COOPERATION AND POLICY INTERVENTIONS:
FIELD EXPERIMENTS
WITH SMALL-SCALE FISHING COMMUNITIES IN TURKEY

PINAR ERTÖR AKYAZI

BOĞAZİÇİ UNIVERSITY

2016

COOPERATION AND POLICY INTERVENTIONS:

FIELD EXPERIMENTS

WITH SMALL-SCALE FISHING COMMUNITIES IN TURKEY

Thesis submitted to the

Institute for Graduate Studies in Social Sciences

in partial fulfillment of the requirements for the degree of

Doctor of Philosophy

in

Economics

by

Pınar Ertör Akyazı

Boğaziçi University

2016

DECLARATION OF ORIGINALITY

I, Pınar Ertör Akyazı, certify that

- I am the sole author of this thesis and that I have fully acknowledged and documented in my thesis all sources of ideas and words, including digital resources, which have been produced or published by another person or institution;
- this thesis contains no material that has been submitted or accepted for a degree or diploma in any other educational institution;
- this is a true copy of the thesis approved by my advisor and thesis committee at Boğaziçi University, including final revisions required by them.

Signature..

Date ....22.08.2016..

ABSTRACT

Cooperation and Policy Interventions:

Field Experiments With Small-Scale Fishing Communities in Turkey

The aim of this dissertation is to contribute to the vast literature on environmental policy instruments and resource use dilemmas via an experimental approach. Based on framed field experiments conducted with members of small-scale fishing communities in Turkey, the study focused on the effects of different policy interventions on cooperation, and demonstrated that analyzing the behavioral impact of environmental policy instruments is crucial to ensure effectiveness. The framed field experiments were used to evaluate the effects of an external fine, a market-like incentive, and a communication condition in a resource use dilemma. The experiments involved twelve sessions for each treatment (baseline, fine, market and communication treatments), with five participants per session. Both communication and a symbolic (low) fine were found to be effective in reducing extraction. The findings further imply that there is a framing effect between the fine and market conditions. When the economic incentive was framed as a fine as opposed to a market-like mechanism, it reduced extraction more. This implies that institutional context matters to individual behavior, and that the same level of economic incentive may perform differently if the institutional context is changed. This finding also supports the hypothesis that market reasoning may lead to unintended outcomes in resource use dilemmas. If this is not taken into account, environmental policies may backfire.

ÖZET

İşbirliği ve Politika Müdahaleleri:

Türkiye’deki Küçük Ölçekli Balıkçı Topluluklarıyla Saha Deneyleri

Çevre politikası araçları ve kaynak kullanımına ilişkin akademik literatüre katkıda bulunmayı hedefleyen bu tezde deneysel yöntemler izlenmektedir. Bu amaçla, Türkiye’deki küçük ölçekli balıkçılıkla uğraşan toplulukların üyeleriyle çerçevelenmiş saha deneyleri yapılmış ve çevre politikası araçlarının etkin olabilmesi için bu araçların davranışsal etkilerinin de incelenmesi gerekliliği vurgulanmıştır. Çerçevelenmiş saha deneyleri, ceza, piyasa ve iletişim koşulları altında kaynak kullanımının nasıl değiştiğini incelemektedir. Deneylerde baz, ceza, piyasa ve iletişim koşulları için 12’şer oturum düzenlenmiş, her oturumda ise beşer katılımcı yer almıştır. Çalışmada iletişim ve ceza koşullarının kaynak kullanımını azaltmada etkili olduğu görülmüştür. Elde edilen sonuçlar, bir yandan da ceza ve piyasa koşulları arasında bir çerçeveleme farkı olduğunu göstermektedir. Kaynak kullanımı, bir ekonomik teşvik ceza olarak çerçevelendiğinde, aynı teşviğin piyasa diliyle çerçevelenmesine kıyasla daha düşük seviyede olmaktadır. Bu bulgu ise, kurumsal bağlamın bireylerin davranışı üzerindeki etkisini ve aynı büyüklükte bir ekonomik teşviğin farklı kurumsal bağlamlarda farklı performans gösterebileceğini ortaya çıkarmaktadır. Sonuçlar ayrıca piyasa temelli muhakemenin, kaynak kullanımında beklenmedik sonuçlara yol açabileceği hipotezini de destekler niteliktedir. Bunlar dikkate alınmadığında, çevre politikaları geri tepebilir.

CURRICULUM VITAE

NAME: Pınar Ertör Akyazı

DEGREES AWARDED

PhD in Economics, 2016, Boğaziçi University

MA in Economics, 2010, Boğaziçi University

BA in Management, 2008, Boğaziçi University

Pre-diploma (Vordiplom) in Management, 2005, University of Cologne

AREAS OF SPECIAL INTERESTS

Experimental and behavioral economics, environmental economics, ecological economics, political ecology, and fisheries management

PROFESSIONAL EXPERIENCE

Teaching and Research Assistant, Department of Economics, Boğaziçi University, 2008 - present

Research Assistant for the EU funded 7th Framework Project EJOLT (Environmental Justice Organizations, Liabilities and Trade), 2010 - 2015

Research Assistant for the EU funded 7th Framework Project ENTITLE (European Network of Political Ecology) and Marie-Curie Scholarship Program, 2012 – 2016

AWARDS AND HONORS

Highest Honors List, Boğaziçi University, 2016

Highest Honors List, Boğaziçi University, 2010

Highest Honors List, Boğaziçi University, 2008

Highest Honors List, University of Cologne, 2005

GRANTS AND SCHOLARSHIPS

B.U. Research Fund (BAP) Doctorate Project Grant (No: 9921), 2015 - 2016

TÜBİTAK International PhD Research Scholarship, 2014

TÜBİTAK PhD Scholarship, 2010 - present

TÜBİTAK MA Scholarship, 2008 - 2010

DAAD (The German Academic Exchange Service) Scholarship, 2003 - 2005

PUBLICATIONS

Journal Articles

Ertör-Akyazı, P., Adaman, F., Özkaynak, B., & Zenginobuz, Ü. (2012). Citizens' preferences on nuclear and renewable energy sources: Evidence from Turkey. *Energy Policy,* 47, 309-320.

MA Thesis

Ertör-Akyazı, P. (2010). The Analysis of Citizens' Preferences for Energy Investment Alternatives in Turkey: Nuclear versus Renewable Energy. Boğaziçi University, Department of Economics.

Book Chapters

Ertör-Akyazı, P. (2015). Çevre ve deneysel iktisat. In D. Dumludağ, Ö. Gökdemir, L. Neyse, & E. Ruben (Eds.), *İktisatta davranışsal yaklaşımlar* (pp. 339-382). İstanbul: İmge Kitabevi.

Conference Proceedings

Sprain, L., Asara, V., Ertör-Akyazı, P., & Tompsett, C. (2011). The 'Wickedness' of Participation in Climate Change Adaptation Governance. In S. Movik & A. Vatn (Eds.), *Thor Heyerdahl Summer School in Environmental Governance, Volume 1.* Aas: Norwegian University of Life Sciences.

ACKNOWLEDGMENTS

First and foremost, I would like to thank my supervisor Prof. Begüm Özkaynak for her encouragement and excellent guidance throughout this dissertation. She has become a dear friend to me during the last eight years in which we have worked together closely. I also would like to express my gratitude to my co-supervisor, Prof. Fikret Adaman, who has always supported my academic career with valuable insights and advice. I am deeply indebted to Prof. Samuel Bowles who has inspired and encouraged me at so many different points of this dissertation. I took his course on Post-Walrasian Economics when he was visiting our department, and this course opened a completely new academic horizon to me.

I would like to express my appreciation for the valuable ideas and suggestions of Prof. Juan Camillo Cardenas, Prof. Ali Çarkoğlu, Prof. Refik Erzan, Prof. Andreas Lange, Prof. John List, Prof. Hüseyin Özbilgin, Prof. Michael Price, Prof. Ali Kerem Saysel, Prof. Ünal Zenginobuz, Assoc. Prof. Devrim Dumludağ, Assoc. Prof. Ayça Ebru Giritligil, Assoc. Prof. Yiğit Gürdal, Assoc. Prof. Vahdet Ünal, Assist. Prof. Erol Akçay, Assist. Prof. Umut Kuzubaş, Assist. Prof. Ceren Soylu, Dr. Emrah Çoraman, Cem İskender Aydın, Çağlayan Işık, and the participants of the EAERE Field Experiments and Environment Summer School.

The terms that I spent at the University of Siena and University of Zurich Laboratory for Social and Neural Systems Research have contributed so much to my academic development; therefore, I cannot thank enough Prof. Samuel Bowles, Prof.

Ernst Fehr, Prof. Luigi Luini, Prof. Ugo Pagano, Assoc. Prof. Charles Efferson, and Dr. Sonja Vogt for these two very productive academic terms.

The fieldwork could not have been accomplished without the help and support of the head of the Istanbul Fisheries Cooperatives Association, Erdoğan Kartal, and his assistant, Fatma Doğru; I should likewise express my indebtedness to them both. I also would like to thank all colleagues, friends and especially BÜKOOP members who participated in the preliminary versions of the experiments, and of course, all the fishers who took part in the field experiments during August 2015 – June 2016. Cansu Çetin, Mustafa Kaba, Kayıhan Kesbiç, and Emir Kutlu have assisted, encouraged and supported the fieldwork with extreme care and dear friendship. They made the field study so much more fun and interesting. Special thanks go to them, and to Gülşah Seral and Kyle Heatherly, who have edited this dissertation with much care and patience.

I would like to acknowledge the Scientific and Technological Research Council of Turkey (TÜBİTAK) for granting me a graduate scholarship (2211) and international PhD research scholarship (2214) during this dissertation. I would also like to acknowledge the support of Boğaziçi University Research Fund Grant No: 9921 on which the fieldwork of this dissertation is based. I also appreciate the academic conference support funds of SBE and BÜVAK at Boğaziçi University.

I dedicate this thesis to my parents, Gönül and Salih, my sister, Irmak, my husband, İbrahim, my aunt, Mehpare, and to the rest of my beloved family, who have taught me to smile and to love, and always supported and encouraged me throughout my life.

TABLE OF CONTENTS

LIST OF TABLES

LIST OF FIGURES

ABBREVIATIONS

CAC	Command-and-control
CBA	Cost-benefit analysis
CPR	Common-pool resource
CV	Contingent valuation
EPA	Environmental Protection Agency
FAO	Food and Agriculture Organization
GFCM	General Fisheries Commission for the Mediterranean
ICCAT	International Commission for the Conservation of Atlantic Tuna
ITQ	Individual transferable quota
MPCR	Marginal per capita return
NOAA	National Oceanic and Atmospheric Administration
OECD	Organization for Economic Co-operation and Development
PES	Payments for ecosystem services
PG	Public good
PO	Pareto optimality
PPO	Potential Pareto optimality
UN	United Nations
UNCLOS	United Nations Convention on the Law of the Sea
UNEP	United Nations Environment Programme
WTA	Willingness to accept
WTP	Willingness to pay

INTRODUCTION

Environmental problems such as climate change, overexploitation of natural resources, biodiversity loss, and pollution are among the most pressing issues of the 21st century. The environmental crisis is affecting humankind in an unprecedented way. In parallel, an increasing number of citizens all over the world are concerned about the environment (Malmqvist & Whan, 2016). These facts are accompanied by an ongoing heated debate as how to address environmental problems.

Environmental policy instruments, such as direct regulation, environmental taxes, and markets, have been designed and implemented since the 1960s to deal with the anthropogenic ecological problems created by industrialization and the increasing world population (Barde, 1994, 1999). Public economics and environmental economics have arisen as the two main sub-disciplines of economics aimed at the study of environmental policy instruments. Assuming that individuals behave opportunistically, their approach has been rather technical and focused on a mechanistic evaluation of economic efficiency, environmental effectiveness, and the distribution of economic costs and benefits. Meanwhile, they have neglected issues such as social norms, the effects of institutions on human behavior, and the role of informal institutional arrangements among resource users (Ostrom, 2006; Spash, 2010; van den Bergh, Ferrer-i-carbonell, & Munda, 2000; Vatn, 2005).

However, public policies that assume universal self-payoff maximizing behavior and neglect the possibility of other-regarding behavior may have unintended consequences and can even backfire (Bowles & Polanía-Reyes, 2012; Bowles, 2008; Gneezy, Meier, & Rey-Biel, 2011). That is, if the analysis of policy instruments does not address preferences such as reciprocity, altruism, trust, and fairness, one cannot be certain about the environmental, social, and economic consequences of the interventions. This can be even more problematic in the environmental realm, in which ethical and normative aspects play a particularly important role (Vatn, 2009a). The institutional context may also affect the way that an individual reasons (Muradian & Cardenas, 2015; Rode, Gómez-Baggethun, & Krause, 2015; Sagoff, 2008; Söderbaum, 1990; Vatn, 2005, 2009b). Accordingly, a citizen perspective that entails moral values may be adopted in certain institutional contexts, whereas a self-payoff maximizing perspective may be more appropriate in other contexts. Institutions are not merely constraints on behavior, as new institutional economics claims, but can substantially alter the decisional context of individuals by signaling socially acceptable behavior in different settings.

Behavioral and experimental studies are important tools that can be used to test claims regarding the effect of institutional context on human behavior. Behavioral and experimental economics have created awareness of framing effects in public goods games, which imply that changes in the framing of the game affect behavior by changing the decisional context (Bowles & Polanía-Reyes, 2012; Bowles, 2008). This indicates that institutions matter in economic decisions. However, these findings are largely ignored in mainstream environmental policy analysis, both in academia and in real world policy-making.

Since the introduction of the first environmental policies in the 1960s and 1970s, the pendulum of environmental policy-making has swung between state regulation and environmental markets, although the latter has attracted much more attention in the last decade (OECD, 2006, 2008; UNEP, 2004). Today, the most visible signs of the expansion of markets towards the environmental sphere are in the areas of climate change, overfishing, and biodiversity conservation. The popularity of markets rests primarily on practical flexibility and the theoretical prediction of economic efficiency (Kolstad, 2009; Sandmo, 2000). Because ecological complexities such as irreversibility and the non-linear effects of biophysical dynamics have interacted with social processes, it has been difficult to assess the effectiveness of markets in terms of environmental outcomes. Nonetheless, the implementation of carbon markets and individual transferable quotas to deal with climate change and overfishing, respectively, have been unable so far to bring about the intended results (Pearce & Böhm, 2015; Smith et al., 2009).

One could argue that environmental markets have not been fully operationalized as a result of obstacles to implementation. Regardless, we still do not know much about the behavioral aspects of markets. How markets interact with other-regarding behavior, social norms, and ethical considerations is largely unknown. Indeed, the effect of market reasoning and language on economic behavior has not yet been fully investigated in experimental economics. The experimental study by Falk and Szech (2013), which demonstrated that moral behavior is significantly undermined upon the introduction of a market setting into the experiment, marks a turning point in this line of research. However, this evidence needs to be supported by further experimental studies that directly address this effect.

The penetration of market values into the environmental realm has also been criticized from different perspectives, such as institutional and ecological economics (O'Neill, 2001; Spash, 2010). Accordingly, markets may undermine moral values and crowd-out voluntary efforts to preserve the natural environment (Frey & Stutzer, 2008a;

Reeson & Tisdell, 2010; Rode et al., 2015). Yet, despite some anecdotal evidence, these theoretical claims have not been tested empirically in the context of environmental policies and resource use. Experiments present an invaluable opportunity to understand the effect of markets on human behavior and ecological consequences in a controlled environment.

Against this background, this study focuses on the intersection of three distinct research areas:

- Traditional environmental policy instruments, based on the public and environmental economics literature;
- Alternative perspectives on environmental policies involving pro-social behavior, institutions, and community-based resource management; and
- Behavioral and experimental economics within these two domains.

Based on a field study involving framed field experiments with members of small-scale fishing communities in Turkey, this study focuses on the effects of different policy interventions on cooperation in a resource use dilemma—taking a methodological individualistic standpoint. In the first stage of the game, participants play a public goods game representing an open-access regime of resource use. In the second stage, three different policy interventions are imposed. These include a symbolic (low) fine, a market-like intervention involving the same level of material disincentive as the fine, and face-to-face communication. More specifically, the experiments aim to answer the following research questions:

- To what extent do individuals choose an extraction level that is close to the socially efficient outcome in a resource dilemma in the absence of any policy intervention? (Baseline treatment)
- How does a symbolic (low) fine affect resource use? Does a weak material incentive framed in the form of a fine for over-extraction reduce or increase resource use? (Fine treatment)
- How does a market-like incentive affect resource use? Does this material incentive—framed as the fixed price of an extraction right—reduce or increase resource use? (Market treatment)
- If face-to-face communication is allowed within the group, but no incentive mechanism is applied, how is resource use affected? (Communication treatment)

The experiments involved twelve sessions for each treatment, and each session or group consisted of five individuals. Sixty individuals participated in each treatment, with 240 individuals in total taking part in the study. The sessions lasted for around one hour, including a short survey administered at the end of each session.

The main contribution of the experiments is the investigation of framing effects between a fine and a market-like incentive in a resource use dilemma. While the existence of framing effects is well-established for public goods games, they have not been investigated for resource use problems. Moreover, despite its relevance for determining the effectiveness of environmental policy instruments, the effect of market-like incentives on behavior has been under-investigated. The field study addresses these two aspects in a framed field experiment and combines them with a short follow-up survey on the socio-economic status and other characteristics of the participants.

Exposed to a similar resource use dilemma in real life and aware of the dynamics inherent to it, as prior fieldwork has revealed, the target group of small-scale fishers in Turkey suits these purposes well. Nonetheless, the insights gained from this study are not limited to the problem of overfishing, and instead, are relevant to other types of resource use dilemmas such as water use, use of forest products and grazing.

The results of the field study confirm previous experimental findings on the effects of communication and external fines. That is, communication as well as the low fine were effective at reducing extraction in the game. Moreover, in line with previous studies, even under the baseline treatment representing the open-access regime, a substantial amount of cooperation occurred among participants in the study. The comparison between the fine and the market treatments suggests the presence of a framing effect between these two settings. When an economic incentive is framed as a fine as opposed to a market-like mechanism, it reduces more extraction in the game. This implies that the institutional context matters for individual behavior, and that the same level of economic incentive may perform differently if the context is changed. This is the first experimental finding, to the best of my knowledge, to support the hypothesis that market reasoning may lead to unintended environmental outcomes in resource use dilemmas. Moreover, the study revealed that selfish and non-selfish individuals may respond differently to a market-like incentive. While the market treatment reduced the extraction of non-selfish individuals on average, selfish individuals did not significantly change their extraction under the same treatment.

These results have important implications for real world policy making in resource use dilemmas. First, as demonstrated in the institutional and ecological economics literatures, institutions do not merely change underlying incentive structures;

instead, they also change the decisional context. If this is not taken into account, environmental policies may actually have unintended consequences, as the present study confirms. Second, selfish and non-selfish individuals may respond differently to policy instruments, therefore, assuming all resource users will behave opportunistically may be counterproductive. Adopting a one-size-fits-all approach entailing universal implementation of environmental markets is not suitable in all contexts. This is not to suggest that markets are good or bad per se, but that the interaction between markets and pro-social behavior needs to be better understood to address environmental problems in a more effective and efficient way.

The structure of this dissertation is as follows: Part I reviews and synthesizes different perspectives on environmental policy instruments and experimental studies. Chapter 1 focuses on the traditional view of environmental policy instruments, and presents a common classification of them. Command-and-control type instruments, policies based on economic incentives (environmental taxes, subsidies, environmental markets, payments for ecosystem services), voluntary instruments and policy mix approaches are discussed. Chapter 2 presents criteria used to evaluate environmental policy instruments in public and environmental economics. In public economics and environmental economics, the commonly used criteria include economic efficiency, distributional concerns, and environmental effectiveness. Chapter 2 also underlines that behavioral impacts of environmental policies are also crucial for the effectiveness, and as such, they should be considered alongside the more traditional criteria of economic efficiency, effectiveness and distribution. The chapter at the end points out that behavioral considerations have been missing from these discussions. Chapter 3 summarizes the main challenges to the traditional approach, presented in Chapter 1 and

2, by summarizing discussions on opportunism, externalities, commensurability, and pseudo-markets and shadow-pricing. It also relates issues regarding the performativity of economics to environmental policies, and introduces recent research on alternative perspectives, such as the role of institutions, pro-social behavior, and community-based management. Chapter 4 reviews experimental studies on the environment, beginning with experiments on environmental valuation, which was the first area on the common research agenda of experimental and environmental economists. The chapter proceeds by summarizing empirical results from experiments on environmental policy instruments and resource use dilemmas. While experiments on environmental valuation are presented for the sake of completeness, experimental studies on environmental policy instruments and resource use dilemmas are most relevant—and hence the main theoretical background—for the field experiments in this dissertation.

The fieldwork, which builds on the theoretical and empirical results of the previous chapters, is presented in Part II of the dissertation. Chapter 5 gives a brief overview of fisheries in Turkey by focusing on the role of the state, institutional arrangements, and conflicts and tensions in fisheries in the country. Chapter 6 introduces the research questions, model, hypotheses, and experimental design of the framed field experiments. Chapter 7 describes the data collection procedures in the field. Chapter 8 presents and discusses the empirical results of the experiments. Finally, the Conclusion chapter synthesizes the results and empirical findings of the dissertation, and suggests areas for further research.

PART I

THEORETICAL BACKGROUND

CHAPTER 1

POLICY INSTRUMENTS AND THE ENVIRONMENT:

TYPES OF INSTRUMENTS

The main purpose of this chapter is to review different types of environmental policy instruments by focusing on their advantages and drawbacks. As such, the chapter presents the main points and findings of the vast literature on environmental policy instruments, as well as the practical issues in implementation. The instruments reviewed in this chapter are command-and-control type policies, instruments based on economic incentives (environmental taxes, subsidies, environmental markets, payments for ecosystem services), voluntary instruments and policy-mix approaches. Each of these instruments undoubtedly comes with some advantages and weaknesses, both from a theoretical and a practical standpoint. These issues are discussed in this chapter in order to shed light on the trade-offs that confront policy makers and better position the types of policy interventions used in the field experiments in Part II of this dissertation.

As environmental policy instruments are often designed and implemented based on the neoclassical tradition of economics, it will be useful to first summarize the way the neoclassical school of thought defines environmental degradation. In the neoclassical environmental economics framework, environmental degradation occurs when the value of an environmental good or service is less than the value one would obtain by using or damaging it. Degradation of nature can take one of two forms: overuse of environmental resources—when extraction exceeds economically-optimal levels—or levels of waste or pollution exceed the capacity of the environment to absorb. In either case, the reason for this degradation is presumably the lack of markets or the absence of market prices for environmental goods and services (Adaman & Özkaynak, 2002).

According to this approach, natural resources are over-harvested because access to them is open and property rights are not well-defined. If property rights were clear and well-enforced, economic agents would be able to negotiate the level of harvest with other parties. However, property rights are often not well-defined, and hence, no one takes care of the resource as he or she should in an economical sense, which leads to the infamous "tragedy of the commons" as coined by Garrett Hardin (1968).

Moreover, pollution or waste exceeds what can be economically tolerated when an economic actor does not take into account the costs of pollution it creates through its consumption or production and imposes these costs —the so-called negative externalities—onto third parties (Pigou, 1920). This is again the case when there are no clear property rights dictating who is responsible to clean up the pollution or who has the right to enjoy a cleaner environment. Therefore, as Arthur Pigou argued, the private and the social costs of the economic activity diverge in these two scenarios. That is, in both of these cases, private costs are externalized by the polluters or by over-harvesters and imposed on others, leading to an economically inefficient outcome or to the so-called market failure.

According to the neoclassical approach, externalities need to be internalized to reach Pareto-efficiency, and two solutions are proposed to this end. The first way to restore efficiency is to assign clear and enforceable property rights to economic actors. Well-defined property rights ensure that negotiations take place between economic actors affected by the economic activity in which they can compensate one another. This would improve economic efficiency since each actor acts in their own self-interest and demands payments for the environmental damage that they are subject to, and the other party pays compensation as long as it is smaller than the benefit they enjoy. This is a simple representation of the Coase Theorem (Coase, 1960).

The second way to improve efficiency is to set the rules of economic activities by a regulatory authorities. The regulator can force economic actors to internalize externalities through the use of an environmental policy instrument (Barde, 2000). The regulator has two options: First, it can enforce compliance with command-and-control (CAC) type policies, such as standards, technical specifications, and quotas. Second, it can implement policies based on economic incentives, such as environmental taxes, subsidies, and permit markets, in order to ensure that environmental destruction or resource use entails certain costs. In other words, it can prevent the destruction of natural assets from being free-of-charge.

There are several problems associated with each of these two proposals (Adaman & Özkaynak, 2002): First, the Coasean approach requires zero transaction costs of negotiation or enforcement to lead to an efficient outcome (Kolstad, 2009). In real life environmental conflicts, this is often not the case. Instead, there are substantial asymmetries in access to the justice system, power, and wealth, which may prevent the negotiations envisioned by Coase from happening (Farrell, 1987; Hahnel & Sheeran, 2009). The second approach requires the regulatory authority to act benevolently and with the sole aim of restoring economic efficiency. That is, no lobbying, favoritism, or regulatory capture should occur. Otherwise, government failure will replace or accompany the market failure at hand. Moreover, the regulatory agency, if selfless and free from outside influences as required, needs to calculate the exact level of optimal extraction or pollution. In order to compute this figure, however, it must know the economic value of environmental goods and services, which is quite difficult, since individuals do not usually express their private valuations for natural assets such as forests, parks, and endangered species. Without knowing the marginal benefits to individuals of consuming these natural assets, there is no way to compare these to the marginal benefits of the economic activity that damages the asset—a significant shortcoming of the approach involving the use of environmental policy instruments.

To overcome this issue, environmental economists have developed estimation methods to capture individuals' private values for environmental assets. The most prominent and widely applied method is the contingent valuation (CV) (Mitchell & Carson, 1989). However, this estimation method comes with several problems as well (Spash, 2008; Venkatachalam, 2004). Two of its biggest shortcomings, hypothetical bias and the divergence of results across eliciting methods, are discussed in Chapter 4. Another problem arises from not being able to know the private valuations and production and consumption possibilities of future generations. This is known as the discount factor problem: How does one calculate the benefits and costs of optimal extraction or pollution for people that are yet to be born? How does one choose the appropriate discount factor to find the present value of benefits and costs for future generations? Apart from this intergenerational distribution problem, finally, there is also an intra-generational one: Even within the same generation, different allocations of income may lead to different environmental preferences, prices, and outcomes (Adaman & Özkaynak, 2002). Moreover, given a certain level of income or wealth distribution and various preferences, it appears difficult or may even be impossible to aggregate such preferences to ensure desirable aggregation characteristics for a well-functioning society according to Kenneth Arrow's impossibility theorem (Arrow, 1950).

This standard, neoclassical approach to environmental problems gave rise to the development and implementation of different environmental policy instruments in policy-making. Command-and-control policies and economic incentive approaches are the two traditional categories that have been proposed since the 1920s, as mentioned above. Voluntary approaches and policy-mix perspectives, on the other hand, have emerged more recently.

The broad range of instruments encountered in the real world may be therefore grouped into one of four categories, namely, command-and-control, instruments based on economic incentives, voluntary instruments and policy mix approaches, as Table 1 demonstrates.

Table 1. A Broad Categorization of Environmental Policy Instruments

Command-and-control	Standards, technical specifications, quotas
	Environmental charges, taxes and subsidies Environmental markets Liability and compensation approach Payment for ecosystem services
Voluntary approaches	Unilateral commitments Public voluntary schemes Negotiated agreements
Policy mix approach	A combination of different instruments

Some of the instruments based on economic incentives —charges, taxes, and subsidies—are also called price-based mechanisms, since they prevent the environmental resource from being free-of-charge by effectively establishing a price for it (Beder, 1996). These can be further differentiated from rights-based approaches (for instance, environmental markets and tradable permits), which give polluters or resource users a right to pollute or a right to harvest, respectively. These two subcategories usually have substantially different outcomes regarding effectiveness, efficiency, and distribution in both theory and practice. Another category of instruments is the policy-mix approaches (Lehmann, 2012; Ring & Schröter-Schlaack, 2011). These typically involve using two or more of the above instruments to address different dimensions of an environmental problem more effectively.

Environmental markets, such as tradable permits, markets for water use rights, and individual transferable quotas (ITQs), have become very popular policy approaches over the last few decades. Their prominence is arguably related to their flexibility in abatement, which achieves, at least in theory, economic efficiency (Sandmo, 2000). The same cannot be said, however, for CAC policies and environmental taxes under certain circumstances. While policies that involve direct regulation might achieve pollution abatement targets effectively, in general, they will not be economically efficient. This is because a policy that obliges firms with different levels of marginal abatement costs to achieve the same abatement target cannot ensure efficiency. This policy would require firms with higher abatement costs to undertake costlier abatements. Moreover, environmental taxes may not be efficient if the regulatory agency does not know the firms' abatement costs –which is generally the case (Sandmo, 2000; Weitzman, 1974).

It is theoretically possible to achieve the targeted abatement rates in efficient or cost-effective ways through market-based mechanisms that limit total pollution to a specific level, create a number of emission permits based on this total amount, and enable these permits to be bought and sold in the market. This is widely known as a cap-and-trade mechanism. However, as observed in the EU carbon markets—one of the most vibrant examples of cap-and-trade mechanisms—this method also brings with it various theoretical and practical problems, as discussed in the second section of this chapter.

As previously mentioned, there are two types of approaches to environmental problems. One involves Coasean bargaining, while the other makes use of a regulatory agency and different policy instruments. This chapter will focus on the latter[1] and present a detailed overview of instruments, beginning with CAC type policies, which are the most commonly implemented policy instrument worldwide. The popularity of environmental markets in most of the policy discussions related to the environment is rising as well.

1.1 Command-and-control type policies

Command-and-control (CAC) type policies, also known as "direct regulation," regulate the quantity of resource use or pollution directly via standards, permits, or quotas. The quantity of resource use is specified by, monitored, and enforced by the regulatory authority. Examples include emission standards, product and process standards, and laws of water use. CAC has been widely used since the 1960s, when the first environmental policies were introduced (Sandmo, 2000).

Although CAC approach appears to be a simple and straight-forward way of addressing an environmental problem, it is more complex than it seems due to the fact that it requires the calculation of an exact rate of optimal standard (Adaman &

[1] Coasean bargaining serves well as a theoretical concept, but since it assumes zero transaction and negotiation costs in a bilateral setting, it is not easily applicable in real life policy-making involving commonly non-zero transaction costs and more than two parties to negotiate (Hahnel, 2015).

Özkaynak, 2002). Keeping in mind that there may be a great deal of information asymmetry, uncertainty, and complexity regarding the environmental problem at hand, this is not an easy task for the regulatory authority. Furthermore, decision-makers need to consider environmental, technological, economic, and political criteria simultaneously when deciding on the level of the standard.

In theory, the level of this standard should correspond to the optimal level of environmental degradation where marginal abatement and damage costs are equal. However, this entails some problems in practice. First, costs and benefits must be evaluated in monetary terms. This requires the application of different valuation techniques, which rely either on stated preference or revealed preference methods, such as contingent valuation or travel-cost methods, respectively, and as such, estimation is especially difficult for benefits (Cropper & Oates, 1992; Spash, 2008). It also relies on the assumptions that individuals assign some monetary value to environmental goods and services and that the underlying preferences of individuals are stable and exogenous. Overall, these assumptions as well as the valuation techniques themselves are problematic, which has raised several validity and reliability concerns (e.g. Spash, 2008; Venkatachalam, 2003). Second, firms in the industry may not be willing to reveal their true costs. Hence, misrepresentation of costs may become an important obstacle to determining the level of the standard or quota. Misrepresentation of costs is a substantial problem for taxes and tradable permits as well (Kwerel, 1977).

Of course, it is also important to consider uncertainty and irreversibility in ecosystems when determining the level of standards. Environmental aspects necessitate more or less precise knowledge about ecosystem dynamics and thresholds, environmental risks, and the potential health effects of these risks. Collecting information about these factors is not always possible. Therefore, one should adhere to the pre-cautionary principle when implementing CAC policies. According to this principle, the absence of scientific certainty calls for protective measures until scientific research can discover new facts and generate greater certainty about probable outcomes (Kriebel et al., 2001).

Technological factors are also important in the determination of the standard. What is technically feasible at present is a significant constraint for the industry and decision-makers. Therefore, specific CAC instruments often refer to concepts such as “best available technology” and “best practicable means” (Barde, 2000). Technology poses another difficulty: the industry may not be willing to reveal the best available technology to decision makers if disclosure means that firms will be subject to stricter regulations.

No doubt, a set of political criteria should also be taken into account in the implementation of CAC policies. One of the most important political aspects is the distributive consequences of a policy, which includes questions about how different segments of society will be affected and who is to bear the burden or costs of environmental degradation. The simplicity of implementation is another significant political factor. These criteria are all contingent upon public acceptance (Barde, 2000).

The advantage of CAC-type instruments is that they ensure more predictable environmental outcomes and, therefore, have the potential to be more effective if policy-makers are able to monitor and enforce the regulation well. Therefore, environmental problems entailing irreversibility and threshold effects can be addressed with CAC in a more effective way. Oates et al. (1989) demonstrated theoretically that well-designed CAC policies can perform well. Although CAC policies may lead to over-control, if the beneficial effects of keeping emissions below the standard is significantly high, the discrepancy between CAC and incentive-based mechanisms with respect to efficiency may not be very large.

Yet, there are also several important drawbacks to the use of CAC instruments as pointed out several scholars (Barde, 2000; Cropper & Oates, 1992; Kolstad, 2009; Sandmo, 2000). First, the information requirements are high for determining the optimal level of pollution or damage. Moreover, economic agents may misreport their costs or benefits, which in turn makes this calculation even more difficult. Enforcement and monitoring may be difficult and costly as well. If the probability of being caught is small and the fines associated with non-compliance are low, economic agents may choose not to comply at all. Second, CAC instruments do not satisfy the equimarginal principle. That is, marginal abatement costs cannot be equalized across economic agents, which, since the regulator often does not know the true costs of each firm with certainty, leads to excessive total abatement costs. Therefore, it is impossible for the regulator to impose the correct abatement requirement for each firm. As a result, the outcome will be inefficient. Third, industry representatives may try to lobby policy-makers in order to avoid stricter regulation, which is arguably easier with CAC than with taxes or charges. Fourth, firms or other economic agents will only pay for the abatement of a certain percentage of the pollution. This will in effect function as a subsidy to the polluter, creating distortions (Kolstad, 2009). Finally, and perhaps most importantly, CAC instruments are rather static and do not provide enough incentives to innovate and reduce abatement costs. Firms are not given enough incentives to reduce their abatement costs beyond the given level of the standard. New technologies may be incorporated into these policies, but will only be so gradually.

From a distributional point of view, CAC policies do not make polluting firms pay for the remaining emissions. That is, firms only incur abatement costs to reach the imposed target, but do not have to pay for emissions beyond that. Therefore, they impose a smaller burden on polluting firms compared to environmental taxes or tradable permits with auctioning mechanisms (Goulder & Parry, 2008). Moreover, they do not transfer rents from households to firms, as is the case with emissions taxes or trading mechanisms, since prices, such as those related to energy, do not increase much via CAC policy. Lastly, they do not generate revenues that could be used to reduce other regressive taxes.

1.2 Approaches based on economic incentives

Over the last decades, there has been a strong tendency, especially in Organization for Economic Co-operation and Development (OECD) countries, to move away from CAC measures and towards incentive-based policies (Barde, 2000). Beginning in the 1970s, the adoption of environmental taxes and charges paved the way for a first wave of incentive-based approaches. The primary motivator was to increase the efficiency of environmental policies, since those instruments promised large cost-savings. Moreover, in the real world with imperfect information, incentive-based approaches do not place a heavy burden on regulators in terms of information collection regarding polluters' abatement costs (Cropper & Oates, 1992).

One implicit assumption motivating the strong shift towards economic incentive approaches was the belief that environmental policies in general need to be aligned with industries' interests in order to be effective (OECD, 1996). Moreover, from the 1980s onwards, deregulation was reflected in environmental policies as "regulatory reform" and an alternative to the rather inflexible CAC policies of the previous decades (OECD, 1997a). Tradable permit approaches and individual transferable quotas, as such, became more popular, albeit still limited in terms of implementation.

The most commonly used instruments based on economic incentives are charges, fees, taxes, subsidies, tradable permits, and liability and damage compensation. Fees, charges, and taxes involve a payment for each unit of pollution. The fee or tax rate ensures that all firms' marginal abatement costs are the same. This is called the equimarginal principle as mentioned above. If marginal costs are not equal across economic agents, there would still be some room to reduce the overall costs of the policy by making the low-cost polluter emit one more unit and the high-cost polluter emit one less unit (Kolstad, 2009). There would, therefore, be some potential for Pareto improvement. In turn, this implies that equating the marginal costs can only ensure economic efficiency in a Pareto sense, since it would have minimized the total abatement costs of the policy.[2]

A tradable or marketable permit ensures that the quotas or permits can be bought and sold in a market. That is, agents are able to sell or buy the right to pollute or to use the resource. Due to trading, a permit price is established in the market, which equalizes the marginal costs of abatement across agents. Thus, the equimarginal principle holds for tradable permits as well, leading to Pareto efficiency.

If there is no uncertainty regarding marginal abatement costs, economic theory suggests that the outcome of a Pigouvian tax will be equivalent to that of a tradable permit (Sandmo, 2000). Yet, with substantial uncertainty surrounding the marginal

[2] More on Pareto efficiency and Pareto improvement principles will follow in Chapter 2.

abatement costs of polluters, things can turn out different. If the regulator does not know the marginal abatement costs of polluters, a tax leads to an uncertain quantity of pollution, but to marginal costs that are known for sure prior to the implementation. Tradable permits, on the other hand, would achieve a known level of pollution (due to the cap on quantity), but the marginal cost of pollution control would remain unknown in that case (Weitzman, 1974).

In yet another type of policy based on economic incentives, liabilities or damage compensation, the regulator makes the polluting agent explicitly responsible for the entire cost of a risky activity in cases where an accident occurs. Theoretically, the threat of being held responsible for the full damage caused by an accident should give the polluting agent a sufficient incentive to take the economically optimum level of precaution (Barde, 2000). This would also satisfy the equimarginal principle, since the polluting firm then needs to equate its marginal abatement costs to the marginal damage costs. However, this approach is predominantly based on the following assumptions: Damage costs can be determined precisely; polluters and victims are identifiable; there is a causal relationship between the pollution and the damage, which the regulator can clearly identify; and, finally, the transaction costs of enforcing such a procedure are not too high. However, these prerequisites are not easy to satisfy.

The following advantages of economic incentive approaches make them very popular in both theoretical and practical discussions of policy choice (Kolstad, 2009): First, informational requirements are lower compared to CAC policies. Second, economic incentives drive firms to innovate to find cheaper means of pollution abatement. Moreover, unlike taxes, the polluting agents have to pay for both the abatement as well as for the residual pollution that they do not abate. Therefore, agents are not implicitly subsidized. Finally, due to the fulfillment of the equimarginal principle, instruments based on economic incentives involve cost-saving advantages.

Instruments based on economic incentives also have some drawbacks (Barde, 2000; Cropper & Oates, 1992; Kolstad, 2009; Sandmo, 2000): First, designing economic incentives that address the full scale of complexity and irreversibility involved in ecosystems is very difficult. There are many dimensions of pollution across factors such as space, time, and concentration. It is not realistic to assume that all these factors can be taken into account and that efficient outcomes will be achieved. Second, many economic incentive approaches entail a transfer of revenue from the private sector to the government. This is more apparent for taxes and fees, but it is also the case with tradable permits when permits or quotas are auctioned off rather than grandfathered, e.g., given for free. This is also one of the reasons why industry actors often raise objections about economic incentive policies. Another factor that makes the use of economic incentives rather difficult is that the level of incentive should be continuously adjusted whenever new information about different aspects of environmental degradation is discovered. This is even more important if the environmental problem at hand involves a great deal of uncertainty.

Following this general introduction to economic incentive approaches, the chapter now focuses on each of them by distinguishing between the four types of policies most commonly used in practice. These include environmental taxes and subsidies, the more recent and popular environmental markets, the liability approach, and, the most recently developed instrument, the payments for ecosystems approach.

1.2.1 Environmental charges, taxes, and subsidies

To internalize a negative externality, one option is to use a tax or a charge so that the polluter includes the costs inflicted on other parties into his or her own calculations. Similarly, a charge can be imposed on the polluter so that a payment follows for each unit of the damage. This approach is based on the assumption that the damage can be measured precisely.

The difference between a tax and a charge is that, in general, a charge is a compulsory payment, in proportion to which a government provides a service, while a tax is a compulsory payment to the government with a non-proportional service or benefit in return (Barde, 2000). Emission charges or taxes are, on the other hand, usually paid in proportion to the quantity or quality of emissions. Another category of taxes, product taxes, increases the relative prices of industrial outputs that cause pollution (energy taxes being the most prominent example). Subsidies, on the other hand, are payments made to firms or consumers to encourage an economic activity with a positive externality. Yet, some subsidies provide perverse incentives for economic agents, such as subsidies provided for fossil fuel use. As these encourage environmentally damaging activities, there is much debate in Europe and the United States about how to eliminate them.

Theoretically, the Pigouvian tax (named after Pigou's work in the 1920s) should be imposed at a level that ensures the optimum level of pollution, where the marginal abatement cost is equal to the marginal damage cost. The tax would then minimize total abatement costs in the economy, since firms with lower abatement costs would make more abatement and firms with higher abatement costs would make less abatement.

Since all firms equate their marginal abatement costs to the same tax rate, the equimarginal principle would be reached.

This is in contrast to a uniform CAC policy requiring each firm to reach the same level of a standard. Therefore, a Pigouvian tax implies substantial cost savings relative to CAC measures (Tietenberg, 1990). Moreover, a tax provides an incentive to reduce pollution beyond a standard, since firms will search for better abatement technologies in order to reduce their total costs. If a firm can make use of a more efficient technology, then this is translated into two types of cost savings: lower abatement costs and lower tax payments. Therefore, a tax is usually seen as providing a permanent incentive to search for better technologies or to innovate (Cropper & Oates, 1992). Another advantage of an environmental tax is the tax revenue that it generates. This revenue may be earmarked for specific projects of environmental improvement. The so-called double dividend hypothesis implies that tax revenue gained from environmental taxes can be used to reduce other distortionary taxes in the economy, which would further enhance overall efficiency not only by correcting for market failure, but also by reducing inefficiencies stemming from distortionary taxes (Sandmo, 2000).

On the other hand, the industry may oppose taxes much more than CAC measures, since negotiating a tax rate is more difficult than negotiating a certain standard in practice. Moreover, taxes inflict a second source of cost on firms: firms need to abate and therefore have to pay abatement costs in addition to the tax. With a standard, however, firms do not bear any additional costs beyond reaching the standard for pollution abatement (Barde, 2000). Opposition to taxes may also come from the general public. Objections are often based on the idea that paying environmental taxes is like “purchasing indulgences,” and it is considered to be “purchasing the right to pollute" (Goodin, 1994). Another argument against the use of environmental taxes, that it may damage the international competitiveness of certain sectors, comes mainly from the industry. However, from the perspective of cost-savings and flexibility, taxes can be seen as a viable and important device for better environmental quality.

How effective an environmental tax is at correcting market failure depends heavily upon the presence of a competitive market. In the case of market power, when there is already a distortion in the economy, the monopolist restricts its output to maximize profits, which keeps pollution rather low. Theoretically, pollution may actually be too low, or lower than the optimum level. Since it would even further reduce output and pollution, implementing a tax may not be the best strategy in this case. From the perspective of economic efficiency, it is not desirable for pollution levels to be too low, since this would mean that too many resources are being allocated to pollution abatement (Kolstad, 2009). A monopoly is an important exception where the Pigouvian tax cannot correct the market failure. Under certain conditions, the monopolist should even receive a subsidy to increase output to the socially-efficient level (Buchanan, 1969).

If there is uncertainty regarding marginal abatement costs—in cases when the regulator does not have perfect information about the firms' pollution control costs and firms are not willing to reveal it to the regulator—a tax will produce an uncertain environmental outcome. CAC measures or cap-and-trade mechanisms, however, lead to known levels of pollution. Thus, if there are significant threshold effects so that a small variation in environmental damage is detrimental for the ecosystem, a tax should be avoided. On the other hand, if the marginal damage is constant, a tax is preferable

(Sandmo, 2000; Weitzman, 1974).[3] Weitzman's results imply that when the costs and benefits of pollution abatement are uncertain, one needs to be more careful about the probable outcomes of the policy. Distributive problems may also arise due to the tax incidence. For instance, a carbon tax that increases fuel prices would impose a heavier burden on low income groups due to its regressive nature (Pearce, 1991).

Instead of a tax, it is also theoretically possible to provide a subsidy to the polluting firm in order to encourage it to reduce emissions. Economic theory suggests that in the short-run and with identical firms, this should lead to exactly the same outcome regarding emissions reduction (Cropper & Oates, 1992). However, when firms are heterogeneous regarding costs, the result will be different. Due to the subsidy, inefficient firms may continue to operate in the short-run, which they would not do if they were required to pay a tax. Therefore, such a subsidy is not economically efficient. Another argument against subsidies is that they involve lump-sum transfers to the industry that could be used for other public purposes. In the long-run, subsidies also allow more firms to operate in the industry despite their inefficiency. Hence, output and pollution may expand. Moreover, the prices of goods that generate pollution will be lower than their true costs due to the subsidy, which does not send the right signal to economic agents in the market. Therefore, especially in the long-run, subsidies are not efficient measures, nor are they in line with the polluter-pays principle. Even worse, firms may keep their emissions deliberately high in order to qualify for a subsidy. Subsidies also fail to provide an incentive for firms to reduce emissions through research and development. Such distortionary subsidies are very different from the

[3] These results should be evaluated with caution, since the model assumes linear marginal benefit and cost functions and additive uncertainty.

environmental subsidies discussed above (Barde, 2000). Distortionary subsidies are those that provide perverse incentives to firms in polluting industries such as coal and transportation. Environmental policy should be directed towards the elimination of these environmentally harmful distortionary subsidies. However, there may be substantial resistance against the elimination of such subsidies by industry.

Regarding distributional aspects, the impact of taxes on different income groups depends on how tax revenue is used. Most environmental taxes, especially taxes in the energy sector, adversely affect low-income groups, as energy-related expenditures make up a larger share of their income. As such, they are usually regressive taxes (Barde, 2000; Parry, Sigman, Walls, & Williams, 2006). To compensate for this, governments can earmark their environmental tax revenues to reduce other regressive taxes. In that case, low-income groups can actually benefit from the introduction of an emissions tax.

1.2.2 Environmental markets

The work of Coase (1960) and Dales (1968) inspired the creation of markets for pollution rights. Cap-and-trade systems and tradable permits are widely endorsed today by resource economists and industries for being both efficient and effective measures of environmental policy. Theoretically, environmental markets overcome several drawbacks of CAC measures and environmental taxes. By fixing the allowed amount of pollution or extraction, policy makers can directly control environmental effectiveness. Another big advantage is that environmental effectiveness can be achieved without compromising efficiency, since tradable permits are in line with the equimarginal principle. As long as the market for tradable permits functions well, the total costs of pollution abatement will be minimized. Theoretically, the advantages of both CAC policies and taxes are merged in this instrument, while the disadvantages of inefficiency and uncertainty regarding environmental outcomes are avoided. Moreover, while implementation of an environmental tax requires continuous adjustment of the tax rate to the changing economic conditions, the regulator can avoid this cumbersome procedure with tradable permits. In contrast to taxes, tradable permits easily adjust to inflation, as permit prices increase without any intervention by the regulator (Cropper & Oates, 1992). Tradable permits are, therefore, popular measures, especially in the United States (Kolstad, 2009). Another advantage is that permits can be bought from existing firms such that new firms can enter the industry without compromising environmental effectiveness.

There are, however, a number of disadvantages to tradable permits. Some of these problems are related to implementation, but there are also theoretical problems associated with market power and transaction costs. From an economic efficiency perspective, the initial allocation of permits should not matter at all. That is, market equilibrium exists and should be efficient regardless of which firms are initially allocated permits, assuming price-taking behavior in the permit market (Montgomery, 1972). However, Hahn (1984) demonstrated that when there is market power in the permit market, the initial allocation of permits can be problematic for efficiency as well as equity purposes. In this case, the dominant firm with market power will manipulate the market by selling or buying more or less than the efficient amount that the theory predicts. Its emissions will therefore not coincide with the efficient level. The only way to overcome this loss of efficiency is to allocate to the large firm the exact efficient amount of permits (Hahn, 1984). Another problem may arise when firms competing in the output market withhold permits for the purpose of preventing new firms from entering the industry.

The initial allocation of permits will also matter if the transaction costs of trading permits are relatively high, which will reduce economic efficiency (Stavins, 1995). Transaction costs may be large due to: i) search and information costs; ii) bargaining and decision making costs; and iii) monitoring and enforcement costs (Kolstad, 2009). In fact, tradable permit systems often require sophisticated rules and infrastructures for monitoring and enforcement. The rules are complex and entail high administrative costs. All of these factors can act as impediments to economic efficiency. A survey carried out by Hahn (1989) in the United States and Europe implied that high transaction costs are an important reason why such markets fail in practice.

There are two primary ways of allocating permits. One way is to auction off the permits, which forces a massive resource transfer from the industry to the government and is very likely to be resisted. Another means of initial permit allocation is known as "grandfathering," in which permits are simply given to firms for free. This is preferred by existing firms in the industry, but is problematic for new entrants since they will need to purchase them from existing firms. Grandfathering is also opposed on the grounds that it creates rents and windfall profits for certain firms. One proposed solution to these problems is a zero-revenue auction in which permits are first auctioned and the revenue is then distributed to firms in an ex-ante agreement, based on current emissions, historic emissions, or the ability to pay (Hahn & Noll, 1982; Kolstad, 2009).

Furthermore, if the permit market is thin—that is, there are only a few firms in the market and an insufficient number of transactions—then permits do not change hands enough to ensure efficiency. The pollution level will be fixed, but cost-savings will not be achieved. Compared to emission taxes, tradable permits markets are more difficult and costly to implement, since setting them up requires heavy government involvement and regulation, contrary to environmental markets, which work without any intervention. They require very complex rules and infrastructures to implement, making transaction costs high.

Spash (2010) and Lohmann (2010) further argue that complexities in carbon market, such as measurement and accounting issues, uncertainty, corporate power, and speculation, are obstacles to achieving economic efficiency in practice. Accordingly, basic economic models on carbon trading are abstracted from such real world complexities and neglect important issues such as scientific limits in climate science, the impact of powerful interest groups and lobbies, and behavioral and ethical considerations. Similar criticism often comes from radical environmental movements, which oppose tradable permits markets on the grounds that they amount to the commodification and marketization of nature.

Moreover, Spash (2010) points out that voluntary efforts by individual consumers to contribute to emissions reduction may be crowded out if permits markets convey the message that the problem of climate change has been taken care of. That is, those individuals who have pro-environmental preferences may be relieved from the moral responsibility of pro-environmental action if they come to believe that permit markets are solving the problem of climate change, which has not been confirmed empirically.[4] Environmental markets are usually questioned on the grounds that they

[4] There is mixed evidence concerning the U.S. experience with tradable permits in the 1990s in terms of environmental effectiveness (Cropper & Oates, 1992). However, there is no empirical evidence that the European Emissions Trading System for carbon permits has been able to prevent global warming so far.

cannot convey any message regarding social and ethical norms, nor can they be used as a platform for the expression of pro-social or pro-environmental values (O'Neill, 2001). Thus, even if such markets work efficiently and effectively, they need to be complemented by public policies that facilitate voluntary action (Spash, 2010; Vatn, 2005).

In terms of distribution of economic benefits, grandfathering of permits is clearly distinguishable from auctioning. The free allocation of permits creates rents for polluting firms and may actually increase their profits if the abatement targets are low (Goulder & Parry, 2008). In fact, the first stage of the European Emissions Trading System generated windfall profits for many polluting firms subject to the system. When these permits are auctioned, part of these rents are transferred to the governments, which may then be used to reduce other regressive taxes or to increase the amount of publicly-provided goods and services. In those cases, the distributional impacts of the system on low-income groups are beneficial. However, this type of allocation of permits is likely to be opposed by polluting firms—usually, large-scale multinationals operating in the energy and transportation sectors—with immense resources and lobbying power. Moreover, the grandfathering of permits is most likely to have adverse distributional effects across households if this allocation increases the firms' equity values. As high-income groups are more likely than middle- or low-income groups to own such equities, the latter will become relatively poorer (Goulder & Parry, 2008).

With respect to the distribution of environmental benefits, on the other hand, the implementation of an environmental market for local pollutants, such as sulfur dioxide, may lead to leakages to other regions. In other words, polluting industries may be displaced to other regions of the world in which such markets are not operational. This may create hot spots of pollution in certain parts of the world where environmental regulations are lax, leading to an unequal distribution of environmental costs and benefits.

1.2.3 Liability and compensation

Requirements that make polluters pay for the damage they cause and compensate the victims of pollution make sense on both efficiency and equity grounds. The liability approach creates an expected price for pollution or environmental degradation. In theory, it should be as efficient as a Pigouvian tax. However, this approach can only lead to economically efficient outcomes if rather unrealistic conditions are met (Barde, 2000): Damage costs must be evaluated precisely; polluters and victims must be identified; the causal chain between the pollution and the damage must be established; and all of these procedures must be enforced without excessive costs. However, these conditions are difficult to satisfy in real-life policy-making contexts. Moreover, the expected price may not be equivalent to the actual price due to weaknesses in the legal system. Firms may, for instance, declare bankruptcy to avoid compensation payments (Cropper & Oates, 1992). Therefore, CAC and incentive-based approaches are often preferred to liability approaches.

Yet, there is also a tendency in public discussions to put more pressure on polluters, hold them responsible for the damage a specific industry or firm causes, and demand compensation. This pressure may come either from the governments, NGOs, or a larger public. Especially following disastrous accidents, such as large-scale oil spills, compensation measures dominate public discussions. But wide scale implementation of such measures is difficult if the polluting firms are economically powerful and have resources to lobby the regulatory authorities or the government (Barde, 2000).[5]

1.2.4 Payments for ecosystem services

An incentive-based approach that has been offered to address the problem of overuse or degradation of natural resources is the Payment for Ecosystem Services (PES). Ecosystem services are usually defined as “the benefits people obtain from ecosystems” (Millennium Ecosystem Assessment, 2003, 2005). Accordingly, PES schemes are aimed at translating non-market values of the environment into monetary incentives for local actors so that they both provide and protect these services (Engel, Pagiola, & Wunder, 2008). They rest upon the belief that “the protection and long-term sustainability of diverse ecosystems will only be viable if the full range of services provided by these ecosystems are economically accounted for” (Corbera, Soberanis, & Brown, 2009, p. 743). However, this view has been criticized by some scholars for prioritizing economic reasoning over a more traditional, conservationist approach based on existence and non-use values (Liverman, 2004).

More precisely, PES can be defined as a voluntary transaction in which an environmental service is bought from a service provider conditional upon the actual provision of this environmental service (Wunder, 2005). For instance, a farmer who would like to use a forest area as cropland would generally receive more benefits if it were converted to cropland than if it were kept as a forest. A PES scheme would increase the benefits of forest conservation over the benefits of alternative uses of this

[5] Of course, lobbying is an issue for all other types of environmental policy instruments as well.

land by paying the farmer for conservation. The PES buyer can be a governmental body or a private third-party with a substantial interest in the conservation of the area. Theoretically, the economic incentive provided to the service provider (here, the farmer) internalizes the negative externality of turning the forest into farmland. The protection of the given ecosystem thereby becomes more attractive to the owner of the land, and this private negotiation improves economic efficiency. PES can be seen as an application of the Coase Theorem (Coase, 1960). The incentive can also be seen as an environmental subsidy provided to the service provider coupled with a fee for the beneficiary of the ecosystem service.

Along with other incentive-based mechanisms, a PES measure with a fixed per hectare payment scheme would be preferable to CAC measures from the perspective of efficiency. This is because it would encourage landholders with higher marginal costs of conservation to conserve less and those with lower costs to conserve more. Arguably, PES can also overcome the problems of weak enforcement, information problems, and high transaction costs typically encountered in developing countries. Distributional considerations may also favor PES if CAC measures are imposed as restrictions on relatively poor resource users who depend on this resource for their livelihoods (Engel et al., 2008)

There are, however, several shortcomings of PES schemes as underlined by Muradian et al. (2013), and Pagiola, Arcenas, and Platais (2005). Accordingly, the payment to the service provider may be insufficient to induce social efficiency; payment may follow for a service that would have been provided anyway; the change in land use may be only temporary; destructive land use may leak to other areas where the PES is not applied; or the PES scheme may lead to a price increase in a resource, which then increases environmental degradation in other areas. Moreover, the causal relationship between the land use practice and the environmental service may be difficult to demonstrate. It may also be difficult to measure the value of the ecosystem service. Further, although PES schemes are widely assumed to improve conditions regarding the poverty of ecosystem services providers, there is little empirical evidence to support this. For instance, it is unclear whether poor households can actually participate in PES schemes. Researchers also warn that poorer households may be indirectly and adversely affected by some PES measures.

Several experimental studies have also tested the behavioral outcomes and effectiveness of PES-like measures in a controlled field setting, providing mixed results so far (Handberg & Angelsen, 2015; Midler, Pascual, Drucker, Narloch, & Soto, 2015; Narloch, Pascual, & Drucker, 2012). Muradian and Cardenas (2015) argue that policy makers take it for granted that correcting market failures will solve such conservation problems. The implementation of PES schemes takes a similar approach. However, this issue is not merely a technical problem that can be dealt with by a given set of policy instruments. Rather, since such problems are embedded within the broader societal context of local communities, values and morality also need to be considered when addressing conservation problems. Moreover, it is argued that although PES is appealing in many cases, policy-makers should not apply PES blindly to every conservation issue. Providing more information to service providers and creating awareness can sometimes better improve land use practices.

1.3 Voluntary approaches

As an alternative to CAC and incentive-based approaches, voluntary approaches were proposed in industrialized countries in the 1970s to give more flexibility to both the regulator and the regulated actors. Voluntary measures usually involve commitments from polluting firms to improve their environmental performance (Barde, 2000). There are three main categories: i) unilateral commitments, which are developed by private sector actors to prevent future regulation and/or to improve their public image; ii) public voluntary schemes, which imply voluntary compliance with principles and guidelines proposed by regulatory authorities; and iii) negotiated agreements between public authorities and the private sector, which are endorsed by the private sector to avoid stricter future regulation (Börkey et al., 1999, as cited in Barde, 2000).

The main advantage of voluntary approaches is the flexibility they provide to the private sector as well as to government authorities. Usually, this approach introduces a hypothetical cap on pollution that the industry complies with using the most cost-effective measures. Voluntary approaches can lead to cost savings, but the equimarginal principle is not necessarily satisfied (Barde, 2000). Since they convey a message of cooperation and proactive action on the side of industry, voluntary measures are usually endorsed by different actors, including NGOs and trade unions. However, it is difficult to detect whether positive outcomes are a result of the voluntary measure or would have occurred anyway as a result of business-as-usual. Therefore, it is unclear whether these measures actually improve environmental performance.

Yet, voluntary approaches may effectively improve environmental performance, especially when the government threatens to impose CAC-type measures or taxes on agents who do not comply with them (Barde, 2000). This will be more effective when the number of polluters is low, so that the parties can more easily negotiate and sustain the agreement. However, when a large number of polluters are present, it is more difficult to sustain collusion that lacks any economic incentives.

Some of the drawbacks of these measures include the potential for free-riding, high transaction costs due to negotiations involving numerous actors, high administrative or legal costs for implementation, regulatory capture, and weak controls of the policy (Sandmo, 2000). It is important to note that most economists do not support voluntary approaches since standard economic theory is based on the assumption that individuals have stable preferences and that behavior cannot be altered without economic incentives (Stigler & Becker, 1977, as cited in Sandmo, 2000). Hence, most economists doubt that protection of the environment is possible via voluntary behavior.

At least on an individual basis, however, economic agents sometimes act as citizens with a moral responsibility to protect the environment. That is, they recycle waste, do not litter public places, and do voluntary work for environmental NGOs. This is very unlikely to occur from the perspective of mainstream economics, since the costs associated with these types of voluntary behavior are larger than the material benefits one gets. However, people may adopt voluntary pro-environmental behavior when they derive some non-material benefits from this. There are two major points to consider in this respect: The first concerns the assumption of stable preferences. A preference, as a ranking of alternatives, depends not only upon tastes but also upon available information. Even if tastes are relatively stable, available information about the natural environment changes drastically from year to year. When the public has more information about environmental change, individual preferences may change accordingly (Sandmo, 2000). The second is related to incentives. Even if there is adequate information available about the negative consequences of environmental degradation, a majority of individuals may not modify their behavior accordingly. Therefore, economists usually call for a change in underlying incentive structures. However, there is also evidence that under certain circumstances people may behave as citizens acting for the common good as will also be underlined in Chapter 3 (Sagoff, 2008; van den Bergh et al., 2000; Vatn, 2005). Another related aspect is that economic incentives aimed at internalizing negative externalities may conversely crowd out moral sentiments or civic virtues (Bowles & Polanía-Reyes, 2012; Bowles, 2008; Frey, 1997).

1.4 Policy mix approaches

The consideration of a diverse set of economic, social, and environmental criteria usually leads to a policy mix approach. Due to the substantial trade-offs between environmental effectiveness and economic efficiency, policy makers may prefer to employ a combination of policy instruments rather than only one. Given that the real-world is much more complicated than the policy problems found in textbooks, as Cropper and Oates (1992) point out, distributional concerns can be better addressed using a combination of instruments. Recognizing that no instrument alone is a panacea for complex environmental problems, direct regulation approaches, for instance, may be complemented by incentive-based or voluntary approaches, as well as educational and information campaigns. The exact combination used usually depends on the nature of the environmental problem and the existing regulations in a country.

The policy-mix approach is expected to give rise to new and more innovative approaches to environmental policy in the future (Barde, 2000). More recent studies emphasize the need for a policy-mix approach especially for biodiversity conservation (Ring & Schröter-Schlaack, 2011; Santos, Schröter-Schlaack, Antunes, Ring, & Clemente, 2015). Accordingly, this is likely to improve the consequences of the intervention in terms of public acceptability, distributive fairness, cost-savings and environmental effectiveness (Ring & Schröter-Schlaack, 2011).

1.5 Towards a thorough analysis of environmental policy instruments

In sum, policy-makers have a diverse set of policy options to deal with environmental problems thanks to the developments in mainstream economics since the 1920s. Several institutions such as OECD, Environmental Protection Agency (EPA) and United Nations Environment Programme (UNEP) publish reports to guide policy makers in environmental protection as well (EPA, 2015; OECD, 1997a, 2008; UNEP, 2004). While these institutions have lately put a stronger emphasis on economic instruments, in practice, CAC-type policies are still common, presumably because they are easier to implement. On the other hand, instruments based on economic incentives are also attractive for policy-makers since they promise substantial cost-savings.

Indeed, studies reviewed in this chapter pointed out that the main criteria used in environmental policy appraisals are usually three-fold: environmental effectiveness, economic efficiency and distributional impacts. The next chapter reviews the theoretical underpinnings of each criterion in more detail to show how well these concepts have been studied in the literature so far. Despite this strong focus on these three criteria, behavioral impacts, an important aspect to consider for the environmental effectiveness of interventions, have not been part of these discussions. Fortunately, OECD and World Bank have recently begun to draw attention to this aspect with respect to public policies in general (Lunn, 2013; The World Bank, 2015). This is a promising avenue for a more thorough analysis of environmental policy instruments as well, as this dissertation will point out.

CHAPTER 2

CRITERIA FOR EVALUATING ENVIRONMENTAL POLICY INSTRUMENTS

As Chapter 1 has demonstrated, environmental policy instruments are usually evaluated based on their environmental effectiveness, economic efficiency, and distributional impacts. Some of these criteria, however, are more complex than they seem at first glance. It is important to understand what is really meant with each criterion, as there are substantial trade-offs between them. This chapter therefore aims to review the detailed theoretical studies on these three criteria used in environmental policy evaluations, before arguing that behavioral impacts need to become another criterion in policy appraisal studies.

The chapter will reveal that economic efficiency is often used in public policy discussions to simply imply cost-savings. However, the concept is multi-faceted. A review of the theory reveals that efficiency, cost-effectiveness, Pareto efficiency, Pareto improvement, and potential Pareto improvement are unique theoretical concepts that are all referred to as efficiency in real world policy discussions (Griffin, 1995; Vatn, 2002). One needs to understand and carefully distinguish between these different concepts in order to understand the difficulties of operationalizing efficiency in environmental policy-making.

A second criterion used in policy appraisal is distribution and fairness. This criterion requires careful analysis as well. From a philosophical perspective, important distinctions exist between consequentialist (e.g. Bentham & Mill, 1987), procedural (e.g. Nozick, 1973; Rawls, 1971), and rights-based theories (e.g. Scheffler, 1982; Sen, 1987). Economists, moreover, have traditionally distinguished between equitable and fair outcomes, refining these concepts even further. Time is also an important dimension: inter-generational and intra-generational justice are two separate issues of distribution. The third criterion—environmental effectiveness—seems to be a less controversial concept at first glance. Yet, it is usually not clear whether a given instrument will be able to achieve the intended outcome given the complexities of real life, such as information requirements, uncertainty, irreversibility, and threshold effects.

While neoclassical economics has traditionally focused on efficiency, effectiveness and fairness much in detail, behavioral impacts have so far been not part of the discussions. However, the need to better understand human behavior and motivation has become increasingly urgent given the acceleration and spread of alarming environmental problems in the 21st century. After a through review of the three main evaluation criteria, this chapter will argue that the lack of attention to behavioral aspects may be detrimental to effectiveness of policy instruments. A more thorough analysis of policy instruments, therefore, requires a careful consideration of behavioral impacts as a separate criterion, in addition to efficiency, distribution and effectiveness.

2.1 Economic efficiency and its close relatives

An efficient policy can be defined in operational terms as a policy that maximizes the positive difference between its benefits and its costs (Zylics, 2000). Theoretically, efficiency is achieved when the marginal abatement costs of pollution are equal to the marginal damage costs of pollution across polluters. According to the standard theory, this is the optimal emission or pollution level that needs to be set as an environmental target. However, in real life, this optimal level is often difficult to determine due to the complexities of the environmental problem at hand or the difficulty in measuring the costs and benefits of pollution abatement in monetary units. Therefore, efficiency may not be attainable for many policy instruments in practice.

An alternative concept, cost-effectiveness, is often used in practice to refer to economic efficiency—even though the two concepts are significantly different. As a less stringent concept, cost-effectiveness implies that a given emission target is to be achieved with the lowest possible cost. Among a set of policy alternatives that are supposed to achieve the same outcome, the one with the lowest cost should be chosen.[6]

In economic theory, there is a further refinement of the concept of efficiency: economists distinguish between Pareto optimality, Pareto improvement, and potential Pareto optimality criteria (Bromley, 1990; Griffin, 1995; Vatn, 2002). Pareto optimality (PO) is a state in which no individual's utility can be increased without reducing the utility of other individuals. Pareto improvement, on the other hand, implies that there is room to increase the utility of some individuals without reducing the utility of others. Potential Pareto optimality (PPO) is substantially different in that it refers to a situation where some individuals face a reduction in utility, while others are set to gain in utility and the winners are able to compensate the losers.

The concept of Pareto optimality plays two major roles in policy analyses. First, it is used to establish a reference point to evaluate market outcomes. That is, competitive market outcomes are expected to be Pareto optimal, as the First Welfare Theorem proves, albeit under strict assumptions. Based on that finding, policy analysts

[6] According to this concept, costs should include not only implementation and administration costs, but also other indirect social costs, for instance, those resulting from the encouragement of specific technologies and the discouragement of alternatives (Davies & Mazurek, 1998). One way to achieve cost-effectiveness may be to keep compliance costs as low as possible. This is more likely if policies involve simple operational procedures and the number of new institutions is limited (Gupta et al., 2007).

usually point to competitive markets as desirable structures. Yet, the strict assumptions of the First Welfare Theorem—which include, among others, atomistic agents, decreasing returns, absence of externalities, public goods, and merit goods—are rarely satisfied in real life. Therefore, when policy makers refer to the merits of competitive markets, the reference to Pareto optimality is rather weak and not well-established (Griffin, 1995). More specifically, in the case of environmental policy, the relationship between competitive markets and Pareto optimality is even weaker since environmental issues often involve externalities and public goods.

A second use of the PO criterion can be found in the application of compensation tests. Compensation tests are undertaken when there will be losers and winners of a public policy. However, since PO is not a highly discriminating concept and is difficult to apply to real world policy issues, compensation tests cannot use the PO criterion in its pure form. Instead, compensation tests use the PPO criterion, which is substantially different from the PO rule in that it refers to a change in policy, which can cause some individuals to gain and others to lose and in which it is possible for the gainers to compensate the losers (regardless of whether compensation is actually undertaken after the effects of the public policy are felt). Applying the PPO rule requires the aggregation of welfare measures across individuals. This is in stark contrast to the PO rule, which maximizes individual utility instead of aggregate utility. In practice, efficiency analyses are usually based on the PPO rule. However, this is rarely stated explicitly in reports and analyses of environmental policy. This can be confusing and misleading since the PO and PPO rules have different theoretical backgrounds and implications (Bromley, 1990; Griffin, 1995; Vatn, 2002).

Although it is easier to apply to real world public policy problems, the PPO rule has several fundamental shortcomings. First, compensation poses a problem. An argument used to provide normative support for the PPO criterion is that even if some harm is done to certain individuals in the short-run, repeated application of the PPO rule will benefit each individual over time (Hicks, 1941, as cited in Griffin, 1995). However, this only holds if there are no biases associated with the application of the PPO criterion systematically favoring certain groups or individuals in society. In contrast, the PPO criterion has a status quo bias in that it compares each previous state of the economy to the next policy proposal such that the initial states to which the policy is compared are never independent of each other. This can bring harm to politically powerless, marginalized groups in society in a systematic manner. This normative argument in favor of the PPO rule is therefore not well justified (Griffin, 1995)

Second, while the PO criterion maximizes an arbitrary agent's utility for all possible utility levels and for all other individuals, the PPO criterion does not preserve individual utility at any certain level. Instead, it admits reductions in the utility of certain individuals as long as these are offset by increases in the utility of others. This is usually justified by Kaldor's (1939, as cited in Griffin, 1005) argument on the separability of production and income distribution. Accordingly, production and income distribution can be analyzed as two separate objects. Given separability, compensation tests that rest on the PPO criterion are fine since any harm can be offset by redistributive policies if they can be undertaken without considerable costs. However, empirical evidence suggests that redistribution usually involves excessive costs (see for instance, Ballard, 1988).

Third, monetarization poses another problem for the PPO rule. As already mentioned, compensation tests employ the PPO rule. The tests use either the Kaldor or the Hicks criteria, but this choice has strong normative implications, since these two criteria differ in their comparison bases. Two measures are used in compensation tests to approximate a change in utility: Compensating variation and equivalent variation. Compensating variation rests on the Kaldor criterion and makes use of a "status quo ante" approach (Graaaf, 1957; Sen, 1979). That is, it takes the initial utility level of an individual as the baseline and refers to the monetary amount that she must be given to reach her initial utility level after a policy is introduced. The equivalent variation rests on the Hicks criterion and focuses on the monetary change that would have been necessary to bring the individual to the final utility level in the absence of the policy proposal. Using different utility levels as the referent of the analysis, these two measures often provide different solutions to the PPO problem.

The use of the Kaldor criterion may be preferable if the starting state of the economy is seen as legitimate, assuming that the social conditions of the initial position or status quo have already been negotiated and accepted by the public at large. Yet, this is not a realistic assumption, given the presence of impoverished individuals with few opportunities to influence the current state of the economy. The Hicks criterion, on the other hand, is preferable if the final outcome of the policy is a state that society would like to achieve. In sum, the decision of which criterion to use is not value-neutral or objective. Griffin (1995) further argues that considering only the initial and final utility levels is insufficient. He argues that the best utility level might be neither the initial nor the final utility level. Therefore, he asserts, the number of compensation tests cannot be confined to two (one based on Kaldor and another based on Hicks), but that there are an infinite number of compensation tests among which to choose. Which one is actually selected is therefore a normative question.

In cost-benefit analyses (CBA), the PPO criterion is employed (Graham, 1981). CBA does not simply add up the welfare measures across periods. Instead, it assigns different weights to each period. Welfare effects in different periods are thereby discounted by a pre-selected rate. If the discounted sum of welfare measures or the net present value is positive, the policy should be undertaken. When the outcome of a CBA study implies that a public policy is efficient, the result is actually based on the PPO criterion, not on the pure PO criterion. Usually, this is not clearly stated in the policy analysis. This is problematic since the normative basis of the PO and PPO criterion are substantially different.

Some scholars even argue against the use of other criteria and claim that the PO rule is the only legitimate meaning of efficiency (Lang, 1980, as cited in Griffin, 1995). Similarly, Griffin (1995) asserts that the PO criterion in its pure form is “capable” as a tool of conceptual policy analysis. It is able to incorporate different variable entitlements into the analysis and does not have a status quo bias or bias towards any pre-specified utility level. Unlike other compensation tests, it also allows policy analysts to consider a wide range of efficient actions. Griffin encourages policy analysts to work more with the Pareto optimality rule since it considers more policy options and trade-offs. Sen (1979) also admits that the pure form of PO is normatively more appealing than other Pareto criteria, despite the difficulty in operationalizing PO in the real world.

Several critiques of the efficiency concept have been made by the old institutional school of economics. Economists such as Bromley (1990), Samuels (1992), and Schmid (1987), for instance, have argued that using prices or surplus measures, which result from already existing entitlements, to evaluate new policy proposals creates circularity, since the new entitlements of proposed policies cannot be judged objectively using data from existing policies and entitlements. They argued that both the PO and PPO rules reflect the status quo rights. Bromley (1990) argued that to justify the use of the PO criterion, either the distribution must be seen as a separate, non-economic problem, or it must be assumed that the distribution is optimal at the initial stage. Mishan (1980) asserts that the use of the PPO rule creates ethical problems due to its implicit acceptance of harm to some individuals in return for greater benefits to others.

Moreover, Vatn (2002) argues that efficiency analysis in policy appraisals is strongly normative in two respects: First, efficiency analysis currently accepts the initial distribution of rights as a given. Second, it does not consider the distribution of gains and losses as long as gains outweigh losses, with the underlying assumptions that efficiency and equity are two separate issues and that redistribution is able to deal with the problem of equity. Yet, redistribution is costly and difficult to achieve in the real world. Vatn therefore suggests that an elaboration of whose interests count in policy analyses would be more fruitful than simply adopting the individualistic, welfarist approach, which does not account for socially accepted rights and procedures in a society.

These issues are highly relevant and even more difficult to deal with when crafting environmental policies. This is because environmental goods are usually more complex than other goods and subject to physical interdependencies. Moreover, there are substantial time lags, uncertainties, and novelties within socio-ecological systems. Therefore, the environment poses a significant challenge for both theorists and policy analysts and may cause inconsistencies in the economic models at hand (Vatn, 2002).

For instance, employing the PPO rule would be problematic if time were an important dimension for the environmental problem at hand. The case of global climate change is a perfect example in this case. The costs of abating CO_2 are incurred immediately whereas the benefits will not come for many years. The PPO rule or cost-benefit analysis would discount benefits projected to occur decades later. The optimal abatement level of CO_2 would then be rather low as is the case in many of these analyses (Nordhaus, 1993). Discounting the benefits that would accrue to future generations is a controversial ethical question and considered illegitimate by several scholars (Mishan, 1975; Spash, 1993). On the other hand, this criticism is rejected by scholars who argue that future generations will benefit from man-made physical capital anyway. According to this view, as long as the selected discount rate is in line with future income growth, discounting should not be a problem. However, this argument assumes there is weak substitutability between nature and man-made capital, which is another controversial claim (Özkaynak, Devine, & Rigby, 2004). Vatn (2002) argues further that Kaldor's separation of production and distribution would not hold for environmental problems because of the strong interdependency between efficiency and fairness in environmental issues.

The labor theory of value poses another significant challenge to the concept of Pareto efficiency. Accordingly, the PO criterion utilizes market prices to measure costs and benefits. However, it is not clear which prices should be used to evaluate the costs and benefits of a policy proposal. The theory suggests that the value of goods should be derived from the labor time that is embodied in the production of the goods rather than from market prices (Sills, 1968). Another critique of the use of market prices contends that the economic value of a good should be measured by its natural resource content, or the value that can be derived from the energy embodied in the goods (Burkett, 2006).

The incommensurability of values constitutes another challenge to Pareto efficiency. Efficiency requires the precise measurement of marginal costs and benefits, which in turn requires these to be translated into a single unit of measurement. However, such an approach implicitly assumes the commensurability of values, which had been criticized as far back as the 1920s by Otto Neurath within the socialist calculation debate (Neurath, 1973). Incommensurability is defined as "the absence of a common unit of measurement across plural values, entail[ing] the rejection not just of monetary reductionism but also any physical reductionism (such as eco-energetic valuation, etc.)" (Martinez-Alier, Munda, & O'Neill, 1998, p. 280). Kapp (1983) also raised concerns about the reduction of social costs and benefits into a single monetary unit. He argued:

> Social costs and social benefits have to be considered as extra-market phenomena; they are borne and accrue to society as a whole; they are heterogeneous and cannot be compared quantitatively among themselves and with each other, not even in principle. (p.49)

This argument is even more applicable to environmental costs and benefits, to which markets cannot attribute any economic value since no markets usually exist for them.

2.2 Distribution and fairness

There are two main approaches to fairness or justice in Western philosophy. One is "advantage to all," or the idea that justice requires rules that benefit all parties involved. This view is based on the philosophies of Hobbes and Hume. The other point of view focuses on the principle of impartiality. The impartiality principle advanced by Kant and Rawls implies that rules should apply beyond the self-interest of individuals, but should not necessarily provide an advantage to all parties. Thus, there is a tension between individual advantage and impartiality. However, the two approaches also overlap in the work of different philosophers. For instance, Rawl's (1971) concept of the "veil of ignorance" is said to merge the two views (Barry, 1989). The concept is based on a hypothetical original position, in which individuals do not yet know their exact position in society (i.e., their social status, class, abilities, assets, etc.) and must decide on the principles of justice from behind a "veil of ignorance." Therefore, principles of justice result from this fair agreement or social contract. This initial situation is fair since no one can design principles to favor himself or herself. In that sense, it is in line with both the advantage to all and the impartiality principles.

It is possible to further distinguish between three types of theories regarding justice and fairness in Western philosophy (Vatn, 2002). The first approach is called consequentialist (utilitarian or welfarist). Consequentialist theories investigate the effects of the allocation of goods on individual welfare or utility. Cardinal and ordinal utility theories are two examples of strands within this tradition. They focus on individual advantage, albeit with substantially different perspectives on the admissibility of utility comparisons between individuals.

An important issue here is whether the motivations or intentions behind certain outcomes matter for justice or not. A particular outcome may occur even in the absence of specific intentions to bring it about. Therefore, a second strand, called pure procedural justice theories, focuses more on procedures than on consequences. Rawls differentiates further between perfect and imperfect procedural justice. The first implies that procedures are to be designed such that particular outcomes will be reached with certainty, whereas the latter allows for some uncertainty based on an acceptable level of probability. Some procedural justice theorists, especially those studying environmental risks, underline the importance of participation as an element of procedure. Beck (1992)'s "risk society" is a prime example of this approach (Vatn, 2002).

Another strand of justice theories are those that focus on rights (deontological approach) (Scheffler, 1982; Spash, 1997). Taken to the extreme, these theories imply that, whatever the outcome, rights cannot be violated (Vatn, 2002). However, some have questioned this by considering whether, for instance, a person's right to life can be violated in order to save more lives. Amartya Sen (1987) tried to deal with this tension between rights and consequences by arguing that consequences could be measured along indicators that focus not only on individual welfare but also on the distribution of rights. Procedural and rights-based theories may be based on the principles of individual advantage or impartiality.

Historical developments have contributed to different understandings of justice. Public economics, for example, became more popular in the aftermath of the Second World War. The state was very active in this period and intervened into different issues, including not only market failures such as public goods and imperfect competition, but also normative issues such as welfare, fairness, and redistribution. These policies were mainly based on the Second Fundamental Theorem of Welfare, which states that any Pareto-optimal outcome can be reached as a competitive equilibrium for a certain level of appropriately distributed purchasing power (Arrow, 1963). In the 1970s, however, this era of social-democratic New Deal liberalism more or less came to an end due to the emergence of neoliberalism. The focus of public economics also shifted from fairness and justice to economic efficiency. The redistribution of income and wealth began to be considered ideologically and politically illegitimate (Harvey, 2007; Madra & Adaman, 2010). Moreover, some procedural justice theorists were arguing that any type of taxation is undesirable independent of its outcomes since it always constitutes a violation of ownership rights. In this view, the role of the state should be minimal. Its only functions should be the protection of basic rights and conflict resolution between different actors (Nozick, 1974). Sen (2002)'s rights-based capability approach to justice from a left-liberal perspective was also formulated in this period. Ultimately, the approach of public economics to redistribution, stabilization, and the provision of merit goods in this period was divided along two lines, with one view based on Rawls' justice-as-fairness approach, and the other relying upon proceduralist and constitutionalist theories (Madra & Adaman, 2010).

Historically, Pareto efficiency was the first and most important term used within the mainstream economics literature to refer to distributional issues (Varian, 1975). This baseline concept, however, was insufficient for the analysis of distributive issues. For instance, an allocation in which one agent has everything and others have nothing can also be Pareto efficient. Mainstream economists have thus proposed other concepts to address the problem of distributional justice. For instance, economists defined an allocation as "equitable" if no agent wishes to have the bundle of another agent. That is, when offered the bundle of another agent, no one should be willing to exchange their own bundle in return for this offer. In other words, no one should envy another agent, a situation referred to as "envy-freeness." But this equitable allocation is not necessarily Pareto efficient. If an allocation is both equitable and Pareto efficient, then this allocation was called "fair" (Foley, 1967, as cited in Varian, 1975). This approach had several advantages: It was intuitive, involved symmetry across individuals and ordinal representation of individual preferences, and did not need to address interpersonal comparisons (Chavas, 1994).

This approach was criticized by Nozick (1973) for ignoring the question of production. As a procedural justice theorist, Nozick argued that the question of who has actually contributed to a social product is important for distributional justice. (Varian, 1975) responded to this criticism by extending his model to include production and replacing equity with wealth-equity and fairness with "wealth-fairness". According to this new model, total consumption goods were first divided equally between agents, but the model let the labor differ between agents. Individuals then traded with their equal wealth until market equilibrium was reached. This led to a "wealth-fair" outcome. Similarly, Pazner and Schmeidler (1978) put forth the concept of "income-fair" allocations to deal with the problem of production in a just allocation. Their concept corrected for ability differences between agents. Individuals started with equal shares of labor power, which they then traded until market equilibrium was reached.

A widely-held belief among economists is that there is always a trade-off between efficiency and equality. The main claim is that efforts to reduce inequality often impair economic efficiency (Okun, 1975). However, Sandmo (2000) argues that this need not be always the case, especially for environmental policy. For instance, Pigouvian taxes are an important exception to this rule. When environmental taxes are introduced, economic efficiency can improve without compromising equality if certain complementary measures are taken. In the words of Sandmo (2000, p.20), "the equity-efficiency conflict is not inescapable". Policy instruments set at an optimal level can be analyzed independently of their distributional consequences. This is because the government is assumed to be able to use lump-sum transfers to redistribute income. But

Sandmo cautions that this is difficult to implement in practice since the government does not have perfect information about the tastes and productivities of individual economic agents (Sandmo, 2000).

So far, the arguments and analyses have mainly referred to intra-generational equity. In turning to inter-generational equity, issues of sustainability and discounting enter the discussion. Solow (1991) conceptualized sustainability within mainstream economic theory as follows: Future generations should be provided with an "option or capacity to be as well-off as we are" (p. 181). This reduced the problem of inter-generational equity to a choice between current consumption and saving, which was also the basis for the so-called weak sustainability approach. Assuming perfect substitutability between man-made and other types of capital (e.g., human, natural), total capital stock needs to be kept non-decreasing in order to ensure weak sustainability. Interestingly, several philosophers engaging with the issue of inter-generational equity adopted a position similar to that of Solow. For instance, Rawls (1971) argued in favor of maintaining a fair savings rate to deal with inter-generational equity.

Concerning inter-generational equity, a second important issue relates to discounting. The discounting of future costs and benefits is usually justified on the grounds that people have a positive time preference, which means that individuals attribute more importance to the present than they do to the future. This is usually assumed to describe both consumers' and firms' behavior. However, several economists have questioned whether this is appropriate for public decision-making since it seems to contradict sustainability goals by making almost no considerations beyond one generation (Howarth & Norgaard, 1991). Hence, for instance, Weitzman (1998) argues for the “lowest possible” discounting rate. Chichilnisky (1997) asserts that discounting is not even necessary for inter-temporal efficiency. As an alternative, other scholars have proposed a social rate of discount that is lower than its private counterpart (Brown & Jackson, 1986). Yet, each positive discount rate, however small, implies that costs and benefits are weighted unequally across time.

The arguments in favor of a positive discount rate are as follows: First, generations overlap, meaning that the current generation cares for its children and grandchildren, which implies that the interests of future generations are being taken into account. Second, using a discount rate of zero when the market interest rate is positive means that the current generation is impoverished. Instead, inter-generational equity could be addressed via lump-sum taxation without distorting the prices for the current generation. Third, given the trajectory of economic growth, it would be unwise to redistribute resources in favor of the future generation, which will presumably be richer than the present generations anyway (Hussen, 2004). But economists such as Sen (1982) and Mishan (1988) oppose the use of positive discount rates on the grounds that discounting effectively diminishes the value of costs and benefits that accrue after a long time period to zero.

Solow’s approach, which rests on weak sustainability, has been criticized for reducing the fairness issue merely to a choice of “the proper mix of investment and consumption” (Norton, 2002, p. 37). In contrast, the strong sustainability approach maintains that natural capital is essential for the well-being of future generations and rejects the substitutability of man-made capital and natural capital (Özkaynak et al., 2004). However, since it prescribes nothing more than a positive savings rate and does not require much deviation from the current path of economic growth, weak sustainability is very appealing to policy makers and governments (Norton, 2002).

Discounting is even more controversial within environmental policy debates. Empirical and experimental studies demonstrate that people have different discount rates for different goods (Luckert & Adamowicz, 1993), with environmental goods having perhaps the lowest rate. Decisions made by the current generation may lead to irreversible events (e.g., climate change and species extinction) about which future generations will not have gotten a say. The effects of the decisions of the present generations on future generations are called inter-generational externalities which cannot be corrected even via theoretical Coasean negotiations. Nor can the Pigouvian solution be achieved, since future generations are unable to participate in the market valuation process (Padilla, 2002). Accepting discounting as a means of dealing with inter-temporal distribution therefore involves an inescapable moral judgment (Page, 1977, as cited in Spash, 1993). Summing up, Howarth and Norgaard (1991) argue:

> The question of inter-generational equity is not a technocratic problem of selecting the appropriate rate at which future costs and benefits are to be discounted. It is rather a broader question of identifying the kind of world we wish to leave behind to members of future generations and imposing the social institutions required to bring that future into being. (p. 340)

2.3 Environmental effectiveness

Environmental effectiveness, defined as “the extent to which a policy meets its intended environmental objective or realizes positive environmental outcomes” (Gupta et al., 2007, p. 751), is the ultimate aim of environmental policies. Since human behavior and nature itself are complex and uncertain, it is difficult to ensure the ex-ante effectiveness of environmental policies. Policies may not lead to intended or theoretically predicted outcomes. Therefore, scholars as well as policy makers need to pay careful attention to the interaction between different bases of human motivation, ecological thresholds, irreversibility, imperfect information, information and power asymmetries, and other factors discussed below. If such interactions are overlooked, policies may be counterproductive or even lead to the exact opposite outcome from what was intended.

It is well known that certain instruments are better suited to certain environmental objectives. For instance, CAC measures are most effective at preventing hazards and environmental problems that entail irreversible effects. Once a CAC-type standard is set and enforcement is undertaken effectively, one will be more able to predict the environmental outcome. Yet, monitoring and enforcement are usually difficult and costly. In the case of tradable permits, the cap provides direct control over emissions since it is determined ex-ante as an upper limit. Therefore, if monitoring and enforcement are successfully realized, the environmental outcome of this measure will be relatively certain (Barde, 2000).[7]

Environmental taxes, on the other hand, are ill-suited for controlling toxic and hazardous materials. This is due to the fact that uncertainty and/or imperfect information make it difficult for policy-makers to determine the optimal tax amount. In cases where the tax rate is too low, the exact costs of abatement are unknown to the regulator, tax evasion is likely or linkages between the tax and the pollutants are weak, and the effectiveness of taxes will be questionable (Barde, 2000). In sum, their effectiveness depends on whether they are set at the “right” amount to induce behavioral change (Gupta et al., 2007). Determining this rate is very difficult in practice due to the

[7] As discussed in Chapter 1, several problems in implementation imply that this is usually not the case.

complexity and uncertainty inherent to real world policy contexts. In contrast, a definite ban (as a CAC measure) on toxic and dangerous substances will lead to predictable outcomes, given effective enforcement.

In the case of voluntary measures, the assessment of environmental effectiveness is rather difficult. It is not easy to determine whether the outcome can be attributed to the specific policy or results from other factors. One cannot assess with precision whether the observed outcome is similar to counterfactual state of affairs, or what the business-as-usual scenario would have produced. Since voluntary measures are not subject to sanctions, the environmental performance will also be rather weak (Barde, 2000). Gupta et al. (2007) emphasize that voluntary approaches usually achieve better results when they are applied as part of a broader policy package (policy mix) as opposed to being used as an isolated instrument.

The effectiveness of payments for ecosystem services is controversial as well. Their effectiveness is said to be contingent on “targeting, conditionality, additionality, permanence, leakage, and social sense of fairness” (Porras, Chacón-Cascante, Robalino, & Oosterhuis, 2011, p. 7). Studies investigating their interaction with human motivation and intrinsic pro-environmental behavior is currently very popular in both resource and experimental economists. It is often argued that the material rewards inherent in PES schemes may undermine intrinsic motivation to conserve the resource (Muradian & Cardenas, 2015).

The degree of environmental effectiveness may also depend on the type of pollution as well: For instance, Hahn and Stavins (1992) argue that for pollution with a nonlinear damage function (i.e., for environmental problems with threshold effects) “source-specific standards” may perform better, whereas market-based instruments would be “desirable” for uniformly-mixing global pollutants such as CO_2, given political and technological constraints.

The effectiveness of environmental policy instruments is arguably dependent on their “design, implementation, participation, stringency and compliance” (Gupta et al., 2007, p. 751). Moreover, the information requirements of specific instruments also play an important role. For instance, in the presence of imperfect information about preferences and costs, the environmental performance of taxes may not be predictable, whereas quantity-based instruments may produce more certain environmental outcomes when compliance is ensured. Calculation of the optimum tax level requires excessive information (Cropper & Oates, 1992), while tradable permits require less information. Yet, in the presence of significant transaction costs, this information advantage disappears (Stavins, 1995).

Feasibility is another important factor that affects environmental outcomes. A simple mode of operation, political acceptability, flexibility, and familiarity of the institution may all be seen as parts of the institutional context in which certain instruments are to be applied. Their presence significantly contributes to the effectiveness of policy instruments by facilitating implementation. CAC regulations might be more popular and effective, for instance, in countries without well-functioning markets, while tradable permits require deep markets in order to be effective (Gupta et al., 2007). Policy-makers and firms are usually more familiar with taxes. This might increase the acceptability and therefore the effectiveness of taxes. Taxes might also be opposed by the industry due to their potential negative impacts on international competitiveness. Tradable permit schemes, on the other hand, entail complex rules and require sophisticated monitoring mechanisms (Barde, 2000). Yet, they are more flexible in terms of adjustment to inflation and economic growth. Taxes, meanwhile, require an iterative trial-and-error mechanism until the "right" tax rate is achieved. When inflation is present, this requirement becomes even more crucial. Moreover, the political acceptability of an instrument also depends on the distribution of costs and benefits as well as on the sense of fairness among different groups in society. In practice, the effectiveness of incentive-based instruments may be difficult to evaluate as noted by Barde (1999), since

- they are usually not implemented in isolation, but in combination with other instruments;
- data required for an ex-post evaluation is often lacking;
- instruments based on economic incentives may serve different targets at the same time (e.g., environmental taxes may be used both to provide incentives for behavioral change and to raise revenue); and
- a culture of policy evaluation might be lacking in a country, and/or evaluation may require different governmental bodies to cooperate with one another.

2.4 Behavioral factors

Behavior is another important factor that influences the outcomes of environmental policies; it is especially crucial for environmental effectiveness (Frey & Stutzer, 2008; Rode et al., 2015; Vatn, 2009b). Yet, traditional policy evaluation studies usually leave out behavioral considerations, even though there is a considerable number of experimental studies demonstrating that public policies may perform poorly if these are not taken into account. More recently, studies conducted by the World Bank and OECD started to pay a great deal of attention to behavioral impacts of public policies (Lunn, 2013; The World Bank, 2015).

A review of experimental findings since the 1970s implies that policies based on economic incentives, such as fines, taxes, subsidies, and other rewards, may crowd-out the voluntary behavior of resource users, as discussed in Chapter 3 and 4 of this dissertation. If this phenomenon is not taken into account, policies may lead to unintended consequences (Frey & Stutzer, 2008; Gneezy et al., 2011). This may occur, for instance, if a policy is perceived to compromise individual control (control-aversion), conveys a bad message about the intentions of the regulator, or leads to moral disengagement (Bowles, 2008). On the other hand, some policies based on economic incentives may crowd-in voluntary behavior if they are perceived to support or lead to moral engagement (Rode et al., 2015). In other words, policies based on economic incentives may complement and reinforce pro-environmental behavior in certain cases, while discouraging it in other cases (Bowles & Polanía-Reyes, 2012). Hence, empirical evidence suggests that it is important to understand the interaction between voluntary behavior and different policy interventions. Policies may be detrimental to environmental outcomes if this interaction is not considered prior to implementation.

CHAPTER 3

CHALLENGES TO AND NEW PERSPECTIVES ON ENVIRONMENTAL POLICY INSTRUMENTS

Chapter 1 and 2 provided an overview of the standard neoclassical approach to environmental policy instruments by focusing on different types of instruments and evaluation criteria. There are, however, various challenges to this approach. Public economics, for instance, is criticized for neglecting pro-social behavior, assuming universally opportunistic human behavior, offering one-size-fits-all solutions, and failing to consider performativity issues (e.g. Bowles & Polanía-Reyes, 2012; Callon, 2007; Madra & Adaman, 2010; Ostrom, 1990). Meanwhile, the social psychology, institutional economics, ecological economics, and science philosophy literatures offer new perspectives on environmental policies and further our understanding of human behavior and environmental policy-making.

The purpose of this chapter is, therefore, twofold. The chapter will first discuss several challenges and criticisms directed at the neoclassical approach to environmental policy instruments. Next, it will introduce new approaches to environmental policy-making, including the role of institutions, pro-social behavior, and community-based management relying on Elinor Ostrom’s framework. Overall, the chapter underlines that a mechanistic, technocratic approach to environmental policy-making alone cannot address environmental problems effectively. Instead, other-regarding behavior and local participatory mechanisms manifest themselves as potentially important aspects to be taken into account to better deal with environmental problems.

3.1 Criticisms towards the neoclassical approach to environmental policy instruments

3.1.1 Opportunistic behavior

Public economics, mainly inspired by Arthur Pigou's work, emerged as a sub-discipline of economics in the 1920s. It later became a well-established strand of the economics literature with the contributions of other scholars such as Richard Musgrave, Ronald Coase, Paul Samuelson, William Baumol, and James Buchanan. Public economics mainly relied on the assumption of opportunistic behavior, and on the general idea that price mechanisms have to be supplemented, and if necessary, limited by governments in order to achieve economic efficiency and a well-working society (Musgrave, 1986).[8] In the aftermath of WWII, public economics mainly focused on market failure, welfare issues and redistribution, but from the 1970s onwards and in parallel to the rise of neoliberalism, its focus shifted from welfare and redistribution to efficiency and government failure. In this new neoliberal era, a language based on economic incentives emerged as the dominant approach to analyzing economic actors and activities. Information failure was also seen as an important problem to deal with. In this era, the focus on efficiency led to a strong preference in favor of marketization policies such as privatization, financial liberalization, and deregulation—in both scientific and policy circles. The shift in focus from Pigouvian taxes to marketable permits was a prominent example of this transformation in policy-making (Madra & Adaman, 2010). The logic of economic incentives was strongly expanding to include the social aspects of life as well. The citizen-subject of the welfare state in the post-war era was being replaced by

[8] This is in stark contrast to the Austrian school's approach of spontaneous order, and the Chicago school's belief in markets as selection mechanisms (Madra & Adaman, 2010).

the economic subject of the neoliberal state, or *homo economicus*—a rational, opportunistic individual who responds to the changes in its environment in a systematic way (Foucault 2008 [1978/79], as cited in Madra and Adaman, 2010).[9]

Today, public economics is still based on the assumption that individuals are opportunistic. The assumption also applies to policy-makers and governments. In fact, so-called "government failure" is one of the central problems of public economics. Consequently, public economics deals with the manipulative and opportunistic behavior of citizens and bureaucrats by focusing on incentive-compatible mechanisms and manipulation-proof institutions (for tax collection, direct regulation, etc.). Individual preferences are considered exogenous and stable in these analyses. It is taken for granted that individuals are able and willing to manipulate social aggregation processes. Nevertheless, this approach has been challenged by different scholars. A rather neglected book by Adam Smith addressed the issue of moral sentiments in the 1750s. Recently, behavioral and experimental economists underline the role of pro-social behavior (e.g. Frey & Stutzer, 2008; Gneezy et al., 2011). Institutional and ecological economists argue that the adoption of opportunistic or other-regarding behavior in real-life social dilemmas depends on institutional context (Sagoff, 2008; Söderbaum, 1990; van den Bergh et al., 2000; Vatn, 2005). If economic models fail to incorporate this conditionality, policies may be doomed to fail. Moreover, the focus on opportunistic behavior may prevent policy-makers from considering participatory decision-making models. Madra and Adaman (2010), for instance, argue that due to the assumption that

---

[9] This view, as advanced by the Chicago School, did not imply that *homo economicus* is a real life phenomenon. Rather, it was used as an approximation or a device for theoretical modelling, policy analysis, and institutional reform (Madra and Adaman, 2010).

people are opportunistic, public economics traditionally ignored alternative solutions based on "public discussions and interactive social processes […] and social cooperation through persuasion, compromise and commitment" (p. 1094). One reason for the emergence and popularity of carbon markets may be this focus on opportunistic human behavior. So far, environmental policies geared to address climate change mitigation have by and large ignored deliberation and participation (Spash, 2010).

3.1.2 Externalities

The neoclassical conception of externalities is another source of criticism. Although this has no direct relevance for our empirical purposes in the present study, a brief discussion of why this is considered problematic is provided in this section. Externalities play a major role in both public economics and environmental economics. They are often seen as unintended by-products of economic activities (Baumol & Oates, 1988; Mishan, 1971). However, several scholars criticize a theory based on such conception of externalities. Vatn and Bromley (1997) argue, for instance, that the theory of externalities is inconsistent due to the interdependencies between what is considered internal and what is considered external to the model. Consequently, in public economics, when a polluter emits a pollutant, an externality only exists if a victim is physically present in the area affected by the pollution. In other words, causing the emission may not always coincide with causing the externality. This physical interdependency is problematic for the standard models of public economics, since they are based on independent economic actors and methodological individualism (Vatn & Bromley, 1997). Moreover, modeling externalities as unintended by-products of economic activities may be confusing because according to the standard model, rational choices by self-interested agents cannot be unintended. Several scholars, therefore, argue that externalities instead should be seen as deliberate and intended "cost-shifting" actions by rational actors (Kapp, 1950; Muradian & Martinez-Alier, 2001). Arguably, if costs can be shifted to other actors without violating any formally enforceable rules or laws, then it is rational for self-interested actors to do so, in order to minimize the costs of economic activities. Indeed, within the market model, economic agents look for ways to shift costs to "external" parties. More specifically, (Kapp, 1950) argues:

> Hence, a system of decision-making operating in accordance with the principle of investment for profit cannot be expected to proceed in any way other than by trying to reduce its costs whenever possible and by ignoring those losses that can be shifted to third persons or to society at large. (p. xiii)

Accordingly, the presence of externalities should be seen a rational outcome of market processes instead of a "failure" of the market (Vatn & Bromley, 1997). Therefore, strictly focusing on the internalization of externalities via pricing mechanisms may not perform well in terms of environmental outcomes. Arguably, any policy proposal should consider other factors such as moral commitments, social norms, collective decisions on standards, network processes, and moral suasion.

3.1.3 Pseudo-markets and shadow pricing

Another problematic area concerns environmental markets and shadow prices. The neoclassical view on environmental policy instruments implies that the underlying reason of environmental problems is lack of markets (Arrow, 1984, as cited in O'Neill, 2001). Two potential solutions are put forth to deal with their absence: to either i) precisely define property rights about environmental goods so that they can be traded in markets, or ii) extract individuals' willingness to pay or to accept damages in the form of shadow prices[10], which can then be used in cost-benefit analyses (CBA) to evaluate the welfare gains or losses of specific policies (O'Neill, 2001, 2012). The "old" school of institutional economics has criticized both of these approaches. Regarding the latter, (Sagoff, 2008), for instance, has argued that the shadow pricing approach confuses public judgments and private preferences about environmental goods, for it considers moral and aesthetic values as private consumer preferences, does not differentiate between people's values and consumption opportunities, and conflates market processes with political institutions. He asserts that decision-making on environmental issues should belong to the political realm, and that it cannot be subject to market-mimicking approaches such as shadow pricing.[11]

Conversely, the introduction of pseudo-markets for non-market goods commodifies environmental assets and opens the environmental sphere to market norms. Yet this may be problematic, because it is questionable whether unregulated markets can produce outcomes that are socially and ecologically benign (O'Neill, 2012). Market skepticism with respect to environmental problems has a long tradition; the work of scholars such as Karl Polanyi Polanyi (1957), Otto Neurath (1945), William Kapp (1950), Joan Martinez-Alier (1987) and John O'Neill (2001) are important in this respect, although the extent of and main reasoning behind their criticism varies. For instance, while Karl Polanyi did not oppose markets per se, he expressed concerns about disembedding markets from social and environmental constraints; Neurath was against using a single monetary unit in economic decision-making and proposed other types of

[10] These are the prices that would have been paid had there been a market for these goods.
[11] Several scholars from the neoclassical tradition also reject this approach based on ideas from public choice theory (e.g. Barnett & Yandle, 2009; Zerbe & McCurdy, 1999).

measures based on physical, social and environmental aspects of welfare. Meanwhile, Kapp's work merged these two traditions (O'Neill, 2012). Overall, extending markets to environmental problems was the source of the problem according to these scholars; not the solution. In particular, they were critical of expanding markets to include previously unrelated items (such as the human body, biodiversity), as well as the expansion of market relations, norms and attitudes to other spheres of life, such as social life (e.g. universities treating their students as consumers) and nature (e.g. decision-making on nature via cost-benefit analyses). The field study in Part II of this dissertation addresses some specific issues related with environmental markets and market reasoning in an experimental setting.

3.1.4 Commensurability

Another criticism of the neoclassical approach to environmental policies addresses the (in)commensurability of different values. Otto Neurath was one of the early and most influential writers to argue that some elements in the economy may not be commensurable. William Kapp also argued against the commensurability of values, and did this within the realm of environmental policies:

> To place a monetary value on and apply a discount rate (which?) to future utilities or disutilities in order to express their present capitalized value may give us a precise monetary calculation, but it does not get us out of the dilemma of a choice and the fact that we take a risk with human health and survival. For this reason, I am inclined to consider the attempt at measuring social costs and social benefits simply in terms of monetary or market values as doomed to failure. Social costs and social benefits have to be considered as extra-market phenomena; they are borne and accrue to society as a whole; they are heterogeneous and cannot be compared quantitatively among themselves and with each other, not even in principle. (Kapp, 1983, p. 49, as cited in Martinez-Alier et al., 1998)

The incommensurability of values implies a rejection of any type of reductionism, be it monetary or bio-physical; however, it does not imply incomparability. Rather, it acknowledges the complexity of the world and the plurality of values (Martinez-Alier et al., 1998; Özkaynak et al., 2004).

Evidence in the form of protest bids—usually encountered in contingent valuation (CV) studies for environmental assets—may be seen as a case in point for the incommensurability of values as discussed below. CV surveys aim to measure the value individuals privately ascribe to environmental amenities by asking them how much they would be willing to pay for an environmental improvement or to accept an environmental damage. In these studies, often some participants refuse to put a price on improvements or damages. Similar to how one would be unwilling to put a price on friendship, love, or the lives of family members, a not negligible fraction of CV participants are opposed to putting a price on nature. They argue that putting a price tag on nature is unacceptable since accepting a price for such social relations would imply betrayal and corruption. Moreover, if an environmental asset is acquired from previous generations (rather than bought from another individual, for instance), some CV participants question its alienability and tradability in terms of property rights. Who has the right to the environmental good at hand, and who has to pay for or accept damage to it are apparently controversial issues. Environmental economists who conduct CV surveys usually exclude these protest bids from their analyses. Yet this is problematic, because protest bids imply both a refusal of commensurability, and the extension of market norms to the environmental sphere. Moreover, protest bids can also be seen as refusal of cost-benefit analyses in general. As discussed in Chapter 2, CBA relies on the potential Pareto improvement criterion, and as such, implies that a policy will be implemented as long as the winners can compensate the losers of the policy. Compensation may or may not take place; the idea is rather that the winners' gains exceed the damage suffered by the losers. This is another area of contestation (O'Neill, 2001; Spash, 2008).

Arguably, instead of assuming commensurability between conflicting values, one should see the environment as "a site of conflict between competing values and interests, and institutions and communities that articulate those values and interests." (O'Neill, 2001, p. 1868). Accordingly, rational decision-making does not necessarily require reducing all conflicting values into a single measure of welfare; rather, it necessitates deliberation. CV and CBA studies are also criticized for being "reason-blind." That is, preferences and their intensity count in the analysis, but not their reasons. Preferences for societal values, esthetics, science et cetera are not distinguished from preferences regarding private goods; meaning, policies based on economic valuation and CBA do not rely on reasoning or debate. This is also revealed by the finding that the more CV participants are provided with information about the environmental good at hand, the greater becomes their monetary valuation. Yet this is problematic, because each response is then a function of how much information is provided to the participant, and the process of economic valuation actually turns into a process of education—with the remaining question of how much information to provide to extract "actual" preferences. Moreover, since CBA studies are reason-blind, market interactions for environmental assets are reason-blind as well. Accordingly, the market may be "in principle unprincipled" (Hirsch, 1977, as cited in O'Neill, 2001). That is, market choices and decisions are made without dialogue. Markets can ensure coordination only because certain actors or goods enter or exit a market, not because there is some type of reasonable dialogue. Therefore, subjecting the nature and the commons indiscriminately to the economic logic of markets may lead to “unwarranted consequences” (O’Neill, 2001, p. 1098). Consequently, applying market logic to the uncertain and complex problems of environmental degradation may lead to different outcomes than would a collective, deliberative mechanism of negotiation (Lohmann, 2010; Schellnhuber, Cramer, Nakicenovic, Wigley, & Yohe, 2006; Vatn, 2002).

3.1.5 Performativity

The issue of performativity poses another challenge for public economics; especially in terms of environmental policies. Some scholars note that the concept of *homo economicus* can be actually “performative” instead of descriptive; meaning, policies or incentive structures based on the assumption that *homo economicus* will engage in opportunistic behavior may actually elicit said behavior (Callon, 2007; Madra & Adaman, 2010). If this is the case, then environmental policies—which take self-interested opportunistic behavior as a given—may in turn generate such behavior, instead of diminishing it.

The concept of performativity has its origins in philosophy (Austin, 1962), but has been adopted by a diverse set of social science studies; including feminist theory (Butler, 1990) and science philosophy (Beck, Giddens, & Lash, 1994). Callon (1998) was one of the first scholars to think and write about the performativity of economics. Callon (1998) argued that “economics […] performs, shapes and formats the economy, rather than observing how it functions” (p.2). However, one must note that in this formulation, “economics” refers not only to the academic discipline of economics, but also includes economic actors, instruments, practices, procedures, technologies, and so forth. Neither should performativity be reduced to the power of language or to “doing things with words” as it was first proposed by Austin (1962). It is clearly more complex and multifaceted than that (MacKenzie, Muniesa, & Siu, 2007).

Analyzing the role of theoretical economic models in the economic activities of real life, Callon disagrees that economists solely describe and analyze economic activities. Rather, he argues that causality may sometimes go the other way round; namely, from economic models to real-life economic behavior. He underscores that economists are important actors who shape economies, and as such, cannot be seen as separate, isolated actors. In brief, economists do not simply observe the economy, but also reconstruct it; in other words, they perform it. However, Callon prefers the term “co-performation” instead of “performativity” since the former implies that different actors, institutions, and practices in the realm of economics jointly shape the economy we end up with. There is always conflict and competition between different approaches to the economy, and co-performation points out the interaction between these heterogeneous approaches and actors.

Co-performation can be observed, for instance, in the preference for carbon markets in environmental policy. When the initial proposal for carbon markets was put forth in the 1970s, a competing view emphasized the importance of carbon taxes instead. Despite the presence of these two opposing approaches, carbon markets became a popular environmental policy instrument and environmental taxes did not. Analyzing this process, Callon argues that the implementation of carbon markets was enabled not only by economists, but also by lobby groups, different technologies regarding the measurement of carbon molecules and new data collection techniques; and so this was an incidence of co-performation by economists, economic actors, and technologies in the carbon market.

Another example in environmental policy where performativity seems to be at work is the case of individual transferable quotas (ITQs) in fishing. Based on the works of Gordon (1954) and Scott (1955), economists argued that assigning fishers the total allowable catch in smaller units—quotas—and letting them trade these quotas would combat problems of overcapacity and overfishing (Squires, Kirkley, & Tisdell, 1995). This would be possible because fishers would then not race for fish as they do under open access regimes, and allowing them to trade quotas would improve efficiency in the sense that inefficient fishers would sell their quotas to more efficient fishers and exit the fishery. Obviously, this approach transforms fishers into capital owners and the fish into private property. In other words, the fisheries problem becomes an investment problem that involves rational and efficient actors, and predictable outcomes. This model is actually implemented in several fisheries around the world, including Norway, New Zealand, and Iceland. It became possible to implement ITQs when fish—once a common good for all fishers—were deemed property and turned into a commodity to be traded in markets in line with the fisheries models of resource economists.

Holm (2007) argues that this may be seen as an instance of economics "performing" the economy in the sense that real life phenomena were shaped to fit economic models. In other words, economics did not only observe and describe fisheries, but also performed a certain economic model by restructuring and shaping their context. Again, however, this was due not only to the academic discipline of economics; other factors were also essential in the enactment of ITQs. Holm (2007) underlines that economists' theoretical models would not have been implemented in real life if it were not for the organizational and institutional preparations that took place prior to the implementation of ITQs. Hence, fish that were once untraceable, slippery, and difficult to track, turned into a “cyborg fish” that is now easy to track, identify, and control. Statisticians, technicians, politicians, and many other professionals contributed to this process, without which the economic model would not work at all. Accordingly, fishers became self-interested calculative individuals in response to these changes. Holm calls this process of slow and deliberate preparations the “invisible revolution”. He does not agree that the ITQ system, as a theoretical model, was imposed on the fisheries from outside in a top-down manner by economists. Rather, he argues that the very actors, namely the fishers and even the fish itself, were transformed beforehand in such a way that they were ready for the implementation of the market in fishing quotas. This transformation was enabled by the new sampling, data collection, documentation, and measurement methods that were introduced to make the resource and resource users fit the new model of resource management; namely, ITQs. The fishers were expected to adopt a certain kind of behavior—opportunistic behavior—and being dispossessed of what was previously their resource, some of them indeed became a *homo economicus*, a self-interested capital manager, while most others were disconnected from their local livelihood and way of life. Callon (2007) asserts that without the performativity of economics, the ITQ market would not have come to existence. Once the market existed, fishers’ calculative and self-interested behavior became visible. “The individualization of the agency” became evident; that is, fishers who maximized their own payoff were performed by economics.

3.2 New approaches to environmental policy instruments

Building on these criticisms and other empirical evidence, alternative approaches have been developed to address environmental problems. The role of institutions, pro-social or other-regarding behavior, and community-based management are among those alternative perspectives, as discussed below in the second part of this chapter.

3.2.1 The role of institutions

It is widely agreed that institutions are important factors that influence human behavior. This statement is generally accepted both within the methodologically individualist, and the social constructivist traditions of social sciences. However, these two literatures approach the role of institutions in two distinct ways. The methodologically individualist model (neoclassical economics) relies on the assumption of calculative, self-interested individuals, and assumes further that these individuals operate and choose on the basis of external constraints. The other fundamental assumptions of this model are maximizing behavior and stable preferences, and in here the equilibrium approach dominates analyses (Becker, 1976; Eggertsson, 1990, as cited in Vatn, 2005). Institutional aspects such as property rights and markets are also considered important structures; however, individual preferences and type of rationality used are assumed to be independent of institutional context. Due to the assumptions of full information and zero transaction costs, institutions cannot play a role in providing meaning to different alternatives and the communication of needs. Theorizing within the same paradigm, new institutional economists such as North (1990) and Williamson (2000)—agreeing on the importance of positive information and transaction costs—argue that modern institutions are important elements of economic life that reduce these costs and uncertainty. Therefore, they define institutions as rules of the game or constraints on individual behavior (North, 1990). Consequently, institutions are additional constraints on standard individual maximizing behavior.

The social constructivist perspective conceptualizes the role of institutions in a clearly different manner. Relying on the “social man” instead of the “economic man,” the social constructivist approach conceives the individual as a socially constructed creature who makes choices based on internalized norms, rules, and values. Individuals behave according to what is considered appropriate in different choice situations. Social processes shape and constrain individual behavior. According to this view, therefore, institutions are not merely constraints on behavior. Rather, they “constitute the individual and create meaning” (Vatn, 2005, p. 204). Classical institutional economics, mainly relying on Veblen’s and Common’s work, is based on a similar understanding. Accordingly, individual behavior is created and shaped socially.

There are two main traditions within the social constructivist perspective: the cognitive and the normative position. The cognitive position underlines that the world is too complicated to understand fully, and therefore individuals need “typifications” about natural and social phenomena. These collectively-produced typifications or concepts of common knowledge—language and other cognitive models, for instance—are used to make sense of the world and to be able to cooperate. Institutions are created when there is a common acceptance of these typifications. The typifications are then objectified; that is, they exist independent of who created them. In other words, they are taken for granted and seen as given, natural objects used to make sense of the complex phenomena in the world. In contrast to the methodological individualist perspective, the cognitive position does not rely on the assumption that individuals are perfectly able to understand their own needs and all relations about the social and the natural world. It rather posits that institutions are social constructs that enable people to act. This is also in line with the classical institutional economics' approach to institutions. Moreover, institutions give meaning to the actions of individuals.

In a similar vein, the normative position underlines the importance of common values and obligations for human actions. Accordingly, normatively appropriate behavior is signaled by social norms and expectations about roles in society. Social norms define what is seen as appropriate and acceptable. Individual behavior also rests on reasoning, but not so often on calculated expected returns from alternative behaviors. Hence, both the cognitive and the normative positions highlight that norms and conventions offer individuals help in complex behavioral situations. In sum, the cognitive position asserts that institutions help to distinguish between appropriate and inappropriate behavior, whereas the normative position emphasizes the prescriptive dimension of institutions (Vatn, 2005).

Vatn (2005) suggests merging the methodological individualist and social constructivist perspectives in order to more thoroughly conceptualize the role of institutions within the realm of environmental policy. Based on the cognitive position, one can argue that the presence of particular institutions may lead to the use of particular types of rationalities. For instance, selfish calculative behavior may be elicited by one institutional structure, whereas other-regarding behavior may be elicited by another. Vatn's (2005) approach therefore underlines that individual calculative behavior may not be the only type of rationality that individuals use to arrive at a choice; instead "the logic of the situation is institutionally formed" (p. 208). It may be appropriate to use calculative logic in one type of situation, but it may be "impossible, irrelevant or embarrassing" (Vatn 2005, p.208) to adopt it in another institutional context. Of course, one also needs to consider bounded rationality; that is, one cannot be fully sure that humans have the cognitive capacity to maximize utility without the help of collectively produced cognitive models, even if individuals are using a calculative self-interested logic. These cognitive structures are needed for communication and coordination. Moreover, some institutions may actually evoke and support calculative rationality. Therefore, the performativity of institutions is not to be underestimated. Money, double-entry book-keeping, markets—and most relevant for environmental policy—CV surveys for environmental goods, are some of the institutions Vatn offers as examples that simplify complex cognitive processes and foster calculative behavior. However, normative aspects are also important in each of these institutions; for instance, markets function well if the norms of "good conduct" are strong enough to overcome fraud and opportunism. Vatn's synthesis implies that preferences are to be understood as a combination of individual tastes and socially constructed typifications.

Turning to environmental issues and environmental policy-making, Vatn notes that it is a "category mistake" to evoke the logic of individual maximization and markets for environmental common goods. This is because the relevant issue for these goods is the consideration of "we" instead of "I". If the dominant understanding is that there is only one type of rationality—as is common in public and environmental economics—namely, that of individual calculative behavior, then marketization in environmental issues will naturally introduce rationality to the realm of the environment. However, this traditional approach to environmental issues entails three problems:

First, in neoclassical economics, preference formation is assumed to be independent of the social context. However, the social constructivist perspective accepts that preferences are formed by the social sphere. Obviously, there are also physical needs like food and shelter that lead to certain preferences; nevertheless, how one satisfies these needs is again contingent on the social sphere of life. Another problem related to environmental goods is that people usually have neither any experience nor any cognitive models to understand and value some of these environmental goods, because the environment is subject to rapid change. For example, the climate change problem is a new concept or typification that humanity has been encountering only for the last few decades. One cannot expect individuals to have complete preferences about environmental issues that are so recent. Vatn therefore points to the recent literature on endogenous preferences, which posits that institutions are structures that play a significant role in the formation of norms, preferences, and human behavior (Bowles, 1998; Hodgson, 1998).

The social context and norms are important factors especially in cases where the preferences of an individual affect the possibilities of others. This is an important element in environmental problems; particularly in the case of environmental common goods. Moreover, normative issues play a significant role in environmental problems. These normative concerns may be perceived as issues related to being a citizen, rather than a consumer. Obviously, what is considered a consumer issue versus a citizen issue is itself, socially constructed. In other words, this may also evolve over time if one assumes that preferences are not stable over the long-term. Unstable preferences imply that individuals may be eager to discuss and deliberate on their preferences, rather than merely expressing them (in the market, or in CV studies). Individuals may be unwilling to accept a trade-off between the normative aspects of life and consumption possibilities, if they can switch between different sets of preferences from time to time; between normative and calculative logic, for instance.

Second, if we are to take the normative and ethical aspects of environmental issues seriously, we need to consider the incommensurability of values. Reducing different values to a single unit of measurement may be impossible, and individual calculative logic may often fail when it comes to environmental conflicts. Consequently, a communicative rationality needs to be adopted rather than a calculative one (Habermas, 1984). Instead of imposing a certain type of institution based on a certain type of rationality (for instance, market institutions based on calculative logic), participating in and deliberating on which institutional structures are appropriate a particular environmental problem will be more fruitful.

Third, and most relevant for our purposes in this dissertation, institutions or policy instruments may change both the material incentives and the logic or rationality adopted when making a choice. Hence, in addition to individual calculative logic, normative considerations, reciprocity, trust, and a desire for social approval also play important roles, as confirmed by the recent literature on experimental and behavioral economics (Bowles & Polanía-Reyes, 2012; Bowles, 2008; Gintis, 2000; Fehr & Falk, 2002; Ostrom, 2000). Policy instruments that involve economic incentives may influence the intrinsic motivations of individuals. If policy instruments assume strategic opportunistic behavior, this may signal to individuals that this type of rationality is appropriate in a particular context. Consequently, policy instruments that aim to mitigate opportunistic behavior may themselves create such behavior. In other words, certain kinds of policy instruments may undermine other-regarding motives or the "citizen" logic; namely, trust, reciprocity, a focus on common responsibilities, and moral obligations, which are in fact all very crucial in terms of environmental problems. In such cases, policies that intend to achieve improved environmental performance may actually backfire and lead to opposite, counterproductive outcomes. This is explained in detail in the next section.

In a similar vein, (Sagoff, 2008) distinguishes between citizen and consumer preferences, and argues that this distinction has a long history in public economics. Sagoff refers to Gerhard Colm's arguments to explain this claim:

> He [Gerhard Colm] holds that the individual voter dealing with political issues has a frame of reference quite distinct from that which underlies his allocation of income as a consumer. In the latter situation the voter acts as a private individual determined by self-interest and deals with his personal wants; in the former, he acts as a political being guided by his image of a good society. The two, Colm holds, are different things. (Musgrave, 1959, as cited in Sagoff, 2008, pp. 48-49)

Another excerpt in Sagoff's work from Stephen Marglin puts forth similar arguments:

> The preferences that govern one's unilateral market actions no longer govern his actions when the form of reference is shifted from the market to the political arena. The Economic Man and the Citizen are for all intents and purposes two different individuals. It is not a question, therefore, of rejecting individual […] preference maps; it is, rather, that market and political preference maps are inconsistent. (Marglin, 1963, as cited in Sagoff, 2008, p. 49)

Hence, in contrast to altruistic and moral motives, individuals may have conflicting interests when it comes to different economic motives (Milgrom, 1993). Sagoff argues that each of these conflicting preferences is expressed in contexts appropriate to each. These contexts may range from family situations to the market, or the political and professional realms. In certain contexts, the role of the citizen or community member is invoked, and the individual behaves according to his/her judgments about what the group or the community should do, as opposed to what his/her own interests would dictate him/her to do. Therefore, Sagoff posits that citizen judgments should not be confused with personal preferences; that is, there is big difference between what "we" should do as a community, and what "I" prefer as an individual.

3.2.2 The role of pro-social behavior

There is much anecdotal and experimental evidence that implies public policies may backfire when they address the behavior of solely self-regarding individuals (Bowles, 2008; Frey & Stutzer, 2008; Gneezy et al., 2011; Gneezy & Rustichini, 2000). At least since Titmuss' blood donation example, several scholars argue that economic incentives may undermine intrinsic motivations, leading to the crowding-out of pro-social behavior (Gneezy et al., 2011; Titmuss, 1971). Therefore, material incentives as well as moral considerations need to be taken into account when designing public policies, especially in realms where normative and moral behavior is prevalent and much needed. Environmental policy-making is obviously an important case in this respect.

The motivation crowding idea was very controversial when it was first proposed, and several scholars argued against it (e.g. Solow 1971; Arrow 1972, as cited in Bowles & Polanía-Reyes, 2012). However, subsequent research on social preferences and incomplete contracts cleared the way for a change in how economic incentives are analyzed in standard economic models (Bowles & Polanía-Reyes, 2012; Gneezy & Rustichini, 2000). Modifying the assumptions of standard models, experimental studies led to an extensive amount of research on issues such as trust, reciprocity, fairness, et cetera, often summarized as "pro-social" or "other-regarding" behavior. According to the motivation crowding theory, the effects of incentives need to be analyzed in two parts: First, economic incentives have a direct (relative) price effect; meaning, they make the activity at hand relatively more or less expensive to pursue, and as such, alter the behavior towards the predetermined policy goal. Second, if individuals are intrinsically motivated enough prior to the imposition of an economic incentive, a crowding-out or crowding-in effect will result (Frey & Oberhozer-Gee, 1997; Frey, 1997; Gneezy et al., 2011).

Although the standard economic theory makes no claims that other-regarding behavior and moral motivations are non-existent, it still assumes that there is no interplay between economic incentives and intrinsic motivations. That is, it posits that economic incentives and moral motivations are additive, rather than interdependent (Bowles, 2008). Recently, this assertion was largely refuted by theoretical and empirical studies (see, for instance, Bowles & Polanía-Reyes, 2012; Cardenas, Stranlund, & Willis, 2000; Cardenas, 2000). Moreover, the assumption of complete contracts in the standard economic theory does not consider any spillover effects between agents. Accordingly, the external effects of an economic activity on third parties can be corrected when economic actors take all benefits and costs into account. This becomes possible when clear property rights dictate who will be held liable for relevant external effects. This way, self-interested individuals are made to care about the effects they impose on others as if they were moral individuals. Therefore, if contracts are complete, "morality has no application to market interaction under the conditions of perfect competition" (Gauthier, 1986, p. 93, as cited in Bowles, 2008). This is also the general approach to market failure that involves negative externalities in much of the environmental economics literature.

However, as Bowles (2008) argues, contracts are often incomplete: There are information asymmetries between principals and agents; that is, the quantity and/or quality of goods and services is often unknown to at least one of the parties, or they may be known, but may not be enforceable. Labor markets, credit markets, and environmental problems are all subject to the problem of incomplete contracts, which standard economic theory has developed mechanisms to overcome. Mechanism design theory is a recent and popular subject in public economics that deals with this problem, but it ignores moral and other-regarding motivations. Civic virtues are left out of the discussion, despite the fact that they are often used in real life to overcome the very problems that incomplete contracts create. Such values are evidenced, for instance, in tax paying behavior and employer-employee relations (e.g. Andreoni et al., 1998; Bewley, 1999, as cited in Bowles, 2008). In real life, market failure is usually addressed by interventions consisting of a combination of economic incentives and intrinsic motivations.

In their review of 41 economic experiments, Bowles and Polanía-Reyes (2012) posited that economic incentives and moral motivations cannot be seen as separable issues; they claimed that the two may complement each other in some cases, and act as substitutes in others. That is, economic incentives may crowd-in moral considerations (they may reinforce the effect of economic incentives) or crowd-out civic virtues (they may undermine moral motivations). Accordingly, potential interactions between economic incentives and intrinsic motivations (i.e., non-separability) may be related to four factors: i) framing, ii) information content of incentives, iii) self-determination, and iv) endogenous preferences.

Framing implies that incentive structures may signal the appropriate behavior in different contexts. Therefore, one effect of economic incentives may be that they offer hints about the appropriate behavior in different settings, meaning they “frame” the choice situation. As a result, frames may have a significant influence on human behavior. Experimental evidence also points to this so-called framing effect. For instance, using market-like language (calling the interaction an "exchange"; calling the game a "Wall Street Game") as opposed to adopting neutral language leads to less cooperation (Dufwenberg, Gächter, & Hennig-Schmidt, 2011; Hoffman & Spritzer, 1985; Liberman, Samuels, & Ross, 2004). The presence of market-like competition may lead to behavior which had previously been unjustified (Schotter, Weiss, & Zapater, 1996). Frames may give information about the anonymity of the choice situation. For instance, in economic experiments where the room was darker or where the respondents wore dark glasses, respondents had a sense of greater anonymity which made them less generous (Zhong & Loewenstein, 2007). Bowles and Polania-Reyes (2012) argue that thinking about the differing levels of anonymity in the different realms of social life—family, business, community, and markets, for instance—may offer a hint about the relevance of framing effects. Similarly, Fiske (1992) distinguishes between four types of psychological models that individuals adopt when deciding: the authoritarian, communal, egalitarian, and market models. Each one implies a particular type of behavior that is acceptable or appropriate, and therefore elicits a particular behavior. Some of the framing effects related to markets is explained by the "moral disengagement" argument in social psychology (Bandura, 1991). This implies that market-like incentives may switch people's moral motivations on and off (Shu, Gino, & Bazerman, 2011). In such cases, economic incentives provide information about permissible behavior, leading to moral engagement or disengagement.

The second observation related to the interactions between economic incentives and moral motivations is on the information content of economic incentives. In principal-agent problems, economic incentives may convey information about the intention or beliefs of the principal who is imposing an incentive (punishment or reward) on the agent. If the incentive implies distrust towards the agent, the agent may act dishonestly, for instance, or respond with less effort in working relations (e.g., Fehr & List, 2004; Fehr & Rockenbach, 2003). Yet if the incentives indicate the principal has good intentions, then their effect is reinforced (e.g. Borges & Irlenbusch, 2007; Fehr, Klein, & Schmidt, 2007).

The third factor is related to the self-determination of individuals. Whenever individuals are intrinsically motivated to undertake an activity and an external economic incentive is imposed, this intervention may "overjustify" the activity and undermine their feelings of self-determination by undercutting their previous autonomy. This type of control may be considered unacceptable and lead to resistance. While negative incentives such as penalties may be perceived as controlling and hostile, positive incentives such as rewards may be perceived as supportive, possibly leading to the crowding-in of intrinsic motivations (Frey & Stutzer, 2008; Vollan, 2008). These three effects discussed so far are examples of "state-dependent" preferences (Bowles & Polanía-Reyes, 2012). Their presence changes individuals' "sense of situation" and the context of the choice environment, and as such, lead to a change in behavior (even if temporary) when economic incentives are employed.

The fourth category of how incentive mechanisms may change behavior is more long-lasting, and concerns "endogenous" preferences. Endogenous preferences are related to a change in cultural norms that begin in youth and persist over the lifespan. In other words, they involve a long-term learning process that is difficult to replicate in a short-term economic experiment. If a newly acquired behavior persists after the economic incentive is removed in an experiment, this could be seen as evidence for endogenous preferences. For example, if individuals continue to fasten their seat belts after the fine for not wearing one has been lifted, this may be taken as evidence for endogenous preferences.[12] The matter of endogenous preferences is still a controversial topic for mainstream economists. The difficulty of proving the presence of endogenous preferences in experiments obviously adds to the controversy.

Applying the motivation crowding theory to the realm of environmental problems requires that environmental policies are reconsidered in a different light, because most of these instruments are based on economic incentives. Fines, taxes, subsidies, and tradable permits all make use of material incentives. Frey and Stutzer (2008) argue that environmental morale should play a major role in environmental policy-making due to the following four reasons:

First, instruments based on economic incentives may be inapplicable or too costly to implement in many instances of environmental policy (Baumol & Oates, 1979). This can be the case when polluters are scattered across a given landscape, and/or generate pollution for short periods, making it too costly to monitor each and every one. Consequently, voluntary behavior becomes necessary to reduce pollution. Second, environmental goods are usually public goods by nature. However, unless there are sanctions for non-contributors, contribution to public goods often remains inadequate. People with environmental morale would likely be willing to sanction each other, even at a cost to themselves. Evoking feelings of guilt and shame in non-contributors requires that contributors are reciprocal and fair—to a certain extent—in

[12] A review for endogenous preferences is provided by Bowles (1998, 2004) and Bowles and Gintis (2011).

their preferences. Third, complying with laws and regulations also requires moral motivations. Naturally, this is only the case when laws and regulations are perceived to be legitimate and fair. Finally, the fact that environmental problems by nature are a public goods problem is also important in voting. Voters will support policies needed to protect the environment only if they have environmental morale to a certain degree, because voting in favor of environmental protection usually implies a private cost for citizens.

There are actually many examples where environmental morale is at work in real life. People do not litter public spaces, boycott firms that harm the environment, and sort their trash even if there is no legal obligation to do so—as long as the costs of undertaking such behavior are not too high for them. Crowding-out may occur in these cases following the imposition of economic incentives. A crowding-out hypothesis was also formulated for corporations in terms of pollution issues (Frey, 1992; Gawel, 2001).

In real life environmental policy-making, economic incentives play an increasingly important role in conservation (Pirard, 2012; Rode et al., 2015; WBCSD, 2011). However, there are concerns about these policies (Child, 2009; McCauley, 2006; Redford & Adams, 2009). One of the main reasons is their potential to crowd-out conservation motivations, at least in the long-run (Gomez-Baggethun & Ruiz-Perez, 2011; Kosoy & Corbera, 2010; Luck et al., 2012; Rode et al., 2015; Vatn, 2010). Crowding effects are, in fact, especially relevant in terms of payments for ecosystem services schemes, which are very popular new tools for conservation (Rode et al., 2015; Vatn, 2010). Despite its immediate relevance, however, crowding effects have so far been under-investigated in relation to resource use and conservation.

Building on the motivation crowding theory in social psychology and experimental studies on public goods, Frey and Stutzer (2008) formulated a theoretical model based on the following four hypotheses regarding crowding-out:

First, command-and-control (CAC) type policies (i.e., direct regulation) are likely to undermine environmental morals. This is to be expected, because regulations, which are prescriptive by nature, compromise an individual's sense of self-determination. However, a counteractive effect is also potentially likely, since regulations can also point to the normatively correct or appropriate behavior. Accordingly, regulations might support pro-environmental behavior if they are easy to understand, and the link between cause and damage is simple to establish. If the penalty for getting caught is small and its probability low, then the relative price effect of the economic incentive will be small. Given that individuals do not always possess the cognitive capacity to calculate the expected penalty in a fully rational manner (Dawes, 1988; Kahneman, Slovic, & Tversky, 1982; Thaler, 1992), the final outcome cannot be known with certainty (Schoemaker, 1982, as cited in Frey and Stutzer, 2008). The outcome depends on the relative magnitudes of crowding-out and relative price effects. In real life, the probability of being penalized and the penalty itself is rather small, so one would not expect the relative price effect to dominate the crowding effect.

Second, according to Frey and Stutzer (2008), tradable permits will have a strong crowding-out effect on environmental morale. A price is established when a market is created for tradable rights to pollute or extract. This price acts as an external material incentive that changes the relative price of the environmentally harmful activity. If seen as a "right to pollute," tradable permits signal that it is legitimate to pollute the environment as long as economic actors own the pollution rights that are paid for within a market context. As such, this instrument does not involve a sense of punishment for a harmful activity. Rather, it grants the polluters a “license to pollute” not unlike the “indulgences” of Medieval times (Goodin, 1994). In this view, tradable permits are similar to Medieval-era indulgences in the sense that one can dispense of the responsibility of a sin (environmental damage) as long as one pays for it. Tradable permits, therefore, are concerned with neither environmental morale nor an intrinsic motivation to protect the environment. There is no point in employing moral considerations in this context, since there is a fixed amount of pollution rights anyway. If one refrains from using them, another economic actor probably will not. Hence, Frey and Stutzer (2008) expect a crowding-out effect in the case of tradable permits, and with environmental taxes as well.

Third, environmental taxes or charges will also lead to a crowding-out effect, but to a smaller extent compared to tradable permits. This is because the connotation of taxes is different. They, albeit indirectly, express the idea that an environmentally harmful activity is an undesirable one. Still, a crowding-out effect can be expected since taxes may also undermine people’s sense of self-determination. Moreover, taxes (or charges) can be expected to perform better than tradable permits as long as the damage costs are identical, and hence, the relative price effect is the same, since the crowding-out effect would be smaller.

A fourth proposition is that low or high taxes will perform better than intermediate level taxes in terms of environmental performance. This rests on the idea that low taxes have symbolic meaning in support of environmental morale, but at the same time, they do not compromise people’s sense of self-determination much, since the amount of payment is low, as is the relative price effect. Experimental studies suggest that low taxes or charges can indeed work well (Thogersen, 2003). On the other hand, high taxes are expected to perform well due to a stronger relative price effect that dominates the crowding effect, especially if environmental morale was already weak prior to the tax. An intermediate level of taxes, however, would have a strong crowding-out effect on intrinsic motivations, and the relative price effect would not be strong enough to support a lessening in the environmentally harmful activity. This may be in line with the "Pay enough or don't pay at all" approach by Gneezy and Rustichini (2000) who investigated parental behavior in childcare centers at Haifa, where a crowding-out effect occurred after the introduction of a monetary fine for parents who picked up their children late. Parents' intrinsic motivation was apparently crowded-out by the moderate monetary incentive.[13]

Frey and Stutzer (2008) put forth the following three hypotheses on crowding-in: First, they proposed that crowding-in in terms of environmental motivation can be achieved with appeals and participation in the short-run, and with education in the long-run. This is because these instruments would support people's sense of self-determination, and signal that it is their own responsibility to protect environment. There may even be some favorable spill-over effects; when such an instrument is introduced in one area of an environmental problem, this may lead to a crowding-in in other cases of environmental degradation. Empirical evidence for this can be found in issues related to recycling and water conservation (Baumol & Oates, 1979; De Young, 1985; Hopper & Nielsen, 1991). However, researchers also caution that there are important limitations to using policies based on intrinsic motivations. Accordingly, the

[13] Even more interestingly, this effect continued after the fine was removed.

relationship between the cause and effect of environmental damage must be clear enough so that one can actually rely on intrinsic motivations. Even more importantly, sustaining the positive effects of environmental morale is difficult, and it may drop significantly when there is substantial free-riding. Therefore, one cannot design policies that rely on intrinsic motivations alone. However, intrinsic motivations are still a very important aspect of environmental policy-making. In particular, the authors suggest that market-based instruments should be complemented with a considerable amount of environmental morale to be effective.

The second hypothesis is that legal regulations would reinforce environmental morale with their expressive function. Such regulations differ from standard CAC-type policies in that they do not emphasize the punishment aspect. The function of the law is rather to point to socially-desirable moral behavior to protect the environment. Frey and Stutzer (2008) assert that this aspect is often neglected by standard economic theory, even though the law and economics literature has long been aware of this effect (Cooter, 1984; Sunstein, 1997).

The final hypothesis of Frey and Stutzer (2008) concerns environmental subsidies. They argue that environmental subsidies have an ambiguous effect on environmental morale. Subsidies point to desirable behavior, and as such, can have an expressive function leading to crowding-in. The relative price effect would also work into the same direction. However, there is also the possibility that subsidies may be perceived as “controlling bribes,” and therefore undermine environmental morale. So the final outcome would be ambiguous.

3.2.3 Community-based management

Another alternative perspective to address environmental problems, especially resource use dilemmas, was developed by Elinor Ostrom and her colleagues. Their approach underlines the importance of community-based institutional arrangements, such as social norms, informal rules, and sanctions. Collecting and analyzing numerous case studies on the communal use of common-pool resources (CPRs) in different countries and contexts, Ostrom (1990) and her colleagues identified several conditions under which communities can manage CPRs sustainably. The work of Elinor Ostrom also inspired the rich experimental literature on lab and field experiments with resource users. A review of her contribution to the literature on resource use dilemmas is therefore very important for the empirical purposes of this dissertation.

Ostrom et al. (1999) were the first to clarify the difference between resource systems and property right regimes. Accordingly, CPRs are defined as natural or man-made resources where i) extraction by one user diminishes other users' extraction potential and, ii) excluding any user is extremely difficult and/or costly via institutional or physical means. This creates short-term perverse incentives for individuals to extract as much as possible, whereas they would cooperate and conserve the resource if they were to act in line with their long-term interests. Fish stocks, man-made irrigation systems, the internet, et cetera can all be viewed as CPRs because it is difficult to exclude other users, and there is rivalry among users. Free-riding is common in CPR use: users over-extract the resource and/or do not contribute enough to preserve or maintain the resource. Their use of the term CPR is independent of the property right regime with which the resource is governed.

Property regimes, on the other hand, can be divided into four categories: open access, common property, individual property, and government property regimes. Under an open access regime, Garrett Hardin's "tragedy of the commons"—which implies that in the use of common-pool resources, over-extraction is inescapable in the absence of state intervention or the assignment of private property rights—is more likely to occur since there are no enforceable property rights. However, this may not be the case if resource users are able to alter the regime to one of group or individual property. There is much empirical evidence that indeed they can (Feeny, Berkes, Mccay, & Acheson, 1990; Ostrom, Gardner, & Walker, 1994).[14]

Situated within the methodological individualism paradigm, the Ostrom framework is mainly based on the approach of new institutional economics. As such, it primarily analyzes formal and informal institutions, such as collective-choice rules, and monitoring and sanctioning processes, but it also focuses on other bio-physical characteristics of social-ecological systems, such as physical resource characteristics, harvest methods and technology, and the extraction rate of resource users. The strong focus on community-management does not preclude the analysis of characteristics external to the system, such as constitutional rules, non-governmental organizations, and government organizations (Basurto & Ostrom, 2009). The framework's main contribution to environmental policy discussions may be seen as the shift in focus from

---

[14] While both individual and common property regimes give users the ability to exclude other users, the difference between them is related to the ease of selling and buying a share of the resource. Under a government property regime, on the other hand, a local, regional, or national public authority can impose restrictions or support mechanisms on users, since it holds the resource rights. The authors underline, however, that none of the property regimes work well universally in all cases of CPR problems, i.e. there are no panaceas. Despite strict regulations, many CPR problems may remain unsolved and over-extraction may continue (OECD, 1997a).

state versus market dichotomy, towards community-based institutional arrangements (Ostrom, 2000).

Moreover, according to this framework, the main source of environmental problems is related not to market failure, but the presence of a collective action dilemma, which results from the conflicting interests of resource users. Such dilemmas may arise under a common property regime, an open access regime, or a private property regime. That is, in contrast to the neoclassical view, the framework does not conceptualize environmental problems as externalities inflicted on third parties, but rather as coordination failures among economic agents. Therefore, while the advice of the neoclassical approach is to internalize externalities by getting the prices right, the Ostrom framework advises getting the rules—for instance, the institutions—right. In line with the school of new institutional economics, institutions are seen as formal and informal rules and habits that shape and limit agents' social and economic actions (North, 1990). Moreover, individuals can have different motivations, both intrinsic and extrinsic. They may be motivated by selfish interests, or by reciprocity, altruism, and civic motivations. Ethical considerations, values such as justice and equity, et cetera are seen as part of institutional arrangements, which are context and culture-dependent (Kral, García & Aber, 2011; Muradian & Cardenas, 2015). Solving collective action dilemmas requires institutional change, which primarily implies a transformation in social norms and rules. Institutions are context-dependent, and so are the proposed solutions to environmental degradation or resource use problems. In contrast to the neoclassical approach, the framework also pays a great deal of attention to procedural justice as well as efficiency gains. Participation and legitimacy are important aspects to consider in terms of environmental policies (Biermann, Betsill, Gupta & Kanie, 2010).

Accordingly, environmental policies cannot be implemented as techno-fixes; that is, they are not universally applicable regardless of context. There are no panaceas in environmental policies. Rather, policy-making requires careful consideration of the specificities of the collective action problem at hand. Some collective action dilemmas can be solved by neither direct regulation nor marketization, but a combination of such policies may perform better depending on the characteristics of the resource, the resource users, their local arrangements (or governance structure), and external parties.

The standard economic model used to analyze resource use dilemmas is based on selfish, myopic, utility maximizing individuals, and predicts, in line with Hardin, the overuse and destruction of the CPR. Ostrom (1999) challenged Hardin's (1968) tragedy of the commons, based on its three fundamental assumptions: First, the tragedy of the commons assumes that resource users are "norm-free maximizers of immediate gains" (Ostrom, 1999, p. 496). In other words, they are unable to cooperate for long-term benefits and cannot overcome the coordination problem they are facing unless an external authority intervenes with a particular policy. This approach further takes it as granted that government authorities act in the public interest and can devise optimal policies that benefit resource users. The second widely-held belief regards designing the optimal policy as a simple technical task, and views different CPRs in different localities as being similar enough so that each one of them can be managed with more or less the same rules. The final assumption of this approach is that CPR management requires central management and policy-making; in other words, rules and regulations that originate from a central governmental authority. Ostrom (1999) argued that this was a poor theoretical basis to analyze different public policies regarding CPR use. She asserted that a model based on myopic, utility maximizing individuals cannot be supported by empirical evidence, and modified it with the assumption of "fallible, boundedly rational, and norm-using" (p. 496) individuals. That is, individuals are adaptive; they learn from previous mistakes and able to devise rules and tools to overcome social dilemmas they face repetitively in their lives. One important finding that originated from the field studies was that there were numerous different rules and designs that local resource users devised to overcome collective action dilemmas, and they were much more diverse than those encountered in the public policy literature.

Both empirical studies and data from laboratory and field experiments provide evidence that resource users are able to coordinate if they can communicate, sanction each other, and devise their own rules (Bowles, Boyd, Fehr, & Gintis, 1997; Ostrom, 1998). Individuals may adopt selfish behavior in various settings, but in many other contexts, they may have reciprocal preferences, for instance. More specifically, individuals may adopt one of the following positions: i) free-riding individuals (those who never cooperate and always behave in a selfish manner), ii) individuals who do not want to cooperate until it is made sure that they will not be exploited by free-riders, iii) reciprocal cooperators, who hope that other users will answer with a similar trusting response in order to overcome the dilemma at hand, and iv) altruists (a rather small group) who always try to improve total benefits for the group.[15] Empirical evidence suggests that if the proportion of free-riders is initially low in the group, then cooperation can be achieved and sustained, and may even grow (Axelrod, 1986). Moreover, conditional cooperators may build up a reputation for trustworthiness, and this may benefit themselves and their offspring (Kreps & Wilson, 1982; Nowak &

[15] Individuals in the second and third groups are called conditional cooperators since they are willing to reciprocate if others do not cheat and free-ride.

Sigmund, 1992). If group members can monitor each other and identify who is of which type, it is easier for them to sustain cooperation based on trust, reciprocity, reputation, and social norms. Resource users usually weigh the benefits of cooperation against the costs of monitoring, negotiating, and enforcement, in order to decide whether it makes sense to continue to cooperate. Benefits are higher; the more valuable the products of the preserved resource are. Costs are higher; the more complex the resource regime, the more users disagree on certain aspects (such as regeneration rates etc.), and the more their different interests diverge. Social norms are not always sufficient to sustain cooperation, however. Users themselves and external actors can enhance cooperation if they devise rules and mechanisms for monitoring, negotiating, sanctioning, and enforcing those who fail to comply with informal rules.

Several shortcomings have been identified regarding the Ostrom framework. First, theorizing within the methodological individualist tradition, this approach is said to neglect the importance of history, power issues, and poverty (Agrawal, 2003; Johnson, 2004, as cited in Knudsen, 2009). Further, it is argued that the role of politics and knowledge enter the analysis only in respect to institutions (Agrawal, 2006). Accordingly, several scholars argue that it needs to be complemented either by a political economy approach that focuses on power issues and the distribution of economic surplus, or a political ecology approach that analyzes the unequal distribution of environmental risks, burdens, and benefits (Knudsen, 2009; Muradian & Cardenas, 2015).

Indeed, power structures can prevent institutional change or exclude certain actors from the transformation process altogether. Consequently, some individuals or groups may be asymmetrically subject to environmental degradation while others play an active role in institutional change geared to furthering their own interests. Moreover, this view’s strong emphasis on context-specificity should not imply that there are no universal facts or more generalizable patterns in environmental policy. In fact, experimental studies aim to undercover such commonalities and general results (Muradian & Cardenas, 2015). It is further argued that this approach reduces the problem to an individual cost-benefit calculation of resource use disregarding the complexity of the problem in terms of cultural context, knowledge creation, and the embeddedness of social norms in society (Acheson, 2003, as cited in Knudsen, 2009). Yet overall, the approach underlines the importance of community norms, local institutions, and communal management, and thus not only provides valuable insights for CPR management, but also opens up new horizons to design and implement effective environmental policy instruments.

3.3 Looking ahead

One the most important lessons that can be drawn from these critiques and alternative approaches to environmental policy-making seems that one needs to conceive the individual as a citizen and a member of several different communities, rather than solely as a consumer looking to maximize personal payoffs. Individuals need to be analyzed as entities who are shaped and reshaped by different social structures, rather than who have exogenously-given and stable preferences (Madra & Adaman, 2010; Sagoff, 2008). Nevertheless, this does not imply that regulations and mechanism design are unimportant for environmental policies. The effects of material incentives on behavior must not be underestimated. However, one must also acknowledge that reducing everything to a mere incentive problem cannot be fruitful in terms of environmental problems.

Empirical evidence strongly suggests that individuals and communities can be intrinsically motivated to conserve environmental resources. If policies fail to address such pro-social motivations, they may backfire (Bowles, 2008; Frey & Stutzer, 2008; Gneezy et al., 2011). Similarly, under certain conditions, the internal rules and endogenous social norms of communities may perform better than government regulation or marketization in dealing with incentive problems and manipulation (Ostrom, 2000). Ostrom framework motivated numerous experimental studies on resource use and community-management of such resources. The next chapter discusses the findings of experimental studies on environment, including experiments on environmental valuation, environmental policy instruments and resource use dilemmas.

CHAPTER 4

ENVIRONMENTAL ECONOMICS MEETS EXPERIMENTAL ECONOMICS: A REVIEW OF EXPERIMENTS ON ENVIROMENTAL VALUATION, ENVIRONMENTAL POLICY INSTRUMENTS, AND RESOURCE USE DILEMMAS

Experiments are important tools to analyze human behavior, test assumptions and predictions of economics, and design new institutions. Since the 1970s, both field and laboratory experiments have been increasingly conducted by economists studying environmental problems. Economists—including, among others, Peter Bohm, Charles Plott, Vernon Smith, Jack Knetsch, and Ronald Cummings—working on issues such as negative externalities, Pigouvian taxes, and environmental valuation have made significant contributions to experimental and behavioral economics as well. As Vernon Smith argued recently, these contributions have challenged public economics so fundamentally that "[p]ublic goods theory will never be the same. We now think differently, more openly and positively on these issues" (Smith, 2008, p. xx).

This chapter aims to review the rich literature on experiments and environment. It first provides a categorization of different types of experiments by introducing the issues of context and framing. Then, it addresses various experiments conducted on environmental valuation, the earliest item to appear on the shared research agenda of environmental and experimental economists. Next, the chapter reviews experiments on environmental policy instruments, especially those on direct regulation, Pigouvian taxes, and environmental markets. The final section summarizes the main results of experiments on resource use dilemmas, with a specific emphasis on framed field experiments. While the section on environmental valuation serves the purpose of providing a complete picture of experiments on environmental issues, the sections on environmental policy instruments and resource use dilemmas prepare the ground for the field experiments in Part II of this dissertation.

4.1 The use and classification of experiments

Experiments are useful for environmental economists in at least three respects (Cherry, Kroll, & Shogren, 2008). First, experiments can be used to test economic theories. As in the case of public goods and resource use, a researcher can use an experiment to test whether the theoretical predictions of an economic model are robust, whether the assumptions of the model are valid, and to what degree the intended behavioral change actually occurs (Plott, 1983; Stranlund & Murphy, 2013). Second, experiments can be used as tools to better understand human behavior. Environmental valuation surveys, for instance, can identify behavioral anomalies and investigate the effects of information and contextual frames on behavior (e.g. Kahneman, Knetsch, & Thaler, 1990; Shogren, Shin, Hayes, & Kliebenstein, 1994). Using experimental methods, one can also study how and why people coordinate in social dilemmas, as well as how to improve cooperation in resource use (e.g. Cardenas et al., 2000; Velez, Stranlund, & Murphy, 2009). Finally, experiments can be used to test new institutions, mechanisms, and regulations that aim to achieve efficiency, such as tradable permit markets (e.g. Friesen & Gangadharan, 2013; Godby, Mestelman, Muller, Welland, & Muller, 1997).

An important distinction is usually made between laboratory and field experiments. In the initial phases of experimental economics, as it was trying to establish itself as a sub-discipline investigating economic behavior, laboratory experiments were preferred to ensure a controlled environment. In lab experiments, subjects are typically recruited from university students, and experimental sessions are conducted in an isolated environment with the aim of achieving experimental control and internal validity. However, as Harrison and List (2004) argue, if subjects perceive the game introduced by the experiment as being artificial, or the abstract task given to the subjects is not suitable for the context at hand, the experimenter's control may be compromised and the subjects may impose their own context on the game. Therefore, the use of field experiments, in which subjects are studied in the exposure of their everyday, real-life economic problems, has become increasingly wide-spread. As Harrison and List (2004) argue, field and lab experiments can be seen today as complementary tools. They also propose a four-fold taxonomy to distinguish between different types of economic experiments. These are:

- Conventional lab experiments are ones in which subjects are university students and the framing of the game is abstract;
- Artefactual field experiments are similar to conventional lab experiments except that the subjects are non-students;
- Framed field experiments resemble artefactual field experiments in that they use a non-standard subject pool, but the resource at hand is framed as in the actual field or the task given to subjects is based on the field context; and
- Natural field experiments are similar to framed field experiments except that subjects do not know that they are participating in an experiment; they perform the task at hand naturally as part of their daily life.

Whereas laboratory experiments are usually preferred for the testing of environmental policy instruments, artefactual and framed field experiments are mostly used to better understand human behavior in resource use dilemmas. By using lab and field experiments, researchers can also account for the shortcomings of certain environmental valuation techniques. Because context matters in resource use dilemmas, researchers usually prefer framed field experiments that recruit subjects from the group of actual resource users. Another reason is that abstract, context-free instructions of lab experiments increase the risk that subjects impose their own experiences on the game and reduces experimental control. In a similar vein, Eckel and Grossman (1996) point to the importance of framing and context:

> As experimenters, we aspire to instructions that most closely mimic the environments implicit in the theory, which is inevitably a mathematic abstraction of an economic situation. We are careful not to contaminate our tests by unnecessary context. But it is also possible to use experimental methodology to explore the importance and consequence of context. Economists are becoming increasingly aware that social and psychological factors can only be introduced by abandoning, at least to some extent, abstraction. This may be particularly true for the investigation of other-regarding behavior in the economic arena. (p.188)

4.2 Experiments on environmental valuation

Since the 1950s, economists have developed valuation methods for environmental assets that are outside the markets and do not have a price. The primary motivation for doing this is that it allows for cost-benefit analyses to be conducted. Economists must gauge the value people assign to the environment before coming up with recommendations on the efficient allocation of scarce environmental resources. Experimental methods are frequently used for this purpose. In fact, it is argued that environmental valuation and experimental economics "have grown up together"(Horowitz, McConnell, & Murphy, 2013). Experimental studies in this field have detected a series of anomalies in human behavior –such as loss aversion, endowment effects and status quo bias—and fundamentally challenged several assumptions of neoclassical economics, as will be summarized in this section.

Among the methods used to assign a value to environmental assets outside the market, the most frequently used is contingent valuation (CV). CV has been an especially useful method in the context of environmental disasters as well as projects that aim to improve the environment. However, as the CV method has become increasingly widespread and attracted more attention from environmental and experimental economists, the problems inherent to it have also gradually come to the fore, as will be addressed below.

The aim of CV surveys is to ascertain the value individuals assign to the improvement or destruction of environmental amenities. This value can be obtained in two ways: i) Measuring a respondent's willingness to pay (WTP) for one additional unit of environmental benefit; and ii) Measuring a respondent's willingness to accept (WTA) undesirable environmental damage, by asking them how much monetary compensation they would like to receive in return for tolerating one unit of environmental destruction. Theoretically, a respondent's answers to both types of questions should be very close to one another (provided that the income effect is negligibly small). Yet, CV surveys consistently deliver results that are just the opposite: Average WTA is up to three or four times the average WTP. This large discrepancy between the WTP and WTA figures is one of the main areas of environmental valuation in which experimental methods are employed.

The second application area of experiments in environmental valuation is related to “hypothetical bias”. Hypothetical bias arises from the fact that questions in surveys are usually based on a hypothetical situation. That is, respondents neither pay the WTP amount nor receive the WTA amount in CV studies. The prevailing wisdom among economists is not to trust the monetary value respondents provide unless the amount they state actually leaves or enters their pocket.[16] Several studies comparing real life to hypothetical situations suggest that this bias is problematic for CV studies (Cummings, Harrison, & Osborne, 1995; Horowitz et al., 2013; Sturm & Weimann, 2006).

More than in perhaps any other area, experimental studies have been vital to addressing these two important problems with CV surveys. This is because—while it is possible to observe and analyze real-life events, including social dilemmas or institutional arrangements and regulations, in environmental economics—it is not possible to place a value on non-market environmental benefits and damages. The next section provides a more detailed summary of experimental studies that address these two fundamental problems inherent in CV studies.

4.2.1 Experiments on the WTA/WTP disparity

The difference between WTA and WTP measures was one of the first behavioral anomalies to be uncovered in environmental economics. This disparity might even be considered one of the first anomalies to be discovered throughout the entire field of economics (Horowitz et al., 2013). In the first study addressing this disparity, Hammack and Brown (1974) found the WTA for hunting rights to be nearly four times the WTP.

[16] For additional problems with CV surveys, see Hanemann (1994), Spash, (2008), Venkatachalam (2003).

This large difference between WTA and WTP would also be found in numerous CV surveys to come, in both laboratory and field experiments (Horowitz et al., 2013; Knetsch, 1989; List, 2004; Rowe, D'Arge, & Brookshire, 1980).

Experiments examining the WTA/WTP disparity are typically designed as follows: Subjects are divided into two groups, and only one group is given the good in question. WTA is measured for the members of the group given the good, while WTP is measured for the members of the other group not given the good. Because subjects are randomly assigned to groups, it is expected that the average WTP of the group given the good will be equal to the average WTA of the group denied the good—unless a large anomaly or income effect is at play.

Almost all of these experiments result in a similar outcome: Average WTA is significantly greater than average WTP. In fact, regardless of survey design, this outcome persists. In 2002, Horowitz and McConnell (2002) conducted a meta-analysis that examined 45 such studies. They found the WTA/WTP ratios ranged from 0.74 to 113, and was 7.2 on average. WTA and WTP values for various goods—including chocolate, pens, movie tickets, hunting licenses, bad-tasting beverages, sandwiches that have gone bad, and wildlife conservation areas—were compared in both this meta-analysis and subsequent studies, which obtained similar results (Horowitz et al., 2013).

Sturm and Weimann (2006) grouped the results of experiments that explain the WTA/WTP disparity under three categories: The endowment effect, the Hanemann argument, and the learning effect, as explained below.

One of the arguments used to explain the WTA/WTP disparity is the fact that individuals are usually ready to sacrifice more in order not to lose what they already have. This is called the "endowment effect." In other words, people tend to dislike losses more than they enjoy gains. This phenomenon is also sometimes referred to as "loss aversion."[17]

The endowment effect is especially important for environmental goods. If an individual can state the amount of compensation they would require to tolerate environmental destruction, then this implies that they are already in possession of the environmental good or benefit in question. Consequently, given evidence for loss aversion or endowment effects, the value they assign to giving up the good is greater than the value they assign to paying for it and receiving something they do not currently have. This might be one reason why WTA is usually found to be greater than WTP.

Kahneman et al. (1990) conducted one of the most prominent experiments on the endowment effect. In this study, half of the subjects were in possession of coffee mugs decorated with a university logo and half were not. The selling price demanded by those who owned the mugs was greater than theoretically expected. Consequently, the volume of trade was also lower than theoretically expected.[18] Meanwhile, Knetsch and Wong (2009) examined the endowment effect in terms of reference points. The researchers claimed that changes in the situation that serves as a reference point can significantly affect the value people assign to their losses and gains. If people place greater value on their losses than their gains, the impact on economic welfare of a positive or negative change in losses will be greater than a positive or negative change in gains of equal magnitude. Therefore, they argue that for the economic welfare criteria to reflect the true change in people's level of welfare, one needs to account for the change with

[17] For more on this effect, see Kőszegi and Rabin (2006).
[18] For other studies that examine the endowment effect, see Dijk and Knippenberg (1996), Knetsch and Sinden (1984), Knetsch (1989).

respect to the reference point. Using WTP as a measure for all kinds of change regardless of loss or gain (as recommended by the U.S. Department of Commerce's National Oceanic and Atmospheric Administration (NOAA) Panel for the implementation of CV studies) seriously underestimates losses or reduced losses. Using WTA instead of WTP would cause actions that destroy the environment to be encouraged more, and prevent adequate reduction measures from being taken (Knetsch, 2013).

To explain WTA/WTP disparity, Hanemann (1991) refers to income elasticity and the substitution elasticity of the good under consideration. He argues that when income elasticity is held constant, if the good in question has a high elasticity of substitution, or can be easily substituted by another good, then the difference between WTA and WTP should be close to zero. As it is generally difficult or impossible to substitute environmental assets with some other goods, the difference between WTA and WTP for environmental goods is still expected to be large.

Shogren et al. (1994) empirically tested Hanemann's hypothesis by conceiving personal health as a good with considerably low substitution elasticity. To this end, the researchers compared sandwiches contaminated with a pathogen, a good that will negatively affect personal health, to candy bars, which can easily be substituted with another good. The experiment had two treatments. In the WTP treatment, subjects were given a contaminated sandwich and asked how much they would be willing to pay for a fresh, uncontaminated one. In the WTA treatment, subjects were given a fresh sandwich and asked how much they payment they would require to exchange it for a contaminated one. (The subjects had to actually eat the sandwiches at the end of the experiment.) The researchers' results confirmed the Hanemann hypothesis: WTP and

WTA values were close for candy bars, an easily substitutable good, while WTA was much greater than WTP for the contaminated sandwiches, as it is difficult to find a substitute for health.

The endowment effect and the Hanemann hypothesis are closely related. Kahneman et al. (1990) stated that the endowment effect depends on the elasticity of substitution. This means that the endowment effect will be weaker for goods that are easily substitutable. In other words, both of the explanations for WTA/WTP disparity actually point in the same direction. This matter was addressed in Horowitz and McConnell (2002) meta-analysis, which found the WTA/WTP ratio was 10.4 on average for non-market goods (such as health or quality of environment) and 2.9 for goods that can be exchanged on the market. This second ratio, however, does not fit well with Hanemann's hypothesis. If the Hanemann argument is correct, the ratio for market goods that can be substituted relatively easily should be around one (Sturm & Weimann, 2006). This finding questions therefore the validity of the Hanemann argument.

Another problem with the Hanemann argument is that it is not possible to know the elasticity of substitution of a good provided by researchers to the experimental subjects. In other words, the argument that candy bars can be substituted more easily than health risks caused by contaminated sandwiches cannot be tested empirically. Moreover, it is not possible to know whether the degree of substitutability is large enough to explain the observed difference between WTA and WTP (Horowitz et al., 2013).

Another argument put forth to explain the difference between WTA and WTP is the learning effect. Accordingly, the endowment effect is expected to disappear when the game is repeated enough times to give the subject the opportunity to learn. An examination of several experimental studies, however, shows that this argument is not sufficiently supported (Sturm & Weimann, 2006). Although the learning effect was seen to be important in some studies (Coursey, Hovis, & Schulze, 1987; List, 2003), Kahneman et al. (1990) replicated previous experiments and found out that the endowment effect did not disappear with learning. Moreover, in their meta-analysis, Horowitz and McConnell (2002) also emphasized that the endowment effect weakens but does not completely disappear when the game is repeated. In sum, there is not enough evidence to conclude that the WTA/WTP disparity will decrease as subjects gain experience.

Given that researchers frequently observe differences between WTA and WTP, which one should be used in CV surveys? The recommendations of the NOAA Panel advise are to measure WTP, since WTA is always several times greater than WTP, meaning that WTP will always deliver more conservative estimates. However, many researchers argue that use of either WTA or WTP depends on the environmental problem at hand (for example, see Ahlheim and Buchholz, 2000). Accordingly, as a basic principle, WTP should be used in cases of environmental improvements, while WTA should be used in cases of environmental degradation.

4.2.2 Hypothetical bias

One of the most important criticisms of CV studies is that responses to survey questions do not have immediate monetary consequences in real life. While hypothetical bias has been known about since the 1930s, it became even more contested after CV surveys were used to study the 1989 Exxon Valdez oil spill disaster[19] (Horowitz et al., 2013).

Numerous experiments have been conducted since the 1970s to test whether the WTA and WTP values obtained in CV studies would still be found if subject made and received actual payments. In these experiments, subjects in one group are asked how much money they would hypothetically be willing to pay to obtain a certain good (hypothetical treatment), while subjects in a second group are made an offer that would actually require them to pay for that same good (real payment treatment). If the number of people who state that they will buy the good at hand at the indicated price turns out to be equal in each group, this would support the validity of the WTP or WTA values obtained via CV surveys. In other words, it would demonstrate that hypothetical bias does not in fact exist.

Bohm (1972) and Bishop and Heberlein (1979) were among the first to investigate this question. While they were able to detect hypothetical bias in their studies, their modeling methods and surveys were criticized later on, and it was not possible to replicate hypothetical bias in later studies (Andreoni, 1990; Dickie, Fisher, & Gerking, 1987; Hanemann, 1984; Mitchell & Carson, 1989; Sinden, 1988). Results from other studies conducted in the 1980s provided conflicting results, supporting both the existence and non-existence of hypothetical bias (Brookshire & Coursey, 1987). However, some of the studies carried out during this time were also considered to be problematic, as there were differences in elicitation procedures between the hypothetical and real payment treatments (Horowitz et al., 2013). Recent studies have found

[19] This disaster is considered as one the largest tanker spills and most serious environmental disasters in the history. 11 million gallons of crude oil were released into the sea (Carson et al., 2003).

responses given to hypothetical questions to be up to two to three times greater than in cases where the payment was actually made. For instance, Cummings et al. (1995) found a bias for private goods such as calculators and chocolate, while Cummings et al. (1997) found the same for public goods.

Unlike the WTA/WTP disparity, hypothetical bias cannot be considered a behavioral anomaly, nor does it indicate an inconsistency in neoclassical economics. This is because the neoclassical demand theory does not make any predictions about situations that do not have real economic consequences. Nonetheless, because the difference between hypothetical and real responses is rather large, the bias can still have a large impact on whether or not a policy is implemented (Horowitz et al., 2013). Consequently, two different calibration methods were put forth to overcome hypothetical bias: instrument calibration and statistical calibration (Cummings et al., 1997; Horowitz et al., 2013; Sturm & Weimann, 2006).

The experimental examination of whether hypothetical bias disappears when certain instructions are added to surveys is called "instrument calibration." It is thought that the additional instructions will enable respondents to review their answers so that they more closely reflect reality. Generally, four different methods are used in instrument calibration: Adding reminders to survey instructions about the budget; engaging in cheap talk; writing instructions geared to convincing subjects that their responses will have real-life consequences; and asking subjects how sure they are of the answers they provided.

Instructions that caution respondents about the budget have not been found to reduce hypothetical bias. Loomis et al. (1994, 1996), for instance, found no difference between the answers given by respondents who were reminded about the budget and those who were not.

Findings are still inconclusive regarding the cheap talk method.[20] Cummings et al. (1995) gave subjects information about the bias and asked them to take it into account while responding. In one treatment the reminder was brief (light cheap talk), while hypothetical bias was explained in more detail in the other (heavy cheap talk). The lengthy reminder eliminated bias whereas the brief reminder did just the opposite and boosted it. However, there is at present no robust theory or explanation as to why cheap talk works in one setup but not the other (Horowitz et al., 2013; Shogren, 2005).

Another instrument calibration method is to convince respondents that their decision could have real-life consequences. One way to do this is to announce to the respondents that the outcomes of the survey will be shared with the relevant authority. For example, flipping a coin to determine whether the study results would have real-life consequences has been shown to eliminate hypothetical bias (Horowitz et al., 2013; Landry & List, 2007).

Another method is to ask the respondents how sure they are of their answers. This method was found to be more effective in coping with hypothetical bias compared to cheap talk (Blumenschein, Blomquist, Johannesson, Horn, & Freeman, 2008; Blumenschein, Johannesson, Blomquist, Liljas, & O'Conor, 1998; Champ, Bishop, Brown, & McCollum, 1997).

By using data from the real payment and hypothetical payment treatments in experiments, and data on subjects' socioeconomic characteristics, researchers are

[20] Theoretically, cheap talk is not expected to affect experimental results because it does not change the economic incentives offered to people.

making efforts to predict the magnitude of hypothetical bias through statistical methods (Horowitz et al., 2013). Blackburn et al. (1994), for instance, used a multinomial logit model to predict the probability that a person who says, “Yes, I will pay this price,” in response to a hypothetical question will also respond affirmatively to a question that requires real payment based on the person’s socioeconomic characteristics. However, one important problem in statistical calibration is the difficulty in transferring the value predicted for a specific good and context to another context. Just as this value cannot be transferred from one context to another, it would be unwarranted to assume that hypothetical bias is systematically similar in different contexts.

Overall, experimental studies that aim to examine problems in environmental valuation surveys are not only effective at revealing and mitigating the weaknesses in these surveys, but they also point to various behavioral anomalies that have brought some of the predictions of neoclassical theory into question. Rational behaviors that are inconsistent and irrational from the perspective of neoclassical theory, such as the endowment effect, are even more important in a normative area, such as the environment. However, the behavioral anomalies detected in experimental economics are not mentioned in environmental policies and are not currently being given adequate attention by researchers and policymakers. If these findings are not taken into account, especially in the cost-benefit analyses calculated on the basis of data obtained from valuation surveys, social welfare may actually decline. Consequently, researchers who aim to ascertain people’s preferences about the environment via CV surveys and politicians who use these valuations in decision-making processes about the environment should reconsider the validity of these analyses and policies.

4.3 Experiments on environmental policy instruments

The purpose of this section is to review experimental studies on environmental policy instruments such as fines, taxes and environmental markets. Experiments on this domain often test the effects of existing instruments on the environmental outcome with respect to economic efficiency and distributional aspects, and they also help to devise new institutions and regulations to deal with environmental problems.

A common denominator of today's most pressing environmental problems such a climate change and industrial pollution is that they are considered to be a result of negative externalities. As explained in Chapter 1, there are two methods that enable the internalization of negative externalities. The first, that property rights need to be well-defined and easily enforceable so that the polluter and the victim can negotiate on compensation payments, was famously proposed by Coase (1960).[21] The second involves environmental policy instruments developed via intervention by a regulatory agency. The primary objective of environmental policy instruments such as direct regulation, Pigouvian taxes, and environmental markets is to ensure polluters internalize the negative externality they generate by paying a fee, fine, or tax, or by buying a pollution permit on the market in return for the per unit cost of the pollution they cause.

The theoretical grounds for environmental taxes are rooted in studies conducted from the 1920s onwards (Pigou, 1920), while studies on environmental markets are

[21] In order to internalize negative externalities via Coase negotiations, several unrealistic assumptions need to hold. For instance, the two parties involved, namely, the polluter and the victim, should be able to negotiate easily and transaction costs have to be zero. Because there will often be more than two actors involved in issues such as pollution and wastes, the transaction costs of holding negotiations the way Coase imagined them will also be quite high. In cases like these, external intervention will be necessary. Therefore, putting Coase negotiations into effect as an environmental policy tool is rather unlikely. For this reason, this chapter will not review experiments that examine Coase negotiations (see, for instance, Hoffman and Spitzer, 1982; Rhoads and Shogren, 1999; Shogren and Kask, 1992).

relatively more recent (Crocker, 1966; Dales, 1968). Despite the recent popularity of environmental markets, the most prevalent environmental policy used in practice today still involves direct regulation such as usage fees, technical standards, and fines. Experimental studies conducted since the 1970s have focused on the weaknesses of environmental policy instruments as well as on how to overcome those weaknesses.[22] In this context, environmental markets have been put forth as alternatives to direct regulation and taxes and are among the most commonly examined environmental policy instruments.[23]

4.3.1 Direct regulation and taxes

Plott (1983) was among the first to compare environmental policy instruments in an experimental setting, examining the relative performance of direct regulations, taxes, and environmental markets with respect to economic efficiency and distribution. In his study, six buyers and six sellers traded in some imaginary good, and participants earned a certain income as a result of their transactions. However, as the total number of transactions, or total trade volume, increased, each participant's income decreased by a certain amount. This represents a negative externality in this experimental setting. All transactions were carried out via double auctions[24] once subjects were assigned the role

[22] Several of these problematic areas are discussed in Chapter 1.

[23] This is probably because direct regulations are based on very simple financial incentives, while environmental taxes are neither very popular among the public, which make them potentially not very applicable, nor are their effects much different from other relatively more familiar taxes (Bohm, 2003; Friesen & Gangadharan, 2013).

[24] In a double auction, there are two different parties: buyers and sellers. Each buyer (seller) proposes a bid (an offer) simultaneously. The highest bid (or the lowest offer) is the current or standing bid (offer); this price is announced to everyone involved in the auction. Any buyer (seller) can either accept the proposed current offer (current bid) or propose a new bid (offer); the proposed new bid (offer) has to be higher (lower) than the current bid (current offer). In many experiments that use the double auction

of either buyer or seller. There were four different treatments in the experiment: i) no intervention; ii) optimal taxation; iii) direct regulation, where total amount of trade is restricted at an optimal level; and iv) pollution licensing market, where pollution rights are restricted at an optimal level and are then bought and sold. Plott compared these four treatments on the basis of efficiency, price behavior and distributional aspects. He found that the no intervention and direct regulation treatments generated the least amount of total earnings, while the pollution licensing market treatment generated the highest. The licensing market's earnings were followed by those of the taxation treatment. As a result, Plott (1983) concluded that the models in the economics literature on direct regulation, taxation, and licensing markets were all "amazingly accurate" (p. 119).

At present, only a few studies have investigated Pigouvian taxes via experimental methods. This is probably because Pigouvian taxes are rather unpopular and not considered feasible by the general public (Friesen & Gangadharan, 2013). A recent example is a study conducted by Kallbekken et al. (2011), who examined the effects of presenting a tax in different contexts on tax aversion. The researchers found that framing the financial incentive as a "tax" instead of a "fee" reduced support for the policy, whereas stating what the tax revenue would be used for increased support for it. In another study on taxes, Heres and Lin (2012) compared taxes and permit markets, and ascertained that the two markets could lead to equivalent outcomes in cases where there was uncertainty about the marginal damage that pollutants caused.

---

method, licenses are observed to be allocated efficiently only after several periods of buying-and-selling (Sturm & Weimann, 2006).

4.3.2 Environmental markets

Environmental markets are constructed based on a cap or upper total limit put on a resource or pollutant. This amount is then divided into smaller allowances that can be bought and sold in the permit market. Also called “cap-and-trade” mechanisms, environmental markets are thought to be both efficient and effective in achieving targeted abatement, and, consequently, are considered superior to direct regulation and taxation mechanisms. Theoretically, economic efficiency will be achieved since equimarginal principle[25] is satisfied, and the pre-defined cap will ensure environmental effectiveness. However, in the presence of firms with monopoly power, the effectiveness of such markets cannot be ensured. In other words, the target level of abatement may not be achieved (Hahn, 1984).

The first notable real-life application of environmental markets was the sulfur dioxide markets established in the U.S. in the 1990s to prevent acid rain. Following the 2005 Kyoto Protocol, carbon markets were implemented in the EU in the field of climate change. Environmental markets are used in many other areas, including, but not limited to, conservation of biodiversity, water rights, and fishing rights. Although carbon markets enjoy substantial support from free market environmentalists—as they give the impression that it does not require state intervention—they have been heavily criticized by radical environmentalist movements (Hahnel, 2015).[26] Another case of environmental markets, namely, the market for individual transferable fishing rights, has been strongly criticized as well, on the basis that fishing rights will amass in the

[25] Equimarginal principle within the context of environmental markets implies that the marginal costs of abatement will be equal across all pollution sources due to the equilibrium price of permits, as Chapter 1 indicates.
[26] For these arguments, see Lohmann (2010) and Spash (2010).

hands of a few large firms, as a result of which small-scale fishing practices—a traditional lifestyle—will disappear, causing local economies and communities to suffer.

Provided below is a brief review of experiments on environmental markets that focuses on two prominent cases: carbon markets and individual transferable quotas. First, however, it will serve us well to touch upon how carbon markets work.

The first step in establishing a carbon market is for the state or regulatory agency to announce the total amount (in tons, for instance) of emissions permitted during a certain period of time. The announced amount is set at a level lower than the total annual emissions emitted during previous periods.[27] The regulatory agency divides the total amount into smaller pieces or permits and allocates them to firms on the basis of auctions or grandfathering. It then periodically monitors whether firms' emission levels and the amount they are permitted to emit are consistent with one another.

In a case such as this, firms have a number of options. They can: i) reduce their emission levels to correspond with the allocated permits; ii) reduce their emission levels a little and buy the rest from the market, depending on their abatement obligations; iii) reduce their emission levels more than they are obligated to and sell the surplus permits to others in the market; or iv) reduce their emission levels more than they are obligated to and bank the surplus to use during later periods. Since emission reduction costs usually differ from firm to firm, it is expected that firms will buy and sell permits in this

[27] One has to note that the determination of the level of the cap is controversial. In economic terms, it should be set at a level to equate marginal abatement costs to the marginal damage costs of the pollution. However, different actors such as lobby groups, governments and NGOs usually play an important role in this process so that the cap may not coincide with this optimal level. Moreover, the determination of marginal abatement and marginal damage costs is also very difficult.

market. Firms that are able to reduce emissions at a higher marginal cost will be able to save costs by buying permits from firms that reduce emissions at a lower marginal cost—provided that the price is lower than the marginal abatement costs. Firms that can also reduce emissions at a lower marginal cost and sell their surplus permits to earn extra income (Friesen & Gangadharan, 2013).

In theory, as long as transaction costs are low and markets, both the permit market and the output market, are competitive, the outcome of permit markets will be efficient. As mentioned, Plott (1983) tested this theoretical deduction and attained results that confirmed it. In the three decades following Plott's study, numerous other studies on the emissions trade have been conducted. These studies have mostly consisted of laboratory experiments, and have analyzed different market regulations and their impact on market activity, participation, trade volume, permit prices, transaction costs, compliance, and enforcement. Market uncertainties and differences between actors, for instance, monopoly power, have also been frequently examined (Friesen & Gangadharan, 2013). Other questions that have been investigated include the following: How should emission permits be organized? What kinds of market regulations should be developed? What kind of relationship is there between market structure and market performance?

Experiments on the emission permit markets involve at least eight to twelve subjects to ensure permits will be bought and sold on the market. They are usually carried out with the help of computers. In experiments with auction designs, the double auction method is the one most frequently used. Even with a small number of subjects, this method makes it possible to attain substantially competitive prices and efficient outcomes (Cason & Gangadharan, 2013). The game and its tasks are mostly described to the subjects in abstract terms. For instance, subjects are not informed that the experiment concerns the environment or the climate.[28] The reason for this abstract framing is to reduce the chances that "green preferences" on the part of the subjects harm the generalizability of the game (Sturm & Weimann, 2006).

One frequently studied topic is banking, where firms save permits unused during a given period for use in upcoming periods. Since the permit market will be highly uncertain during its initial periods, price fluctuations are expected. Theoretically, under the assumption of perfect foresight, the ability of firms to bank and save permits for use during the next period should attenuate price fluctuations and thus ensure an efficient intertemporal distribution of permits (Cronshaw & Brown-Kruse, 1999). From the perspective of the regulatory authority, however, firms' ability to bank permits compromises their control over the amount of pollution emitted during each time period. In an experiment conducted using the double auction method, Godby et al. (1997) found that having the option to bank emission permits improved market efficiency. However, this option also resulted in reduced compliance and thus increased emissions (Cason & Gangadharan, 2013; Friesen & Gangadharan, 2013). In other studies, when given the option to bank permits, some firms banked them in surplus of the optimal amounts for the next period (Cronshaw & Brown-Kruse, 1999; Mestelman & Muller, 2006).

[28] There are some exceptions. Cason and Raymond (2011), for instance, used abstract concepts to explain the game and its rules under one treatment, but used instructions that suggested the experiment was about the environment in the other treatment, and examined the impact this difference in framing had on whether emissions were reported correctly to the regulatory agency or not. Another exception is Bohm and Carlén (1999).

Since investments are usually irreversible and require large capital investments, one of the most important targets in the emissions trade is technology investments geared toward reducing emissions to be realized at optimal levels and enabling firms to shift to relatively low-cost emission abatement technologies. Currently, however, only a limited number of experimental studies have investigated the impact of emissions trade on abatement technology investments (Friesen & Gangadharan, 2013). The few examples include Ben-David et al. (1999), who demonstrated that when abatement technology costs differed from firm to firm, the emissions trade and market efficiency experienced a drop. Kusakawa and Saijo (2003) also found that efficiency decreased when uncertainty surrounded investments. Finally, Friesen and Gangadharan's (2013) study revealed that firms were likely to overinvest in abatement technologies.

There are several experimental studies that examine the impact of market uncertainty around how much of the total emission permits will fall in the upcoming period on market performance. Theoretically, in the presence of market uncertainties and the transaction costs of emissions, the market is not expected to perform well (Hahn, 1989). Experimental studies testing this theory have found that in cases where there was uncertainty around how much and when the total emissions limit would be reduced, firms adopted what is called "wait-and-see" behavior (Ben-David et al., 2000). Meanwhile, studies analyzing the relationship between market uncertainty and banking permits for later use have found that this option helps to stabilize permit prices, but at the cost of market efficiency.

Studies have also examined market performance in cases of imperfect enforcement.[29] While some have directly investigated compliance with the rules of emissions markets (Cason & Gangadharan, 2006; Murphy & Stranlund, 2006, 2007), others have examined the social costs and benefits of non-compliance. Stranlund and Murphy (2013) found that even in cases where enforcement is not perfect, the emissions trade could be efficient in allocating permits and non-compliance would occur much less often than theoretically expected. This means that even if enforcement cannot be fully realized, social costs could be lower than what is predicted by standard models.

Another important issue that has been examined in market experiments is how emissions markets are impacted when there is a monopolization trend in the market, that is, when some firms have market power. Theoretically, under the assumption of perfect competition, markets allocate permits in an efficient manner independent of the initial distribution of permits (Montgomery, 1972). However, in the absence of perfect competition, the strongest firm can manipulate prices (simple manipulation), which entails decreased market efficiency (Hahn, 1984). If the firm monopolizes not only the permits market but also the output market, even more negative consequences will result. In such cases, emissions permits will be accumulated by the firm with market power and can be used to push other firms out of the sector (exclusionary manipulation). Laboratory experiments examining market power have looked at these two types of manipulation. While the outcomes in simple manipulation treatments were close to competitive equilibrium, the outcome in the exclusionary manipulation treatment was far from it (Brown-Kruse, Elliott, & Godby, 1995; Sturm & Weimann, 2006).

[29] Imperfect enforcement refers to cases where audits and fines are not high enough to ensure full compliance to the rules and regulations.

License exchange markets are put into effect for other commodities as well, such as hunting and water-use rights (Tietenberg, 2002). Markets on fishing rights, for instance, have been widely debated over the past several decades and are used widely in some countries, including Canada, Iceland, and New Zealand. The following section reviews experiments on individual transferable quota markets as another prominent example of environmental markets.

Individual transferable quotas (ITQs) work similarly to emissions permits markets. The ITQ market is theoretically expected to be efficient because resource users who are not efficient enough, for example, a fisher who catches fish at a high marginal cost, can earn more income by transferring his/her resource use permits, or quotas that determine how much fish he/she can catch, to another market actor who catches fish at a lower marginal cost. Although these markets are easy to defend in economic terms based on the argument that they improve economic efficiency, they also cause some to worry due to the potential for a large number of quotas to accumulate in the hands of few efficient fishers, thereby hurting local economies. Consolidation expectations of this kind make it difficult to implement market-based mechanisms on issues that have historically involved a great deal of stakeholder participation in decision-making mechanisms, such as fishing.[30] This is why the most important issue addressed in experiments examining transferable fishing quotas is whether or not the market gives rise to consolidation. Theoretically, one natural outcome would be for relatively

[30] The consolidation observed in the North Sea fishing grounds seems to confirm the concerns raised on this matter. After transferable quotas were put into effect in this area, the number of fishers who held a quota dropped 29 percent between 1988 and 1997, and the number of fishing boats dropped 33 percent between 1987 and 1998 (Davidse, 2001). With respect to the environmental effectiveness of ITQs, on the other hand, some studies suggest ITQ markets can reduce overfishing (for example, Anderson et al., 2008; Arnason, 2002; OECD, 1997a). However, other studies have provided mixed results in this respect (Smith, Gibbs, & Smith, 2009).

inefficient fishers to sell the quotas allocated to them and for the number of users to fall as a result. However, a significant question concerns whether or not consolidation will occur at excessive levels due to market volatility or the market power of large-scale fishers. An excessive or higher than the optimal level of consolidation is desired neither by the stakeholders in the fishing industry nor by resource economists.

Numerous experiments have investigated the conditions under which consolidation occurs. For instance, Anderson et al. (2008) made use of two treatments: consolidation and diffusion. That is, while one treatment involved cost parameters expected to lead to consolidation, the other had a cost structure expected to result in diffusion. Quota-allocation methods and optimal level of fishing efforts were held constant in both of the treatments. The researchers reported that under the consolidation treatment, the shares of large-scale operators increased as expected, but, beyond that, no other findings were obtained with regard to excessive consolidation. Meanwhile, under the diffusion treatment, small-scale operators purchased some quotas from their large-scale counterparts, albeit in small amounts. In summary, the researchers concluded that transferable quota markets will not always lead to excessive consolidation, and that whether or not consolidation occurs basically depends on market structure (e.g., costs).[31]

The prevention of large price fluctuations in transferable quota markets observed when the regulation is first put into effect is another notable topic of study. During the initial establishment of the ITQ market, because the asset is new and the market actors have not yet had sufficient time to observe supply and demand, prices will be highly

[31] A notable limitation of this study, however, is that in neither treatment were operators given the opportunity to withdraw from the market. In other words, the experiment allowed only for small-scale changes. As any fisher may exit the market in the real world, this raises significant concern about the external validity of this study.

volatile. Meanwhile, price volatility will affect the accumulation of quotas. Unpredictable prices can cause small-scale actors, such as those engaged in fishing as a family business, to withdraw from the market. This is perceived as unfavorable by stakeholders and constitutes a serious obstacle to putting the market mechanism into effect. Studies examining methods to reduce price volatility and speculative bubbles in these initial stages of equilibrium search have demonstrated the benefits of an initial lease period in which a permanent transaction of quotas is prohibited for a limited time (Anderson & Sutinen, 2006).

4.3.3 Some limitations of experiments on environmental markets

Several issues need to be considered for experimental studies on environmental policy instruments. First, a noteworthy problem concerns external validity of such experiments. For instance, it is impossible to exactly replicate real-life institutional arrangements on the emissions trade or transferable quotas in a laboratory setting.[32] Nonetheless, experiments examining possible patterns of human behavior and market performance have made substantial contributions to the field. Moreover, thanks to controlled field experiments and the use of electronic markets[33], which have become popular in tandem with recent technological advances, it is becoming easier to conduct experiments on environmental markets and to test their institutional mechanisms (Sturm & Weimann, 2006).

---

[32] In contrast, this does not constitute a great problem in a social dilemma experiment, for instance, because even if the real-life dilemma is not replicated exactly in the experimental setting, creating a dilemma that involves similar incentives and examining human behavior when faced with this dilemma seems to be sufficient.

[33] These are online platforms bringing together buyers and sellers of various goods and services, a prominent example of which is Ebay,

Second, since in experiments on environmental markets the goods and relevant institutional arrangements in question are presented to the subjects as abstract, context-free concepts, and the experiment's relevance to the environment is not openly stated, the extent to which parallels can be drawn with real-life institutions is debatable (Sturm & Weimann, 2006). Some of the experiments in the literature are geared toward understanding whether or not this constitutes a significant problem. In some of these studies, the additional information provided to the subjects on the market or good in question has had a positive impact on market performance (Cummings, Holt, & Laury, 2004; Tisdell, Ward, & Capon, 2004; Ward & Tisdell, 2008). Others, however, have generated contrasting findings. Cason and Raymond (2011), for instance, found that subjects who had the game and its environmental framework explained to them followed the rules less than those to whom the game was framed neutrally. Consequently, although context has a notable effect in market experiments, the direction of this effect is still unclear.

Third, as the number of experiments on environmental markets increase, new debates arise in relation to the subject groups used, the method of exchange in the market, and the cognitive difficultly level of the experiments (Sturm & Weimann, 2006). Regarding the first aspect, subjects in experiments on environmental markets are usually selected from among university students. To determine whether this constitutes a potential problem in the specific case of environmental markets, some studies have compared subjects highly knowledgeable and experienced in these kinds of markets with student subjects. As of yet, significant differences have not been found between student subjects and subjects with experience in environmental markets (Bohm & Carlén, 1999; Bohm, 1997; Burns, 1985). Concerning the second aspect, experiments on environmental markets usually adopt the double auction method, which slight differences in implementation from one experiment to the next. Because the theoretical underpinnings of the double auction method are relatively less established, the source of these small differences observed in different experiments on market performance is still not fully understood (Sturm & Weimann, 2006). Another factor that differs from one experiment to the next is the level of cognitive difficulty. Uncovering individual-optimal behavior in permit markets is a rather difficult task, especially if subjects are not familiar with this type of experiment. This is why the amount of practice that subjects have had in this market as well as how much opportunity they have had to learn are also important factors. How learning affects the findings of these experiments is an area in need of further research.

Lastly, a more general and noteworthy criticism of the experiments on environmental policies is that these studies have yet to adequately focus on the behavioral changes caused by different policy instruments. For instance, how individuals' responses to a given policy instrument differ from one another, and how such individual differences might affect policy development, is still not fully understood. Another question that awaits detailed examination is whether the behavioral characteristics individuals have acquired due to a policy instrument in effect spill over into other areas of interaction (Frey & Stutzer, 2008; Friesen & Gangadharan, 2013).

4.4 Experiments on resource use dilemmas

Since Hardin's (1968) prominent paper on the "tragedy of the commons," use of common-pool resources has attracted much attention from experimental economists as well. Experiments in both the lab and in the field have tested whether the behavior predicted by traditional, non-cooperative game theory occurs in these settings. The prediction of this theory is that resource users will end up in the Nash equilibrium, which is economically inefficient.

Real-life examples of over-extraction, such as overfishing, overgrazing, and overuse of water resources, have long been treated as empirical evidence validating these theoretical predictions. These predictions were based on the fact that one person's use of the resource decreases the amount consumable by others. Due to rivalry, it is difficult to restrict people's access to common-pool resources. Therefore, resource users and the society at large would be better off if they could coordinate and overcome the collective action problem. This would restore economic efficiency and ensure sustainable use of the resource.

Since the 1970s, Hardin's prediction has been challenged by researchers investigating communities that are able to sustain their resources through social norms or informal rules. These rules are extremely effective, especially when resource users are able to monitor one another's behavior and resource consumption (Anderies et al., 2011). Such internal institutional mechanisms devised by resource users themselves have inspired researchers from various disciplines, including social psychologists, sociologists, anthropologists, geographers, and economists. Without any external intervention by the government or regulatory agencies, how were communities able to enforce the rules? How were they able to establish long-lasting arrangements to successfully conserve resources? What were the characteristics of such communities, resources, and rules? How were they able to cooperate, despite strong material incentives to act selfishly? These questions and many others have been investigated by Elinor Ostrom and her colleagues from the 1970s onwards. Following up case studies with a series of laboratory and field experiments, researchers have shown that, subjects may be able to use the resources in a sustainable manner. This has provided substantial evidence that Hardin's tragedy is not inescapable (Dietz, Ostrom, & Stern, 2003; Ostrom, 1990).

One of the first notable experiments to provide a baseline for many other common-pool resource experiments was conducted by (Ostrom, Walker, & Gardner, 1992). While static and simple, this experiment captured the main incentive structure inherent in resource use dilemmas. University students were each given a certain endowment which they could allocate between two markets: they could divide their endowment between a market with a fixed return or a market in which the return was determined by the individual investment level as well as the aggregate investment level of the group. More specifically, this second market captured the non-excludability and rivalry of the resource use dilemma. The aggregate level of investment in the second market was announced each round. The individually-optimal level of investment in the second market would be higher than the amount that was socially efficient. If subjects could coordinate amongst themselves and reduce the group's total investment in the second market, then the total earnings would increase. Due to the incentive structure inherent in resource use dilemmas such as this one, subjects were not expected to be able to maximize total payoffs. In fact, as was expected by the theory, subjects were found to overinvest into the second market, and total investment in resource use reached Nash equilibrium in line with theory. Accordingly, the findings from the study were in line with Hardin's outlook—the tragedy of the commons.

4.4.1 Communication

When subjects were allowed to communicate face-to-face during the game, resource use decreased and departed from the Nash equilibrium towards social optimum. In fact, studies following this experiment have consistently arrived at similar results regarding the effect of face-to-face communication on cooperation: Allowing for non-binding face-to-face communication results in resource use that is closer to the socially efficient level than in the absence of communication (Ahn, Ostrom, & Walker, 2010; Balliet, 2009; Cardenas, 2004; Ostrom et al., 1992; Sally, 1995). In these and other experiments where the effect of face-to-face communication is examined, subjects are briefly allowed to talk about the game either once at the beginning or before each round of decisions on resource use. In fact, the mode of communication does not seem to matter: face-to-face, over a computer, or by passing notes. Any type of communication among subjects increases cooperation (Sally, 1995). On the contrary, in mainstream economics, non-binding communication is called "cheap talk." That is, since the introduction of communication did not change the incentive structure, it was assumed not to have any influence on behavior. Experiments involving communication, however, have proven consistently otherwise. Similar results have been achieved even if the subjects' initial endowments differ from one another (Hackett et al., 1994). However, when there are significant discrepancies in subjects' real-life personal economic wealth, the positive impact of communication on cooperation is somewhat reduced (Cardenas, 2003).

In another influential study investigating the effect of communication on cooperation (Falk, Fehr, & Fischbacher, 2000), the following hypotheses are put forth: First, the subjects tend to use communication as a coordination tool. In a game involving two equilibria, cooperation and over-extraction, subjects are able to coordinate at the cooperative equilibrium by communicating if they have a preference for reciprocity (i.e., if they respond to cooperation with cooperation and to greedy consumption with greed). As a result, the game might end at a level of extraction close to the socially-efficient one, which would generate higher earnings for the subjects. A second potential benefit of communication is that it can be used as a sanctioning device. The act of sanctioning occurs when subjects express their approval or disapproval of other subjects through communication. Individuals can communicate approval or disapproval of certain behaviors, making it possible for the game to achieve equilibrium at a point of cooperation.

4.4.2 Types of individuals

One common finding regarding cooperation and social dilemmas is that there are different types of subjects in each experiment. Usually, a significant portion of the subjects is made up of reciprocators (conditional cooperators), while a smaller fraction is composed of free-riders (Fischbacher, Gächter, & Fehr, 2001; Rustagi, Engel, & Kosfeld, 2010). There is also a small group of individuals who can be characterized as altruists. In a similar study, Rustagi et al. (2010) found a strong link between the share of conditional cooperators and real life forest management practices. In a framed field experiment they conducted in Ethiopia, groups with a larger share of conditional cooperators performed better in real-life forest management. By identifying conditional cooperators and selfish subjects, Carpenter and Seki (2005) were also able to show that a larger share of conditional cooperators makes groups more productive.

In fact, many recent experimental studies suggest that there is substantial intrinsic motivation to cooperate and sustain the resource at hand. Cardenas et al.

(2000), for instance, argue that resource users seem to seek a balance between self-interest and group-interest. This is based on the observation that subjects rarely over-extract resources in baseline treatments entailing no intervention. Accordingly, there seems to be pro-social and/or pro-environmental motivations at work (Rode, Gómez-Baggethun, & Krause, 2015).

Moreover, findings from common-pool resource experiments demonstrate that only a small number of subjects consistently engage in free-riding, while the behavior of a substantial number of subjects can be explained on the basis of reciprocity. It is presently not clear why subjects mostly refrain from free-riding, despite the fact that they are theoretically expected to do so and to use excessive resources. Explanations for this behavior are that subjects take the status of other group members into consideration, try to avoid inequality among group members, or place great value on the environment (Cardenas, 2011). While a robust overarching theory pertaining to this kind of behavior—termed social preferences or pro-social behavior—is not well-developed yet, new experiments in this field are expanding the empirical scope of the literature (Bowles & Polanía-Reyes, 2012; Fehr & Schmidt, 2003; Sturm & Weimann, 2006).

4.4.3 Crowding effects

The investigation of pro-social behavior usually goes hand in hand with another strand of literature on motivational crowding effects. This vast literature has been developed initially by social psychologists, although its original idea dates back at least to Titmuss' (1971) classic study on blood donation, as discussed in Chapter 3. The main idea is that individuals can be motivated intrinsically as well as extrinsically (Deci, Ryan, & Koestner, 1999; Deci, 1971). Intrinsic motivations are at work when an individual undertakes an activity because it gives them pleasure or satisfaction to do so. This may be based on the activity being fun or challenging, or because they hold a conviction about it. An individual is extrinsically motivated if an activity has instrumental value for them. That is, undertaking the activity provides a monetary or non-monetary benefit to the individual apart from the intrinsic pleasure of it. These two types of motivation may or may not complement each other. Bowles (2008) and Bowles and Polania-Reyes (2012) argue that they may act as substitutes or complements. Arguably, external interventions may undermine intrinsic motivations, a phenomenon referred to as "crowding-out" of voluntary behavior, whereas the reinforcement of intrinsic motivations is called "crowding-in." Since Bruno Frey's (1992, 1993, 1997) studies, experimental and behavioral economists have been interested in the interaction between motivations and economic incentives. This research agenda is especially important for environmental issues, since intrinsic motivations are usually relevant to nature conservation and resource use. Therefore, the effect of different economic incentives, such as fines, taxes, rewards, and subsidies, on resource use behavior is an important research area in the common-pool resources literature as well.

As mentioned, an economic incentive may reduce or strengthen intrinsic motivations to cooperate or to conserve a resource. That is, economic incentives and intrinsic motivations are not always separable as is usually assumed by public economics and mechanism design theories. Rather, they may interact, which is why economic incentives may be ineffective or even counterproductive at times. Bowles and Polanía-Reyes (2012) distinguish between three different mechanisms that may, at least temporarily, lead to crowding-out or crowding-in effects:

First, policies involving economic incentives can signal whether the regulator has good or bad intentions about the regulated individual. This is the informational aspect of economic incentives. The regulator may be self-interested or benevolent, and may be perceived as fair or unfair. Therefore, if economic incentives convey a message about the person or authority who designed them, they may lead to crowding-out or crowding-in.

Second, certain incentives may frame the decision situation. They may provide cues about what is appropriate or permissible behavior in a given decisional context. They may trigger self-interested behavior or economic reasoning, the so-called "moral disengagement" mechanism (Bandura, 1991), which supposes that "people can switch their ethicality on and off" (Shu et al., 2011, p. 31).

Third, economic incentives may crowd-out intrinsic motivations when individuals have a certain degree of "control aversion." For instance, an externally-enforced fine may reduce an individual's internal motivation to preserve the resource or their desire to keep the situation under control. Compromising the individual's autonomy, this can lead to resistance and, as a consequence, crowd-out voluntary behavior.

The fourth mechanism, endogenous preferences, consists of stable tendencies that may lead to crowding effects. Different economic incentives can change the way preferences are acquired and learned in the society. If the fraction of the population with pro-social behavior reaches a certain threshold, the kinds of preferences, selfish as opposed to pro-social, that individuals learn will change, and this effect may be persistent.

Vollan (2008) has identified and tested some other determinants of the crowding-out effect. Levels of in-group trust and self-determination, as well as whether the external intervention is perceived as controlling or supporting, have a significant influence on crowding effects. Focusing on the issues of resource use and biodiversity conservation, Rode et al. (2015) refined the psychological processes behind crowding-out and crowding-in effects. According to these authors, crowding-out effects arise when economic incentives lead to i) control aversion; ii) frustration; iii) reduced internal satisfaction (reduced "warm glow"); iv) reduced image motivation; v) release from moral responsibility; vi) frame shifting; and vii) changes in values or mindsets. Explanations follow for each of these seven factors:

First, intrinsic motivations to cooperate or to preserve a resource may be crowded-out when individuals feel they have a high degree of control over the external intervention. This is the control aversion aspect: Individuals react adversely to external interventions and economic incentives when they think that their capacity to act independently is compromised through the incentive or intervention (Bowles & Polanía-Reyes, 2012; Bowles, 2008; Frey & Stutzer, 2006).

Second, the introduction of an external regulation will frustrate individuals if they think that the intention behind the intervention is bad, untrustworthy, or unfair. This is the case when the intervention transmits a negative message or "bad news" about the regulator's intentions (Bowles & Polanía-Reyes, 2012; Falk & Kosfeld, 2005; Gawel, 2001). Third, economic incentives may reduce the "warm glow" effects or satisfaction one gets from conserving nature voluntarily (Andreoni, 1990; Bowles & Polanía-Reyes, 2012). Fourth, if people are motivated to project and maintain an image as a person with pro-environmental habits to others, this image may become indistinguishable to outside parties when an external intervention dictating environmental conservation is introduced, leading people to be less inclined to preserve nature or to conserve resources.

The fifth psychological factor is one of the most relevant to both environmental policies and nature conservation efforts. If an economic incentive signals that it is morally right to extract a resource or to pollute the environment as long as one pays for it, individuals may feel released from conservational duties or responsibilities. That is, the payments, be it in the form of a fine, tax, or the price of a marketable permit, may convey the message that it is one's "right" to harm the environment. Indeed, previous authors have noted the similarity between the Polluters Pay Principle and environmental "indulgences" (Goodin, 1994; Spash, 2010). The logic of marketable permits may invoke similar responses when individuals perceive the market context as one in which they are unable to change or influence the level of overall extraction: "If I don't buy the marketable permit, someone else will, so my behavior is not pivotal at all" (Falk & Szech, 2013; Frey, 1992).

The sixth factor has been labeled frame shifting or the framing effect. Economic incentives frame a decision situation by providing clues about appropriate behavior in each context. For instance, there are a few experiments invoking the market, as opposed to a community context, in which cooperation is more appropriate (Bowles & Polanía-Reyes, 2012; Liberman et al., 2004). Upon the introduction of certain economic incentive structures, economic reasoning may dominate (Heyman & Ariely, 2004). This may be closely related to the psychological factor of moral disengagement, which may be evoked by market-like incentives (Bandura, 1991), and market logic may dominate in instances where a more cooperative logic was previously in place. This may be the case for payments for ecosystem services (PES), where a certain level of intrinsic motivation involving environmentally-benign land use practices may have existed prior to the introduction of material compensation aimed at encouraging better conservation, which may actually shift the focus away from the community and towards more selfish and material concerns (Gomez-Baggethun, de Groot, Lomas, & Montes, 2010; Vatn, 2010).

Finally, the dominance of economic reasoning may influence values or mindsets, triggering more long-term effects through learning and adaptation. This may be the case when “incentives affect the environment in which preferences are learned and therefore the stationary distribution of preference types in the population (i.e. the fraction of the population with social preferences)” (Bowles & Polanía-Reyes, 2012, p. 376). This presents a long-term change in behavior and may be an indication of endogenous preferences.

It is also possible that economic incentives and intrinsic motivations work together. That is, they may act as complements rather than as substitutes, reinforcing rather than counteracting one another. Crowding-in seems to occur more often in public goods or common-pool resource experiments that involve more than three participants. That is certainly good news for policy makers who usually have to deal with a large groups of resource users when they impose a certain regulation (Bowles & Polanía-Reyes, 2012).

Several experiments provide evidence on the crowding-in of voluntary behavior. Crowding-in may occur when members of a group impose a fine on free-riders (peer punishment) (Fehr & Gächter, 2000) or simply express disapproval towards them (Barr, 2001; Masclet, Noussair, Tucker, & Villeval, 2003). Experimental evidence also suggests that low fines, which theoretically would not be sufficient to induce social optimum, can lead to the crowding-in of pro-social behavior (e.g. Lopez, Murphy, Spraggon, & Stranlund, 2012; Rodriguez-Sickert et al., 2008). Accordingly, even small, symbolic fines seem to "stabilize[…] cooperation by preventing a spiral of negative reciprocation" (Rodriguez-Sickert & Guzmán, 2008, p. 223). Similarly, Cardenas (2004) found that varying the size of the penalty imposed did not make much of difference: both high and low fines led to more cooperative behavior, the reason probably being that the presence of any fine is enough to signal that cooperation is the normatively-desirable behavior. In the same experiment, a self-governed mechanism entailing communication also led to the almost same level of cooperation (or crowding-in in this experiment). As mentioned, a well-established finding in the experimental economics literature is that communication leads to the crowding-in of voluntary behavior. Cardenas (2011) also found that external fines provide normative clues about appropriate behavior in different decision contexts. In that study, a treatment involving no fine, but with a public announcement of free-riders, led to crowding-in as well.

Rode et al. (2015) identify four psychological reasons for crowding-in that can be observed in resource use dilemmas:

- Enhanced internal satisfaction through social recognition: Resource users feel more motivated when they see that their conservation effort is appreciated;
- Reinforced positive attitudes or trust: Resource users are motivated to cooperate when economic incentives increase their level of trust towards regulators or support positive attitudes;

- Prescriptive effect: The incentive or the intervention in general signals morally correct behavior. This effect will be more permanent if it is able to change perceptions or norms in the long run.
- Reinforcement achieved by compelling non-intrinsically motivated individuals to comply: Incentives or interventions signal that free-riding is not tolerated and will be punished. Conditional cooperators will also reduce their extraction once they see that they will not be exploited by free-riders. Bowles and Polanía-Reyes (2012, p. 410) explain this process as follows: "The incentives and constraints typical of the rule of law and other institutional designs that limit the most extreme forms of antisocial behavior and facilitate mutually beneficial interactions on a large scale may enhance the salience of social preferences by assuring people that those who conform to moral norms will not be exploited by their self-interested fellow citizens." That is, "betrayal aversion" can be avoided when there are certain incentives that punish exploitative free-riding behavior (Bohnet, Greig, Herrmann, & Zeckhauser, 2008).

4.4.4 Peer punishment

Another finding consistently obtained in common-pool resource experiments is that subjects are inclined to punish uncooperative subjects in their group (peer punishment), even if this reduces their own payoffs (Ostrom et al., 1992). In experiments examining the effects of costly peer punishment, the individual extraction of each participant is announced each round but assigned to each subject on the basis of participant numbers. Each subject is then given the opportunity to punish another participant in the group if he/she does not approve of the amount of that person's resource use. However, it is also costly for the participant to impose the punishment. The cost of punishment to the punishing individual and the amount of penalty are fixed and known at the outset, and each participant can be fined by more than one participant each round. Once individual resource use amounts have been announced at the end of each round, participants are given a penalty form. After the penalty forms are collected, each participant learns whether he or she was fined or not and, if so, how much total penalty will be deducted from his or her payoff. Participants are not told which group member(s) fined them. Usually, penalties are about two-to-four times more than the cost to fine someone.

Contrary to theoretical predictions, the common finding from these experiments is that individuals who use a resource excessively are often fined by the other participants. Of course, due to the cost of punishment, average payoffs are reduced. However, resource use on average decreases; in other words, cooperation increases (Fehr & Gächter, 2000). Conversely, in a public goods game experiment without any framing, Gächter and Herrmann (2011) found that even with a great deal of peer-punishment incidents, peer-punishment did not effectively improve cooperation. Rather, it seemed to crowd-out other-regarding behavior. Visser and Burns (2015) demonstrated that when resource users are asymmetric with respect to their endowments in the game, unequal groups are more cooperative on average than equal groups in the context of peer punishment games. In a public goods game without any framing that was undertaken with rural subjects in Russia, peer punishment did not improve cooperation (Gächter & Herrmann, 2011).

4.4.5 Fines, taxes, subsidies and rewards

When a fine is charged not by subjects within the group but by an external party, findings can differ substantially. A modestly-enforced weak external fine was found on average to crowd-out other-regarding behavior (Cardenas et al., 2000). In similar studies, average extraction fell upon the introduction of the externally imposed fine, but then increased again (Lopez et al., 2012; Velez, Murphy, & Stranlund, 2010). Gatiso et al. (2015) also observed crowding-out upon the introduction of an external fine when the resource was abundant. However, when a fine was introduced under the scarcity treatment, crowding-out did not occur. In contrast, findings also show that external fines may increase cooperation on average (Anderies et al., 2011; Cardenas, 2011; Castillo & Saysel, 2005; Handberg & Angelsen, 2015). When the enforcement is stricter, or the probability of being caught increases, cooperation increases (Rodriguez-Sickert et al., 2008). Another framed field experiment in Colombia demonstrated that a weakly-enforced, a low as well as a high fine was effective in reducing extraction (Cardenas, 2004). Rodriguez-Sickert et al. (2008) argue that low fines are effective in reducing extraction levels because they “prevent[…] a spiral of negative reciprocation” (p. 223). Consequently, the effects of externally imposed fines are still under investigation.

Framing a material disincentives as a tax seems to reduce average extraction (Castillo & Saysel, 2005). In another experimental study, a collective tax was introduced in a framed field experiment among subsistence resource users (Reichhuber, Camacho, & Requate, 2009). The authors found that when a symbolic, low tax was imposed collectively on the group, a significant level of cooperation occurred and then disappeared over the course of the game. Yet, since there was no baseline treatment, it is not clear how this collective tax changed average behavior.

Another significant area for research on cooperation and resource use concerns the effects of rewards on resource extraction. One of the reasons why rewards are seen as important stems from the popularity of payments for ecosystem services. In some other experiments, rewards are framed as subsidies or payments to encourage users' compliance with a use quota. Such experiments on rewards, however, seem to have provided mixed evidence. For instance, Narloch et al. (2012) demonstrated that collective rewards reduce cooperation, while individual rewards are more effective at strengthening conditional cooperation. In line with this finding, Midler et al. (2015) found that individual rewards are more effective than collective rewards at improving cooperation. However, they also demonstrated that collective rewards are effective when combined with face-to-face communication. On the other hand, Kerr et al. (2012) conducted field experiments to test participation in real community tasks under different reward schemes, which suggested that framing rewards as PES seems to reduce intrinsic motivation to preserve the resource. Vollan (2008) found that rewards increased cooperation in the sub-set of the sample with higher levels of trust. Jack (2009) tested the effect of weak compensation paid by upstream resource users to downstream ones and found that the other-regarding behavior of upstream users was crowded-out under the temporary enforcement of compensation payments.

4.4.6 Market-based incentives

So far, the effect of market-based incentives on resource use has not been sufficiently investigated. Considering the recent dominance of environmental markets for common-pool resources, such as climate, fish stocks, and water use rights, it is important to experimentally investigate the effects of market-like mechanisms on cooperation and resource use. However, there are several difficulties with trying to examine this issue in a field experiment with actual resource users. First, a high level of cognitive capacity and some familiarity with such markets are needed. Uncovering individually-optimal behavior in permit markets is a rather difficult task, especially if subjects are unfamiliar with this type of experiment. Second, a double auction mechanism is usually applied in experiments investigating environmental markets. However, some theoretical aspects of double auctions have still not been clearly examined (Sturm & Weimann, 2006).

A recent paper published in *Science* by Falk and Szech (2013) suggested that market-like mechanisms may undermine moral behavior in general.[34] On the other hand, familiarity with market interactions or priming with market language has been shown to lead to crowding-in. In the setting of a trust game, subjects primed with words related to markets and exchange were found to be more trusting towards their partners in the game (Al-Ubaydli, Houser, Nye, Paganelli, & Pan, 2013). In an ultimatum game, subjects from market-integrated societies gave more to their partners (Henrich et al., 2005). Thus, there remains a need for further studies on market-like incentives. Another recent lab experiment involving a public goods game and a competitive tender demonstrated that market institutions make individuals behave in a more self-interested manner (Reeson & Tisdell, 2010).

4.4.7 Ecological dynamics

While the basic findings related to resource use can be achieved through simple and static experiments, recent experimental studies have attempted to account for the

[34] The study is controversial since it involved a rather unnatural setting (saving the life of a mouse via payments) and is in need of cross-cultural validation.

dynamic ecological characteristics of a resource as well (Janssen, 2010; Kimbrough, 2013; Muller & Vickers, 1996; Walker & Gardner, 1992). Given that most common-pool resources are at risk of depletion, it is important to include these dynamics into the game. However, such experiments have not yet delivered conclusive results regarding resource use (Cardenas, Janssen, & Bousquet, 2013). Experiments that reflect some of the spatial and biological peculiarities of the resource at hand have demonstrated that resource users seem to follow strategies based on their own experiences in the field (Castillo, Bousquet, Janssen, Worrapimphong, & Cardenas, 2011), but there is also some evidence that they may behave similar to student subjects in the lab or in other static field experiments (Janssen, Anderies, & Cardenas, 2011).

4.4.8 Framing and subject pools

The interaction of frames with decisions has been investigated recently in the behavioral economics literature, especially for public good games. Since Deutsch's (1958) study, it has been well documented that social frames influence the decisions of individuals. Even the name of the game may affect the average level of cooperation in public goods games. For instance, calling the game an “Investment Game” (or a “Wall Street Game”) as opposed to a “Community Game” changes the results (Kay & Ross, 2003; Liberman et al., 2004). This finding is called “label framing” or the “pure framing effect,” and has been observed in cases where the material incentives and the relevant reference points are held constant (Dufwenberg et al., 2011; Elliott, Hayward, & Canon, 1998). Recent studies have demonstrated that such framing effects are closely related to changing beliefs about other subjects’ behavior (Dufwenberg et al., 2011; Ellingsen, Johannesson, Mollerstrom, & Munkhammar, 2012). That is, if subjects believe that others will behave selfishly (such as in the case of the Wall Street Game), they will decide to act selfishly as well. On the other hand, if the subjects believe that other subjects will cooperate, they will also cooperate. In other words, frames may change one’s beliefs about others, and those beliefs may in turn change one’s behavior. However, so far, the framing effects have not been investigated in resource use experiments.

There is strong experimental evidence that student subjects are less pro-social and cooperative than non-students (e.g. Cardenas, 2005; Fehr & List, 2004). A number of studies have compared subjects who experience dilemmas in real-life situations (such as fishers or villagers whose livelihood depends on forestry) with student subjects. Cardenas (2011), for instance, compared villagers who used the resource in their daily lives with university students in a common-pool resource game, and found that while the results pointed in the same direction, their magnitudes varied. Similarly, Carpenter and Seki (2005) demonstrated that fishers were more cooperative than student subjects in a voluntary contribution mechanism.

4.4.9 Framed field experiments

Providing a context that resembles the real-life experience of subjects is important for experiments in general and for resource use and environmental policy experiments in particular (Cherry et al., 2008; Eckel, Grossman, & Johnston, 2005; Harrison & List, 2004). In laboratory experiments where the game and the rules are presented to the subjects in an abstract, context-free manner, subjects may perceive the game differently based on their prior experiences, which can weaken the internal validity of the experimental manipulation. The game would then turn into another game unintended by the researcher. Consequently, several researchers emphasize that making concrete references to context—especially in framed field experiments—will improve both the internal and external validity of such experiments (Sturm & Weimann, 2006). This also explains why researchers generally prefer framed field experiments when it comes to resource use dilemmas.

Appendix A provides an overview of numerous framed field experiments on resource use and cooperation. As can be seen from the table in Appendix A, a diverse set of aspects have attracted researchers' attention. The effect of penalties, rewards, taxes and subsidies, peer punishment, reputation (guilt, shame, or pride), voting on different types of institutions, communication, co-management among resource users, resource availability, and spatial and biological resource dynamics are all important dimensions that have been investigated so far. In all of these experiments, the choice set of the subjects and the task itself were framed according to the characteristics of the resource in question. Fishing, harvesting wood from the forest, land use, grazing, water use, irrigation, and conservation of biological resources are the most common contexts for resource use experiments. Most of these experiments have been conducted with rural communities in developing countries such as Colombia, Peru, South Africa, and Namibia. Some of the experiments addressing conservation of biodiversity or land use practices explicitly referred to PES rewards, probably due to the popularity of these new mechanisms in environmental policy in developing countries. Public goods games and common-pool resource games were widely used in the experiments. Some experiments were explicitly conducted to assess the effect of context variables and framing (Bouma & Ansink, 2013; Castillo et al., 2011), while others tested external validity (Gelcich, Guzman, Rodrigues-Sickert, Castilla, & Cardenas, 2013) or explicitly addressed real-world policies, such as co-management, command-and-control, and PES (Handberg &

Angelsen, 2015; Saldarriaga-Isaza, Villegas-Palacio, & Arango, 2015). One study measured the fraction of conditional cooperators and free-riders among resource users (Rustagi et al., 2010). Another recent study used social network measures to test links between network centrality and other-regarding behavior (Mantilla, 2015a). Overall, a robust finding demonstrated by many of these studies is the positive effect of communication on cooperation (e.g. del Pilar Moreno-Sanchez & Maldonado, 2010; Ghate, Ghate, & Ostrom, 2013; Handberg & Angelsen, 2015). Another common finding is the positive effect of externally-enforced, low (symbolic) penalties and public announcements of harvest levels (Cardenas, 2004, 2011), although exceptions exist (Cardenas et al., 2000; Lopez et al., 2012). The effect of peer punishment is another interesting question that has been addressed by these studies (e.g. Gächter & Herrmann, 2011). Other studies have combined peer punishment with scarcity (Gatiso et al., 2015) as well as with heterogeneous endowments in the game (Visser & Burns, 2015).

4.4.10 Internal and external validity of experiments on resource use

Achieving high internal validity in resource use experiments remains a challenge for researchers. Fines applied toward excessive resource use at times may decrease collaboration and at times may increase it, but the reason for this is still unclear. People may be interpreting the game differently, or there may be another behavioral reason for it. Nevertheless, one can argue that common-pool resource experiments enable us to see that fines may in fact lead to mixed outcomes, a finding which has repercussions for real-life policy choices. Still, experimental studies continue to examine the reasons for these seemingly conflicting findings (Sturm & Weimann, 2006).

The external validity of common-pool resource experiments, or whether the experimental findings can be generalized to real-life situations, is also still being debated. Fehr and Leibbrandt (2011), for instance, measured how inclined fishers and shrimp catchers were to collaborate in a series of public goods experiments and then compared these results with real-life behavior. Individuals who collaborated more in the public goods game were more likely to use tools or devices that depleted fish and shrimp stocks at a slower pace in the real-life common-pool resource problem.[35] Rustagi et al. (2010) also found resource use experiments to have external validity. Moreover, they found that groups with more conditional cooperators were more successful in real-life forest management activities. Carpenter and Seki (2005) found that the subjects who had contributed more in a public goods game were also more likely to be members of fishing cooperatives, which were associated with more sharing and cooperation, than their counterparts. In contrast, some scholars claim that behavior in an experiment can never be sufficiently predictive of real-life behavior (e.g. Gurven and Winking, 2008). Nonetheless, common-pool resource use experiments continue to be a fruitful research area. New experiments that allow for comparisons with behavior outside the experimental setting will without a doubt continue to shed light on questions of external validity.

[35] Shrimp catchers who collaborated more in the game used bigger mesh nets in their real lives, which enabled smaller shrimp to escape from the net and grow bigger. So although they caught less shrimp, their behavior reflects a cooperative approach that will prevent the species from going extinct.

4.5 The value-added of experimental studies on environment

Summing up the chapter, it can be concluded that behavioral and experimental economics aligns with environmental economics in three main areas—environmental valuation, environmental policy instruments, and resource use dilemmas. This review of the broad literature on this subject provides several takeaways.

First, experiments on environmental valuation and contingent valuation surveys complement one another. Experimental studies have ascertained the weaknesses of CV surveys, understood why they came about, and improved upon them. The fact that it is not possible to directly observe the values people ascribe to the environment makes it clear that valuation methods such as CV surveys and the experiments used in conjunction with them are important.

Second, setting up experiments to test environmental policy instruments (direct regulation, taxation, or market-based methods) and how changes in these mechanisms influence human behavior has been a real challenge, as it is relatively difficult to replicate an institutional mechanism within a laboratory setting (Sturm & Weimann, 2006). Still, experiments on environmental policies have provided significant insights that could not have been gleaned from real-life experiences or other empirical studies.

Third, although the real-life dilemma that resource users' experience cannot be exactly reproduced in an experimental setting, experiments can provide us with strong clues as to the kinds of behavioral responses that people might give when faced with such a dilemma. Again, it is thanks to experiments that we are better able to understand certain factors, such as communication and peer punishment, can prevent excessive resource use.

Meanwhile, we need to better analyze how subjects perceive the game in experiments on common-pool resources and environmental policies, and which real-life experiences they liken it to. To this end, many researchers emphasize the need to provide subjects with sufficient information about the context of resource use. To obtain more generalizable results, experimenters sometimes prefer to present the game to the subjects in neutral language that does not offer many clues, thereby hoping to have better control over the experiment. However, if the context is not thoroughly described to the players, then the chances are greater that they will bring their own contexts to the game and make decisions accordingly. This could actually decrease the level of experimental control. Given that the normative dimension is extremely important for environmental policies and resource use dilemmas, specifying the context of experiments will strengthen these studies. One advantage of experimental methods is that they allow for different combinations of control and context via different treatments. This makes it possible to determine the extent and direction that context influences players' preferences. Moreover, subjects should be selected from among the people who will be affected by the environmental policy that is likely to be put into effect. In this sense, field experiments are becoming increasingly important for environmental economists. Moreover, it is not difficult to design field experiments in a way that takes the factors described above into account (Cherry et al., 2008).

In sum, thanks to experimental and behavioral economics, it is possible to better understand how institutional interventions put forth to resolve environmental dilemmas affect human behavior. Experimental methods can reveal the links among institutions, incentives, and behavior, and offer predictions as to how successful various policies will be in resolving the problem at hand. As such, Part II of this dissertation builds on the review provided in this chapter, especially on the empirical findings of experimental studies on environmental policy instruments and resource use dilemmas, and presents the findings and insights gained from the framed field experiments conducted with small-scale fishing communities in Turkey.

PART II

EVIDENCE FROM THE FIELD:

FRAMED FIELD EXPERIMENTS

WITH SMALL-SCALE FISHING COMMUNITIES IN TURKEY

CHAPTER 5

BACKGROUND OF THE FIELDWORK: THE ROLE PLAYED BY THE STATE, INSTITUTIONAL ARRANGEMENTS, AND CONFLICTS AMONG RESOURCE USERS IN FISHERIES IN TURKEY

This chapter will introduce, as a background, the institutional context of the fieldwork conducted with fishing communities in Turkey. First, a history of fisheries management in the country will be provided as an overview of the role played by the state in fisheries. Next, specific local, national, and international institutional arrangements will be discussed. Finally, some key information on the current tensions and conflicts between different resource users in fisheries will be presented.

5.1 A brief history of fisheries in Turkey and the role of the state

Fisheries are an important source of food and income for the citizens of Turkey, and play a quite significant role not only in the country's economy, but also in its culture. Even before the Republic, income from fish resources was prone to state regulation; fishing was a source of tax revenue for the Ottoman State and a food source for the Sultan and his palace. With the Tanzimat Reforms in the 1880s, ideas of economic progress began to make their way onto the state's agenda. However, this was also a time when the Ottoman State had a significant amount of foreign debt, which was administered by the *Düyûnu Umumiye* (Public Debt Administration) to the benefit of foreign credit sources. From the 1880s onwards, part of the funds managed by the *Düyûnu Umumiye* originated from tax revenues collected from fisheries, mainly those in the Marmara Region. A number of foreign and local experts began to express ideas related to economic development and modernization; for instance, Karekin Deveciyan, director of the Istanbul Fish Hall in the early 1900s and author of the first book on fisheries in the Ottoman State and Turkey (Deveciyan, 2013). Until the mid-1930s, however, these ideas never really made it onto the state agenda for fisheries.

The period between the 1930s and 1950s marked a break from the previous era in terms of the focus of taxation, in that government officials started to adopt a developmentalist discourse for fisheries. There were plans for a new fisheries law that would promote trade, production, and consumption; the establishment of a specialized bank to support fisheries; the abolishment of taxes on fishing incomes; a reduction in tariffs for fishery equipment; and the elimination of middlemen in fish sales. While these discussions reflected the statist ideals of the new Republic, implementing most of these plans had to wait until the 1950s, probably because bureaucrats were not decisive and consistent enough in their actions. So from the early years of the Republic up until the 1950s, the fish catch declined instead of increasing. This can be traced back to the fact that the population and economy was stagnant in the aftermath of the War of Independence, and that the Greek communities who had been primarily occupied with fishing in the Marmara region had emigrated (Knudsen, 2009).

After WWII, government focus began to shift away from a self-sufficient economy—based on heavy industrial development and the construction of railways—to agricultural development and road construction. These new developments were all made possible by the Marshall Plan, and fisheries attracted much attention as part of the agricultural development ideal. While the Marshall Plan provided funds to the Turkish fishing industry (Tören, 2007), government authorities lifted the taxes on fishing income prior to the 1950 parliamentary elections. *Et ve Balık Kurumu* (The Meat and

Fish Authority) was founded in 1952 to both manage meat and fish resources, and contribute to the scientific research of marine resources. This new institution was also supported financially by the Marshall Plan, and collaborated with the Food and Agriculture Organization (FAO) and U.S. scientists. During this period, the General Directorate for Agricultural Credit was founded, which offered loans to the agriculture and fishing industries. Survey-based data collection on fisheries began in 1955, and findings were published annually from 1967 onwards (Acara, Mert, Şenel & Atik, 2001). The state started constructing new and large harbors on Turkey's Black Sea coast, and endorsed the establishment of fishing cooperatives. However, these measures proved insufficient to develop fisheries and increase the fish catch in the 1950s and 1960s, while plans to regulate the influence of middlemen in the fish trade were never realized.

Fisheries became part of the Five Year Development Plans from 1963 onwards, and responsibilities related to the fishing industry was transferred from the Ministry of Trade to the newly established *Su Ürünleri Genel Müdürlüğü* (General Directorate of Marine Products) within the Ministry of Agriculture. The Ministry of Agriculture began to prepare a set of regulations for fisheries annually. The Law on Marine Products was passed in 1971 to regulate commercial and recreational fisheries (Ünal & Göncüoğlu, 2012). At universities, marine science was developing—albeit rather belatedly compared to their European counterparts—with the support of the Marshall Plan, the FAO, a Turkey-Japan cooperation network, and the contributions of foreign scientists employed at Istanbul University who had escaped from the Nazi regime in Germany (Bilecik, 2003, as cited in Knudsen, 2009). Scientific research was conducted at public universities and State Fisheries Research Institute, and as such, the state played a major role in the scientific advancement of fisheries as well.

Parallel to these developments, the discourse on fisheries changed drastically compared to both the earlier periods of the Republic and the later periods of the Ottoman state. Concepts like modernization, economic development, technological progress, production, and proteins started to dominate the language concerning the fisheries in the country. The western world (especially European countries) and Japan were the role models in this regard; the term "marine products" replaced the word "fish" to refer to the fish caught. The modernization ideal appeared in the "national task and necessity" to modernize and develop fisheries with a "rational, scientific, and economic" management strategy (Özbey, 1989, p. 5, as cited in Knudsen, 2009). The traditional ways of catching fish were regarded as "backward" and in need of transformation; accordingly, fisheries emerged as an economic "industry" that needed to be measured (in terms of per capita consumption and production), assessed, managed, and researched, just like the agricultural industry. The focus on tax revenue shifted from what it was in Ottoman times, to modern and rational economic policies geared to boost "the production of marine products" instead of "catching" the "fish" of previous periods. So fisheries that were once a source of government revenue became an industry that received public investment, subsidies, and loans. The income tax on fishing was reduced from 20 percent (as it was during Ottoman times) to a maximum of 3 percent, which was effectively only paid by a limited number of rather wealthy fishers (Knudsen, 2009).

The establishment of agricultural and fishing cooperatives was another western ideal of the state. The ideal first appeared in the draft plans for a fishing law in the

1930s and 1940s. Fishing cooperatives were established in the early 1950s, mainly due to the promise that fishers would receive financial help from the Marshall Plan if they organized as cooperatives. However, cooperatives were not the main focus of *Et ve Balık Kurumu*, which focused more on technical developments to promote greater fish production. Consequently, during the first wave of fishing cooperatives, most either soon shut down or changed hands from local fishers to middlemen (Çakıroğlu, 1969). Later, towards the late 1960s, the state decided to activate the cooperatives via the Ministry of Trade, especially in the Black Sea Region, and the *Karadeniz Balıkçı Kooperatifleri Birliği* (the Association of Black Sea Fishing Cooperatives) was founded. Thereafter, cooperatives took off in terms of number and use of state loans (Acara et al., 2001). The revival of fishing cooperatives was again state-led and rather top-down, similar to many other modernization processes in Turkey. This is in contrast to, for instance, the fishing cooperative movement in Nordic countries and Japan, which were mainly based on community rights and the local participation of fishers (Pomeroy & Berkes, 1997).

Nevertheless, the role of the state in fisheries remained, arguably, quite weak and ineffective until the late 1970s (Knudsen, 2009). One of the reasons was that state intervention in fishing occurred rather late compared to other industries such as agriculture. While agriculture received strong political and financial support from the government from the 1930s onwards, state-led policies to support certain industries was no longer popular in the liberalization era of the 1950s. As a result, fisheries were unable to directly benefit from supportive public policies in this period. Further, until the 1980s, the main focus of state policies was import-substitution, but since seafood was not an import good in the country, fisheries did not attract much attention.

Difficulties in managing the fish stock and fishers was probably another reason. In contrast to agricultural products, both the fish and the fishers were very mobile, and thus not easy to monitor and control. Moreover, the fish trade required well-organized storage and transportation facilities, and it was rather problematic for state bureaucrats to ensure these conditions.

The fish catch began to rise from the 1970s onwards (Tokaç et al., 2014), thanks mainly to *gırgır*s (purse-seiners) and *trol*s (trawlers) equipped with the latest technology—it seemed as though the state's investments in big fishing harbors, and subsidized credits provided to fishers, were finally paying off. Moreover, the state supported privately-owned fishmeal and oil factories with large subsidies, and offered them import tax exemptions from the 1970s onwards. This increased the number of factories where surplus catch (or fish below the legal size limits) could be processed, and paved the way for large-scale fishing fleets to emerge in the Black Sea, especially for anchovy, since the factories made large-scale fishing operations more profitable (Doğan, 1982; Knudsen, 2009). In parallel, the demand for fresh fish was on the rise and the transportation infrastructure was improved; as a result, several regional fish markets united in a national market for fish. In addition, aquaculture (fish farming) was introduced in this period to meet the increasing domestic demand for fish.[36]

The collapse of the fish stocks in the Black Sea, from 1989 to 1992, was a turning point in terms of the management and the science related to fisheries in Turkey

[36] A significant amount of farmed fish is exported today (FAO, 2005; Giannetto, Acar, & Demir, 2014). Government subsidies for farmed fish, initiated in 2003, have contributed substantially to this fact (Balcı Akova, 2015).

(Knudsen, 2009)[37]. The collapse finally forced marine scientists in the country to look at the sustainability of fish stocks, somewhat behind worldwide scientific trends. Bio-economic models and studies on fish stocks became more common, as it was soon realized fish harvests cannot be increased infinitely. So scientific studies shifted away from technological developments that aim to boost harvests, towards stock assessment and population dynamics models that investigated the bio-physical and economic relationship between fishing efforts and catch.

Overall, the state played a major role in both modernizing and developing large-scale fisheries, and institutionalizing marine research at public universities. It was very influential especially in codifying laws, developing plans, collecting data, conducting research, and monitoring fisheries. Based on its ideals of modernization, Westernization, and economic development, it intervened to generate income for fishers and food for its citizens. However, this implied that the state basically ignored the traditional local knowledge of fishers. Alternative models based on participation and co-management were not considered viable development models. This was in line with the overall outlook of the state in Turkey and its top-down, paternalistic economic development policies (Adaman & Arsel, 2005). Further, the prospect of EU membership motivated the state to align relevant policies and infrastructure with the Common Fisheries Policy of the EU; albeit with limited success so far (European Commission, 2015). Yet one important result was that sustainability issues and the environmental

[37] Fisheries in Norway and Newfoundland had already experienced a similar collapse (in the 1960s and 1970s-to-1980s, respectively). From this point on, scientific studies began to address the sustainability of fish stocks.

impact of fisheries were gradually addressed as part of the discourse in state policies and made it onto the marine research agenda at universities (Ünal & Göncüoğlu, 2012).

5.2 Current institutional arrangements and policies concerning fisheries in Turkey

Overfishing and overcapacity are the two main problems of fisheries management today. Since the 1950s, fish stocks have been declining at an accelerated rate. The exploitation of marine resources peaked in 1996, reaching 86.4 million tons, and has been following a downward trend ever since. As of 2013, stock assessments show that 31.4 percent of all fish stocks are overexploited, and 58.1 percent are fully-exploited; only 10.5 percent seems to be under-fished. The rate of under-fished resources has been steadily declining since the 1970s (FAO, 2016d). The main goal of fisheries management is achieving sustainability, since this is crucial for the livelihood and nutrition of millions of people in the world. Global per capita fish consumption was around 19.7 kg in 2013, and estimated to be over 20 kg in 2015, while it was much lower in the 1960s (9.9 kg). However, this figure varies considerably from country to country. For Turkey, it was 6.3 kg in 2013 (Turkish Statistical Institute [TURKSTAT], 2014).

In the period from the 1970s up to the 2000s, Turkey experienced a rapid expansion in fish catch. To facilitate this process, the government provided considerable subsidies and other incentives such as tax exemptions during this time. The fishing fleet in Turkey grew with the help of state support, in terms of length, number, and engine power. However, this has led to problems of overcapacity and overfishing in parallel to world fisheries (Tokaç et al., 2014). Today, both artisanal (small-scale) and industrial (large-scale) fishers are facing various problems that have been exacerbated by conflicts between different groups of resource users, decreases in catch per unit of effort—leading to a substantial increase in fishing efforts—and problems with regulation enforcement. Moreover, the absence of reliable stock estimates and fisheries data, and a lack of participatory planning have made these problems even worse (Ünal & Göncüoğlu, 2012).

In this context, successive governments aimed to regulate the fisheries in Turkey at the national scale, mainly by using command-and-control type policies, such as technical specifications, closed seasons, size and area restrictions, and quotas for a few species. Most of these regulations can be traced back to the Fisheries Law of 1971. However, the chronic failure to enforce them has rendered them quite ineffective. A similar observation has been made for regulations in the environmental sphere in general (Adaman & Arsel, 2005). Accordingly, although the legal infrastructure to address environmental problems is well-developed, enforcement issues render environmental policies ineffective. Problems of corruption and the patronage mentality further prevent effective enforcement in general. At the international level, the prospect of EU membership has been influential, and governments signed several international agreements on environmental protection in general, and the protection of marine resources in particular. Institutional arrangements at the local level have been rather under-developed, and participatory decision-making methods are either largely absent or have remained rather symbolic so far, in other realms of environmental decision-making as well. A brief discussion regarding the institutional context and the specific arrangements for fisheries in Turkey is provided below.

Small-scale fisheries employ around 90 percent of all fishers in the world, while providing approximately 10 percent of the total fish catch. Although there is no precise definition of small-scale fishers, boats shorter than 12 meters in length—excluding trawlers and draggers—are usually considered small-scale fishing vessels (European Commission, 2016). In the EU, the proportion of small-scale boats in the fleet varies from country to country, between 70 percent (Italy) and 98 percent (Finland), with an EU average of 82 percent. These small-boats are usually family-run, and have few or no employees; their gears put less pressure on ecosystems; and fish are usually caught close to the home harbor and sold in local markets. While small-scale fisheries constitute 47 percent of total employment in fisheries, they represent only 14 percent of the value of landed catch. The European Commission requires EU countries with a thousand or more small-scale vessels to create action plans for the development of small-scale fisheries.

Similarly, in Turkey, small-scale vessels constitute around 90 percent of the whole fishing fleet, while its total catch amounts to less than 10 percent. These vessels are usually between 5 and 12 meters long, with an engine power between 10 and 25 HP. Most of them are wooden vessels and use equipment such as gillnets and longlines. Large-scale industrial fishers, on the other hand, are the trawlers and purse seiners with bigger boats and stronger engines. Recreational fishers constitute yet another group of fishers consisting of both amateur and sport fishers. Their activities are regulated by the Fisheries Law No. 1380, yet there is presently no reliable data on what and how much they catch (Ünal & Göncüoğlu, 2012).

According to FAO figures, there were 16,988 licensed vessels in 2012 in Turkey, of which 8-to-10 percent were inactive (FAO, 2016a). Of the active vessels, 669 (4 percent) were trawlers, 610 (3.6 percent) were purse-seiners, and 15,540 (91.4 percent) were small-scale fishers. In 2013, a total of 53,799 individuals were employed in the fishing industry (including both full-time and part-time work) in the harvesting, aquaculture, and processing sectors (OECD, 2016). In 2014, the total export of fish products amounted to US $700 million, while imports were around US $375 million in the country. The contribution of the fisheries to Turkey's GDP (in constant prices, base year=1998) amounted to only 0.3 percent in 2009 (TURKSTAT, 2010).

The mean annual income of the fishers in Turkey was around $9,500 in 2012, higher than the official minimum wage of $5,904. However, for small-scale fishers, this figure is well below the minimum wage ($4,000) (FAO, 2016a). On average, small-scale vessels went fishing for 117 days in 2012, and landed 1.5 tons of fish by using one ton of fuel in 2012. The major part of their costs consisted of energy (33 percent) and other activity costs (33 percent), while labor costs constituted 17 percent and fixed costs 18 percent of total costs. For purse-seiners and trawlers over 12 meters in length, the major source of costs were attributable to labor costs and energy costs (57 percent and 46 percent, respectively).

In general, in fisheries management, two main approaches dominate the academic literature and policy-making. The approach based on the traditional local knowledge of fishers, participation, and co-management lies at one end of the spectrum, while management based on bio-economic models, which makes use of concepts such as total allowable catch, quotas, and individual transferable quotas (ITQs), lies at the other (Knudsen, 2009). While the former is more applicable to small-scale fisheries, the latter is usually assumed to be more appropriate for large-scale industrial fisheries. Both of these approaches rest on the ideas of sustainability, as opposed to the focus of the first half of the twentieth century where academics and policymakers alike mainly concentrated on technological improvements to increase the fish harvest. The bioeconomics approach is further divided into three subgroups: i) output controls (total allowable catch [TAC][38], vessel catch limits, individual fishing quotas,[39] and individual transferable quotas[40]), ii) input controls (license limits, individual effort quotas, other gear restrictions) and iii) technical measures (restrictions on size and species of the catch, time and area closures). Co-management, on the other hand, is a participatory form of fisheries management that formally provides opportunity and power to different stakeholders and user groups to take part designing and implementing fisheries policies; as such, it is considered a new and promising approach to fisheries management (OECD, 1997b).[41]

5.2.1 National institutional arrangements

In Turkey, institutional arrangements regarding fisheries at the national level consist of a combination of output controls, input controls, and technical measures. Some of the main instruments used by the regulatory authority include limitations on the minimum landing size of fish, minimum mesh size, minimum depth, vessel size, and fishing gear, as well as regulations regarding the catch of certain species, protected areas, closed areas, and vessel licensing. While some of these instruments are directed towards certain regions (e.g. closed areas), others apply to the country in general (e.g. the ban on

[38] These are national catch limits on certain commercial marine resources, which can be considered national quotas (European Commission, 2016). Applying TAC alone, without any complementary instruments, leads to a race-to-fish (FAO, 2016b)
[39] This instrument divides the total TAC per country and assigns each country a fixed number of vessels, for instance. This reduces the race-to-fish. Its success depends on the rate of TAC and enforcement issues; however, it does not provide incentives to reduce overcapacity.
[40] Making individual quotas transferable, the instrument has the advantage of reducing the fleet's capacity, since inefficient fishers or vessels will sell their quotas to efficient ones. However, this puts immense pressure on existing vessels with a quota, as under an open-access regime (FAO, 2016b)
[41] Analyses of the pros and cons of different instruments used in fisheries management abound in the academic literature and policy papers (e.g. Copes & Pálsson, 2000; Copes, 1996; OECD, 1997b), but a detailed analysis is beyond the scope of this manuscript.

certain types of nets). Each vessel involved in commercial fishing needs to be registered and have a fishing license, and the fishers themselves need to have a fishing license. The regulatory authority has stopped licensing new vessels since 2002; however, there are presently no effective instruments that limit the fishing effort (Ünal & Göncüoğlu, 2012). For instance, there are neither specific regulations that prescribe the ports at which fish have to be landed, nor any regional restrictions for vessels. Vessels registered at harbors in the Black Sea or the Sea of Marmara can freely catch fish in the Aegean or the Mediterranean.

In the process of adapting to EU regulations, the regulatory authority initiated a buy-back program in 2012 to reduce the number of vessels. In the first phase of the program, about 20 percent of the vessels above 12 meters were bought back by the state authority. In the second phase, initiated in 2013, the program was expanded to include all vessels above 10 meters (Ünal & Ekmekçi, 2015). The buy-back program was criticized for not being appealing to large-scale vessel owners—which constitute the largest part of the pressure on fish stocks—and for being initiated without first consulting with marine researchers. Its effectiveness in terms of reducing overcapacity is questionable as well. A survey study conducted with a representative sample of the fishers during the second phase of the program revealed that a significant number (27 percent) expressed their desire to buy a bigger boat with the payment received from the buy-back program. Another non-negligible share of the fishers (28 percent) stated that they transferred their second boat to the state authority within the buy-back program, which was inactive and not profitable prior to the program anyway (Ünal & Ekmekçi, 2015).

Currently, there is a TAC and an individual quota system in place for two marine species: the Bluefin tuna and striped Venus clam. The Ministry of Food, Agriculture, and Livestock is planning to expand the TAC system to include anchovy and mackerel (Ünal & Göncüoğlu, 2012). Many fishers express a need for an individual quota system (especially for anchovy), having seen that when the catch increases in certain seasons, the price they receive falls significantly, as does their income—so much so that they cannot even cover their expenses. A few years ago, fishers attempted to introduce a voluntary self-quota for *hamsi* (anchovy) and *istavrit* (horse mackerel). However, the system collapsed after the first few days when prices ran high and some fishers realized they could optimize their profits by catching more fish during this time (E. Menekşe, personal communication, December 12, 2015).

Some marine scholars also endorse the quota system. In fact, individual quotas tend to be effective tools in dealing with overfishing and overcapacity problems (OECD, 1997b). Yet data collection and stock assessment issues pose great challenges to implementing this system in Turkey. There is a strong tendency among fishers to underreport their catch, in the fear that the government will impose taxes on their income. However, this makes it more difficult to implement individual quotas, since a quota system allocates quotas depending on each fisher's catch history. In Turkey, catch data is collected by TURKSTAT via self-report surveys, and thus not very reliable due to misreporting. Moreover, the seas that surround Turkey on three sides host multiple species, which makes applying the quota system very difficult.

As a market-based policy instrument, the individual transferable quota (ITQ) system is also implemented in Turkey for the harvest of Bluefin tuna, a species currently under threat of extinction. Turkey is a member of the International

Commission for the Conservation of Atlantic Tuna (ICCAT), and accordingly is assigned a certain TAC for this species every year. This figure was 1,295 tons in 2016, which was then allocated to 31 large-scale vessels via a lottery. Twelve of these vessels sold their quota to the rest for around € 9-to-15 per kilogram in 2016 (Marine scientist, personal communication, July 8, 2016). In the end, the quotas were used by the remaining 19 vessels. The harvest was then sold to big aquaculture corporations such as Kılıç and Dardanel, which export the fish to Japan—the largest consumer of Bluefin tuna—after feeding them in large pools with smaller species such as sardine, herring, and mackerel (called the "fattening process"). In the Mediterranean, the Bluefin harvest is very competitive; the transferable quota system for it creates a great deal of tension and conflict not only among industrial fishers, but between fishers and governmental regulatory authorities as well, both in Turkey and other countries assigned Bluefin quotas. Cases of illegal fishing are often reported in newspapers and the social media (e.g. Alptekin, 2013, November 27; T24, 2016, February, 16). In more general terms, some scholars criticize ITQs for changing traditional and more sustainable fishing practices for this species without being able to prevent its overfishing (Longo & Clark, 2012). However, there is currently not much discussion on the sustainability, effectiveness, and distributional aspects of the ITQ system in neither academic nor policy circles in Turkey.

Illegal fishing constitutes another important problem for Turkey. Recently, a fishing vessel monitoring system was introduced to reduce illegal fishing, which will soon be expanded to cover all commercial fishing vessels above 12 meters. Illegal fishing is perhaps the most serious problem faced by fisheries worldwide. Illegal, unreported, and unregulated fishing amounts to some 20-to-25 percent of all fishing activities in the world (Ünal & Göncüoğlu, 2012). Other problems identified in fisheries management in Turkey include issues related to law enforcement, data collection, local participatory methods, and coordination between stakeholders and among government regulatory bodies, which are all either lacking or ineffective. Weak cooperation between different state organs, as evidenced in general in the country, is a problem encountered in fisheries as well. The Ministry of Food, Agriculture, and Livestock, the Ministry of Environment and Urbanization, the Ministry of Trade, the State Planning Organization, Harbor Chiefs, the Coast Guard, and local municipalities are among the many public institutions responsible for different aspects of fisheries.

Co-management, ecosystem-based fisheries management, and the use of precautionary principle are concepts that have all been mostly absent from policy discussions, at least until recently. The EU accession process and responsibilities related to adapting the EU Common Fisheries Policy paved the way for consideration of these concepts in academic and policy circles, to a certain degree. Yet co-management is merely understood and implemented as consultation in the country, and all decisions on fisheries are still taken by the regulatory authority without any delegation of responsibility (Tokaç et al., 2014). This leaves little room for regional policies, and does not allow for the meaningful and effective participation of stakeholders. Still, fishing cooperatives and environmental NGOs such as Slow Food and Greenpeace have recently become more involved and partially able to influence decision-making, as evidenced in the preparation phase of the new fishing notification this year. While this is a gradual process, it is hoped to have positive contributions to ecosystem-based management and co-management. The role of marine scientists in policy-making is not negligible either, in contrast to other realms of policy-making, such as the environment, social policy, et cetera. Bureaucrats responsible for fisheries usually back their policies based on scientific arguments from the academic community in Turkey. However, there are also cases where academics are not consulted prior to policy implementation (e.g. the buy-back program).

Exposure to and familiarity with markets varies among fishing communities as well as individual fishers. A few cooperatives are able to sell or marketize their catch, while others need middlemen to sell their fish at the Istanbul Fish Bazaar or other wholesale points. This may be related to the variability of the fish harvests across seasons and days, as well as a lack of financial insurance and loans, as a result of which fishers cannot eliminate the presence of middlemen and other intermediaries with greater financial capital (Ünal, Güçlüsoy & Franquesa, 2009). The fish caught is sold via auctions at the big wholesale Fish Bazaars, and sometimes also at fishing cooperatives that have sales facilities in their area. As a market-based system, the sale of Bluefin tuna ITQs is another form of market exposure for fishers. Fishers understand the problems of excess supply in fishing markets in certain seasons well, and in response, demand to have individual quotas to keep prices high enough and prevent a race-to-fish, which is detrimental to their incomes. The relocation of the old Istanbul Fish Bazaar from Kumkapı to Gürpınar brought with it new infrastructure that enables the electronic auction of fish (Habertürk, 2015, August 30). This is likely to constitute a new type of exposure to markets for the fishers.

5.2.2 Local institutional arrangements

In Turkey, fishing cooperatives are the most important organizations that can be considered local institutional arrangements. They have a potential for successful, local, community-based management and can be effective in dealing with the overfishing problem and improving the livelihoods of their members (Baticados, Agbayani, & Gentoral, 1998; Pomeroy & Berkes, 1997). Despite the fact that the western ideal of establishing cooperatives was mainly endorsed by state bureaucrats and the elite in the 1950s and 1960s, fishing cooperatives remained ineffective until the 2000s. This was presumably because the cooperative movement was initiated not by the fishers themselves, but in a top-down manner by the state (Knudsen, 2009). The first fishing cooperative was founded in Istanbul in 1943, and cooperatives increased in number, albeit slowly, in the 1960s and 1970s. Before the 1980s and 1990s, however, the main incentives to become a member of these cooperatives were being able to import equipment and securing cheap loans from the Agriculture Bank. Once the Customs Union came into effect, though, some of these motives became invalid.

In the 1990s, the state constructed large harbors, granted fishing cooperatives the right to rent and operate fishing harbors, and reduced the boat rents for cooperative members. In response, the number of fishing cooperatives increased from 229 in the 1980s to 413 in 2005, and to 552 in 2016 (336 cooperatives operate in coastal areas, the rest in inland fisheries) with around 30,000 members (Arpa, 2015; Knudsen, 2009). Fishing cooperatives are currently organized in 14 regional associations and one central association (SÜR-KOOP). They operate in accordance with the Cooperatives Law No. 1163 that was introduced in 1969, which will be revised by the regulatory authority in the near future.

Since the 2000s, fishing cooperatives have been cooperating closely with marine scientists and environmental NGOs, and becoming increasingly influential. While they used to be quite weak—members joined mainly so that they could use the fishing harbor and secure cheap loans from the state—this seems to be changing, although the transition has not been a smooth one. Various ups and downs abound, including user conflicts between small- and large-scale fishers, tensions with recreational fishers, the impact of middlemen, pressure from municipalities, and marketing problems. Perhaps the most crucial difficulty fishing cooperatives are presently facing concerns the severe decline in fish stocks, which implies the income source of their members is also shrinking. Accordingly, many cooperatives are spending great effort to bring their members together, and work in cooperation and solidarity. As Berkes (1986) emphasized, their mere presence is not sufficient to ensure cooperation and effectiveness in terms of preventing overfishing and overcapacity. Based on six case studies of coastal and lagoon fishing cooperatives, Berkes argued that the smaller and more homogenous groups of fishers were—enabling them to align their reciprocal preference and preserve the resource— the more successful they would be in fisheries management at the local level. In his work, success was defined as being able to mitigate overfishing and overcapacity; however, the performance of fishing cooperatives could be based on other factors as well. Ünal et al. (2009), for instance, evaluated the success and failure of six selected fishing cooperatives on Turkey's Aegean coast. They concluded that important sources of success included, among others, a cooperative's ability to marketize the catch of its members, provide inputs to its members, and deal with illegal fishing in its region, in addition to how satisfied its members were with the cooperative. Membership homogeneity, the number of members, and criteria for membership are also crucial factors that affect the performance of fishing cooperatives (Pollnac, 1988, as cited in Ünal et al., 2009).

5.2.3 International institutional arrangements

Institutional arrangements at the international level include several international agreements committed to protecting the environment in general, and aquatic resources in particular. For instance, Turkey ratified the Convention on the Protection of the Mediterranean Sea against Pollution (Barcelona Convention) in 2002, the Convention on the Conservation of European Wildlife and Natural Habitat (Bern Convention) in 1984, the Convention on the Protection of the Black Sea Against Pollution in 1992, the Ramsar Convention in 1994, the Convention on the Biological Diversity in 1996, and the Convention on International Trade in Endangered Species of Wild Fauna and Flora in 1996. Turkey is also a member of the following commissions and organizations committed to protecting marine resources: ICCAT (International Commission for the Conservation of Atlantic Tuna), EUROFISH, GFCM (General Fisheries Commission for the Mediterranean), and FAO (Food and Agriculture Organization). But Turkey has yet to sign other important conventions such as the United Nations Convention on the Law of the Sea (UNCLOS), the UN Fish Stocks Agreement, and the Compliance Agreement (Tokaç et al., 2014; Ünal & Göncüoğlu, 2012).

In order to comply with EU regulations, the regulatory authority plans to change the Fishery Law No. 1380 that went into effect in 1971, and has already launched a fisheries data processing system; reestablished the General Directorate of Fisheries and Aquaculture under the Ministry of the Food, Agriculture, and Livestock; and opened new port offices for fisheries. The 2016 EU Screening Report for Turkey states that adopting the new Fisheries Law is an absolute must for the country. Further, according to the screening reports of 2015 and 2016, the country must align its regulations related to state aid, market policy, and structural actions with those of the EU. For instance, “producer organizations” need to be established to regulate the market in line with the requirements of the EU Common Fisheries Policy. Overall, one can safely state that the prospect of EU membership has had a positive effect, leading to the introduction of new institutions and regulations.

5.3 Conflicts and tensions between different resource users in fisheries in Turkey

There are substantial tensions between different resource users in fishing in the country. Conflicts between small- and large-scale fishers seem to be one of the most significant, as interviews that I have conducted in prior fieldwork has revealed. Small-scale fishers often claim that they are in favor of sustainable fishing practices. They state that they do not catch under-sized fish, for instance. They claim to comply with most regulations, such as those related to closed seasons and prohibited equipment, as long as they think the regulations are legitimate and well-justified in terms of protecting fish stocks. However, a widespread complaint that I have encountered about industrial fishers is that they are “merciless” and over-exploit the fish that were once so abundant. The use of sonars (fish finders) by industrial fishers is seen as being “unfair” and “greedy”. Some fishers believe that sonars actually harm or kill the fish, although so far there is no scientific evidence to support this belief (Knudsen, 2009); some even claim that if the regulatory authority would just prohibit their use, there would be no need for closed seasons and other technical regulations to conserve fish stocks.

In contrast, industrial fishers argue that heavy pollution in the seas is responsible for the decreasing fish stocks. Another source of conflict is related to perceptions regarding fish stocks. There is disagreement among industrial fishers as to the current level of fish stocks; although some claim that the stocks are “all right”—at least in terms of anchovy and some other species—, the mean size of the fish caught is getting smaller, which indicates to many other industrial (as well as small-scale) fishers that stocks have been overexploited. The uncertainty regarding the current level of fish stocks poses a great challenge for marine scientists and bureaucrats in terms of convincing industrial fishers to reduce overcapacity. In fact, due to overcapacity, some industrial fishers have recently taken to fishing off the Atlantic coast of Mauritania (in Africa), along with French, Spanish, Chinese, and Japanese fishing vessels. These are the larger and well-equipped boats of family-run companies, which in other seasons are used for harvesting Bluefin tuna (Sabah, 2016, February, 29).

The regulation on minimum depth of fishing in the last fishery notification of 2012 created further tensions between small-scale and industrial fishers—it raised the minimum depth limit for purse-seine fishers from 18 to 24 meters (T24, 2012, September 2). In protest, purse-seine fishers decided not to fish in the first days of the season. They argued that the 24-meter limit was not reasonable for the whole country, since geographical differences ruled out certain parts of fertile fish grounds completely (such as the Bay of Iskenderun and the fishing grounds around the Prince Islands close to Istanbul, among others). The first three days of the season were so windy that they kept to their promise, but afterwards, they could no longer self-organize and went out to sea regardless.

Up until recently, industrial fishers were more effective in lobbying the regulatory authority in line with their own interests. They played a larger role in the preparation of fishery notifications. However, small-scale fishers have become more influential in policymaking over the past 7-to-8 years. The turning point occurred mainly in 2012, when state bureaucrats held a consultation meeting with fishers in

Ankara. The meeting was attended by an unexpectedly large number of small-scale fishers (according to observers, members of small-scale fishing cooperatives filled 10 busses), which surprised both industrial fishers and bureaucrats. This previously unseen state of events was organized by the Executive Board of SÜR-KOOP—the Central Association of Fishing Cooperatives. The newly elected Executive Board of SÜR-KOOP spent a lot of effort to self-organize and ensure that every fishing cooperative in Turkey took part in the decision-making. Considering that small-scale fishers (90 percent of all fishers in the country) outnumber industrial fishers (10 percent of all fishers) in terms of number of vessel owners, this turn of events was not so surprising at all. However, informal interviews that I have conducted prior to fieldwork with industrial fishers reveal that they are currently not very pleased about their shrinking influence on policy-making, and argue that they need to be better organized as well.

Tensions between small-scale fishers and illegal fishing trawlers, or the *trol çeteleri* (trawler gangs), have become so intense that this even led to physical violence in 2012, when the head of a fishing cooperative in Istanbul was shot in the face and lost an eye after admonishing a trawler for engaging in illegal fishing activities (Radikal, 2012, January 30). Trawling is prohibited in the Bosporus Strait. Naturally, the more profitable illegal fishing is, the greater becomes the source of conflict. According to members of fishing cooperatives, financial penalties are so low that illegal fishing continues without interruption. Regulatory authorities do not even enforce the actual payment of fines, and as a result, fishers who are fined never pay them. There are also reports that during the night, trawler gangs damage the nets of the small-scale fishers as well (Denizhaber, 2012, February 1).

More recently, tensions ran high between the metropolitan municipality and (primarily small-scale) fishing cooperatives in Istanbul. One of the reasons was that the fishing cooperatives demanded to have their own sales stand in the new Fish Bazaar in Gürpınar, as specified by the law that governs Municipality Fish Halls. While the metropolitan municipality did not reject this demand outright, the likelihood of its ever being realized is very low. It is very important for fishery cooperatives to have these stands in the Fish Bazaar so that they can earn a reasonable income from the fish they catch. Otherwise, they are obliged to sell their catch via middlemen who charge a substantial amount of commission.

Another point of contention is the pressure the metropolitan municipality puts on fishing cooperatives with respect to the management of fishing harbors. Cooperatives are given the right to use fishing harbors for a reasonable rent every ten years, and have to comply with certain rules in managing them in accordance with the Fisheries Harbor Directive of 1996. However, some cooperatives in Istanbul have not been managing these harbors professionally; there are problems related to planning and accounting, among other things. *İstanbul Su Ürünleri Kooperatifler Birliği* (The Istanbul Fisheries Cooperatives Association) is currently addressing this problem by organizing workshops and meetings with its member cooperatives. Yet, the metropolitan municipality has accused cooperatives of management failure and corruption, and claims that the municipality would be more efficient and effective in running the harbors. Bureaucrats from the Ministry of Food, Agriculture, and Livestock seem unwilling to accept this proposal, but if it does come into effect, there is a real danger that cooperatives could lose an important source of income and the autonomy they presently have.

Another conflict is one that is ongoing between commercial and recreational fishers. Recreational fishers are not allowed to sell their catch, and there are upper limits as to how much they can catch per person per day (for marine species, the upper limit is 5 kg per person per day). However, these limits are considered too high, especially by small-scale fishers, who believe the maximum catch limit for recreational fishers is unfair since stocks and catch per unit of effort are already declining. In truth, the impact of recreational fishing on commercial fishing is largely unknown because no data is collected on the issue so far (Ünal & Göncüoğlu, 2012).

The share of aquaculture (fish farming) in the total supply of fish is following an upward trend, both worldwide and in Turkey. In fact, Turkey's aquaculture industry—with a 25-percent share in the European production of *levrek* (seabass) and *çipura* (seabream) in 2012—is presented as a best-practice example to its European counterparts (Eurofish, 2016). At a global level, aquaculture is seen as making up for the loss in capture fisheries. However, the production of fish via aquaculture requires a considerable amount of wild fish species; fishmeal and fish oil are important ingredients in the feed they use. As a result, aquaculture is another source that puts significant pressure on wild fish stocks. Commercial fishers are aware of this fact and criticize fish farming practices in their own circles, although tensions seem not to have flared up so far.

Overall, one can safely say that the state still plays a very significant role in fisheries management in Turkey. Yet problems of overfishing and overcapacity cannot be addressed meaningfully without substantial participation at the local level. As Pomeroy and Berkes (1997) argued, ensuring the cooperation and participation of different resource user groups is crucial for the effectiveness of regulatory instruments in fisheries management. The demand for the implementation of individual quotas for many additional species will make it necessary for fishing cooperatives and marine scientists to work in close collaboration. Because geography, fish species, and local needs differ in different contexts, institutional arrangements at the regional level are needed to address regional problems. While some problems may be better solved at the national scale, many others will be better dealt with at the local scale. Identifying problems and devising potential solutions is more effective at the local level, with the help of local organizations such as fishing cooperatives. The international trend towards more market-based policies in fisheries (OECD, 2006) will pose another challenge in the near future for fisheries management in Turkey in terms of data collection, monitoring, and the enforcement of fishing regulations. Moreover, it may create social tensions between resource users. Participatory policies in the form of co-management and ecosystem-based management have the potential to facilitate effective fisheries management in the country. At this juncture, it is important to understand the way each policy intervention performs in policy-making to better inform these discussions. Experimental studies provide an invaluable opportunity to address these issues.

CHAPTER 6

EXPERIMENTAL DESIGN

Building on Part I of this dissertation, which has provided the theoretical background for the field study, the purpose of this chapter is to introduce the specifics of the field study in terms of research questions, the model, hypotheses and experimental design. Chapter 7 will then describe the data collection procedure, and Chapter 8 will present and discuss the main results of the fieldwork.

The main focus of the field study is to assess the effect of policy interventions on cooperation in a resource use dilemma. More specifically, the purpose is to determine whether there is any significant difference with respect to cooperation when an economic incentive is framed as a fine, as opposed to a market price of an extraction right. Framing effects have been well-documented in public goods games (Kay & Ross, 2003; Liberman et al., 2004). However, according to a thorough literature search, they have not been specifically investigated so far for resource use, although this is an important aspect for environmental policies in real life. Moreover, there is detailed theoretical discussion on the effect of markets on moral and environmental behavior (e.g. O'Neill, 2001; Spash, 2010; Vatn, 2005) but again, this effect has not been directly tested empirically in a resource use dilemma so far. Considering the recent dominance and importance of environmental markets for common-pool resources such as climate, fish stocks, water use rights, it is very important to investigate experimentally the effects of market-like mechanisms on cooperation and resource use. Assessing the effects of market framing is a first step towards this end.

While the framing effect between the fine and market treatment was set as the main research interest, I have also included a baseline treatment and a (face-to-face) communication treatment. The positive effect of communication is well-established in the experimental literature (Cardenas et al., 2000; Cardenas, 2011; Ghate et al., 2013; Handberg & Angelsen, 2015; Sally, 1995), yet, the influence of fines is still not that clear. And neither of them has been tested in a field study in Turkey. Another interesting aim for this research is therefore to test whether the positive effect of communication also holds for the citizens in Turkey, a country with a significantly low level of trust towards institutions and other members of the society (OECD, 2011; World Values Survey, 2016).

I selected the fisheries as an application area of the field experiments. The resource use dilemma in fisheries is quite visible in the country, as it is even leading to physical violence among resource users from time to time. Preliminary fieldwork revealed that the inherent dynamics of the resource use dilemma were well-known to the fishers, and as such, they appeared as a suitable target group for our purposes in this study.

Framed field experiments were preferred in this study, as they are commonly used to investigate cooperation in resource use dilemmas with actual resource users, since they do not use abstract language for the instructions, and instead precisely specify the context to the participants in the experiments. Therefore, it increases experimental control in the field, since, otherwise, participants may impose their own contexts onto the game and that may compromise internal validity (Harrison & List, 2004). Moreover, to facilitate comprehension of the rules of the game, we used a linear public goods game (similar to the Take Game of Dufwenberg et al., 2011), which is easier to understand for our target group of fishers than a more complex non-linear game.

More specifically, the study aimed at answering the following research questions:

- To what extent do individuals choose an extraction level that is close to the socially efficient outcome in a resource dilemma in the absence of any policy intervention? (Baseline treatment)
- How does a symbolic (low) fine affect resource use? Does a (weak) material incentive framed in form of a fine for over-extraction reduce or increase resource use? (Fine treatment)
- Similarly, how does a market-like incentive (the same level of material incentive as above) affect resource use? Does this material incentive framed as the fixed price of an extraction right reduce or increase resource use? (Market treatment)
- If face-to-face communication is allowed in the group but no incentive mechanism is applied, how does this affect resource use? (Communication treatment)

The economic model that is employed in this study is based on a (linear) public goods game. The difference from the usual public goods game is that the subjects extract from a common-pool resource base, instead of contributing to a common project. As such, the model is very similar to the "Take Game" that was designed by Dufwenberg et al. (2011). This model is very suitable for our purposes for two reasons. First, it facilitates the test of framing effects without complicating the game. And secondly, it is easier to use with our target group of fishers, since the two corner solutions of the game, "extract everything" or "do not extract anything" (the Nash equilibrium and the social optimum, respectively), are easy to understand since similar outcomes are possible in the real life resource use problem of fishers. Overall, the model represents an open-access regime that is currently present for most species in fishing in the country, in which extracting as much as possible gives strictly higher individual payoff regardless of what the others are extracting.

Three different policy interventions are imposed in the second stage of the game on top of this model:

- a monetary (dis)incentive framed as a fine per unit of extraction exceeding a certain quota,
- the same level of the monetary (dis)incentive framed as the fixed price of an extraction right in excess of the same quota,
- face-to face communication prior to each round of the game.

The following payoff function is used:

$$\pi_i = bx_i + \frac{\alpha}{n}\left(\sum_i y_i - \sum_i x_i\right)$$

where x_i denotes individual extraction per period for an individual i, b corresponds to the individual marginal benefit from extracting one unit of the resource which is constant for each subject, and π_i is the individual payoff per period for an individual i. The maximum amount that can be extracted by each individual is denoted by y_i. This is as an upper limit on resource extraction due to a capacity constraint (in the experiments, it is presented as a capacity limit due to the size of the boat). Therefore, for each individual i, extraction is such that $0 \leq x_i \leq y_i$ holds, while y_i is the same for all individuals. The number of subjects in the group is denoted by n. There are n

individuals in the group, and for each individual the capacity limit is y_i, and the resource stock is therefore consisting of ny_i units.

Each round, whatever is left in the pool, $(\sum_i y_i - \sum_i x_i)$, is multiplied by α and this is divided by the number of group members, namely, n (such that $\frac{\alpha}{n} > b$). That is, each unit left in the pool becomes more valuable, and the total value of the remaining stock in the pool is then distributed equally to all individuals in the group. In order this model reflect a common-pool resource problem, $b > \frac{\alpha}{n}$ must hold with $b, \alpha, n > 0$. That is, only when $b > \frac{\alpha}{n}$ holds, the private marginal benefit of extraction is higher than the marginal return from leaving one unit in the pool, and therefore we end up with over-extraction.

Therefore, as in a Prisoner's Dilemma, individually-rational (selfish) behavior (S) is to extract everything possible ($x_i^S = y_i$), whereas the socially-efficient (E) amount of extraction would be reached if every individual i would leave everything in the pool ($x_i^E = 0$). Whenever subjects extract less than the upper limit y_i, this may be interpreted as a tendency to cooperate or a sign for pro-social behavior (e.g., Velez et al., 2005; Cardenas et al., 2000). The game is static in the sense that each round can be seen as separate one-shot games. In other words, there is no regeneration of the resource to affect the available level of the resource in the next round. This simple setting is easier to understand for the fishers in the field compared to a game with non-linear bio-physical dynamics, which would require a more complicated calculation of optimal strategies, but the game at hand is still able to capture most of the inherent characteristics and incentives of resource use.

The parameters of the model are chosen in a way to facilitate easy calculation and comprehension of different outcomes for the participants. The parameters were selected are as follows:

$$n = 5;\ y_i = 20;\ x_i \in [0, 20];\ b = 10;\ \alpha = 20, MPCR = \frac{\frac{\alpha}{n}}{b} = 0.4$$

That is, each group consists of 5 individuals. And each individual's (boat) capacity is 20 units. This upper limit is realistic and easy to explain to the participants. Each individual can extract, therefore, between 0 and 20 units from the pool (we call it the "lake" in the experiments) each round. For each unit of extraction, each individual receives 10 experimental units of money. And whatever is left in the pool becomes at the end of the round more valuable and is worth 20 experimental units of money. This is explained to the participants as each fish getting bigger and therefore more valuable when it is left in the lake. The value of each fish left in the lake is then distributed equally to each participant in the group at the end of the round. Instead of "rounds", I use "days of fishing" in the experiments. Moreover, in order to make calculations easier, I decided to set the experimental units of the money to the real monetary unit used in Turkey. That is, one experimental unit of the money in the experiment is equal to one Kuruş (one hundredth of one Turkish Lira) in real life. Due to this one-to-one relation I did not need to use any imaginary name for the experimental monetary unit in the game.

One of the important parameters for public goods games is the marginal per capita return ($MPCR$). It is a measure comparing the marginal returns from private extraction as opposed to common use. As long as $MPCR = \frac{\frac{\alpha}{n}}{b} < 1$, the individually-rational strategy is to extract everything (or for pure public goods games, it means that it is individually-rational to contribute nothing to the common project). Empirically, as $MPCR$ rises, so does cooperation in public goods games. This is because marginal returns from the common use rise relatively, as $MPCR$ increases. With the parameters that are used in this study, the $MPCR$ is equal to 0.4.

This is a summary of the incentive structure of the basic game in the first stage. In this first stage, there are no interventions, no external or internal rules, and no communication between subjects. Each subject only knows his/her own extraction in each round, his/her own payoff, and the total amount of extraction (but not the individual extraction level of other participants). The first stage of the game is exactly identical for all individuals in all groups. There are 15 rounds in the first stage of the game. After round 15, the first stage of the game is ended by the experimenter without prior announcement.

In the second stage of the game, each group is assigned randomly to one of the four treatments. These treatments are described in Table 2.

Table 2. Treatments in the Second Stage of the Game

Baseline	**Fine framing**	**Market framing**	**Communication**
No intervention	X Kuruş as a fine	X Kuruş as fixed price of extraction right	Face-to-face communication prior to each round

- Baseline treatment (Treatment 1): The groups assigned to this treatment play the game in the second stage without any intervention (without any incentive mechanism and without communication). Therefore, their payoff function will continue to be as follows:

$$\pi_i = bx_i + \frac{\alpha}{n}\left(\sum_i y_i - \sum_i x_i\right)$$

The socially-efficient outcome is reached if each individual extracts 0 units from the lake, whereas individually-rational strategy is to extract the maximum possible amount $y_i = 20$ as before in the first stage.

- Fine framing (Treatment 2): The groups assigned to this treatment play the game in the second stage with a symbolic fine, that is, a low fine that cannot ensure the socially-efficient outcome. The fine will be imposed for each individual for each unit of extraction exceeding a certain quota. This quota is equal to 1 unit of the resource. In fact, the social optimum would be reached when each individual extracts zero units from the lake. However, imposing a quota of zero does not work well in the field, as fishers are usually not willing to accept a quota of zero units. This is damaging their identity as a fisher.[42] Since quotas are a sensitive issue in fisheries in Turkey, I did not call this upper limit on extraction explicitly a "quota" in the instructions, but rather they were described in a neutral language.[43]

  When a subject's extraction exceeds this quota (i.e., $x_i > q = 1$), the subject has to pay a fine $f = 2$ for each unit of overextraction. Therefore, the payoff function is as follows:

[42] In the pilots we encountered statements such as: "What kind of a fisher I am if I do not catch any fish?".
[43] Most fishers demand a quota in order to ensure higher income, yet, quotas are not easy to implement in Turkey due to lack of reliable historical catch data and commercial multispecies nature of the seas around Turkey.

$$\pi_i = bx_i + \frac{\alpha}{n}\,(\sum_i y_i - \sum_i x_i) - f\,max\,\{x_i - q, 0\},\;\; for\;\; x_i \in [0, 20]$$

As long as $-f > \frac{\alpha}{n}$, the fine is theoretically ineffective. With our parameters at hand, this implies $10 - 2 > \frac{20}{5}$. So, f could take on any value between 1 and 5. I chose $f = 2$ in order to make it more salient that the fine is small enough and ineffective. This fine is enforced effectively in this game. That is, the probability of being caught when exceeding the quota is 100%. Perfect enforcement may seem unrealistic, yet, since the level of fine is very low anyway, this regulation has only symbolic meaning. Moreover, studies employing an imperfect enforcement usually assume risk-neutrality of the fishers. This assumption enables the calculation of expected payoffs for the researchers, but it seems arbitrary to assume for a fisher to be risk-neutral.

- Market framing (Treatment 3): The groups assigned to this treatment play the game in the second stage with the same level of material disincentive as under the fine framing treatment. The incentive is, nevertheless, framed under this treatment as the fixed price of extraction rights. Similar to the fine treatment, the fixed price p is imposed for each unit of extraction exceeding the quota $(q = 1)$. When a subject's extraction exceeds this quota (i.e., $x_i > q = 1$), the subject has to pay a fixed price of p for each unit of overextraction. This fixed price is exactly equal to the fine in the previous treatment, that is, $p = f$. Again, there is no room for non-compliance, the rule is enforced with 100% probability. Therefore, the payoff function is as follows:

$$\pi_i = bx_i + \frac{\alpha}{n}\,(\sum_i y_i - \sum_i x_i) - p\,max\,\{x_i - q, 0\},\;\; for\; x_i \in [0, 20]$$

Due to this perfect equivalence between the level of fine and the market price, it is possible to test the presence of a framing effect between the fine and the market. That is, the only difference between the fine and market treatments is the wording used to refer to the fine or the price of the extraction rights. Apart from this change in wording, everything is kept the same across these two treatments. It is also important to note that the payment for either the fine or the extraction right is immediately subtracted from the payoff of each subject at each round.

- **(**Face-to-face) Communication (Treatment 4): The groups assigned to this treatment have the opportunity to communicate prior to each round (for max. 3 minutes) in the second stage of the game. No incentive mechanism is imposed. The incentives do not change, the game is still open-access as in the first stage. Therefore, the baseline payoff function also applies here:

$$\pi_i = bx_i + \frac{\alpha}{n}\left(\sum_i y_i - \sum_i x_i\right)$$

For each of the above treatments, the individually-rational behavior is to extract everything possible. That is, $x_i^S = y_i = 20$ units should hold for each individual in the study based on the theoretical expectations of the Nash best responses and the Nash equilibrium. This is because $b > \frac{\alpha}{n}$ holds for Treatments 1 and 4. The marginal benefit of extracting one unit from the pool is $b = 10$, whereas the marginal benefit of leaving one unit of the resource is $\frac{\alpha}{n} = 4$, which is strictly smaller than b. For Treatments 2 and 3, the marginal benefit of extraction amounts to $b - f = 10 - 2 = 8$ (Treatment 2) or $b - p = 10 - 2 = 8$ (Treatment 3). And the marginal benefit of leaving one unit of the resource in the lake is $\frac{\alpha}{n}$, which is selected to be smaller than $b - f$ or $b - p$. So, $b - f \geq \frac{\alpha}{n}$ and $b - p \geq \frac{\alpha}{n}$ hold, and $f = p$ holds as well (with $f = p = 2$).

Therefore, the marginal fine f is a symbolic (low) fine paid with certainty, and the fixed price of the extraction right is low and ineffective as well. Hence, at least theoretically, there are no actual material incentives for individuals to change their behavior in the second stage of the game compared to the first stage of the game in any of the treatments. Figure 1 summarizes the steps in the experiment:

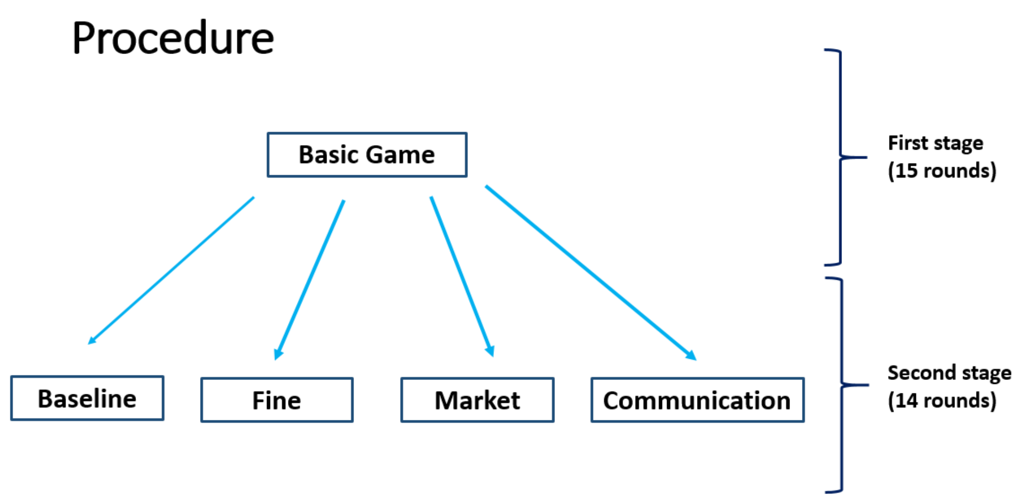

Figure 1. Experimental procedure

The first stage involves 15 rounds of the basic, open-access game. Each group played the precisely same game in the first stage. As outlined earlier in the text, in the second stage, each of the groups was assigned randomly to one the four treatments and played the new game, either baseline, fine, market or communication game, for the rest of experiment, e.g., for 14 rounds. We ended the game in the second stage after round 14 in order to prevent a final round effect.[44] That is, both in the first stage and the second stage, the subjects did not know when the game would end. But since we ended the game after round 15 in the first stage, the participants would expect the second stage to also end after round 15. In order to prevent this, in the second stage we ended the game earlier than in the first stage after, i.e., after round 14.

Similar studies have adopted a shorter version of our game, consisting of 10 rounds for each stage (e.g. Cardenas et al., 2000; Velez et al., 2009). I decided to make each stage a little longer in order to allow for more learning. Moreover, in public goods games, the usual pattern is a relative fall in cooperation towards the end of the 10 rounds. For framing effects, that could also mean that the framing effect disappears the more the game is played. The addition of 5 more rounds into the game in the second stage allows us to observe whether the framing effect, if there is any, persists over a longer time horizon. On the other hand, making the experiment too long could have been boring for fishers. Therefore, I decided not to make it any longer than 15 periods in each stage.

With the data at hand, the following main hypotheses were tested:

- Hypothesis 1: Average cooperation increases upon the introduction of a symbolic fine.

  There are several experimental studies, which have tested the effects of external fines on cooperation. Some of them tested more specifically the effects of low,

[44] In public goods games, a usual finding is that in the last round of the finite games –if subjects know which one is the final round— there is a substantial level of defection (e.g. Ockenfels & Weimann, 1999). Since we want to test a repeated setting that is common in resource use dilemmas, we decided not to announce the final round of the game in line with similar studies (Baggio et al., 2015; Cardenas et al., 2000b).

symbolic fines. On the one hand, several studies underline that there is a tendency of economic incentives such as fines to crowd-out pro-social behavior. The reason may be that such incentives compromise the internal control of resource users, or the warm-glow that they may have for conserving a resource (Bowles & Polanía-Reyes, 2012; Bowles, 2008; Gneezy et al., 2011; Rode et al., 2015). On the other hand, direct regulation involving fines may signal what is the morally appropriate behavior to resource users. Moreover, fines may provide a hint that free-riding will not be tolerated. Therefore, a fine may, instead, increase the amount of pro-social behavior on average (Frey & Stutzer, 2008). These two opposing perspectives have been tested by several studies, but a definite conclusion could not be achieved so far. Some of the recent studies demonstrated that crowding-in may be the case for a low level of fine (e.g., Cardenas, 2011). In line with this recent finding, I expect that the low fine in the present study will decrease, on average, extraction compared to an open-access scenario (the basic game in the first stage and the baseline treatment in the second stage).

- Hypothesis 2: Average cooperation under fine framing is higher than average cooperation under market framing.

  The (low) price for over-extraction is expected to signal that there is no free lunch: there is a price that must be paid in case of over-extraction. In other words, the market price of extraction rights change the resource system from open-access to a rather limited regime. Therefore, it may reduce extraction on average. On the other hand, this market-like incentive may also signal that whatever is not extracted will be extracted by other users. The individual may hence feel less pivotal in the conservation of the resource within this market context. In other words, the payment of the market price for extraction rights may give the feeling that it is one's "right" to exceed the quota as long as one pays for it. Moreover, the price that needs to be paid is very low anyway. Therefore, the introduction of a fixed price for extraction rights exceeding the quota may lead to "moral disengagement" and, hence, may decrease cooperation, or crowd-out pro-social behavior (Falk & Szech, 2013; Frey & Stutzer, 2008). The fine, on the other hand, may reinforce moral engagement (Bowles & Polanía-Reyes, 2012). Therefore, one can expect that average extraction will be relatively lower with fines as opposed to the market treatment.

- Hypothesis 3: Average cooperation increases upon the introduction of face-to-face communication.

  This is a rather well-established finding in the experimental literature (e.g. Ahn et al., 2010; Ostrom et al., 1992). However, it is still important to test it cross-culturally, especially, in countries like Turkey where trust to other individuals in general is significantly lower. Of course, familiarity and trust within the group may still help to increase cooperation. Therefore, it can be expected that face-to-face communication will serve as a coordination device and will reduce extraction (as in Anderies et al., 2011; Falk et al., 2000).

The following chapter will describe how the data were collected by using the framed field experiments methodology. The visual material that was used in the study will be presented in the next chapter as well. This material includes large posters explaining the instructions, the decision cards and other forms used by participants.

CHAPTER 7

DATA COLLECTION

The field experiments were conducted with members of small-scale[45] fishing communities in Turkey. In total, 240 participants took part in the experiments. For each of the four treatments, 12 sessions were conducted. Each group or session consisted of five individuals. Therefore, each treatment involved 60 subjects in total. Each session/group was assigned randomly to one of the four treatments, which lasted around 1 hour and 15 minutes, including the introduction of the experimenter and the scientific purpose, explanations for the game, the experiment itself, the survey and the payment of the reward including the show-up fee. The experiments were conducted by using the pencil-and-paper method. That is, we did not use any computerized system to conduct the experiments.[46] The show-up fee was 10 TL, and the participants could earn up to 90 TL extra based on their performance in the game. According to the Food and Agriculture Organization FAO (2016) figures, the average annual earnings for small-scale fishers is around $4000. That makes a half-day mean earning of around 17 TL, which is comparably lower than the mean net earning that the participants gained in our study (around 57 TL). Each fisher could participate only once in the study. The study

[45] A precise definition for small-scale fishers does not exist (FAO, 2016c). Usually, fishers who use smaller vessels and low power engines are considered as small-scale fishers. While the European Commission calls the vessels under 12 meters small-scale (European Commission, 2016), FAO (2016c) states that small-scale fishers are usually those who use relatively small amount of capital and energy, make short fishing trips from the shore, and sell fish usually for local consumption. According to these criteria, most fishers can be considered as small-scale, while a few exceptions exist.

[46] The target group, the fishers, are on average, over the age of 50 and many of them are not familiar with new technologies such as touch screens, etc.

was funded by the Research Projects Administration of Boğaziçi University (BAP No: 9921).

One experimenter and one assistant were present to conduct each session. The experimenter was responsible for all verbal and visual explanations, while the assistant was responsible for collecting the decision cards, calculating each subject's payoff for each round, distributing the decision cards, and preparing the closed envelopes filled with the rewards at the end of the game. I was myself present as the experimenter in all of the sessions. This is important to control for any differences that the experimenter himself/herself may create. Due to time constraints, four different assistants helped out for the implementation of the experiments, one at a time. They were trained prior to the sessions in the same manner. They did not intervene into the experiment, nor did they answer any questions. All of the explanations were given by the experimenter, and all questions were answered only by the experimenter in order to ensure consistency across sessions.

Prior to the implementation of the experiments with fishers in the field, pilots were conducted with students and colleagues from Boğaziçi University. 10 sessions were conducted with these subjects. Pilots considerably improved the instructions, the material used (such as decision cards), and gave insights as to what can go wrong in the experiments, what is difficult to understand, and so on. The subjects in these first pilots did not receive any reward at the end of the game, yet, they were motivated sufficiently to take the game seriously. Most of them stated after the session that they perceived it as a useful method to think about commons problems and that it was indeed fun for them to participate.

Experimental design and instructions were also discussed with colleagues from the Fisheries Departments of Ege University (Izmir) and Mersin University. They provided valuable insights regarding the acceptability of such a study, given that they have worked often with fishers in the field. They advised on how to approach the fishers and how to convince them to take part in the study. They underlined that survey studies have reached a saturation point in Turkey with fishers, and therefore, fishers are no more willing to take part in surveys. Some of them are even scared that such studies may harm them, if they answer survey questions on sensitive issues such as the level of daily or seasonal catches, the income they receive, etc. One of the professors at Mersin University even warned by stating that “If fishers believe that a study is conducted with the purpose of imposing a tax on them, then they will always under-report their level of catch, whereas if they believe that the study aims at the provision of financial support or subsidies, they will always over-report”.

Therefore, explaining the purpose of the study very clearly to all participants prior to each session was crucial. Moreover, approaching the fishers via the reference of an influential person whom they trust was necessary to convince them to take part in the study. Another professor, from Ege University, indicated that the payment at the end of the game will also be effective to make fishers agree to take part in the study. So, the advice was to contact either the head of the departments of each fishing community via one of the professors that they know well or via the Head of the Cooperatives Association in Istanbul. I already had a good relationship with the Head of the Istanbul Fishery Cooperatives Association, Erdoğan Kartal, who had been an active participant in the “Commons Workshop” organized at Boğaziçi University in 2010. He liked the idea and promised to call the heads of each cooperative prior to the sessions. The head of each cooperative then announced the study to its members and prepared a comfortable setting for the session in the local *kahve* (cafeteria) or in the community building of the cooperative, or even in one case, in one of the relatively bigger boats. The willingness and support of Erdoğan Kartal, as well as the head of each cooperative, was very important for the study, without which the fishers would be suspicious and unwilling to even talk to us. Each time when we arrived at the fishery cooperatives, the trust relationship with participants was established very quickly in a few minutes and fishers were very welcoming and talkative, with only few exceptions.[47] I cannot stress enough how important it is to reach each participant via a person they know well enough, given the trust level in the country is very low towards institutions as well as other individuals (OECD, 2011; WVS, 2016).

Thirty-nine of the 48 sessions were conducted in the local *kahve*s or the community rooms of the fishery cooperatives. The remaining nine sessions were conducted in a fisheries workshop in Antalya, which was attended by members of fishery cooperatives from all over the country. In this workshop, I conducted three sessions for the baseline, two for the fine, two for the market and two for the communication treatments with 45 individuals.[48] In total, I collected experimental data from 39 different small-scale fishing communities organized in fishery cooperatives in

[47] There were two instances in which we felt a little uneasy, both of them in two different cooperatives on the Asian side of Istanbul. In one instance, one fisher did not want to sign the participant form until he actually saw how the game was played and was immediately convinced that there will be no harm for him. In fact, at the end of the game, he even stated that he liked the game a lot despite his concern at the beginning. In the other case, one of the fishers asked us about the “real purpose” of the study. We explained once more what our scientific purpose was and that we would keep everything confidential. It is therefore very important to be clear about the goal of the study prior to each session. Moreover, it also helps a lot to put all contact details of the experimenter in the participation form.

[48] We could have created a bias for a certain treatment, if we had conducted only one treatment in this workshop. We tried to allocate the groups as evenly as possible to different treatments to avoid such bias.

Turkey between February and June 2016, as shown in Table 3. Four of these communities are located at the Black Sea, ten of them are located at the Aegean Sea, and the rest is located at the Sea of Marmara. The list of all sessions indicating the treatments is provided in Appendix B.

Table 3: List of Fishing Communities

	Fishing community	City	Coastline
1	Bakırköy	İstanbul	Sea of Marmara
2	Beykoz	İstanbul	Sea of Marmara
3	Beyoğlu	İstanbul	Sea of Marmara
4	Eminönü	İstanbul	Sea of Marmara
5	Fatih	İstanbul	Sea of Marmara
6	Güzelce	İstanbul	Sea of Marmara
7	İstinye	İstanbul	Sea of Marmara
8	Selimpaşa	İstanbul	Sea of Marmara
9	Silivri	İstanbul	Sea of Marmara
10	Şile	İstanbul	Black Sea
11	Tuzla	İstanbul	Sea of Marmara
12	Üsküdar	İstanbul	Sea of Marmara
13	Yeniköy	İstanbul	Sea of Marmara
14	Zeytinburnu	İstanbul	Sea of Marmara
15	Mimarsinan	İstanbul	Sea of Marmara
16	Küçük Mustafapaşa	İstanbul	Sea of Marmara
17	Poyraz	İstanbul	Sea of Marmara
18	Gemlik	Bursa	Sea of Marmara
19	Kurşunlu	Bursa	Sea of Marmara
20	Eskihisar	Kocaeli	Sea of Marmara
21	Darıca	Kocaeli	Sea of Marmara
22	Hereke	Kocaeli	Sea of Marmara
23	Bağırganlı	Kocaeli	Sea of Marmara
24	Yalova	Yalova	Sea of Marmara
25	Esenköy	Yalova	Sea of Marmara
26	Bandırma	Balıkesir	Sea of Marmara
27	İğneada	Kırklareli	Black Sea
28	Kumbaşı	Ordu	Black Sea
29	Perşembe	Ordu	Black Sea
30	Mordoğan	İzmir	Aegean Sea
31	Aliağa	İzmir	Aegean Sea
32	Çeşmealtı	İzmir	Aegean Sea
33	Foça	İzmir	Aegean Sea
34	Dikili	İzmir	Aegean Sea
35	Mavişehir	İzmir	Aegean Sea
36	Akbük (Didim)	Aydın	Aegean Sea
37	Yalıkavak	Muğla	Aegean Sea
38	Göltürkbükü (Bodrum)	Muğla	Aegean Sea
39	Datça	Muğla	Aegean Sea

Figure 2 shows the exact location of each community on a map.

Figure 2. The fishing communities included in the study

Eleven per cent of the participants (26 individuals) were not a member of the fishery cooperative, but were a member of the local fishing community and were using some of the facilities available at the cooperative for their fishing activities. The rest (214 individuals) were all cooperative members. The mean age of the subjects was around 53, the mean duration of cooperative membership was 12.9 years. On average, they have been fishing for 28 years, and 50% of the participants had at least one person in their family in the previous generation who was also a fisher. The mean boat size was 8.7 meters, and the mean power of engine was around 81 HP. Only one female fisher participated in the study, the rest was male. [49]

[49] In Turkey, fishers are predominantly men, although there are also women fishers especially at the Aegean Sea who either fish with a male family member (usually husbands), or fish alone (Göncüoğlu & Ünal, 2011).

The steps followed for each session are shown in Figure 3 and each step is explained in more detail below.

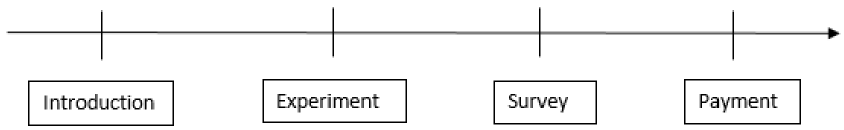

Figure 3. The steps followed in each session

I first called the heads of each fishery cooperative, who had already been informed by the Head of the Istanbul Fishery Cooperatives Association, to organize the sessions and decide where and when to meet. We chose especially the stormy days in which the fishers could not set sails for fishing anyway and would sit together with other fishers in the local *kahve* or the community room of the cooperative. Or if the weather was suitable for fishing, the head of the cooperatives invited us for a time in which the fishers would be back from fishing. This was usually in the morning hours of the day. When we arrived, the head of the cooperative introduced us (the experimenter and the assistant) to the fishers, and asked who was willing to participate. In many cases, the promise of monetary reward was attractive to convince fishers to participate.[50]

[50] Several fishers stated that they would have participated in the experiment anyway, since it looked fun, since they cared about young researchers, or simply since they wanted to be useful. Many of the fishers also expressed their empathy by stating that their sons, daughters or grandchildren were at the university, doing similar home works or theses. Some of the fishers expressed their willingness to help researchers in general, since this is what is needed to improve fisheries in the country and for "being heard" in policy circles.

Then, we introduced ourselves and explained the purpose of our study in rather general terms. I paid attention for not using terms such as “cooperation”, “overfishing” or “over-exploitation of resources” in order not to bias the fishers. Instead, I tried to explain the purpose of the study by using more neutral concepts such as “resource use”, “economic decisions”, which will be seen in the complementary material of this research. This introduction was made via large posters that we brought with us. As such, it was the same across sessions. I also used big posters as visual material to explain the game and to read out the instructions. I preferred not to give any written instructions (apart from the participation form), since pilots with our target group had showed us that fishers do not like reading long texts and prefer receiving verbal instructions instead. This verbal introduction surely improved comprehension. The posters that were used for introducing the study were as follows (See Appendix C for the Turkish version of the posters):

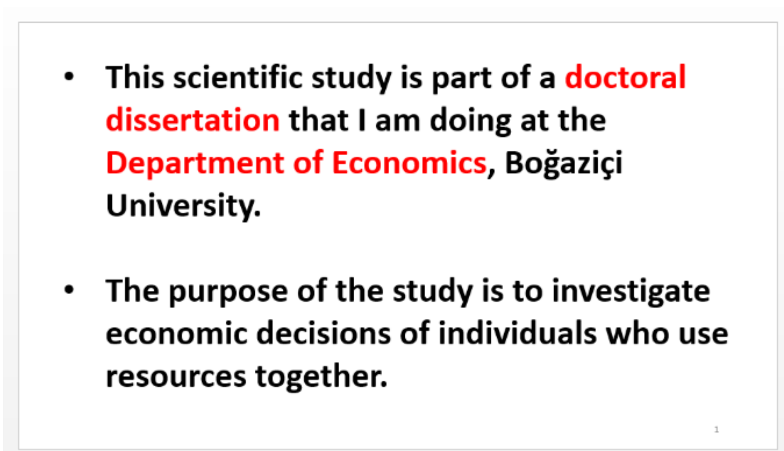

Figure 4. Visual material 1

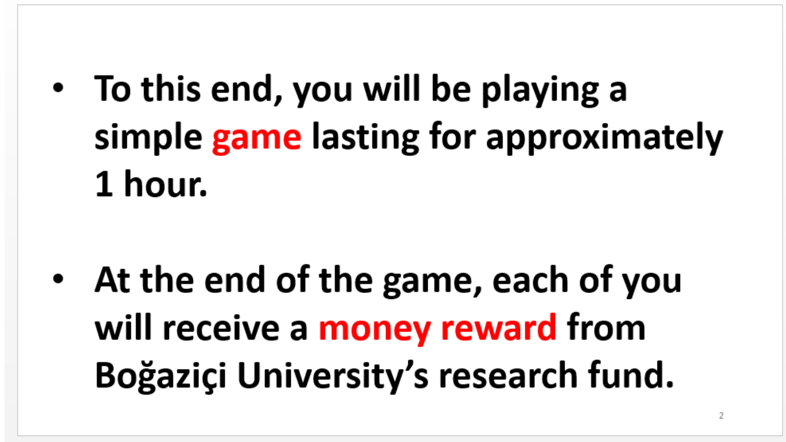

Figure 5. Visual material 2

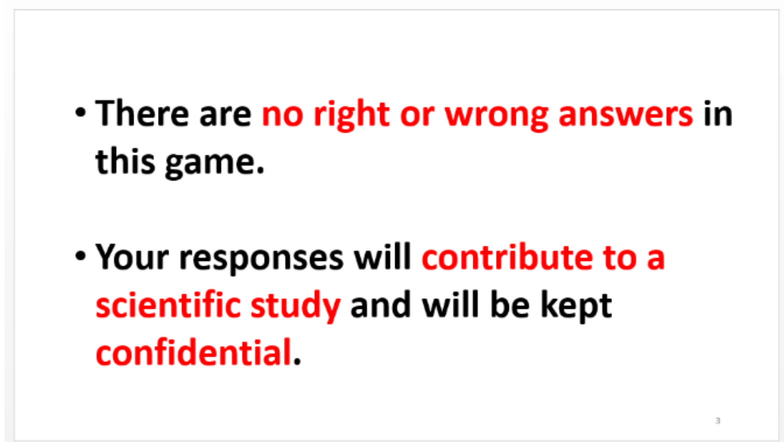

Figure 6. Visual material 3

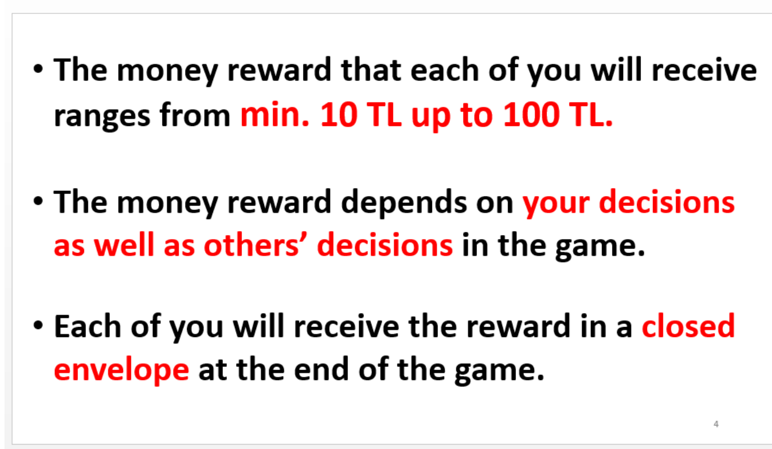

Figure 7. Visual material 4

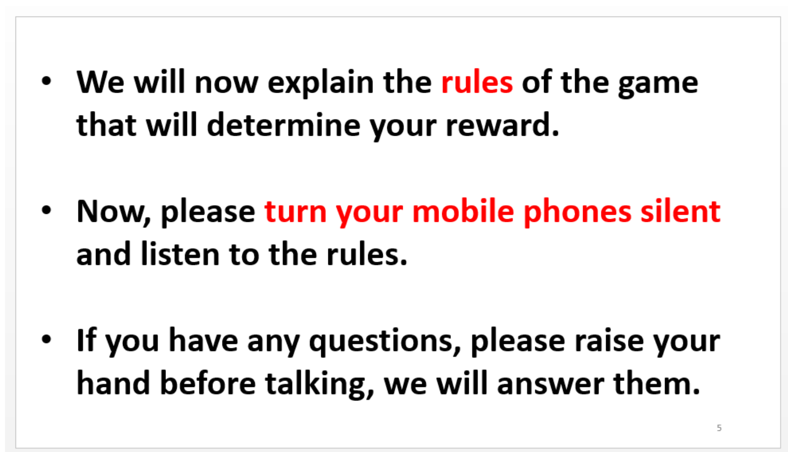

Figure 8. Visual material 5

After this first introduction, we distributed the participation forms (See Appendix D). The participation form was used to explain the subjects their rights and ethical aspects of the study. More specifically, it was clear in the text that they could leave the game whenever they felt uncomfortable, in which case they would be paid the show-up fee of 10 TL. It was also stated that if they needed they could contact any local ethics committee about this study. The text also included the name of the study, the name of the researcher, and the contact details of the researcher. This participation form was signed by each participant, either at the beginning or end of the game. One copy was handed out by the subjects to the experimenter after having signed, and the subjects kept the other copy for themselves.

Then the experiment proceeded with the explanation of the rules of the game, again based on visual material, namely, the large posters. The material that was used as instructions for the first stage of the game was as follows:

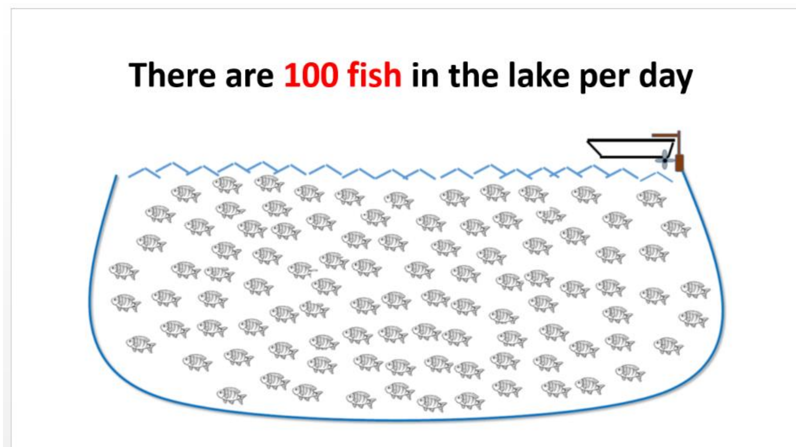

Figure 9. Visual material 6

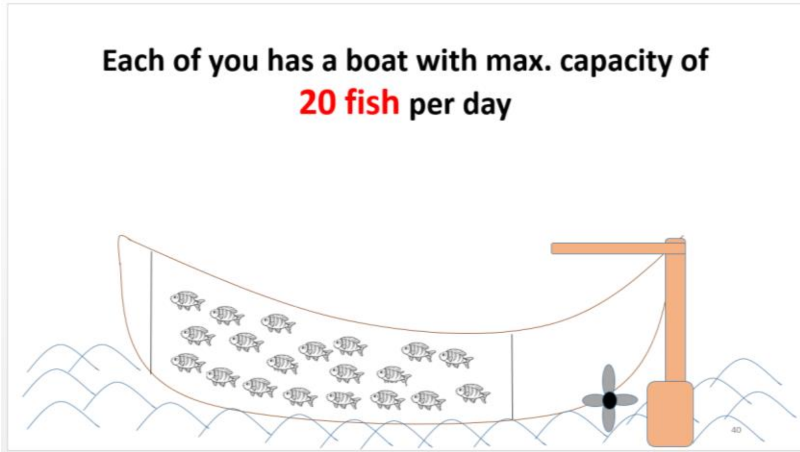

Figure 10. Visual material 7

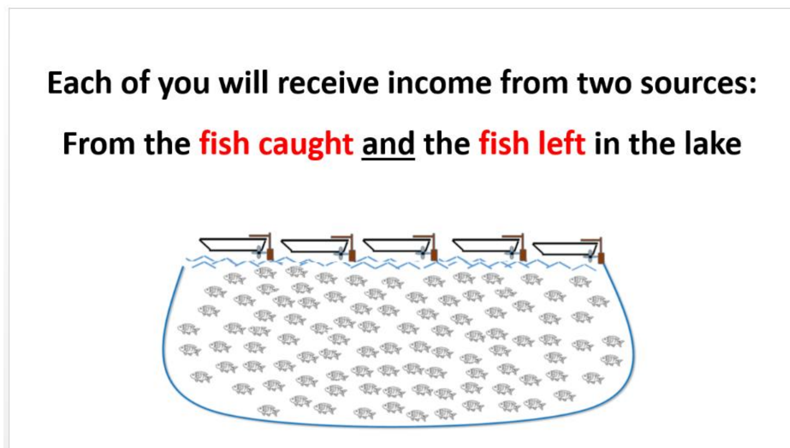

Figure 11. Visual material 8

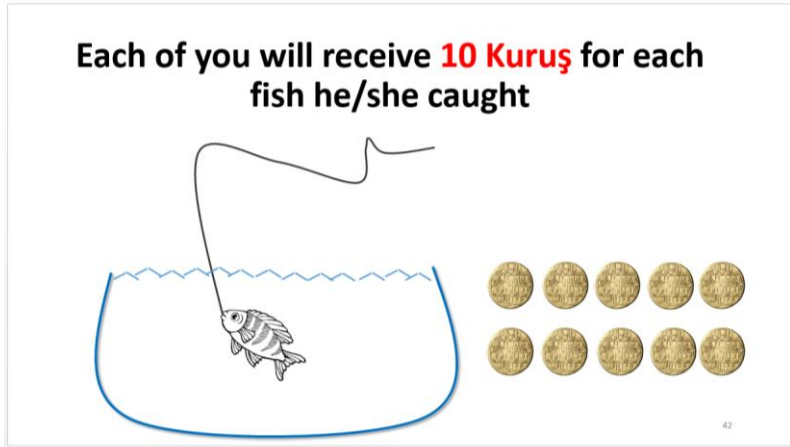

Figure 12. Visual material 9

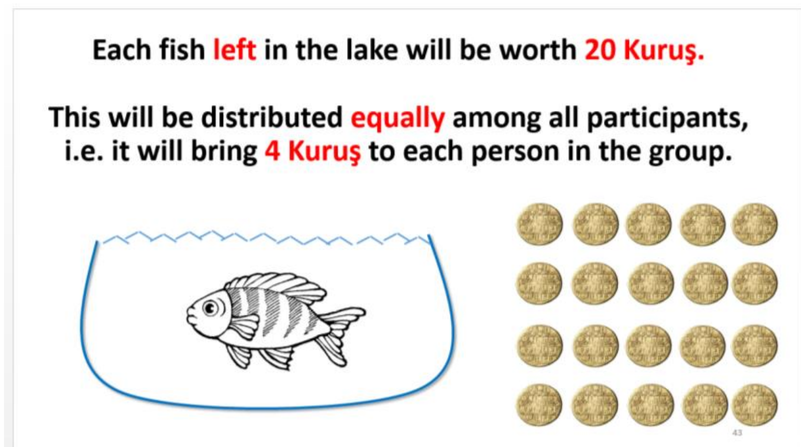

Figure 13. Visual material 10

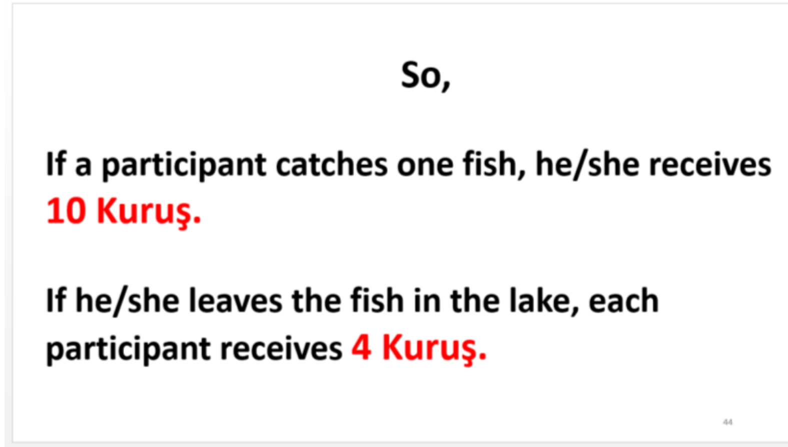

Figure 14. Visual material 11

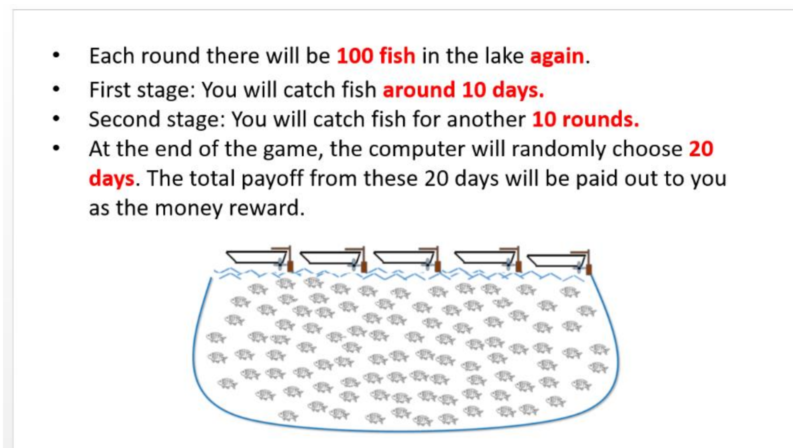

Figure 15. Visual material 12

The pilots undertaken with fishers prior to actual implementation of experiments revealed that fishers had difficulties to calculate and figure out the social optimum as well as the selfish best response. That is, a significant portion believed that taking a small amount from the lake was the selfish best response. They could not see that deviation from this strategy would bring strictly higher payoffs to each person. Therefore, I designed a further visual explanation. After reading the rules from the posters and answering questions, I showed the participants a bowl filled with 100 identical small sticks. These sticks and the bowl represented the 100 fish available each round in the lake. Then, the experimenter gave different (extreme case) scenarios and calculated together with the fishers the payoff that each one would receive under different scenarios.

The first example the experimenter gave was the case in which each participant extracts 20 fish, e.g., the maximum possible amount in this game. So, the experimenter distributed 20 fish to each of the participants, and calculated and announced that each of one would receive a payoff of 200 Kuruş in this case. Then, the experimenter collected all sticks, put them back into the bowl and proceeded with the second case. This is when no one catches any fish, i.e. all sticks remained in the bowl. Now, the experimenter announced that each of the fishers would then receive a payoff of 400 Kuruş and explained how this was calculated. After that, the experimenter proceeded to the third scenario in which one person extracts 20 fish whereas each other in the group extracts 0 units. Then, the experimenter explained that the participant extracting 20 fish would receive 520 Kuruş, whereas each of the remaining participants would receive 320 Kuruş. This last example made it clear that deviation from the social optimum brings extra benefits for ach deviating participant.[51] All these explanations and calculations were done in a neutral language and in the same manner across all sessions in order not to bias the results. [52]

---

[51] This was first tried out with fishers in the pilots. The fishers stated that with this example they immediately saw the parallels to the real life.

[52] For lab experiments conducted with university students, such explanations are usually not given by the experimenters. The experimenters rather expect the subjects to be able to make these calculations on his or her own. This is because students are rather used to this type reasoning. However, with the fishers, it was impossible to let them figure out the underlying conflicts between social optimum and selfish best response

I deliberately did not let the participants calculate their payoffs themselves, as this would have been rather difficult for our target group with rather low level of formal education. I also did not make use of any practice questions as is often used in lab experiments with student subjects. The pilots indicated that fishers would feel intimidated or at least uncomfortable for being tested with questions necessitating math skills like in school times. Instead, we calculated together with participants in different (extreme) cases of the game so that they got an understanding of the nature of the game and the inherent social dilemma in the game. Indeed, as fishers were experiencing a similar problem in their real life, they indicated that the game resembled a lot to their everyday problems with respect to fishing activities.[53] Moreover, I did not employ any non-linearities or probability distributions in the game, since this would also be more difficult to comprehend for our target group, in addition, it would probably not add much to our insights for the purposes of our study. The linear public goods game with corner solutions, on the other hand, was rather easy to understand for the participants of the study, since it closely resembled everyday experience of the fishers. To prevent any confusion regarding the calculation of probabilities and any arbitrary assumption

as pilots revealed. In fact, many field experimenters also advice to explain the incentives structure in a neutral language.

[53] In fact, some of the fishers stated that this game was very familiar to them given their real life experiences. If too much fish is caught and supplied to the market, then "the market shrinks" ("*piyasa düşer*" in Turkish). That is, prices fall a lot, and no one benefits really from this. If there is a lesser amount of fish that is caught on a certain day, then "the market expands" ("*piyasa yükselir*" in Turkish), that is the prices rise substantially, so that the fishers earn a lot more. But if some "reis" (boat owners) catch much fish, while others cannot catch that much, then these boat owners will benefit a lot, while the others will suffer. This idea is in parallel to the extraction levels and payoffs that subjects could gain in our study. For anchovy ("*hamsi*") and horse mackerel ("*istavrit*") there were two different attempts to set voluntary self-quotas in the 2000s. The system collapsed, however, in the last few days, when some of the boat owners realized that deviation from the social optimum (from the quota in this case) brings much more profit while others adhere to the self-quota system. The Prisoner's Dilemma seems actually to be part of the fishers' real life.

regarding the risk-attitude of fishers, perfect enforcement was used regarding the payments of fines and purchase of extraction rights.

Next, the decision cards were distributed to be used by each participant in the study (See Appendix E for their Turkish version). One decision card was used for the first stage of the game, another one for the second stage of the game. The second stage decision cards were only distributed after the first stage of the game was finished. The subjects were informed that there would be a second stage of the game, but they were not informed beforehand how the rules of the game would change in the second stage. The decision card for the first stage was the same for all subjects across all treatments, since all subjects played exactly the same game in the first stage. This is the game without any intervention. The decision card ends in the 16th round (or the day of fishing). However, I ended the game in the 15th period, without informing the subjects beforehand, in line with the literature, to prevent the final round effect.[54]

[54] See, for instance, the experimental protocol created by Cardenas et al. (2000).

FIRST STAGE				
Participant number:				
		This part will be filled in by our assistant		
Day	**HOW MANY FISH do I catch?**	**How many fish are caught IN TOTAL?**	**How many fish are left IN THE LAKE?**	**HOW MUCH IS MY PAYOFF?**
Day 1				
Day 2				
Day 3				
Day 4				
Day 5				
Day 6				
Day 7				
Day 8				
Day 9				
Day 10				
Day 11				
Day 12				
Day 13				
Day 14				
Day 15				
Day 16				

Figure 16. Decision card for the first stage of the game

Upon receiving the decision card, each subject wrote down his/her participant number on the sheet. Then, the experimenter explained how to use the decision card. Accordingly, each participant first decided how many fish to extract from the lake and wrote it down in the first cell reserved for the first day of fishing (i.e. the first round). Without showing this decision and the decision card to any other participants, the participants waited until all decision cards were collected by the experimenter. Then, the assistant calculated (via an excel sheet in the laptop that we brought with us) the total extraction of the group for this round, the number of fish left in the lake and each participant's payoff, and wrote it down on each decision card. Each decision card was then distributed back to each fisher based on their participant numbers. The total amount of fish caught was also announced by the experimenter loud enough so that each participant can hear. The participants inspected their own decision cards, especially the payoff they received for each round, and then decided how much to extract in the next round. The decision cards were then collected by the experimenter again, the total extraction, the fish that is left in the lake and each participant's payoff were calculated by the assistant and the game continued as before. This procedure was followed until the end of the 15th round. After the 15th round, the participants were informed that the first stage ended and that they would now proceed with the second stage of the game. They also received a new decision card for the second stage of the game and were asked to write down their participant number on this card again. They were allowed to keep their own decision cards from the first stage as well.

In the second stage, each group was assigned randomly to one of the four treatments. If the group was assigned to the baseline treatment, each participant in that group was given the decision card shown below in Figure 17. For this treatment, the participants received no new rule, apart from the fact that they were reminded that they should not speak to each other during the game. The second stage consisted of 14 rounds for all of the four treatments. That is, again, the game was stopped at an earlier round so that the participants do not anticipate the final round and act accordingly. This was done to prevent the final round bias.

SECOND STAGE				
Participant number:				
	This part is to be filled in by our assistant			
Day	**HOW MANY FISH do I catch?**	**How many fish are caught IN TOTAL?**	**How many fish are left IN THE LAKE?**	**HOW MUCH IS MY PAYOFF?**
Day 1				
Day 2				
Day 3				
Day 4				
Day 5				
Day 6				
Day 7				
Day 8				
Day 9				
Day 10				
Day 11				
Day 12				
Day 13				
Day 14				
Day 15				
Day 16				

Figure 17. Decision card for the second stage of the game: Baseline treatment

If the group was assigned to the fine treatment, then each participant received the decision card in Figure 18 below. Here, a new column was added to indicate how much fine would be paid by the participant in total in this round. This total amount of fine was directly subtracted from the total payoff for each round, so the last column indicated the net earning per round, after subtracting the fine. This was announced to all participants prior to the second stage of the game.

SECOND STAGE					
Participant number:					
		This part is to be filled in by our assistan			
Day	**HOW MANY FISH do I catch?**	**How many fish are caught IN TOTAL?**	**How many fish are left IN THE LAKE?**	**How much did I pay as FINE?**	**How much is my payoff AFTER HAVING PAID THE FINE?**
Day 1					
Day 2					
Day 3					
Day 4					
Day 5					
Day 6					
Day 7					
Day 8					
Day 9					
Day 10					
Day 11					
Day 12					
Day 13					
Day 14					
Day 15					
Day 16					

Figure 18. Decision card for the second stage of the game: Fine treatment

If the group was assigned to the market treatment, each participant received the decision card in Figure 19. This looks very much like the same as in the previous figure, apart from the fact that we have, instead of a “fine” payment, now, a payment corresponding to the “total price paid to buy extraction rights”. Again, this was shown in the fifth column, and the net earning was shown in the last column of the card.

SECOND STAGE					
Participant number:					
		This part is to be filled in by our assistant			
Day	**HOW MANY FISH do I catch?**	**How many fish are caught IN TOTAL?**	**How many fish are left IN THE LAKE?**	**How much did I pay TO BUY EXTRACTION RIGHTS?**	**How much is my payoff AFTER HAVING PAID FOR EXTRACTION RIGHTS?**
Day 1					
Day 2					
Day 3					
Day 4					
Day 5					
Day 6					
Day 7					
Day 8					
Day 9					
Day 10					
Day 11					
Day 12					
Day 13					
Day 14					
Day 15					
Day 16					

Figure 19. Decision card for the second stage of the game: Market treatment

For those groups which were assigned to the communication treatment, the new rule was the permission to talk to each other prior to each round of the game (for max. 3 minutes). Since this does not change the monetary incentives, the decision card was the same as with the baseline treatment. The decision card for the communication treatment is shown below in Figure 20.

SECOND STAGE				
Participant number:				
		This part is to be filled in by our assistant		
Day	HOW MANY FISH do I catch?	How many fish are caught IN TOTAL?	How many fish are left IN THE LAKE?	HOW MUCH IS MY PAYOFF?
Day 1				
Day 2				
Day 3				
Day 4				
Day 5				
Day 6				
Day 7				
Day 8				
Day 9				
Day 10				
Day 11				
Day 12				
Day 13				
Day 14				
Day 15				
Day 16				

Figure 20. Decision card for the second stage of the game: Communication treatment

Again, the experimenter used posters to explain the new rules for the second stage of the game. The posters for fine, market, communication and baseline treatments are provided below in the respective order:

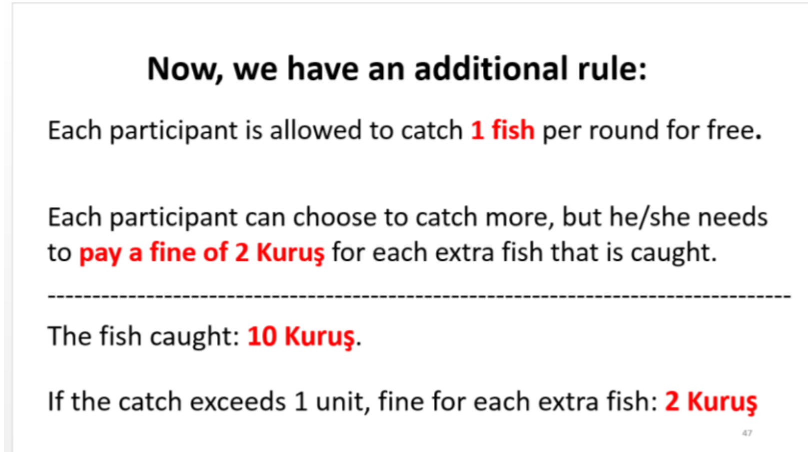

Figure 21. The rules for the fine treatment

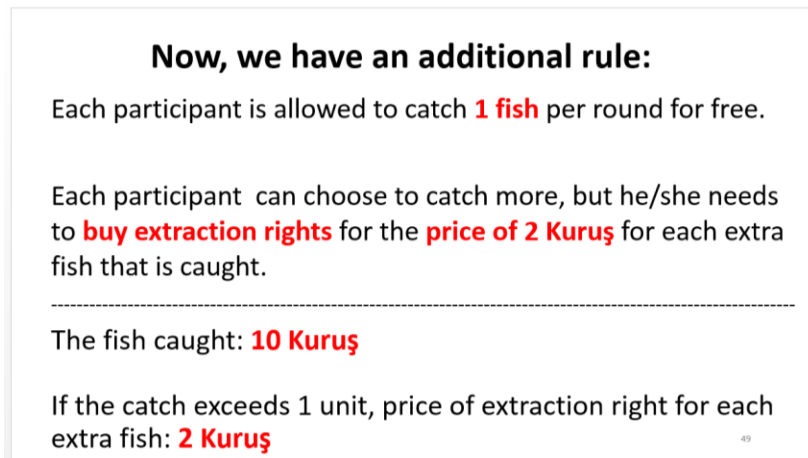

Figure 22. The rules for the market treatment

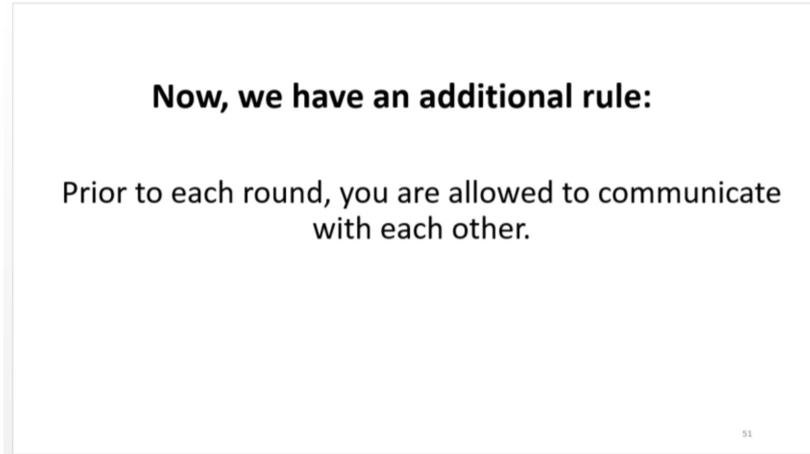

Figure 23. The rules for the communication treatment

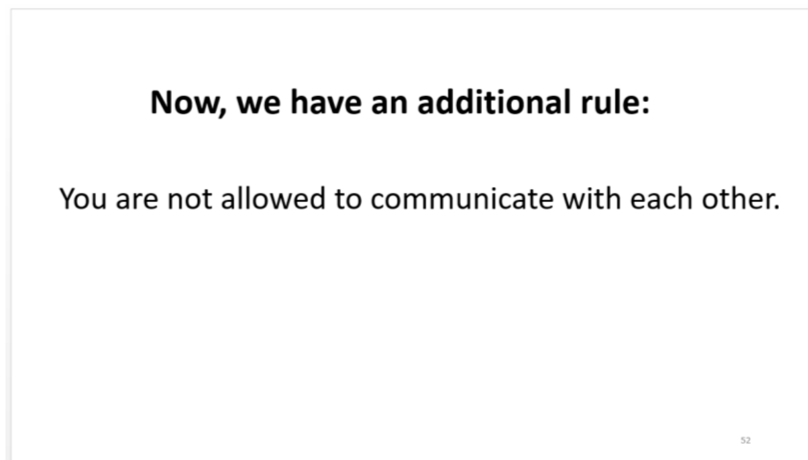

Figure 24. The rules for the baseline treatment

After collecting the data by using the decision cards, the experimenter announced that the next part of the study was a short questionnaire. The participants were also informed that after the questionnaire each participant would receive a closed envelope in which the money reward (including the show-up fee) would be arranged by the assistant.

Appendix B provides a list of all sessions, indicating the location of each session and the treatment that the group was assigned to. The list is organized based on the historical order of actual implementation of the sessions. One of the sessions (not shown here) was conducted in one of the fishery cooperatives in Istanbul, but their data were removed from the study because one of the participants was unwilling to hide his decision cards from other participants, despite all warnings from the experimenter.

The survey items covered the following questions indicated in Table 4. The first five items were addressing the perceived advantages of being a coop member. Items 6-10 addressed the question of how active the participant is within the cooperative. Items 11-13 and 15-18 involved questions regarding fishing activities of the participant. Item 19 was about the perceived income (in)equality within the cooperative, while items 14, 20 and 21 were questions regarding socio-economic characteristics. Each participants answered the survey questions on his/her own, after the experimenter explained how to answer the questions. A five-point scale (1-5) was used instead of a more complicated scale such as 0-10, in order to facilitate understanding of our target group. In some cases, the participants asked for help if they could not understand how to fill in the survey. After a few minutes of extra explanation, they were able to fill it in on their own. The results of the survey regarding socio-economic characteristics are provided in Chapter 8 and the rest is in the Appendix F. The questionnaire (in Turkish) is presented in Appendix G.

Table 4. Survey Items

	Item	Type of response
	How much does being a coop member facilitate the following activities for you?	
1	Use of fishing harbor	5-point Likert scale (1= It does not facilitate at all; 5=It facilitates a lot)
2	Obtaining the necessary documents for fishing	5-point Likert scale (1= It does not facilitate at all; 5=It facilitates a lot)
3	Being informed about new regulations	5-point Likert scale (1= It does not facilitate at all; 5=It facilitates a lot)
4	Sales of fish	5-point Likert scale (1= It does not facilitate at all; 5=It facilitates a lot)
5	Cooperation with other fishers	5-point Likert scale (1= It does not facilitate at all; 5=It facilitates a lot)
	How often do you undertake the following activities?	
6	Attending the general assembly of the cooperative	5-point Likert scale (1= Never; 5=Always)
7	Volunteering to take part of the steering committee of the cooperative	5-point Likert scale (1= Never; 5=Always)
8	Informing the cooperative about a problem regarding fishing activities	5-point Likert scale (1= Never; 5=Always)
9	Trying to find a solution to a cooperative member's problem regarding fishing	5-point Likert scale (1= Never; 5=Always)
10	Trying to find a solution to a cooperative member's problem regarding other matters	5-point Likert scale (1= Never; 5=Always)
11	Was any family member of the last generation occupied with fishing activities?	1=Yes, 2=No
12	What is the main income source in your household?	1=Fishing; 2=Agriculture; 3=Self-employment; 4=Retirement benefit; 5=Other
13	For how many years have you been occupied with fishing activities?	Open-ended
14	How old are you?	Open-ended
15	For how many years have you been a member of the cooperative?	Open-ended
16	How long is your boat?	Open-ended
17	What is the engine power of your boat?	Open-ended
18	Who is the owner of the boat that you are using?	1=Me; 2=I am the co-owner; 3=It belongs to another person
19	Which statement reflects the matters regarding the income that fishers are receiving in your cooperative best?	1=Income of coop members is more or less the same; 2=Income of coop members is a little different from each other; 3=Income of coop members is extremely different from each other
20	Which is the last educational degree that you received?	1=Primary school; 2&3=Secondary school; 4=University and higher; 5=None
21	Which of the following items do you have in your household?	1=Credit card; 2=Personal computer; 3=Internet; 4=Dish washer; 5=Car; 6=Land

The last step of each session was the payment to the participants. In total, each participant played 29 rounds of the game (15 rounds in the first stage and 14 in the second stage). The computer selected 20 rounds of the game randomly and added up the payoffs of these 20 rounds for each participant, as it had been announced at the beginning of the game. This amount —including the show-up fee—was put by the assistant in closed envelopes. Before distributing the envelopes one by one, each participant was asked to sign a payment form that needs to be kept by the experimenter and handed to the person responsible for BAP funds at the University. Participants were only allowed to see their own forms, and not the forms of the others. Some of the participants asked the experimenter who performed the best in terms of the payoff, but the experimenter did not announce it to prevent any conflicts within the community. Handing the envelopes one by one to the participants was a good strategy, otherwise it could become very chaotic during the payment as the pilots showed. The amount of the reward seemed to be satisfactory for the participants. Indeed, the show-up fee plus the extra reward was a substantial amount compared to the mean half day earning of small-scale fishers in Turkey (FAO, 2016a). The net payments that the participants received ranged from 32.25 TL to 77.85 TL, with a mean of 56.79 TL.[55]

[55] The regulation regarding BAP funds necessitates the payment of 20% income tax for each participant by the experimenter. The numbers reported here are the net payments.

CHAPTER 8

RESULTS AND DISCUSSION

This chapter aims to present and discuss the results of the field experiments. First, some descriptive statistics regarding the characteristics of the participants will be presented. Next, I will summarize the experimental results using graphical tools and non-parametric tests. These results will provide the first insights about the study in terms of averages. Then, I will present the results of the regressions and difference in difference analyses controlling for other factors such as socio-economic characteristics of the subjects, period and group fixed effects, and other individual and group level data. Thereafter, a classification of the subjects regarding their types (e.g. selfish and non-selfish) and their responses to different treatments will be provided. The final section will present a synthesis and discussion of the findings as well as some limitations of the study.

8.1 Socio-economics characteristics of the participants

Socio-economic data regarding the characteristics of the participants in the study were collected via a survey at the end of each session. A brief look at these characteristics gives a first idea about our target group of small-scale fishers. Table 5 summarizes the main results.

Table 5. Socio-Economic Characteristics of the Participants

Socio-economic characteristics of participants	N	Mean	Standard deviation	Min	Max	
How old are you?	239	53.49	12.19	17	87	
For how many years have you been occupied with fishing activities?	227	28.01	13.51	2	65	
For how many years have you been a member of the cooperative?	208	12.88	6.80	1	40	
How long is your boat? (in meters)	227	8.65	4.33	4.5	50	
What is the engine power of your boat? (in HP)	226	81.24	128.21	5	1200	
Which is the last educational degree that you received?	Primary	Secondary	University and higher	Missing		
	43.8%	49.6%	5.4%	1.2%		
Which of the following items do you have in your household?	Credit card	Personal computer	Internet	Dish washer	Car	Land
	58.9%	63.3%	62.0%	80.2%	38.8%	28.3%
What is the main income source in your household?	Fishing	Retirement benefit	Self-employment	Agriculture	Other	Missing
	39.6%	31.7%	10.8%	0.8%	5.4%	11.7%
Who is the owner of the boat that you are using?	Me	Co-owner	Non-owner	Missing		
	73.8%	15.8%	7.1%	3.3%		
Was any family member of the previous generation occupied with fishing activities?	Yes	No	Missing			
	50.0%	37.1%	12.9%			

First, the age of the fishers who participated in the field experiments ranges from 17 to 87 with a mean of 53.5. Among the 240 participants of the study, there is only one female participant. Indeed, the number of female fishers is rather small in the country (Göncüoğlu & Ünal, 2011); some of them work with their husbands, some of them on their own. Most sessions were conducted in the local *kahve*s of the fishing communities and in the community rooms of the fisheries cooperatives, in which the presence of female fishers is limited.

The mean years of fishing was 28 years and the mean years of cooperative membership was 12.9 years, implying that most fishers have a substantial level of experience with fishing activities. The length of the boats of the participants ranges from 4.5 to 50 meters, with a mean of 8.65 meters, and the power of the engines range from 5 to 1200 HP, with a mean of 81.24 HP. This confirms that our sample is indeed consisting of primarily small-scale fishers, as small-scale boat length is usually understood to vary between 5-12 meters (Ünal & Göncüoğlu, 2012). 49.6% of the participants attended secondary school (either "*ortaokul*" or "*lise*", i.e. lower or higher secondary school) and 43.8% attended only the primary school. University degree is rather rare among the participants with 5.4%. This is in line with the general finding that fishers often have a lower level of formal education compared to non-fishers, both in the world and in Turkey (Güngör, 1998; McGoodwin, 1990, as cited in Knudsen, 2009). None of the participants was without a formal education, and all participants were literate.

The main income source of 39.6% of the participants is related to fishing activities. Among the participants, 31.7% of fishers are retirees who live on their retirement benefits rather than fishing incomes[56]. 73.8% of the fishers own the boat that they use for fishing activities, whereas 15.8% are one of the co-owners of the boat, and 7.1% do not own the boat or a share of the boat they are using. 50% of our participants have at least one person in the previous generation in their families who was also occupied with fishing activities. Among the items that are owned by the household, the

[56] Some of the fishers stated that in the past the income from fishing activities was instead higher than any retirement benefit that one could receive. But now, most live on the regular income of their retirement benefit. The income from fishing activities is low and very irregular.

most common one is a dish washer (80.2%), followed by personal computer (63.3%), internet (62%) and a credit card (58.9%). Only 38.8 % of the participants own a car, and only 28.3% own a piece of land. The sessions that we have conducted have been predominantly in and around urban centers of the country, especially in Istanbul. The data on ownership of household items seem to reflect this characteristic of the participants.

Summary statistics regarding the age and education level of the participants across treatments is provided in Table 6. There is no significant difference across treatments at p=0.05 level with respect to age and education groups, as ANOVA analyses confirm.[57] These findings are in line with the random assignment of groups to treatments, i.e. the randomization procedure resulted in a balanced sample with respect to socio-economic characteristics of age and education.

Table 6. Summary Statistics for Age and Education across Treatments

	Age			Education		
Treatment	N	Mean	Standard deviation	N	Mean	Standard deviation
Baseline	60	55.93	11.10	59	1.53	0.54
Communication	60	53.25	11.81	59	1.63	0.69
Fine	60	55.30	12.33	60	1.52	0.54
Market	59	49.39	12.68	59	1.78	0.56
Total	239	53.49	12.19	237	1.61	0.59

[57] The mean age is within the 45-64 years for all treatments (Age classification based on UN [1982]. The age groups are classified as 0-15, 15-24, 25-44, 45-64 and 65+). This was not rejected by the ANOVA analysis across treatments with respect to age groups (p=0.0921). The educational groups consist of primary school, secondary school, and university and higher. The difference across treatments with respect to these three educational groups were not found to be significant (p=0.0533).

8.2 Experimental results: Descriptive statistics

The descriptive statistics for the experiments are provided in Tables 7, 8 and 9. Table 7 summarizes the average results for the first stage of the game, in which the basic open-access game was identical for all treatment groups. The subjects in each treatment played exactly the same game without any intervention and any communication. In the first stage, the average extraction was just above 11 units of fish, while the choice set varied between 0 and 20 units. There was much individual variation around these averages, though (see Appendix H). Yet, only four individuals consistently employed the selfish best response of 20 units in all rounds. Moreover, only 25% of all observations in the first stage were close to the Nash best response of 20 units. That is, extraction equal or above 17 units amounted to the 25% of all observations in the first stage of the game. The rest of the observations were below 17 units. That is, even in the basic game representing the open-access regime, there was a substantial level of cooperation contrary to the theoretical expectations regarding the Nash equilibrium implying 20 units. Forty-seven of the fishers (19.6%) have extracted in the first period of the game between 9-11 units, i.e. somewhere in the middle of the choice set which was between 0 and 20 units, while only 3 individuals (1.25%) extracted 0 units, and 21 individuals (8.75%) 20 units in the first period of the game.

The maximum possible payoff that could have been reached when all the five participants in the group cooperated by leaving everything in the lake was 400 Kuruş for each participant. This would be the efficient outcome in this basic model. Therefore, the efficiency figures in Table 7 and 9 indicate the ratio of the average payoff achieved in each treatment as compared to the efficient payoff of 400 Kuruş. As one would expect with randomly assigned groups, there is no significant difference across treatments in the first stage of the game. This is confirmed with Wilcoxon (Mann-Whitney) rank sum tests as shown in Table 8.

Table 7. Average Extraction, Average Payoff and Efficiency in the First Stage

First stage: Periods 1-15			
Treatments	Average extraction (Units of fish)	Average payoff (Kuruş)	Efficiency
Baseline	11.34	286.54	0.716
Communication	11.17	288.28	0.721
Fine	11.81	281.92	0.705
Market	12.03	279.72	0.699

Table 8. Comparisons of Average Extraction and Average Payoff in the First Stage

Pair-wise comparisons	Rounds	Average extraction		p-value	Average payoff		p-value
Base vs Fine	1-15	11.34	11.81	0.525	286.54	281.92	0.525
Base vs Market	1-15	11.34	12.03	0.299	286.54	279.72	0.299
Base vs Comm.	1-15	11.34	11.17	0.863	286.54	288.28	0.863
Fine vs Market	1-15	11.81	12.03	0.686	281.92	279.72	0.686
Fine vs Comm.	1-15	11.81	11.17	0.603	281.92	288.28	0.603
Market vs Comm.	1-15	12.03	11.17	0.225	279.72	288.28	0.225

p values are from Wilcoxon (Mann-Whitney) rank sum tests.
***, **, and * indicate a significance level of 1%, 5% and 10%, respectively.

In the second stage of the game, a new rule was introduced for all groups, except for those who were assigned to the baseline treatment. As such, the baseline treatment is an indication of how extraction would have changed if there was no incentive mechanism or no communication at all. As Table 9 demonstrates, the average extraction is highest under the baseline treatment, and lowest under the fine treatment. The average efficiency is highest, on the other hand, for the groups under the communication treatment, while it is lowest for those under the market treatment. It is important to note that although the average extraction is highest under the baseline treatment, the market treatment performs the worst in terms of efficiency. This is because on top of the higher level of extraction, subjects were required to make a payment for exceeding the quota. This is why the market treatment performed worst in terms of payoffs and, therefore, also worst in terms of efficiency.[58]

Table 9. Comparison of Average Extraction, Average Payoff and Efficiency in the Second Stage

Second stage: Periods 16-29				
Treatments	Average extraction (Units of fish)	Average payoff (Kuruş)	Efficiency	Average payment due to exceeding quota (Kuruş)
Baseline	12.20	277.98	0.695	None
Communication	9.28	307.19	0.768	None
Fine	8.86	295.60	0.739	15.84
Market	11.07	269.08	0.673	20.23

Average payoff for fine and market treatments are net payoffs, that is, they are the payoffs the subjects have received after payment for fine or extraction rights was subtracted from the total payoffs.

The data collected are analyzed first by using non-parametric tests. The experiments involve both a within- and between-subject design. I use Wilcoxon (Mann-Whitney) rank sum tests for between-subject design and Wilcoxon signed rank tests for within-subject design.

Under the baseline treatment, the average extraction is significantly higher in the second stage compared to the first stage of the game. Yet, one has to note that, average

[58] Note that for the second stage, the efficient outcome does not change. Under the baseline and communication treatments, the incentives do not change at all. And under the fine and market treatments, each subject receives 400 Kuruş, if each one of them extracts zero units, while if they adhere to the quota and extract only 1 unit, each receives 10+(95x4)=390 Kuruş. And all other levels of extraction give strictly lower payoffs. Therefore, the efficient outcome is still reached when everyone extracts 0 units under each of the treatments in the second stage.

results under the baseline treatment are still substantially lower than the Nash equilibrium prediction of 20 units. On the other hand, average extraction falls significantly when a fine (2 Kuruş per each fish in excess of the quota) is imposed in the second stage. Although there is a slight fall in the average extraction under the market treatment, this fall is not statistically significant. Communication is also effective in significantly reducing the average extraction as Table 10 demonstrates. The average payoff, on the other hand, falls significantly under the baseline treatment, and it increases significantly under the communication treatment compared to the first stage of the game. Even though there is a significant fall in the average extraction under the fine treatment, this does not translate into a significant rise in payoffs. This is because the subjects still pay a substantial amount in terms of fines. This does not give much room for an improvement in the average payoffs under the fine treatment.

Table 10. Average Extraction and Average Payoff for the First vs. Second Stage

Pair-wise comparisons	Rounds	Average extraction		p-value	Average payoff		p-value
Base vs Base	1-15 vs 16-29	11.34	12.20	0.010***	286.54	277.98	0.001***
Fine vs Fine	1-15 vs 16-29	11.81	8.86	0.003***	281.92	295.60	0.308
Market vs Market	1-15 vs 16-29	12.03	11.07	0.209	279.72	269.08	0.308
Comm. vs Comm.	1-15 vs 16-29	11.17	9.28	0.028**	288.28	307.19	0.028**

p values are from Wilcoxon signed rank tests.
***, **, and * indicate a significance level of 1%, 5% and 10%, respectively.

The comparison of average extraction and payoff across treatments is also revealing. Table 11 indicates that the average extraction is lower under all the three treatments compared to the baseline treatment. This negative difference is significant at 1% for communication treatment, at 5% for fine treatment and 10% for market treatment. With regard to the payoffs, the difference between the baseline and communication is statistically significant at 1%, and also the difference between the market and the communication are statistically significant at 1%. It seems to be counterintuitive for the latter case, since there is no significant difference between average extractions under these two treatments. However, the difference between payoffs is significant because under the communication treatment the average extraction fell without subjects having to pay anything extra, while with the market treatment, the average extraction was a little higher, and subjects who exceeded the quota had to pay a substantial amount. This explains the significant difference between the payoffs of the market and the communication treatments.

Table 11. Comparison of Average Extraction and Average Payoff across Treatments in the Second Stage

Pair-wise comparisons	Rounds	Average extraction		p-value	Average payoff		p-value
Base vs Fine	16-29	12.20	8.86	0.033**	277.98	295.60	0.204
Base vs Market	16-29	12.20	11.07	0.094*	277.98	269.08	0.419
Base vs Comm.	16-29	12.20	9.28	0.008***	277.98	307.19	0.008***
Fine vs Market	16-29	8.86	11.07	0.166	295.60	269.08	0.166
Fine vs Comm.	16-29	8.86	9.28	0.817	295.60	307.19	0.603
Market vs Comm.	16-29	11.07	9.28	0.119	269.08	307.19	0.009***

Average payoff for fine and market treatments are net payoffs, that is, they are the payoffs the subjects have received after the payment for fine or extraction rights was subtracted from the total payoffs.
p values are from Wilcoxon (Mann-Whitney) rank sum tests.
***, **, and * indicate a significance level of 1%, 5% and 10%, respectively.

The average extraction is lower under the fine treatment compared to the market treatment by more than 2 units, although this was not found to be statistically significant at 10% level, as Table 11 indicates. However, this difference, albeit not statistically significant at this stage where we have not yet controlled for other factors, may be a first indication of a framing effect between the fine and the market treatments. Further, Figure 25 presents this difference with respect to the rounds, giving another strong indication for a framing effect between the fine and the market treatments. That is, average extraction is lower under the fine treatment compared to the market treatments, for all rounds in the second stage. The average extraction under the fine and communication treatments are rather undistinguishable from each other, but both of them consistently lower than under the baseline during the second stage. The average extraction under the market treatment is lower than under the baseline treatment, except for one round. However, the difference of the fine treatment compared to the baseline treatment is much larger.

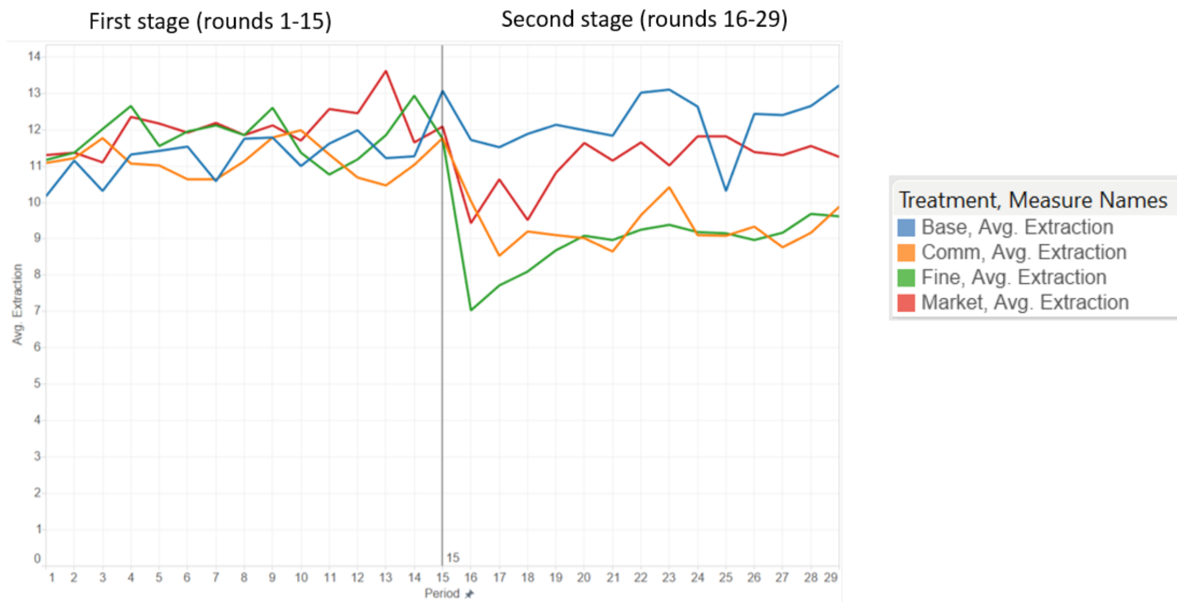

Figure 25. Average extraction under baseline, fine, market and communication treatments

So far, we seem to have some preliminary evidence to support all of our three hypotheses that were presented in Chapter 6. That is, Hypothesis 1 seems to be confirmed, since the average extraction is lower under the fine treatment compared to the baseline treatment for all rounds. Hypothesis 3 is also lent some support since communication was very effective to reduce the average extraction compared to the baseline. Moreover, Hypothesis 2 seems to have some empirical support, too, since average extraction is consistently lower under the fine treatment compared to the market treatment. Yet, further analyses will control for other factors such as socio-economic characteristics, and period and group effects, and as such, they will provide more reliable results.

Figure 26 below presents a similar analysis with respect to average payoffs. The average payoff is highest under the communication treatment except for the first round of the second stage. It is followed by the average payoff under the fine treatment, and then by the average payoff under the baseline treatment. On average, the lowest payoff is received under the market treatment, except for the 16th and 18th rounds. One has to note that the payoffs presented in all graphs and tables in this chapter are net payoffs, that is, payoffs that subjects received after subtracting the payment due to exceeding the quota.

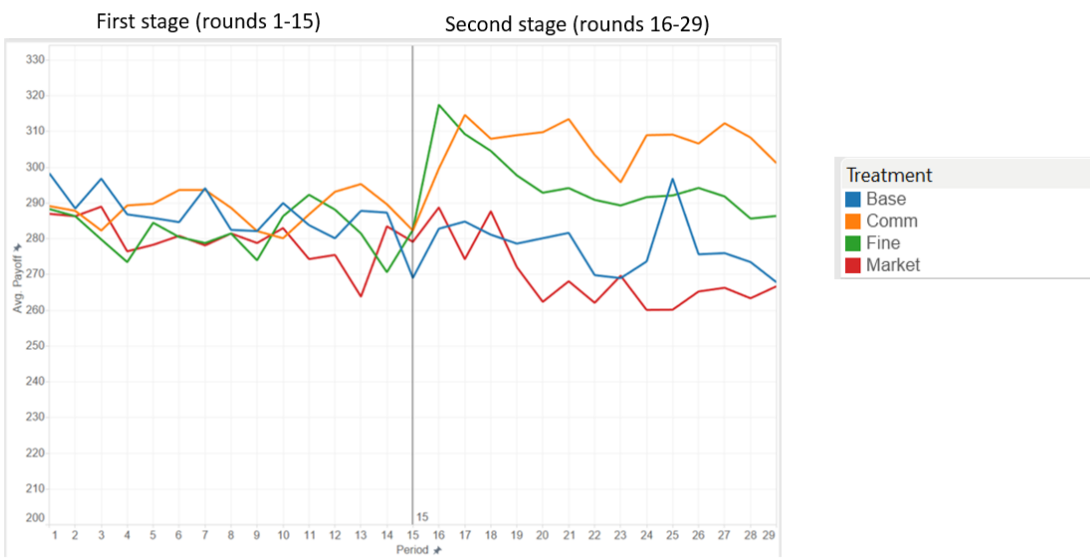

Figure 26. Average payoff under baseline, fine, market and communication treatments

Figure 27 further demonstrates that there is a substantial level of group variation around these averages. In this figure, each colored line stands for a group average (or session average). The black line represents the average extraction for each treatment across all groups playing the game under this condition.

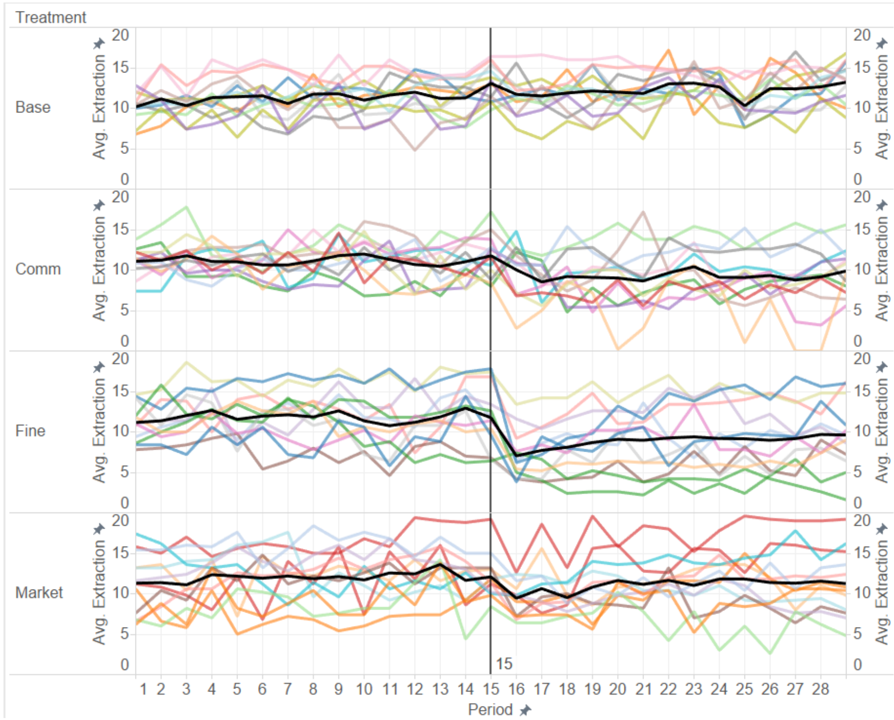

Figure 27. Group averages across treatments

8.3 Regression analyses: Explaining the responses to different treatments

The preliminary results from the non-parametric tests provided a first idea about differences in average extraction under different treatments. Simple OLS regressions are then run to control for other factors such as age, education, and period and group fixed effects. In these regressions, dummy variables were used for each treatment, leaving the dummy variable for the baseline treatment out in order to prevent the dummy variable trap. Therefore, the results regarding the treatment effects provided in the Table 14 need to be interpreted in relation to the baseline treatment. The dependent variable in all of the four regressions provided in Table 14 is the individual extraction for each round of the second stage. I used pooled regressions, i.e. I clustered observations for each individual in order to get robust standard errors.[59]

Regression I in Table 12 shows the effects of fine, market and communication treatments on individual extraction in each round as compared to the baseline treatment. Accordingly, each of the fine and communication treatments significantly reduces individual extraction in each round compared to the baseline treatment. The effect of the market treatment is negative as well, but it is not found to be statistically significant.

In regression II three more independent variables are added: previous period's group average in the second stage (Group average t-1), each individual's average from the first stage (Individual average in the first stage) and each individual's extraction in the previous period of the second stage (Individual extraction t-1 in the second stage). These three variables turned out to be highly significant in all of our regressions as was expected. That is, an average individual seems to respond to an increase in group average in the previous period with increasing his/her own individual extraction. Moreover, each individual's average extraction over the 15 rounds in the first stage of the game is positively correlated with individual extraction in the second stage. And an average individual's extraction in period t seems to be positively correlated with the same person's extraction in period t+1 in the second stage. All treatment effects (fine, communication and market) were found to be negative and statistically significant compared to the baseline treatment in this estimation.

---

[59] Clustering is a widely used statistical tool in experimental economics in order to deal with heteroscedasticity and dependency of error terms within each group of observations.

In regression III I included two more independent variables regarding socio-economic characteristics of the participants, namely, age and education[60]. There is a slightly negative effect of age on individual extraction for each period. That is, older individuals seem to extract a little less than the younger ones. However, this effect is not statistically significant. There is a significantly positive effect on individual extraction of having attended the secondary school compared to having attended only the primary school. Again, all treatment effects are significantly negative, as the statistically significant coefficients of dummy variables for fine, communication and market demonstrate.

In regression IV, the period and group (session) fixed effects are added. That is, I added 13 dummy variables for period fixed effects, and 47 dummy variables for group/session fixed effects.[61] This changed both the sign and significance of market and communication treatments. That is, in regression IV, only the fine treatment was found to be significantly negative, while the treatment effect for market and communication disappeared. Yet, since there are 48 groups and 14 periods in total in this last regression, the degrees of freedom is reduced significantly, and as such, regressions seem to become very sensitive to the adding and removing of independent variables. Therefore, I have decided to conduct difference in difference analysis with the same variables, which would give more robust results.

[60] The results of the regressions including other socio-economic factors such as boat size, boat length, ownership etc are not shown here. This is because those factors were not found to be significant, and adding and removing them in the regressions does not change the main results of the study. Including them in the regressions, on the other hand, reduces the degrees of freedom without presenting any new insights. Therefore, I only added age and education as socio-economic factors with significant impact on individual extraction.

[61] Again, in order to prevent the dummy variable trap, the number of dummy variables used in the regressions is one less than the number of categories.

Table 12. Regression Analyses

	I	II	III	IV
Fine	-3.346***	-1.767***	-1.725***	-2.589***
	(0.862)	(0.305)	(0.300)	(0.954)
Market	-1.133	-0.828**	-1.022***	1.147
	(0.869)	(0.325)	(0.343)	(1.087)
Communication	-2.921***	-1.303***	-1.311***	0.154
	(0.713)	(0.295)	(0.310)	(0.523)
Group average t-1 (in the second stage)		0.119***	0.138***	-0.265***
		(0.039)	(0.040)	(0.050)
Individual average (in the first stage)		0.419***	0.413***	0.446***
		(0.045)	(0.044)	(0.049)
Individual extraction t-1 (in the second stage)		0.438***	0.426***	0.411***
		(0.034)	(0.033)	(0.034)
Age			-0.012	-0.013
			(0.012)	(0.013)
Education: Secondary			0.576**	0.717**
			(0.271)	(0.267)
Education: University			0.954	0.621
			(0.506)	(0.549)
Period fixed effects	No	No	No	Yes
Group fixed effects	No	No	No	Yes
Constant	12.202***	0.653	0.983	5.355***
	(0.507)	(0.464)	(0.883)	(1.332)
R squared	0.048	0.476	0.482	0.5334
F test	7.790	259.270	186.32	59.03
P value	0.000	0.000	0.000	0.000
N	3360	3360	3318	3318
Number of individuals	240	240	237	237

Dependent variable: Individual extraction in each round of the second stage
Robust standard errors (adjusted for clusters of individuals) are reported in parentheses.
***, **, and * indicate a significance level of 1%, 5% and 10%, respectively.

8.4 Difference in difference estimations: Identifying treatment effects

I have conducted difference in difference estimations to make pairwise comparisons between baseline and fine, baseline and market, baseline and communication, and finally, fine and market treatments, while controlling for group average t-1, individual average in the first stage, and individual extraction t-1 in the second stage, age, education as well as period and group (i.e. session) fixed effects.

The main idea of the difference in difference estimation is to subtract the average change in the non-treated group from the average change in the treated group, as Table 13 shows. This assumes that the treatment group is exposed to the treatment in the Post-period, but not in the Pre-period, as in line with the experimental design in the present study. The non-treated group is not exposed to any treatment, neither in Pre- nor in Post-period. The coefficient for the interaction term of the Post and the Treated (Post x Fine or Post x Market in Table 14) gives the treatment effect, i.e. if the coefficient for this interaction term, namely β_3, is significant, one can conclude that there is a treatment effect. Moreover, the sign and magnitude of the interaction term indicates the direction and size of the treatment effect.

The difference in difference estimation has the advantage of using the full data set with all periods in the first and second stage, where the pre-and post-treatment periods are distinguished from each other by a dummy variable "*Post*". This dummy variable takes the value 1 if the period belongs to the post-treatment stage and 0 if the period belongs to the pre-treatment stage. The "Treatment" itself is a dummy variable taking the value 1 if the observation comes from a treated individual and the value 0 if the observation comes from a non-treated (baseline) individual.

Therefore, three separate difference in difference models are estimated to compare each of the treatments, namely, fine, market and communication, to the baseline as the following model indicates:

$$Extraction_{i,t} = \beta_0 + \beta_1 Post_t + \beta_2 Treatment_i + \beta_3 (Post\ x\ Treatment)_{i,t} + \varepsilon$$

Table 13. The Coefficients of Interest in the Difference in Difference Model

	Pre	Post	Difference between Pre and Post
Non-treated	β_0	$\beta_0 + \beta_1$	β_1
Treated	$\beta_0 + \beta_2$	$\beta_0 + \beta_1 + \beta_2 + \beta_3$	$\beta_1 + \beta_3$
		Difference in Difference	β_3

The identifying assumption of the difference in difference estimation is that the measure of the baseline and treatment groups should have a similar trend in the absence of the treatment. This common trend assumption can be tested by the interaction of the pre-treatment periods and treatment dummies in the first stage of the game where there was no treatment. If the coefficients of these interaction terms turn out to be insignificant, then the common trend assumption is satisfied. This assumption is satisfied for our data; the interaction terms for pre-treatment periods and treatment dummies were not found to be statistically significant at 5% level in neither of the treatments.

Table 14. Difference in Difference: Baseline vs Fine & Baseline vs Market

Baseline vs Fine	I	II	Baseline vs Market	I	II
Post	0.994*	1.176**	Post	1.006*	1.052*
	(0.572)	(0.583)		(0.551)	(0.569)
Fine	-0.142	0.168	Market	-0.168	-1.926**
	(0.095)	(0.722)		(0.118)	(0.802)
Post x Fine	-2.212***	-2.501***	Post x Market	-1.171***	-1.210***
	(0.363)	(0.408)		(0.436)	(0.449)
Group average t-1	0.152***	0.094	Group average t-1	0.058	0.062
	(0.042)	(0.058)		(0.045)	(0.051)
Individual average (in the first stage)	0.681***	0.721***	Individual average (in the first stage)	0.683***	0.739***
	(0.057)	(0.058)		(0.053)	(0.050)
Individual extraction t-1	0.249***	0.230***	Individual extraction t-1	0.251***	0.226***
	(0.040)	(0.042)		(0.042)	(0.040)
Age	-0.016	-0.021*	Age	-0.006	0.002
	(0.010)	(0.012)		(0.010)	(0.012)
Education: Secondary	0.484**	0.735***	Education: Secondary	0.331	0.407
	(0.212)	(0.198)		(0.240)	(0.255)
Education: University	0.553	0.013	Education: University	-0.034	0.529
	(0.785)	(0.579)		(0.464)	(0.680)
Period fixed effects	Yes	Yes	Period fixed effects	Yes	Yes
Group fixed effects	No	Yes	Group fixed effects	No	Yes
Constant	-0.186	1.028	Constant	0.201	-0.325
	(0.774)	(1.265)		(0.840)	(0.985)
R squared	0.500	0.513	R squared	0.4697	0.4804
F test	67.090	98.47	F test	86.23	94.09
P value	0.000	0.000	P value	0.0000	0.0000
N	3332	3332	N	3304	3304
Number of individuals	119	119	Number of individuals	118	118

Dependent variable: Individual extraction per period
Robust standard errors (adjusted for clusters of individuals) are reported in parentheses.
***, **, and * indicate a significance level of 1%, 5% and 10%, respectively.

The first part of Table 14 analyzes the treatment effect of the fine as opposed to the baseline, where the dependent variable is individual extraction per period. In model I, the coefficient for the interaction term between Post x Fine is found to be negative and statistically significant, and this continues to hold when group fixed effects are added to the model in model II. That is, fine treatment reduces individual extraction by about 2.5 units and this is a significant difference compared to the (non-treated) baseline treatment. In model I, in the absence of the group fixed effects, the group average t-1 is highly significant but this effect diminishes once group fixed effects are added in model II. This was expected since group averages can be seen as an indication of group fixed effects. In model II, individual average in the first stage and individual extraction in t-1 continue to be significant factors affecting individual extraction. The effect of age on individual extraction is negative and significant at 10%. That is, older individuals tend to extract a little less than the younger ones. Secondary education as opposed to primary education seems to have a significantly positive effect on individual extraction.

The second part of Table 14 analyzes the treatment effect of the market. Again, the dependent variable is individual extraction per period. For this analysis I use the data from individuals under the baseline and market treatments. Again, in model I, the coefficient for the interaction term between Post x Market is negative and this difference is statistically significant. Both model I and II indicate that individuals treated with market seem to extract, on average, 1.2 units less than the individuals under the baseline treatment. Individual average in the first stage and individual extraction in t-1 are significantly positive.

Table 15. Difference in Difference: Baseline vs Communication

Baseline vs Communication	I	II
Post	1.095*	1.230**
	(0.612)	(0.617)
Communication	-0.167	-1.280***
	(0.115)	(0.367)
Post x Communication	-1.620***	-1.851***
	(0.416)	(0.423)
Group average t-1	0.171***	0.095
	(0.048)	(0.060)
Individual average (in the first stage)	0.693***	0.718***
	(0.055)	(0.058)
Individual extraction t-1	0.195***	0.185***
	(0.037)	(0.040)
Age	-0.011	-0.017
	(0.009)	(0.013)
Education: Secondary	0.301	0.298
	(0.231)	(0.210)
Education: University	0.645	-0.049
	(0.392)	(0.353)
Period fixed effects	Yes	Yes
Group fixed effects	No	Yes
Constant	0.010	1.543
	(0.853)	(1.435)
R squared	0.429	0.442
F test	59.37	72.57
P value	0.000	0.000
N	3304	3304
Number of individuals	118	118

Dependent variable: Individual extraction per period
Robust standard errors (adjusted for clusters of individuals) are reported in parentheses.
***, **, and * indicate a significance level of 1%, 5% and 10%, respectively.

Table 15 provides a similar analysis for the treatment effect of the communication compared to the baseline. Again, the coefficient for the interaction term of Post x Communication is negative and statistically significant, implying that communication was effective to reduce individual extraction. The effect continues to be significant after adding group fixed effects in model II. That is, being subject to communication treatment reduces an average individual's extraction by around 1.8 units according to model II.

Consequently, the difference in difference analyses demonstrate that being treated with either fine, market or communication reduces individual extraction significantly compared to the baseline treatment. In order to test whether there is any framing effect with respect to the fine and the market treatments, I have included a new interaction term Post x Treated x FM. The following new model aims to analyze whether there is any framing effect:

$$Extraction_{i,t} = \beta_0 + \beta_1 Post_t + \beta_2 Treated_i + \beta_3\, FM + \beta_4\, (Post\ x\ Treated)_{i,t} + \beta_5\, (Post\ x\ Treated\ x\ FM)_{i,t} + \varepsilon$$

In this model, "Post" takes the value 1 if the observation comes from a post-treatment stage and 0 otherwise, "Treated" takes the value 1 if the subject received any treatment (either fine or market) and 0 otherwise (baseline), and, finally, FM takes the value 1 if the treatment was the fine treatment and 0 if the treatment was market treatment. In other words, we can now differentiate how much of the reduction in individual extraction is attributable to the negative incentive (payment of 2 Kuruş for each unit of extraction exceeding the quota) and how much to the framing effect.

Table 16. Difference in Difference: Fine vs Market

Fine vs Market	I	II
Post	0.895*	1.010**
	(0.461)	(0.474)
Treated (Treated=1, Non-Treated=0)	-0.249**	-0.415
	(0.119)	(0.883)
FM (FM=1 if Fine, FM=0 if Market)	0.134	0.660
	(0.121)	(1.040)
Post x Treated	-1.01**	-1.102**
	(0.408)	(0.428)
Post x Treated x FM	-1.218**	-1.316**
	(0.510)	(0.542)
Group average t-1	0.115***	0.089**
	(0.036)	(0.044)
Individual average (in the first stage)	0.636***	0.689***
	(0.044)	(0.046)
Individual extraction t-1	0.283***	0.258***
	(0.035)	(0.035)
Age	-0.008	-0.007
	(0.009)	(0.010)
Education: Secondary	0.429**	0.598***
	(0.196)	(0.192)
Education: University	0.269	0.538
	(0.472)	(0.564)
Period fixed effects	Yes	Yes
Group fixed effects	No	Yes
Constant	-0.194	0.156
	(0.673)	(0.923)
R squared	0.4992	0.5120
F test	105.19	112.07
P value	0.000	0.000
N	4984	4984
Number of individuals	178	178

Dependent variable: Individual extraction per period
Robust standard errors (adjusted for clusters of individuals) are reported in parentheses.
***, **, and * indicate a significance level of 1%, 5% and 10%, respectively.

As Table 16 demonstrates, being treated with either the fine or the market reduces individual extraction by around 1.1 units according to model II. However, if the individual was subject to a fine treatment, the reduction is even bigger: individual extraction is further reduced by around 1.3 units. This is shown by the negative and statistically significant coefficient of the interaction term Post x Treated x FM. This seems to be in line with the comparison baseline vs fine in Table 14. That is, based on Table 16 above, an overall reduction of 2.4 units can be decomposed into two effects: reduction of 1.1 units is related to the negative incentive (either as a fine or market), the remaining 1.3 is related to the framing effect. Therefore, facing the negative incentive, an average individual tends to reduce extraction, but when this negative incentive is framed as a fine, this further reduces individual extraction. Therefore, this tends to confirm our Hypothesis 2, that there is strong evidence for a framing effect between the fine and market treatments, indicating that it substantially matters whether an economic incentive is framed as a fine or as a market-like incentive.

8.5 Classification of types

Based on Carpenter & Seki (2005) and Carpenter et al. (2009), I ran a random coefficients model to estimate conditional and unconditional cooperation coefficients for each individual. This enables us to identify different types of individuals, namely, selfish, altruistic, reciprocal, and both altruistic and reciprocal types as based on the first stage of the game. This is done by regressing each individual's extraction in each period, $x_{i,t}$, on the group average in the previous period, on $\overline{x_{t-1}}$ (for the first stage of the game). The model that I have estimated is therefore the following:

$$x_{i,t} = a_i + b_i \, \overline{x_{t-1}} + \varepsilon_{i,t} \quad for\ periods = 1, \ldots, 15$$

where $x_{i,t}$ denotes each individual i's extraction in each period t, a_i is the intercept for each individual, and therefore a measure for the unconditional cooperation, b_i is the coefficient for $\overline{x_{t-1}}$, and as such it may be interpreted as individual propensity to reciprocate (i.e., conditional cooperation), given that each individual responds to the group extraction in period $t - 1$, and $\varepsilon_{i,t}$ is the error term.

The estimation of the random coefficients model provides the following results:

At the population level, the coefficient for the group average in the t-1 is 0.16 and the constant is 9.77, both of them statistically significant at 1%. That is, on average, an individual would extract 9.77 units even if group average in t-1 was zero units. And the positive coefficient on group average in t-1 implies that on average individuals respond to increasing group average in t-1 with increasing their own extraction (and vice versa). Hence, the individual constants can be interpreted as a measure of altruism/spite, and individual slopes on group average in t-1 as a measure of reciprocity (as in line with Carpenter et al., 2009; Carpenter & Seki, 2011). On the other hand, results of the random coefficients model imply that individual coefficients of conditional cooperation, b_i's, vary from -1.31 to 1.63 with a mean of 0.17. And individual coefficients of conditional cooperation, a_i's, vary from -8.80 to 31.29, with a mean of 9.60.

I classify those individuals with an altruism/spite coefficient above the mean of 9.60 and a reciprocity coefficient below zero, as "selfish" individuals. Individuals with a negative reciprocity estimate, but a below mean spite/altruism parameter are classified as "altruists", while individuals with positive reciprocity estimates and above mean spite/altruism estimate are classified as “reciprocators”. Accordingly, individuals with positive reciprocity coefficients and below average altruism/spite estimate are identified as “altruistic and reciprocal” individuals. This is demonstrated in Figure 28 below. In this figure, each individual who has participated in the experiment is represented by a dot with a different color. Four of the subjects in the game extracted consistently the maximum possible amount (20 units) for all rounds in the first stage of the game. The model does not provide any estimates for them, and as such they are not represented in the below figure, but they can be classified among the most selfish individuals without any reciprocal preferences.

According to this classification, there are 77 selfish individuals (including those four individuals who have consistently extracted 20 units each in the first stage), 13 altruists, 40 reciprocators, and 110 individuals belong to the group of altruist and reciprocator individuals. That is, conditional cooperators (those who have a positive reciprocity coefficient) amount to 150 individuals in total in our sample.

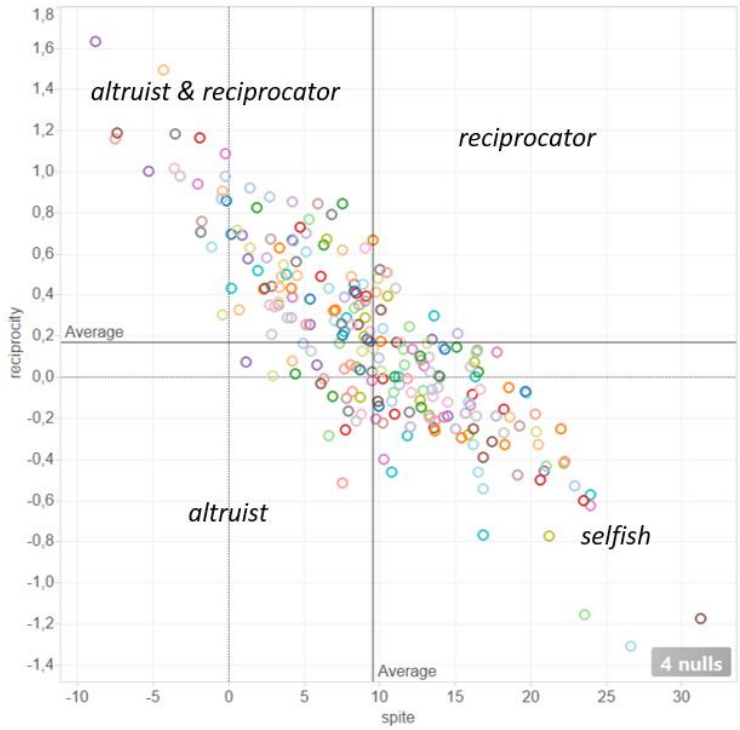

Figure 28. Classification of types

It is also interesting to observe how different type of individuals responded to the treatments in the second stage. That is, whether these types are stable or, in contrast, change in response to treatments is another question that was tested. To this end, first the graphs in Figure 29 were inspected. Then, I have conducted difference in difference analyses separately for selfish and non-selfish individuals, as the section below demonstrates.

8.6 Responses to treatments by selfish and non-selfish individuals

Next, I have investigated whether there was any difference between selfish and non-selfish individuals with respect to treatments. Based on the classification of types in Figure 28, I have regrouped individuals as selfish and non-selfish ones (i.e. those who are altruists, reciprocators, and individuals who are both altruists and reciprocators are reclassified as non-selfish individuals).

According to Figure 29, the baseline treatment does not seem to change the average extraction of selfish and non-selfish individuals, although there seems to be a slight positive trend for non-selfish individuals in the second stage of the game under the baseline treatment. The selfish individuals extract consistently higher than the non-selfish ones under the baseline treatment, and their average extraction does not seem to change in the second stage. On the other hand, both the fine and the communication treatments seem to be effective to reduce average extraction of selfish individuals, while market treatment does not seem to have the same effect. Under the market treatment, average extraction of selfish individuals stays rather stable, while for non-selfish individuals there is a slight fall in average extraction in the initial periods of the market treatment; however, this effect seems to disappear towards the end of the game. Fine treatment, on the other hand, seems to be effective to reduce average extraction of both the selfish and non-selfish individuals. So, overall the visual inspection of Figure 29 seems to suggest the following points:

- Communication treatment is effective to reduce average extraction for selfish individuals, and only slightly effective for non-selfish individuals.
- Fine treatment is effective for both types of individuals.
- The effect of market treatment on average behavior seems to have a slight effect to reduce average extraction for both types of individuals, but the effect is not that clear as under the fine treatment.

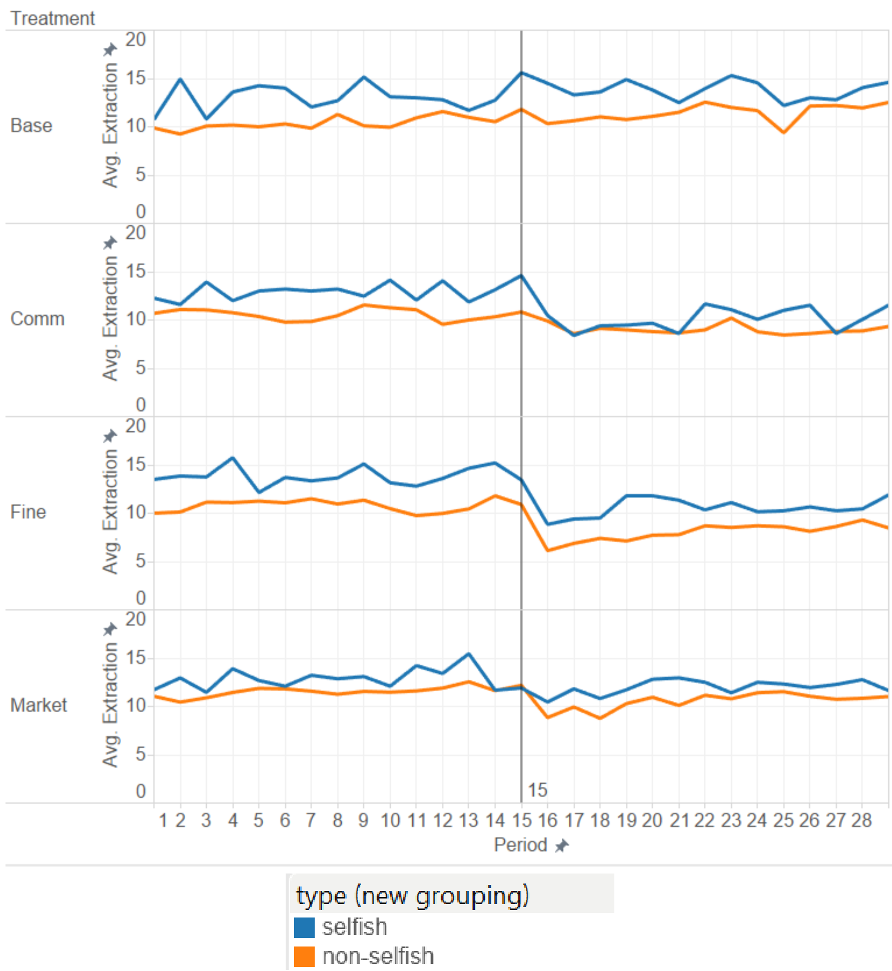

Figure 29. Average extraction of selfish vs non-selfish individuals across treatments

To test these visual preliminary findings, I have estimated difference in difference models separately for the subgroups of selfish and non-selfish individuals. In line with the graphs in Figure 29, the fine treatment was found to be very effective to reduce extraction of both selfish and non-selfish individuals ($p < 0.01$). However, with respect to the market treatment, the results are rather different. As Table 17 shows, the coefficient for the interaction term Post x Market is not found to be statistically significant for selfish individuals ($p > 0.10$), while it is highly significant for non-selfish individuals ($p < 0.05$). That is, the market treatment is not effective to reduce extraction of an average selfish individual significantly, while an average non-selfish individual seems to have reduced his/her extraction as a response to the market treatment.

Table 17. Response by Selfish vs Non-Selfish Individuals to Fine and Market Treatments (As Opposed to the Baseline)

Baseline vs Fine	selfish	non-selfish	Baseline vs Market	selfish	non-selfish
Post	0.059	1.443**	Post	-0.319	1.813***
	(1.036)	(0.676)		(1.001)	(0.611)
Fine	-0.362**	-0.023	Market	-0.071	-0.133
	(0.179)	(0.135)		(0.204)	(0.158)
Post x Fine	-2.532***	-2.040***	Post x Market	-1.370	-1.073**
	(0.654)	(0.438)		(0.897)	(0.481)
Group average t-1	0.123*	0.167***	Group average t-1	-0.111	0.129**
	(0.072)	(0.053)		(0.083)	(0.050)
Individual average (in the first stage)	0.739***	0.653***	Individual average (in the first stage)	0.886***	0.579***
	(0.127)	(0.063)		(0.081)	(0.062)
Individual extraction t-1	0.205***	0.276***	Individual extraction t-1	0.139**	0.330***
	(0.070)	(0.049)		(0.068)	(0.047)
Age	-0.023	-0.014	Age	0.004	-0.008
	(0.017)	(0.012)		(0.019)	(0.011)
Education	0.353	0.529**	Education	0.161	0.226
	(0.475)	(0.244)		(0.345)	(0.243)
Period fixed effects	Yes	Yes	Period fixed effects	Yes	Yes
Group fixed effects	No	No	Group fixed effects	No	No
Constant	1.498	-1.721*	Constant	1.602	-1.195
	(1.226)	(1.014)		(1.647)	(1.068)
R squared	0.436	0.5048	R squared	0.3906	0.5161
F test	50.700	51.87	F test	109.06	85.86
P value	0.000	0.000	P value	0.000	0.000
N	1120	2212	N	1176	2128
Number of individuals	40	79	Number of individuals	42	76

Dependent variable: Individual extraction per period
Robust standard errors (adjusted for clusters of individuals) are reported in parentheses.
***, **, and * indicate a significance level of 1%, 5% and 10%, respectively.

Moreover, communication is found to be effective to reduce extraction for both selfish and non-selfish individuals significantly, as the coefficient for the interaction term Post x Communication is negative and significant for both types of individuals. One has to note that the coefficient for the selfish individuals is larger for the non-selfish individuals, though. Hence, communication seems to be more effective to reduce extraction for selfish individuals, as Table 18 demonstrates.

Table 18. Response by Selfish vs Non-Selfish Individuals to Communication Treatment (As Opposed to the Baseline)

Baseline vs Communication	selfish	non-selfish
Post	-0.115	-0.208
	(0.524)	(0.347)
Communication	-0.329	-0.132
	(0.239)	(0.113)
Post x Communication	-3.195***	-1.250***
	(0.988)	(0.437)
Group average t-1	-0.050	0.228***
	(0.091)	(0.046)
Individual average (in the first stage)	0.709***	0.679***
	(0.109)	(0.065)
Individual extraction t-1	0.103	0.230***
	(0.065)	(0.044)
Age	-0.011	-0.011
	(0.016)	(0.011)
Education	0.610	0.231
	(0.382)	(0.173)
Period fixed effects	Yes	Yes
Group fixed effects	No	No
Constant	2.627	-1.588*
	(2.369)	(0.866)
R squared	0.298	0.4491
F test	28.730	177.80
P value	0.000	0.000
N	952	2352
Number of individuals	34	84

Dependent variable: Individual extraction per period
Robust standard errors (adjusted for clusters of individuals) are reported in parentheses.
***, **, and * indicate a significance level of 1%, 5% and 10%, respectively.

8.7 Discussion of the results

The findings reveal that, on average, there is substantial amount of intrinsic motivation to cooperate among the participants of the study. Even under the baseline treatment, both in the first and the second stages of the game, there is a significant level of cooperation which does not erode even in the last rounds of the game. This seems to be in line with the statements that one encounters in informal interviews with members of small-scale fishing communities. Small-scale fishers usually state that they pay a great deal of attention to sustainability of the resource and complain about excessive "greedy" fishing activities of industrial fishing fleets. These self-reported statements of small-scale fishers about their own cooperative behavior seems to be in line with experimental findings. Yet, obviously, one needs to be careful about generalizability of this preliminary evidence.

There is also some evidence that fishers do not want to exploited by their peers, either. That is, even though they understand that extraction of a small amount from the lake increases the overall payoff for each one of them in the game, they first want to observe how others respond and then they adjust their behavior accordingly. This is suggested by the evidence that 19.6% of the participants extracted in the first period of the game between 9-11 units, i.e. somewhere in the middle of the choice set which was between 0 and 20 units, while only 1.25 % (3 individuals) extracted 0 units in the first period of the game. On the other hand, most participants did not jump towards the other extreme, to 20 units, in the first period. Only 8.75% of fishers extracted 20 units in the first period. This seems to be a first indication of conditional cooperation among our participants. Yet, as the random coefficients model that I have estimated demonstrates, there is some variability regarding the types of participants. While 150 of the participants (62.5%) are identified as conditional cooperators, 77 were identified as selfish (32% of participants) and only 13 were identified as altruists (5.4%). These fractions are roughly consistent with previous findings in the literature (e.g. Fischbacher, Gachter, & Fehr, 2001; Rustagi, Engel, & Kosfeld, 2010).

Interestingly, individual extraction both under the fine and the market treatments was lower compared to the baseline treatment, when controlling for other factors such as age, education and period and group fixed effects. Theoretically, this should not be the case since both of these material disincentives are set at a very low level so that no one should deviate from the Nash best response of 20 units. The finding regarding the effect of a symbolic low fine lends support to Hypothesis 1 of the present study. It is also in line with some recent findings on that matter (Cardenas & Ostrom, 2004; Cardenas, 2011; Rodríguez-Sickert et al., 2008). This effect was presumably related to normative aspects of the fine (Frey & Stutzer, 2008; Rode et al., 2015). That is, even though the fine was only symbolic and did not change the underlying incentive structure to over-extract the resource, it implied that over-fishing is not the right thing to do in this context. Another factor may have been at work as well: In line with the findings of Rodriguez-Sickert and Guzmán (2008), the symbolic fine was probably effective to convince the conditional cooperators among the participants that over-extraction will not go unpunished. With this expectation, conditional cooperators may have reduced their own extraction.

The market treatment had a similar effect on average extraction. That is, even though the market price of extraction rights was low, individual extraction fell down compared to the baseline. This seems to be puzzling, though. The market treatment does not involve any normative signal with regard to the right behavior to adopt, in contrast to the fine. Still, the market price of the fishing rights probably implied that there was no open access regime anymore.[62] That is, fishing was not free anymore, the subjects were required to share a certain amount of their payoff with the experimenter when they overfished. Therefore, even though the market price was very low, it was signaling that there was no free lunch, so that the average subject decreased his/her extraction.

But the fall in individual extraction due to the fixed market price of the fishing rights was still smaller than the fall due the fine. This difference can be traced back to the absence of any strong normative message in the case of the market treatment. That is, while a fine is to be avoided from a normative perspective, the payment for a fishing right may imply, to the contrary, that over-extraction is normatively all right as long as one pays for it. Previous experimental studies did not provide any evidence to hint to a similar finding regarding the market treatment. To the best of my knowledge, no experimental study has specifically focused on the effect of a fixed market price of an extraction right in a resource use context. Second, framing effects have not been studied so far to investigate whether there is any difference between an incentive framed in a market-like language, as opposed to the same incentive framed in a language implying a direct regulation such as fines or penalties. The only specific hypothesis that had been theoretically formulated but not been tested empirically was by Frey and Stutzer (2008), which suggested that tradable permits will strongly crowd-out environmental morale. This was partially confirmed by the present study. That is, the market-like incentive reduced average extraction instead of crowding-out cooperation, but the reduction was

[62] In fact, fisheries management in Turkey is dominated by an understanding of open-access. “The sea is ours”, in other words, the access to seas is seen as “free”, despite there are several limitations regarding seasons, equipment, and so on. This open-access regime has been endorsed by state authorities since the 1950s (Knudsen, 2009).

lower than under the fine treatment even though the level of the material incentive was the same. Therefore, this finding lent support to Hypothesis 2 in the present study. Controlling for other factors, the disincentive framed as a fine reduced extraction by around 2 units, whereas the market reduced individual extraction by around 1 unit, despite the fact that the material disincentive was exactly the same. This implies that there is a framing effect between the fine and market treatments. In sum, the findings indicate that the fall in extraction can be decomposed into two effects: one is the effect of the incentive implying that the regime has changed from an open-access to a more restricted case in which fishing is not free anymore. The second effect is the framing effect, i.e. the difference between the two normative messages of the fine and the market. While the fine is sending the message that it is normatively not appropriate to over-fish, the market does not convey the same message. Instead, it presumably implies that one can exceed the quota as long as one pays for it.

The effect of face-to-face communication on cooperation and average payoffs was positive, confirming our Hypothesis 3. That is, communication seems to be an effective tool to reduce extraction and to increase overall payoffs. This is in line with the consensus in the experimental literature on the effect of communication so far (Ahn et al., 2010; Cardenas et al., 2000; Ostrom et al., 1992). Although the internal dynamics of this effect are not well-known yet, it has been suggested that factors like familiarity, empathy and trust may play a role (Sally, 1995). Even though trust towards other individuals and institutions is low in Turkey, interpersonal trust among the members of the small-scale fishing communities is likely to be stronger. Small-scale fishing communities are usually organized in fishery cooperatives established and physically settled around the fishing harbor of each village or neighborhood. Small-scale fishers spend time with each other on a day-to-day basis for many years around these harbors, either in local *kahve*s or community rooms or cafeterias of fishery cooperatives. This probably increases familiarity and degree of trust among the members of fishing communities, which can render communication effective. It is interesting to note that except for a few groups, groups assigned to the communication treatment had some conversation about the game only at the beginning of the second stage. Despite the fact that they were allowed to communicate prior to each round in the second stage, most of them did not make use of this opportunity. Even though communication was therefore just a one-shot experience for some of the groups, on average, introduction of communication into the experiment was an effective intervention to reduce average extraction. The findings show that the effect of a symbolic low fine and of communication on extraction seem to be similar. However, communication is more effective to increase average payoffs since in that case participants approach social optimum without foregoing some of their payoffs, while under the fine treatment, participants need to pay a fine due to exceeding the quota, even though extraction is lower like under the communication treatment.

I have classified the participants according to their extraction in the first stage of the game, where there was open-access. The aim was to study whether the types change in response to treatments. The graphical analyses implies that the types are rather stable, but still there seems to be a difference between the responses of selfish and non-selfish individuals to different treatments. Selfish individuals did not respond much to the market treatment, whereas fine treatment was very effective for both selfish and non-selfish individuals. Communication, on the other hand, was very effective for selfish individuals and a little less effective for non-selfish individuals.

To the best of my knowledge, the present study provides the first behavioral finding comparing a framing based on a command-and-control type intervention, as opposed to another framing resting on a market permit, in resource use dilemmas. Further studies are needed to test the framing effect across different resource use groups as well as across cultures. Moreover, the study implies that different individuals (selfish as opposed to non-selfish) may respond differently to different policy interventions. Treating all individuals with a one-size-fits-all approach may not be very fruitful, or may even be a waste of scarce resources. The findings also tend to suggest that selfish individuals may be better approached with direct regulation, but not so much with market-based instruments.

The results of the present study can have implications for real life policies on fishing as well as other types of common-pool resources, in which environmental markets such as tradable quotas and marketable permits are popularly used. Yet, two issues need to be considered for the generalizability of the results:

First, experimental design in the present study focused on framing effects, but not on beliefs, which may have explanatory power for changing behavior when frames change. Recent studies suggest that frames change beliefs about others, and in turn, this change in beliefs alters individual behavior (e.g. Dufwenberg et al., 2011). This can be tested in case of resource use dilemmas by manipulating the beliefs externally and then measuring individual behavior using sequential public goods games, for instance.

Second, an assessment of external validity could constitute an important improvement area. The generalizability of results strongly hinges upon that. For that purpose, studies need to identify a real life individual measure for cooperation among resource users, which is then compared to the experimental measure of cooperation. There are studies that have, for instance, measured the mesh size of fishers, the membership to high-performing cooperatives, etc. as directly observable outside measures of cooperation. These studies demonstrated that there was a positive correlation between experimental and real-life measures of cooperation (e.g. Fehr & Leibbrandt, 2011; Gelcich, Guzman, Rodrigues-Sickert, Castilla, & Cardenas, 2013). Such an approach could not be adopted in the present study since most fishers were very reluctant to reveal any information regarding their catches, equipment, etc. These are very sensitive issues in fisheries in Turkey, since fishers usually fear that heavy taxes will be imposed on them when they reveal such data. Yet, figuring out ways to establish a link between the experimental and the real life cooperative behavior could further improve the results.

CONCLUSION

Given the increasing pressure on natural resources and the acceleration of ecological problems, this dissertation reviewed the literature on environmental policy instruments and contributed to it by adopting an experimental approach. In particular, it challenged the traditional policy analysis in the environmental realm by taking a closer look at the role of institutions, other-regarding behavior, and community-based solutions, such as face-to-face communication, using framed field experiments conducted with small-scale fishers in Turkey.

After a brief review of the traditional approach to environmental policy instruments, alternative views on environmental policies and experimental studies on environmental issues were discussed in Part I. The experimental design, data collection procedures and results of the framed field experiments were presented in Part II of the dissertation. More specifically, the framed field experiments were conducted with 240 participants, who were members of small-scale fishing communities in Turkey. There were four treatments: baseline, fine, market and communication. Twelve sessions were conducted for each of the treatments, and in each session, five participants played the public goods game that was framed as a common-pool resource use dilemma. The first stage of the game represented an open-access regime without any external intervention for all sessions. In the second stage of the game, each group was assigned randomly to one of the four treatments, and played the game either under the baseline, fine, market or communication treatment.

Overall, the results of the field study suggested that external fines and face-to-face communication are important tools for achieving a higher level of cooperation in resource use dilemmas. The results also implied that the way in which an economic incentive is framed matters a great deal. Although economic incentives play a significant role in attenuating resource use dilemmas, the existence of framing effects implies that the final outcomes of incentive-based instruments will not always coincide with the theoretical expectations of mainstream economic models or the intentions of policy-makers. This can be traced back to the existence of other-regarding behavior, which has so far been largely ignored in traditional evaluations of environmental policy instruments.

Indeed, even under the open-access condition (baseline treatment) there was a substantial level of cooperation, indicating the presence of other-regarding behavior. This finding aligns with similar field studies, which already challenged the mainstream assumption of purely-selfish individuals (e.g. Cardenas et al., 2000; Vollan, 2008). In line with several studies, introduction of face-to-face communication was an effective intervention to reduce extraction (Ahn et al., 2010; Cardenas, 2011; Ostrom et al., 1992; Sally, 1995). The field experiments also confirmed that the institutional context significantly affects individual economic decisions. External fines reduced extraction compared to the open access regime. The other institutional context, corresponding to a market-like incentive involving the purchase of extraction rights, also reduced extraction by transforming the decision context from open-access to a limited regime. Yet, individual extraction was higher compared to the fine treatment under this condition. This is presumably because, unlike the fine, the market did not trigger any moral considerations among participants, nor did it signal normatively appropriate behavior. This framing effect between the fine and the market conditions suggests that institutions involving economic incentives are likely to change both the underlying incentive structure as well as the decision context.

The field study is also one of the few experiments to assess the responses of selfish and non-selfish individuals to different policy interventions in resource use dilemmas. The main contribution of this analysis is to highlight the difference between the responses of selfish and non-selfish individuals to market-like incentives. A one-size-fits all approach that assumes opportunistic behavior and endorses market-based policies for all resource use contexts and all resource users may render environmental policies ineffective. Of course, to generalize these findings, more studies need to address this issue in the future.

Several aspects can be studied more in depth as future work. For instance, regional characteristics of fishing activities may have an effect on the results of the game. Fishers in the Marmara, Black Sea and Aegean regions may have acted differently in terms of extraction in the experiments. Moreover, the treatments may have a significant impact not only on the average outcomes, but also on the distribution of payoffs. This can be studied more in detail. Finally, certain characteristics of the fishery cooperatives may have an impact on the results. This is another aspect that can be studied by collecting more data on the cooperative level.

While the results of the present study contribute to a more thorough understanding and analysis of environmental policy instruments regarding framing effects and the behavioral impacts of markets, some issues related with the generalizability of the results need to be considered. First, the results of the experimental study are based on individuals, not on large-scale companies, government agencies, or non-governmental organizations. As such, the findings need to be evaluated with some level of caution. In certain contexts, individuals may feel an obligation to act as responsible citizens with civic duties, while in others, they may feel that being selfish is socially acceptable (Bowles, 1998; van den Bergh et al., 2000; Vatn, 2005). However, firms and larger organizations may be subject to other constraints, lacking the choice between citizen versus self-payoff maximization roles. Further studies may address this question by changing the subject pool from individual resource users to representatives of firms or governments and altering the rules of the game accordingly.

Second, the framing effect between a fine and a market-like incentive constitutes a first step towards understanding the behavioral aspects of markets in resource use dilemmas. Further advances can be made by studying more complicated market interactions involving an endogenous price formation mechanism with buyers and sellers actually trading quotas or permits. Market experiments are complex, requiring a higher degree of cognitive capacity and a group of subjects who have some prior experience participating in such experiments. Yet, the formal education level of the target groups—artisanal fishers, small-scale farmers, and other similar resource users—tends to be rather low. Therefore, simple versions of such experimental markets need to be designed. This is a challenging but crucial next step in studying markets and their effects on the behavior of resource users.

Lastly, future experimental studies can address the problem of larger groups in resource use dilemmas. The involvement of a large number of individuals in environmental problems, such as overfishing, climate change, and conservation, is another important topic for future research. No doubt, one can also introduce other real world phenomena, such as negotiation and transaction costs, in future work.

Experimental studies have the advantage of being flexible enough to incorporate these aspects of real world policy making.

Experimental studies can also be designed to test social norms, the performativity of economics, and other aspects of community-based management that were presented in Chapter 3. For instance, the role of participatory mechanisms and deliberation can be investigated with the help of experimental studies to improve the understanding of within-group communication. Since participation and deliberation are important to policy makers, these types of studies are needed to improve environmental decision making. There are several problems in implementation regarding the distribution of power in the community, rent-seeking behavior, and the exclusion of marginalized groups. The main advantage of experimental studies is that, if cleverly designed, they provide a controlled environment for researchers to deliberately manipulate a single variable and then assess the results. Moreover, the effect of policy mix approaches that combine different instruments to address an environmental problem may also be tested using experimental studies. Coupling external fines or market-like incentives with face-to-face communication may prove to be more effective than using one instrument at a time. Another aspect for future work to focus on is the content of communication. Communication is consistently found to be effective in resource use dilemmas, but the underlying reason for this finding is still discussed.

In conclusion, the value-added of framed field experiments conducted with actual resource users should be acknowledged in the environmental policy literature and by policy-makers. By using simple versions of the real world that represent inherent social dilemmas, framed field experiments can be revealing in many cases where one cannot analyze the outcomes of environmental policies based on aggregate data. The analysis of aggregate data is prone to difficulties due to confounding factors involving uncertainty about dynamic ecological systems. This is the case with carbon markets and individual transferable quotas, for instance. Empirical evidence about their environmental performance is mixed (Anderson et al., 2008; Betz, Rogge, & Schleich, 2006; Hepburn, 2007; T. Smith, Gibbs, & Smith, 2009). Even if a positive environmental outcome can be identified after the introduction of such markets, it cannot be ruled out that other factors have indeed contributed more to environmental improvement. However, behavioral studies allow for the investigation of this issue in a simpler manner, and present an invaluable opportunity to researchers to understand the behavioral and environmental impacts of different environmental policies. The framed field experiments presented in this study are the first to analyze the cooperative behavior of resource users in Turkey, and the results constitute a baseline to compare the findings of future experimental work on resource use and cooperation.

APPENDIX A

FRAMED FIELD EXPERIMENTS ON RESOURCE USE

Table 19. List of Various Framed Field Experiments on Resource Use Dilemmas

Authors	Country	Context	Type of game	Treatments
Alpizar & Gsottbauer, 2015	Costa Rica	Recycling	PG (with threshold)	Shame (bad reputation); pride (good reputation)
Aswani et al., 2013	Solomon Islands	Fishing	PG	Socio-economic factors
Barr et al., 2014	Albania	Time and effort allocated to participation	PG	Participation in accountability institutions
Bouma & Ansink, 2013	Costa Rica	Conservation	CPR (linear)	Penalty; communication; framing
Braaten, 2014	Peru	Land use	PG	Effect of ownership
Cardenas, 2000	Colombia	Forestry	CPR (non-linear)	Communication
Cardenas, 2003	Colombia	Forestry	CPR (non-linear)	Heterogeneity in real wealth
Cardenas, 2004	Colombia	Forestry	CPR (non-linear)	High penalty; low penalty; communication
Cardenas, 2011	Colombia	Forestry	CPR (non-linear)	High penalty; low penalty; communication (repeated); communication (one-shot)
Cardenas et al., 2000	Colombia	Forestry	CPR (non-linear)	Low penalty; communication
Cardenas et al., 2002	Colombia	Forestry	CPR (non-linear)	Asymmetric marginal returns to alternative activity
Castillo & Saysel, 2005	Colombia	Fishing	CPR (non-linear)	Communication; subsidy; tax
Castillo et al., 2011	Colombia, Thailand	Fishing	CPR (non-linear with spatial aspects)	Voting on an institution (random access, rotation, quota); context variables (trust among local fishers, trust to authorities)
Dannenberg & Martinsson, 2015	Ethiopia, Nepal	Forestry	PG	Externally-defined non-binding agreement
del Pilar & Maldonado, 2010	Colombia	Fishing	CPR (dynamic)	Co-management

Fehr & Leibbrandt, 2011	Brazil	Fishing	PG	Impatience; external validity
Gächter & Herrmann, 2011	Russia	No context	PG	Peer punishment
Gatiso et al., 2015	Ethiopia	Forestry	CPR (dynamic)	Penalty; peer punishment; resource availability
Gelcich et al., 2013	Chile	Fishing	CPR game (linear)	Confiscation of harvest (if quota is exceeded); dependence on resource in real life; external validity
Ghate et al., 2013	India	Forestry	CPR (non-linear)	Communication
Handberg & Angelsen, 2015	Tanzania	Forestry	CPR (dynamic)	Penalty; communication; individual reward
Hayo & Vollan, 2012	South Africa, Namibia	Grazing	CPR game (non-linear)	Voting on an institution (penalty, reward, communication)
Jack, 2009	Kenya	Conservation	Investment Game	Compensation payment between users
Janssen et al., 2011	Thailand, Colombia	Irrigation	CPR (non-linear with spatial dynamics)	Asymmetric access to resources
Javaid & Falk, 2015	Pakistan	Irrigation	CPR (non-linear with spatial dynamics)	Peer punishment; communication; institution choice (individual); communication with traditional authority
Lopez et al., 2012	Colombia	Fishing	PG	High penalty; low penalty; guilt; shame (bad reputation)
Maldonado & del Pilar, 2009	Colombia	Fishing	CPR (dynamic)	Penalty; communication; resource availability; co-management (fine & communication)
Mantilla, 2015a	Colombia	Fishing	CPR (non-linear)	Social network measures
Mantilla, 2015b	Colombia	Fishing	CPR (non-linear)	Communication (with controlled content)
Midler et al., 2015	Peru	Conservation	PG (impure)	Communication; individual reward; collective reward
Narloch et al., 2012	Bolivia, Peru	Conservation	PG (impure)	Collective reward; individual reward
(Pfaff et al., 2015)	Colombia	Water use	Dictator Game	Resource availability
Prediger et al., 2011	Namibia & South Africa	Grazing	CPR (non-linear with spatial dynamics)	Resource availability
Rustagi et al., 2010	Ethiopia	Forestry	PG	Share of types (conditional cooperators and free riders)

Saldarriaga-Isaza et al., 2015	Colombia	Small-scale gold mining	PG (with threshold)	Exclusion; co-management
Travers et al., 2011	Cambodia	Conservation	CPR (linear)	High penalty; low penalty; individual reward; collective reward
Velez et al., 2009	Colombia	Fishing	CPR (non-linear)	Testing for self-interest, altruism, reciprocity, inequity aversion, conformity
Velez et al., 2010	Colombia	Fishing	CPR (non-linear)	Communication; communication and low penalty; communication and medium penalty
Velez et al., 2012	Colombia	Fishing	CPR (non-linear)	Voting on an institution (low or medium fine); communication and low penalty; communication and medium penalty; communication and voting for medium or low penalty
Visser & Burns, 2015	South Africa	Fishing	PG (heterogeneous endowments)	Unequal endowments; peer punishment
Vollan, 2008	South Africa, Namibia	Grazing	Trust game & CPR game (non-linear)	Voting on an institution (gradually increasing penalty, randomized reward, communication)

PG denotes public goods games, CPR denotes common-pool resources games.

APPENDIX B

LIST OF SESSIONS

Table 20. List of 48 Sessions in the Study

Session No	Location	Treatment
1	Bakırköy Fishery Cooperative	Fine
2	Silivri Fishery Cooperative	Market
3	Silivri Fishery Cooperative	Market
4	Silivri Fishery Cooperative	Fine
5	Silivri Fishery Cooperative	Fine
6	Bakırköy Fishery Cooperative	Market
7	Selimpaşa Fishery Cooperative	Communication
8	Selimpaşa Fishery Cooperative	Market
9	Beykoz Fishery Cooperative	Baseline
10	Beykoz Fishery Cooperative	Market
11	Üsküdar Fishery Cooperative	Market
12	Üsküdar Fishery Cooperative	Baseline
13	Zeytinburnu Fishery Cooperative	Fine
14	Zeytinburnu Fishery Cooperative	Baseline
15	Zeytinburnu Fishery Cooperative	Communication
16	Fatih Fishery Cooperative	Fine
17	Fatih Fishery Cooperative	Baseline
18	Fatih Fishery Cooperative	Communication
19	Eskihisar Fishery Cooperative	Market
20	Antalya Fisheries Workshop	Baseline
21	Antalya Fisheries Workshop	Fine
22	Antalya Fisheries Workshop	Communication
23	Antalya Fisheries Workshop	Market
24	Antalya Fisheries Workshop	Communication
25	Antalya Fisheries Workshop	Fine
26	Antalya Fisheries Workshop	Baseline
27	Antalya Fisheries Workshop	Market
28	Antalya Fisheries Workshop	Baseline
29	Küçükmustafapaşa Fishery Cooperative	Communication
30	Küçükmustafapaşa Fishery Cooperative	Fine
31	Eminönü Fishery Cooperative	Fine
32	Eminönü Fishery Cooperative	Communication
33	Bakırköy Fishery Cooperative	Communication

34	Bakırköy Fishery Cooperative	Communication
35	Hereke Fishery Cooperative	Baseline
36	İstinye Fishery Cooperative	Baseline
37	Beykoz Fishery Cooperative	Baseline
38	Beykoz Fishery Cooperative	Fine
39	Güzelce Fishery Cooperative	Communication
40	Güzelce Fishery Cooperative	Market
41	Tuzla Fishery Cooperative	Baseline
42	Tuzla Fishery Cooperative	Fine
43	Eminönü Fishery Cooperative	Baseline
44	Şile Fishery Cooperative	Market
45	Şile Fishery Cooperative	Communication
46	İğneada Fishery Cooperative	Communication
47	İğneada Fishery Cooperative	Market
48	İğneada Fishery Cooperative	Fine

APPENDIX C

POSTERS

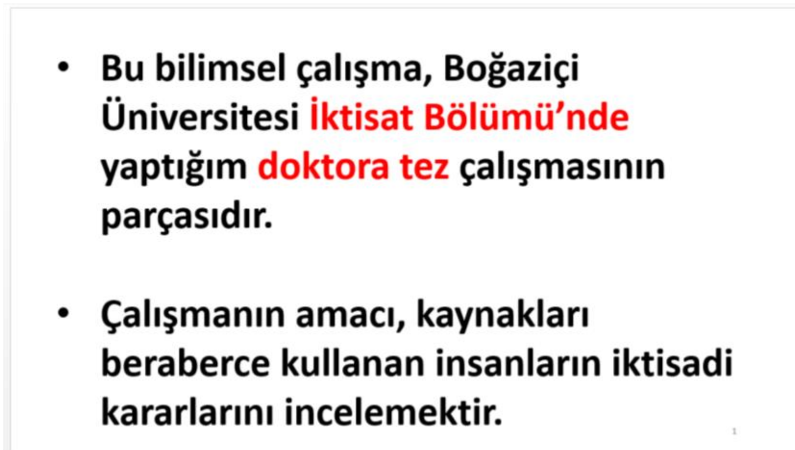

Figure 30. Poster 1

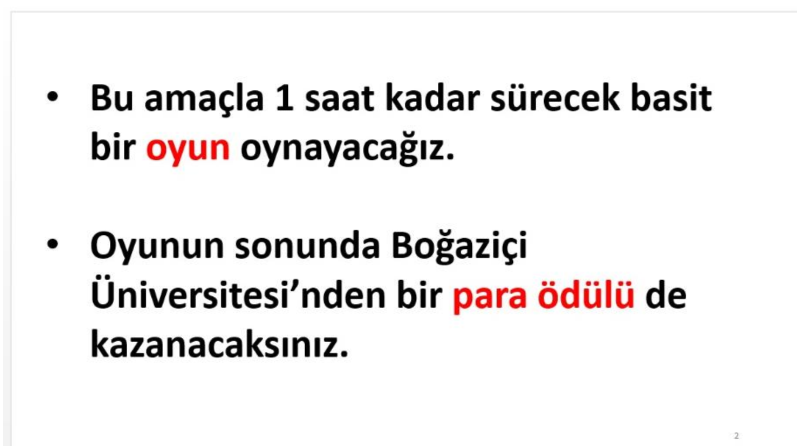

Figure 31. Poster 2

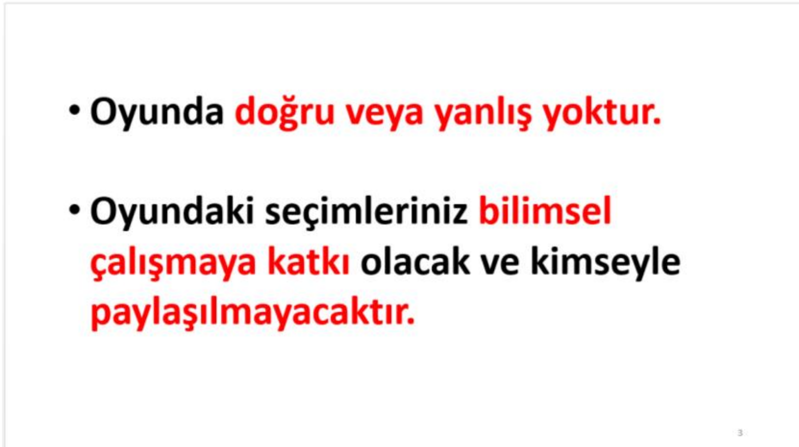

Figure 32. Poster 3

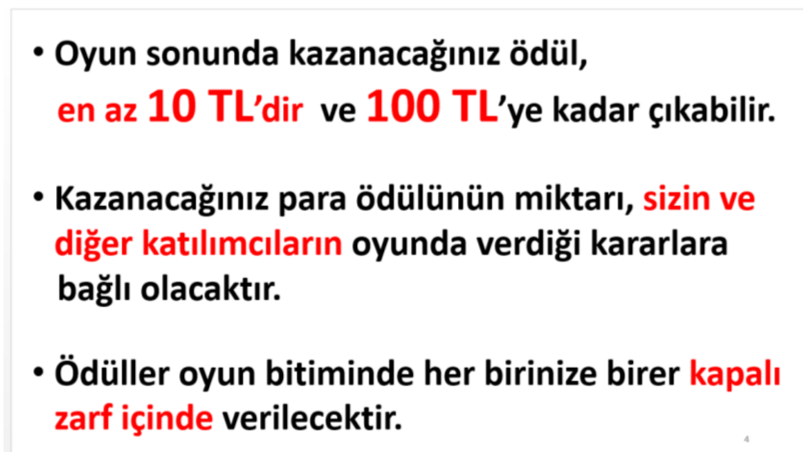

Figure 33. Poster 4

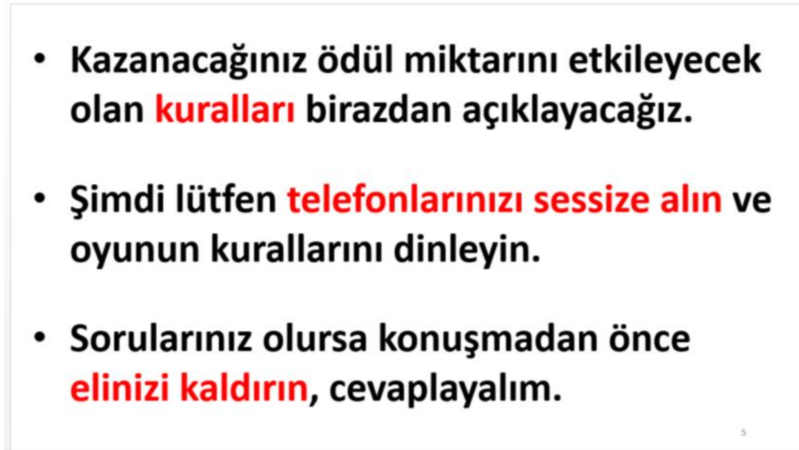

Figure 34. Poster 5

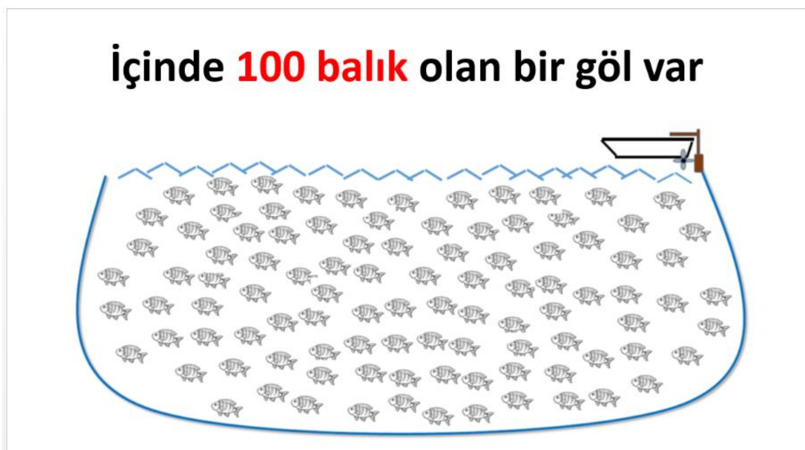

Figure 35. Poster 6

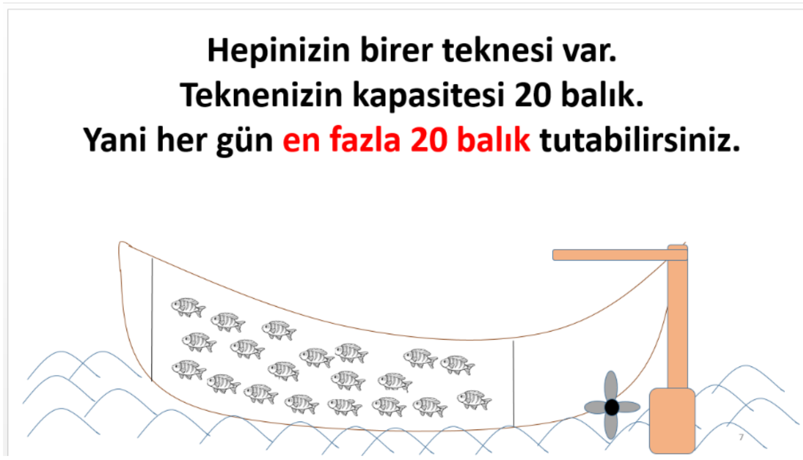

Figure 36. Poster 7

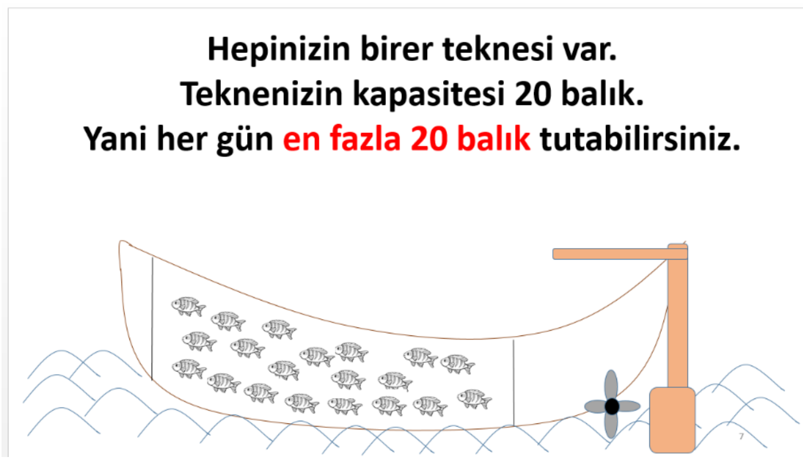

Figure 37. Poster 8

Figure 38. Poster 9

Figure 39. Poster 10

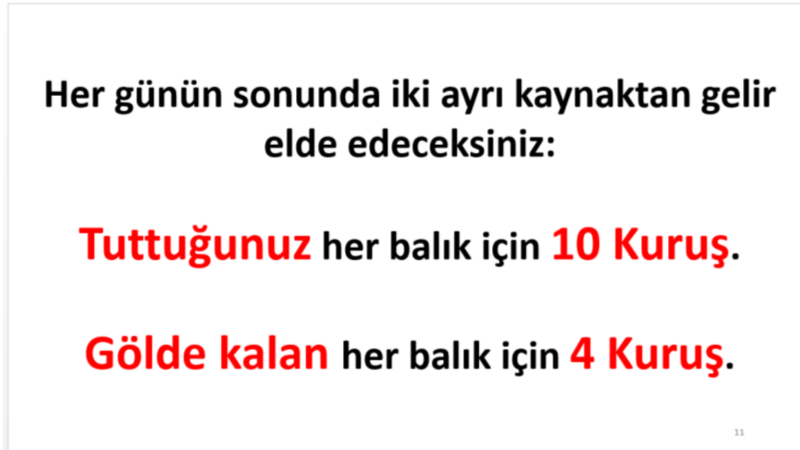

Figure 40. Poster 11

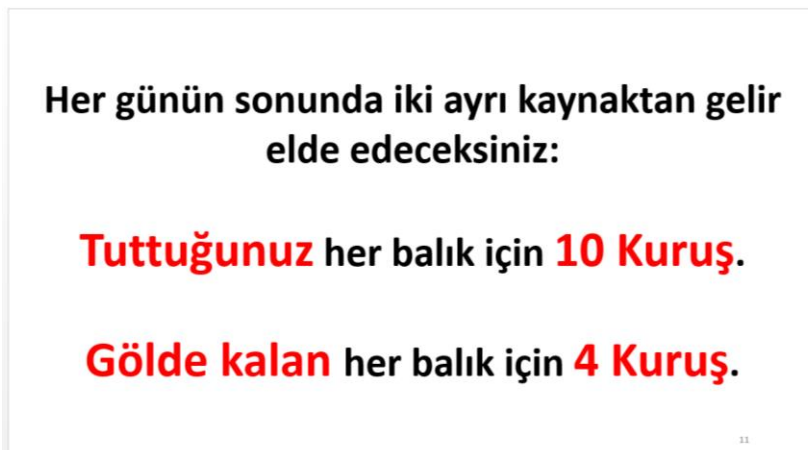

Figure 41. Poster 12

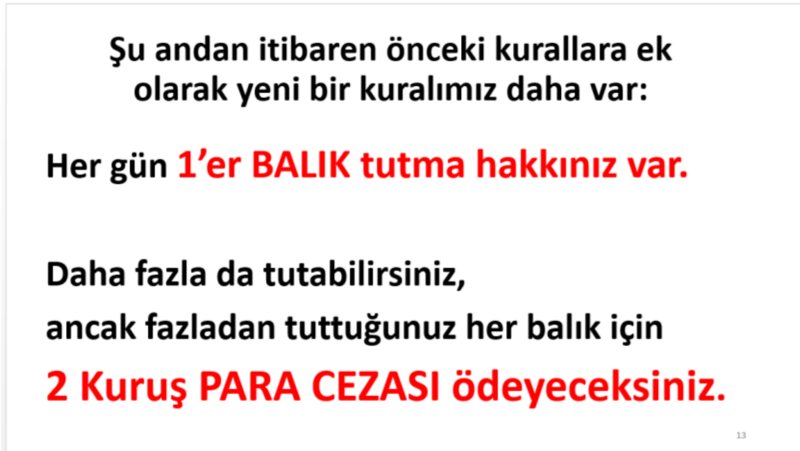

Figure 42. Poster 13

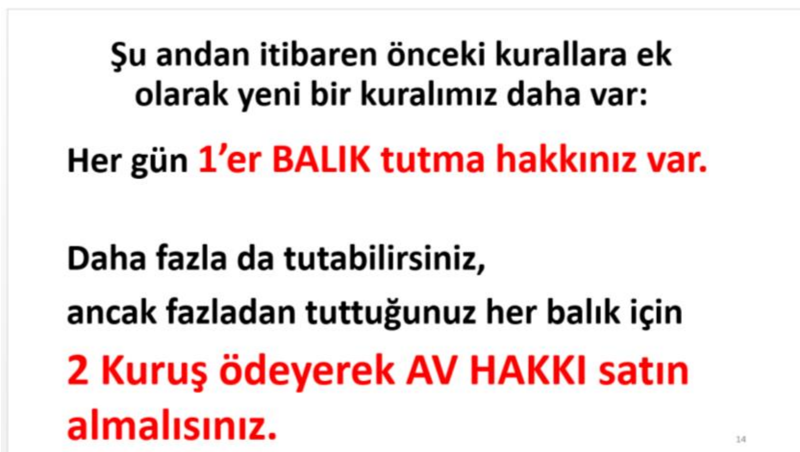

Figure 43. Poster 14

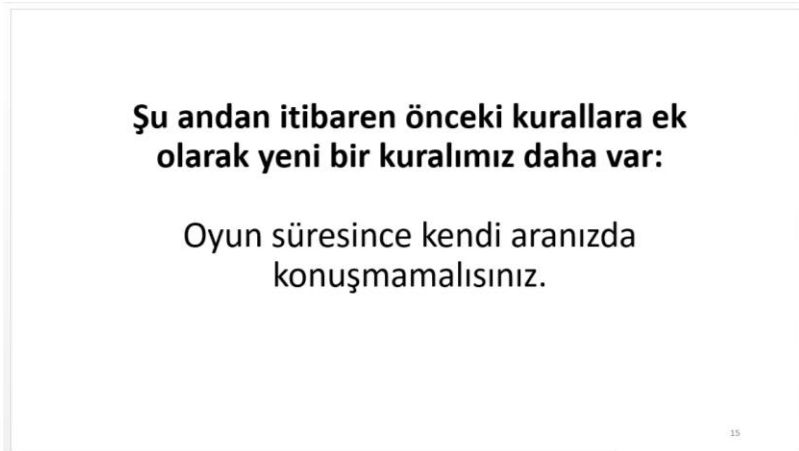

Figure 44. Poster 15

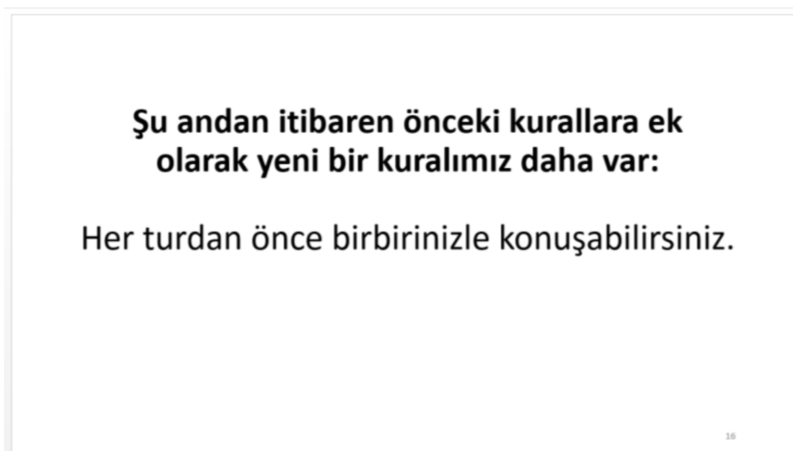

Figure 45. Poster 16

APPENDIX D

PARTICIPATION FORM

Araştırma Projesinin Adı: Ekonomik Kararların Analizi
Proje Yürütücüsü/Araştırmacının adı: Ar. Gör. Pınar Ertör
Adresi: Boğaziçi Üniversitesi Ekonomi Bölümü, 34342, Bebek/İstanbul
E-mail adresi: pinar.ertor@boun.edu.tr
Telefonu: 0 212 359 65 05

Proje Konusu ve izlenecek prosedür: Boğaziçi Üniversitesi Araştırma Fonu'nun desteklediği bu projede, bireylerin ekonomik kararları nasıl aldığını incelemeyi amaçlıyoruz. Çalışmaya katılım ödülü ve buna ek olarak çalışma esnasında kazandığınız para ödülü oyun sonunda nakit olarak ödenecektir. Yaptığınız seçimlere bağlı olarak kazanabileceğiniz para ödülü (10 TL'si katılım ödülü olmak üzere) 10 ile 100 TL arasında değişebilir.

Onam: Çalışmaya katılım tamamıyla gönüllüdür. Seçimleriniz tamamen gizli tutulacak ve sadece araştırmacılar tarafından değerlendirilecek; elde edilecek bilgiler bilimsel yayınlarda kullanılacaktır. Adınız/soyadınız hiç bir zaman verdiğiniz kararlarla ilişkilendirilmeyecek, yani veriler tamamen isimsiz şekilde, sadece size vereceğimiz ve kimliğinizle hiçbir ilgisi bulunmayan bir "katılımcı numarası" kullanılarak incelenecektir.

Çalışma, kişisel rahatsızlık verecek soruları içermemektedir. Ancak, katılım sırasında sorulardan ya da herhangi başka bir nedenden ötürü kendinizi rahatsız hissederseniz yarıda bırakıp çıkmakta serbestsiniz. Çalışmayı yarıda bırakıp çıksanız dahi 10 TL'lik katılım ödülü sizlere nakit olarak ödenecektir.

Bu çalışmaya katıldığınız için şimdiden teşekkür ederiz. Çalışma hakkındaki sorularınızı sormak veya daha fazla bilgi almak için Tel: 0 212 359 65 05 ya da E-posta: pinar.ertor@boun.edu.tr yoluyla iletişim kurabilirsiniz. Araştırmayla ilgili haklarınız konusunda yerel etik kurullarına da danışabilirsiniz.

Bana anlatılanları ve yukarıda yazılanları anladım. Bu formun bir kopyasını aldım. Bu çalışmaya tamamen gönüllü olarak katılıyorum. Yaptığım seçimlere ilişkin verilerin bilimsel amaçlı yayınlarda kullanılmasını kabul ediyorum.

(Formu doldurup imzaladıktan sonra uygulayıcıya geri veriniz).

Adı Soyadı	Tarih	İmza
	----/----/-----	

APPENDIX E

DECISION CARDS

BİRİNCİ BÖLÜM				
Katılımcı numarası:				
		Bu kısımlar asistanımız tarafından doldurulacaktır.		
Gün	**KAÇ BALIK tuttum?**	**TOPLAM kaç balık tutuldu?**	**GÖLDE kaç balık kaldı?**	**NE KADAR KAZANDIM?**
1. gün				
2. gün				
3. gün				
4. gün				
5. gün				
6. gün				
7. gün				
8. gün				
9. gün				
10. gün				
11. gün				
12. gün				
13. gün				
14. gün				
15. gün				
16. gün				

Figure 46. Decision card for the first stage

İKİNCİ BÖLÜM					
Katılımcı numarası:					
		Bu kısımlar asistanımız tarafından doldurulacaktır			
Gün	**KAÇ BALIK tuttum?**	**TOPLAM kaç balık tutuldu?**	**GÖLDE kaç balık kaldı?**	**NE KADAR CEZA ödedim?**	**Ceza ödedikten sonra NE KADAR KAZANDIM?**
1. gün					
2. gün					
3. gün					
4. gün					
5. gün					
6. gün					
7. gün					
8. gün					
9. gün					
10. gün					
11. gün					
12. gün					
13. gün					
14. gün					
15. gün					
16. gün					

Figure 47. Decision card for the fine treatment

İKİNCİ BÖLÜM					
Katılımcı numarası:					
		Bu kısımlar asistanımız tarafından doldurulacaktır			
Gün	**KAÇ BALIK tuttum?**	**TOPLAM kaç balık tutuldu?**	**GÖLDE kaç balık kaldı?**	**AV HAKKI SATIN ALMAK İÇİN ne kadar ödedim?**	**Av hakkı satın aldıktan sonra NE KADAR KAZANDIM?**
1. gün					
2. gün					
3. gün					
4. gün					
5. gün					
6. gün					
7. gün					
8. gün					
9. gün					
10. gün					
11. gün					
12. gün					
13. gün					
14. gün					
15. gün					
16. gün					

Figure 48. Decision card for the market treatment

İKİNCİ BÖLÜM				
Katılımcı numarası:				
		Bu kısımlar asistanımız tarafından doldurulacaktır.		
Gün	**KAÇ BALIK tuttum?**	**TOPLAM kaç balık tutuldu?**	**GÖLDE kaç balık kaldı?**	**NE KADAR KAZANDIM?**
1. gün				
2. gün				
3. gün				
4. gün				
5. gün				
6. gün				
7. gün				
8. gün				
9. gün				
10. gün				
11. gün				
12. gün				
13. gün				
14. gün				
15. gün				
16. gün				

Figure 49. Decision card for the communication treatment

İKİNCİ BÖLÜM				
Katılımcı numarası:				
		Bu kısımlar asistanımız tarafından doldurulacaktır.		
Gün	**KAÇ BALIK tuttum?**	**TOPLAM kaç balık tutuldu?**	**GÖLDE kaç balık kaldı?**	**NE KADAR KAZANDIM?**
1. gün				
2. gün				
3. gün				
4. gün				
5. gün				
6. gün				
7. gün				
8. gün				
9. gün				
10. gün				
11. gün				
12. gün				
13. gün				
14. gün				
15. gün				
16. gün				

Figure 50. Decision card for the baseline treatment

APPENDIX F

SUMMARY STATISTICS

Table 21. Summary Statistics for the Remaining Items in the Survey

Item	N	Mean	Standard deviation	Min	Max
How much does being a coop member facilitate the following activities for you?					
Use of fishing harbor	239	4.23	1.07	1	5
Obtaining the necessary documents for fishing	235	3.90	1.27	1	5
Being informed about new regulations	235	4.11	1.11	1	5
Sales of fish	235	3.32	1.50	1	5
Cooperation with other fishers	234	3.08	1.42	1	5
How often do you undertake the following activities?					
Attending the general assembly of the cooperative	231	4.40	1.03	1	5
Volunteering to take part of the steering committee of the cooperative	228	3.71	1.39	1	5
Informing the cooperative about a problem regarding fishing activities	230	4.37	1.03	1	5
Trying to find a solution to a cooperative member's problem regarding fishing	230	4.24	1.08	1	5
Trying to find a solution to a cooperative member's problem regarding other matters	233	3.54	1.39	1	5
Which statement reflects the matters regarding the income that fishers are receiving in your cooperative best?	More or less same	Little different	Very different	Missing	
	27.1%	47.5%	11.7%	13.8%	

APPENDIX G

QUESTIONNAIRE

	Kooperatif	
	Katılımcı numarası	
	Tarih	

SORU 1: Kooperatif ortağı olmanız aşağıdaki faaliyetlerinizi ne derece kolaylaştırıyor?

- Liman/balıkçı barınağı kullanımı:

Hiç kolaylaştırmıyor				Çok kolaylaştırıyor
1	2	3	4	5

- Balıkçılık için gereken belgelerin edinilmesi:

Hiç kolaylaştırmıyor				Çok kolaylaştırıyor
1	2	3	4	5

- Balıkçılıkla ilgili yeni yasalardan/kurallardan haberdar olma:

Hiç kolaylaştırmıyor				Çok kolaylaştırıyor
1	2	3	4	5

- Tutulan balığın satışı:

Hiç kolaylaştırmıyor				Çok kolaylaştırıyor
1	2	3	4	5

- Diğer balıkçılarla imece usulü iş yapma:

Hiç kolaylaştırmıyor				Çok kolaylaştırıyor
1	2	3	4	5

SORU 2: Kooperatif ortağı olarak aşağıdakileri ne sıklıkla yapıyorsunuz?

- Kooperatif genel kuruluna katılmak:

Hiçbir zaman				Her zaman
1	2	3	4	5

- Kooperatif yönetiminde görev almak için gönüllü olmak:

Hiçbir zaman				Her zaman
1	2	3	4	5

- Balıkçılıkla ilgili bir sorunu kooperatife bildirmek:

Hiçbir zaman				Her zaman
1	2	3	4	5

- Bir kooperatif ortağının balıkçılıkla ilgili bir sorununa çözüm bulmaya çalışmak:

Hiçbir zaman				Her zaman
1	2	3	4	5

- Bir kooperatif ortağının, balıkçılık dışındaki bir sorununa çözüm bulmaya çalışmak:

Hiçbir zaman				Her zaman
1	2	3	4	5

SORU 3: Ailenizde sizden önceki kuşakta balıkçılıkla uğraşanlar var mıydı?

1> Evet

2> Hayır

SORU 4: Hanenizde gelirinizin çoğunu aşağıdakilerden hangisi sayesinde elde ediyorsunuz?

1> Balıkçılık
2> Tarım
3> Başka bir serbest meslek
4> Emekli
5> Diğer

SORU 5: Kaç senedir balıkçılıkla uğraşıyorsunuz? ………………….. sene

SORU 6: Kaç yaşındasınız? …………………..

SORU 7: Kaç senedir kooperatif ortağısınız? ………………….. sene

SORU 8: Kullandığınız tekne kaç metre? …………………… metre

SORU 9: Kullandığınız teknenin motoru kaç beygir? ………………….. beygir

SORU 10: Kullandığınız tekne kime ait?

1> Tekne bana ait.
2> Tekne ortaklarından biriyim.

3> Tekne bir başkasına ait.

SORU 11: Kooperatifinizin ortaklarının balıkçılıktan elde ettikleri gelir ile ilgili aşağıdakilerden hangisi şu andaki durumu en iyi yansıtıyor?

1> Ortakların kazandıkları gelir aşağı yukarı aynı

2> Ortakların kazancı birbirinden biraz farklı

3> Ortakların elde ettiği gelir birbirinden aşırı farklı

SORU 12: En son bitirdiğiniz okul nedir?

1> İlkokul

2> Ortaokul/ilköğretim

3> Lise

4> Üniversite ve üstü

5> Hiçbiri

SORU 13: Aşağıda saydıklarımızın hangileri hanenizde bulunmaktadır?

Kredi Kartı	VAR	YOK
Bilgisayar	VAR	YOK
İnternet	VAR	YOK
Bulaşık makinası	VAR	YOK
Araba	VAR	YOK
Arsa	VAR	YOK

APPENDIX H

INDIVIDUAL EXTRACTION ACROSS TREATMENTS

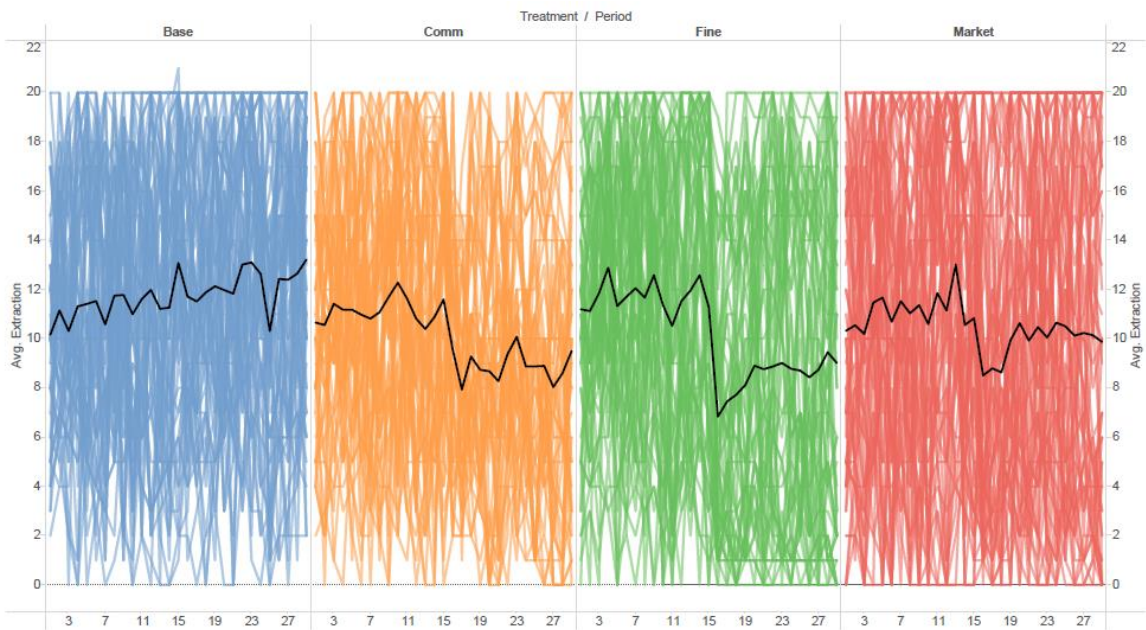

Figure 51. Individual extraction across treatments

Colored lines indicate extraction of different individuals for each round, and black line indicate averages for each treatment

REFERENCES

Acara, A., Mert, İ., Şenel, G., & Atik, F. (2001). *Su ürünleri ve su ürünleri sanayii özel ihtisas komisyonu raporu*. Retrieved from http://www.kooptr.com/ekitap/raporlar/su_urunleri_oik588.pdf

Adaman, F., & Arsel, M. (2005). *Environmentalism in Turkey: Between democracy and development?* Burlington: Ashgate Publishing Limited.

Adaman, F., & Özkaynak, B. (2002). The economics-environment relationship: Neoclassical, Institutional, and Marxist approaches. *Studies in Political Economy*, *69*, 109–135.

Agrawal, A. (2006). *Environmentality: Technologies of government and the making of subjects*. Durham: Duke University Press.

Ahlheim, M., & Buchholz, W. (2000). WTP or WTA – Is that the question? *Zeitschrift Für Umweltpolitik & Umweltrecht*, *23*, 253–271.

Ahn, T.-K., Ostrom, E., & Walker, J. (2010). A common-pool resource experiment with postgraduate subjects from 41 countries. *Ecological Economics*, *69*, 2624–2633.

Alpizar, F., & Gsottbauer, E. (2015). Reputation and household recycling practices: Field experiments in Costa Rica. *Ecological Economics*, *120*, 366–375.

Alptekin, E. (2013, November 27). Marmara'da orkinos katliamı. *Radikal*. Retrieved from http://www.radikal.com.tr/hayat/marmarada-orkinos-katliami-1163260/

Al-Ubaydli, O., Houser, D., Nye, J., Paganelli, M. P., & Pan, X. S. (2013). The causal effect of market priming on trust: An experimental investigation using randomized control. *PLoS ONE*, *8*(3), 1–8.

Anderies, J. M., Janssen, M. A., Bousquet, F., Cardenas, J.-C., Castillo, D., Lopez, M.-C., … Wutich, A. (2011). The challenge of understanding decisions in experimental studies of common pool resource governance. *Ecological Economics*, *70*(9), 1571–1579.

Anderson, C. M., Freeman, M. A., & Sutinen, J. G. (2008). A laboratory analysis of industry consolidation and diffusion under tradable fishing allowance management. In T. L. Cherry, S. Kroll, & J. F. Shogren (Eds.), *Environmental economics: Experimental methods*. Londra ve New York: Routledge.

Anderson, C. M., & Sutinen, J. G. (2006). The effect of initial lease periods on price discovery in laboratory tradable fishing allowance markets. *Journal of Economic Behavior & Organization*, *61*, 164–180.

Andreoni, J. (1990). Impure altruism and donations to public goods: A theory of warm-glow giving. *The Economic Journal*, *100*(401), 464–477.

Arnason, R. (2002). ITQs in practice: An international review. In A. Hatcher, S. Pascoe, R. Banks, & R. Arnason (Eds.), *Future options for fish quota management: A report to the Department for the Environment, Food and Rural Affairs* (pp. 54–70).

CEMARE Report, *58*.

Arpa, H. (2015). *Balıkçılık tarihimizden notlar*. Ankara: TMMOB Ziraat Mühendisleri Odası.

Arrow, K. J. (1950). A difficulty in the concept of social welfare. *The Journal of Political Economy*, *58*(4), 328–346.

Arrow, K. J. (1963). Uncertainty and the welfare economics of medical care. *The American Economic Review*, *53*(5), 941–973.

Aswani, S., Gurney, G. G., Mulville, S., Matera, J., & Gurven, M. (2013). Insights from experimental economics on local cooperation in a small-scale fishery management system. *Global Environmental Change*, *23*(6), 1402–1409.

Austin, J. L. (1962). *How to do things with words*. Oxford: Clarendon.

Axelrod, R. (1986). An evolutionary approach to norms. *American Political Science Review*, *80*(04), 1095–1111.

Baggio, J. A., Rollins, N. D., Pérez, I., & Janssen, M. A. (2015). Irrigation experiments in the lab: Trust, environmental variability, and collective action. *Ecology and Society*, *20*(4).

Balcı Akova, S. (2015). Aquaculture and its distribution in Turkey. *Journal of Aquaculture Engineering and Fisheries Research*, *1*(4), 160–190.

Ballard, C. L. (1988). The marginal efficiency cost of redistribution. *The American Economic Review*, *78*(5), 1019–1033.

Balliet, D. (2009). Communication and cooperation in social dilemmas: A meta-analytic review. *Journal of Conflict Resolution*, *20*(10), 1–19.

Bandura, A. (1991). Social cognitive theory of moral thought and action. In W. M. Kurtines & J. L. Gewirtz (Eds.), *Handbook of moral behavior and development* (pp. 45–103). Hillsdale: Lawrence Erlbaum Associates.

Barde, J.-P. (1994). *Economic instruments in environmental policy: Lessons from the OECD experience and their relevance to developing economies*. Development Centre of the Organisation for Economic Co-operation and Development.

Barde, J.-P. (1999). *Economic instruments for pollution control and natural resources management in OECD countries: A survey*. (No.72). OECD.

Barde, J.-P. (2000). Environmental policy and policy instruments. In *Principles of environmental and resource economics* (pp. 157–198). Cheltenham: Edward Elgar.

Barnett, A. H., & Yandle, B. (2009). The end of the externality revolution. *Social Philosophy and Policy*, *26*(02), 130.

Barr, A. (2001). *Social dilemmas and shame-based sanctions: Experimental results from rural Zimbabwe* (No. WPS/2001.11).

Barr, A., Packard, T., & Serra, D. (2014). Participatory accountability and collective action: Experimental evidence from Albania. *European Economic Review*, *68*, 250–269.

Barry, B. (1989). *Theories of justice: A treatise on social justice*. Berkeley: University of California Press.

Basurto, X., & Ostrom, E. (2009). Beyond the tragedy of the commons. *Economia Delle Fonti Di Energia E Dell'Ambiente*, *1*, 35–60.

Baticados, D. B., Agbayani, R. F., & Gentoral, F. E. (1998). Fishing cooperatives in Capiz, central Philippines: Their importance in managing fishery resources. *Fisheries Research*, *34*(2), 137–149.

Baumol, W. J., & Oates, W. E. (1979). *Economics, environmental policy, and the quality of life*. Englewood Cliffs, N.J: Prentice-Hall.

Baumol, W. J., & Oates, W. E. (1988). *The theory of environmental policy*. Cambridge: Cambridge University Press.

Beck, U. (1992). *Risk society: Towards a new modernity*. London: Sage.

Beck, U., Giddens, A., & Lash, S. (1994). *Reflexive modernization: Politics, tradition and aesthetics in the modern social order*. Cambridge: Polity.

Beder, S. (1996). Charging the earth: The promotion of price-based measures for pollution control. *Ecological Economics*, *16*, 51–63.

Ben-David, S., Brookshire, D. S., Burness, S., McKee, M., & Schmidt, C. (1999). Heterogeneity, irreversible production choices, and efficiency in emission permit markets. *Journal of Environmental Economics and Management*, *38*, 176–194.

Ben-David, S., Brookshire, D. S., Burness, S., Mckee, M., Schmidt, C., Ben-david, S., … Schmidt, C. (2000). Attitudes toward risk and compliance in emission permit markets. *Land Economics*, *76*(4), 590–600.

Bentham, J., & Mill, J. S. (1987). *Utilitarianism and other essays*. London: Penguin Books.

Berkes, F. (1986). Local-level management and the commons problem: A comparative study of Turkish coastal fisheries. *Marine Policy*, *10*(3), 215–229.

Betz, R., Rogge, K., & Schleich, J. (2006). EU emissions trading: An early analysis of National Allocation Plans for 2008-1012. *Climate Policy*, *6*(4), 361–394.

Biermann, F., Betsill, M. M., Gupta, J., Kanie, N., Lebel, L., Liverman, D., … Zondervan, R. (2010). Earth system governance: A research framework. *International Environmental Agreements*, *10*, 277–298.

Bilecik, N. (2003). *Herkes yerine: Denizler çölleşmeden, balıklar yok olmadan*. İzmir: Final Ofset.

Bishop, R. C., & Heberlein, T. A. (1979). Measuring values of extramarket goods: Are indirect measures biased? *American Journal of Agricultural Economics*, *61*(5), 926–930.

Blackburn, M., Harrison, G. W., & Ruström, E. (1994). Statistical bias functions and informative hypothetical surveys. *American Journal of Agricultural Economics*, *76*(5), 1084–1088.

Blumenschein, K., Blomquist, G. C., Johannesson, M., Horn, N., & Freeman, P. (2008). Eliciting willingness to pay without bias: Evidence from a field experiment. *The Economic Journal*, *118*, 114–137.

Blumenschein, K., Johannesson, M., Blomquist, G. C., Liljas, B., & O'Conor, R. M. (1998). Experimental results on expressed certainty and hypothetical bias in contingent valuation. *Southern Economic Journal*, *65*(1), 169–177.

Bohm, P. (1972). Estimating demand for public goods: An experiment. *European Economic Review*, *3*(2), 111–130.

Bohm, P. (1997). *A joint implementation as emission quota trade: An experiment among four Nordic countries*. Copenhagen: Nordic Council of Ministers.

Bohm, P. (2003). Experimental evaluations of policy instruments. In K. G. Maler & J. Vincent (Eds.), *Handbook of experimental economics, Vol. 1 of handbooks in economics*. Amsterdam: Elsevier.

Bohm, P., & Carlén, B. (1999). Emission quota trade among the few: Laboratory evidence of joint implementation among committed countries. *Resource and Energy Economics*, *21*(1), 43–66.

Bohnet, I., Greig, F., Herrmann, B., & Zeckhauser, R. (2008). Betrayal aversion: Evidence from Brazil, China, Oman, Switzerland, Turkey, and the United States. *The American Economic Review*, *98*(1), 294–310.

Borges, G., & Irlenbusch, B. (2007). Fairness crowded out by law: An Experimental study on withdrawal rights. *Journal of Institutional and Theoretical Economics*, *163*(1), 84–101.

Bouma, J., & Ansink, E. (2013). The role of legitimacy perceptions in self-restricted resource use: A framed field experiment. *Forest Policy and Economics*, *37*, 84–93.

Bowles, S. (1998). Endogenous preferences: The cultural consequences of markets and other economic institutions. *Journal of Economic Literature*, *36*(1), 75–111.

Bowles, S. (2004). *Microeconomics: Behavior, institutions, and evolution*. Princeton: Princeton University Press.

Bowles, S. (2008). Policies designed for self-interested citizens may undermine "the moral sentiments." *Science*, *320*, 1605–1609.

Bowles, S., Boyd, R., Fehr, E., & Gintis, H. (1997). Homo reciprocans: A research initiative on the origins, dimensions, and policy implications of reciprocal fairness. *Advances in Complex Systems*, *4*(2/3), 1-30.

Bowles, S., & Gintis, H. (2011). *A cooperative species: Human reciprocity and its evolution*. Princeton: Princeton University Press.

Bowles, S., & Polanía-Reyes, S. (2012). Economic incentives and social preferences: Substitutes or complements? *Journal of Economic Literature*, *50*(2), 368–425.

Braaten, R. H. (2014). Land rights and community cooperation: Public goods experiments from Peru. *World Development*, *61*, 127–141.

Bromley, D. W. (1990). The idelology of efficiency: Searching for a theory of policy analysis. *Journal of Environmental Economics and Management*, *19*, 86–107.

Brookshire, D. S., & Coursey, D. L. (1987). Measuring the value of a public good: An empirical comparison of elicitation procedures. *The American Economic Review*, *77*(4), 554–566.

Brown, C. V., & Jackson, P. M. (1986). *Public sector economics*. Oxford: Wiley-Blackwell.

Brown-Kruse, J., Elliott, S. R., & Godby, R. (1995). Strategic manipulation of pollution permit markets: An experimental approach. *Department of Economics, McMaster University Working Paper*, 95-10.

Buchanan, J. M. (1969). External diseconomies, corrective taxes, and market structure. *The American Economic Review*, *59*(1), 174–177.

Burkett, P. (2006). *Marxism and ecological economics: Toward a red and green political economy*. Boston: Brill.

Burns, P. (1985). *Experience and decision-making: A comparison of students and businessmen in a simulated progressive auction* (No. 00135). The Field Experiments Website.

Butler, J. (1990). *Gender trouble: Feminism and the subversion of identity*. New York: Routledge, Chapman & Hall.

Callon, M. (1998). *The laws of the markets*. Oxford: Blackwell Publishing Ltd.

Callon, M. (2007). What does it mean to say that economics is performative? In D. MacKenzie, F. Fabian Muniesa, & L. Siu (Eds.), *Do economists make markets? On the performativity of economics* (pp. 311–357). Princeton: Princeton University Press.

Cardenas, J.-C. (2000). How do groups solve local commons dilemmas? Lessons from experimental economics in the field. *Environment, Development and Sustainability*, *2*, 305–322.

Cardenas, J.-C. (2003). Real wealth and experimental cooperation: Experiments in the field lab. *Journal of Development Economics*, *70*, 263–289.

Cardenas, J.-C. (2004). Norms from outside and from inside: An experimental analysis on the governance of local ecosystems. *Forest Policy and Economics*, *6*, 229–241.

Cardenas, J.-C. (2005). Groups, commons and regulations: Experiments with villagers and students in Colombia. In *IEA Conference Volume Series* (Vol. 142). London: Palgrave Macmillan UK.

Cardenas, J.-C. (2011). Social norms and behavior in the local commons as seen through the lens of field experiments. *Environmental and Resource Economics*, *48*, 451–485.

Cardenas, J.-C., Janssen, M. A., & Bousquet, F. (2013). Dynamics of rules and resources: Three new field experiments on water, forests and fisheries. In J. A. List & M. Price (Eds.), *Handbook on experimental economics and the environment* (pp. 319–345). Cheltenham: Edward Elgar.

Cardenas, J.-C., & Ostrom, E. (2004). What do people bring into the game? Experiments in the field about cooperation in the commons. *Agricultural Systems*, *82*, 307–326.

Cardenas, J.-C., Stranlund, J. K., & Willis, C. (2000). Local environmental control and institutional crowding-out. *World Development*, *28*(10), 1719–1733.

Carpenter, J., Bowles, S., Gintis, H., & Hwang, S. H. (2009). Strong reciprocity and team production: Theory and evidence. *Journal of Economic Behavior and Organization*, *71*, 221–232.

Carpenter, J., & Seki, E. (2005). *Competitive work environments and social preferences: Field experimental evidence from a Japanese fishing community*. *IZA Discussion Paper No. 1691*.

Carpenter, J., & Seki, E. (2011). Do social preferences increase productivity? Field experimental evidence from fishermen in Toyama Bay. *Economic Inquiry*, *49*(2), 612–630.

Carson, R. T., Mitchell, R. C., Hanemann, M. W., Kopp, R. J., Presser, S., & Ruud, P. A. (2003). Contingent valuation and lost passive use: Damages from the Exxon Valdez oil spill. *Environmental and Resource Economics*, *25*, 257–286.

Cason, T. N., & Gangadharan, L. (2006). Emissions variability in tradable permit markets with imperfect enforcement and banking. *Journal of Economic Behavior and Organization*, *61*(2), 199–216.

Cason, T. N., & Gangadharan, L. (2013). Cooperation spillovers and price competition in experimental markets. *Economic Inquiry*, *51*(3), 1715–1730.

Cason, T. N., & Raymond, L. (2011). Framing effects in an emissions trading experiment with voluntary compliance. In R. M. Isaac & D. A. Norton (Eds.), *Research in experimental economics* (Vol. 14). Bingley: Emerald Group Publishing Limited.

Castillo, D., Bousquet, F., Janssen, M. A., Worrapimphong, K., & Cardenas, J.-C. (2011). Context matters to explain field experiments: Results from Colombian and Thai fishing villages. *Ecological Economics*, *70*, 1609–1620.

Castillo, D., & Saysel, A. K. (2005). Simulation of common pool resource field experiments: A behavioral model of collective action. *Ecological Economics*, *55*, 420–436.

Champ, P. A., Bishop, R. C., Brown, T. C., & McCollum, D. W. (1997). Using donation mechanisms to value non-use benefits from public goods. *Journal of Environmental Economics and Management*, *33*(2), 151–162.

Chavas, J.-P. (1994). Equity considerations in economic and policy analysis. *American Journal of Agricultural Economics*, *76*, 1022–1033.

Cherry, T. L., Kroll, S., & Shogren, J. F. (2008). *Environmental economics: Experimental methods*. New York: Routledge.

Chichilnisky, G. (1997). The costs and benefits of benefit-cost analysis. *Environment and Development Economics*, *2*(2), 195–221.

Child, M. F. (2009). The Thoreau ideal as a unifying thread in the conservation movement. *Conservation Biology*, *23*(2), 241–243.

Coase, R. H. (1960). The problem of social cost. *Journal of Law & Economics*, *3*(1).

Cooter, R. (1984). Prices and sanctions. *Columbia Law Review*, *84*(6), 1523–1560.

Copes, P. (1996). *Adverse impacts of individual quota systems on conservation and fish harvest productivity* (No. 96-1). Burnaby.

Copes, P., & Pálsson, G. (2000). Challenging ITQs: Legal and political action in Iceland, Canada and Latin America. In *IIFET Proceedings* (pp. 1–6).

Corbera, E., Soberanis, C. G., & Brown, K. (2009). Institutional dimensions of Payments for Ecosystem Services: An analysis of Mexico's carbon forestry programme. *Ecological Economics*, *68*, 743–761.

Coursey, D., Hovis, J., & Schulze, W. (1987). The disparity between willingness to accept and willingness to pay measures of value. *The Quarterly Journal of Economics*, *102*(3), 679–690.

Crocker, T. D. (1966). The structuring of atmospheric pollution control systems. In H. Wolozin (Ed.), *The economics of air pollution* (pp. 61–86). New York: Norton.

Cronshaw, M. B., & Brown-Kruse, J. (1999). An experimental analysis of emission permits with banking and the Clean Air Act Amendments of 1990. In C. Holt & R. . Isaac (Eds.), *Research in experimental economics vol. 7*. Stanford: JAI Press.

Cropper, M. L., & Oates, W. E. (1992). Environmental economics: A survey. *Journal of Economic Literature*, *30*(2), 675–740.

Cummings, R. G., Elliott, S., Harrison, G. W., & Murphy, J. (1997). Are hypothetical referenda incentive compatible? *Journal of Political Economy*, *105*(3), 609.

Cummings, R. G., Harrison, G. W., & Osborne, L. L. (1995). *Can the bias of contingent valuation surveys be reduced?* (No. B-95-03).

Cummings, R. G., Holt, C. A., & Laury, S. K. (2004). Using laboratory experiments for policy-making: An example from the Georgia irrigation reduction auction. *Journal of Policy Analysis and Management*, *23*(2), 341–363.

Çakıroğlu, S. B. (1969). *Karadeniz'de balıkçılığımız*. Ankara: Bilgi Basımevi.

Dales, J. H. (1968). *Pollution, property and prices*. Toronto: Toronto University Press.

Davidse, W. P. (2001). The effects of transferable property rights on the fleet capacity and ownership of harvesting rights in the Dutch demersal North Sea fisheries. *FAO Fisheries Technical Paper*, (2), 258-266.

Davies, C. J., & Mazurek, J. (1998). *Pollution control in the United States: Evaluating the system*. New York: Resources for the Future.

Dawes, R. M. (1988). *Rational choice in an uncertain world*. San Diego and New York: Harcourt, Brace, Yovanovich.

De Young, R. (1985). Encouraging environmentally appropriate behavior: The role of intrinsic motivation. *Journal of Environmental Systems*, *15*(4), 281–292.

Deci, E. (1971). Effects of externally mediated rewards on intrinsic motivation. *Journal of Personality and Social Psychology*, *18*(1), 105–115.

Deci, E. L., Ryan, R. M., & Koestner, R. (1999). A meta-analytic eeview of experiments examining the effects of extrinsic rewards on intrinsic motivation. *Psychological Bulletin*, *125*, 627–668.

del Pilar Moreno-Sanchez, R., & Maldonado, J. H. (2010). Evaluating the role of co-management in improving governance of marine protected areas: An experimental approach in the Colombian Caribbean. *Ecological Economics*, *69*, 2557–2567.

Denizhaber (2012, February 1). İstanbullu balıkçılar Valilik kapısını çaldı. Retrieved from http://www.denizhaber.com.tr/istanbullu-balikcilar-valilik-kapisini-caldi-haber-39944.htm

Deutsch, M. (1958). Trust and suspicion. *The Journal of Conflict Resolution*, *2*(4), 265–279.

Deveciyan, K. (2013). *Türkiye'de balık ve balıkçılık*. Ankara: Aras Yayıncılık.

Dickie, M., Fisher, A., & Gerking, S. (1987). Market transactions and hypothetical demand data: A comparative study. *Journal of the American Statistical Association*, *82*(397), 69–75.

Dietz, T., Ostrom, E., & Stern, P. C. (2003). The struggle to govern the commons. *Science*, *302*(5652), 1907–1912.

Doğan, K. (1982). Balık unu ve yağı ile ilgili sorunlar. In *Su Ürünleri Üretimi Artırma ve Kredi Yönlendirme Sempozyumu* (pp. 278–298). Ankara: Türkiye Cumhuriyeti Ziraat Bankası.

Dufwenberg, M., Gächter, S., & Hennig-Schmidt, H. (2011). The framing of games and the psychology of play. *Games and Economic Behavior*, *73*, 459–478.

Eckel, C. C., Grossman, P. J., & Johnston, R. M. (2005). An experimental test of the crowding-out hypothesis. *Journal of Public Economics*, *89*, 1543–1560.

Eckel, C., & Grossman, P. (1996). Altruism in anonymous dictator games. *Games and Economic Behaviorconomic Behavior*, *16*, 181–191.

Ellingsen, T., Johannesson, M., Mollerstrom, J., & Munkhammar, S. (2012). Social framing effects: Preferences or beliefs? *Games and Economic Behavior*, *76*, 117–130.

Elliott, C. S., Hayward, D. M., & Canon, S. (1998). Institutional framing: Some experimental evidence. *Journal of Economic Behavior & Organization*, *35*, 455–464.

Engel, S., Pagiola, S., & Wunder, S. (2008). Designing payments for environmental services in theory and practice: An overview of the issues. *Ecological Economics*, *65*, 663–674.

Environmental Protection Agency (EPA). (2015). *The European environment: State and outlook 2015*. Office for Official Publ. of the European Communities.

Eurofish. (2016). Overview of the Turkish fisheries and aquaculture sector. Retrieved from http://www.eurofish.dk/index.php?option=com_content&view=article&id=123%3Aturkey&catid=37&Itemid=53

European Commission. (2015). *Commission Staff Working Document: Turkey 2015 Report*. Retrieved June 28, 2016, from http://ec.europa.eu/enlargement/pdf/key_documents/2015/20151110_report_turkey.pdf

European Commission. (2016). *Small-scale coastal fleet in the EU*. Retrieved June 28, 2016, from http://ec.europa.eu/fisheries/documentation/publications/2016-small-scale-coastal-fleet_en.pdf

Falk, A., Fehr, E., & Fischbacher, U. (2000). *Appropriating the commons: A theoretical explanation* (No. 55).

Falk, A., & Kosfeld, M. (2005). The hidden costs of control. *The American Economic Review*, 96(5), 1611-1630.

Falk, A., & Szech, N. (2013). Morals and markets. *Science*, *340*(6133), 707–711.

Farrell, J. (1987). Information and the Coase Theorem. *The Journal of Economic Perspectives*, *1*(2), 113–129.

Feeny, D., Berkes, F., Mccay, B. J., & Acheson, J. M. (1990). The tragedy of the commons: Twenty-two years later. *Human Ecology*, *18*(1), 1–19.

Fehr, E., & Falk, A. (2002). Psychological foundations of incentives. *European Economic Review*, *46*, 687–724.

Fehr, E., & Gächter, S. (2000). Cooperation and punishment in public goods experiments. *The American Economic Review*, *90*(4), 980–994.

Fehr, E., Klein, A., & Schmidt, K. M. (2007). Fairness and contract design. *Econometrica*, *75*(1), 121–154.

Fehr, E., & Leibbrandt, A. (2011). A field study on cooperativeness and impatience in the tragedy of the commons. *Journal of Public Economics*, *95*, 1144–1155.

Fehr, E., & List, J. A. (2004). The hidden costs and returns of incentives: Trust and trustworthiness among CEOs. *Journal of the European Economic Association*, *2*(5), 743–771.

Fehr, E., & Rockenbach, B. (2003). Detrimental effects of sanctions on human altruism. *Nature*, *422*, 137–140.

Fehr, E., & Schmidt, K. M. (2003). Theories of fairness and reciprocity: Evidence and economic applications. In *Advances in economics and econometrics: Theory and applications* (pp. 208–257).

Fischbacher, U., Gächter, S., & Fehr, E. (2001). Are people conditionally cooperative?: Evidence from a public goods Eexperiment. *Economics Letters*, *71*, 397–404.

Fiske, A. P. (1992). The four elementary forms of sociality: Framework for a unified theory of social relations. *Psychological Review*, *99*(4), 689–723.

Food and Agriculture Organization (FAO). (2005). National aquaculture sector review: Turkey. Retrieved from http://www.fao.org/fishery/countrysector/naso_turkey/en

Food and Agriculture Organization (FAO). (2016a). *A sub-regional analysis of the socio-economic situation of Eastern Mediterranean fisheries*. Retrieved July 14, 2016, from http://www.fao.org/documents/card/en/c/cd49499d-cdbf-48ea-b436-7f72bda96b4a/

Food and Agriculture Organization (FAO). (2016b). Fisheries management and safety. Retrieved May 23, 2016, from http://www.fao.org/docrep/003/x9656e/X9656E03.htm

Food and Agriculture Organization (FAO). (2016c). *Technical and socio-economic characteristics of small-scale coastal fishing communities, and opportunities for poverty alleviation and empowerment*. Retrieved June 15, 2016, from http://www.fao.org/3/a-i5651e.pdf

Food and Agriculture Organization (FAO). (2016d). *The state of world fisheries and aquaculture 2016: Contributing to food security and nutrition for all*. Rome: Food and Agriculture Organization of United Nations.

Frey, B. S. (1992). Pricing and regulating affect environmental ethics. *Environmental & Resource Economics*, *2*, 399–414.

Frey, B. S. (1993). Motivation as a limit to pricing. *Journal of Economic Psychology*, *14*(4), 635–664.

Frey, B. S. (1997). A constitution for knaves crowds out civic virtues. *The Economic Journal*, *107*, 1043–1053.

Frey, B. S., & Oberhozer-Gee, F. (1997). The cost of price incentives: An empirical analysis of motivation crowding-out. *American Economic Review, 87*(5), 746–755.

Frey, B. S., & Stutzer, A. (2008). Environmental morale and motivation. In A. Lewis (Ed.), *The Cambridge handbook of psychology and economic behaviour* (pp. 406–428). Cambridge: Cambridge University Press.

Friesen, L., & Gangadharan, L. (2013). Environmental markets: What do we learn from the lab? *Journal of Economic Surveys*, *27*(3), 515–535.

Gächter, S., & Herrmann, B. (2011). The limits of self-governance when cooperators get punished: Experimental evidence from urban and rural Russia. *European Economic Review*, *55*, 193–210.

Gatiso, T. T., Vollan, B., & Nuppenau, E. A. (2015). Resource scarcity and democratic elections in commons dilemmas: An experiment on forest use in Ethiopia. *Ecological Economics*, *114*, 199–207.

Gawel, E. (2001). Intrinsische Motivation und umweltpolitische Instrumente. *Perspektiven Der Wirtschaftspolitik*, *2*(2), 145–165.

Gelcich, S., Guzman, R., Rodrigues-Sickert, C., Castilla, J. C., & Cardenas, J.-C. (2013). Exploring external validity of common pool resource experiments: Insights from artisanal Benthic fisheries in Chile. *Ecology and Society*, *18*(3).

Ghate, R., Ghate, S., & Ostrom, E. (2013). Cultural norms, cooperation, and communication: Taking experiments to the field in indigenous communities. *International Journal of the Commons*, *7*(2), 498–520.

Giannetto, D., Acar, U., & Demir, O. (2014). Current status and future prospective for aquaculture sector in Turkey. *Austin Journal of Aquaculture and Marine Biology*, *1*(1), 1–3.

Gintis, H. (2000). Beyond homo economicus: Evidence from experimental economics. *Ecological Economics*, *35*, 311–322.

Gneezy, U., Meier, S., & Rey-Biel, P. (2011). When and why incentives (don't) work to modify behavior. *Journal of Economic Perspectives*, *25*(4), 191–210.

Gneezy, U., & Rustichini, A. (2000). A fine is a price. *Journal of Legal Studies*, *29*, 1–17.

Godby, R., Mestelman, S., Muller, R. A., Welland, J. D., & Muller, R. A. (1997). Emissions trading with shares and coupons when control over discharges is uncertain. *Journal of Environmental Economics and Management*, *32*, 359–381.

Gomez-Baggethun, E., de Groot, R., Lomas, P. ., & Montes, C. (2010). The history of ecosystem services in economic theory and practice. *Ecological Economics*, *69*, 1209–1218.

Gomez-Baggethun, E., & Ruiz-Perez, M. (2011). Economic valuation and the commodification of ecosystem services. *Progress in Physical Geography*, *35*(5), 613–628.

Goodin, R. E. (1994). Selling environmental indulgences. *Kyklos*, *47*, 573–596.

Gordon, S. H. (1954). The economic theory of a common-property resource: The fishery. In *Classic papers in natural resource economics* (pp. 178–203). London: Palgrave Macmillan UK.

Goulder, L. H., & Parry, I. W. H. (2008). Instrument choice in environmental policy. *Review of Environmental Economics and Policy*, *2*(2), 152–174.

Göncüoğlu, H., & Ünal, V. (2011). Fisherwomen in the Turkish fishery, southern Aegean Sea. *Journal of Applied Ichthyology*, *27*, 1013–1018.

Graaaf, J. (1957). *Theoretical welfare economics*. Cambridge: Cambridge University Press.

Graham, D. A. (1981). Cost-benefit analysis under uncertainty. *The American Economic Review*, *71*(4), 715–725.

Griffin, R. C. (1995). On the meaning of economic efficiency in policy analysis. *Land Economics*, *71*(1), 1–15.

Gupta, S., Tirpak, D. a., Burger, N., Gupta, J., Höhne, N., Boncheva, A. I., … Sari, A. (2007). Chapter 13: Policies, instruments, and co-operative arrangements - AR4 WGIII. Retrieved March 15, 2015, from http://www.ipcc.ch/publications_and_data/ar4/wg3/en/ch13.html

Habermas, J. (1984). *The theory of communicative action*. Boston: Beacon Press.

Habertürk (2015, August 30). İstanbul'da balık e-mezat ile satılacak. Retrieved from http://www.haberturk.com/ekonomi/ekonomi/haber/1122176-istanbulda-balik-e-mezat-ile-satilacak

Hahn, R. W. (1984). Market power and transferable property. *The Quarterly Journal of Economics*, *99*(4), 753–765.

Hahn, R. W. (1989). Economic prescripions for environmental problems: How the patient followed the doctor’s orders. *Journal of Economic Perspectives*, *3*(2), 95–114.

Hahn, R. W., & Noll, R. G. (1982). Designing a market for tradable emission permits. In W. Magat (Ed.), *Reform of environmental regulation*. Lexington: Lexington Books.

Hahn, R. W., & Stavins Source, R. N. (1992). Economic incentives for environmental protection: Integrating theory and practice. *The American Economic Review*, *82*(2), 464–468.

Hahnel, R. (2015). *Green economics: Confronting the ecological crisis*. New York: Routledge.

Hahnel, R., & Sheeran, K. A. (2009). Misinterpreting the Coase Theorem. *Journal of Economic Issues*, *43*(1), 215–238.

Hammack, J., & Brown, G. M. (1974). *Water-fowl and wetlands: Toward bioeconomic analysis*. Baltimore: The Johns Hopkins University Press for Resources for the Future.

Handberg, Ø. N., & Angelsen, A. (2015). Experimental tests of tropical forest conservation measures. *Journal of Economic Behavior & Organization*, *118*, 346–359.

Hanemann, M. W. (1984). Welfare evaluations in contingent valuation experiments with discrete responses. *American Journal of Agricultural Economics*, *66*(3), 332–341.

Hanemann, M. W. (1991). Willingness to pay and willingess To accept: How much can they difffer? *The American Economic Review*, *81*(3), 635–647.

Hanemann, M. W. (1994). Valuing the environment through contingent valuation. *Journal of Economic Perspectives*, *8*(4), 19–43.

Hardin, G. (1968). The tragedy of the commons. *Science*, *162*(3859), 1243–1248.

Harrison, G. W., & List, J. A. (2004). Field experiments. *Journal of Economic Literature*, *42*(4), 1009–1055.

Harvey, D. (2007). *A brief history of neoliberalism*. New York: Oxford University Press.

Hayo, B., & Vollan, B. (2012). Group interaction, heterogeneity, rules, and co-operative behaviour: Evidence from a common-pool resource experiment in South Africa and Namibia. *Journal of Economic Behavior and Organization*, *81*, 9–28.

Henrich, J., Boyd, R., Bowles, S., Camerer, C. F., Fehr, E., Mcelreath, R., … Sciences,

B. (2005). “Economic man” in cross-cultural perspective: Behavioral experiments in 15 small-scale societies. *Behavioral and Brain Sciences*, *28*, 795–855.

Hepburn, C. (2007). Carbon trading: A review of the Kyoto mechanisms. *Annual Review of Environment and Resources*, *32*, 375–393.

Heres, D. R., & Lin, C. C. (2012). *Quantity and price controls for the correction of externalities under uncertain damages: Evidence from a laboratory experiment*.

Heyman, J., & Ariely, D. (2004). Effort for payment: A tale of two markets. *Psychological Science*, *15*(11), 787–793.

Hodgson, G. M. (1998). The approach of institutional economics. *Journal of Economic Literature*, *36*(1), 166–192.

Hoffman, E., & Spitzer, M. L. (1982). Experimental tests of the Coase Theorem. *Journal of Law and Economics*, *25*(1), 73–98.

Hoffman, E., & Spritzer, M. L. (1985). Entitlement, rights and fairness: an experimental examination of subjects concept of distributive justice. *Journal of Legal Studies*, *14*, 259–297.

Holm, P. (2007). Which way is up on Callon? In D. A. MacKenzie, F. Muniesa, & L. Siu (Eds.), *Do economists make markets? On the performativity of economics* (pp. 225–243). Princeton: Princeton University Press.

Hopper, J. R., & Nielsen, J. M. (1991). Recycling as altruistic behavior: Normative and behavioral strategies to expand participation in a community recycling program. *Environment and Behavior*, *23*(2), 195–220.

Horowitz, J. K., & McConnell, K. E. (2002). A Review of WTA/WTP studies. *Journal of Environmental Economics and Management*, *44*(3), 426–447.

Horowitz, J. K., McConnell, K. E., & Murphy, J. J. (2013). Behavioural foundations of environmental economics and valuation. In J. A. List & M. K. Price (Eds.), *Handbook on experimental economics and the environment* (pp. 115–156).

Howarth, R. B., & Norgaard, R. B. (1991). Sustainability and discounting the future. In R. Costanza (Ed.), *Ecological economics: The science and management of sustainability* (pp.88-101).

Hussen, A. M. (2004). *Principles of environmental economics* (2bd ed.). New York: Routledge.

Jack, B. K. (2009). Upstream-downstream transactions and watershed externalities: Experimental evidence from Kenya. *Ecological Economics*, *68*, 1813–1824.

Janssen, M. A. (2010). Introducing ecological dynamics into common-pool resource experiments. *Ecology and Society*, *15*(2).

Janssen, M. A., Anderies, J. M., & Cardenas, J.-C. (2011). Head-enders as stationary bandits in asymmetric commons: Comparing irrigation experiments in the laboratory and the field. *Ecological Economics*, *70*, 1590–1598.

Javaid, A., & Falk, T. (2015). Incorporating local institutions in irrigation experiments: Evidence from rural communities in Pakistan. *Ecology and Society*, *20*(2).

Kahneman, D., Knetsch, J. L., & Thaler, R. (1990). Experimental test sof the endowment effect and the Coase Theorem. *Journal of Political Economy*, *98*, 1325–1348.

Kahneman, D., Slovic, P., & Tversky, A. (1982). *Judgement under uncertainty: Heuristics and biases*. Cambridge: Cambridge University Press.

Kallbekken, S., Kroll, S., & Cherry, T. L. (2011). Do you not like Pigou, or do you not understand him?: Tax aversion and revenue recycling in the lab. *Journal of Environmental Economics and Management*, *62*, 53–64.

Kapp, K. W. (1950). *The social costs of private enterprise*. New York: Schocken Books.

Kapp, K. W. (1983). *Social costs, economic development, and environmental disruption*. Lanham: University Press of America.

Kay, A. C., & Ross, L. (2003). The perceptual push: The interplay of implicit cues and explicit situational construals on behavioral intentions in the prisoner's dilemma. *Journal of Experimental Social Psychology*, *39*, 634–643.

Kerr, J., Vardhan, M., & Jindal, R. (2012). Prosocial behavior and incentives: Evidence from field experiments in rural Mexico and Tanzania. *Ecological Economics*, *73*, 220–227.

Kimbrough, E. O. (2013). The social and ecological determinants of common pool resource sustainability. *Journal of Environmental Economics and*, *72*, 38–53.

Knetsch, J. L. (1989). The endowment effect and evidence of nonreversible indifference curves. *American Economic Review*, *79*(5), 1277–1284.

Knetsch, J. L. (2013). Values of gains and losses: reference states and choice of measure. In J. A. List & M. K. Price (Eds.), *Handbook on experimental economics and the environment*. Cheltenham: Edward Elgar.

Knetsch, J. L., & Sinden, J. A. (1984). Willingness to pay and compensation demanded: Experimental rvidence of an unexpected disparity in measures of value. *The Quarterly Journal of Economics*, *99*(3), 507–521.

Knetsch, J. L., & Wong, W.-K. (2009). The endowment effect and the reference state: Evidence and manipulations. *Journal of Economic Behavior & Organization*, *71*, 407–413.

Knudsen, S. (2009). *Fishers and scientists in modern Turkey: The management of natural resources, knowledge and identity on the eastern Black Sea coast*. New York: Berghahn Books.

Kolstad, C. D. (2009). *Environmental economics*. New York: Oxford University Press.

Kosoy, N., & Corbera, E. (2010). Payments for ecosystem services as commodity fetishism. *Ecological Economics*, *69*, 1228–1236.

Köszegi, B., & Rabin, M. (2006). A model of reference-dependent preferences. *The Quarterly Journal of Economics*, *CXXI*(4), 1133–1165.

Kral, M. J., Ramierez García, J. I., Aber, M. S., Masood, N., Dutta, U., & Todd, N. R.

(2011). Culture and community psychology: Toward a renewed and reimagined vision. *American Journal of Community Psychology*, *47*, 46–57.

Kreps, D. M., & Wilson, R. (1982). Reputation and imperfect information. *Journal of Economic Theory*, *27*(2), 253–279.

Kriebel, D., Tickner, J., Epstein, P., Lemons, J., Levins, R., Loechler, E. L., … Stoto, M. (2001). The precautionary principle in environmental science. *Environmental Health Perspectives*, *109*(9), 871–876.

Kwerel, E. (1977). To tell the truth: Imperfect information and optimal pollution control. *The Review of Economic Studies*, *44*(3), 595.

Landry, C. E., & List, J. A. (2007). Using ex-ante approaches to obtain credible signals for value in contingent markets: Evidence from the field. *American Journal of Agricultural Economics*, *89*(2), 420–429.

Lehmann, P. (2012). Justifying a policy mix for pollution control: A review of economic literature. *Journal of Economic Surveys*, *26*(1), 71–97.

Liberman, V., Samuels, S. M., & Ross, L. (2004). The name of the game: Predictive power of reputations versus situational labels in determining prisoner's dilemma game moves. *Personality and Social Psychology Bulletin*, *30*(9), 1175–1185.

List, J. A. (2003). Does market experience eliminate market anomolies? *Quarterly Journal of Economics*, 41–71.

List, J. A. (2004). Neoclassical theory versus prospect theory: Evidence from the marketplace. *Econometrica*, *72*(2), 615–625.

Liverman, D. (2004). Who governs, at what scale and at what price? Geography, environmental governance, and the commodification of nature. *Annals of the Association of American Geographers*, *94*(4), 734–738.

Lohmann, L. (2010). Uncertainty markets and carbon markets: Variations on Polanyian themes. *New Political Economy*, *15*(2), 225–254.

Longo, S. B., & Clark, B. (2012). The commodification of bluefin tuna: The historical transformation of the Mediterranean fishery. *Journal of Agrarian Change*, *12*(2-3), 204–226.

Loomis, J., Brown, T. C., Lucero, B., & Peterson, G. (1996). Improving validity experiments of contingent valuation methods: Results of efforts to reduce the disparity of hypothetical and actual willingness to pay. *Land Economics*, *72*(4), 450–461.

Loomis, J., Gonzalez-Caban, A., & Gregory, R. (1994). Do reminders of substitutes and budget constraints influence contingent valuation estimates? *Land Economics*, *70*(4), 499–506.

Lopez, M.-C., Murphy, J. J., Spraggon, J. M., & Stranlund, J. K. (2012). Comparing the effectiveness of regulation and pro-social emotions to enhance cooperation: Experimental evidence from fishing communities in Colombia. *Economic Inquiry*, *50*(1), 131–142.

Luck, G. W., Chan, K. M. A., Eser, U., Gómez-Baggethun, E., Matzdorf, B., Norton, B., & Potschin, M. B. (2012). Ethical considerations in on-ground applications of the ecosystem services concept. *BioScience*, *62*(12), 1020–1029.

Luckert, M. K., & Adamowicz, W. L. (1993). Empirical measures of factors affecting social rates of discount. *Environmental and Resource Economics*, *3*, 1–21.

Lunn, P. (2013). *Behavioural economics and regulatory policy*. Paris: OECD Publishing.

MacKenzie, D. A., Muniesa, F., & Siu, L. (2007). *Do economists make markets? On the performativity of economics*. Princeton University Press.

Madra, Y., & Adaman, F. (2010). Public economics after neoliberalism: A theoretical–historical perspective. *The European Journal of the History of Economic Thought*, *17*(4), 377–406.

Maldonado, J. H., & del Pilar Moreno-Sanchez, R. (2009). *Does scarcity exacerbate the tragedy of the commons? Evidence from fishers' experimental responses*. *Documentos CEDE* (Vol. 22). Bogota.

Malmqvist, T., & Whan, E. (2016). *Greendex 2014: Consumer choice and the Environment - A Worldwide Tracking Survey*. Toronto. Retrieved June 24, 2016, from http://images.nationalgeographic.com/wpf/media-content/file/2014_Global_Report-cb1411689801.pdf

Mantilla, C. (2015a). Communication networks in common-pool resource games: Field experimental evidence. *Journal of Economic Behavior and Organization*, *118*, 215–226.

Mantilla, C. (2015b). To suggest is to commit? A common pool resource experiment with non-enforceable recommendations. *Journal of Behavioral and Experimental Economics*, *59*, 13–20.

Martinez-Alier, J. (1987). *Ecological economics*. Oxford: Blackwell Publishing Ltd.

Martinez-Alier, J., Munda, G., & O'Neill, J. (1998). Weak comparability of values as a foundation for ecological economics. *Ecological Economics*, *26*, 277–286.

Masclet, D., Noussair, C., Tucker, S., & Villeval, M.-C. (2003). Monetary and non-monetary punishment in the voluntary contributions mechanism. *The American Economic Review*, *93*(1), 366–380.

McCauley, D. J. (2006). Selling out on nature. *Nature*, *443*, 27–28.

Mestelman, S., & Muller, A. R. (2006). Share trading and coupon banking interact to improve performance in emission trading markets. In C. R. Plott & V. Smith (Eds.), *The handbook of experimental economics: Results 1*. Amsterdam: North-Holland.

Midler, E., Pascual, U., Drucker, A. G., Narloch, U., & Soto, J. L. (2015). Unraveling the effects of payments for ecosystem services on motivations for collective action. *Ecological Economics*, *120*, 394–405.

Milgrom, P. (1993). Is sympathy an economic value? Philosophy, economics and the contingent valuation method. In J. A. Hausman (Ed.), *Contingent valuation: A critical assessment* (Vol. 42, pp. 417–435). Emerald Group Publishing Limited.

Millennium Ecosystem Assessment. (2003). *Ecosystems and human well-being: A framework for assessment*. Washington: Island Press.

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

Mishan, E. J. (1971). The postwar literature on externalities: An interpretive essay. *Journal of Economic Literature*, *9*, 1–28.

Mishan, E. J. (1975). *Cost-benefit analysis: An informal introduction*. London: George Allen and Unwin.

Mishan, E. J. (1980). How valid are economic evaluations of allocative changes? *Journal of Economic Issues*, *14*(1), 143–161.

Mishan, E. J. (1988). *Cost-benefit analysis* (4th ed.). London: Unwin Hyman.

Mitchell, R. C., & Carson, R. T. (1989). *Using surveys to value public goods: The contingent valuation method*. Washington: Resources for the Future.

Montgomery, W. D. (1972). Market in licenses and efficient pollution control programs. *Journal of Economic Theory*, *8*, 395–418.

Muller, A., & Vickers, M. (1996). *Communication in a common pool resource environment with probabilistic destruction* (No. 96-06). *McMaster University, Department of Economics, Working Paper*. Retrieved May 25, 2016, from http://www.economics.mcmaster.ca/documents/1996-06.pdf

Muradian, R., Arsel, M., Pellegrini, L., Adaman, F., Aguilar, B., Agarwal, B., … Urama, K. (2013). Payments for ecosystem services and the fatal attraction of win-win solutions. *Conservation Letters*, *00*, 1–6.

Muradian, R., & Cardenas, J.-C. (2015). From market failures to collective action dilemmas: Reframing environmental governance challenges in Latin America and beyond. *Ecological Economics*, *120*, 358–365.

Muradian, R., & Martinez-Alier, J. (2001). Trade and the environment: From a “southern” perspective. *Ecological Economics*, *36*, 281–297.

Murphy, J. J., & Stranlund, J. K. (2006). Direct and market effects of enforcing emissions trading programs: An experimental analysis. *Journal of Economic Behavior and Organization*, *61*(2), 217–233.

Murphy, J. J., & Stranlund, J. K. (2007). A laboratory investigation of compliance behavior under tradable emissions rights: Implications for targeted enforcement. *Journal of Environmental Economics and Management*, *53*(2), 196–212.

Musgrave, R. A. (1986). *Public finance in a democratic society, Volume I: Social goods, taxation and fiscal Policy*. New York: New York University Press.

Narloch, U., Pascual, U., & Drucker, A. G. (2012). Collective action dynamics under external rewards: Experimental insights from Andean farming communities. *World Development*, *40*(10), 2096–2107.

Neurath, O. (1945). Alternatives to market competition. In R. S. Cohen & T. Uebel (Eds.), *Otto Neurath: Economic writings 1904–1945* (pp. 546–548). Dordrecht: Kluwer.

Neurath, O. (1973). Through war economy to economy in kind. In *Empiricism and sociology* (pp. 123–157). Dordrecht: Springer Netherlands.

Nordhaus, W. D. (1993). Rolling the "DICE": An optimal transition path for controlling greenhouse gases. *Resource and Energy Economics*, *15*(1), 27–50.

North, D. C. (1990). *Institutions, institutional change and economic performance*. Cambridge: Cambridge University Press.

Norton, B. (2002). The ignorance argument: What must we know to be fair to the future? In D. W. Bromley & J. Paavola (Eds.), *Economics, ethics, and environmental policy* (pp. 35–52). Oxford: Blackwell Publishing Ltd.

Nowak, M. A., & Sigmund, K. (1992). Tit for tat in heterogeneous populations. *Nature*, *355*, 250–252.

Nozick, R. (1973). Distributive justice. *Philosophy & Public Affairs*, *3*(1), 45–126.

Nozick, R. (1974). *Anarchy, state and utopia*. New York: Basic Books.

O'Neill, J. (2001). Markets and the environment: The solution is the problem. *Economic & Political Weekly*, *36*(21), 1865–1873.

O'Neill, J. (2012). Austrian economics and the limits of markets. *Cambridge Journal of Economics*, *36*(5), 1073–1090.

Oates, W. (1989). The net benefits of incentive-based regulation: A case study of environmental standard setting. *American Economic Review*, *79*(5), 1233–1242.

Ockenfels, A., & Weimann, J. (1999). Types and patterns: An experimental East-West-German comparison of cooperation and solidarity. *Journal of Public Economics*, *71*(2), 275–287.

Okun, A. M. (1975). *Equality and efficiency: The big trade-off*. Washington: Brookings Institution Press.

Organization for Economic Co-operation and Development (OECD). (1996). *Integrating environment and economy: Progress in the 1990s*. Paris: OECD.

Organization for Economic Co-operation and Development (OECD). (1997a). *Reforming environmental regulation in OECD countries*. Paris: OECD.

Organization for Economic Co-operation and Development (OECD). (1997b). *Towards sustainable fisheries: Economic aspects of the management of living marine resources*. Paris. Retrieved July 4, 2016, from http://www.oecd.org/tad/fisheries/studytowardssustainablefisheries-economicaspectsofthemanagementoflivingmarineresources.htm

Organization for Economic Co-operation and Development (OECD). (2006). *Using market mechanisms to manage fisheries: Smoothing the path*. Paris: OECD. Retrieved March 15, 2016, from http://www.keepeek.com/Digital-Asset-Management/oecd/agriculture-and-food/using-market-mechanisms-to-manage-fisheries_9789264036581-en#page1\nhttp://www.oecd.org/tad/fisheries/usingmarketmechanismstomanagefisheriessmoothingthepath.htm\nhttp://www.oecd-ili

Organization for Economic Co-operation and Development (OECD). (2008). *An OECD framework for effective and efficient environmental policies*. Retrieved May 16, 2016, from http://www.oecd.org/greengrowth/tools-evaluation/41644480.pdf

Organization for Economic Co-operation and Development (OECD). (2011). *Trust. Society at a Glance 2011: OECD Social Indicators*. Paris: OECD.

Organization for Economic Co-operation and Development (OECD). (2016). *OECD review of fisheries: Country statistics 2015*. Paris: OECD. Retrieved from http://dx.doi.org/10.1787/rev_fish_stat_en-2015-en

Ostrom, E. (1990). *Governing the commons: The evolution of institutions for collective action*. Cambridge: Cambridge University Press.

Ostrom, E. (1998). A behavioral approach to the rational choice theory of collective action: Presidential address, American Political Science Association, 1997. *American Political Science Review*, *92*(01), 1–22.

Ostrom, E. (1999). Coping with tragedies of the commons. *Annual Review of Political Science*, *2*(1), 493–535.

Ostrom, E. (2000). Collective action and the evolution of social norms. *Journal of Economic Perspectives, 6,* 232-252. http://doi.org/10.1257/jep.14.3.137

Ostrom, E. (2006). The value-added of laboratory experiments for the study of institutions and common-pool resources. *Journal of Economic Behavior and Organization*, *61*(2), 149–163.

Ostrom, E., Burger, J., Field, C. B., Norgaard, R. B., & Policansky, D. (1999). Revisiting the commons: Local lessons, global challenges. *Science*, *284*(5412), 278–282.

Ostrom, E., Gardner, R., & Walker, J. M. (1994). *Rules, games and common-pool resources*. An Harbor: University of Michigan Press.

Ostrom, E., Walker, J., & Gardner, R. (1992). Covenants with and without a sword: Self-governance is possible. *American Political Science Review*, *86*(2), 404–417.

Özkaynak, B., Devine, P., & Rigby, D. (2004). Operationalising strong sustainability: Definitions, methodologies and outcomes. *Environmental Values*, *13*(3), 279–303.

Padilla, E. (2002). Intergenerational equity and sustainability. *Ecological Economics*, *41*, 69–83.

Pagiola, S., Arcenas, A., & Platais, G. (2005). Can payments for environmental services help reduce poverty? An exploration of the issues and the evidence to date from Latin America. *World Developmentevelopment*, *33*(2), 237–253.

Parry, I. W. H., Sigman, H., Walls, M., & Williams, R. C. (2006). The incidence of pollution control policies. In T. Tietenberg & H. Folmer (Eds.), *The international yearbook of environmental and resource economics 2006/2007* (pp. 1–25).

Cheltenham: Edward Elgar.

Pazner, E. A., & Schmeidler, D. (1978). Egalitarian equivalent allocations: A new concept of economic equity. *The Quarterly Journal of Economics*, *92*(4), 671–687.

Pearce, D. (1991). The role of carbon taxes in adjusting to global warming. *The Economic Journal*, *101*(407), 938–948.

Pfaff, A., Vélez, M. A., Ramos, P. A., & Molina, A. (2015). Framed field experiment on resource scarcity & extraction: Path-dependent generosity within sequential water appropriation. *Ecological Economics*, *120*, 416–429.

Pigou, A. C. (1920). *The economics of welfare*. London: Macmillan.

Pirard, R. (2012). Market-based instruments for biodiversity and ecosystem services: A lexicon. *Environmental Science and Policy*, *19-20*, 59–68.

Plott, C. R. (1983). Externalities and corrective policies in experimental markets. *The Economic Journal*, *93*(369), 106–127.

Polanyi, K. (1957). *The great transformation*. Boston: Beacon Press.

Pomeroy, R. S., & Berkes, F. (1997). Two to tango: The role of government in fisheries co-management. *Marine Policy*, *21*(5), 465–480.

Porras, I., Chacón-Cascante, A., Robalino, J., & Oosterhuis, F. (2011). PES and other economic beasts: Assessing PES within a policy mix in conservation. In *Instrument Mixes for Biodiversity Policies. POLICYMIX Report 2/2011* (pp. 119–144). Istanbul: ESEE.

Prediger, S., Vollan, B., & Frölich, M. (2011). The impact of culture and ecology on cooperation in a common-pool resource experiment. *Ecological Economics*, *70*, 1599–1608.

Radikal (2012, January 30). Trol çetesi kör etti. Retrieved from http://www.radikal.com.tr/turkiye/trol-cetesi-kor-etti-1077169/

Rawls, J. (1971). *A theory of justice* (Vol. 1). Cambridge: The Belknap Press of Harvard University Press.

Redford, K. H., & Adams, W. M. (2009). Payment for ecosystem services and the challenge of saving nature. *Conservation Biology*, *23*, 785–787.

Reeson, A. F., & Tisdell, J. G. (2010). The market instinct: The demise of social preferences for self-interest. *Environmental and Resource Economics*, *47*, 439–453.

Reichhuber, A., Camacho, E., & Requate, T. (2009). A framed field experiment on collective enforcement mechanisms with Ethiopian farmers. *Environment and Development Economics*, *14*(05), 641.

Rhoads, T. A., & Shogren, J. F. (1999). On Coasean bargaining with transaction costs. *Applied Economics Letters*, *6*, 779–783.

Ring, I., & Schröter-Schlaack, C. (2011). *Instrument mixes for biodiversity policies*. Leipzig.

Robert N. Stavins. (1995). Transaction costs and tradeable permits. *Journal of Environmental Economics and Management*, *29*, 133–148.

Rode, J., Gómez-Baggethun, E., & Krause, T. (2015). Motivation crowding by economic incentives in conservation policy: A review of the empirical evidence. *Ecological Economics*, *117*, 270–282.

Rodríguez-Sickert, C., Guzmán, R. A., & Cardenas, J.-C. (2008). Institutions influence preferences : Evidence from a common pool resource experiment. *Journal of Economic Behavior & Organization*, *67*(1), 215–227.

Rodriguez-Sickert, C., Guzman, R., & Cárdenas, J. C. (2008). Institutions influence preferences: Evidence from a common pool resource experiment. *Journal of Economic Behavior and Organization*, *67*(1), 215–227.

Rowe, R. D., D'Arge, R. C., & Brookshire, D. S. (1980). An experiment on the economic value of visibility. *Journal of Environmental Economics and Management*, *7*(1), 1–19.

Rustagi, D., Engel, S., & Kosfeld, M. (2010). Conditional cooperation and costly monitoring explain success in forest commons management. *Science*, *890*, 961–965.

Sabah (2016, February 29). Moritanya'da "Vira Bismillah" dediler. Retrieved from http://www.sabah.com.tr/egeli/2016/02/29/moritanyada-vira-bismillah-dediler

Sagoff, M. (2008). *The economy of the earth: Philosophy, law, and the environment* (2nd ed.). Cambridge: Cambridge University Press.

Saldarriaga-Isaza, A., Villegas-Palacio, C., & Arango, S. (2015). Phasing out mercury through collective action in artisanal gold mining: Evidence from a framed field experiment. *Ecological Economics*, *120*, 406–415.

Sally, D. (1995). Conversation and cooperation in social dilemmas: A Meta-analysis of experiments from 1958 to 1992. *Rationality and Society*, *7*(1), 58–92.

Samuels, W. J. (1992). Welfare economics, power and property. In *Essays on the economic role of government* (pp. 56–138). New York: New York University Press.

Sandmo, A. (2000). *The public economics of the environment*. New York: Oxford University Press.

Santos, R., Schröter-Schlaack, C., Antunes, P., Ring, I., & Clemente, P. (2015). Reviewing the role of habitat banking and tradable development rights in the conservation policy mix. *Environmental Conservation*, *42*(04), 294–305.

Scheffler, S. (1982). *The rejection of consequentialism*. Oxford: Oxford University Press.

Schellnhuber, H. J., Cramer, W., Nakicenovic, N., Wigley, T., & Yohe, G. (2006). *Avoiding dangerous climate change*. Cambridge: Cambridge University Press.

Schmid, A. A. (1987). *Property, power, and public choice*. New York: Praeger.

Schotter, A., Weiss, A., & Zapater, I. (1996). Fairness and survival in ultimatum and dictatorship games. *Journal of Economic Behavior & Organization*, *31*(1), 37–56.

Scott, A. (1955). The fishery: The objectives of sole ownership. *Journal of Political Economy*, *63*(2), 116–124.

Sen, A. K. (1979). *Collective choice and social welfare* (2nd ed.). Amsterdam: North-Holland.

Sen, A. K. (1982). Approaches to the choice of discount rates for social benefit-cost analysis. In R. Lind, K. J. Arrow, & G. Corey (Eds.), *Discounting for time and risk in energy policy* (pp. 172–206). Washington: Resources for the Future.

Sen, A. K. (1987). *On ethics and economics*. Malden: B. Blackwell.

Sen, A. K. (2002). *Rationality and freedom*. Cambridge: The Belknap Press of Harvard University Press.

Shogren, J. F. (2005). Experimental methods and valuation. In K. G. Mäler & J. R. Vincent (Eds.), *Handbook of environmental economics, volume 2: Valuing dnvironmental changes* (pp. 969–1027). Amsterdam: North-Holland.

Shogren, J. F., & Kask, S. B. (1992). Exploring the boundaries of the Coase theorem: Efficiency and rationality given imperfect contract enforcement. *Economics Letters*, *39*(2), 155–161.

Shogren, J. F., Shin, S. Y., Hayes, D. J., & Kliebenstein, J. B. (1994). Resolving differences in willingness-to-pay and willingness to accept. *American Economic Review*, *84*(1), 255–270.

Shu, L. L., Gino, F., & Bazerman, M. H. (2011). Dishonest deed, clear conscience: When cheating leads to moral disengagement and motivated forgetting. *Personality and Social Psychology Bulletin*, *37*(3), 330–349.

Sills, D. L. (1968). *International Encyclopedia of Social Sciences*. New York: Macmillan.

Sinden, J. (1988). Empirical tests of hypothetical bias in consumers' surplus surveys. *Australian Journal of Agricultural Economics*, *32*(2-3), 98–112.

Smith, T., Gibbs, M., & Smith, D. (2009). Fishing for more effective incentives. *Science*, (January), 337–339.

Smith, V. (2008). Foreword. In T. L. Cherry, S. Kroll, & J. F. Shogren (Eds.), *Environmental economics: Experimental methods*. London and New York: Routledge.

Solow, R. M. (1991). Sustainability: An economist's perspective. In *Economics of the environment, selected readings* (Vol. 3, pp. 179–187). Retrieved October 10, 2014, from http://www.isites.harvard.edu/fs/docs/icb.topic203569.files/Solow.Sustainability_An_Economists_Perspective._1993.pdf

Söderbaum, P. (1990). Neoclassical and institutional approaches to environmental economics. *Journal of Economic Issues*, *XXIV*, 27–30.

Spash, C. L. (1993). Economics, ethics, and long-term environmental damages.

Environmental Ethics, *15*(2), 117–132.

Spash, C. L. (1997). Ethics and environmental attitudes with implications for economic valuation. *Journal of Environmental Management*, *50*(4), 403–416.

Spash, C. L. (2008). Contingent valuation design and data treatment: If you can't shoot the messenger, change the message. *Environment and Planning C: Government and Policy*, *26*(1), 34–53.

Spash, C. L. (2010). The brave new world of carbon trading. *New Political Economy*, *15*(2), 169–195.

Squires, D., Kirkley, J., & Tisdell, C. A. (1995). Individual transferable quotas as a fisheries management tool. *Reviews in Fisheries Science*, *3*(2), 141–169.

Stranlund, J. K., & Murphy, J. J. (2013). Imperfect enforcement of emissions trading and industry welfare: A laboratory investigation. In J. A. List & M. K. Price (Eds.), *Handbook on experimental economics and the environment*. Cheltenham: Edward Elgar.

Sturm, B., & Weimann, J. (2006). Experiments in environmental economics and some close relatives. *Journal of Economic Surveys*, *20*(3), 419–457.

Sunstein, C. R. (1997). *Free markets and social justice*. New York: Oxford University Press.

T24. (2012, September 2). Balıkçılar eyleme gitti, balık fiyatları tavan yaptı. Retrieved July 10, 2016, from http://t24.com.tr/haber/balikcilar-eylemde-gitti-balik-fiyatlari-tavan-yapti,212163

T24. (2016, February 16). Yasa dışı avcılık denizleri tüketiyor. Retrieved July 5, 2016, from http://t24.com.tr/haber/yasa-disi-balikcilik-denizleri-tuketiyor,328357

Thaler, R. (1992). *The winner's curse: Paradoxes and anomalies of economic life*. New York: Free Press.

The World Bank. (2015). *Overview: Human decision making and development policy*. *World Development Report 2015 "Mind, Society, and Behavior."*

Thogersen, J. (2003). Monetary incentives and recycling: Behavioural and psychological reactions to a performance-dependent garbage fee. *Journal of Consumer Policy*, *26*(2), 197–228.

Tietenberg, T. (1990). Economic instruments for environmental regulation. *Oxford Review of Economic Policy*, *6*(1), 17–33.

Tietenberg, T. (2002). The tradable permit approach to protecting the commons: What have we learned? In E. Ostrom (Ed.), *The drama of the commons* (pp. 197–232). Washington: National Academy Press.

Tisdell, J., Ward, J., & Capon, T. (2004). Impact of communication and information on a heterogeneous close water catchment environment. *Water Resources Research*, *40*(9).

Titmuss, R. M. (1971). *The gift relationship: From human blood to social policy*. New York: Pantheon Books.

Tokaç, A., Ünal, V., Tosunoğlu, Z., Lök, A., Ceyhan, T., & Özbilgin, H. (2014). Summary and analysis of the available information for the ecosystem approach to fisheries management in Turkey. *Scientia Marina*, *78,* 29–36.

Tören, T. (2007). *Yeniden yapılanan dünya ekonomisinde Marshall planı ve Türkiye uygulaması*. Sosyal Araştırmalar Vakfı.

Turkish Statistical Institute (TURKSTAT). (2010). *Türkiye istatistik yıllığı 2009*. Anakara: Türkiye İstatistik Kurumu.

Turkish Statistical Institute (TURKSTAT). (2014). *Fishery statistics 2013*. Ankara: Turkish Statistical Institute Yücetepe.

United Nations Environment Programme (UNEP). (2004). *The use of economic instruments in environmental policy: Opportunities and challenges*.

Ünal, V., & Ekmekçi, B. (2015). *Türkiye'de on metre ve üzeri balıkçı teknelerinde geri-alım programının değerlendirilmesi*. İzmir.

Ünal, V., & Göncüoğlu, H. (2012). Fisheries management in Turkey. In A. Tokaç, A. C. Gücü, & B. Öztürk (Eds.), *The state of the Turkish fisheries* (pp. 516–550). Turkish Marine Research Foundation.

Ünal, V., Güçlüsoy, H., & Franquesa, R. (2009). A comparative study of success and failure of fishery cooperatives in the Aegean, Turkey. *Journal of Applied Ichthyology*, *25*, 394–400.

van den Bergh, J. C. J. M., Ferrer-i-carbonell, A., & Munda, G. (2000). Alternative models of individual behaviour and implications for environmental policy. *Ecologial Economics*, *32*, 43–61.

van Dijk, E., & van Knippenberg, D. (1996). Buying and selling exchange goods: Loss aversion and the endowment effect. *Journal of Economic Psychology*, *17*, 517–524.

Varian, H. R. (1975). Distributive justice, welfare economics, and the theory of fairness. *Philosophy & Public Affairs*, *4*(3), 223–247.

Vatn, A. (2002). Efficient or fair: Ethical paradoxes in environmental policy. In D. W. Bromley & J. Paavola (Eds.), *Economics, ethics, and environmental policy* (pp. 133–147). Oxford: Blackwell Publishing Ltd.

Vatn, A. (2005). Rationality, institutions and environmental policy. *Ecological Economics*, *55*, 203–217.

Vatn, A. (2009a). An institutional analysis of methods for environmental appraisal. *Ecological Economics*, *68*, 2207–2215.

Vatn, A. (2009b). Cooperative behavior and institutions. *The Journal of Socio-Economics*, *38*, 188–196.

Vatn, A. (2010). An institutional analysis of payments for environmental services. *Ecological Economics*, *69*, 1245–1252.

Vatn, A., & Bromley, D. W. (1997). Externalities: A market model failure. *Environmental and Resource Economics*, *9*, 135–151.

Velez, M. A., Murphy, J. J., & Stranlund, J. K. (2010). Centralized and decentralized management of local common pool resources in the developing world: Experimental evidence from fishing communities in Colombia. *Economic Inquiry*, *48*(2), 254–265.

Velez, M. A., Stranlund, J. K., & Murphy, J. J. (2009). What motivates common pool resource users? Experimental evidence from the field. *Journal of Economic Behavior and Organization*, *70*(3), 485–497.

Velez, M. A., Stranlund, J. K., & Murphy, J. J. (2012). Preferences for government enforcement of a common pool harvest quota: Theory and experimental evidence from fishing communities in Colombia. *Ecological Economics*, *77*, 185–192.

Venkatachalam, L. (2004). The contingent valuation method: A review. *Environmental Impact Assessment Review*, *24*(1), 89–124.

Visser, M., & Burns, J. (2015). Inequality, social sanctions and cooperation within South African fishing communities. *Journal of Economic Behavior and Organization*, *118*, 95–109.

Vollan, B. (2008). Socio-ecological explanations for crowding-out effects from economic field experiments in southern Africa. *Ecological Economics*, *67*, 560–573.

Walker, J. M., & Gardner, R. (1992). Probabilistic destruction of common-pool resources: Experimental evidence. *The Economic Journal*, *102*(414), 1149–1161.

Ward, J. R., & Tisdell, J. G. (2008). Aligning policy and real world settings. In T. L. Cherry, S. Kroll, & J. F. Shogren (Eds.), *Environmental economics, experimental economics* (pp. 100–130). London and New York: Routledge.

World Business Council for Sustainable Development [WBCSD]. (2011). *Guide to corporate ecosystem valuation: A Framework for improving corporate decision-making*. Geneva.

Weitzman, M. L. (1974). Prices vs. Quantities. *The Review of Economic Studies*, *41*(4), 477–491.

Weitzman, M. L. (1998). Why the far-distant future should be discounted at its lowest possible rate. *Journal of Environmental Economics and Management*, *36*(3), 201–208.

Williamson, O. E. (2000). The new institutional economics: Taking stock, looking ahead. *Journal of Economic Literature*, *38*(3), 595–613.

World Values Survey (WVS). (2016). World values survey, wave 6: 2010-2014.

Wunder, S. (2005). Payments for environmental services: Some nuts and bolts. *CIFOR Occasional Paper No. 42*, (42).

Zerbe, R. O., & McCurdy, H. E. (1999). The failure of market failure. *Journal of Policy Analysis and Management*, *18*(4), 558–578.

Zhong, C.-B., & Loewenstein, J. (2007). Speaking the same language. *Journal of Conflict Resolution*, *51*(3), 431–456.

Zylics, T. (2000). Goals, principles and constraints in environmental policies. In H. Folmer & L. h: Gabel (Eds.), *Principles of environmental and resource economics* (pp. 130–153). Cheltenham: Edward Elgar.