* 2007). Refrigeration is a large component of auxiliary engine effort, and there are more and more studies investigating how refrigeration itself can become more efficient (for example: Johnston *et al.* 1994; Fernandez-Seara *et al.* 1998; Wang and Wang 2005). It is possible that the results may change for the industrial sector if auxiliary engines were included, but the small-scale sector would be unchanged. It is likely then that this methodology underestimates fuel consumption for the industrial sector.

It is widely accepted that the gear used plays a crucial role in fuel consumption for fishing vessels (for example: Sund 2009; Notti *et al.* 2011; Gulbrandsen 2012). A gear factor was not proposed in this methodology for two reasons; the first is because there is only data available on a few select gear types, mainly industrial trawling and longlining. Without more detailed information on more varied gear types it seemed that the confidence of the estimate would be affected. Secondly, gear in much of the database is assigned based on proportional breakdowns and limited data. Without better gear data, it seemed prudent to not include a factor that may substantially alter the results. It is likely that certain gear types, in particular trawling, have a higher impact on annual GHG emissions than more passive gear types like traps or gear designed for targeting schools of low-volume fish (such as purse seining for anchovies). Should the database be used to evaluate the effects of gear on fuel consumption, it would be important to note that the impact of gear on fuel consumption is not included in the current calculations.

The obvious limitation to the current results is the sample size. Only 9 countries are considered and although they do represent a wide range of fisheries, the results may change once future EEZs are added. Despite the small sample size, the results have demonstrated the potential and usefulness of the methodology proposed to estimate GHG emissions of the world`s fishing fleets. Once the database is completed, the trends that were illuminated in the preliminary analysis should be re-examined.

Chapter 4: Conclusion

The primary focus of this thesis was to develop a methodology that would guide future research on global fishing effort and GHG emissions; previous research has been limited by the paucity of lack of analyses and misconceptions in regards to the potential for fisheries to contribute to global climate change. Although tentative hypotheses were proposed, and results were discussed, the primary objective was to develop a methodology that could be applied to all fishing countries in the near future as part of *Sea Around Us*'s work. The following discussion is based on the findings of the preliminary analysis which suggests that overall, when fishing effort is adequately accounted for, the sustainability of fisheries operating as they are currently is questionable.

4.1 Fishing effort, sustainability and food security

The importance of marine resources to food security cannot be overstated, particularly in coastal developing countries (Kent 1997; Bell *et al.* 2009; Zeller *et al.* 2014). According to the FAO, the global average per capita fish consumption continues to rise, and current levels are twice what they were in the 1960s (FAO 2012). Although the global average is relatively low, approximately 18 kg per capita, consumption patterns are not distributed evenly, with the inhabitants of some countries, e.g., in the Pacific, consuming more than 100 kg per capita (FAO 2012). Furthermore it has been shown that countries with higher per capita incomes are less likely to be affected by decreases in fish availability because they have alternative protein options, and more money to purchase fish at higher prices. However, it is mostly low-income countries that depend the most on marine resources and are the most affected by decreasing fish stocks (Allison *et al.* 2009). Whether or not fisheries are exploited sustainably is a global issue, but amongst the many initiatives and management plans, there is little discussion as to what constitutes a sustainable fishery.

The idea of sustainability and its application to policy writing has increased substantially in the last two decades (Beckerman 1994). Simply stated whether or not something is 'sustainable' can be defined as "able to be maintained at a certain rate or level" (www.merriam-webster.com; Accessed October 8th 2014). In principle, when applied to environmental science and resource use, the concept is beautiful; it suggests that natural resources should be managed so that not only can it be used in the years to come by future generations, but that it does not damage an ecosystem and that the ecosystem state remains as close to pristine as possible. Sustainable development implies a moral obligation to resource use, but in practicality may not accurately define what society's objectives are and can therefore limit its applications in policy making (Beckerman 1994; Parris and Kates 2003). Plans and policies written with sustainable development in mind are consequently ambiguous and have difficulty reconciling both the conservation and development of a resource. Fisheries management is no exception.

The definition of sustainability is often assumed to be static, but what it means to be sustainable has changed dramatically in the last century; this is especially true in fisheries management. Once the false notion that fisheries was an "inexhaustible resource" was overcome, much fisheries management was on the concept of Maximum Sustainable Yield (MSY) whereby the population was maintained at its maximum growth rate and only individuals that would be added to the population each year would be taken (Clark 1973). MSY is rooted in single-species stock assessments and its ability to manage fisheries resources greatly depends on accurate population census and structure data. Here, fishing was considered sustainable if the target population numbers remained stable after an initial depletion. It did not however include any ideas regarding the economic sustainability of fisheries, considered to be governed by an "invisible hand"(Smith 1776) which would make it superfluous to consider the socioeconomic aspect of sustainability. As this proved incorrect as well, the concept of Maximum Economic Yield (MEY) was then introduced to fisheries management (Dichmont *et al.* 2010).

In the case of MSY, the focus is on managing the resource whereas MEY focuses on managing the resource users where revenue margins are peaked, instead of catch (Dichmont *et al.* 2010). MEY was identified as a "win-win" for fishers and conservationists: reduced exploitation would result in higher profits and sustainable stocks (Dichmont *et al.* 2010). The primary problem with MEY, as in all economic theory, is the assumption of rational actors; it assumes that fishers and policy makers will act rationally. And although it is able to be applied to multi-species fisheries, it does not consider the effects fishing may have on other ecosystem components that likely contribute to the sustainability of the fishery as well. Furthermore the applicability of MEY as a management tool is generally undermined by government subsidies that enable fishing to continue once it is no longer economically sustainable (Sumaila *et al.* 2008).

More recently, with the growing appreciation for the overexploited state of many of the world's fisheries, there has been a shift from target species and economic sustainability to ecosystem sustainability. Ecosystem sustainability is intended to be more holistic and includes not only the preservation of ecosystem processes but all the goods and services they provide (Murawski 2000). The problem, however, is that neither MSY or MEY necessarily capture ecosystems processes and it is not really known how much impact any one ecosystem can withstand. The problem is further complicated by what has already been exploited – it is impossible to know whether or not the ecosystem in question has already changed. Recent studies suggest that there are few, if any, ecosystems that remain unaffected by exploitation or development (for example: Nilsson and Berggren 2000; Halpern *et al.* 2008; Knowlton and

Jackson 2008). It must then be determined if sustainability includes recovery of ecosystems or if it aims to conserve what is currently still present.

Whether or not current fishing practices are sustainable is an important question and one that is not easily answered. Understanding fishing effort patterns over time can help answer this question. Fishing effort has mostly stabilized, but CPUE continues to decline for most of the EEZs included here suggesting that fishing effort is being maintained at a higher level than what is sustainable in the long term. Ecosystem sustainability requires an understanding of changes outside of target species and is beyond the scope of this thesis. There have been studies that have focused on changes in catch type that have found that over time, as larger species become more rare, fishers begin to target smaller species causing trophic cascades to occur (Pauly *et al.* 1998; Daskalov 2002; Morato *et al.* 2006). There is also evidence for ecosystem state changes, or 'regime shifts', in many of the world's oceans (Hughes *et al.* 2005). It is difficult to quantify what role fishing has in regime shifts, but its contribution is probably best described as reducing resilience: fishing removes species that perform certain ecosystem functions that may not be performed by any (or many) other species. Reduced resilience increases the vulnerability of an ecosystem to other impacts such as disease, and climate change (Hughes *et al.* 2005).

The results presented here, pertaining to fishing effort, do suggest that the type of fishing that occurs may be relevant to at least catch rate sustainability, if not ecosystem sustainability. Small-scale fishing composed approximately 30% of total fishing effort, but included over 90% of boats in the database. Also, the EEZ with only a small-scale fishing sector (Bahamas) was also the only EEZ with an increasing CPUE. However, the fleet composition – small-scale versus industrial – was not that different between 1950 and 2010: in 1950 small-scale fishing constitutes approximately 95% of the vessels in the database versus 93% in 2010. It is difficult to know whether or not this slight change in global fleet composition could in fact drive declining CPUE, especially given that fishing effort in 2010 for all sectors is 20 times what it was in 1950. Furthermore CPUE trends were declining since the beginning of the time period, suggesting that overfishing was already occurring in the majority of these EEZs. The data may not paint a clear picture whether or not a particular sector is responsible for overfishing, but what it does say is that reducing the number of large, industrial vessels will result in a substantially bigger reduction in fishing effort per boat than small-scale vessels.

Reducing the number of large vessels not only reduces overall fishing effort, but also limits the spatial capacity of fishers. Small-scale vessels typically fish in-shore areas and coastal zones within 20 nautical miles from shore (Schorr 2005) whereas industrial vessels are capable of off-shore fishing, and fishing on the high seas. The spatial limitation of smaller vessels combined with reduced capacity for refrigeration technology provides a refuge for fish. Although the impact of small-scale fisheries may still be important, it is more contained than that of industrial fisheries. Additionally the number of fishers impacted by fishing effort reductions in the industrial sector is much less than would be impacted by a reduction in small-scale fishers, helping to secure the economic sustainability of the industry (Jacquet and Pauly 2008).

As the world's population continues to grow, so does the demand for marine resources. Capture fisheries are increasingly more unable to keep up and aquaculture is the fastest growing food-producing sector (FAO 2014). The potential for aquaculture to help meet growing food demands is promising, but it cannot replace the value of capture fisheries. Aquaculture, like farming on land, also impacts the ecosystem and is actually second-best option in terms of ecosystem sustainability. It is the solution to declining CPUE in marine capture fisheries, and continual increases in demand for marine resources. Aquaculture has become a desirable option because fishing is becoming less desirable.

Managers and policy makers need to be clear on what constitutes sustainable fishery; importantly an understanding of whether or not fisheries that are managed sustainably meet the growing nutritional and economic needs of the population is required, or has the ability of marine populations to recover been damaged. The reduction of fishing effort, especially that of industrial sector, may lead to a more sustainable future for fisheries, depending on how 'sustainable' is defined. Furthermore the contribution that managing fishing effort can make to sustainable fisheries will be better understood once fishing effort time series have been estimated for all relevant EEZs.

4.2 Eco-labelling, sustainability and the carbon footprint of fisheries

Public interest in the sustainability of fisheries has never been greater and, over the last decade, a huge number of eco-labelling certification options have emerged aimed to guide consumers to the relative sustainability of a product (Jacquet 2011). Eco-labelling is a form of social marketing whereby the responsibility of sustainability lies on the consumer; it is based on the principle that consumers will choose to purchase products that are ethically and sustainably sourced over those products that are not (Jacquet and Pauly 2007; Potts and Haward 2007). Therefore changes in consumer behaviour aid fisheries management by rewarding those fisheries with sustainable practices with higher profit margins. Certification schemes originally were government initiatives, the first of which occurred in Germany in 1977, followed by Canada 11 years later (Gulbrandsen 2010). These first initiatives were multi-product programs with little fishery emphasis – the first, and perhaps most famous, fishery directed eco-label was "Dolphin-safe" tuna (Jacquet and Pauly 2007). The most established and widely used fishery eco-label to date is the Marine Stewardship Council (MSC), a multi-product scheme that attempts to provide ratings based on the entire production chain from ocean to consumer (Thrane *et al.* 2009).

The efficacy of certification schemes as a useful management tool is contentious, and some research indicates none, or very little change, in consumer behaviour (e.g.Gulbrandsen 2009). The essence of the discussion, however, is not whether or not certification schemes alter human behaviour but rather if they adequately capture sustainability. As discussed in the previous section, sustainability not accompanied by a definition is rather ambiguous and depends on the management goals. Eco-labels range in their coverage from addressing one single environmental issue, such as by-catch in the tuna industry by the Dolphin-Safe certification, to a multi-environmental impact certification such as that claimed by the MSC. Primarily, current fishery certification schemes focus on the "fishing stage" and for good reason: LCA studies do agree that the fishing stage has the greatest environmental impact (Thrane *et al.* 2009). Currently no international, multi-impact certification scheme includes fishing's GHG emissions as a part of the certification criteria, despite the role of carbon dioxide in global warming and that the majority of CO_2 emissions in the industry occur during fishing.

The carbon footprint of products tends to be excluded from eco-labelling criteria, and is instead, the focus of carbon-specific certification initiatives such as Carbon Trust (www.carbontrust.com) and CarbonFree Certified (www.carbonfund.org). Certification schemes currently, by nature, are voluntary and fisheries are conspicuously absent from carbon focused eco-labels. There are some seafood wholesale distributors that are aiming to reduce the carbon footprint of their products, for example, ProFish (www.profish.com). However, these businesses focus mainly on the carbon footprint of fish once they are landed, and do not include, to quote Thrane *et al.* (2009), the "fishing stage". This may be a reflection of the limited data readily available in regards to the carbon footprint of fisheries and may be changing; ProFish says that *"On the water, there are several initiatives to reduce seafood's carbon footprint"*, but no quantitative information regarding what constitutes a carbon-friendly fishery was provided.

In Europe, where the majority of fisheries LCA studies have been conducted, there has been some attempt to include energy reductions – and therefore reduced carbon emissions – into eco-certification by the Swedish KRAV eco-label and the Danish NGO, the Danish Society for Living Seas (DSLS). The former has only certified 4 fisheries and the latter is no longer in use (Thrane *et al.* 2009). Energy reduction initiatives in fisheries are driven by rising fuel costs to fishers as opposed to their contribution to climate change through CO_2 emissions (Schau *et al.* 2009b; Notti *et al.* 2011). However the preliminary results presented in this thesis found that not only were the CO_2 emissions from fishing far from negligible – around 4% of total global emissions – but that despite increases in fuel efficiency, the amount of CO_2 emissions per tonne of catch is increasing. With such a high carbon footprint, it is a wonder that any multi-attribute eco-label could not include fishing's GHG emissions during the fishing stage as part of their sustainability criteria.

The results of the thesis suggest that most industrial fisheries would not likely get certified under sustainable carbon criteria, again depending on what was deemed sustainable. Industrial fishing was found to emit approximately 18 tonnes of CO_2 per tonne of catch in 2010, compared to about 5 tonnes by the small-scale sector. This figure places fish procured through the industrial sector with a similar carbon footprint to beef (Nijdam *et al.* 2012). Obviously there is wide variation between gear types, probably not dissimilar than the differences between cattle raising strategies, but the point is that industrial fisheries on average have a very high carbon footprint. This greatly affects the magnitude of environmental impact fishing- that is not captured by eco-labelling initiatives – and questions the sustainability of the resource in a global warming world.

The role that CO_2 emissions play in the sustainability of fisheries is critical. It may not be as obvious as direct environmental impacts observed by destructive fishing techniques or trophic cascades, but it directly contributes to an arguably bigger problem: global warming. If fisheries are to be considered sustainable, then adequately considering their contribution to climate change is a crucial component. Most sustainable development policies include points on CO_2 emissions, primarily on reducing them, and it is time that fisheries be part of the international discussion.

4.3 Closing comments and future research

The primary focus of this thesis was to develop a methodology that would adequately quantify and consider the "Effort Factor" in fisheries, and then use this data to estimate CO_2 emissions from fishing. It was important to do so independent of catch and consumption-based statistics. The presented results are limited by the sample size; however the trends, if they persist once the global analysis is complete, will be able to give managers a more comprehensive idea of the environmental costs of fishing and fill a considerable knowledge gap in fisheries science.

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Appendices

Appendix A - Days fished

Information on mean number of days fished per year by vessels of different gear classes was first compiled by Gelchu and Pauly (2007) from FAO reports on economic performance of fishing fleets from selected EEZs of each region in the years 1995 to 2000 (Le Ry *et al.* 1998; Tietze *et al.* 2001). Mean number of days fished information from these countries were assumed to represent average fishing activities in the region and were therefore used to assign mean days fished per year for corresponding gear class categories for EEZs within a given region. The majority of data used by Gelchu and Pauly (2007) were based on European and North American countries and may not be accurate for other continents. A continent factor was subsequently developed by Anticamara *et al.* (2011) to account for variation between continents, where North America and Europe equal 1.

Table 7 Mean days fished and the continent factor by region and gear type used in the *Sea Around Us* global fishing effort database

Region	Gear type	Mean days fished	Continent factor
Africa	Pelagic Trawler	270	0.9
	Liner	180	0.9
	Seiner	148	0.9
	Bottom trawler	180	0.9
	Gillnet	150	0.9
	Driftnet	150	0.9
	Trapper	150	0.9
	Dredger	180	0.9
Asia- Pacific	Driftnet	207	1.1
	Gilnet	207	1.1
	Dredger	180	1.1
	Bottom trawler	233	1.1
	Trap	180	1.1
	Pelagic trawler	196	1.1
	Liner	213	1.1
	Seiner	181	1.1
Europe and North America	Bottom trawler	231	1
	Pelagic trawler	294	1
	Driftnet	161	1
	Seiner	181	1

Region	Gear type	Mean days fished	Continent factor
	Dredger	200	1
Europe and North America cont'd	Gillnet	150	1
	Liner	185	1
	Trap	120	1
South America and Caribbean	Driftnet	163	1
	Dredger	213	1
	Gillnet	163	1
	Pelagic trawler	209	1
	Liner	163	1
	Bottom trawler	213	1
	Trap	111	1
	Seiner	209	1

Appendix B - Fuel coefficient

The fuel coefficient from 1950 – 2010 was estimated to account for changes in the fuel efficiency over the study time period. A specific fuel rate (SFR) range in g/kWh was supplied from Yanmar, an international builder and supplier of marine engines since 1912, for the years 1960, 1974, 1980, 1985, 1989, 1996 and 2001 (Table 1; Fig 8). Given that for each year, the SFR had two values, a lower and an upper range value, an average of these two values was used. For years outside of the supplied data range (1950 – 1959 and 2002 – 2010), the SFR was assumed to the same as the closest raw data point. The remaining missing years were linearly interpolated.

The data supplied by Yanmar was in g/kWh, a measure of mass, and in order to estimate CO_2 emissions the SFR time series needed to be in l/kWh, a measure of volume of fuel consumed. To convert mass to volume, the mass was divided by the density:

$$V_{l/kWh} = M_{g/kWh} \big/ D_{g/l}$$

Where V is volume, M is mass and D is density. The density of the fuel used was supplied by Yanmar and was 885 g/l. The year 2000 was set as the benchmark, i.e., the FC for the year 2000 = 1. The FC was then calculated by dividing the SFR for a given year by the SFR for the year 2000 to yield the change in SFR over time (Table).

Table 8 Fuel coefficient from 1950 - 2010 that was calculated using Yanmar engine SFR data

Year	g/kWh	L/kWh	Fuel coefficient	Data description
1950	224.79	0.25	1.22	Carried backwards
1951	224.79	0.25	1.22	Carried backwards
1952	224.79	0.25	1.22	Carried backwards
1953	224.79	0.25	1.22	Carried backwards
1954	224.79	0.25	1.22	Carried backwards
1955	224.79	0.25	1.22	Carried backwards
1956	224.79	0.25	1.22	Carried backwards
1957	224.79	0.25	1.22	Carried backwards
1958	224.79	0.25	1.22	Carried backwards

Year	g/kWh	L/kWh	Fuel coefficient	Data description
1959	224.79	0.25	1.22	Carried backwards
1960	224.79	0.25	1.22	Data point
1961	224.19	0.25	1.22	Linear interpolation
1962	223.60	0.25	1.21	Linear interpolation
1963	223.00	0.25	1.21	Linear interpolation
1964	222.41	0.25	1.21	Linear interpolation
1965	221.82	0.25	1.20	Linear interpolation
1966	221.22	0.25	1.20	Linear interpolation
1967	220.63	0.25	1.20	Linear interpolation
1968	220.03	0.25	1.19	Linear interpolation
1969	219.44	0.25	1.19	Linear interpolation
1970	218.84	0.25	1.19	Linear interpolation
1971	218.25	0.25	1.18	Linear interpolation
1972	217.66	0.25	1.18	Linear interpolation
1973	217.06	0.25	1.18	Linear interpolation
1974	216.47	0.24	1.17	Data point
1975	212.99	0.24	1.16	Linear interpolation
1976	209.52	0.24	1.14	Linear interpolation
1977	206.05	0.23	1.12	Linear interpolation
1978	202.58	0.23	1.10	Linear interpolation
1979	199.11	0.22	1.08	Linear interpolation
1980	195.64	0.22	1.06	Data point
1981	194.74	0.22	1.06	Linear interpolation
1982	193.83	0.22	1.05	Linear interpolation
1983	192.93	0.22	1.05	Linear interpolation
1984	192.03	0.22	1.04	Linear interpolation
1985	191.58	0.22	1.04	Data point
1986	190.57	0.22	1.03	Linear interpolation
1987	189.55	0.21	1.03	Linear interpolation
1988	188.54	0.21	1.02	Linear interpolation
1989	187.53	0.21	1.02	Data point
1990	187.57	0.21	1.02	Linear interpolation
1991	187.60	0.21	1.02	Linear interpolation
1992	187.64	0.21	1.02	Linear interpolation
1993	187.67	0.21	1.02	Linear interpolation
1994	187.71	0.21	1.02	Linear interpolation
1995	187.74	0.21	1.02	Linear interpolation
1996	187.78	0.21	1.02	Data point
1997	186.89	0.21	1.01	Linear interpolation
1998	186.01	0.21	1.01	Linear interpolation

Year	g/kWh	L/kWh	Fuel coefficient	Data description
1999	185.12	0.21	1.00	Linear interpolation
2000	184.23	0.21	1.00	Linear interpolation
2001	183.35	0.21	1.00	Linear interpolation
2002	182.46	0.21	0.99	Data point
2003	182.46	0.21	0.99	Carried forwards
2004	182.46	0.21	0.99	Carried forwards
2005	182.46	0.21	0.99	Carried forwards
2006	182.46	0.21	0.99	Carried forwards
2007	182.46	0.21	0.99	Carried forwards
2008	182.46	0.21	0.99	Carried forwards
2009	182.46	0.21	0.99	Carried forwards
2010	182.46	0.21	0.99	Carried forwards

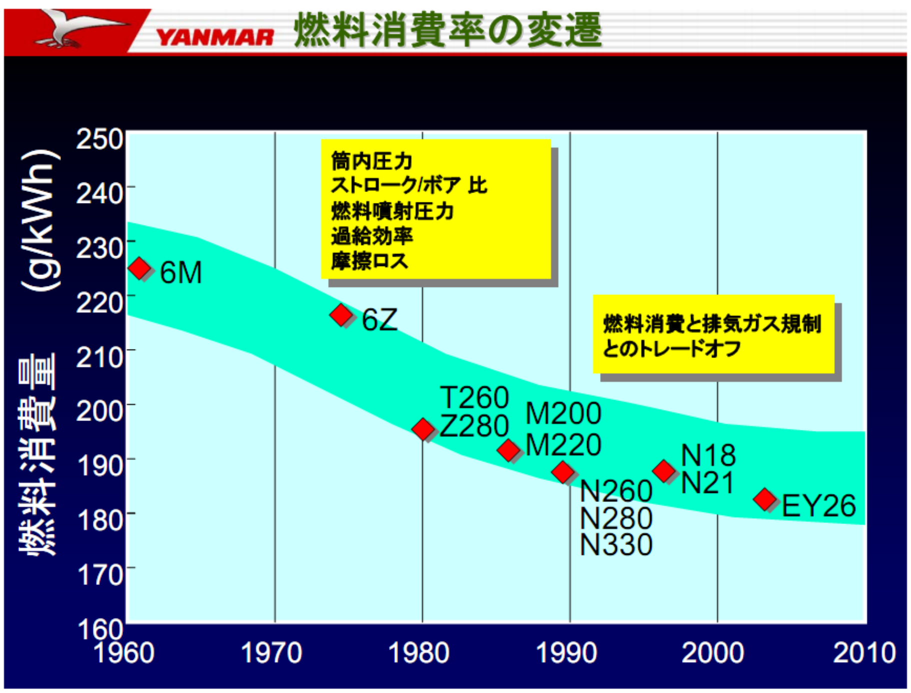

Figure 8 Fuel coefficient of diesel engines (g/kWh). This was the original data supplied by Yanmar engines. The numbers beside the diamonds refer to specific engine types. The text refers to changes in technology.