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Fisheries Management under Individual Transferable Quota

Outcomes for Ecology and Equity

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Dissertation submitted in partial fulfillment of a double *Philosophiae Doctor* degree in Environment and Natural Resources at the University of Iceland and in physical geography at the Department of Physical Geography at Stockholm University

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Summary

The management of marine resources pose a difficult commons problem as monitoring behavior is difficult and benefit flows from the resources are uncertain. Implementing individual transferable quota (ITQ) is a management regime in which quasi-property rights are assigned for an often mobile and uncertain environmental resource, fish or marine invertebrates. This thesis addresses sustainability impacts of ITQ's as a fisheries management tool. The findings demonstrate that fisheries management regimes in which fisheries opportunities are allocated as quota and / or are allocated individually experience reduced overfishing compared to controls that do not have these attributes (**Paper I**), however the analysis found less support for transferability and no support for longer duration being associated to any change in the probability of overfishing. In addition, a longitudinal study showed that with an adaptive design ecological and economic goals could be balanced in an important mixed fishery in Iceland (**Paper II**), and based on such findings suggested that several policy changes could be implemented to modify the ecological risk of catch-quota balancing allowances. Additional longitudinal analyses allowed to conclude that rapid consolidation in an important small-boat fishing sector in Iceland, which may have had negative implications for local fishing communities (**Paper III**), and that on average since the introduction of ITQ's total amount of quota traded stayed below around 60% for the main commercial species in the Icelandic ITQ system. Moreover, the results of **Paper IV** also show that in case of a credible announcement of quota revocation in the future there would be scope for policy reform. Finally, research is beginning to emerge that shows that marine species are unequally affected by climate change. In a final chapter the analyses show that under different scenarios of global change a re-shaping of the Icelandic foodweb is likely (**Paper V**). The re-shaping of the foodweb will be to the benefit of some resource users and to the loss of others. In general, the findings from all the analyses together demonstrate that there could be benefits to individual quota implementation for fisheries sustainability and that some of the hypothesized trade-offs could potentially be balanced, the thesis highlights ways forward in investigating the common pool problems in fisheries management.

Útdráttur

Stjórnun fiskveiðiauðlinda skapar vandamál varðandi sameign þar sem eftirlit með hegðun er ekki án vandkvæða og flæði ávinnings frá auðlindinni er óljós. Innleiðing framseljanlegs kvóta er stjórnunarfyrikomulag þar sem quasi-eignarétti er úthlutað fyrir umhverfisauðlind, fisk eða hryggleysingja sjávar, sem er hreyfanleg og ríkir um óvissa. Þessi doktorsritgerð fjallar um sjálfbærnisáhrif framseljanlegs kvóta sem stjórnun fiskveiðiauðlinda. Niðurstöðurnar sýna fram á að stjórnunarkerfi fiskveiða sem byggja tækifæri til veiða á kvótum eða úthlutunum til einstaklinga, leiða frekar til minni ofveiði heldur en þau sem hafa ekki þá eiginleika (fræðigrein I), en samt sem áður benti greiningin til þess að lítil tengsl væru á milli tegundatilfærslu og líkinda á minni ofveiði, og engin tengsl á milli lengra tímabils án breytinga á kerfinu og líkinda á ofveiði. Þar að auki sýndi langtímarannsókn fram á að með aðlögunarhönnun væri hægt að gæta jafnvægis milli vistfræðilegra og efnahagslegra markmiða í mikilvægum blönduðum veiðum við Íslandsstrendur (fræðigrein II). Lagðar voru fram tillögur um innleiðingu margsháttar stefnubreytinga til þess að draga úr vistfræðilegum áhættum sem snúa að tegundatilfærslum sem byggðar voru á þeim niðurstöðum. Frekari langtímarannsóknir gáfu tilefni til að draga þá ályktun að hröð samþjöppun hafi átt sér stað í þeim mikilvæga geira sem fiskveiði smábáta er á Íslandi, sem gæti hafa leitt til neikvæðra áhrifa á samfélög sem byggist upp á fiskveiðum (fræðigrein III), og að síðan kvótakerfið var innleitt hafi að magn þess kvóta sem höfð voru viðskipti með, haldist að meðaltali um tæplega 60% fyrir þær tegundir sem eru efnahagslega mikilvægastar fyrir íslenska kvótakerfið. Enn fremur benda niðurstöður úr fræðigrein IV til að ef kæmi til trúverðugrar tilkynningar um afturköllun kvóta í framtíðinni gæfi það færi á umbótum á stefnu. Að lokum benda rannsóknir sem fram hafa komið nýlega til þess að frekar megi gæta áhrifa hlýnunar jarðar á sjávartegundir umfram aðrar. Í lokakapla ritgerðarinnar benda greiningar til að íslenskur fæðuvefur tekur breytingum undir mismunandi atburðarásam breytinga á jörðinni (fræðigrein V). Þær breytingar á fæðuvefnum munu gagnast einum hóp auðlindanotkenda á meðan aðrir tapa á þeim. Almennt benda niðurstöður greininganna þegar þær eru settar saman til þess að innleiðingar einstaklings kvóta gætu haft í för með sér kosti fyrir sjálfbærni fiskveiðiauðlinda og að hægt væri að jafna út suma af þeim fórnarskiptum sem gerðar hafa verið tilgátur um, og þessi ritgerð dregur upp mynd af frekari rannsóknum sem varpað gætu ljósi á sameigna vandamál í fiskveiðistjórnun.

Sammanfattning

Förvaltningen av fiskeresurser utgör ett svårt "commons"-problem, dvs ett problem rörande förvaltningen av en gemensamt utnyttjad men inte enskilt ägd resurs, eftersom övervakningen av resurserna liksom nyttan från dessa är osäkra. Implementeringen av individuellt transfererbara kvoter (individual transferable quotas ITQ) är en metodik för att skapa kvasiäganderättigheter till en ofta rörlig och osäker naturresurs som fisk eller marina evertebrater. Den här avhandlingen berör flera viktiga frågor inom fiskeriförvaltningen men med fokus på ITQ:er som förvaltningsverktyg. Avhandlingen visar att förvaltningsregimer där fiskemöjligheterna fördelas som kvoter och/eller individuellt ger minskat överfiske jämfört med kontroller som saknar dessa egenskaper (**artikel I**). Den fann emellertid inget stöd för att transfererbarhet eller längre tidshorisonter var kopplade till några förändringar i sannolikheten för överfiske. Dessutom visades för ett viktigt isländskt blandat fiske att, med en adaptiv design, ekologiska och ekonomiska mål kunde balanseras gentemot varandra (**artikel II**) samtidigt som resultatet pekar mot flera olika policyförändringar för att modifiera den ekologiska risken i kvotbalanserade tilldelningar. Avhandlingen visar vidare på en snabb konsolidering inom fiskesektorn för mindre fiskebåtar på Island vilket kan ha en negativ påverkan på mindre fiskesamhällen (**artikel III**), och att i snitt kvothandeln inom det Isländska ITQ-systemet för de viktigare kommersiella arterna låg under 60%. Avhandlingen påvisar dessutom att vid en trovärdig deklaration av ett kvotåterkallande i framtiden så skulle det finnas utrymme för policyformer (**artikel IV**). Slutligen, en framväxande forskning pekar mot att marina arter påverkas av klimatförändringar på ett skiljaktigt sätt. Avhandlingens sista kapitel visar att olika scenarier för globala förändringar sannolikt kommer att omforma den isländska marina näringsväven (**artikel V**). En omformning av näringsväven kommer att gynna vissa naturresursbrukare medan andra kommer att missgynnas.

Samenvatting

Het beheer van marine vis en ongewervelden stelt het visserijbeleid voor uitdagingen, gezien het feit dat het lastig is het gedrag van vissers te monitoren. Ook zijn er vele onzekerheden voor de vissers zelf omtrent de vangst, vanwege deze onzekerheden weegt het kortetermijnbelang soms zwaarder dan de (duurzaamheids-)belangen op de lange-termijn. Individuele (en verhandelbare) quota zijn een vorm van visserijbeleid waarin quasi-eigendomsrechten aan vissers worden gegeven/verkocht. Dit proefschrift adresseert verschillende duurzaamheidsaspecten van individuele quota. De resultaten demonstreren dat beleidsvormen waarin toegang tot de visserij wordt verleend in de vorm van quota en/of aan individuele bedrijven wordt verleend geassocieerd zijn aan een reductie in overbevissen (**Paper I**). Echter, de analyse vond weinig bewijs voor een verband tussen duurzaamheid en verhandelbaarheid van quota, en de analyse vond geen enkel bewijs voor een langere allocatie (langer dan één jaar) van individuele quota en duurzame exploitatie. In een tijd-serie analyse werd gevonden dat met een adaptief ontwerp van visserijbeleid ecologische en economische doelen voorzichtig tegen elkaar kunnen worden afgewogen, in een visserij waarin de vangst gemend is (**Paper II**). Dit artikel richtte zich op een specifiek mechanisme in IJsland waarin vissers vangst in overeenstemming kunnen brengen met quota, na de vangst. Op basis van deze bevindingen worden verschillende beleidsvoorstellen gedaan om het risico voor de visbestanden te verminderen. Verdere tijd-serie analyses demonstreerden dat er snelle consolidatie is opgetreden in een belangrijke visserij met kleinere boten in IJsland (**Paper III**). Zulke consolidatie zou negatieve consequenties kunnen hebben gehad in de afgelegen dorpen in IJsland waar visserij een belangrijke economische sector is. **Paper IV** laat zien dat er ruimte is voor herziening van het quota-beleid in IJsland wanneer er een geloofwaardige aankondiging wordt gedaan van een herroeping van de quota (quota wordt bijvoorbeeld voortaan geveild). In een laatste artikel wordt door middel van een analyse met een ecosysteem-model het IJslandse ecosysteem bestudeerd in de toekomst, onder verschillende scenario's van opwarming en verzuring van de oceaan (**Paper V**). De analyse laat zien dat sommige soorten zullen toenemen in biomassa en anderen zullen verminderen, wat ten gunste zal zijn van sommige vissers en ten nadele van anderen. In het algemeen, demonstreren de bevindingen van alle analyses in dit proefschrift dat de introductie van individuele quota gunstig van zijn voor de duurzaamheid van de visserij en dat adaptief beleid noodzakelijk is. Het proefschrift benadrukt belangrijke onderwerpen voor toekomstig onderzoek in visserijbeheer.

To my family

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Abbreviations

EEZ – Exclusive Economic Zone

CE– Cod Equivalent

DiD – Difference in Differences

I – Individual

IQ – Individual Quota

ITQ – Individual Transferable Quota

OA – Ocean Acidification

pH – Potential of Hydrogen

T – Transferable

TAC – Total Allowable Catch

TGT – Transitional Gains Trap

Q – Quota

Thesis papers

This doctoral thesis consists of two published papers and three manuscripts, listed below (I-V). The published papers are reprinted under the terms of the Creative Commons Attribution 4.0 International License (<http://creativecommons.org/licenses/by/4.0/>).

I Maartje Oostdijk & Griffin Carpenter, Which attributes of fishing opportunities are associated with sustainable fishing? (*in review* ***Environmental Research Letters***)

II Maartje Oostdijk, Conor Byrne, Gunnar Stefansson, Maria J. Santos, Pamela J. Woods, Catch–quota matching allowances balance economic and ecological targets in a fishery managed by individual transferable quota ***Proceedings of the National Academy of Sciences***, 117 (40) 24771-24777, <https://doi.org/10.1073/pnas.2008001117>

III Maartje Oostdijk, Maria J. Santos, Sveinn Agnarsson, Pamela J. Woods (2019) Structure and evolution of cod quota markets networks in Iceland over times of financial volatility ***Ecological Economics***, 159: 279-290, <https://doi.org/10.1016/j.ecolecon.2019.01.035>

IV Conor J. Byrne, **Maartje Oostdijk**, Sveinn Agnarsson, Brynhidur Davidsdóttir, Quota Trade and potential measures of the Transitional Gains Trap - A quantitative analysis of the Icelandic ITQ system (*Manuscript*)

V Maartje Oostdijk, Erla Sturludóttir, Maria J. Santos, Risk assessment for key socio-economic and ecological species in a sub-arctic marine ecosystem under combined ocean acidification and warming (*in review* ***Ecosystems***)

Author contributions

I MO collected the data for several regions jointly with GC, MO lead the research design and carried out the analysis. MO wrote the first draft of the paper, GC helped writing.

II MO collected the data jointly with CB, MO lead the research design and MO and CB jointly carried out the analysis. PW and MJS also designed the research, GS aided with the methodology. MO wrote the first draft of the paper all co-authors helped writing.

III MO conceived of the research idea, lead the data collection and carried out the analysis, MO wrote the article with help of MJS SA and PJW.

IV CB conceived of the idea and designed the study. MO collected data and carried out part of the analysis. CB wrote the first draft and MO, SA and BD helped with the writing.

V MO conceived of the research idea, carried out literature research and ran modeling scenarios with the help of ES. MO wrote the first draft of the manuscript and ES and MJS helped with writing.

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1 Introduction

Advances in fishing techniques, globalization, the fast development of new markets, illegal fishing and lack of institutions have been claimed to cause a true “tragedy of the commons” where “roving bandits” with no incentives to conserve marine resources deplete populations that could be beneficial for societies and ecosystems around the globe (Berkes et al. 2006). Fortunately, after decades of severe exploitation, efforts to restore marine populations are finally paying off (Duarte et al., 2020). Fish is a renewable resource and if well managed can serve as a high value food source as well as offer economic and cultural benefits to societies around the world now as well as in the distant future (Costello et al., 2016). Moreover, we rely on healthy marine ecosystems for many non-monetized ecosystem services (Worm et al., 2009). Sustaining lower exploitation rates can be beneficial for both the ecosystem as well as catches (Costello et al. 2016), whereas decline and rebuilding can incur significant costs (Worm et al., 2009). The ultimate risk of overfishing being a shift towards an undesirable state in which the population is “collapsed” and rebuilding is difficult or impossible due to population and ecosystem feed-backs (Holling, 1996). For instance, having too few individuals to sustain a healthy population (i.e. allee effects such as mate limitation or reduced cooperation (Winter et al., 2020)). Because of climate change decreased productivity of fish populations is projected globally, but if well-managed, even under decreased productivity of the environment, yield and profits are forecast to increase (Gaines et al., 2018). Fisheries science and management are inherently interdisciplinary fields, and success of fisheries management should thus not hinge on a single of the three pillars of sustainability; ecological, social and economic concerns are equally important. In fact, there are indications that fisheries management can address economic, social and ecological objectives synergistically (Asche et al., 2018).

Current fisheries management counts several successful rebuilt and well-managed stocks, under a wide range of management solutions (Melnychuk et al., 2013). An often-coined solution to end the tragedy of the commons is the implementation of property rights in the form of individual transferable quota (ITQ) or individual quota (IQ), to align fishers economic incentives with a long-term interest in the sustainability of the target stock (Branch, 2009). Currently ~25% of global landings are caught under some form of quota management (EDF, 2018). Individual quota systems are usually managed by implementing a total allowable catch (TAC) based on scientific evidence and by dividing this TAC into individual shares. In the majority of these systems there is some form of transferability, permanent and/or temporary (Essington 2010; Melnychuk et al. 2016). ITQ systems have been praised for their potential economic efficiency (Newell, Sanchirico, and Kerr, 2005) and possible positive outcome on the status of marine stocks (Costello, Gaines, and Lynham 2008). On the other hand, in many ITQ systems there has been substantial consolidation of quota rights into the hands of fewer holders (Agnarsson et al., 2016), which has led to negative impacts on fishing communities (McCay

1995; Putten and Gardner 2010). In addition, in most systems, fishing rights have been freely granted to fishers in proportion to their historical catches which is another controversial (and possibly avoidable) aspect of these regimes with possible adverse societal, economic and even ecological consequences (Lynham, 2014; McCay, 1995). This aspect is controversial as the most aggressive harvesters (having caused potentially the most environmental destruction) are allocated the largest shares, and new entrants to the system will have to pay for their harvest rights in contrast to older incumbents remaining in the system. Such implications need to be considered, especially since it is becoming apparent that global climate change will have severe implications for the equitable distribution of marine resources (Cheung et al., 2010).

Thébaud, Innes, and Ellis (2012) pointed out that there were few systematic empirical assessments of ITQ regimes and their effects on marine populations. A similar lack of empirical information has been signaled for the quota market (Matthiasson 2012; Holland et al. 2015), the instrument through which fishing operations under an ITQ system achieve greater efficiency. In recent years the number of studies on ITQs and ecological outcomes has been growing. Some of this work is case-study based (Grimm et al., 2012), some of these studies compare across case studies (Chu, 2009; Essington, 2010) while others use global databases which contain stock assessments or catch levels of fisheries around the world (Costello, Gaines, and Lynham 2008; Essington et al. 2012; Melnychuk et al. 2012; Melnychuk et al. 2016). While there is some evidence that quota-systems (including ITQ and IQ) meet fisheries management goals, the evidence on which aspects of quota management regimes are causing these beneficial effects is still sparse. Is *transferability*, *individual allocation* or management by means of *quota* associated with good management outcomes? (Gibbs, 2009). Moreover, other fisheries management regimes have gotten disproportionately little attention. This thesis contributes to both of these gaps in the literature in paper I, by studying systems by their attribute (I, T and Q) and perform a comparative study over a wide array of fisheries management systems to examine which attribute is associated with sustainable exploitation. Some see tradability as a key element to the success of ITQ's as an economic tool: quota is presumed to flow to the most efficient companies. Other systems have not implemented tradability because prevention of consolidation of quota into the hands of fewer holders and away from rural areas are key issues in these systems (e.g. Norway and Japan). The implementation of ITQ's is phrased as a trade-off between economic efficiency and predictability on the one hand and equity on the other hand. Predictability of ITQ systems has also been linked to better ecological outcomes (Essington, 2010; Melnychuk et al., 2012). In this thesis fisheries management under individual transferable quota are investigated, aiming at the unraveling of impacts of fisheries management attributes on the health of marine populations, and the impact of ITQ systems on equity with a focus on access to fishing opportunities. The papers in this thesis focus largely on the Icelandic system as a case study of ITQ management. The overarching question guiding the research herein is: how do individual transferable quota management in fisheries perform across the different domains of sustainability?

1.1 Theoretical background and previous research

1.1.1 Ocean fisheries a (tragic) common pool resource

Ocean fisheries are a difficult natural resource to manage collectively, even when they are taking place in a countries' exclusive economic zone (EEZ). This is because 1) the boundaries of the resource are difficult to define and usually extend over large areas, 2) the resource is mobile, fish migrate (Fulton, 2011), 3) there are uncertainties regarding the harvestable levels due to marine ecosystem dynamics (Poos et al., 2010; Punt et al., 2013), and 4) it is difficult to monitor appropriators (i.e. harvesters), and appropriators are often even in different nations (Bromley, 2009; Ostrom et al., 2007). The phrasing "tragedy of the commons" (popularized by: Hardin, 1969) is frequently invoked in fisheries and fisheries-economics literature. This concept is invoked to sketch two alternatives: either the government manages the resource by a strong top-down control ("command and control", (Birkenbach et al., 2017)), or some way is found to align economic incentives of the harvesters to the sustainability of the resource in the long run through privatizing the resource (and in the fisheries case the access to the resource) (Arnason, 2007a).

This type of thinking forgoes advances in our understanding of how humans have self-organized and have been perfectly able to manage resources without privatization or government interventions probably since the first agricultural societies (Kohler, 1992). There are ample examples of successfully managed commons, e.g. the famous case of lobster fisheries on the North East coast of the US or community controlled Amerindian hunt for beavers (Berkes et al., 1989), and it has long been demonstrated that the "tragedy of the commons" is not a default trajectory for all shared resources (Ostrom, 2010). Even in complicated resources as marine fish, successful cases of collective "self-organized" management have been shown to exist. In such cases, for instance, cooperatives form that set their own total allowable catches and regulate fishing activity amongst a group (Leslie et al., 2015).

Moreover, it is not exactly clear that privatization of resources necessarily results in better outcomes. For instance, there are examples of privatized agricultural fields with degrading soil and occasionally even erosion which is a good example of why privatization and sustainability incentives do not always seem to align and where additional regulation/government intervention has been needed to guard against top-soil loss (Bromley, 2009). In addition, thinking that privatization versus government intervention are the only two options in solving common pool resource problems forgoes the complexity of social-ecological systems and their management. It is usually not one single intervention or one single level of governance/government that is associated to the management of natural resources, but empirical studies have shown that there are many layers and many structures associated to these (Ostrom, 1990). For instance, individually assigned rights may be (partially) grouped for a collective benefit (Ostrom, 1990); further, even if rights are individually assigned, governments often play a role with management and enforcement and collective initiatives will exist in combination with private property and government intervention.

Designing a system in a top-down fashion and imposing it on users of the common pool resource is not as successful or adaptive as working with the users of a common pool resource over time to develop a system that matches the ecological system and the practices of local resource users, their norms or the long-term economic wellbeing of the harvesters (Ostrom, 2010a). An example of the perceived importance of certain management measures is that when contacting fisheries experts for Paper I, the experts often told us that the fishery was managed for 100% under a form of quota management, but also for 100% under a form of effort management. The divide that was made in Paper I between input (control) systems and output systems is to some extent artificial and most output systems have input restrictions at the same time, and these can be perceived as very important by those who are managing a local resource. In the literature, this type of (even more simplistic!) type of classification is common, but it may overlook important combinations of management measures for the local context. In Paper II of this thesis a system in Iceland's multi-species fishery is investigated, the high level of tailoring to the context makes this system a good example of how resource management can be adapted over time to the local economic and ecological context.

1.1.2 Sustainability effects of I(T)Q's, theoretical background

Marine fish stocks do not naturally lend themselves to property rights as it is impossible to demarcate individual fish as belonging to different owners (or even resources within a given national territory), or for owners to catch the fish that only belong to them. The mobility of most fish stocks prevents ownership by sea area. The solution found by fisheries management has been to develop ownership of the right to fish ('fishing opportunities') rather than the fish themselves (e.g. Arnason, 2013). Under individual quota systems (IQ or ITQ when quota is transferable), a quota limit on the amount of fish harvest is held by a specified fisher, company, or license. When the time duration of IQs is sufficiently long and secure, it is argued that the alignment of the costs and benefits of fishing extend forward in time such that the long-term sustainability of the fish stocks is in the fisher's own interest as they will bear the consequences of (un)sustainable behavior. These aspects of IQs ought to make fishers good stewards of their target stocks (Arnason, 2007a). A possible link between such stewardship and health or marine stocks is proposed to follow through lobbying for different quota settings, i.e. it is said that if long-run ownership of the asset is not certain, individuals should have a greater incentive to lobby for a higher TAC. There is also anecdotal evidence of fishers asking managers for lower quota under IQ systems (Costello and Grainger, 2018; Isaksen and Richter, 2019), although there is also anecdotal evidence of the reverse (The conversation, 2017).

For marine fisheries in particular there are several reasons to be skeptical of the often-claimed link between individual ownership and resource stewardship. First, since the common pool nature of the resource remains even when fishing opportunities are privately owned, gains from overfishing still accrue individually while costs are still shared collectively (Bromley, 2009; Sumaila, 2010). Second, there are many opportunities to 'cheat' in marine fisheries. Examples include high-grading, when selecting more valuable individuals of a stock and throwing undersized individuals overboard, quota-busting, when fishing more than quota allocations and finally, through discarding by throwing undesired species over board (Bromley, 2009; Sumaila, 2010).

Other ecological benefits from IQs are said to arise from the reduction of the ‘race to fish’ that occurs when a quota limit is held commonly and harvested on a ‘first come, first serve’ basis (Birkenbach et al., 2017). By eliminating this racing behavior, IQs should reduce the amount of lost fishing gear by causing a less rushed pace of fishing and thus limit indirect mortality through ‘ghost fishing’. A longer, more planned fishing season should also allow greater selectivity as fishers avoid certain times and areas of the sea to protect vulnerable stocks (Branch et al., 2006), IQs would thus reduce incentives for discarding (Branch et al., 2006).

1.1.3 Empirical ecological findings regarding I(T)Q’s

Because many of the theoretical claims are controversial, empirical work on the question whether sustainability is linked to the attribution of property rights in marine systems is needed. The seminal empirical study on management systems and the sustainability of marine fisheries comes from Costello et al. (2008), which concluded that a fisheries collapse is less likely to occur in systems managed by ITQs. In this study, ITQ systems were compared to all non-ITQ of fisheries management, which includes a great diversity of fisheries management systems including fisheries that have no management system in place at all. It is therefore unclear whether the ITQs outperform IQs, output-based systems fished collectively (Q) or even having any management system in place. There are also issues with the indicator of fisheries sustainability in Costello et al. (2008) as landings data (or even catch data) may not reflect the sustainability of fish stocks (Branch et al., 2011).

Recently Isaksen and Richter (2019) followed the approach of Costello et al. (2008) but used stock assessment data as well as landings data. In this study, ‘property rights’ in fisheries are defined as IQs, ITQs, or individual spatial management (territorial user rights for fishing). Similar to Costello et al. (2008), the conclusion drawn is that these systems decrease the risk of stock collapse. However, Isaksen and Richter (2019), while distinguishing between tradable and non-tradable property rights regimes do not distinguish between systems of IQs and ITQs and their individual allocation (I), their market-based aspect (T), or their output-based aspect (Q). The control group contained unmanaged regimes, effort regimes as well as fisheries with fleet level quota, while the “property rights group” also contained spatial property rights (territorial use rights for fishing, TURFS).

Testing ITQs specifically, neither the before-after control-impact approach used in Essington (2010) nor the time-series analysis used in Essington et al. (2012) found significant impacts on target species status (control stocks in Essington (2010) were not all under quota management). However, Essington (2010) did find reduced variation in fisheries status indicators, possibly linked to increased harvesting security under IQ/ITQ. Chu (2009) found that results differed among systems: in some cases, stock biomass recovered after ITQ introduction, in other cases biomass was stable or continued to decline. Biomass, however, can be impacted by historical overexploitation in such a way that restoration is complicated by population dynamics and as such insensitive to management intervention (at least one can expect significant delays between intervention and result (Duarte et al., 2020)).

Conversely, empirical studies that have compared IQ to Q systems find no significant difference between the effect the two systems have on average stock health (Melnychuk et al., 2012), although the authors did find Q systems are associated with lower fishing mortality than effort-based management and that IQ systems are associated with a lower occurrence of high overfishing.

To date, Melnychuk et al. (2012) and Melnychuk et al. (2016) remain the only studies to compare stock outcomes for different aspects of fisheries management. With Paper I this thesis contributes to the empirical study of fisheries management regimes by their attributes.

1.1.4 Equity issues in fisheries (-management)

While the equity implications of fisheries management solutions are a decades old concern (McCay et al., 1995; Pálsson and Helgason, 1995), it is only recently that equity implications of fisheries management solutions have gotten wider recognition (Costello, 2019; Symes and Phillipson, 2009) and that these concerns are explicitly considered when designing new methods to allocate fisheries opportunities. Concerns regarding equity implications of fisheries management have mainly focused on the concentration of quota into the hands of few companies within individual transferable quota systems (ITQ's) (Byrne et al., 2020; Pálsson and Helgason, 1995) and on geographical concentration and the loss of quota in (remote) rural areas (Edvardsson et al., 2018). Inequality in fisheries opportunities, a reduction in jobs and viable fishing communities are of pressing concern and will likely be exacerbated by climate change (Gaines et al., 2019). Larger companies with more financial resources may have an easier time adapting to future changes (Holland et al., 2017), and climate change is likely to have more severe impacts in fisheries that suffer from overexploitation, fisheries which are often found in lower income countries where fishing is a more vital part of livelihoods (Asche et al., 2018).

Initial allocation of harvest rights is also a contentious issue in fisheries management and research. Initial allocation of individual quota is often done at a moment where the industry is not highly profitable (Gunnlaugsson et al., 2020) and quota are often freely gifted to industry participants (this practice is also called grandfathering) (Lynham, 2014). However, over time, inefficient resource users are incentivized to sell their harvesting rights and cash out (Gunnlaugsson et al., 2020), while remaining participants consolidate and become more profitable (while overcapacity is reduced), especially if stocks restore and the cost of fishing goes down (Merayo et al., 2018). In Iceland the largest companies have been present from the start of the ITQ system (Paper IV), likely because new entrants have a disadvantage compared to companies already in the industry: they will have to buy or lease quota which has increased in value and becomes a larger investment over time (Copes and Charles, 2004). And in the Icelandic coastal fisheries the fleet is “greying” due to barriers of new entry, the average age of coastal fishers being around 60 years of age, with an experience of over 30 years on average (Nielsen et al., 2017).

Another issue with the grandfathering approach, aside from the difficulty to allow for a fair way to deal with new entrants in the industry, is the problem that quota are gifted only to those that own vessels, this means that those working on the vessels or in the processing industry are not

directly compensated if the owner of the vessel decides to sell their quota (Matthiasson, 2012). A third issue with the grandfathering approach to individual allocation (addressed in Paper IV of this thesis) is that new entrants to the industry have to buy the quota, and that if the government decides to revoke the privilege it may be the case that these new entrants lose out, which would cause industry resistance to changes in fisheries governance (Tullock, 1975). This gap between rights that are grandfathered over time, and those that are bought by new entrants widens over time, until it is supposedly complete (a 100% of rights have been bought by new entrants), this situation is also called “Transitional Gains Trap” (Tullock, 1975). The perceived unfairness of government revocation of rights for new entrants represents a lock-in situation for individual quota systems, especially for those systems in which quota is permanently transferable. Paper IV shows that this lock-in situation may not be as major as originally thought and policy change is theoretically possible.

ITQ systems are often implemented for their perceived economic benefits, and generally, years since the introduction of ITQ’s in the 1990’s Icelandic fisheries have been increasingly profitable. However, not all segments of the fleet are in an equally good financial state, and it is in particular the smallest and largest companies that have the most healthy financial outlook (Gunnlaugsson and Saevaldsson, 2016). There has however been a downward trend in employment in Icelandic fisheries, due to consolidation of quota (in 2013 only 10 firms held about 58% of total quota (Gunnlaugsson and Saevaldsson, 2016). I.e. fishing jobs and jobs in fish processing decreased by more than a third since the introduction of quota in 1991 (Gunnlaugsson and Saevaldsson, 2016). Moreover, the majority of coastal fishing communities in Iceland can be classified as vulnerable, regarding the status of the local fishing industry based on data of the year 2014 and a social resilience assessment presented in Kokorsch and Benediktsson (2018). Remuneration in fishing jobs in Iceland is generally competitive with other sectors, however coastal fishers earn less than half of the average wages granted in the highly concentrated pelagic sector (Nielsen et al., 2017), stressing again the important differences between fleet segments. Moreover, well-performing fisheries in Iceland correlate with a more sustainable state of rural demography and socio-economics, even in remote areas, showing the importance of the fishing sector for rural development (Kokorsch and Benediktsson, 2018).

1.1.5 Individual quota in mixed fisheries

Another challenging aspect of quota systems in fisheries is their predominant focus on single species management, even though ITQs are often implemented in combination with input management (Thébaud, Innes, and Ellis, 2012) or implemented in fisheries that are inherently multi-species in nature (Squires et al., 1998). Shifting ranges of species may complicate this matter even further (Gaines et al., 2018).

Harvesters in mixed-species fisheries are often uncertain how much of each fish they will catch on a fishing trip. If individual quota (IQs or, if transferable, ITQs) are implemented in a mixed fishery fishers may face a problem: what if they run out of quota in one species before they have used up remaining quota in other species (Squires et al., 1998). Discarding the excess catch in the unwanted species is one possible response, but this is economically wasteful and

complicates stock assessments (Batsleer et al., 2015; Sturludottir, 2018). Discarding is now also prohibited in a growing number of fisheries (Condie et al., 2014; Mcquaw and Hilborn, 2020). Allowing trade of quota helps with this dilemma (Melnychuk et al., 2016), but this is not always possible: sometimes trade is prohibited (Copes and Charles, 2004) or quota may be scarce because of an overall shortage (Mcquaw and Hilborn, 2020). System wide quota shortages (or so called “choke species”) have been a reason for low quota uptake in other mixed fisheries (Mcquaw and Hilborn, 2020). For these species it can be for instance that the total allowable catch is set low for rebuilding purposes and therefore the total amount of quota will be limited.

Catch–quota balancing mechanisms are 1) banking (i.e., transfer of quota between periods) 2) transformation (i.e., exchange of quota in one species for quota in another species), and 3) surrender (i.e., catch in excess of quota is “sold” at a prescribed price to the fishery manager). These mechanisms give harvesters flexibility within limits to balance quota to catch after harvesting. Banking is a common mechanism across mixed fisheries and has been positively associated with stock status across fisheries (Melnychuk et al., 2016). Transformation is allowed in Iceland and has been allowed previously in certain fisheries in New Zealand and Canada, but has been abandoned there mainly because of concerns of systematic overfishing (e.g. fishers used certain species quota to actively target species for which they did not hold quota (Sanchirico et al., 2006)).



Figure 1: A mixed haul from a demersal trawler in Iceland with Atlantic cod (*Gadus morhua*), redfish (*Sebastes spp.*) and Atlantic wolffish (*Anarhichas lupus*). Picture credit: Svanhildur Egilsdóttir, Hafrannsóknastofnun

1.1.6 Challenges ahead: reshaping of current marine ecosystems

The ocean has stored more than 90% of the global heat in the climate system since the 1950’s (IPCC, 2014). Global warming and decreasing pH levels are causing a re-shaping of marine ecosystems (Cheung et al., 2011). The effect of global warming on marine species alters phenology and causes mismatches in time-events (Cheung et al., 2011; Pankhurst and Munday, 2011; Sumaila et al., 2011), such as a mismatch between a plankton bloom and a spawning

event, causing less food available for the new larvae (Asch et al., 2019). It is also well-known that warming has caused migration to for example greater depth and/or higher latitudes (Dulvy et al., 2008). Warming itself can also cause decreases in aerobic performance due to lower oxygen levels in non-mobile species (Pörtner and Knust, 2007). Shifts in primary productivity could result in altered productivity in higher trophic levels (Cheung et al., 2011). At a theoretical level combined pressures such as overfishing and warming could result in unwanted system transitions (Holling, 1996) by enhanced allee-effects (Winter et al., 2020), complicating stock recoveries.

Globally, ocean acidification is predicted to increase mortality, and impact growth and survival of many marine species (Falkenberg et al., 2018) and is likely to re-shape marine ecosystems in the near future. This re-shaping is due to not all organisms being equally likely to be affected by ocean acidification. Species traits like shells of aragonite or calciferous exoskeletons may make some species more vulnerable to future changes (McLaskey et al., 2016), other species lacking these traits may become increasingly dominant due to competitive exclusion (Olsen et al., 2018). These projected changes and the emergence of novel species assemblages and ecosystems may therefore have a yet unknown effect on productivity and affect people that directly depend on these ecosystems for their livelihoods (Marshall et al., 2017). Paper V indicates strong direct but also strong indirect consequences of warming and acidification on the Icelandic marine foodweb when projecting forward under different scenarios of combined ocean acidification and warming. This re-shaping of ecosystems will be of the benefit of some fishers and the losses of others, and policy makers should be aware of these dilemmas. Fishers with more diverse quota-portfolios may be more resilient to future ocean changes (Fulton, 2011; Holland et al., 2017).

1.2 Gaps in knowledge and research questions

From the above there are several knowledge gaps arising. First, the empirical literature often does not distinguish between the different aspects of such systems and it is not yet known if there is much benefit to ITQ implementation beyond the benefits of quota (a total allowable catch) (Bromley, 2009). There are indications that individual allocation aids preventing high levels of overfishing (Melnychuk et al., 2012), but the evidence is not yet conclusive.

Secondly, due to the difficulty of implementing individual quota in fisheries that are inherently multi-species, such fisheries often suffer from low levels of quota uptake (McQuaw and Hilborn, 2020). With such low levels of quota uptake, ITQ's in mixed fisheries may fall short of delivering their proposed economic benefits (Arnason, 1993). Balancing mechanisms have been implemented in many fisheries to aid harvesters harvesting under uncertainty to match their quota to their catches (Sanchirico et al., 2006). Iceland has one of the world's most elaborate balancing mechanisms and also relatively high quota uptake, but especially its transformation system (quota in one species can be transformed into quota of other species) could carry pervasive incentives (National economics institute, 1999). Despite Iceland propensity to pervasive behavior, persistent overfishing of any one species has not been shown

(Woods et al., 2015). It is thus of interest to know what the drivers are for the usage of balancing mechanisms in Iceland, and to know why this system seems to be working relatively well.

Thirdly, while development of quota markets over time have been studied (Newell et al., 2005; Sanchirico and Newell, 2003), detailed information on these markets has been lacking especially regarding markets of permanent shares (Holland et al., 2015). Analysis of drivers of quota trade is very limited (Innes et al., 2014). It is also not known how much of originally grandfathered quota changes hands in the quota market, and how much political leeway this would give policy makers to change original quota allocations. Finally, a re-shaping of the Icelandic marine foodweb is likely (and already occurring) due to the impacts of climate change and the differences in species responses, it is not (yet) known how major fisheries with large catch-values and many participants will be affected in terms of their catches. Given that climate change is an ongoing and continuing process, understanding species responses and interactions will be a challenging moving target.

The following overarching question guiding the research in this thesis is: how do individual transferable quota management in fisheries perform across the different domains of sustainability? The main focus lies on the Icelandic ITQ system. The socio-economic dimension is mainly addressed by investigating equity impacts in terms fair access to fishing opportunities, while acknowledging that other aspects in terms of the socio-economics of fishing systems are crucial for fisheries sustainability as well. The overarching research question is addressed with the following research questions:

RQ1: What attribute of ITQ system (the I, the T or the Q) is associated with sustainable outcomes for fish populations? (Paper I)

RQ2: How have ITQ's been implemented in Iceland's multi-species demersal fishery? (Paper II)

RQ3: How has the quota market for permanent and temporary quota trades evolved over time? (Paper III and Paper IV)

RQ4: What are drivers for quota trade? (Paper III)

RQ5: How is Iceland's marine ecosystem likely to be affected by climate change and; what are the implications of climate change for the fisheries with most catch-value and participants (i.e. fishing companies) in the Icelandic ITQ system? (Paper V)

2 Methods and region of study

2.1 Case study region: Iceland

Iceland is located in the sub-arctic, but the waters south off the island are relatively warm because of the Atlantic meridional overturning circulation (AMOC), while the northern waters are characterized by polar waters off the Greenland current. This also results in highly variable conditions and ecosystem productivity. Iceland's EEZ is relatively large compared to the size of the total population, making fishing an important contributor to GDP and exports as well as to national culture (Arnason, 2007b). In Iceland a history of overexploitation and collapse of several commercially important fish stocks (mainly Atlantic herring (*Clupea harengus*), (Arnason, 1993)) has prompted a series of management interventions. Iceland is often exemplified as a case of good fisheries management mainly for the touted economic efficiency (Arnason, 2012). The local fisheries management is now also said to be one of the more ecologically sustainable in the world (Melnychuk et al., 2017).

The first non-transferable individual quota were issued in Iceland in 1976 for herring (a stock that collapsed in the 1960's). In 1984 individual quota were introduced for the demersal fishery (with an effort quota option) and in 1986 IQ's were introduced in the capelin fishery (Arnason, 1993). The current ITQ system in Iceland finds its origin in the Fisheries Management Act of 1990. With this act, quota were allocated for an undetermined duration and were freely transferable (within years as temporary leases and as permanent transactions) within concentration limits (Arnason, 1993). The ITQ system has expanded from only a few species in 1990 to 26 species 2016, divided in some cases by region or sub-stock (Byrne et al., 2020). The most commercially important demersal species have been Atlantic cod (*Gadus morhua*), haddock (*Melanogrammus aeglefinus*), saithe (*Pollachius virens*), redfish (*Sebastes spp.*) and Greenland halibut (*Reinhardtius hippoglossoides*) while the most commercially important pelagic species have been capelin (*Mallotus villosus*) and herring (Byrne et al., 2020). Blue whiting (*Micromesistius poutassou*) and Atlantic mackerel (*Scomber scombrus*) are currently becoming increasingly important pelagic fish species as both stocks have recently migrated pole-wards (Campana et al., 2020). The above species have been targeted by different vessel types, from small coastal boats to large trawlers, and in some important species (Atlantic cod and haddock) the share caught by hook and line boats has been increasing in recent years (Sigurardóttir et al., 2014). The small-boat sector has relatively more participants but has also become increasingly concentrated over time (i.e. the number of participants has become smaller over time, see Paper III and Paper V of this thesis).

A discard ban or landing obligation was implemented in Iceland alongside the quota system in 1989 (Condie et al., 2014). Individual quota in combination with a ban on discarding can complicate fishing in multi-species fisheries, where harvesters are not certain about the exact catch composition of the species-mix prior to harvesting. If quota in a species runs out before the quota in other species, fishers will have to revert to quota trade which can sometimes be difficult, especially if there are fleetwide shortages in certain species (i.e. so called "choke species") (Paper II). This is why Iceland designed an elaborate system of catch-quota balancing mechanisms in which quota shortages can be resolved by borrowing from the next year or transferring quota between species. This elaborate and unique system is the subject of Paper II.

2.2 Methods of inference

Based on the need for longitudinal assessment of fisheries management interventions, this thesis research makes use of three main types of methods: (i) statistical analysis of time-series (Papers I-II), (ii) network analysis (Paper III and V) and (iii) ecosystem modeling (Paper V). The specific methods employed in all papers and the type of data used are listed in Table 1. Paper I employed mixed effects models of time-series to investigate associations between fisheries management systems as well as a Difference in Differences analysis that mimics an experiment setting (“natural experiment”) by comparing stocks undergoing changes in management to control stocks that don’t undergo such an intervention. Paper III and Paper V use descriptive network statistics, and Paper III also uses a statistical modeling technique for network data to investigate drivers of trade-connections in a quota market in Iceland. The last section of this chapter explains the main study region, subject of Paper II-V, Iceland in relation to its ITQ system as well as the extensive possibilities in the Icelandic ITQ system to balance catch to quota after harvesting (Paper II).

Table 1: Summary of data used and methods applied in the research for this thesis

Paper	Data	Methods
I	Stock assessments: RAM legacy database (http://ramlegacy.org/), global database of stock assessments.	Impact analysis: General linear mixed effects models, Difference in Difference analysis.
	Management data: combined approach reviewing literature & legislation and expert consultation	
II	Total yearly individual catches and allowed catches per vessel and company, and conversion ratios and lease prices of all main demersal species in the Icelandic ITQ system. Vessel and company characteristics.	General linear mixed effects models, general linear model, descriptive statistics.
III	All permanent and temporary quota trades in the Icelandic ITQ system between 2004 and 2016. Detailed company level information, lease prices.	Descriptive network statistics, statistical network models
IV	All permanent quota holdings in the Icelandic ITQ system between 1991 and 2016. Estimations of permanent quota prices 2001-2009.	Descriptive statistics, spreadsheet modeling.
V	Historical assessments of fish stocks for Iceland and historical catches (Sturludottir et al., 2018).	Network analysis, dynamic ecological modeling (process-based)

2.2.1 Longitudinal data and analyses

Mixed-effect models: Mixed-effects modelling frameworks allow for the introduction of random effects for variables where the sustainability indicators are more likely to share a similar response (Verbeke and Molenberghs, 2009). For example, a response of a stock in one region

to a management system studied in Paper I will be more likely to correlate to the response of another stock in the same region. Both Paper I and Paper II also account for autocorrelation in the timeseries: e.g. the response of one fishing vessel studied in one year in Paper II will more likely correlate to the response of the same fishing vessel in another year. The fact that these observations are not independent violates a major assumption of linear regressions, and if not considered in the modeling process may give rise to type I statistical errors: rejecting a null hypothesis when there is in fact no relation (Verbeke and Molenberghs, 2009).

Difference-in-Differences analyses: Difference in Differences (DiD) analysis was used in Paper I. DiD analysis is commonly used for analyzing time series data where systems that undergo a change (i.e., treatment) are compared to systems that remain the same (i.e., control). An important assumption in this approach is that treatment stocks would have followed a similar trajectory to control fisheries if no change had occurred (Shadish et al., 2004).

2.2.2. Network analysis

Research on networks is part of the broader study of complex systems, with its origination in graph theory (Scott, 1987). Network analysis is based on a theoretical framework which studies interactions or flow of information (called links or edges) between elements of systems (nodes), represented by a matrix. Information on the nature of interactions and their importance can also be included in the study of networks. Many natural and social phenomena have been represented as networks and maybe this is not surprising as a network is an intuitive depiction of connections between entities, may they be animals (including humans) and their social interactions, countries trading with one another, stations in a transportation network, computers connected through the internet etc. Intriguingly, despite this enormous diversity of networks observed in natural and human-made systems, diverse networks can share several universal structural characteristics (Barzel and Barabási, 2013), and many network own properties (endogenous properties of the network, in network terminology) can be found to explain a large share of the connections in a network. Sometimes shared network-processes underlie these observed similarities between networks. An example of such well-known real-world processes can be explained in terms of networks is the “the rich get richer” process. This is a cliché, but it’s also a true representation of real-world networks. In network terms the “rich get richer” is a dynamic that is called “preferential attachment”, those with more network connections are more likely to form even more new connections by a statistical likelihood that is greater than chance (Barabasi, 2009) (for instance a research article with many citations is going to attract proportionately more citations for several reasons including the quality/novelty/pioneering aspects of the work itself). Such a dynamic would cause extremely skewed distributions of connections in social networks, something that is frequently observed in real-world networks (Barabasi, 2009).

Paper III studied if such a skewed distribution of trades was present in a quota trade network and if this meant that more connected traders would more easily form new trade-connections in a quota market. This had been indicated previously for quota markets in e.g. Tasmania (Innes et al., 2014). For this purpose, the quota market for Atlantic cod in Iceland was modeled as a

weighted directed network. Networks were created for each year in the dataset so that timeseries of network statistics could be computed. The number of incoming and outgoing trades per quota holder (in- and outdegree respectively) and fitted power-law, exponential and log-normal relationships to the in- and out-degree distributions (Clauset et al., 2009) were measured. Power-law distributions are extremely skewed and could be a result of network endogenous processes such as the “preferential attachment” process that is described above. Some support was found for a scalable distribution (power-law) in the cod quota market which could have been indicative of network processes such as popularity and the formation of hubs, which in turn could indicate a market that is not easily accessed by all individuals (Innes et al., 2014; van Putten et al., 2011).

In Paper V network metrics were used to analyze which species or functional group in the Icelandic marine ecosystem could be considered as ecological key-stone species. Key stone species are defined as a species or functional group through which many other species (or functional groups) are indirectly connected or species that are a main prey species for many other species/functional groups. The Google page rank indicator gives a high score to species that support many other (important) species directly and indirectly through predator-prey interactions (Allesina and Pascual, 2009). Indegree centrality, which quantifies the relative importance of a species by how many predators depend on that species as a diet species (Chen et al., 2008), was also calculated. A centrality measure was also calculated, betweenness that, similar to the Google page rank, indicates how many species are directly and indirectly connected through that species or functional group (McDonald-Madden et al., 2016).

Statistical modeling of networks allows for the investigation of drivers of connectivity. Exponential random graph modeling is a method that is used in this thesis to help quantify the effect of characteristics of quota holders on trade formation (i.e., characteristics of ‘nodes’ on ‘edges’ in network terminology) (Fischer and Jasny, 2017). In the ERG model in Paper III the number of trades between two quota holders can be viewed as a value in the response variable of a regression model, and the predictor variables were internal network characteristics of the quota holders (e.g., reciprocity) as well as external characteristics (e.g., vessel size). The approach is divided into two steps. In the first step a well-fitting statistical model is searched for the empirical network, the second step simulates models generated in the first step and tests whether the observed network-structure is well-replicated (Pol, 2017). The outcome of the ERG models are the log-odds for a set of parameters, similar to those of a logistic regression. The value of the log-odds characterizes the strength and the direction (\pm) of the influence of a parameter on the likelihood of link-formation, i.e. zero (no link) or one (link is present).

2.2.3 Ecological food-web modeling

For Paper V the Atlantis ecosystem model was used for the Icelandic trophic relationships. The Atlantis model is a full ecosystem model built on an oceanographic model and includes all major marine functional groups and species in the Icelandic exclusive economic zone (EEZ). The oceanographic part of the model contains 51 three-dimensional spatial boxes that exchange

water flows, salinity levels, and temperature. The oceanographic data were adapted from a hydrodynamic model developed by (Logemann et al., 2013). The ecological model contains 52 functional groups (see Figure 2), where vertebrates are generally modeled with a higher level of detail than invertebrates. Vertebrate groups have age structure and recruitment is modeled using the Beverton-Holt function while invertebrates and plankton groups are simple biomass pools. Sturludottir et al. (2018) provide a detailed description of the Icelandic Atlantis model. The Atlantis model was used to test the effects of several different scenarios, namely ocean acidification and warming climate change. First, a literature review was performed to determine the parameter ranges for impacts of ocean acidification and the temperature niches of species in the Icelandic marine ecosystem, then the last ten years of the model run (2002-2012) were repeated five times to create a baseline scenario until the year 2100, the model was then run for several scenarios of warming or combined warming and ocean acidification.

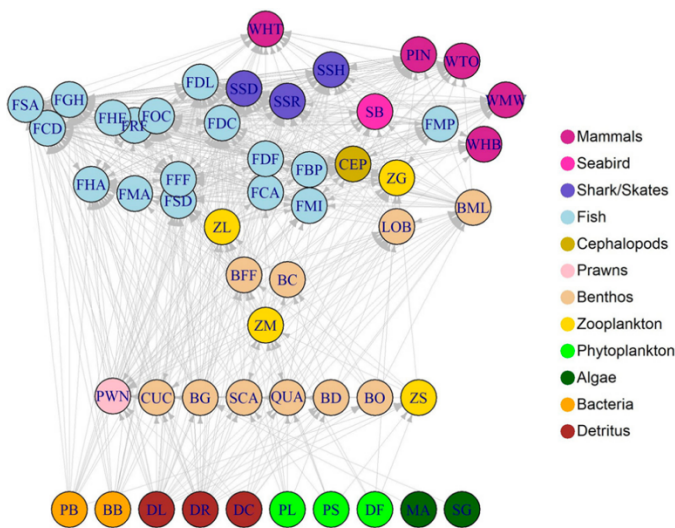


Figure 2: Food web connections between modeled functional groups in the Icelandic modeled functional groups, image from (Sturludottir, 2018).

3 Paper summaries and insights from the thesis papers

3.1 Paper I: Which attributes of fishing opportunities are associated with sustainable fishing?

Maartje Oostdijk & Griffin Carpenter, Which attributes of fishing opportunities are associated with sustainable fishing? *In review Environmental Research Letters*

Individual transferable quotas (ITQs) are an increasingly used system to allocate fishing opportunities. While there have been several prominent studies that link these systems to sustainable outcomes (occurrence of overfishing (fishing mortality) and overfished stocks (biomass), there is little information on which specific attributes of this system (individual, transferable, or quota) of this system, or indeed any other system, that leads to sustainable outcomes. To analyze the impact of different allocation systems on target species, systems of allocating fishing opportunities were classified for 423 fish stocks from 1990-2018, producing the largest global database of its kind. A decision tree was designed to enable classification of fisheries management regimes by their attributes (Figure 3).

Research questions:

Overarching research question (RQ1): What attribute of ITQ system (the I, the T or the Q) is associated with sustainable outcomes for fish populations?

This research question is divided into the following sub questions:

- What fisheries management systems are associated with healthy marine stock status?
- What attribute of ITQ systems (the I/the T/or the Q) is associated with healthy marine stock status?
- How does duration of harvesting rights affect stock status?

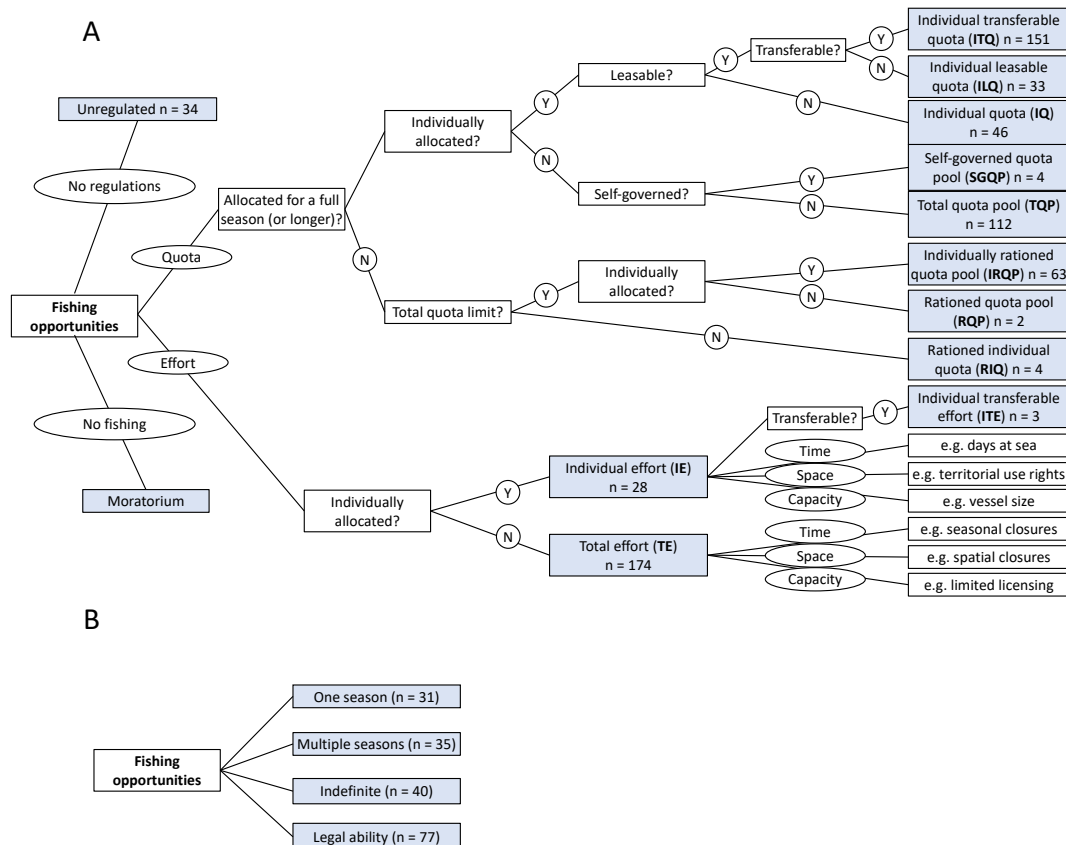


Figure 3: A) Decision tree containing a classification of fisheries opportunities by their attributes. The blue terms are the thirteen exhaustive classifications used in this study for all fisheries management systems. Generally, the blue terms are the final classifications with all assigned attributes, no distinction between spatial, temporal and capacity limitations were made for the effort classifications due to the difficulty assigning percentages to the respective categories. B) Decision tree containing a classification of fisheries opportunities duration, the blue terms are the four exhaustive classifications used for duration. The number of each unique stock and classification combination is noted in each of the final classifications (Figure corresponds to Figure 1 in **Paper I**).

Methods

Stocks were classified as experiencing *overfishing* if fishing mortality was higher than 1.1 times maximum sustainable yield, stocks were classified as *overfished* if biomass was lower than 0.8 times biomass at maximum sustainable yield. Mixed effects models were used to predict the incidence of overfishing by the different management regimes and attributes (I, T, Q). A difference in differences strategy was employed for stocks where attributes changed during the time-period of the study (1990-2018), see 2.1.

Main results

Systems that use quota limits and allocate limits individually were associated with reduced overfishing (Figure 4). Little significant benefit of transferability nor the permanent allocation of fishing rights as opposed to a fixed single season was found (Figure 4). Difference in difference analysis revealed that stocks that change from effort management to quota experienced less overfishing, as well as stocks adding attributes T, L and I. The analysis of 19

treatment fisheries that transition from general quota management to individual (transferable) quota management showed no significant changes for any of the indicators compared to control fisheries. There is thus some evidence that quota and individual (transferable) quota reduce the probability of overfishing, but these results are less reflected in biomass and causal evidence is weak and not confirmed when pairing the same species in the same or a nearby geographical region with different management approaches (i.e. quota versus I(T)Q). Little support was found for the hypothesis that tradability and is associated with more sustainable outcomes and no support was found for durability of harvesting rights.

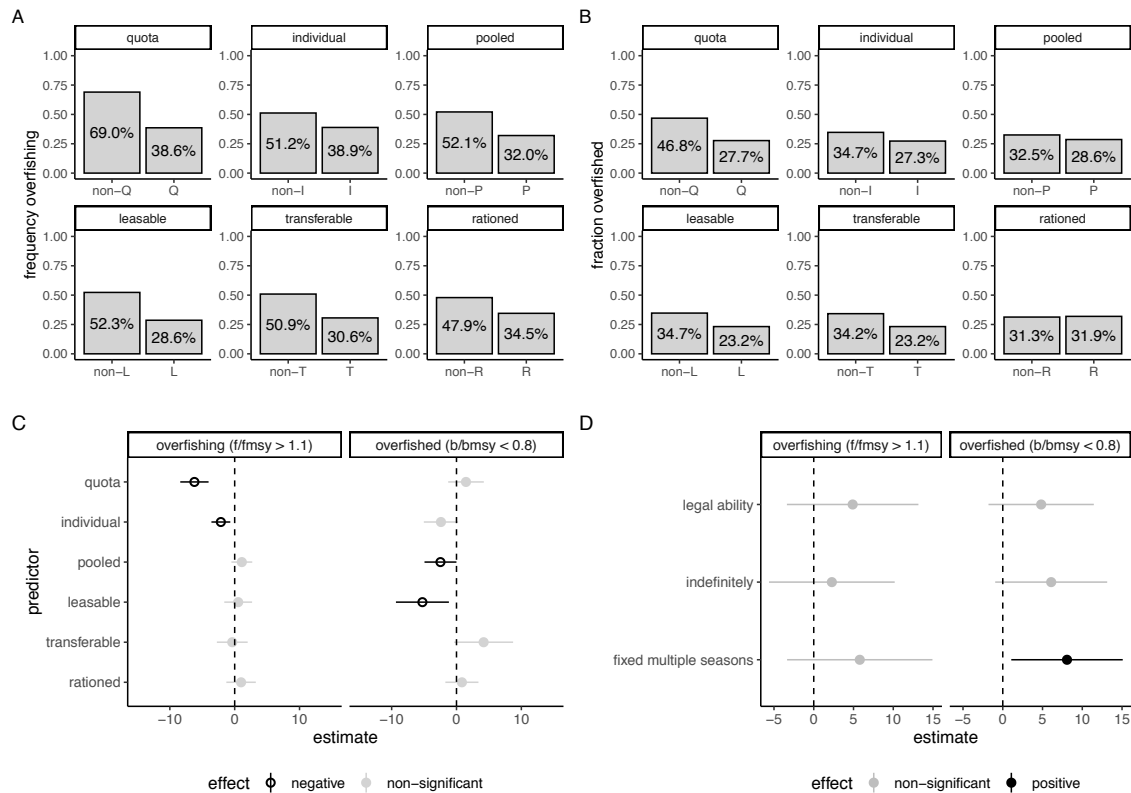


Figure 4: A) frequency of overfishing ($F/F_{msy} > 1.1$) and B) frequency of overfished observations ($B/B_{msy} < 0.8$) for the attributes I, T, Q, P, L and R. Each observation is a stock-year combination. C) Mixed-effects results for the attributes I, T, and Q. Negative (black, open circles) effects indicate a reduced probability of overfishing for I and Q (overfishing: 343 stocks with 6803 observations; overfished: 299 stocks with 6875 observations) and a reduced probability of overfished for pooled and leasable. D) Effects for the duration of fishing opportunities compared to a single season. The positive (black, closed circle) value indicates an increased probability of the overfished state for fixed multiple seasons (Figure corresponds to Figure 3 in Paper I).

3.2 Paper II: Catch–quota matching allowances balance economic and ecological targets in a fishery managed by individual transferable quota

Maartje Oostdijk, Conor Byrne, Gunnar Stefansson, Maria J. Santos, Pamela J. Woods, Catch–quota matching allowances balance economic and ecological targets in a fishery managed by individual transferable quota: *Proceedings of the National Academy of Sciences* <https://doi.org/10.1073/pnas.2008001117>².

Many economic benefits of IQ systems could be compromised in mixed fisheries owing to fisher’s uncertainties related to the catch mix. A fisher could face problems when s/he reaches quota of one species but not the others, and incentives for discarding these fish become strong. Mixed fisheries managed by quota have solved this issue in different ways by implementing catch-quota balancing mechanisms (Sanchirico et al., 2006). While previous studies investigated catch-quota balancing behaviour at the level of the whole fleet (Sanchirico et al., 2006; Woods et al., 2015), individual vessel behaviour was investigated in the most elaborate catch-quota balancing system in the world - Iceland. Catch-quota balancing allowances can theoretically lead to overfishing if total allowable catches (TACs) are consistently exceeded. In the Icelandic case several methods of balancing catch to quota after harvesting exist: 1) transferring quota between vessels (i.e. leasing quota), 2) transferring quota between years (i.e. quota banking) 3) transferring quota between species (i.e. species transformations, 4) “grace take” (i.e. a small percentage of the catch can be sold directly to the directorate of fisheries for a fraction of its value). Banking and species transformations in the Icelandic system are further detailed in Figure 5. This paper focussed on the species transformation system because this system carries a risk of persistent TAC overages and is therefore the most ecologically risky. Conversion ratios used in species transformation systems could contribute to this risk because, if these ratios do not reflect the profitability of the different species, they could open up avenues to profit from differences in the exchange rates (i.e. arbitrage) which could exacerbate the ecological risk to certain species especially if fishers could target such species.

¹ **Author contributions:** MO collected the data jointly with CB, MO lead the research design and me and CB jointly carried out the analysis. PW and MJS also designed the research, GS aided with the methodology. MO wrote the first draft of the paper all co-authors helped writing.

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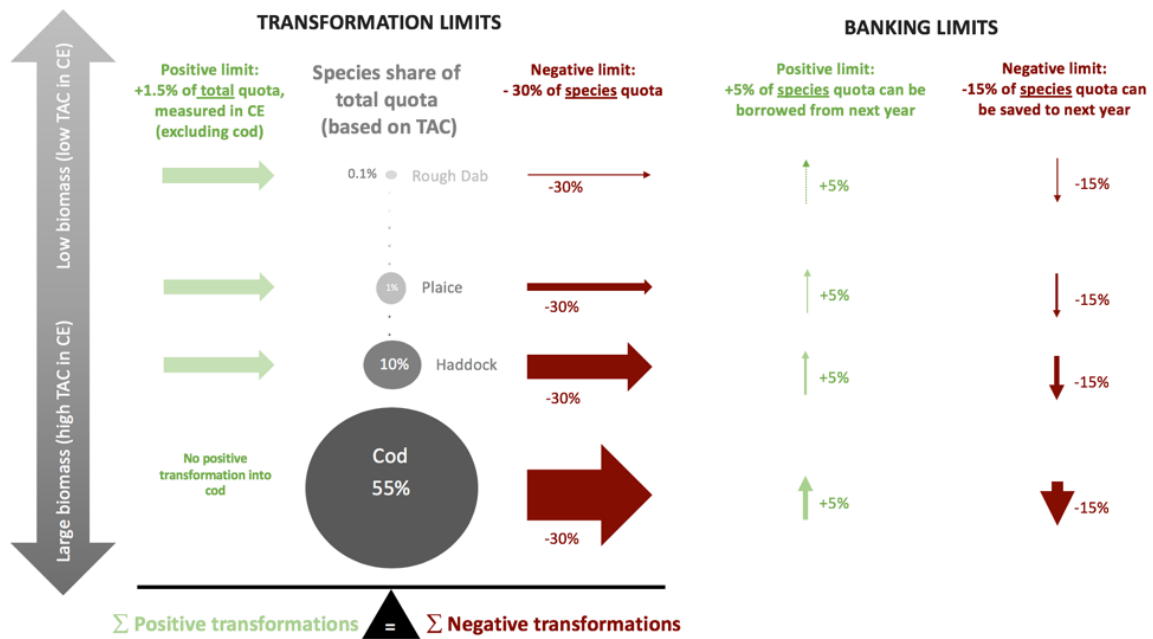


Figure 5: Transformation and banking limits in the Iceland ITQ system. Positive (light green)/negative (red) limits restrict increases (decreases) in the permitted catch of the relevant species in the current period. Total cod equivalents (CEs) (units converting species quota to the same unit based on last year's price relative to cod) are summed across species before applying the 1.5% positive transformation and 5% total transformation limits; all others are applied as percentages of the originating species quota. The arrows are scaled relative to the total percentage of CE of the relevant species. The left-hand side of the figure shows how transformation limits are designed asymmetrically; the positive limits are based on a vessel's total CE quota, aggregated across species, which is the same for all species, while the negative limits are based on quota in the relevant species. This means that the overall fleetwide limit on positive transformations can potentially be several times TAC for low biomass species (it is 75 times TAC for common dab) but will be small relative to TAC for high biomass species and may be further constrained by the limited "supply" of negative transformations from low biomass species (Figure corresponds to Figure 2 in **Paper II**).

Research questions:

Overarching research questions (RQ2): How have ITQ's been implemented in Iceland's multi-species demersal fishery?

This research question is divided into the following sub questions:

- How does fleet -level catch-quota balancing differ between the different species and years in the Icelandic system?
- What incentives are created by the Icelandic balancing system?
- What differences exist between companies and fleet segments in their balancing behavior?

Methods

An index was designed that describes the similarity of balancing across individual vessels in the Icelandic fleet (the directionality index). The overall directionality of balancing adjustments, was defined as Ds for species s and calculated as follows:

$$D_S = \frac{\sum_i^I P_{si} - \sum_i^I N_{si}}{\sum_i^I P_{si} + \sum_i^I N_{si}} \quad (1)$$

where P_s is the positive quota adjustment for vessel i (0 when negative), and N_s is the negative quota adjustment for vessel i (0 when positive). This index takes values between -1 and +1; the former implies that transformation or banking are purely negative, the latter that transformation or banking are purely positive, while 0 indicates equal volumes of positive and negative flows. The directionality index was calculated separately for quota transformed and quota banked at the end of each year. This index was then predicted by several potential drivers of catch-quota balancing in the Icelandic fleet (opportunities for arbitrage, and indicator of a general shortage (“choke indicator”) and an indicator of the ability to target individual species. Moreover, individual level models were employed to predict catch, in which also the impact of vessel- and company characteristics on the catch-quota balancing behavior was studied.

Main results

Evidence was found that balancing behaviour was frequently similar across the fleet. Transformations could be predicted by an indicator of a potential for arbitrage caused by differences in conversion ratios used for transformation and lease prices and the total allowable catch (Table 2).

Table 2: Directionality of transformations model with fractional logit estimates of the contribution of each of the predictor variables, standard errors, z values and probabilities. It can be observed that arbitrage potential and TAC are the most important predictors of transformation directionality with a positive effect. Asterisks represent significance levels, with *** at the <0.01 level (Table corresponds to Table 1 in **Paper II**).

Predictor	Estimate	standard error	z value	Pr(> z)
Arbitrage potential	1.47	0.21	7.14	< 0.001***
Choke indicator (dummy variable)	-0.17	0.44	0.38	0.70
Total allowable catch (TAC)	0.99	0.29	3.41	< 0.001***
Targeting indicator	-0.09	0.14	-0.63	0.53
Targeting indicator * arbitrage potential	-0.21	0.15	-1.35	0.18
Cox & Snell's $R^2 = 0.27$				
Nagelkerke's $R^2 = 0.58$				

Larger companies contribute more to differences between catch and quota. Despite these findings, TAC overages tended to be modest and especially so in recent years (< 30%). The only species with substantial TAC overages in recent years was Lemon sole. Key reasons for the limited TAC overages appear to be a recent (2011-2012) tightening of vessel transformation limits and the central role of Atlantic cod. Cod is the main target species in the Icelandic demersal fishery but cannot be persistently overfished due to a specific prohibition on positive transformations into the species. These results show how the tailored design of the Icelandic catch-quota balancing system has helped in balancing economic and ecological goals in fisheries management. The usage of the system is though not similar across the fleet and future

research may be needed to indicate if larger companies can make more use of arbitrage opportunities due to their ability to hire staff to deal with optimising catch-quota balancing. We also suggested a few policy changes, on order to make the design of the catch-quota balancing system less ecologically risky, which include the prioritisation of between year transfers over species transformations as between year transfers cannot lead to systematic TAC overages or underages, and a limit for transformation into species (currently the amount of transformations are only constrained by amounts out of species, and the limit into species is based on aggregate quota holdings, which is risky for small biomass species).

3.3 Paper III: Structure and evolution of cod quota markets networks in Iceland over times of financial volatility

Maartje Oostdijk, Maria J. Santos, Sveinn Agnarsson, Pamela J. Woods (2019) Structure and evolution of cod quota markets networks in Iceland over times of financial volatility *Ecological Economics*, 159, 279-290, <https://doi.org/10.1016/j.ecolecon.2019.01.035>³

This paper explores changes in the quota markets over a period of time that included the financial crash (2004-2016). While quota markets are the main instrument through which individual transferable quota systems are supposed to reach their efficiency, there are few studies on their functioning and empirical information remains scarce. It is also through the quota market that consolidation occurs in ITQ systems and the investigation of drivers for trade is important to distinguish between e.g. vessel-level consolidation or company level consolidation. Namely, consolidation can occur on two levels: 1) accumulation of quota onto fewer vessels and accumulation of quota into the hands of fewer holders (often also due to mergers of companies). The exit of the less profitable harvesters from the fishery can reduce over-capacity, which is a necessary step towards economic sustainability in fisheries (Branch, 2009). However, quota consolidation can also occur in a fleet where overcapacity is no longer a problem, caused by other economic benefits such as profitability and the resilience gained by owning a diversity of fishing quota (Holland et al., 2017).

³ **Author contributions:** MO conceived of the research idea, lead the data collection and carried out the analysis, MO wrote the article with help of MJS, SA and PJW.

Research questions:

Overarching research questions (RQ3 and RQ4): How does the quota market for permanent and temporary quota trades evolve over time? And What are drivers for quota trade?

These research questions are divided into the following sub questions:

- How did financial volatility shape the largest quota trade networks in Iceland?
- How does the concentration of quota in the largest quota market in Iceland develop over time?
- How do network characteristics of companies impact the prices obtained in the quota markets?
- What are main drivers for quota trade for cod quota share and lease markets in Iceland?

Methods

Changes in quantity and amount of quota trade over time in the Icelandic cod quota markets for permanent and lease quota were described, for the small boat segment of the market and the regular segment. Changes in network indicators over time were described, and network models were employed (exponential random graph models) to study drivers of quota trade.

Main results

While prior to the crash the market for permanent quota displayed a large amount of activity, the activity became sparser and never fully recovered after the crash. The lease quota market displayed a dip around 2009-2010 but regained its activity. Some support for power law fits was found for 9 out of the 24 distributions of quota trades (per vessel) in quota share markets. Especially the smaller boat market consolidated rapidly between 2004 and 2008 which is also reflected in a rapid increase in network connectivity (the networks are displayed in Figure 6).

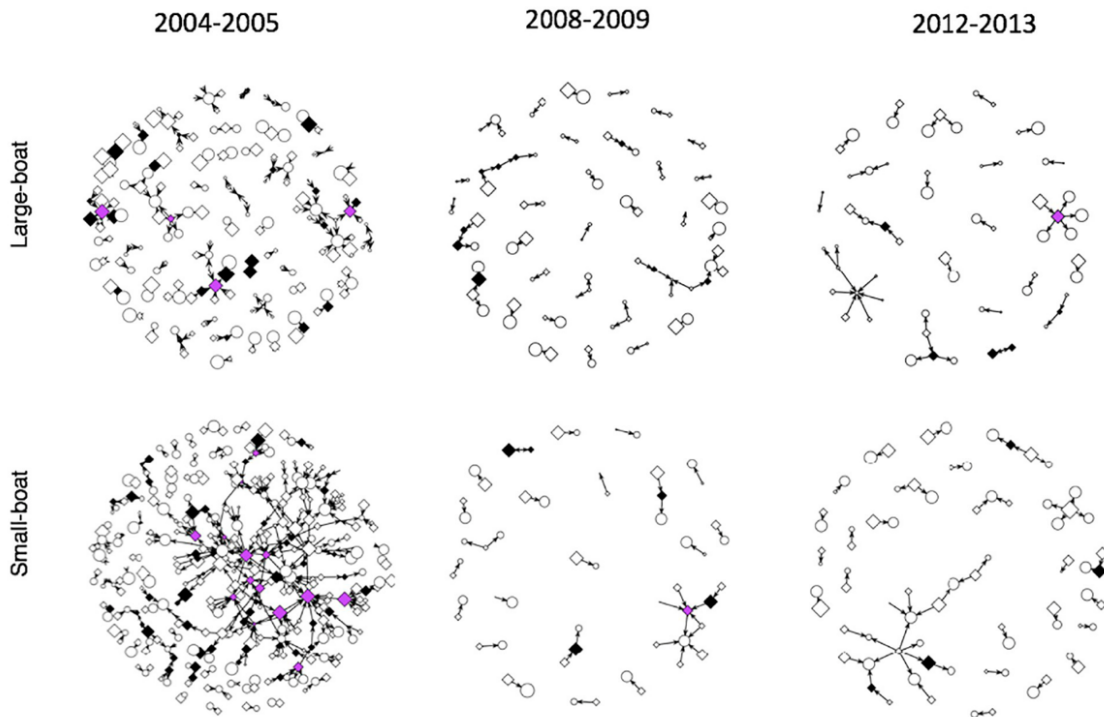


Figure 6. Owner networks of the small-boat quota share market for permanent shares for fishing years 2004–2005, 2008–2009 and 2012–2013. Upper panels show the large-boat networks and lower panels show the small-boat networks. Nodes are scaled to the amount of quota owned. Open diamond nodes have outgoing trades, open circle nodes have incoming trades, black colored nodes have both incoming and outgoing trades and purple nodes have both incoming and outgoing trades as well as five or more total trades in the same fishing year (“broker nodes”) (Figure corresponds to Figure 4 in **Paper III**).

Drivers associated with consolidation (gross tonnage of vessels or allocated quota to the vessel owner) were more important factors shaping trade relations than the need to match quota to catches in the markets for permanent cod quota. There were slight indications that network position mattered for the prices obtained in the lease-markets. Broker nodes got significantly higher prices for quota that they leased out in both the small- and large-boat as well as central nodes as measured by betweenness centrality in the large-boat network, although surprisingly the opposite result was found for betweenness centrality in the small- boat network.

3.4 Paper IV: Trade and potential measures of the Transitional Gains Trap - A quantitative analysis of the Icelandic ITQ system

Conor J. Byrne, Maartje Oostdijk, Sveinn Agnarsson, Brynhidur Davidsdóttir, Quota Trade and potential measures of the Transitional Gains Trap - A quantitative analysis of the Icelandic ITQ system (*in review Marine Resource Economics*)⁴

The Transitional Gains Trap has been cited as an argument against reallocation of grandfathered transferable fishing quota; the initial recipients gradually sell until all quota have been purchased at which point industry profitability returns to normal and it becomes politically impossible to reallocate the quota. Despite this, there do not appear to be any studies examining the extent of the trap in ITQ fisheries or more generally. This paper proposes potential quantitative measures of the trap which reflect not only the total volume of quota purchased but also the extent to which these purchases may be offset by quota sales or rent from retained grandfathered quota.

Research questions:

Overarching research questions (RQ3): How does the quota market for permanent and temporary quota trades evolve over time?

This research questions was divided into the following sub questions:

- How can TGT be empirically calculated?
- What is the extend of TGT in the Icelandic quota system?

Methods

A quantitative measure of TGT in the Icelandic ITQ system was designed. The proposed measure of TGT reflects potential loss faced by individual harvesters due to revocation of purchased quota. However, it also considers the offsetting effect on these harvesters of accumulated profits due to the initial grandfathered allocations as well as investment in additional or new quota. This measure of TGT was then calculated for Iceland's ITQ system under a range of assumptions on the gains of (grandfathered and purchased) quota.

⁴**Author contributions:** CB conceived of the idea and designed the study. MO collected the data and carried out part of the analysis. CB wrote the first draft and MO, SA and BD helped with the writing.

Main results

Figure 7 shows the (stylized) effect of considering rent accumulation when calculating TGT. In the figure it can be seen that TGT can in fact decline over time if part of a company's quota was grandfathered; accumulation of resource rent from the grandfathered quota can gradually offset the cumulative net investment cost of purchased quota.

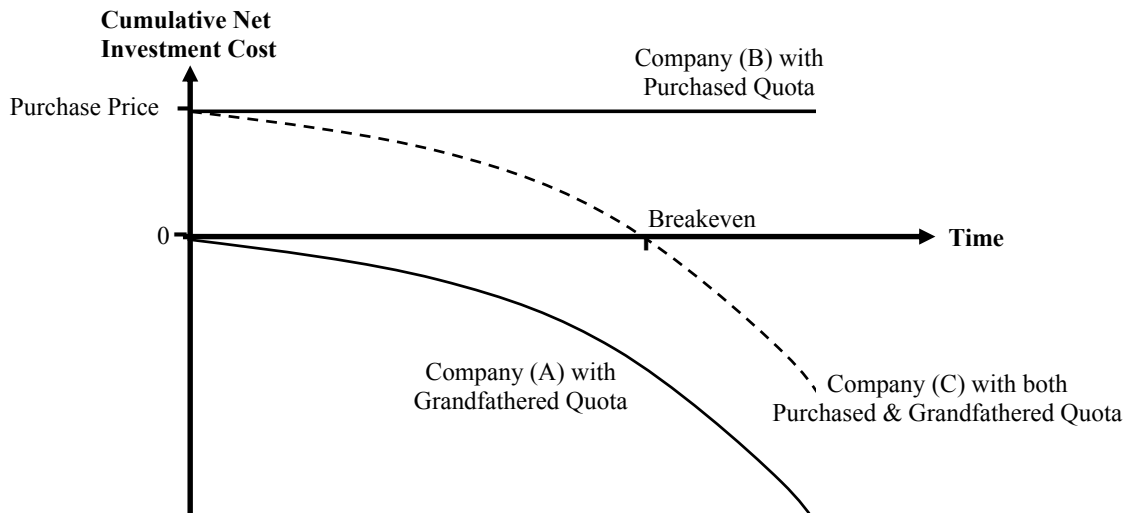


Figure 7: Effect of rent accumulation on cumulative net investment cost of quota holders (Figure corresponds to Figure 4 in *Paper IV*).

The net investment of company (A) which received grandfathered quota is initially zero and becomes increasingly negative (i.e. surplus) as resource rent is accumulated and compounds in value. The net investment of company (B) with purchased quota is initially the purchase price of the quota. Calculations for Iceland's ITQ system, from their introduction in 1991 until 2016, suggest that the extent of the trap varies considerably by species but generally remained below 60%. Finally, it is shown that the trap can decrease over time. This potential decrease of the trap is especially apparent in a scenario where the government announces a revocation in the future. This paper shows that, under a broad base of assumptions (regarding the rents generated from owned and purchased quota), there is political scope for reform of quota systems where quota have been initially freely gifted.

3.5 Paper V: Risk assessment for key socio-economic and ecological species in a sub-arctic marine ecosystem under combined ocean acidification and warming

Maartje Oostdijk, Erla Sturludóttir, Maria J. Santos, Risk assessment for key socio-economic and ecological species in a sub-arctic marine ecosystem under combined ocean acidification and warming *Manuscript*⁵

The Arctic as a region may be particularly vulnerable for consequences of ocean acidification and is projected to be largely corrosive to aragonite by the end of the century. Moreover, warming in the Arctic happens at a faster pace than the global average. As different species or even individuals have different vulnerabilities to lower pH levels and have different responses to warming re-shaping of ecosystems is likely. Recent lab experiments also indicate that the larval stage of several fish species may be very vulnerable for ocean acidification (Frommel et al., 2011; Stiasny et al., 2016), and recent empirical literature already shows decreased average productivity linked to warming (Free et al., 2019).

Research questions:

Overarching research questions (RQ5): How is Iceland's marine ecosystem likely to be affected by climate change and; what are the implications of climate change for the fisheries with most participants (i.e. fishing companies)?

This research question is divided into the following sub questions:

- Which species in the Icelandic EEZ are likely to be impacted by increasing ocean temperatures?
- Which species in the Icelandic EEZ are likely to be impacted by lowering pH levels?
- How is the marine foodweb around Iceland impacted under combined warming and acidification scenarios?
- How does this impact the fisheries with the most participants in Iceland?

⁵ **Author contributions:** MO conceived of the research idea, carried out literature research and ran modeling scenarios with the help of ES. MO wrote the first draft of the manuscript and ES and MJS helped with writing.

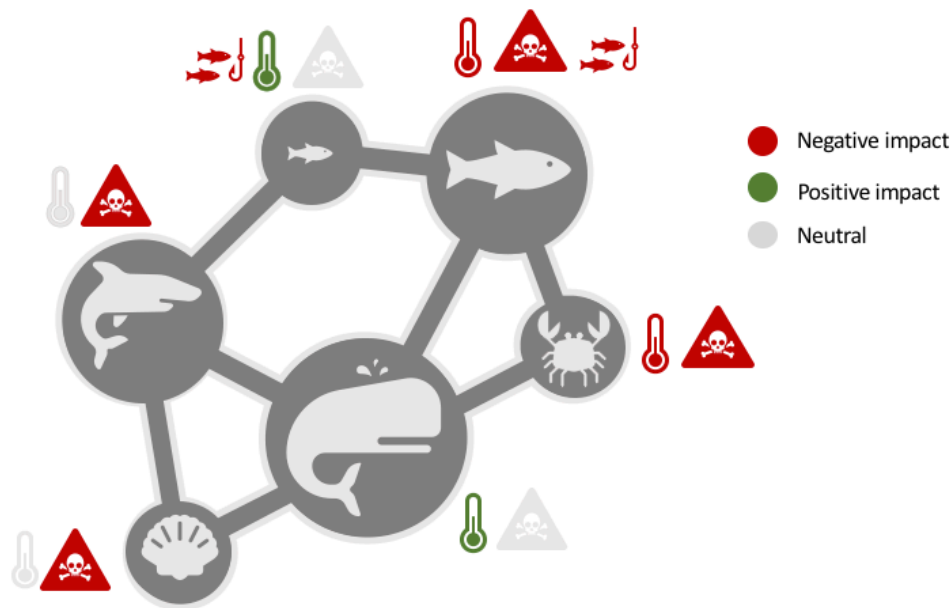


Figure 8: A hypothetical foodweb with different effects of ocean acidification or warming on species. Species are directly exposed to pressures of ocean acidification, warming and fishing but not all species are equally affected by the same drivers and species will likely experience indirect effects due to impacts on diet species or changes in competition and predation (Figure corresponds to Figure 1 in **Paper V**).

Methods

Species of interest were selected based on the number of participants in the fishery and catch-value or their ecological function in the foodweb. Network indicators were used to select species with key roles in the foodweb (McDonald-Madden et al., 2016), such keystone species have a disproportionately large impact on the functioning of the foodweb (Valls et al., 2015). The network indicators chosen were shown to have an important indicative value as to how the food-web would be impacted if such a species were to be removed (McDonald-Madden et al., 2016). Then a literature review was performed on the possible impacts of ocean acidification on the species/functional groups and searched the literature for temperature niches or optima. Then the Atlantis end-to-end ecosystem model (Fulton et al., 2011; Sturludottir et al., 2018) was parametrized based on the literature review and projected forward under twelve scenarios of combined warming and acidification. Two scenarios were developed for ocean acidification, namely a moderate and a severe acidification scenario. For the moderate scenario we adjusted (i) growth of benthic groups by -20%, and (ii) reduced cod recruitment by -20%. For the severe acidification scenario, we adjusted (i) growth of benthic groups by -30%, and (ii) reduced cod recruitment by -30%. These are relatively small reductions, as studies have suggested that recruitment may be reduced by 76% to 92% (Stiasny et al., 2016). If information on temperature optima or niches were found, these were used to parameterize the temperature optima in Atlantis as was done in (Griffith et al., 2012).

Main Results

Divergent species responses were found to warming and acidification levels; certain species benefited while others decreased in biomass under warming and acidification scenarios. With conservative harvest rates, Atlantic cod (the largest catch-value species) population size is

projected to remain stable under even the harshest ocean acidification and warming scenario modeled, while without ocean acidification nor warming the population size is projected to slightly increase and stabilize around 2040. Figure 9 displays the biomass (left-hand panels) and catches (right-hand panels) of the most important sectors in the Icelandic ITQ system (>5% of catch-value) under the different scenarios of combined ocean acidification and warming, over time. Although the biomass of the Atlantic cod stock was surely impacted by the forced reduction in recruitment, biomass levels seemed to stabilize in all scenarios at levels higher than present, while in the baseline scenario biomass increased further (Figure 9). The model did not forecast a collapse or a strong decrease in biomass compared to current levels. Biomass of Atlantic cod has been increasing steadily since the reduction of harvest rates in the early 2000's and if this harvest rate is kept constant the Atlantis model predicts a rather stable biomass (Figure 9) and catches (Figure 9). However reduced biomass of haddock was predicted compared to the present, probably because of predation pressure of Atlantic cod (Figure 9). In scenarios where warming and acidification resulted in reductions in biomass of Atlantic cod other species in the ecosystem increased, by reduced competition and predation compared to a baseline scenario with no global change. These results highlight the interdependencies of multiple global change drivers and their cascading effects on trophic organization, as well as the impacts on important species within the Icelandic ITQ system. The different responses of the Icelandic species to global change are an important consideration when building climate-resilient fishing operations. However, there are high levels of uncertainty regarding individual species responses to ocean acidification and warming. Especially given that there is not much experimental research on Icelandic species specifically.

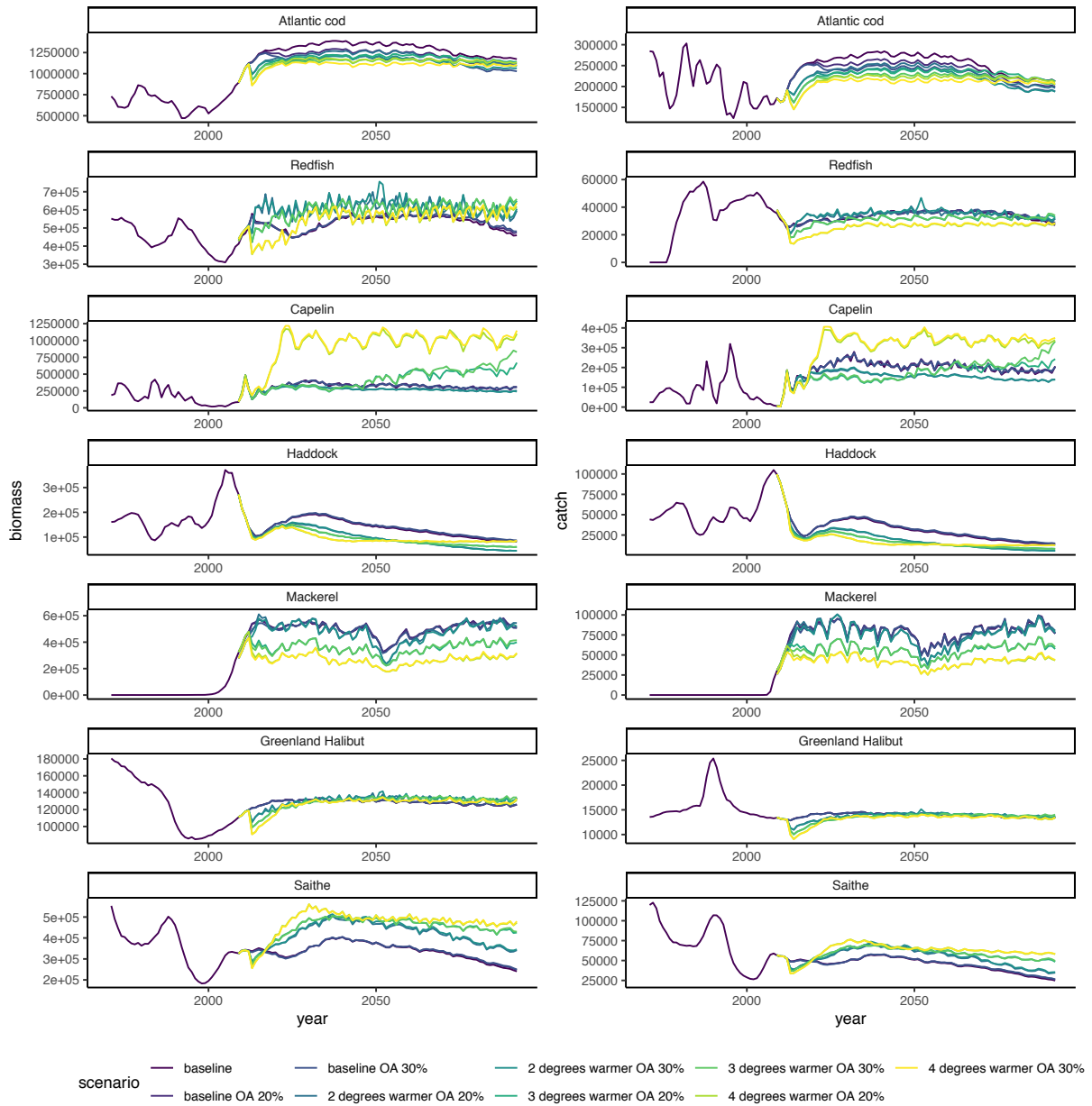


Figure 9: Biomass (left panels) and catches (right panels) of the main Icelandic fisheries (5% or more of Icelandic catch-value, organized from highest to lower catch values) in Atlantis under the baseline scenario and eight different global change scenarios. Note that the y-axes are on very different scales. (Figure corresponds to Figure 7 in Paper V).

4 Discussion

4.1 Scientific insights – individual papers

This thesis contributed to the field of fisheries science with the study of ecological impacts on target stocks of fisheries management regimes and the study of the evolution of socio-economic indicators over time in an ITQ regime. Moreover, the thesis contributed to the literature by projecting forward under climate change scenarios and studying impacts on the most important stocks in an ITQ regime. Table 3 lists the gaps in knowledge and the contribution to the field of each of the individual papers.

The effectiveness of the different fisheries management regimes for conserving and re-building marine stocks while improving socio-economic conditions for fishers remains unclear. First, a comparative analysis of fisheries management systems (RQ1, Paper 1, see Table 3) aimed to give more clarity on what aspects of fisheries governance systems provide positive outcomes on fish stocks.

RQ1: What attribute of ITQ system (the I, the T or the Q) is associated with sustainable outcomes for fish populations? The strongest effect was found for **quota in reducing the probability of overfishing**, also **individual allocation was found to reduce the probability of overfishing**. No support was found for longer allocation and little support was found for transferability of harvesting rights in preventing overfishing. This research contributes to which aspects of a quota system are relevant for management of fish and invertebrate populations. In the past individual transferable quota regimes have been only compared to control fisheries under different (undifferentiated) management styles, some of which are open access regimes (Costello et al., 2008). This simplistic comparison design made it impossible to differentiate the effects of stewardship claims associated to individual quota and simply the effect of setting of a biologically reasonable total allowable catch on target stocks (Bromley, 2009). Fisheries managers can opt for different designs of so-called rights-based management, possibly also to avoid some of the social costs associated to ITQs as has happened in many European fisheries (Gibbs, 2009). Allocating harvesting rights individually but for a limited duration, for instance via auction seems to be a viable policy option according to the outcomes of Paper I. What also became clear from Paper I is that there is an enormous heterogeneity worldwide in how fishing opportunities are allocated and, in several systems, there is some shared / pooled quota next to quota that is individually assigned. This mix of rights is common in other resource systems as well and might be overlooked due to a focus on singular solutions (Ostrom, 2010b).

Table 3: Knowledge gaps, papers and insights from this thesis

Knowledge gap	Paper	Main insight
It is unclear which (if any) attribute of individual quota systems preserves the health of marine populations	I	Many types of quota systems reduce the probability of overfishing compared to effort systems, individual allocation is also associated with reduced overfishing, but the strongest effect was found for quota.
It is not clear what the drivers are for the use of Icelandic balancing mechanisms	II	The Icelandic balancing system in its mixed fishery has a relatively high quota uptake, but overfishing is relatively modest. Pervasive incentives were nonetheless observed, overfishing may be limited by the fact that transformations flow from small to large biomass species which makes the volume of transformations relatively smaller (see summary figure from Paper II).
It is not clear how quota markets respond to financial volatility and what are drivers for quota trade	III	Quota trade reduced after the financial crisis and never regained the same activity. Drivers for buying quota were vessel size and company sizes, also reciprocity was significant in shaping trade.
The theoretical model of the Transitional Gains Trap (TGT) has not been empirically thought out nor implemented	IV	TGT can be calculated empirically, illustrative calculations show that TGT is not complete for the ITQ in Icelandic fisheries, moreover, under certain assumptions TGT reduces over time.
It is not known how the Icelandic marine foodweb will respond to combined warming and acidification.	V	Combined warming and acidification impact certain species/functional groups more than others. The Atlantic cod stock, the most important fishery in terms of catch-value and participants, is projected to be less productive but still increase compared to current levels under scenarios of warming and acidification.

Secondly, this research assessed the function of the fishery with the most elaborate catch-quota balancing options around the world, Iceland's demersal fishery (Paper II, see Table 3).

RQ2: How have ITQ's been implemented in Iceland's multi-species demersal fishery?

The research indicated **that pervasive incentives are likely to exist** because conversion ratios used for transferring quota between species were indicated **but not to reflect their relative profitability**. However, **overfishing because of the system was limited** as has been shown before (Woods et al., 2015). Indications were found that **this is the case because the system is tuned over time to the local socio-economic context**: a central role of Atlantic cod and the

limits being more restrictive for high biomass species, and it is exactly these species that tend to be the sink species for the species transformations.

The Icelandic quota mixed demersal fishery has a much higher quota uptake than in many other mixed fisheries around the world (Kuriyama et al., 2016; Mcquaw and Hilborn, 2020), and thus shows that if a catch-quota matching design is tailored well to the local context, economic and ecological goals of management could be balanced. These results are likely to be of growing relevance due to the increased implementation of individual quota systems and discard bans around the world (Chu, 2009; Condie et al., 2014; Mcquaw and Hilborn, 2020), and shifting distributions of marine species due to climate change (Campana et al., 2020 ; Fulton, 2011).

Thirdly, the research in this thesis studied drivers for quota trade in the largest quota market in Iceland (the market for cod quota). Accumulation of allocated quota to companies with more allocated quota is a driving mechanism for consolidation (a form of “the rich getting richer” which is a known social network mechanism). Consolidation has been observed earlier in the Icelandic quota system (Agnarsson et al., 2016; Pálsson and Helgason, 1995).

RQ 3: How does the quota market for permanent and temporary quota trades evolve over time? The boat’s gross tonnage (GRT) in our ERG models (Table 3) had a strong explanatory power, but only in the first fishing years studied (the initial years after introduction of the small-boat quota system) and the effect was the strongest for the small-boat market. The significant effect of boat size may have been explained by the switch to output controls which would have reduced each vessels landing-potential as previous to implementing ITQ’s in the small-boat segment effort quota were consistently overfished (Matthiasson and Agnarsson, 2010). Those that chose to remain fishing rather than exit the fishery would need to buy enough quota to remain profitable, and for larger boats, this would be a larger share of the quota. High reciprocity (in the lease markets around 20% of ties was reciprocal) and modularity pointed to the importance of personal ties in the quota trade networks.

Fourth, this thesis contributed with the development of a model for the Transitional Gains Trap (TGT) and empirical calculations based on the Icelandic ITQ system. In a previous analysis of the Icelandic ITQ system the TGT was invoked as an impediment to reallocation of grandfathered quota (Kristofersson, 2010): virtually all originally grandfathered quota in the main species in the Icelandic system were likely to be in the hands of new incumbents. If the government would consider reallocation of these quota many participants in the industry would be duped.

RQ4: What are drivers for quota trade? Paper IV (see Table 3) details a new methodology in which originally grandfathered quota are traced over time. Paper IV also presents analyses based on a set of assumptions on rent already generated by grandfathered quota; any excess profit from gifted profit could have technically been used to acquire new quota, thus **many quota purchases of older industry participants have in a way been compensated by gains from gifted quota**. Moreover, if quota were purchased (even if purchased by a new entrant) returns from this quota should be considered when calculating TGT. Under a set of realistic broad-based assumptions TGT is shown to decline over time in the Icelandic ITQ system, which would give policy makers leeway to consider reallocation of grandfathered quota.

Fifth, Icelandic fisheries are likely to be affected by climate change. It would be expected that, even in the case of increased productivity in case of warming, Icelandic fisheries will likely be affected through the pathway of ocean acidification and warming (Lam et al., 2016).

RQ5: How is Iceland's marine ecosystem likely to be affected by climate change and; what are the implications of climate change for the fisheries with most participants? Paper V (see Table 3) showed that ***different sensitivities of species for warming and acidification together re-shaped the Icelandic foodweb under scenarios of global change*** projected until 2100. Generally, ***the sensitivity of a few predatory fish caused changes in the foodweb by decreasing predation and competition***. Indirect effects due to climate changes impacts on important species have been addressed with other ecosystem models modeling different ecosystems around the world (Marshall et al., 2017; Olsen et al., 2018). Catches of the main fisheries in Iceland were projected to still be high (confirming findings in (Lam et al., 2016)) but catches for Atlantic cod and redfish were lower in scenarios of global change than under a baseline scenario with no changes. As the fishery for Icelandic cod is of key socio-economic (i.e. the fishery is of socio-economic importance since it has the most companies participating in all Icelandic fisheries and the highest landed value) importance in Iceland it is crucial that more information becomes available on the species' sensitivity for ocean acidification.

4.2 Scientific contributions – synthesis of the thesis results

The overarching question guiding the analyses in this thesis was: how do individual transferable quota management in fisheries perform across the different domains of sustainability? This chapter synthesizes results from the different chapters, focusing on ecological sustainability of fisheries management tools (Chapter 4.2.1.; addressing RQ1, RQ2, RQ5) and the socio-economic impacts of ITQ's (Chapter 2.2.2 addressing RQ4), as well as trade-offs between the different sustainability domains.

1.2.1 Ecological sustainability and fisheries management regimes (Papers I, II and V)

As mentioned above in chapter 3, Paper I demonstrated a benefit of individual allocation for the health of marine stocks. One explanation for this finding is a decreased race to fish in such systems (Birkenbach et al., 2017) and/or better adherence to the stipulated total allowable catch (Melnychuk et al., 2012). Another possible explanation could be that individual quota are often implemented alongside a discard ban (this has happened for instance in Iceland, but also on the west coast of the US). Implementing individual quota alongside a discard ban often results in very low quota uptake in mixed fisheries (Mcquaw and Hilborn, 2020; Melnychuk et al., 2012). This in itself could explain part of the findings in Paper I and is linked to the study performed in Paper II. The low quota uptake in mixed fisheries represents a trade-off: overfishing may be limited in such systems, but there is also a large loss in potential catch-value (Mcquaw and Hilborn, 2020). In Paper II this potential trade-off is studied in the Icelandic system, which has the most elaborate catch-quota balancing system in the world to deal with this trade-off in IQ systems. A harvesters mis-match between catch and quota can also be an incentive for (in some systems illegal) discarding (Acheson et al., 2015; Sturludottir, 2018). Transferability can

resolve this difference between catch and quota, and theoretically could reduce incentives for illegal discarding. However, transferability has not been positively associated with marine fish or invertebrate stock biomass in empirical studies (Melnychuk et al. (2016) found a negative association between allowing for either permanent or temporary transferability and marine fish or invertebrate stock biomass). Despite the lack of evidence supporting ecological benefits of transferability of quota in IQ systems, transferability may still be very important resolving differences between catch and quota, especially in mixed fisheries (Bromley, 2009). However, transferability is not always enough to resolve the difference between catch and quota, for instance in the presence of a general quota shortage (Holland, 2013) or high frictional costs. This is where more elaborate balancing mechanisms come into play which can help balance the mis-match between catch and quota and reduce incentives for illegal discarding for individual harvesters. Between-year transfers have been shown to be positively associated with stock biomass (Melnychuk et al., 2016), possibly caused by such reduced incentives for discarding. In Iceland between-species transfers (species transformations) are allowed, which have been the cause of overfishing of several stocks in Iceland, albeit not systematically so (Woods et al., 2015). Paper II showed that because of the tuning to the local context the economic and ecological goals of this fishery are relatively well-balanced. However, Paper II did show that the system is likely to carry pervasive incentives and in other systems with less constraining factors these could result in persistent overfishing. From a conservation point of view, as addressed in Paper II it would be prudent to add an upper limit to species transformations in Iceland into particular species. Nonetheless, in Iceland's mixed fishery overfishing is to some extent allowed for economic reasons and whether these catch-quota balancing mechanisms actually reduce discarding in individual quota fisheries is not yet known. This kind of nuance within a case-study in Paper II is not a part of Paper I, where whether balancing mechanisms are in place or not was not recorded for instance, which could cause our threshold for *overfishing* to be crossed (despite being an intentional element of the systems design).

In Paper II the central role of Atlantic cod in Iceland's socio-economic system is addressed; due to a prohibition on transforming quota into Atlantic cod (and other factors) ecological risks associated to the transformation system are likely reduced. Paper V addresses the central role of Atlantic cod in the Icelandic food-web: under scenarios of climate change in which Atlantic cod biomass is reduced, biomass of other species is increasing due to reduced predation and competition (mainly haddock and capelin). Moreover, future changes in the biomass of different species and the re-shaping of the marine ecosystem will have implications for important fishery sectors, as well as the usage of balancing mechanisms in the mixed fishery. In Paper V harvests levels remain at the current level, under which biomass increases for Atlantic cod are projected even if the stock is negatively impacted by climate change.

4.2.2 Socio-economic dimension of ITQ's: focus on quota markets (papers III and IV)

The socio-economic dimension of ITQ systems addressed in this thesis focused mainly on the instruments through which fishers gain access to fishing opportunities in Iceland, namely quotas and the quota markets. Papers III and IV studied quota markets in Iceland, one of the longest standing ITQ systems globally. Consequences for equity are addressed by investigating the

amount of quota held by those that got quota freely gifted, as well as those that could pay for the quota by windfall gains from grandfathered quota (Paper IV) and by addressing consolidation of quota (Paper III). While Paper III focusses on the largest market for lease and permanent quota in Iceland (the market for cod quota), Paper IV focusses on the 16 most important quota markets for permanent quota. Paper III distinguishes between the small-boat segment of the market for which quota was introduced more recently, while Paper IV studies both segments together, focusing on the total amount of quota traded over time. From Paper IV it's clear that most trade occurs in the first few years of the introduction of ITQ's, and after a few years there is little difference in the cumulative trade. As the large-boat market was already under quota for several years at the start of the study period (ITQ's were introduced in 1991 while the study period of Paper III starts in 2004) confirming the suspicion in Paper III that much of the quota trade had already occurred for the large-boat market. The rapid consolidation observed in the small-boat segment in Paper III probably had important implications for the fishing communities largely relying on landed catch from small-boat sector (Chambers et al., 2017) especially given the larger amounts of companies participating in the small-boat sector (Paper V). A recent survey demonstrated that many rural fishing towns are in a vulnerable state and that this is linked to decreased employment in the fishing sector / a vulnerable state of the local fishing sector (a vulnerability that can be also related to several fishing communities being dependent on a single quota holding company) (Kokorsch and Benediktsson, 2018). Iceland has implemented community quota and a coastal fishing opportunity in the summer to buffer the consequences of the ITQ for rural communities, but these represent very small amount of quota and have thus far not been enough to counteract the impacts in the fishing towns (Kokorsch and Benediktsson, 2018). The observed rapid consolidation in the small-boat fleet may have been facilitated by the fact that ITQ introduction in this fleet in 2004 coincided with the financial boom years in Iceland. There was a great willingness of the banks to provide loans to Icelandic fishing companies since the early 2000s and moreover loans were provided for other industry spending using fishing quota as a collateral (Paper III; Gunnlaugsson and Saevaldsson, 2016).

The small-boat segment and demersal fisheries have many more participants and a somewhat more equal distribution of quota holdings (Paper III, Paper V, Byrne et al., 2020), while the pelagic fisheries are highly concentrated (Byrne et al., 2020). This is reflected in the fact that quota trades are frequently occurring in the cod quota market and that TGT is much larger in those fisheries. Moreover, trade in recently grandfathered quota is much more limited than trade in species that were grandfathered in the early 90's probably because quota was gifted to already more established fishing companies. This gives some food for thought regarding new introductions of species in the Icelandic ecosystem due to climate change: if quota is gifted to the fishing industry based on for instance harvesting capacity it could be that quota holdings and the wealth generated from these is moving even further away from an equitable distribution. Despite the large amount of trade observed (in both Paper III and Paper IV), Paper IV concludes that the transitional gains trap is not complete and that there would be scope for policy reform, especially if there is a credible announcement of this reform in the future. However, implications for such reform and impacts on fishing industry and communities would need to be carefully considered.

4.3.3 Trade-offs in fisheries management

Potential trade-offs could arise in different quota management regimes. For instance, allowing for transferability could be economically more efficient as the fleet size reduces and fewer and more efficient harvesters spend less time harvesting (Merayo et al., 2018). On the other hand, quota are allocated only to vessel owners and rapid consolidation of quota (as studied in Paper III) may leave others in the fishing community without employment and without windfall from allocated quota impacting social dynamics in fishing communities (Chambers et al., 2017). Potential trade-offs related to different aspects of quota management regimes are shown in Figure 10. This thesis did not indicate a trade-off invoked by allowing for transferability between social and ecological goals, simply because transferability did not significantly impact marine stock status (Paper I). As addressed before in Paper I, hypothesized mechanisms for transferability to benefit the ecological sustainability dimension of fisheries would be, for instance 1) allowing better matching between catch and quota or 2) facilitation of monitoring with smaller fleet size 3) strength of property rights would be larger in case of allowing transferability. However, Paper I (and Melnychuk et al., 2016) found little empirical support for these theories. On the other hand Paper I did suggest a link between individual allocation and reduced overfishing. Quota trade in the study of Paper IV also took place indirectly (through vessel and permit transfers) which could enhance consolidation and a trade-off may thus exist between the ecological domain and social domain caused by individual allocation (Figure 10). Potential trade-offs between the economic domain and social domain were not investigated here (see Figure 10). An example of such a potential trade-off is that transferability may enhance the economic performance of fisheries (Arnason, 2012), potentially at the cost of fishing communities and maybe especially so in small-boat ITQ systems (Paper III, Chambers and Carothers, 2017). In Paper II a specific trade-off was studied between ecological fisheries management goals and economic ones; the fact that catch-quota balancing mechanisms allow for a higher quota uptake in mixed fisheries at the cost of allowing overfishing. Paper II shows that such trade-offs could be relatively well balanced, if management is tuned to a local context. This lesson may apply to a wider array of possible trade-offs associated with fisheries management tools.

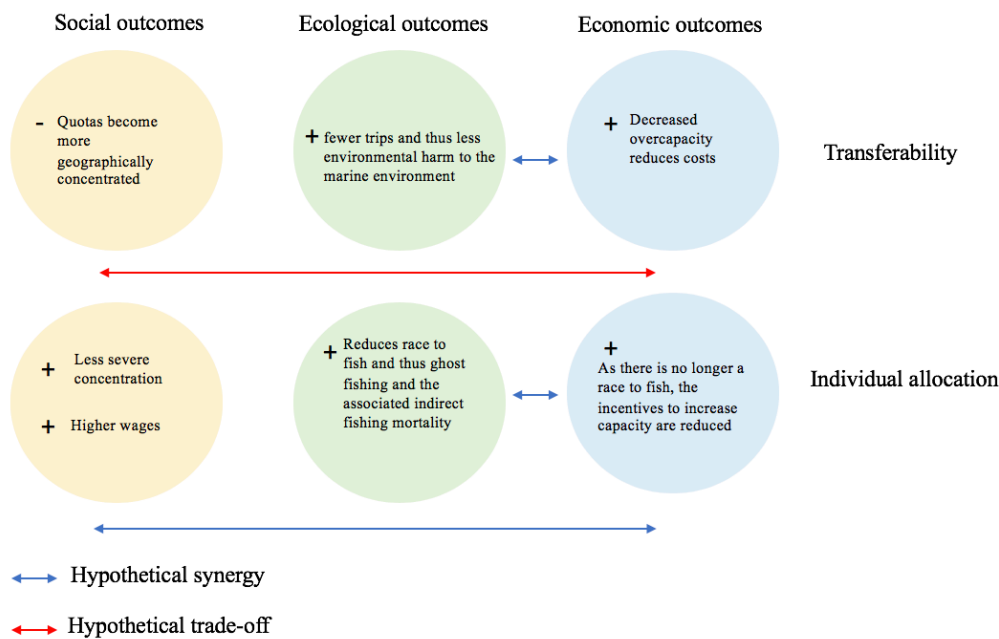


Figure 10: Hypothesized sustainability outcomes, synergies and trade-offs between these arising from fisheries management. This figure is adapted from the literature review in Paper I. Yellow circles represent hypothetical social outcomes of fisheries management attributes, green circles represent hypothetical ecological outcomes of fisheries management attributes, blue circles represent hypothetical economic outcome of fisheries management attributes.

4.3 Practical implications – synthesis of the thesis results

Outcomes of the research in this thesis have several practical implications. For instance, almost all output systems studied in Paper I outperformed a system based on total effort in reducing overfishing. A switch from input management to output management can be costly (Beddington et al., 2007) but our research, as well as that of (Melnychuk et al., 2012) indicates that it may be worthwhile for the sustainability of the resource in the long run. While our research also indicated that individual allocation aided with reducing overfishing, further research is needed to study the underlying mechanisms but it's likely that reduced racing behavior plays a role (Birkenbach et al., 2017), allocating quota individually may thus be desirable but may depend as well on certain social goals of policy. If harvest rights are individually allocated for a longer duration, this could cause issues with allowing for new entrants (Copes and Charles, 2004). Experimentation with other methods of allocation, e.g. leasing quota from the government or auctioning quota (possibly with different auction markets for different fleet segments) are possible policy options to deal with the new-entrants trade-off of individual allocation (Lynham, 2014). Another practical implication from this thesis arises from Paper II. As explained above implementing IQ's in mixed fisheries comes with new issues that could reduce possible economic benefits of IQ implementation. Paper II shows that some of the trade-offs in this fisheries management dilemma can be balanced if the design of the management regime is tailored to the local social-ecological context. Pervasive incentives were nonetheless indicated. In addition, Paper II contributes with several practical suggestions on how the ecological risks present could be modulated in the Icelandic system: 1) it would be better if balancing was done at the company rather than the vessel level 2) ecological risk could be reduced by prioritizing

between year transfers over species transformations 3) it would be prudent to add a limit on transformation into species to protect small-biomass species. Paper IV demonstrated with illustrative calculations that TGT can be calculated empirically, and that the trap may decrease over time. This potentially increases the political scope for reform of grandfathered quota systems. The reshaping of ecosystems and the “winners” and “losers” in the ecosystem as a result of climate change (addressed in Paper V) will also have economic implications. Especially in combination with the spatial shifts in the distribution of marine populations new species will be allocated and old species will be lost to fishers, some aspects of these shifts can be addressed by policy proactively, especially in the case of new fishing opportunities. ITQ’s are generally a quite flexible policy instrument to address small spatial shifts, i.e. a redistribution of a stock, such as happened with haddock in Iceland (Dobeson, 2018) would theoretically be solvable if it happens within the EEZ. However, the haddock case in Iceland demonstrated that there are notable differences between fleet-segments, for instance smaller companies occupied in coastal fishing do not have the flexibility to shift their fishing effort over a large spatial extend, have less flexibility using the catch to quota balancing mechanisms due to overall lower quota holdings (Paper II) and at the time had trouble accessing the quota market for scarce haddock quota which was under a strict TAC (Dobeson, 2018). Such small shifts that may also be driven by temperature changes can thus also stretch the flexibility limits of ITQ systems at least for some fleet segments. When stock shifts happen across countries quota systems could in theory allow for sustainable fishing of new fishing opportunities. However, conflict between countries over allowable catches often inflate the local TAC and there tends to be a delay in management action (Ojea et al., 2017; Pinsky et al., 2018). It is really expected that due to shifts in temperature and the poleward move of fish, new transboundary stocks will occur, governments could establish agreements both locally and internationally in anticipation of such shifts rather than the ad hoc decisions that are taken after the fact. Anticipatory governance offers such a framework in which decisions on social-ecological systems and climate change can be made in anticipation of the future changes, explicitly considering the large uncertainties (see Paper V) that are accompanying such changes (Guston, 2014). The Icelandic government could use scenario approaches to investigate the impacts of spatial redistribution of fish stocks as well as shifts in productivity (addressed in Paper V) and what objectives the government has in regards to the consequences of different fleet segments.

4.4 Limitations and further research

All papers in this thesis are limited in their scope, and several limitations are important to keep in mind when interpreting their results. For Paper I the scope of the study was limited to governmental policy, and thus in the classification method the study relied on the legal definitions of fisheries management systems. This was at times counter-intuitive for fisheries managers/experts. For instance, in some individual quota systems quota are pooled after individual allocation, such systems may operate “de facto” like a cooperative but legally quota is with the individual company. Moreover, such systems often have an option to “opt out” of the coop and fish the quota share in a competitive fishery outside of the IQ system. The same classification method was applied to the duration of harvesting rights but also the legal definitions of duration may differ from the perceived duration of fishing opportunities based on

historical precedent. Often fisheries experts would classify harvesting rights as indefinite while legislation explicitly states that rights can be revoked. The distinction between our classifications of “legal ability” and “indefinite” are however up for debate as even secure property rights can be modified if governments change their policies (though usually owners can expect compensation in such cases). Paper II was limited by the use of indicators to estimate effects of drivers on the usage of balancing mechanisms. For instance, lease prices were assumed to be “arm’s length” and to reflect the profitability of a fishery. However, lease prices can be impacted by species for which there is a fleetwide quota shortage (so called “choke species”) (Holland, 2013; Mcquaw and Hilborn, 2020). This is the reason for the design of a choke species indicator, but as explained in Paper II both indicators are imperfect and the lease price may still be impacted by a gradually increasing quota shortage throughout the fishing year. Also our use of a targeting indicator in Paper II is rather simplified and an elaborate analysis of technical interactions in Icelandic fisheries should be undertaken to really know which species or species mixes are actively targeted (Katsanevakis et al., 2010). Both Paper III and Paper IV are limited in their scope by studying Icelandic fisheries at the level of single companies. Individuals may own several companies and companies may own shares in other fishing companies (Cooper et al., 2014), if this is the case total holdings of companies/individuals may be larger than those estimated in Paper III and Paper IV. The thesis limits itself in the study of one particular dimension of equity: i.e. the access to fishing opportunities via quota and the quota markets, while other dimensions to equity are of interest as well for a complete sustainability assessment (income distributions in the fishing industry, employment, financial health and vulnerability etc.).

Uncertainties are present in all analyses in this thesis, for instance in Paper I there is a lot of variability in the outcome variables across fisheries and confidence intervals in several tests are wide compared to the effect sizes that are found. While a wide range of cases was studied (more than 400 stocks, over a time period of almost 30 years, although not all of time-series were complete), more data-collection on the topic could increase the statistical power of the tests and reduce the uncertainty in our results. Paper V should be considered illustrative modeling of the Icelandic foodweb under scenarios of climate change. There is very little information on possible Ocean Acidification effects on key species in the Icelandic foodweb. Paper V highlights several important species for which the future under climate change is highly uncertain. Moreover, several broad-based assumptions were made when modeling climate change impacts on the Icelandic ecosystem: recruitment and growth were impacted by a fixed change in growth or recruitment under ocean acidification scenarios rather than a gradual impact of increasing acidity and temperature was increased in one time-step rather than progressively increasing over time. Broad assumptions were also made for the forward projections in Paper IV especially when rent and windfall profits were estimated, as there are no sources for the true price of quota shares in Iceland, as stressed in Paper IV these calculations should be considered illustrative for the Icelandic system and offer no precise estimate of TGT.

Out of the limitations stipulated above several interesting avenues for further research arise. In future research it would be good that an explicit distinction is made between government policies and legislation and “de facto” policies and informal management structures that are in

place. For instance, the category “self-governed-quota-pool” is only one way in which self-governance could come into play. In future work relating fisheries management interventions it would be interesting to record whether e.g. pooling of quota after allocation occurs. The combination of legislation and informal management practices are likely to impact the conservation of marine populations. Moreover, Paper I only looked at single management regimes and attributes on their impact on marine populations, one avenue to expand this research would be to look at combinations of management interventions and their impacts on marine populations (e.g. combinations of spatial effort and quota (Stefansson and Rosenberg, 2005)). As explained above Paper II used a very simplified indicator of the ability of fishers to target individual species. A much-needed future avenue for that work would be an analysis of all technical interactions in the Icelandic demersal fisheries. Such an analysis would record which species are likely to be caught with one another, with which gear and in which spatial locality (Branch and Hilborn, 2008; Katsanevakis et al., 2010). This work can then also be used to parametrize the Icelandic Atlantis model and project forward under climate change. In this way, more realistic catch mixes and possible future constraints (e.g. “choke species”) can be modeled. Moreover, given that fishing companies may own shares of other (Icelandic) fishing companies, a detailed study of the ownership structure of Icelandic fisheries would benefit the research into industry consolidation and would be further extensions of Paper III and IV.

As shown in Paper V different species are likely to have different responses to warming in the marine ecosystem around Iceland, which will also cause changes in species mixes. A possible future issue for the fishery is that this will cause different quota shortages in different locations around the country (Dobeson, 2018). Paper V also made clear that research on ocean acidification impacts on species in the Icelandic waters is lacking. More experiments for ocean acidification on Icelandic key species are needed. The approach in Paper V to select species of interest could be used to select important species to focus on in combination with a traits-based approach. Another possible extension of Paper V could be a comparison between different ecosystem models, e.g. Ecopath with Ecosim, to assess the impact of model structure on the outcomes of the modeled climate change scenarios (Ribeiro et al., 2018).

Further research is also needed into the mechanisms behind the findings in Paper I, as well as possible alternative explanations. For instance, Birkenbach et al. (2017) demonstrated that individual quota slow the race to fish, which could be one pathway through which Individual Quota could reduce overfishing. An interesting new research avenue could be to find out if the decreased racing in individual quota systems results in reduced quota overages (which have been shown to occur less in IQ systems (Melnychuk et al., 2012)) and if this is the underlying cause for the reduced occurrence of overfishing in such systems.

Finally, further research is needed to test if there are trade-offs between social, economic and ecological goals across a wide range of fisheries management systems. This thesis provided an indication that the hypothesized trade-off caused by ITQ’s between ecological fisheries management goals and social fisheries management goals may not exist: no definite proof was found for the ecological benefit of implementing tradable quota. However, this thesis has not studied consolidation of harvesting rights across a range of fisheries management regimes,

which would be needed to assess if, for instance systems with individual allocation are more consolidated through the sale of permits with quota attached than systems without such individual allocations. If this would be the case a possible trade-off could exist between ecological goals and social goals (e.g. fair access to the fishery, distributional consequences of individual quota) caused by individual allocation. Previous research has not indicated such a trade-off (Asche et al., 2018). Asche et al. (2018) indeed showed synergies between social, economic and ecological outcomes of fisheries, which were the strongest in “rights-based management” regimes (any system with individual allocation including ITQ’s). This result is exactly the contrary of the hypothesized trade-offs that have been associated with ITQ systems, regarding social and economic outcomes, and ecological and social outcomes. This dispute in the literature is therefore highlighting the need to better understand and empirically demonstrate trade-offs and synergies associated with fisheries management systems, however, several fundamental questions remain unanswered, such as (i) what are the ecosystem effects, (ii) would better indicators of stock health or of other metrics of stock performance change these results, (iii) what is the impact of the different fisheries management schemes on equity in the fishery (i.e. equity in the broad sense, including fair access to fishing opportunities as addressed in this thesis, but also secure employment, financial health of the industry etc.), and (iv) how do these findings relate to fisheries management systems attributes, beyond merely comparing “rights based management” versus open access regimes and licensing schemes. (v) how do these trends evolve over time?

5 Conclusions

The analyses in this thesis have addressed several important sustainability implications of the implementation of ITQ's, the overarching question guiding the analyses in this thesis was: how do individual transferable quota management in fisheries perform across the different domains of sustainability? Research in this thesis has demonstrated that fisheries management regimes in which fisheries opportunities are allocated as quota and / or are allocated individually experience reduced overfishing compared to controls that do not have these attributes. The thesis however found little support for transferability and no support for longer duration and their impacts on sustainability of fish stocks. The thesis also showed rapid consolidation in an important small-boat fishing sector in Iceland, which may have had negative implications for local fishing communities. The fact that longer duration was not associated with positive ecological outcomes may give policymakers leeway to design fisheries management systems with good ecological outcomes but less severe distributional consequences as past implementation of ITQ's. This will and should be a careful balancing act between economic (e.g. increased efficiency, profitability), ecological (e.g. stock health, ecosystem health) and social goals (e.g. fair access, equitable distribution of gains, quality jobs) of policymakers. In addition, the thesis showed that with an adaptive design ecological and economic goals could be balanced in an important mixed fishery in Iceland. Finally, research is beginning to emerge that shows that marine species are unequally affected by climate change. In a final chapter the thesis shows that under different scenarios of global change a re-shaping of the Icelandic foodweb is likely. The re-shaping of the foodweb will likely be to the benefit of some resource users and to the loss of others, which requires adaptive policy design in the near future.

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Paper I



Which attributes of fishing opportunities are linked to sustainable fishing?

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Title: Which attributes of fishing opportunities are linked to sustainable fishing?

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Abstract

To prevent the overfishing of marine fish populations, governments often limit access through the allocation of fishing opportunities. While some studies have linked particular systems of fishing opportunities to sustainable outcomes (particularly individual transferable quota (ITQ)), it remains unclear, whether it is the use of exclusive property-rights (individual allocation and secure duration), the use of a market for fishing opportunities (tradability), or the quota limits themselves that underlie these positive outcomes. To determine which system attributes lead to sustainable fishing we developed a novel method to systematically classify how fishing opportunities are allocated for 443 global fish stocks from 1990 to 2018 to produce the longest and most comprehensive dataset and longitudinal study of its kind. Our results revealed that quota limits and individual allocation were associated with a reduced probability of overfishing, with the most robust result for quota limits. The leasing of quota was the only attribute associated with a reduced probability of overfished biomass. Whereas some previous studies have emphasized that market-based systems or those with strong property rights (i.e. a long duration) are associated with sustainable fishing, these benefits are small or not significant once proper controls for other system attributes are introduced. These results highlight the importance of considering all attributes of institutional design in the governance of common pool resources.

Keywords

Common pool resources, Fisheries management, Incentives, Market-based instruments

1. Introduction

The “tragedy of the commons”¹, where a common pool resource is overexploited as the benefits of an extra harvest are gained individually while the costs of overuse are shared across many, has been frequently invoked in natural resource management². Often two alternative management solutions are sketched to escape the tragedy, either the state intervenes or the resource is privatised. Property rights (e.g. a fenced off meadow) align the costs and benefits of an action to the same user and therefore ‘solve’ the commons by removing it while ‘top-down’ government regulation forces users to adhere to prescribed rules to limit overall costs³. Progress in the study of common pool resources has nuanced this dichotomy with abundant evidence on community self-organisation (i.e. a community establishes rules for sustainable usage of common pool resource), illustrating that privatisation or direct government management are not the only two pathways to avoid tragedy^{4,5}.

Marine fish populations, particularly highly mobile populations, remain a common pool resource outside of private ownership due to the impracticality of demarcating individual fish as belonging to different owners or for owners to only catch the fish that belong to them. The response in fisheries management has been to develop private ownership over the right to fish (‘fishing opportunities’) rather than the fish themselves⁶. Individual transferable quota (ITQ), where harvest limits are held individually, can be freely transferred, and are generally held for a long duration, are an increasingly common fisheries management system throughout global fisheries^{7,8}. While studies have linked the use of ITQs to sustainable fishing^{9,10}, the effect of each ITQ attribute (i.e., the I, the T, and the Q) remains underexplored and the lessons for policy design unclear³.

For each ITQ attribute, theoretical claims have been advanced to support a link to sustainable fishing, but counterclaims have also been raised. Quota limits have an important advantage compared to limiting fishing effort (e.g. time at sea, number of hooks or pots) because the quantity of fish harvested is more closely tied to fishing mortality^{11,12} and more predictable to control^{13,14}; however, quota limits may be more difficult to enforce with over-quota catches simply discarded at sea^{15,16}. Individual allocation empowers fishers to choose when to use them¹⁷, including during lower impact fishing seasons; however, the common pool aspect of fish stocks remains and with it the incentive for individuals to fish more (or discard less valuable fish) for private gain as the benefits of fishing less are shared¹⁵. Transferability in fishing opportunities leads to concentration in the hands of the most profitable businesses^{6,11} who may be more likely to pay for fisheries management¹⁸ of a smaller fleet¹⁹; however,

profitability is not synonymous with efficiency given unaccounted for externalities^{12,20} and fleet contraction can still occur through vessel sale without transferability of fishing opportunities.

Beyond these ITQ attributes, there is little literature on the attributes used in alternative allocation systems, such as pooling (i.e., opportunities are fished collectively without allocation to individuals), leasing, rationing throughout the year, or allowing the industry to self-govern allocations (e.g. allocating fishing opportunities to a cooperative, which does not include initial individual allocations that are later grouped by cooperatives²¹).

The allocation of fishing opportunities is closely linked to duration, a specific attribute of fishing opportunities. Several studies have claimed that when the duration of fishing opportunities is sufficiently long and secure, fishers themselves will bear the consequences of (un)sustainable behaviour^{22,23}; however, other studies have noted that the common pool aspect of fish stocks remains, hence the continued need for enforcement³, and long-term property rights in other sectors have still led to unsustainable behaviour³.

As many of the theoretical claims linking ITQ attributes to sustainable fishing are contested, it is especially important to test the empirical effect of existing fisheries management systems. Unfortunately, several studies have used contested proxies as metrics of sustainable fishing²⁴ and much of the existing empirical literature focuses on a particular management system and does not distinguish between the specific attributes of the management system (Table 1). This focus on the management system rather than its attributes means that the control groups used in these studies also suffer as all other management systems, including systems with no management at all, are grouped together in one single control group. In contrast to the results of studies comparing ITQs systems to a single control group, the few empirical studies that analyse attributes show no conclusive evidence that individual allocation, transferability, or duration are associated with sustainable fishing beyond the benefits of quota management (Table 1).

Table 1. Empirical research linking attributes of fisheries management systems to sustainable fishing.

Ref	Coverage	Dependent variable	Method	Individual allocation	Transferability	Quota	Duration	Multiple attributes
9	11,153 fisheries, 1960–2003, 121 ITQs, global **,^^	Collapsed landings (binary)	Difference-in-differences	-	-	-	-	ITQ: Lower probability of collapse
8	20 ITQ stocks, global, from 16 to 36 years *,^	Biomass change	Descriptive	-	-	-	-	ITQ: Mixed effect
25	>11,000 fisheries, 1950–2003, 121 ITQs, global, **,^^	Collapsed landings (binary)	Difference-in-differences (subset)	-	-	Lower probability of collapse	-	IT: Lower probability of collapse
26	15 IQ/ITQ in North America *,^	Landings, mortality, biomass, habitat-damaging gear, discards, catch:quota	Before after control impact	-	-	-	-	IQ/ILQ/ITQ: Lower variability of mortality and biomass, no effect on mortality or biomass
10	345 stocks, global, 2000–2004 ***,^	Catch:quota, F:Ftarget, B:Btarget	1) Fixed-effects models, 2) mixed-effects models, 3) propensity score matching	Lower catch:quota, lower probability of high overfishing	-	Lower mortality, no effect on biomass	-	IQ/ILQ/ITQ: Lower variability of catch:quota, lower mortality, no effect on biomass
27	84 IQ/ITQ and 140 reference fisheries ***,^	Landings, mortality, biomass	Difference-in-differences (Bayesian)	-	-	-	Lower variability of landings and mortality	IQ/ILQ/ITQ: Lower variability of landings and mortality, no effect on mortality or biomass
28	167 stocks, global, 2000–2004 ***,^	Catch:quota, F:Ftarget, B:Btarget	1) mixed-effects models 2) random forest models	No independent effect of exclusivity	Lower biomass	-	No effect	-
29	298 MSC-certified fisheries, 170 ITQ/IQ/TURF fisheries, 136	MSC certification scores (includes stock)	1) Bayesian belief networks, 2) statistical association	-	-	-	-	IQ/ILQ/ITQ/TURF: Higher probability of high MSC score

	which are "SET" ***, ^^	assessments)						for stock assessment
30	178 fisheries, 27 countries. 78 transition to ITQ/IQ/TURF ***, ^^	Mortality	1) Regression, 2) non-parametric approach	-	-	-	-	IQ/ILQ/ITQ/TURF: Lower mortality on overexploited stocks, no effect on others
19	ITQ/IQ/TURF ***, ^^	Collapsed (binary)	1) Difference-in-differences 2) Instrumental variable	-	Lower probability of collapse			IQ/ILQ/ITQ/TURF: Lower probability of collapse

Biological data

- * Manual
- ** Sea Around Us database (<http://www.seararoundus.org/>)
- *** RAM legacy database (<https://www.ramlegacy.org/>)
- **** MSC fisheries database

Classification data

- ^ Manual
- ^^ EDF catch share database (<http://fisherysolutionscenter.edf.org/database>)

As there are conflicting theoretical claims and the empirical evidence on specific attributes is limited and ambiguous, an important research question remains: Which attributes of fisheries management systems, if any, are associated with sustainable fishing? To answer this question, we compiled the largest dataset on fisheries management systems to date, covering 1990–2018, and tested different systems and their attributes against two metrics of sustainable fishing: mortality (i.e., whether a fish stock is subjected to overfishing) and biomass (i.e., whether a fish stock is overfished).

2. Methods

Data collection

Management data: To classify fisheries management systems, we used a combined primary and secondary research approach by reviewing government legislation and existing fisheries literature as well as consulting fisheries managers and research specialists. In total, we consulted 230 experts for classification queries; 173 replied; of which 116 either provided or confirmed classifications.

Biological data: For data on our sustainability metrics we used the RAM legacy database v4.491 (<https://www.ramlegacy.org/>, RAM Legacy Stock Assessment Database (2018)). These data were extracted from stock assessment documents with information on estimated annual biomass (spawning stock biomass or total stock, we refer to both of these measures as B) and exploitation rates (instantaneous fixing mortality or exploitation ratios (catch/total biomass), we refer to both of these measures as F). We only used assessments which also estimated target reference points that would generate maximum sustainable yield (i.e., F_{msy} and/or B_{msy}). Stocks were only included if they had five years of data on F/F_{msy} and/or B/B_{msy} from 1990 to 2018.

Management systems classifications

We developed 13 exhaustive classifications of management regimes based on a decision tree of potential attributes (Figure 1). Classifying management systems based on attributes in a decision tree allowed for the standardisation of systems where existing definitions were vague and allowed us to control for system attributes in a straightforward manner. Each branch in the decision tree (Figure 1) indicated the presence or absence of a system attribute (e.g. the first branch indicated whether quota or effort management was used). The blue-shaded boxes in Figure 1 are the 13 exhaustive classifications used in this study. It needs to be noted that due to this approach (i.e., the classification by attribute) our usage of terms can differ from previous usage in the body of literature on fisheries management systems (e.g. we use IQ only for individual quota systems in which no leasing or permanent transferability is allowed, while others have used this term for all systems with individual allocation and quota, including ITQ and ILQ).

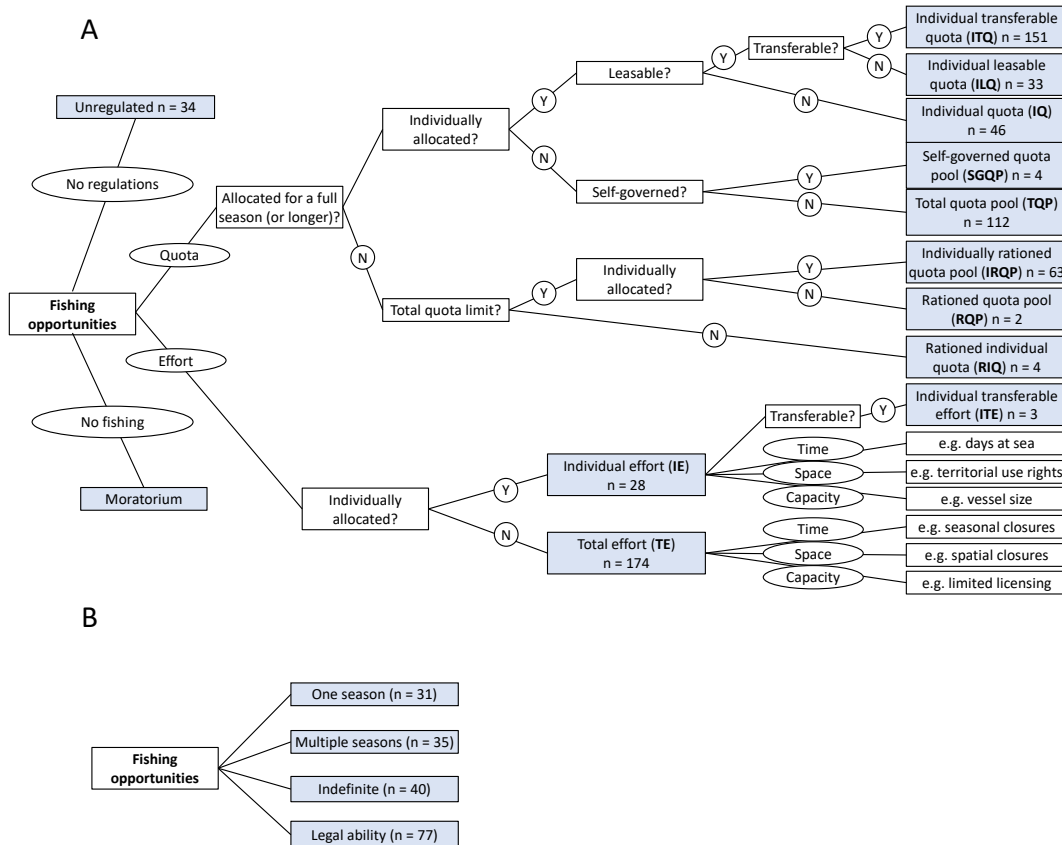


Figure 1. A) The classification decision tree for fisheries management systems based on their attributes. The blue terms are the 13 exhaustive classifications used in this study. B) The classification decision tree for the duration of fishing opportunities. Definitions used for the classifications are recorded in Table A1. For each exhaustive classification the number of unique stocks and management classifications that occurred in the dataset are noted. Not all classification numbers overlap with timeseries of F/Fmsy and or B/Bmsy, Table A3 contains the number of stocks per management classification overlapping with timeseries of F/Fmsy and/or B/Bmsy as well as average and median F/Fmsy and B/Bmsy values. While individually allocated, IRQP is not classified as an individual quota system due to the fact that quota is still fished competitively. Note that because of the new classification method (classifying systems by attribute), our usage of terms can differ from those commonly used in the literature (e.g. we only use the term IQ for non-transferable individual quota).

Where multiple allocation systems were used by a fisheries administration to manage a fish stock (e.g. different systems for coastal and industrial fleets), we assigned a percentage to each system based on the size of the allocation to each subsystem. Similarly, where multiple fisheries administrations exploited the same fish stock, we assigned a percentage to each fisheries administration based on the size of the allocation to each fishing administration, or, if no formal shares existed, the size of catches. The resulting stock classifications were thus a combination of systems used between administrations (where applicable) and within administrations (where applicable). If the use of multiple management systems prevented any one system from representing 75 percent of the fishing pressure for a fish stock, then we did not assign a classification to that fish stock as it was a ‘mixed system’ (following¹⁰). Particular

system attributes (e.g. quota, individual) could reach the 75 percent threshold if they were present in multiple systems leading to the inclusion of these stocks in the attribute-level analysis.

In addition to the method for allocating fishing opportunities, we also classified the duration of fishing opportunities into four categories: single season, fixed multiple seasons, indefinite, or of an unspecified duration with the legal ability to change allocations (Figure 1B, further details in Appendix A). Individual allocation and duration are often used as interchangeable terms (e.g. analysis of ‘catch shares’); however, duration operates as an independent attribute that can vary across all allocation types. We only assessed duration for individual systems, where fishing opportunities were allocated as a separate unit from the fishing licence which may have had its own specified duration.

Sustainability definitions

To define sustainable fishing, we assessed fish stocks against two metrics (in line with ^{10,27}): fishing mortality divided by the fishing mortality needed to achieve maximum sustainable yield (F/F_{msy}), and biomass divided by the biomass that can produce maximum sustainable yield (B/B_{msy}). We defined a fish stock as subjected to *overfishing* when the fishing mortality was higher than 1.1 times F_{msy} (following ¹⁰) and a fish stock as *overfished* when the stock biomass was lower than 0.8 B/B_{msy} (following³¹). We selected this threshold approach as our main analysis as we believe that higher or lower fishing pressure can only be assessed against a defined threshold (i.e. an increase in fishing pressure from a low base could still be sustainable). We only included stocks from the first year that F/F_{msy} was at least 0.5 (where data on F/F_{msy} was available) to control for fisheries that were not yet developed or of little commercial interest.

Sensitivity analyses: We used two alternative thresholds for the definition of overfishing and overfished (both also used in ¹⁰), for *high overfishing* ($F/F_{msy} > 1.5$) and *highly overfished* ($B/B_{msy} < 0.5$). We also modelled continuous levels of F/F_{msy} and B/B_{msy} , the methodology for this approach is presented in Appendix B.

Data analyses

To estimate the effect of management systems and their attributes on fisheries sustainability we used two modelling approaches: (1) a set of mixed-effects models testing both systems and

attributes, and how these were associated to fisheries status; and (2) a difference-in-differences (DiD) approach that tested systems where attributes changed (also using mixed effects).

The mixed-effects modelling framework allowed for the introduction of random effects for variables where the sustainability indicators were more likely to share a similar response. For example, a response of a stock in one region to a management system was more likely to correlate to the response of another stock in the same region³². For each model, we assumed that the residuals followed a first order autocorrelated (AR1) process which controlled for the fact that the time-series observations were serially correlated at the stock-level. Systems and attributes with fewer than 10 classified stocks were excluded from all analyses below.

First, we modelled the sustainable fishing metrics S for region r , stock s (nested within region), and year t as a function of the fisheries management system (and its multiple attributes):

$$S_{r,s,t} = \beta_0 + \beta_1 M_{r,s,t} + R_r + R_t + R_s + \varepsilon_{r,s,t} \quad (1)$$

where $M_{r,s,t}$ is a dummy variable for the management system in place, R_r is a random effect dummy variable for the region, R_t is a random effect dummy variable for the year, R_s is a random effect dummy variable for each stock. If variance explained by any random effect approached 0 we removed those random effects and confirmed that model estimates did not change. We compared the effects of all management systems against *total effort* (TE) as a control group as there are very few *unregulated* fisheries in our dataset.

Second, we modelled the sustainable fishing metrics as a function of the attributes I, T, and/or Q, L, R and P:

$$S_{r,s,t} = \beta_0 + \beta_1 Q_{r,s,t} + \beta_2 I_{r,s,t} + \beta_3 T_{r,s,t} + \beta_4 L_{r,s,t} + \beta_5 R_{r,s,t} + \beta_6 P_{r,s,t} + \beta_7 SG_{r,s,t} + R_r + R_f + R_s + \varepsilon_{r,s,t} \quad (2)$$

The metrics of sustainable fishing was modelled by dummy variables Q (quota), I (individual), T (transferable), L (leasable), R (rationed), P (pooled), SG (self-governed). Random effects were the same as in Equation (1). Because estimates for I and T could be impacted by both quota and effort systems we also modelled Equation (2) for quota systems only as a sensitivity (i.e., all predictor variables were kept the same except that the Q (quota) variable was removed).

We modelled the impact of the duration of fishing opportunities as follows:

$$S_{r,s,t} = \beta_0 + \beta_1 D_{r,s,t} + R_r + R_t + R_s + \varepsilon_{r,s,t} \quad (3)$$

where $D_{r,s,t}$ represents the duration of fishing opportunities in a management system. We compared the effects of duration against *single season* as a control group. Random effects were the same as for Equations (1) and (2). We tested for collinearity between duration and individual quota attributes (i.e., leasable and transferable) and since these values were rather low (i.e., $R^2 < 0.25$) we kept Equation 2 and Equation 3 as separate models.

As a second approach, we employed a DiD analysis for all transitions where a Q, I, T, R, or P element was “added”, for instance a transition from non-individual effort management to individual effort management (addition of I), or a transition from individual quota to ITQ (addition of T). This second approach is commonly used for analysing time series data where systems that undergo a change (i.e., treatment) are compared to systems that remain the same (i.e., control). A key assumption in this approach is that treatment stocks would have followed a similar trajectory to control fisheries if no change had occurred³³. DiD modelling was previously employed to study the effects of IQs, ILQs, and ITQs on sustainable fishing (Table 1).

Equation 4 represents the DiD approach where treatment stocks were compared to control stocks:

$$S_{r,s,t} = \beta_0 + \beta_1 Tr_{r,s,t} + R_r + R_s + R_t + \varepsilon_{r,s,t} \quad (4)$$

The sustainable fishing metrics were modelled by dummy variable Tr (treatment, i.e., addition on I, T, or Q in the treatment fishery, a dummy variable which was coded 1 after the addition of the attribute and coded 0 for control stocks or prior to introduction of the attribute in treatment fisheries). The other variables are the same as Equations (1)–(3). Because attribute transitions could be impacted by transitions of multiple attributes (i.e., the transition to I and T occurs simultaneously), as a sensitivity test the DID was also repeated for quota systems only.

For a subset of stocks (n=19), we matched treatment and control stocks for the same species in the same region or regions closely located to one another (Table A2). This approach controlled for confounding circumstances, such as changes in demand or climate change impacts that affected particular species and regions³³. Treatment fisheries transitioned from pooled quota to

individual allocation while control fisheries remained under pooled quota. Previous research requested such an approach be undertaken to separate the effects of attributes of I(T)Q systems from the effects of quota management (Branch, 2009; Bromley, 2009). For this analysis, we grouped all individually allocated quota systems due to the small sample size:

$$S_{r,s,t} = \beta_0 + \beta_1 Tr_{t,s} + \beta_2 P_{t,s} + \beta_3 Tr_{t,s} * P_{t,s} + R_s + \varepsilon_{t,s} \quad (5)$$

The sustainable fishing metrics S were modelled by dummy variable Tr (treatment), P (before and after treatment) and their interaction, β_3 represents the DiD estimator³³. Initially the model contained a dummy variable for the treatment and control pair but near zero variance was explained by this random effect, probably due to the high amount of variance explained by the stock-level autoregressive component.

All models were implemented using the package `GlmmTMB`³⁴ in R studio version 1.1.463³⁵.

3. Results

Frequency of fisheries management systems in the classifications

The most frequently observed fisheries management systems in our dataset were total effort (TE, number of stocks = 174, Figure 1A), where an input to fishing is limited at the fleet level, individual transferable quota (ITQ, n=151), and total quota pool (TQP, n=112), where a quota cap is set and fished collectively by the fleet until it is exhausted. Individually rationed quota pool (IRQP, a collective quota system where quota is allocated in rationed periods over the year, e.g. a weekly limit for vessels) and individual quota (IQ) were also frequently observed (n=63 and n= 46, respectively, Figure 1A). All other allocation systems were extremely rare: rationed individual quota (RIQ, n= 4, where individual quota is allocated for a shorter term than a full season), self-governed quota pool (SGQP, n=4, where quota is formally allocated to a group such as a cooperative), individual transferable effort (ITE, n=3), and rationed quota pool (RQP, n=2, where quota is allocated to the fleet, for a shorter time than one fishing season).

Regarding the duration for which individual quota (IQ, ILQ, ITQ) is held, the most frequent observation was a ‘legal ability’ to change allocations (e.g. by changes in the fisheries management plan, n=77, Figure 1B), although other durations were all also frequently observed

(indefinite, n= 40, multiple seasons, n=35 and for one single season, n=31, Figure 1B). The total amount of IQ systems in Figure 1A is larger than the amount of stocks with classified duration in Figure 1B because a single stock can be managed under multiple IQ systems over the 1990-2018 period (e.g. a stock can be managed under IQ for several years and be managed under ITQ for several subsequent years).

Frequency of sustainable fishing indicators in stock status

We found that *overfishing* frequently occurred across all regions and management systems. The regions with the highest shares of overfishing were the Mediterranean & Black Sea and northern Europe with 84 percent and 69 percent of observations, respectively. TE, unregulated, RQP, RIQ, and ITE management regimes had the largest shares of overfishing occurring, ranging between 71 and 100 percent of observations. In contrast, TQP, IRQP, ITQ, and individual leasable quota (ILQ) management regimes had the lowest share of overfishing ranging between 30 and 35 percent of observations (Figure 2A).

Of the total sample, a smaller number of observations, only 32 percent, were in an *overfished* state. Individual effort (IE) was the management regime with the largest share of fish stocks in an *overfished* state with 71 percent of observations, followed by TE and unregulated with 44 and 45 percent, respectively (Figure 2B).

The effects of fisheries management systems on sustainable fishing indicators

Several fisheries management systems were associated with a reduced probability of *overfishing* and/or being in an *overfished* state when compared to the control system of TE (Figure 2C). Strong effects for reducing *overfishing* were found for individual quota systems (including ITQ, ILQ, and IQ), with the strongest effect associated to ITQ (Figure 2C). One form of pooled quota, TQP, was also significantly associated to a reduced probability of *overfishing* (Figure 2C). Only one management system, ILQ, was linked to a significantly reduced probability of *overfished* biomass (Figure 2C), although the confidence intervals were wide.

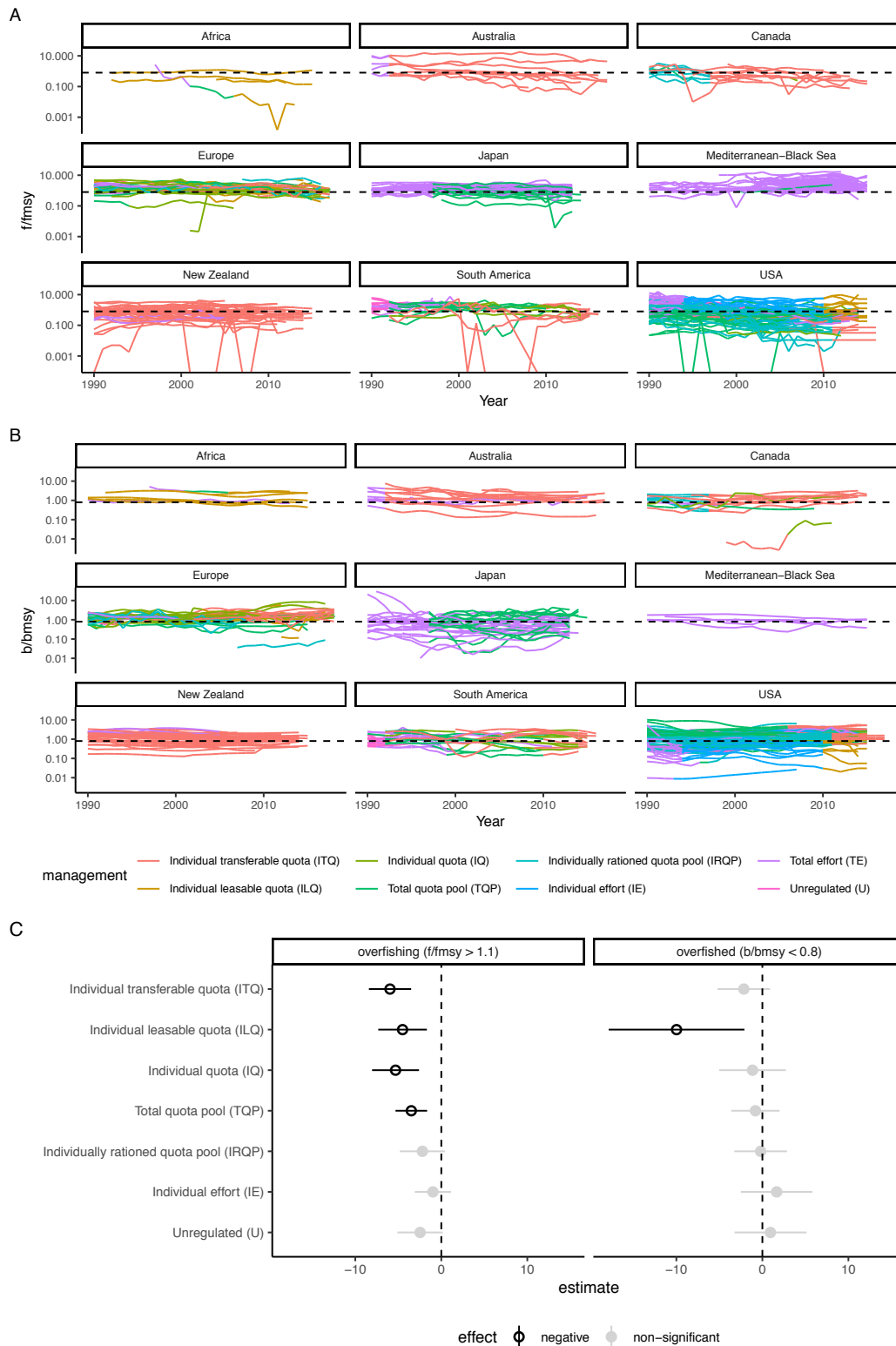


Figure 2. A) F/F_{msy} for classified fisheries management systems with 10 or more observations (dotted line indicates the threshold for *overfishing*, i.e., when $F/F_{msy} = 1.1$). B) B/B_{msy} for classified fisheries management systems with 10 or more observations (dotted line indicates the threshold for *overfished*, i.e., when $B/B_{msy} = 0.8$). C) Estimates and 95 percent confidence intervals of management systems compared to TE (Model Eq.1 in methods section). Negative (black) values indicate that the management system reduces the probability of the outcome variable, for example IQ reduces the probability of overfishing compared to TE. Table B1 shows further model details (i.e., odds ratios, p-values, random effects and R^2 values).

Disentangling the effects of system attributes on sustainable fishing

Without controlling for other factors (i.e., region, autocorrelation, fishery type), management systems with quota limits, individual allocation, pooling, transferability, and leasing had lower frequencies of *overfishing* and *overfished* states, with the largest difference for quota limits (69.0 percent without versus 38.6 percent with quota limits, Figures 3A and 3B).

Association between attributes and sustainable fishing (mixed-effects models): Controlling for other factors in the mixed-model analysis, we found a reduced probability for *overfishing* associated to fisheries under quota limits and/or when fishing opportunities were allocated individually (Figure 3C), with the largest effect found for quota limits. For transferability, leasing, rationing and pooling, despite a lower occurrence of overfishing (Figure 3A), no significant effect was found once other factors were accounted for in the mixed-model analysis (Figure 3C). The leasing and pooling attributes were associated to a reduced probability for stocks being in an *overfished* state (Figure 3C). We found no significant difference in the probability of *overfishing* when systems with longer durations were compared to those allocated for a single season, although there was an increased probability of an *overfished* state associated to fishing opportunities allocated for fixed multiple seasons (Figure 3D).

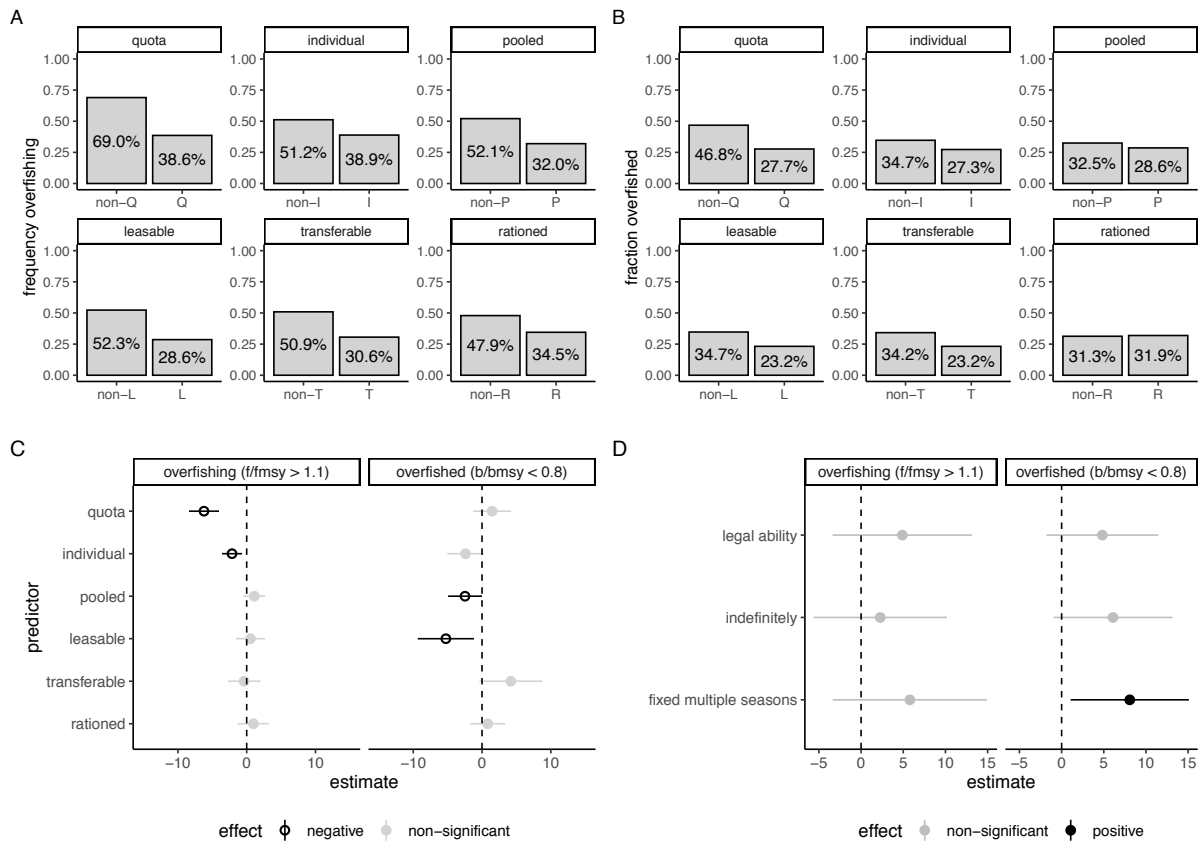


Figure 3. A) Frequency of overfishing ($F/F_{msy} > 1.1$) and B) frequency of overfished observations ($B/B_{msy} < 0.8$) for the attributes I, T, and Q. Each observation is a stock-year combination. C) Mixed-effects results for fisheries opportunity attributes I, T, Q, L, P, and R (Model in Eq.2 in methods section). Negative (black, open circles) effects indicate a reduced probability of overfishing associated to the I and Q (overfishing: 343 stocks with 6803 observations; overfished: 299 stocks with 6875 observations) and a reduced probability of overfished biomass for the L and the P., D). Effects for the duration of fishing opportunities compared to a single season (Model in Eq.3 in methods section). The positive (black, closed circle) value indicates an increased probability of the overfished state for fixed multiple seasons. Table B2 and B3 show further model details (i.e., odds ratios, p-values, random effects and R^2 values).

Because some of the results in Figure 3C could be impacted by effort systems and quota systems differently (i.e., individual quota systems and individual effort systems both impact the estimate for “individual”) we reran the model (i.e., Equation 2 in methods section) using only quota systems. For this subset of observations, we found a significant reduction in the probability for *overfishing* and *overfished* biomass associated to individual allocation (Figure B1) and an increased probability of *overfishing* associated to rationing (Figure B1).

Difference-in-differences: When analysing the change in management system through a difference-in-differences (DiD) analysis, we found a significant reduction in the probability of overfishing associated with the addition of quota limits, individual allocation, leasing, and transferability (Figure 4), with the largest effects found for transferability (effect size = -4.08 and 95% CI [-2.18; -5.97]) and leasing (effect size = -3.51 95% CI [-2.08; -4.95]). Leasing was the only attribute that was associated with a reduced probability of *overfished* biomass (Figure

4). Findings for single attributes in the DiD analysis are likely driven by combinations of attributes (i.e., additions of T and L are almost always accompanied by additions of individual allocation, n=54 of 77 and n=51 of 60), thus the attribute results from the mixed-effects models (Figure 3) are better controlled for the impact of the presence or absence of other attributes. Limiting the DiD analysis to quota systems resulted in similar significant effects for individual allocation, transferability, and leasing for *overfishing* (Figure B3) and leasing for *overfished* biomass (Figure B3).

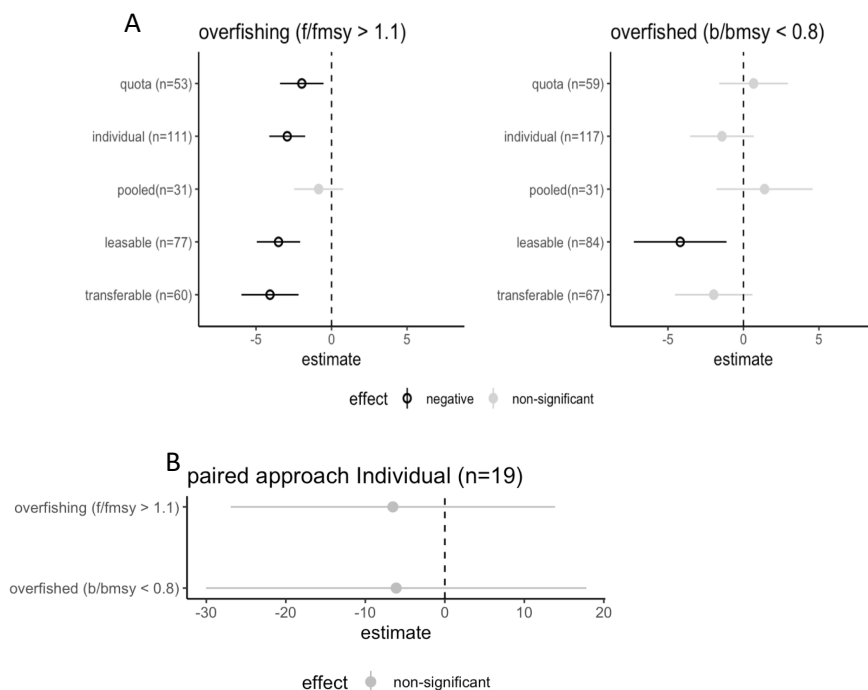


Figure 4. A) Difference-in-differences results for fisheries opportunity attributes Q, I, P, L and T, the attribute R had only 7 observations for the DiD analysis and was not modelled. Negative (black, open circles) effects indicate a reduced probability of *overfishing* associated to the Q, I, T and L and a reduced probability of *overfished* biomass for the L. The number of treatment stocks are noted between brackets for each of the tests (Model in Eq.4 in methods section). **B)** Difference in differences results for fisheries opportunity attribute I and *overfishing* and *overfished* outcomes, for fisheries that were previously under pooled quota management, paired to similar fisheries that remain under pooled quota management (Model in Eq.5 in methods section). Table B4 – B5 show further model details (i.e., odds ratios, p-values, random effects and R^2 values).

Refining the DiD approach to 19 paired treatment and control fisheries, where treatment fisheries transitioned from pooled quota (with or without rationing) to individually allocated quota revealed no significant change in the probability of *overfishing* or *overfished* outcomes (Figure 4B). Confidence intervals were very wide for this result, indicating considerable uncertainty.

Sensitivity test using alternative thresholds for overfishing and overfished indicators

Applying alternative sustainability thresholds (*high overfishing*: $F/F_{msy} > 1.5$; *highly overfished*: $B/B_{msy} < 0.5$) resulted in some changes as more systems recorded a significant effect (Figure B3). Whereas IRQP, IE, SGQP, and unregulated fisheries did not have an effect at the original *overfishing* threshold (Figure 2C), these systems were associated with a reduced probability of *high overfishing* (i.e., all other management systems outperformed TE with regards to *high overfishing*, Figure B3). In general, effect sizes were larger for quota systems. IE systems were associated with an increased probability of *highly overfished* biomass (Figure B3).

At the attribute level, the results were largely unchanged when alternative sustainability thresholds were applied (i.e., a reduced probability of *high overfishing* with individual allocation and quota limits) and the effect sizes increased (Figure B4A). The significant reduction in the probability of *overfished* biomass associated with leasing and pooling no longer held at the *highly overfished* threshold (Figure B4A). The lack of effect for duration remained unchanged (Figure B4B).

The results from the DiD analysis were also similar under the alternative sustainability thresholds with a few exceptions (Figure B5A). The associated reduction in *overfishing* was also found for the same attributes (Q, I, L and T) for *high overfishing*, although effect sizes shifted somewhat and the effect for T was notably smaller for *high overfishing* while the effect for Q was notably larger. While P was not associated with a significant reduction in *overfishing*, it was associated with a significant reduction in *high overfishing*. For *highly overfished* biomass, we found no significant result from the DiD analysis (Figure B5A) as the association between leasing and *overfished* was not robust to the application of the alternative threshold. Again, no significant effects were found in the DiD approach with 19 paired treatment and control fisheries (Figure B5B).

Sensitivity test analysing indicators as continuous variables

Several quota systems reduced the predicted average F/F_{msy} compared to TE (TQP, IQ and ITQ) with the largest effect found for IQ systems (effect size = -0.46, 95% CI [-0.65; -0.27], Figure B6), indicating that the results from the mixed-effects models (Figure 2C) were largely robust to modelling the probability of *overfishing* as continuous F/F_{msy} . ILQ was the only system where the average predicted F/F_{msy} was not lower compared to TE despite having a reduced probability of *overfishing* (and *high overfishing*) occurring. We found no significant

effect of management systems on continuous B/Bmsy (Figure B6), thus the reduced probability of *overfished* biomass associated with ILQ (Figure 2C) was not reflected in the average predicted B/Bmsy.

At the attribute level, the largest reduction in average F/Fmsy was found for quota limits. As quota limits were also associated to the largest reduction in the probability of *overfishing* and *high overfishing*, we believe this result is highly robust (Figure B7A). The significant result found for individual allocation and its association to a reduced probability of *overfishing* and *high overfishing* was not confirmed by the predicted average F/Fmsy. Contrary to the results for the *overfishing* and *high overfishing* thresholds, we found that leasing significantly increased mean predicted F/Fmsy and that transferability significantly decreased mean predicted F/Fmsy (Figure B7A). We found no significant effect of attributes on continuous B/Bmsy, demonstrating that the results found for leasing and pooling were not very robust to assumptions as both of the sensitivity tests did not confirm the results found for *overfished* biomass (Figure 3). The lack of effect for duration remained unchanged for both continuous F/Fmsy and continuous B/Bmsy (Figure B7B).

Applying the same sensitivity test to the DiD analysis, we found a significantly lower F/Fmsy with the addition of quota limits and individual allocation. In contrast to both *overfishing* and *high overfishing*, no significant results were found for transferability and leasing (Figure B8A). No management system attribute impacted the estimated mean B/Bmsy significantly in the DiD analysis (Figure B8B).

4. Discussion

We set out to understand the degree to which fishery management systems affect sustainable fishing. Following the existing literature, a focus was given to I, T, and Q attributes as well as their duration.

After classifying the management systems that govern hundreds of fisheries around the world, we found that management systems using quota limits, particularly those allocated individually (IQ, ILQ, ITQ), were associated with a reduced probability of *overfishing* compared to TE management. ILQ was the only system associated with a reduction in the probability of stocks being *overfished*, albeit with considerable uncertainty.

Disentangling the effects of I, T, and Q attributes, we found that Q and I were associated with large reductions in the probability of *overfishing*, with the largest effect found for Q, and that these effects were stronger when we applied an alternative threshold for overfishing (i.e., *high overfishing*, $F/F_{msy} > 1.5$). The result for Q was also confirmed by a predicted average reduction in F/F_{msy} as a continuous variable, unlike the result for I (Figure B7), making our findings for Q the most robust to modelling assumptions. These results (for Q and I) were however not reflected in biomass indicators, where P and L systems were associated with a reduced probability of *overfished* biomass. In the DiD analysis, Q and I were again associated with a reduction in *overfishing* (and *high overfishing*), as were T and L, and L was also associated with a reduction in *overfished* biomass.

From these results, we conclude that quota systems tend to outperform effort systems in terms of delivering sustainable fishing, and that individual systems tend to outperform systems with total, pooled limits. The result for individual allocation, however, seems to be largely driven by individual quota systems (I+Q, Figure 2) and is thus not entirely independent (i.e., I acts in interaction with Q). Moreover, these results are not reflected in reduced probabilities of *overfished* biomass, where we only found an association for L and a weak association for pooled quota limits (which is in fact the opposite of individual allocation).

We used three main modelling approaches (mixed-effects models for systems and attributes and DiD for systems where attributes changed) for the main analysis as well as for two alternative thresholds for the indicators of sustainability (*overfishing* and *overfished*). To test the robustness of our results we complemented these analyses with an alternative measurement approach (the continuous variable) and two additional sensitivity analyses (Table 2). In total, we generated 10 sets of main results and 26 sensitivity tests. With such a large number of tests it is not surprising that results were not totally uniform for a management system or attribute. This was especially the case for *overfished* biomass where results were generally not very robust to modelling assumptions. However, there was a high-level consistency in the study results for *overfishing*, especially regarding our findings for quota limits, and to a slightly lesser extent for individual allocation (Table 2).

Table 2: Summary of results across the main test and sensitivity tests. Results that are statistically significant are in bolded text where green signifies a decrease in the probability of overfishing or overfished biomass and red signifies an increase. Results that are not statically significant are in bracketed grey text.

Test	Metric of sustainability	Systems	Attributes	Duration	DiD attributes	DiD paired approach
Main test	<i>Overfishing</i>	ITQ IQ ILQ TQP (U IRQP IE)	Q I (T L R P)	(D)	T L I Q (P)	(I)
	<i>Overfished</i>	ILQ (ITQ IQ TQP IRQP U IE)	P, L (I R Q T)	Multiple*	L (I T Q P)	(I)
Only quota systems	<i>Overfishing</i>		IR* (T P L)		I T, L	(I)
	<i>Overfished</i>		IP (L R T)		(I, T, L)	(I)
High Overfishing /Overfished	<i>Overfishing</i>	ITQ IQ ILQ IRQP TQP IE U	Q I (T L R P)	(D)	P Q I L T (P)	(I)
	<i>Overfished</i>	IE* (ILQ IQ ITQ TQP IRQP U)	(Q I T L R P)	(D)	(Q I T L P)	(I)
Continuous variable	<i>F/F_{msy}</i>	IQ ITQ TQP (ILQ U IRQP IE)	Q T L* (I R P)	(D)	Q I (T L P)	(I)
	<i>B/B_{msy}</i>	(ITQ IQ TQP IRQP U IE ILQ)	(Q I T L R P)	(D)	(Q I T L P)	(I)

Quota limits may contribute to fisheries sustainability through their direct link to fisheries mortality (i.e., closing a fishery when the quota has been fished), while effort limits have greater uncertainty in determining their appropriate level³⁶. Moreover, when effort limits are used, fishers can invest in greater efficiencies in catch and mortality per unit of restricted effort (i.e., technological creep or input substitution) which severely complicates the setting of effort limits at sustainable levels^{13,14}.

The reduced probability of *overfishing* in individual systems could be caused by the elimination of the race to fish in individual systems¹⁷, which may result in a more targeted fishery and a reduced need to discard catches^{18,37,38}. It may also result in catches that are lower compared to total allowable catches¹⁰. Longer fishing seasons may aid enforcement (e.g. in a fishery with a very short season it may be more difficult for coastguards to monitor over-quota catches or

illegal discarding)³⁷, as would the accountability of individual allocations as these are held (and exceeded) by a fisher or a company rather than the entire fleet.

The leasing of fishing opportunities produced conflicting results with leasability somewhat associated with a lower probability of *overfished* biomass but also with a significantly higher fishing mortality. This may result from reduced incentives for discarding when allowing leasing and better catch to quota matching especially in multi-species fisheries (which could lead to higher average fishing mortality without resulting in overfishing).³⁹ This finding warrants further research into the mechanistic links between management regimes and marine population biomass levels.

We found little effect for the transferability (only the results of the DiD analysis, in which it was difficult to disentangle the effect of individual allocation) of fishing opportunities and no effect for their duration, which suggests that the casual mechanisms underlying our findings for individual allocation may not be related to secure property rights in fisheries or the use of market-based systems despite both hypothesis featuring in previous literature^{9,17}.

In the first major empirical study, Costello et al. (2008)⁹ found that ‘catch shares’ (specifically ITQs) prevented fisheries collapse, defined as landings below 10 percent of historical levels. While this study was the first of its kind, it suffered from several shortcomings such as the fact that control fisheries were not classified and the comparison group consisted of all non-ITQ fisheries including many unregulated fisheries. With these shortcomings, it is impossible to disentangle whether a reduced probability of collapse was due to I, T, or Q attributes³. In addition, it has been demonstrated that landings data, the proxy used for sustainable fishing, is a poor indicator of stock status²⁴. Subsequent studies have nuanced these results. For instance, a subsequent study by the same authors²⁵ addressed some of the issues by investigating the impact of ITQs on fisheries that already had quota limits in place, and found that effects were still present, although weaker, than in the earlier study (Table 1). More recently, studies have found mixed results for the sustainability benefits of management systems (Table 1), although the few studies that have analysed specific system attributes have consistently found that Q improves sustainable fishing, little or no effect for I, and no consistent effect for either T or D (Table 1). Our findings are similar, but more robust as we studied a longer time-span and used a more detailed classification scheme²⁷. Previous studies have not distinguished between temporal and permanent transferability, and this study is the first to indicate an association between reduced *overfished* biomass and the leasing of quota.

While our study addresses many of the confounding issues in previous literature, several limitations remain. First, we cannot guarantee that our control and treatment fisheries are similar, for example regional circumstances may differ even for adjacent regions⁴⁰, or that fisheries undergoing management change may undergo transitions due to a current or recent fisheries collapse^{19,30}. Second, the scope of this study is limited to governmental policy, and thus in our classification method we relied on the legal definitions of fisheries management systems. Systems may differ from what is described on paper or may develop important attributes in parallel to the governmental system (e.g. producer organisations and fishing co-operatives may pool fishing opportunities that were initially individually allocated). Similarly, the legal definitions of duration may differ from the perceived duration of fishing opportunities based on historical precedent (although our result for duration based on legal definitions aligns with previous research using perceived duration²⁸). More broadly, differentiating between systems as defined by policy and systems as they operate in practice is one area for future research and even further nuance in studying fishing opportunities.

Based on our methodology and the new dataset on fisheries management systems we compiled, we found evidence that both Q and I attributes were associated with a reduced probability of *overfishing* and that the L attribute was associated with a reduced probability of *overfished* biomass. The effect of different management attributes on sustainable fishing was not ubiquitous, however, as the findings for I and Q were not reflected in the probability of a stock being *overfished* and we found no benefit for stocks already under quota transitioning to individual quota or individual transferable quota when we matched these to control fisheries that continued to use pooled quota. Whereas some previous studies have emphasised that market-based systems (i.e., the presence of transferability) or those with strong property rights (i.e., a long duration) are associated with sustainable fishing, these benefits are small or insignificant once proper controls for other system attributes are introduced. These results highlight the importance of considering all attributes of institutional design in the governance of common pool resources.

Data availability: The dataset underlying this study is stored online with open access for further verification and use (fishing-opportunities-database: <https://docs.google.com/spreadsheets/d/1UaKeXxEfVYCp5xzZwOHAnIRf1UzE484G9k1YL4SaynM/edit#gid=1387127720>). Stock data of RAM legacy database v4.491 are publicly available here: <https://www.ramlegacy.org/>. Classification data used for the final analyses will be presented on a personal github account after peer-review.

Code availability: Scripts for the main analyses will be published on <https://github.com>.

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Appendix A. Management system definitions and DiD pairs

Table A1: Management system definitions, for the 13 final management systems and the 4 types of duration of harvesting rights.

Management system	Definition
Individual Transferable Quota	A quantity limit on catches/landings is allocated for the exclusive use of a vessel/license and can be sold to a different vessel/license (leasing is also permitted, quota swapping may also be permitted).
Individual Leasable Quota	A quantity limit on catches/landings is allocated for the exclusive use of a vessel/license and can be sold to a different vessel/license for a fixed time period only (quota swapping may also be permitted but permanent transfer is not).
Individual Quota	A quantity limit on catches/landings is allocated for the exclusive use of a vessel/license and can be swapped for other quota but cannot be leased or permanently sold (i.e. monetary transfers).
Self-Governed Quota Pool(s)	A quantity limit on catches/landings is allocated to a group of vessels/licenses for joint use. The pool is managed by its membership. Fishers have no individual holdings to enter/exit the pool.
Total Quota Pool	A quantity limit on catches/landings is allocated to a group of vessels/licenses for joint use. The pool is managed by the government.
Individually-Rationed Quota Pool	A quantity limit on catches/landings is allocated to a group of vessels/licenses for joint use. These limits are allocated to individual vessels/licenses for exclusive use in multiple time periods within a fishing season (e.g. daily, weekly or monthly limits).
Rationed Quota Pool	A quantity limit on catches/landings is allocated to a group of vessels/licenses for joint use. These limits are administered in multiple time periods within a fishing season (e.g. weekly or monthly vessel limits).
Rationed Individual Quota	A quantity limit on catches/landings is allocated for the exclusive use of a vessel/license. These limits are administered in multiple time periods within a fishing season (e.g. weekly or monthly vessel limits). There is no total quota limit that can be reached, meaning there is no pool and each vessel/license limit is independent.
Individual Transferable Effort	A limit on fisheries inputs (e.g. days at sea, area/territory, vessel capacity) is allocated for the exclusive use of a vessel/license and can be sold to a different vessel/license.
Individual Effort	A limit on fisheries inputs (e.g. days at sea, area/territory, vessel capacity) is allocated for the exclusive use of a vessel/license.
Total Effort	A limit on fisheries inputs (e.g. number of vessels, days at sea, vessel capacity, seasonal closure, spatial closure) is set for the entire fishery.
Unregulated	There is no fisheries legislation limiting the amount of fishing pressure.
Moratorium	There is a ban on fishing
Duration	Definition

Indefinite	In fisheries legislation it is specified that fishing opportunities are held permanently. The size of the fishing opportunity may change as the total changes (e.g. 3% of 100 may become 3% of 150), but fishing opportunity does not change as a relative share of the total. Fishing licenses may be subject to change at a different interval.
Fixed multiple seasons	In fisheries legislation it is specified that fishing opportunities are held for a fixed period that spans multiple fishing seasons (e.g. 10 years) after which the relative shares of fishing opportunities may be revised. Fishing licenses may be subject to change at a different interval.
One season	In fisheries legislation it is specified that fishing opportunities are held for one season (e.g. one year) after which the relative shares of fishing opportunities may be revised. Fisheries legislation requires an active decision each year on allocations (i.e. the default is not necessarily the same allocation as the previous year). Fishing licenses may be subject to change at a different interval.
Legal ability	In fisheries legislation it is specified that the fisheries manager reserves the right to revise the relative shares of fishing opportunities, but as the duration of the fishing opportunities is not specified this can take place at any time. Fisheries legislation does not require an active decision each year on allocations (i.e. the default is the same allocation as the previous year). Fishing licenses may be subject to change at a different interval.

Table A2: Treatment and control stocks for paired difference in difference analysis.

Impact/ control	stock name RAM	Species name	year of quota	year of IQ	minimum year used	final year used	IQ type
impact	Sablefish Eastern Bering Sea / Aleutian Islands / Gulf of Alaska	<i>Anoplopoma fimbria</i>	1977	1995	1982	2011	ITQ
control	Sablefish Pacific Coast	<i>Anoplopoma fimbria</i>	1982	2011	1982	2011	ITQ
impact	Walleye pollock Eastern Bering Sea	<i>Gadus chalcogrammus</i>	1977	2000	1982	2018	ILQ
control	Walleye pollock Gulf of Alaska	<i>Gadus chalcogrammus</i>	1977	no IQ	1982	2018	ILQ
impact	Pacific cod West Coast of Vancouver Island	<i>Gadus macrocephalus</i>	1979	1997	1982	2008	ITQ
control	Pacific cod Bering Sea	<i>Gadus macrocephalus</i>	1977	2008	1982	2008	ITQ
impact	Pacific cod Hecate Strait	<i>Gadus macrocephalus</i>	1992	1997	1992	2018	ITQ
control	Pacific cod Gulf of Alaska	<i>Gadus macrocephalus</i>	1977	no IQ	1992	2018	ITQ
impact	Tilefish Mid-Atlantic Coast	<i>Lopholatilus chamaeleonticeps</i>	2001	2009	1994	2018	ITQ
control	Tilefish Southern Atlantic coast	<i>Lopholatilus chamaeleonticeps</i>	1994	no IQ	1994	2018	ITQ
impact	Red king crab Bristol Bay	<i>Paralithodes camtschaticus</i>	1980	2005	1982	2015	ITQ
control	Red king crab Norton Sound	<i>Paralithodes camtschaticus</i>	1978	no IQ	1982	2015	ITQ
impact	Pacific ocean perch West Coast of Vancouver Island	<i>Sebastes alutus</i>	1979	1997	1982	2011	ITQ
control	Pacific ocean perch Pacific Coast	<i>Sebastes alutus</i>	1982	2011	1982	2011	ITQ
impact	Norway lobster Labadie, Jones and Cockburn (FU 20-21)	<i>Nephrops norvegicus</i>	1980	1997	1982	2018	IQ
control	Norway lobster Smalls (FU 22)	<i>Nephrops norvegicus</i>	1980	no IQ	1982	2018	IQ
impact	Red snapper Gulf of Mexico	<i>Lutjanus campechanus</i>	1990	2010	2006	2018	ITQ
control	Vermilion snapper Southern Atlantic coast	<i>Rhomboplites aurorubens</i>	2006	no IQ	2006	2018	ITQ
impact	Rock sole Hecate Strait	<i>Lepidopsetta bilineata</i>	1980	1997	1982	2007	ITQ
control	Northern rock sole Eastern Bering Sea and Aleutian Islands	<i>Lepidopsetta bilineata</i>	1980	2007	1982	2007	ITQ
impact	Arrowtooth flounder Pacific Coast	<i>Atheresthes stomias</i>	1983	2011	1983	2018	ITQ
control	Arrowtooth flounder Gulf of Alaska	<i>Atheresthes stomias</i>	1978	no IQ	1983	2018	ITQ

impact	Dover sole Pacific Coast	<i>Microstomus pacificus</i>	1983	2011	1983	2018	ITQ
control	Dover sole Gulf of Alaska	<i>Microstomus pacificus</i>	1978	no IQ	1983	2018	ITQ
impact	Megrim ICES 7-8abd	<i>Lepidorhombus whiffiagonis</i>	1980	1997	1982	2018	IQ
control	Fourspotted megrim ICES 8c-9a	<i>Lepidorhombus boscii</i>	1981	no IQ	1982	2018	IQ
impact	Pacific ocean perch Haida Gwaii	<i>Sebastes alutus</i>	1979	1997	1982	2008	ITQ
control	Pacific ocean perch Eastern Bering Sea and Aleutian Islands	<i>Sebastes alutus</i>	1980	2008	1982	2008	ITQ
impact	Atlantic cod North-East Arctic (Norwegian coastal waters)	<i>Gadus morhua</i>	1977	1990	1985	2018	IQ
control	Atlantic cod NAFO 1f and ICES 14	<i>Gadus morhua</i>	1985	no IQ	1985	2018	IQ
impact	Tilefish Gulf of Mexico	<i>Lopholatilus chamaeleonticeps</i>	2004	2010	1994	2018	ITQ
control	Tilefish Southern Atlantic coast	<i>Lopholatilus chamaeleonticeps</i>	1994	no IQ	1994	2018	ITQ
impact	Walleye pollock Aleutian Islands	<i>Theragra chalcogramma</i>	1980	2000	1982	2018	ILQ
control	Walleye pollock Gulf of Alaska	<i>Theragra chalcogramma</i>	1977	no IQ	1982	2018	ILQ
impact	Atlantic cod North-East Arctic	<i>Gadus morhua</i>	1977	1990	1985	2018	IQ
control	Atlantic cod NAFO 1f and ICES 14	<i>Gadus morhua</i>	1985	no IQ	1985	2018	IQ
impact	Northern rockfish Gulf of Alaska	<i>Sebastes polyspinis</i>	1980	2007	1980	2018	ITQ
control	Northern rockfish Bering Sea and Aleutian Islands	<i>Sebastes polyspinis</i>	1980	no IQ	1980	2018	ITQ

Table A3: Number of stocks under the different management regimes with overlap with time-series of F/Fmsy (n stocks F/Fmsy) and timeseries of B/Bmsy (n stocks B/Bmsy), mean and median F/Fmsy and B/Bmsy under the different management regimes.

Management	n stocks F/Fmsy	mean F/Fmsy	median F/Fmsy	n stocks B/Bmsy	mean B/Bmsy	median B/Bmsy
Individual transferable quota (ITQ)	105	1.14	0.66	117	1.41	1.26
Individual leasable quota (ILQ)	26	1.30	0.72	27	1.34	1.11
Individual quota (IQ)	31	1.26	1.10	32	1.53	1.28
Self-governed quota pool (SGQP)	4	1.36	1.15	4	1.05	1.08
Total quota pool (TQP)	74	0.92	0.67	76	1.51	1.37
Individually rationed quota pool (IRQP)	48	0.89	0.72	44	1.30	1.17
Rationed quota pool (RQP)	1	2.05	2.09	1	1.10	1.15
Rationed individual quota (RIQ)	2	2.03	2.13	2	1.34	1.42
Individual transferable effort (ITE)	3	2.23	2.22	3	1.49	1.20
Individual effort (IE)	18	2.09	1.49	18	0.64	0.46
Total effort (TE)	134	2.31	1.58	86	1.15	0.88
Unregulated (U)	18	1.98	1.61	18	0.87	0.86

Appendix B. Additional tables, figures and sensitivity analyses

Table B1. Odds ratios, confidence intervals, p-values, model random effects and R² values for models predicting the impact of management regimes on *overfishing* (F/Fmsy >1.1) and *overfished* (B/Bmsy < 0.8) outcome variables.

<i>Predictors</i>	overfishing			overfished		
	<i>Odds Ratios</i>	<i>CI</i>	<i>p</i>	<i>Odds Ratios</i>	<i>CI</i>	<i>p</i>
(Intercept)	0.62	0.01 – 39.49	0.821	0.00	0.00 – 0.00	< 0.001
management [TQP]	0.03	0.00 – 0.19	< 0.001	0.40	0.02 – 6.75	0.523
management [IE]	0.37	0.04 – 3.06	0.356	4.91	0.08 – 320.46	0.455
management [ILQ]	0.01	0.00 – 0.19	0.002	0.00	0.00 – 0.12	0.013
management [IQ]	0.00	0.00 – 0.08	< 0.001	0.28	0.01 – 13.69	0.524
management [IRQP]	0.14	0.01 – 2.07	0.152	0.87	0.04 – 21.25	0.931
management [ITQ]	0.00	0.00 – 0.03	< 0.001	0.11	0.01 – 2.36	0.159
management [U]	0.08	0.01 – 1.05	0.054	2.41	0.04 – 153.43	0.679
Random Effects						
σ^2	3.29			3.29		
τ_{00}	14.03	region		*		
	53.97	stocklong		550.01	stocklong	
τ_{11}	53.97	stocklong.year1991 - 2017		550.01	stocklong.year1991-2018	
ρ_{01}	0.96	stocklong.year1991		0.97		
	0.92	stocklong.year1992		0.94		
	0.89	stocklong.year1993		0.90		
	0.85	stocklong.year1994		0.87		
	0.82	stocklong.year1995		0.85		
	0.79	stocklong.year1996		0.82		
	0.75	stocklong.year1997		0.79		
	0.72	stocklong.year1998		0.76		
	0.70	stocklong.year1999		0.74		
	0.67	stocklong.year2000		0.71		
	0.64	stocklong.year2001		0.69		
	0.62	stocklong.year2002		0.67		
	0.59	stocklong.year2003		0.65		
	0.57	stocklong.year2004		0.62		
	0.55	stocklong.year2005		0.60		
	0.53	stocklong.year2006		0.58		
	0.50	stocklong.year2007		0.57		
	0.48	stocklong.year2008		0.55		
	0.47	stocklong.year2009		0.53		
	0.45	stocklong.year2010		0.51		
	0.43	stocklong.year2011		0.49		
ρ_{01}	0.41	stocklong.year2012		0.48		
	0.40	stocklong.year2013		0.46		

	0.38	stocklong.year2014	0.45
	0.37	stocklong.year2015	0.43
	0.35	stocklong.year2016	0.42
	0.34	stocklong.year2017	0.40
			0.39
ICC	0.85		0.00
N	9	region	256
	293	stocklong	
<hr/>			
Observations	5218		5280
Marginal R ² / Conditional R ²	0.192 / 0.882		0.593 / 0.593

Table B2. Odds ratios, confidence intervals, p-values, model random effects and R² values for models predicting the impact of management system attributes on *overfishing* (F/Fmsy > 1.1) and *overfished* (B/Bmsy < 0.8) outcome variables.

<i>Predictors</i>	overfishing			bbmsy_overfished		
	<i>Odds Ratios</i>	<i>CI</i>	<i>p</i>	<i>Odds Ratios</i>	<i>CI</i>	<i>p</i>
(Intercept)	25.18	1.62 – 392.37	0.021	0.00	0.00 – 0.00	<0.001
rationed [1]	4.02	0.40 – 40.51	0.238	2.30	0.18 – 29.44	0.522
pooled [1]	2.24	0.45 – 11.21	0.325	0.08	0.01 – 0.99	0.049
leasable [1]	1.51	0.18 – 12.90	0.705	0.01	0.00 – 0.31	0.012
transferable [1]	0.77	0.07 – 8.32	0.831	65.73	0.70 – 6174.61	0.071
individual [1]	0.12	0.03 – 0.51	0.004	0.09	0.01 – 1.28	0.076
quota [1]	0.00	0.00 – 0.02	<0.001	4.33	0.28 – 67.63	0.296
Random Effects						
σ^2	3.29			3.29		
τ_{00}	8.11	region				
	69.65	stocklong		443.23	stocklong	
τ_{11}	69.65	stocklong.year1991-2017		443.23	stocklong.year1991-2018	
ρ_{01}	0.96	stocklong.year1991		0.96		
	0.92	stocklong.year1992		0.92		
	0.89	stocklong.year1993		0.89		
	0.85	stocklong.year1994		0.85		
	0.82	stocklong.year1995		0.82		
	0.79	stocklong.year1996		0.78		
	0.76	stocklong.year1997		0.75		
	0.73	stocklong.year1998		0.72		
	0.70	stocklong.year1999		0.69		
	0.67	stocklong.year2000		0.67		
	0.64	stocklong.year2001		0.64		
	0.62	stocklong.year2002		0.61		
	0.59	stocklong.year2003		0.59		
	0.57	stocklong.year2004		0.57		
	0.55	stocklong.year2005		0.54		
	0.53	stocklong.year2006		0.52		
	0.51	stocklong.year2007		0.50		
	0.49	stocklong.year2008		0.48		
	0.47	stocklong.year2009		0.46		
	0.45	stocklong.year2010		0.44		
	0.43	stocklong.year2011		0.43		
	0.41	stocklong.year2012		0.41		
	0.40	stocklong.year2013		0.39		
	0.38	stocklong.year2014		0.38		
	0.37	stocklong.year2015		0.36		
ρ_{01}	0.35	stocklong.year2016		0.35		

	0.34 _{stocklong_year2017}	0.33
		0.32
ICC	0.71	0.00
N	9 _{region}	299 _{stocklong}
	344 _{stocklong}	
<hr/>		
Observations	6788	6869
Marginal R ² / Conditional R ²	0.437 / 0.838	0.431 / 0.431

Table B3. Odds ratios, confidence intervals, p-values, model random effects and R² values for models predicting the impact of durations of harvesting rights on *overfishing* (F/Fmsy > 1.1) and *overfished* (B/Bmsy < 0.8) outcome variables.

<i>Predictors</i>	overfishing			overfished		
	<i>Odds Ratios</i>	<i>CI</i>	<i>p</i>	<i>Odds Ratios</i>	<i>CI</i>	<i>p</i>
(Intercept)	0.00	0.00 – 0.00	0.001	0.00	0.00 – 0.00	<0.001
multiple seasons	299.65	0.03 – 3093962.90	0.227	3173.33	2.88 – 3492298.31	0.024
indefinitely	9.44	0.00 – 28624.91	0.583	454.08	0.40 – 517772.89	0.088
Legal ability	122.30	0.03 – 537276.77	0.261	123.61	0.17 – 91896.14	0.153
Random Effects						
σ^2	3.29			3.29		
τ_{00}	5.00	region		614.67	stocklong	
	0.13	year				
	492.42	stocklong				
τ_{11}	492.42	stocklong.year1991-2017		614.67	stocklong.year1991-2018	
ρ_{01}	0.97	stocklong.year1991		0.95		
	0.93	stocklong.year1992		0.91		
	0.90	stocklong.year1993		0.86		
	0.87	stocklong.year1994		0.82		
	0.84	stocklong.year1995		0.78		
	0.82	stocklong.year1996		0.74		
	0.79	stocklong.year1997		0.71		
	0.76	stocklong.year1998		0.67		
	0.74	stocklong.year1999		0.64		
	0.71	stocklong.year2000		0.61		
	0.69	stocklong.year2001		0.58		
	0.66	stocklong.year2002		0.55		
	0.64	stocklong.year2003		0.53		
	0.62	stocklong.year2004		0.50		
	0.60	stocklong.year2005		0.48		
	0.58	stocklong.year2006		0.45		
	0.56	stocklong.year2007		0.43		
	0.54	stocklong.year2008		0.41		
	0.52	stocklong.year2009		0.39		
	0.51	stocklong.year2010		0.37		
	0.49	stocklong.year2011		0.35		
	0.47	stocklong.year2012		0.34		
	0.46	stocklong.year2013		0.32		
	0.44	stocklong.year2014		0.30		
	0.43	stocklong.year2015		0.29		
	0.41	stocklong.year2016		0.28		
	0.40	stocklong.year2017		0.26		
				0.25		
ICC	0.61			0.00		

N	7 _{region} 28 _{year} 135 _{stocklong}	149 _{stocklong}
Observations	1999	2368
Marginal R ² / Conditional R ²	0.271 / 0.715	0.571 / 0.571

Table B4. Odds ratios, confidence intervals, p-values, model random effects and R² values for DiD models predicting the impact of management attribute changes of harvesting rights on *overfishing* (F/Fmsy >1.1) outcome variable.

Predictors	overfishing			overfishing			overfishing			overfishing			overfishing		
	Odds Ratio s	CI	p	Odds Ratio s	CI	p	Odds Ratio s	CI	p	Odds Ratio s	CI	p	Odds Ratio s	CI	p
(Intercept)	8.21	0.38 – 176.53	0.179	0.49	0.02 – 10.80	0.654	0.34	0.02 – 6.37	0.468	0.37	0.02 – 6.09	0.488	0.96	0.05 – 19.13	0.976
Q	0.11	0.03 – 0.50	0.004												
I				0.04	0.01 – 0.13	<0.001									
T							0.02	0.00 – 0.11	<0.001						
L										0.03	0.01 – 0.11	<0.001			
P													0.31	0.06 – 1.55	0.155
Random Effects															
σ ²	3.29			3.29			3.29			3.29			3.29		
τ ₀₀	8.82 region			11.97 region			10.44 region			9.23 region			11.60 region		
	41.05 stocklong			63.43 stocklong			63.74 stocklong			63.52 stocklong			58.26 stocklong		
τ ₁₁	41.05 stocklong.year1991-2017			63.43 stocklong.year1991-2017			63.74 stocklong.year1991-2017			63.52 stocklong.year1991-2017			58.26 stocklong.year1991-2017		
ρ ₀₁	0.97 stocklong.year1991			0.97 stocklong.year1991			0.96 stocklong.year1991			0.96 stocklong.year1991			0.97 stocklong.year1991		
	0.94 stocklong.year1992			0.93 stocklong.year1992			0.93 stocklong.year1992			0.93 stocklong.year1992			0.93 stocklong.year1992		
	0.91 stocklong.year1993			0.90 stocklong.year1993			0.89 stocklong.year1993			0.89 stocklong.year1993			0.90 stocklong.year1993		
	0.88 stocklong.year1994			0.87 stocklong.year1994			0.86 stocklong.year1994			0.86 stocklong.year1994			0.87 stocklong.year1994		
	0.85 stocklong.year1995			0.84 stocklong.year1995			0.83 stocklong.year1995			0.83 stocklong.year1995			0.84 stocklong.year1995		
	0.82 stocklong.year1996			0.81 stocklong.year1996			0.80 stocklong.year1996			0.80 stocklong.year1996			0.81 stocklong.year1996		
	0.80 stocklong.year1997			0.78 stocklong.year1997			0.77 stocklong.year1997			0.77 stocklong.year1997			0.79 stocklong.year1997		
	0.77 stocklong.year1998			0.75 stocklong.year1998			0.74 stocklong.year1998			0.74 stocklong.year1998			0.76 stocklong.year1998		
	0.75 stocklong.year1999			0.73 stocklong.year1999			0.72 stocklong.year1999			0.71 stocklong.year1999			0.73 stocklong.year1999		
	0.72 stocklong.year2000			0.70 stocklong.year2000			0.69 stocklong.year2000			0.68 stocklong.year2000			0.71 stocklong.year2000		
	0.70 stocklong.year2001			0.68 stocklong.year2001			0.67 stocklong.year2001			0.66 stocklong.year2001			0.69 stocklong.year2001		
	0.68 stocklong.year2002			0.65 stocklong.year2002			0.64 stocklong.year2002			0.63 stocklong.year2002			0.66 stocklong.year2002		
	0.65 stocklong.year2003			0.63 stocklong.year2003			0.62 stocklong.year2003			0.61 stocklong.year2003			0.64 stocklong.year2003		
	0.63 stocklong.year2004			0.61 stocklong.year2004			0.60 stocklong.year2004			0.59 stocklong.year2004			0.62 stocklong.year2004		
	0.61 stocklong.year2005			0.59 stocklong.year2005			0.57 stocklong.year2005			0.56 stocklong.year2005			0.60 stocklong.year2005		
	0.59 stocklong.year2006			0.57 stocklong.year2006			0.55 stocklong.year2006			0.54 stocklong.year2006			0.58 stocklong.year2006		
	0.57 stocklong.year2007			0.55 stocklong.year2007			0.53 stocklong.year2007			0.52 stocklong.year2007			0.56 stocklong.year2007		
	0.56 stocklong.year2008			0.53 stocklong.year2008			0.51 stocklong.year2008			0.50 stocklong.year2008			0.54 stocklong.year2008		
	0.54 stocklong.year2009			0.51 stocklong.year2009			0.50 stocklong.year2009			0.48 stocklong.year2009			0.52 stocklong.year2009		
	0.52 stocklong.year2010			0.49 stocklong.year2010			0.48 stocklong.year2010			0.47 stocklong.year2010			0.50 stocklong.year2010		
	0.50 stocklong.year2011			0.48 stocklong.year2011			0.46 stocklong.year2011			0.45 stocklong.year2011			0.49 stocklong.year2011		
ρ ₀₁	0.49 stocklong.year2012			0.46 stocklong.year2012			0.44 stocklong.year2012			0.43 stocklong.year2012			0.47 stocklong.year2012		

	0.47 stocklong_year2013	0.44 stocklong_year2013	0.43 stocklong_year2013	0.42 stocklong_year2013	0.45 stocklong_year2013
	0.46 stocklong_year2014	0.43 stocklong_year2014	0.41 stocklong_year2014	0.40 stocklong_year2014	0.44 stocklong_year2014
	0.44 stocklong_year2015	0.41 stocklong_year2015	0.40 stocklong_year2015	0.38 stocklong_year2015	0.42 stocklong_year2015
	0.43 stocklong_year2016	0.40 stocklong_year2016	0.38 stocklong_year2016	0.37 stocklong_year2016	0.41 stocklong_year2016
	0.41 stocklong_year2017	0.38 stocklong_year2017	0.37 stocklong_year2017	0.36 stocklong_year2017	0.40 stocklong_year2017
ICC	0.73	0.78	0.76	0.74	0.78
N	6 region	7 region	7 region	7 region	7 region
	138 stocklong	275 stocklong	294 stocklong	288 stocklong	230 stocklong
Observations	2314	5708	6145	5798	4623
Marginal R ² / Conditional R ²	0.080 / 0.750	0.117 / 0.810	0.117 / 0.788	0.102 / 0.764	0.008 / 0.781

Table B5. Odds ratios, confidence intervals, p-values, model random effects and R² values for DiD models predicting the impact of management attribute changes of harvesting rights on *overfished* (B/Bmsy < 0.8) outcome variable.

Predictors	bbmsy_overfished			bbmsy_overfished			bbmsy_overfished			bbmsy_overfished			bbmsy_overfished		
	Odds Ratio	CI	p	Odds Ratio	CI	p	Odds Ratio	CI	p	Odds Ratio	CI	p	Odds Ratio	CI	p
(Intercept)	0.00	0.00 – 0.05	0.06	0.00	0.00 – 0.00	<0.001	0.00	0.00 – 0.00	<0.001	0.00	0.00 – 0.00	<0.001	0.00	0.00 – 0.00	<0.001
Q	1.94	0.21 – 8.05	0.562												
I				0.24	0.03 – 1.97	0.184									
T							0.10	0.00 – 2.28	0.148						
L										0.03	0.00 – 0.48	0.014			
P													4.00	0.17 – 4.60	0.390
Random Effects															
σ^2	3.29			3.29			3.29			3.29			3.29		
τ_{00}	41.35	region		457.35	stocklong		443.20	stocklong		457.34	stocklong		461.77	stocklong	
	139.88	stocklong													
τ_{11}	139.88	stocklong.year1991-2018		457.35	stocklong.year1991-2018		443.20	stocklong.year1991-2018		457.34	stocklong.year1991-2018		461.77	stocklong.year1991-2018	
ρ_{01}	0.97	stocklong.year1991		0.96			0.96			0.96			0.97		
	0.95	stocklong.year1992		0.93			0.92			0.92			0.93		
	0.92	stocklong.year1993		0.90			0.88			0.88			0.90		
	0.90	stocklong.year1994		0.86			0.85			0.85			0.87		
	0.87	stocklong.year1995		0.83			0.81			0.81			0.84		
	0.85	stocklong.year1996		0.80			0.78			0.78			0.81		
	0.83	stocklong.year1997		0.77			0.75			0.75			0.78		
	0.81	stocklong.year1998		0.74			0.72			0.72			0.76		
	0.78	stocklong.year1999		0.72			0.69			0.69			0.73		
	0.76	stocklong.year2000		0.69			0.66			0.66			0.71		
	0.74	stocklong.year2001		0.67			0.63			0.64			0.68		
	0.72	stocklong.year2002		0.64			0.61			0.61			0.66		
	0.70	stocklong.year2003		0.62			0.58			0.59			0.64		
	0.69	stocklong.year2004		0.60			0.56			0.56			0.62		
	0.67	stocklong.year2005		0.57			0.54			0.54			0.59		
	0.65	stocklong.year2006		0.55			0.51			0.52			0.57		
	0.63	stocklong.year2007		0.53			0.49			0.50			0.56		
	0.62	stocklong.year2008		0.51			0.47			0.48			0.54		
	0.60	stocklong.year2009		0.50			0.45			0.46			0.52		
	0.58	stocklong.year2010		0.48			0.44			0.44			0.50		
	0.57	stocklong.year2011		0.46			0.42			0.42			0.48		

ρ ₀₁	0.55	stocklong.year2012	0.44	0.40	0.40	0.47
	0.54	stocklong.year2013	0.43	0.38	0.39	0.45
	0.52	stocklong.year2014	0.41	0.37	0.37	0.44
	0.51	stocklong.year2015	0.40	0.35	0.36	0.42
	0.50	stocklong.year2016	0.38	0.34	0.34	0.41
	0.48	stocklong.year2017	0.37	0.33	0.33	0.39
	0.47	stocklong.year2018	0.36	0.31	0.32	0.38
	ICC	0.94		0.00	0.00	0.00
N	9	region	224	stocklong	245	stocklong
	93	stocklong			239	stocklong

Observations	2015	5610	6102	5708	4594
Marginal R ² / Conditional R ²	0.002 / 0.939	0.115 / 0.115	0.173 / 0.173	0.343 / 0.343	0.051 / 0.051

Table B6 Odds ratios, confidence intervals, p-values, model random effects and R² values for DiD models predicting the impact of individual allocation of harvesting rights (under quota, paired to control stocks) on *overfished* (B/Bmsy < 0.8) outcome variable.

<i>Predictors</i>	overfishing			overfished		
	<i>Odds Ratios</i>	<i>CI</i>	<i>p</i>	<i>Odds Ratios</i>	<i>CI</i>	<i>p</i>
(Intercept)	0.00	0.00 – 0.00	< 0.001	0.00	0.00 – 0.03	0.020
Before after	0.01	0.00 – 69232331.78	0.671	2.20	0.01 – 669.99	0.786
Treatment	2.88	0.00 – 3657.26	0.771	3570.68	0.00 – 48096963387.26	0.329
Treatment*After	0.00	0.00 – 1042896.88	0.530	0.00	0.00 – 53937841.19	0.616
Random Effects						
σ^2	3.29			3.29		
τ_{00}	1065.94	<small>stocklong</small>		1135.57	<small>stocklong</small>	
τ_{11}	1065.94	<small>stocklong.year1985-2017</small>		1135.57	<small>stocklong.year1985-2017</small>	
ρ_{01}	0.97			0.94		
	0.95			0.89		
	0.92			0.84		
	0.89			0.80		
	0.87			0.75		
	0.84			0.71		
	0.82			0.67		
	0.80			0.64		
	0.78			0.60		
	0.76			0.57		
	0.73			0.54		
	0.71			0.51		
	0.69			0.48		
	0.67			0.45		
	0.66			0.43		
	0.64			0.40		
	0.62			0.38		
	0.60			0.36		
	0.59			0.34		
	0.57			0.32		
	0.55			0.30		
	0.54			0.29		
	0.52			0.27		
	0.51			0.26		
	0.50			0.24		
	0.48			0.23		
	0.47			0.22		
	0.46			0.20		
	0.44			0.19		
	0.43			0.18		
	0.42			0.17		
	0.41			0.16		

ρ_{01}	0.40	0.15
		0.15
ICC	0.00	0.00
N	32 <small>stocklong</small>	32 <small>stocklong</small>
Observations	853	938
Marginal R ² / Conditional R ²	0.861 / 0.861	0.759 / 0.759

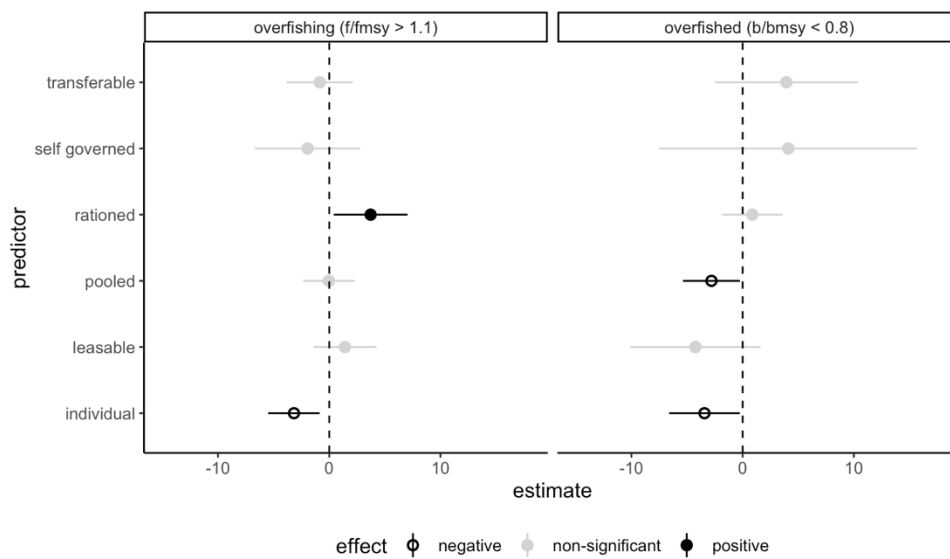


Figure B1. Estimates and 95 percent confidence intervals of management system attributes for quota systems. Black values indicate significant values, open circles indicate negative effects, i.e. the management system attribute reduces the probability of the outcome variable and closed black circles indicate positive effects i.e. the management system attribute increases the probability of the outcome variable, for example individual allocation reduces the probability of overfishing and overfished biomass compared to pooled allocation.

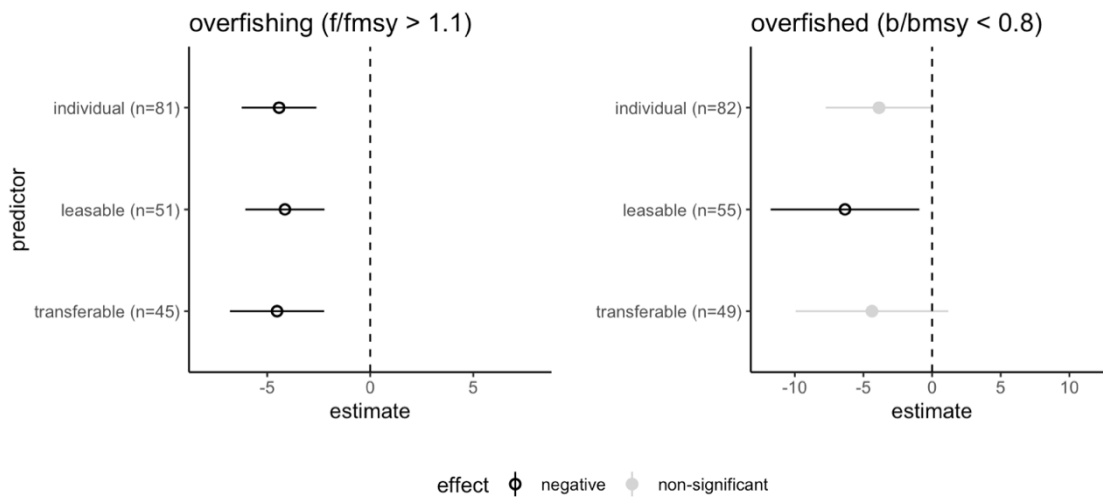


Figure B2. Difference in differences results for fisheries opportunity attributes I, T, and L for quota systems only. Negative (black, open circles) effects indicate a reduced probability of *overfishing* associated to the I, L and T and a reduced probability of *overfished* biomass for the L. Amount of treatment stocks are noted between brackets for each of the tests.

Sensitivity: alternative thresholds for overfishing and overfished

Sensitivity analysis using alternative thresholds for *overfishing* and *overfished*, i.e. *high overfishing* ($f/fmsy > 1.5$) and *highly overfished* ($b/bmsy < 0.5$). A few of the models failed to converge but did converge without inclusion of the random effect for fishery type, a very low amount of variance was explained by that random variable so we do not expect that this impacted the result greatly. This random variable was removed for the model for *high overfishing* ($f/fmsy > 1.5$) and the different management systems (presented in Figure B2) and for the model predicting *highly overfished* ($b/bmsy < 0.5$) by the different fishing opportunity attributes (presented in Figure B3A).

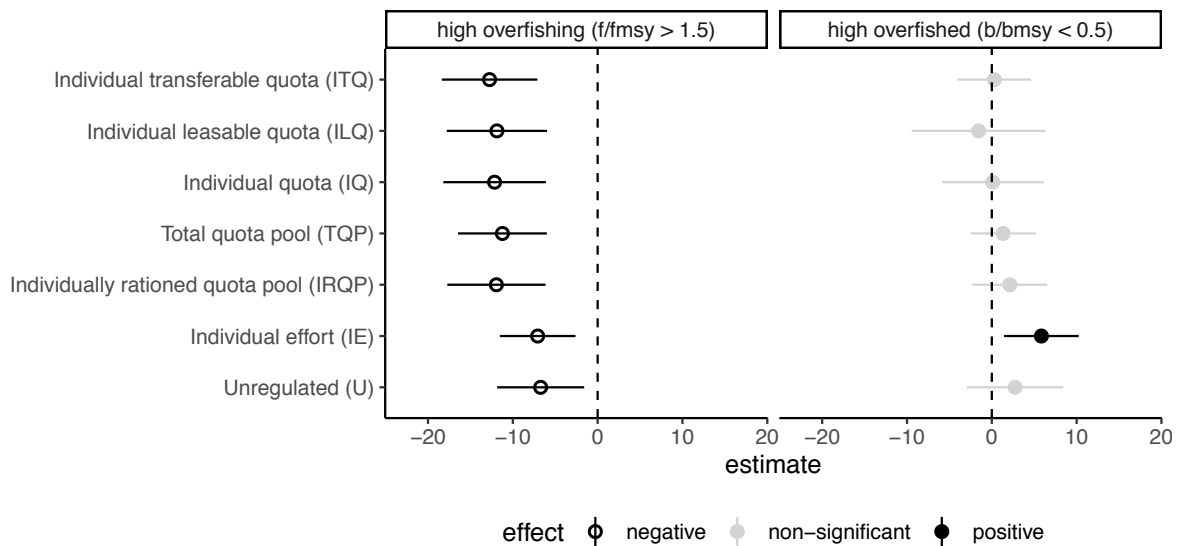


Figure B3. Estimates and 95 percent confidence intervals of management systems impact on high overfishing and high overfished compared to TE. Black values indicate significant values, open circles indicate negative effects, i.e. the management system reduces the probability of the outcome variable and closed black circles indicate positive effects i.e. the management system increases the probability of the outcome variable, for example all other management systems reduced the probability of high overfishing compared to TE.

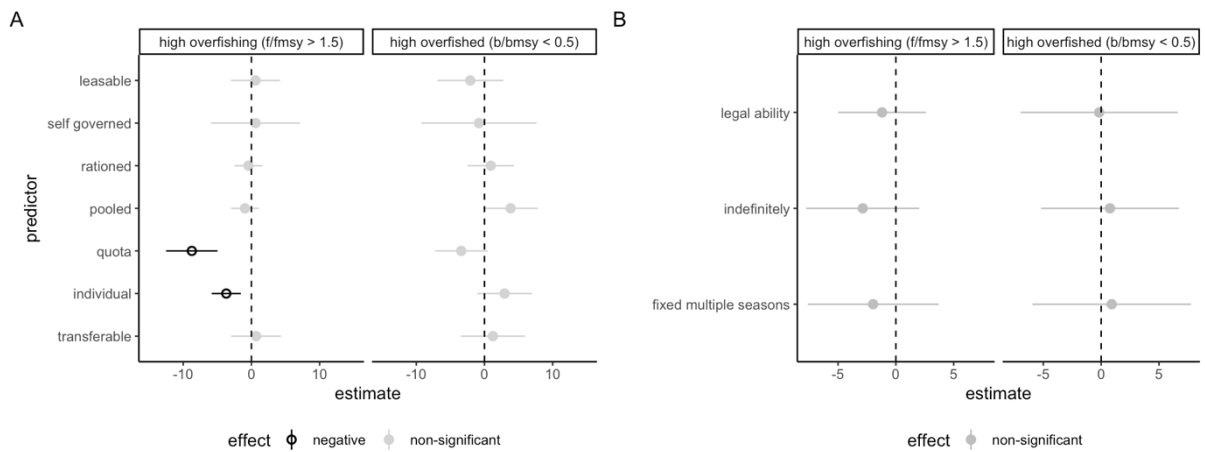


Figure B4. A) Mixed-effects results for the fishing opportunity attributes and their association to alternative thresholds of *overfishing* and *overfished* (i.e. *high overfishing* and *high overfished*) outcome variables. Negative (black, open circles) effects indicate a reduced probability of high overfishing for I and Q (overfishing: 343 stocks with 6803 observations; overfished: 299 stocks with 6875 observations). B) Effects for the duration of fishing opportunities compared to a single season for *high overfishing* and *high overfished*.

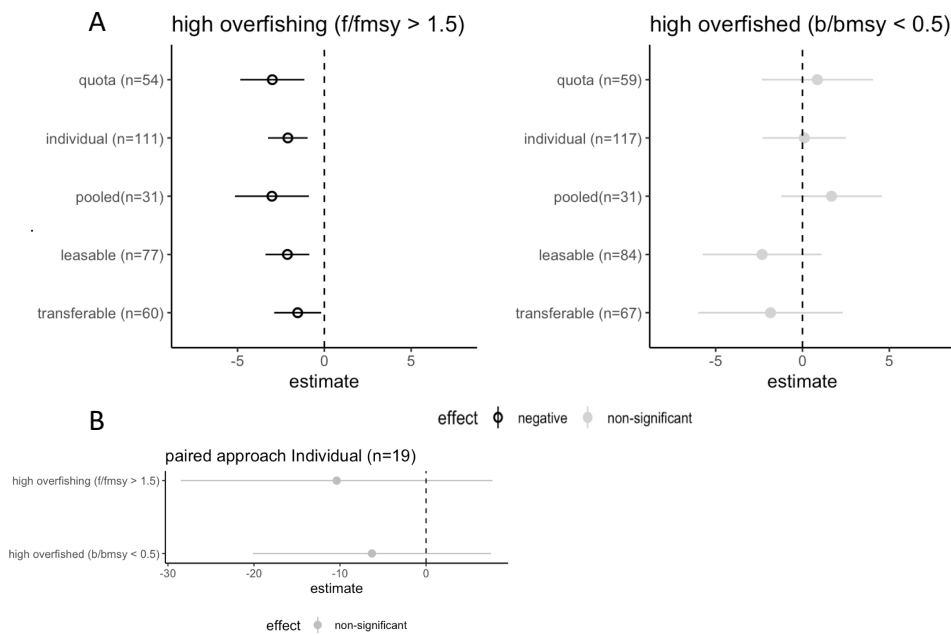


Figure B5. A) Difference in differences results for fisheries opportunity attributes Q, I, P, L and T, the attribute R had only 7 observations for the DiD analysis and was not modelled. Negative (black, open circles) effects indicate a reduced probability of *high overfishing* associated to the P, Q, I, T and L. Amount of treatment stocks are noted between brackets for each of the tests. B) Difference in differences results for fisheries opportunity attribute I and *high overfishing* and *high overfished* outcomes, for fisheries that were previously under pooled quota management, paired to similar fisheries that remain under pooled quota management.

Predicting mean continuous F/Fmsy and B/Bmsy, additional methods and modeling outcomes

Additional methodology

We log-transformed the response variables F/Fmsy and B/Bmsy and confirmed that distributions approximated a normal distribution. We did not back-transform model estimates, effect sizes are thus interpreted qualitatively. For each model, we assumed that the residuals followed a first order autocorrelated (AR1) process which controlled for the fact that the time-series observations were serially correlated at the stock-level.

First, we modelled the transformed mean F/Fmsy (F) and B/Bmsy (B) for region r , stock s , and time t as a function of the fisheries management system (and its multiple attributes):

$$\begin{aligned} Y_{r,s,t} &= \log(F_{r,s,t}); Y_{r,s,t} = \log(B_{r,s,t}) \\ &= \beta_0 + \beta_1 M_{r,s,t} + R_r + R_s + \varepsilon_{r,s,t} \end{aligned} \quad (1)$$

where $M_{r,s,t}$ is a dummy variable for the management system in place, R_r is a random effect dummy variable for the region, R_s is a dummy variable for each stock (nested within region). Like in the main analysis, we compared the effects of all management systems against *total effort* (TE).

Second, we modelled the sustainable fishing metrics as a function of the attributes I, T, and/or Q, L, R and P:

$$\begin{aligned} Y_{r,s,t} &= \log(F_{r,s,t}); Y_{r,s,t} = \log(B_{r,s,t}) \\ &= \beta_0 + \beta_1 Q_{r,s,t} + \beta_2 I_{r,s,t} + \beta_3 T_{r,s,t} + \beta_4 L_{r,s,t} + \beta_5 R_{r,s,t} + \beta_6 P_{r,s,t} + \beta_7 SG_{r,s,t} + R_r + \\ &R_s + \varepsilon_{r,s,t} \end{aligned} \quad (2)$$

The metrics of sustainable fishing was modelled by dummy variables Q (quota), I (individual), T (transferable), L (leasable), R (rationed), P (pooled), SG (self-governed). Random effects were the same as in Equation (1).

We modelled the impact of the duration of fishing opportunities as follows:

$$\begin{aligned} Y_{r,s,t} &= \log(F_{r,s,t}); Y_{r,s,t} = \log(B_{r,s,t}) \\ &= \beta_0 + \beta_1 D_{r,s,t} + R_r + R_s + \varepsilon_{r,f,s,t} \end{aligned} \quad (3)$$

where $D_{r,s,t}$ represents the duration of fishing opportunities in a management system. We compared the effects of duration against *single season* as a control group. Random effects were the same as for Equations (1) and (2).

As a second approach, we employed a DiD analysis for all transitions where a Q, I, T, R, or P element was “added”.

Equation 4 represents the DiD approach where treatment stocks were compared to control stocks:

$$\begin{aligned}
 Y_{r,s,t} &= \log(F_{r,s,t}); Y_{r,s,t} = \log(B_{r,s,t}) \\
 &= \beta_0 + \beta_1 Tr_{r,f,s,t} + R_r + R_s + \varepsilon_{r,f,s,t} \quad (4)
 \end{aligned}$$

The sustainable fishing metrics were modelled by dummy variable Tr (treatment, i.e., addition on I, T, or Q in the treatment fishery, a dummy variable which was coded 1 after the addition of the attribute and coded 0 for control stocks or prior to introduction of the attribute in treatment fisheries). The other variables are the same as Equations (1)–(3). Because attribute transitions could be impacted by transitions of multiple attributes (i.e. the transition to I and T occurs simultaneously).

For a subset of stocks ($n=19$), we matched treatment and control stocks for the same species in the same region or regions closely located to one another (Table A2):

$$\begin{aligned}
 Y_{r,s,t} &= \log(F_{r,s,t}); Y_{r,s,t} = \log(B_{r,s,t}) \\
 &= \beta_0 + \beta_1 Tr_{t,s} + \beta_2 P_{ct,s} + \beta_3 Tr_{t,s} * P_{t,s} + R_s + \varepsilon_{c,t,s} \quad (5)
 \end{aligned}$$

The sustainable fishing metrics S were modelled by dummy variable Tr (treatment), P (before and after treatment) and their interaction, β_3 represents the DiD estimator³³.

All models for continuous F/F_{msy} and B/B_{msy} were implemented using the package `nlme`⁴¹ in R studio version 1.1.463³⁵.

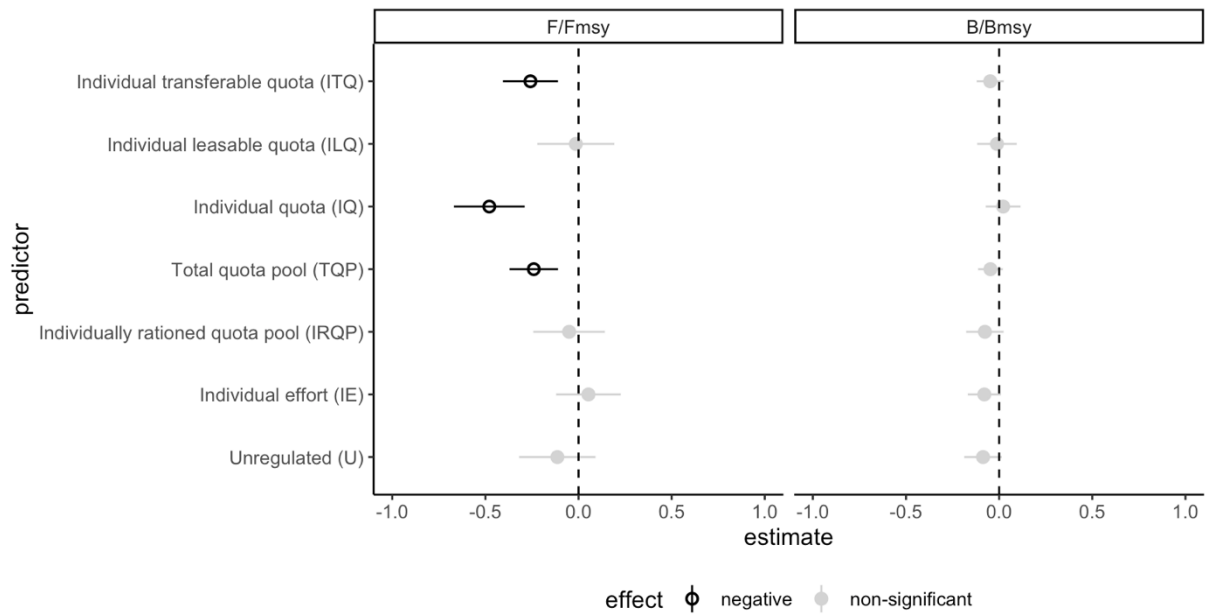


Figure B6. Estimates and 95 percent confidence intervals of management systems impact on mean $\log(F/F_{msy})$ and mean $\log(B/B_{msy})$ compared to TE. Negative (black, open circles) effects indicate a reduced $\log(F/F_{msy})$ for ITQ, IQ and TQP systems.

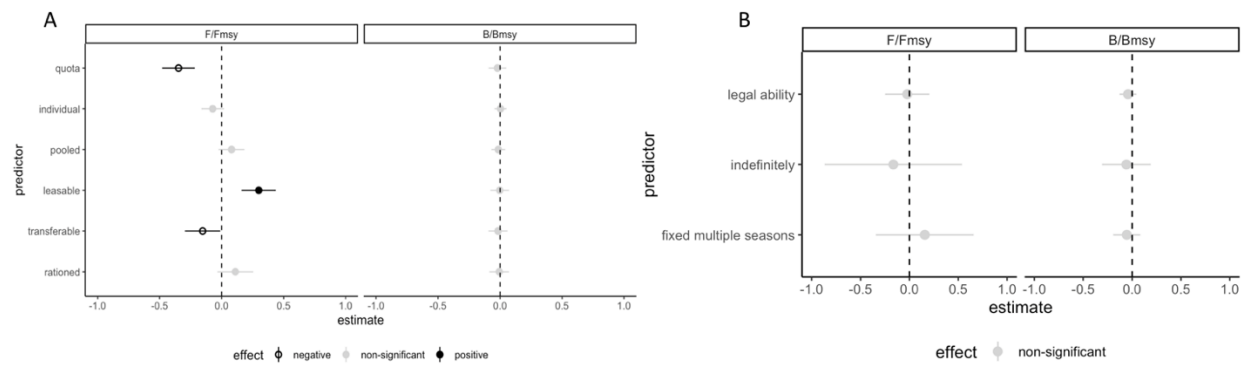


Figure B7. A) Mixed-effects results for the fishing opportunity attributes and their association to mean $\log(F/F_{msy})$ and mean $\log(B/B_{msy})$. Negative (black, open circles) effects indicate a reduced $\log(F/F_{msy})$ for Q and T. Positive (black, closed circles) indicated an increased $\log(F/F_{msy})$ for L (F/Fmsy: 343 stocks with 6803 observations; B/Bmsy: 299 stocks with 6875 observations). B) Effects for the duration of fishing opportunities compared to a single season for F/Fmsy and B/Bmsy

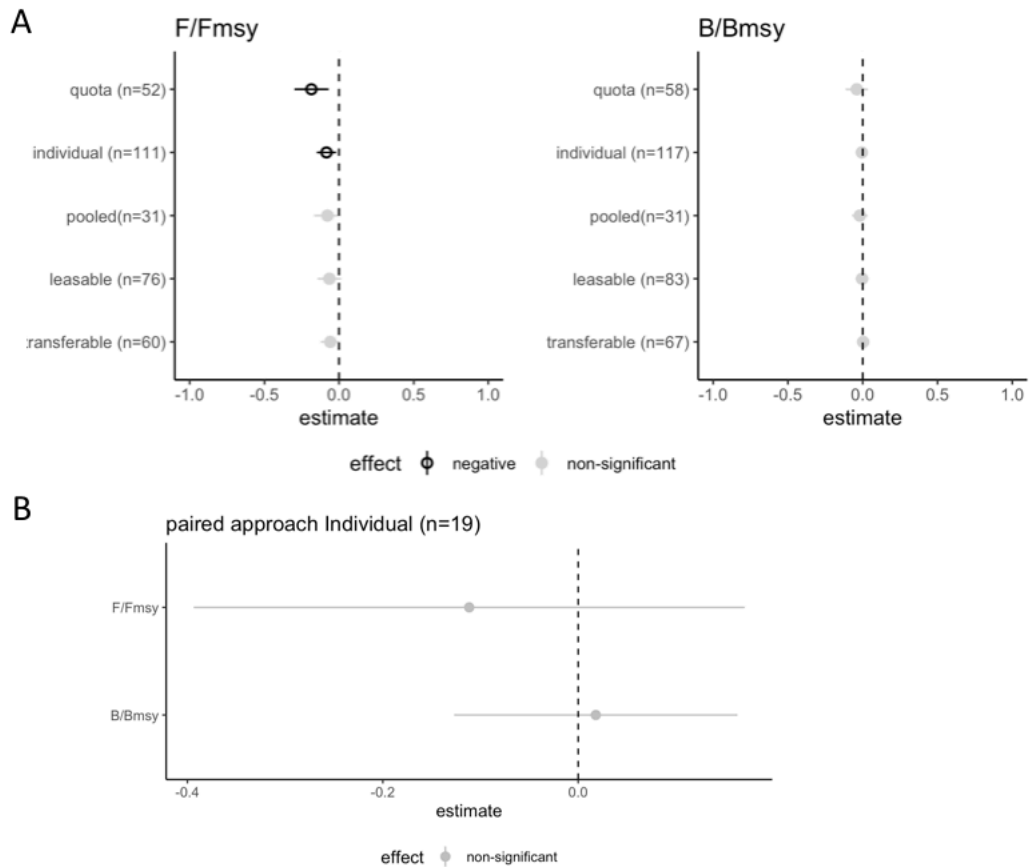


Figure B8. A) Difference in differences results for fisheries opportunity attributes Q, I, P, L and T, the attribute R had only 7 observations for the DiD analysis and was not modelled. Negative (black, open circles) effects indicate a lower mean $\log(F/F_{msy})$ associated to the Q and I. Amount of treatment stocks are noted between brackets for each of the tests. B) Difference in differences results for fisheries opportunity attribute I and $\log(F/F_{msy})$ and $\log(B/B_{msy})$ outcomes, for fisheries that were previously under pooled quota management, paired to similar fisheries that remain under pooled quota management.

Model performance

To assess model predictive performance (of the models presented in Figure 2-4, as a measure of “fit” to the data, we split the data in a training and testing sample. If the model predicted probability was 60% we determined that that fishery was predicted to experience *overfishing* (mortality) or was predicted to be in an *overfished* state (biomass). We then calculated model accuracy by comparing actual states in *overfishing* and *overfished* to model predicted states. Even if it was not our intention to predict *overfishing* and *overfished* states with our modeling effort, such a test will give a sense of the reliability of model results and fit (beyond R^2 values reported in Table B1-B6). We also plot model predicted outcomes versus actual *overfishing* and *overfished* states in the test data below.

Outcome variable	Model	Model accuracy
<i>Overfishing</i> ($F/F_{msy} > 1.1$)	Management model (Figure 2, Eq. 1)	92%
<i>Overfished</i> ($B/B_{msy} < 0.8$)	Management model (Figure 2, Eq. 1)	95%
<i>Overfishing</i> ($F/F_{msy} > 1.1$)	Attribute model (Figure 3C, Eq. 2)	91%
<i>Overfished</i> ($B/B_{msy} < 0.8$)	Attribute model (Figure 3C, Eq. 2)	94%
<i>Overfishing</i> ($F/F_{msy} > 1.1$)	Attribute model (Figure 3D, Eq. 3)	91%
<i>Overfished</i> ($B/B_{msy} < 0.8$)	Attribute model (Figure 3D, Eq. 3)	95%
<i>Overfishing</i> ($F/F_{msy} > 1.1$)	DiD model Q (Figure 4A, Eq. 4)	89%
<i>Overfished</i> ($B/B_{msy} < 0.8$)	DiD model Q (Figure 4A, Eq. 4)	92%
<i>Overfishing</i> ($F/F_{msy} > 1.1$)	DiD model I (Figure 4A, Eq. 4)	91%
<i>Overfished</i> ($B/B_{msy} < 0.8$)	DiD model I (Figure 4A, Eq. 4)	94%
<i>Overfishing</i> ($F/F_{msy} > 1.1$)	DiD model T (Figure 4A, Eq. 4)	90%
<i>Overfished</i> ($B/B_{msy} < 0.8$)	DiD model T (Figure 4A, Eq. 4)	94%
<i>Overfishing</i> ($F/F_{msy} > 1.1$)	DiD model L (Figure 4A, Eq. 4)	90%
<i>Overfished</i> ($B/B_{msy} < 0.8$)	DiD model L (Figure 4A, Eq. 4)	93%
<i>Overfishing</i> ($F/F_{msy} > 1.1$)	DiD model P (Figure 4A, Eq. 4)	91%
<i>Overfished</i> ($B/B_{msy} < 0.8$)	DiD model P (Figure 4A, Eq. 4)	95%
<i>Overfishing</i> ($F/F_{msy} > 1.1$)	DiD model paired (Figure 4B, Eq. 5)	95%
<i>Overfished</i> ($B/B_{msy} < 0.8$)	DiD model paired (Figure 4B, Eq. 5)	96%

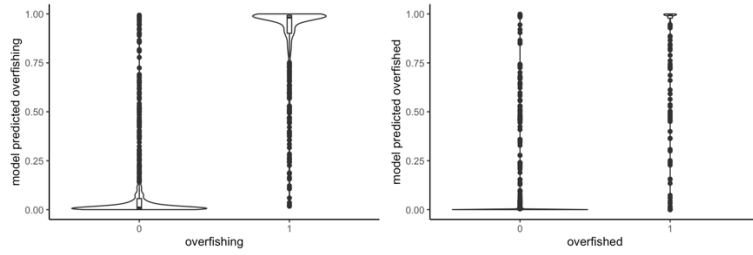


Figure B9. Violin plots for model predictions in test data (y-axis) versus actual *overfishing* and *overfished* outcomes in test data for models using management regimes as predictors (Figure 2 in main body of text, Eq. 1 in methods section).

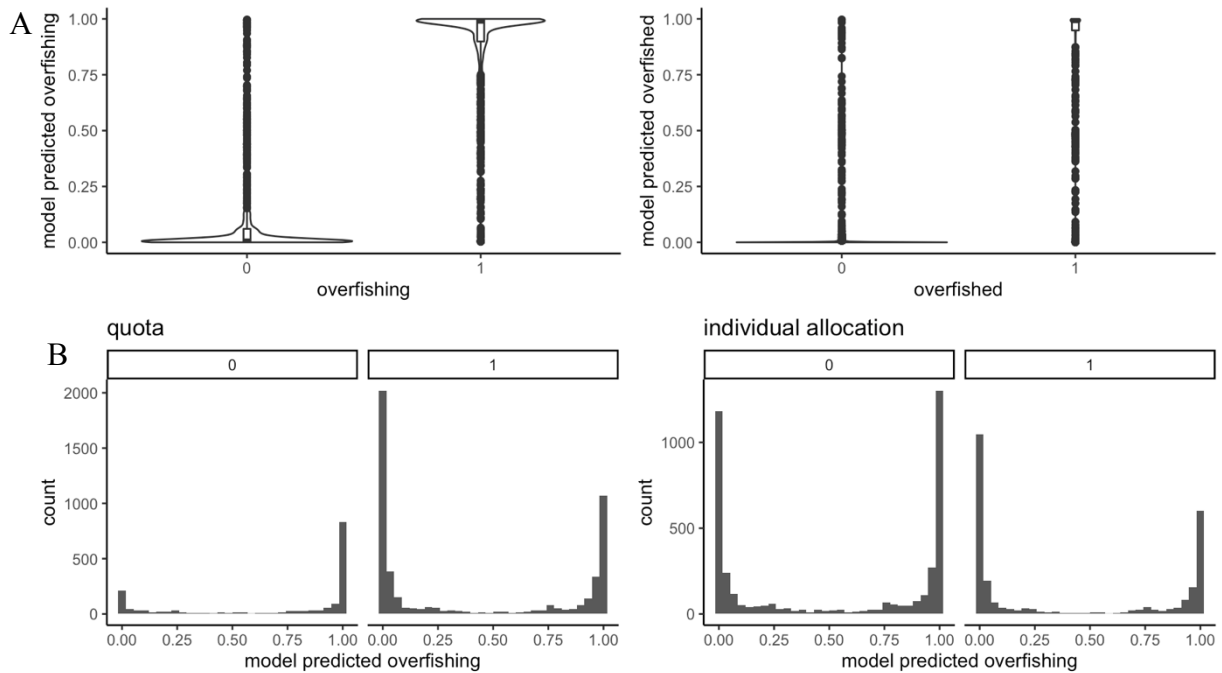


Figure B10. A) Violin plots for model predictions in test data (y-axis) versus actual *overfishing* and *overfished* outcomes in test data for models with management system attributes as predictors (Figure 3C in main body of text, Eq. 2 in methods section). B) model predicted probabilities for *overfishing* in the absence (0) and presence (1) of the attributes quota and individual allocation.

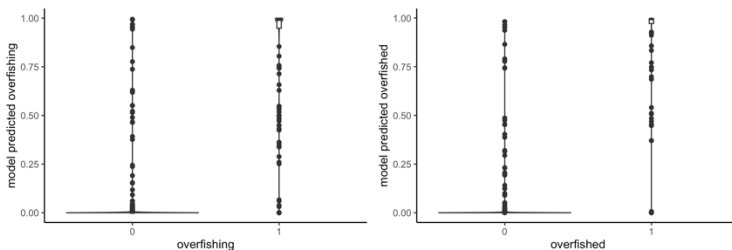


Figure B11. Violin plots for model predictions in test data (y-axis) versus actual *overfishing* and *overfished* outcomes in test data for models with different durations of harvesting rights (for IQ systems) as predictors (Figure 3D in main body of text, Eq. 3 in methods section).

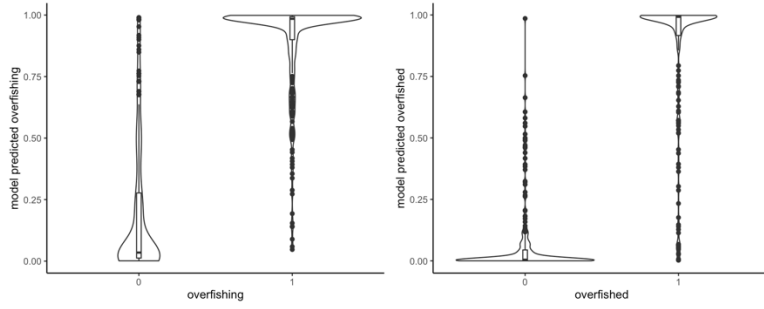


Figure B12. Violin plots for model predictions in test data (y-axis) versus actual *overfishing* and *overfished* outcomes in test data for models with DiD analysis for system attribute Q (Figure 4A in main body of text, Eq. 4 in methods section).

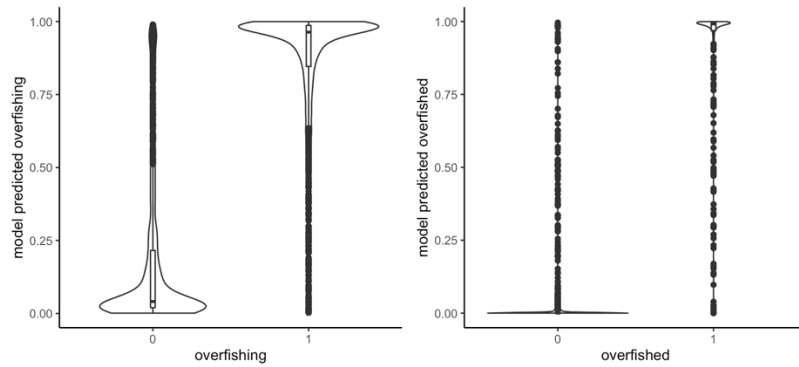


Figure B13. Violin plots for model predictions in test data (y-axis) versus actual *overfishing* and *overfished* outcomes in test data for models with DiD analysis for system attribute I (Figure 4A in main body of text, Eq. 4 in methods section).

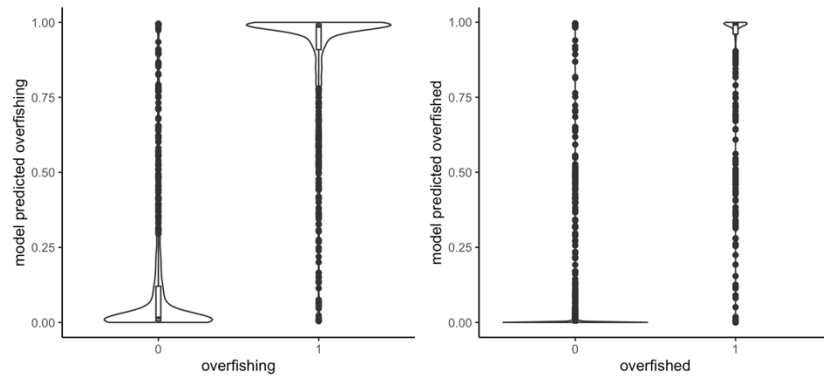


Figure B14. Violin plots for model predictions in test data (y-axis) versus actual *overfishing* and *overfished* outcomes in test data for models with DiD analysis for system attribute T (Figure 4A in main body of text, Eq. 4 in methods section).

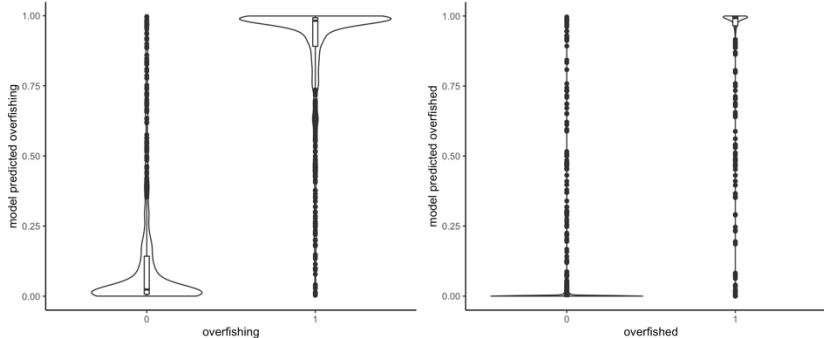


Figure B15. Violin plots for model predictions in test data (y-axis) versus actual *overfishing* and *overfished* outcomes in test data for models with DiD analysis for system attribute L (Figure 4A in main body of text, Eq. 4 in methods section).

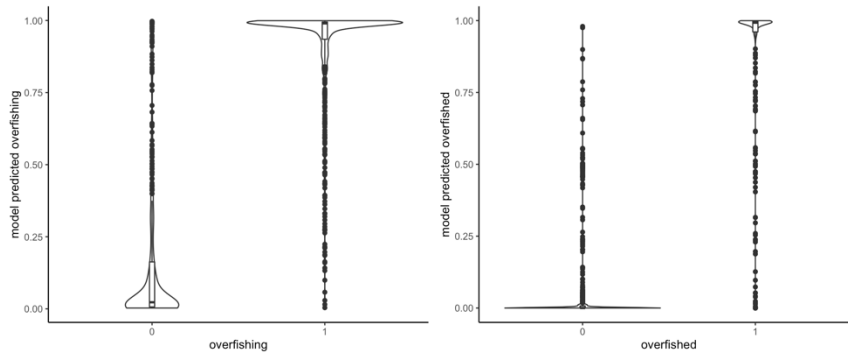


Figure B16. Violin plots for model predictions in test data (y-axis) versus actual *overfishing* and *overfished* outcomes in test data for models with DiD analysis for system attribute P (Figure 4A in main body of text, Eq. 4 in methods section).

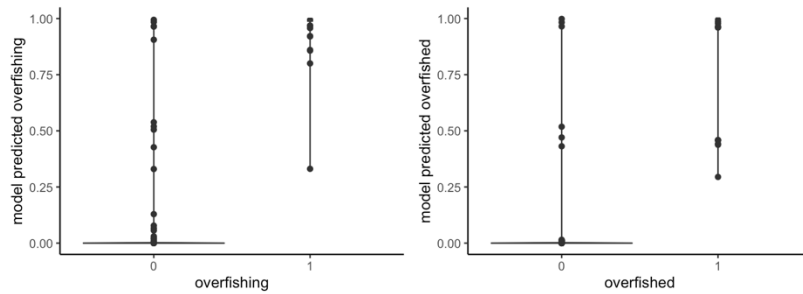


Figure B17. Violin plots for model predictions in test data (y-axis) versus actual *overfishing* and *overfished* outcomes in test data for models with DiD analysis using the paired approach (Figure 4B in main body of text, Eq. 5 in methods section).

Paper II



Catch–quota matching allowances balance economic and ecological targets in a fishery managed by individual transferable quota

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Author contributions: MO collected the data jointly with CB, MO lead the research design and MO and CB jointly carried out the analysis. PW and MJS also designed the research, GS aided with the methodology. MO wrote the first draft of the paper all co-authors helped writing.



Catch–quota matching allowances balance economic and ecological targets in a fishery managed by individual transferable quota

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Fishers with individual catch quota, but limited control over the mix of species caught, depend on trade and catch–quota balancing allowances to fully utilize their quota without discarding. However, these allowances can theoretically lead to overfishing if total allowable catches (TACs) are consistently exceeded. This study investigates usage of balancing allowances by the Icelandic demersal fleet over 2001–2017, for over 1,900 vessels. When a vessel’s demersal catch exceeds owned and leased quota for a given species, the gap can be bridged by borrowing quota from the subsequent fishing period or transforming unutilized quota in other species, restricted by limits. Conversely, excess quota can be saved or transformed into quota for species where there is a shortfall. We found evidence that balancing behavior is frequently similar across the fleet. Transformations are consistent with indicators of a general quota shortage and potential for arbitrage caused by differences in conversion ratios used for transformation and lease prices. Larger companies contribute more to these patterns. Nevertheless, TAC overages are generally modest especially in recent years—key reasons appear to be the tightening of vessel transformation limits and the central role of Atlantic cod, which is the main target species but cannot be persistently overfished due to a specific prohibition on positive transformations into the species. These results show how the tailored design of the Icelandic catch–quota balancing system has helped in balancing economic and ecological goals of management. We suggest policy changes that could further reduce ecological risks, e.g., prioritizing between-year transfers over transformations.

catch–quota balancing | fisheries management | incentives

Harvesters in mixed-species individual quota fisheries (IQs or, if transferable, ITQs) potentially face a dilemma; what to do if they run out of quota in one species before they have used up remaining quota in other species (1, 2). One possible response, continuing to fish but discarding excess catch, has negative consequences and is now prohibited in many fisheries (3, 4). Purchasing additional quota can help but is sometimes not possible: If trade is prohibited for broader reasons (5), a particular quota is scarce due to a systemwide imbalance (6, 7), or frictional trading costs are high. Then harvesters may have to choose between illegal discarding and forfeiting unused quota. For these reasons, catch–quota balancing mechanisms have been introduced in a number of fisheries (8). Despite their limited track record, balancing mechanisms are likely to play an increasingly important role in fisheries management due to proliferation of ITQ systems (9) and discard bans (4), climate change-driven perturbation of marine ecosystems (10, 11), as well as the low amount of catch compared to quota in several mixed ITQ systems and the resulting loss of potential catch value (12, 13).

Catch–quota balancing mechanisms include banking (i.e., transfer of quota between periods; Fig. 1), transformation (i.e., exchange of quota in one species for quota in another species), and surrender (2, 8) (i.e., catch in excess of quota is “sold” at a prescribed price to

the fishery manager). These mechanisms give harvesters limited flexibility to balance quota to catch after fishing. Experience of catch–quota balancing mechanisms has been mixed; while banking is common and has been positively associated with stock status across fisheries (5), transformation has been introduced and later abandoned in Canada and New Zealand due to concerns about overfishing (8) but survived, with modifications, in Iceland (14). A chief concern regarding these mechanisms is that they allow for implicit quota exchange rates (between quota in different periods or species), which may not be aligned with the equivalent exchange rates in quota markets. Where the quota exchange rates implied by balancing mechanisms differ from market exchange rates, harvesters will have an incentive to use balancing (15) to exploit the differences, effectively engaging in arbitrage (16). Such incentives are of concern to fishery managers because they are systematic, potentially causing larger gaps between harvest and total allowable catch (TAC). This does not necessarily mean that all instances of systematic behavior must be due to arbitrage; they may also be due to species for which there is a general quota shortage, for instance when, for rebuilding purposes, quota are set at low levels compared to actual biomass (if such species are caught together with target species, they can constrain the amount of catch of target species and function as so-called “choke”

Significance

The trend toward individual quota and discard bans presents a challenge for mixed fisheries: how to avoid widespread underutilization of quota due to choking effects of individual species for which quota is exhausted. Iceland’s demersal fishery has met this challenge using the most elaborate set of balancing mechanisms in the world. We investigated the performance of the Icelandic system and find pervasive incentives inherent in the system’s design. The absence of persistent overfishing of individual stocks is attributed to limits that have been tightened over time and are very strict for the primary target species. These results highlight the potential for balancing mechanisms to facilitate sustainable exploitation of distinct interconnected resources and the importance of adapting implementation to local circumstances.

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The authors declare no competing interest.

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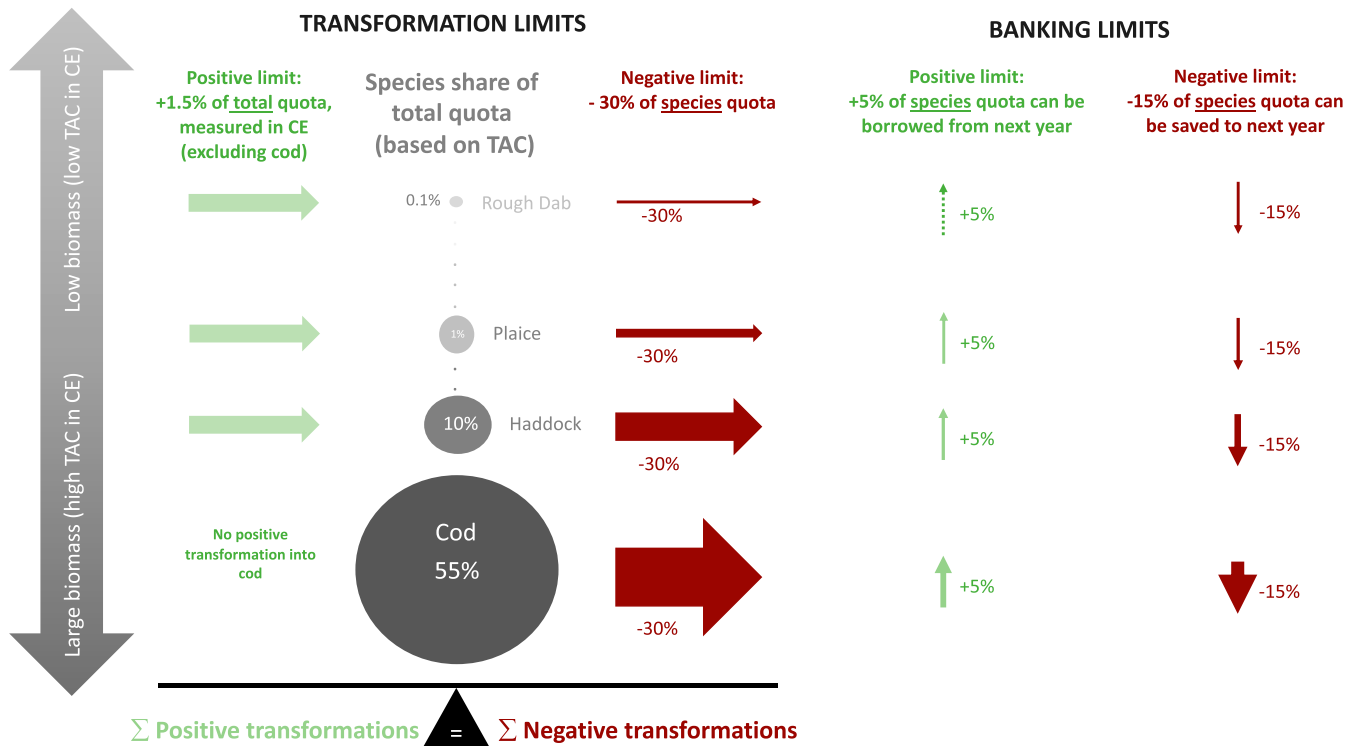


Fig. 1. Transformation and banking limits in the Iceland ITQ system. Positive (light green)/negative (red) limits restrict increases (decreases) in the permitted catch of the relevant species in the current period. Total cod equivalents (CEs) (units converting species quota to the same unit based on last year's price relative to cod) are summed across species before applying the 1.5% positive transformation and 5% total transformation limits; all others are applied as percentages of the originating species quota. The arrows are scaled relative to the total percentage of CE of the relevant species. The left-hand side of the figure shows how transformation limits are designed asymmetrically; the positive limits are based on a vessel's total CE quota, aggregated across species, which is the same for all species, while the negative limits are based on quota in the relevant species. This means that the overall fleetwide limit on positive transformations can potentially be several times TAC for low biomass species (it is 75 times TAC for common dab) but will be small relative to TAC for high biomass species and may be further constrained by the limited "supply" of negative transformations from low biomass species.

species). The distinction between drivers, a general quota shortage as opposed to arbitrage, is important because the latter can be sufficient to be the cause of persistent overfishing while the same is not true for the former.

The Icelandic ITQ-managed mixed demersal fishery is a suitable system for investigating catch–quota balancing behavior due to its use of all of the above-mentioned mechanisms (Fig. 1) as well as quota trade over an extended period, and the availability of detailed vessel- and company-level data. Previous analysis of aggregate balancing outcomes found that TAC overages were modest and did not occur consistently in any species from 2001 to 2013 (14). The current study extends this research, using a complete dataset of individual catch–quota balancing of over 1,900 vessels between 2001 and 2017, to explore the extent to which balancing behavior cancels out at the fleet level and if the pattern of behavior is consistent with hypothesized incentives and constraints. We would expect unpredictable local variation in catch to lead to balancing behavior that cancels out substantially at the fleet level, whereas systemwide constraints or incentives would be more likely to result in similar, seemingly systematic, behavior across vessels. We investigate both banking and transformation behavior, although we place greater emphasis on the latter mechanism since it can theoretically lead to persistent and significant overfishing of particular species. In contrast, the long-term risk from bringing quota forward is relatively low since the maximum amount is limited relative to annual quota and the impact is therefore diluted over longer time periods. The Icelandic ITQ system also allows for surrender of catch, but the associated volumes for demersal species are low, equating to 0.5%

of total demersal quota between 2002 and 2017 (*SI Appendix, Table S3*), and have therefore been excluded from the analysis.

We began by investigating the extent to which balancing behavior (i.e., positive and negative flows) was similar across vessels for each species–year combination. In order to quantify behavioral similarity in balancing, we created a standardized index of the overall directionality of balancing adjustments, defined as D_s for species s and calculated as follows:

$$D_s = \frac{\sum_i^I P_{si} - \sum_i^I N_{si}}{\sum_i^I P_{si} + \sum_i^I N_{si}}, \quad [1]$$

where P_s is the positive quota adjustment for vessel i (0 when negative), and N_s is the negative quota adjustment for vessel i (0 when positive). This index takes values between -1 and $+1$; the former implies that transformation or banking are purely negative, the latter that transformation or banking are purely positive, while 0 indicates equal volumes of positive and negative flows. The directionality index was calculated separately for quota transformed and quota banked at the end of each year.

We then used a regression model to examine the drivers of the directionality of transformations (we refer to this model as the "transformation directionality model"). We developed quantitative proxy measures of proposed behavioral drivers, namely, the following: potential for arbitrage (*arbitrage potential*), the ability to target species (*targeting indicator*), as well as a systemwide quota shortage (*choke indicator*). The potential for arbitrage arises when the quota conversion rate set by the fishery manager differs from the conversion rate that can be achieved in the quota market,

i.e., by simultaneously selling quota in one species and purchasing quota in other species (17). We defined a proxy for each species' *arbitrage potential* in a given fishing year based on the ratio between the average lease cost of quota and transformation cost (fixed by the fishery manager as the cod equivalent [CE] value from the previous fishing year). The ratio is then normalized, dividing it by the weighted average ratio across all species (excluding Atlantic cod) to yield the proxy. An *arbitrage potential* value of 1 corresponds to parity with a notional basket of the remaining species, while a value >1 implies that it would be cheaper to obtain the relevant species quota indirectly by leasing and then transforming quota in other species rather than leasing the desired species quota directly; a value <1 implies the converse. This proxy is only a rough indicator of arbitrage potential over the fishing year as it is based on comparing average lease prices across species, whereas arbitrage involves risk-free exploitation of contemporaneous price disparities (16). Harvesters' ability to exploit *arbitrage potential* opportunities is increased when they can proactively target species, which can be transformed into cheaply (or avoid species with a high transformation value), potentially exacerbating the risk of overfishing, i.e., we would expect an interaction effect between the ability to target species and their *arbitrage potential*. To assess this possible behavior, we created an indicator of species *targeting*, defined as the percentage of each species total annual catch occurring on trips where the species contributed at least two-thirds of trip catch (*SI Appendix, Supplementary Methods*) and interact this variable with *arbitrage potential*.

Similar behavior across vessels and a high arbitrage potential could be caused by a general shortage of quota in the relevant species ("choke" species) or by arbitrage potential. In order to distinguish the two phenomena, we included a species *choke indicator*, calculated as a binary presence/absence variable where a choke effect was considered present whenever the average lease price exceeded the average marginal catch value (defined as ex vessel price less estimated crew share, quota fee, and fuel cost) (1). We also included TAC in the transformation directionality model as larger TAC species are more likely to be targeted due to economies of scale (18) and therefore more likely to be species for larger positive transformation flows ("sink" species).

We also developed a set of multispecies and single-species regression models to investigate balancing behavior (banking and transformation) at the individual vessel level and the influence of different resource user characteristics, including company and vessel size, and permit type. We refer to these models as the "vessel-level" models (i.e., single-species and multispecies vessel-level models). We expected larger companies to more fully utilize the balancing mechanisms for arbitrage since they would have more management resources and potentially have more scope to alter the species mix, especially if they have multiple vessels. This study examines catch-quota balancing behavior at the vessel and company level.

The Icelandic system has been adjusted over time, particularly the limits for transformation and banking, most notably in 2011/2012 (more detail in *SI Appendix*). The main rule change reduced the limit on negative transformations from 100% of species quota held by each vessel to only 30%, and we included this change as a dummy variable in the vessel-level models to investigate whether this change was effective. We use the vessel-level models rather than the transformation directionality model as only a change in underlying transformation volume would be expected, which is not captured by the directionality index.

Results and Discussion

There is behavioral similarity in both transformation and banking as the directionality indices deviate strongly from zero (Fig. 2B and C). A small set of species have predominantly negative transformations effectively acting as "source" of additional quota for other species. For example, transformations for both dab species were consistently

below -0.5 . Similarly, there are "sink" species for which the direction of transformation appears to be mostly positive (e.g., haddock being above 0.5 for all years after the rule change). For the majority of species, the transformation directionality varies from year to year. Directionality of banking was predominantly negative (i.e., saving quota to the next year, Fig. 2C), meaning that harvesters prefer to save quota rather than borrow it, which is a pattern also observed in other fisheries (8). This result suggests risk aversion on the part of harvesters in the face of uncertainty regarding future TAC levels and the potential for choke effects. Overall, catches in the Icelandic system have been relatively well-aligned with the TAC, and in 2017 on average 88% (82% when excluding all TAC overages) of TAC was caught in Iceland's mixed fisheries (Fig. 2A). This is high compared to the 30 to 60% of TAC caught in mixed fisheries that same year in the United States (13).

We found large variation in arbitrage potential and a clear difference between species (Fig. 2D), suggesting that it may often be more profitable for companies to use species transformations rather than using the lease market. A few species have consistently low arbitrage potential values (e.g., both dab species and Greenland halibut), implying that harvesters with surplus quota in these species would have an incentive to transform out of the quota rather than lease it out. In contrast, other species exhibit high *arbitrage potential*, albeit not for every year, indicating that transformation into these species may be more profitable than leasing them in. There were few choke observations in the Icelandic system: Atlantic cod was indicated as a choke species in all years and haddock and redfish in several years (Fig. 2E). The *targeting indicator* also displays large variability, both between species and years (Fig. 2G). It is important to notice that some species, for example monkfish, could be vulnerable species for the transformation system, as they show both relatively high values for *arbitrage potential* and the *targeting indicator* and a low TAC; TAC overages for monkfish are, however, modest (Fig. 2A).

Arbitrage potential was the strongest statistically significant predictor of directionality of transformations (Table 1), consistent with the hypothesis that harvesters respond to the incentives arising from misaligned transformation costs and lease prices. The *arbitrage potential* predictor was also positively associated with the catch: quota ratio in the multispecies vessel-level model as well as 8 out of 14 individual-species vessel-level models (Fig. 3). Contextual evidence exists to support these findings. For example, several source species have material amounts of unused quota that are effectively forfeited (*SI Appendix, Fig. S3*), and it is logical to expect the owners of this quota to have fully utilized opportunities to transform quota of these species into more valuable species, as predicted by theory (19, 20). We find circumstantial evidence that transformations may sometimes be driving quota trade, with an average of 54% of negative transformation volume occurring when the quota was first leased in and then transformed (*SI Appendix, Table S5*)—with this ratio reaching 70% for some species.

On the other hand, we found that the *choke indicator* does not significantly predict directionality of transformations, which suggests that the alternative explanation that general quota shortages would drive up both transformation and relative lease prices is less supported, strengthening the case for arbitrage-driven behavior. In the vessel-level models, we found that the choke indicator also showed no significant effect in the multispecies model as well as most of the individual-species models, with the exception of a higher catch-to-quota ratio for redfish and common dab in choke years and a negative effect on the catch-to-quota ratio for ling (Fig. 3). The effects for the *choke indicator* should be read with caution, however; it could be that the presence/absence indicator is too coarse to capture a gradual shift in case a species turns out to be a choke during the fishing year (as lease prices may rise throughout the year). For redfish, it seems that *arbitrage potential*

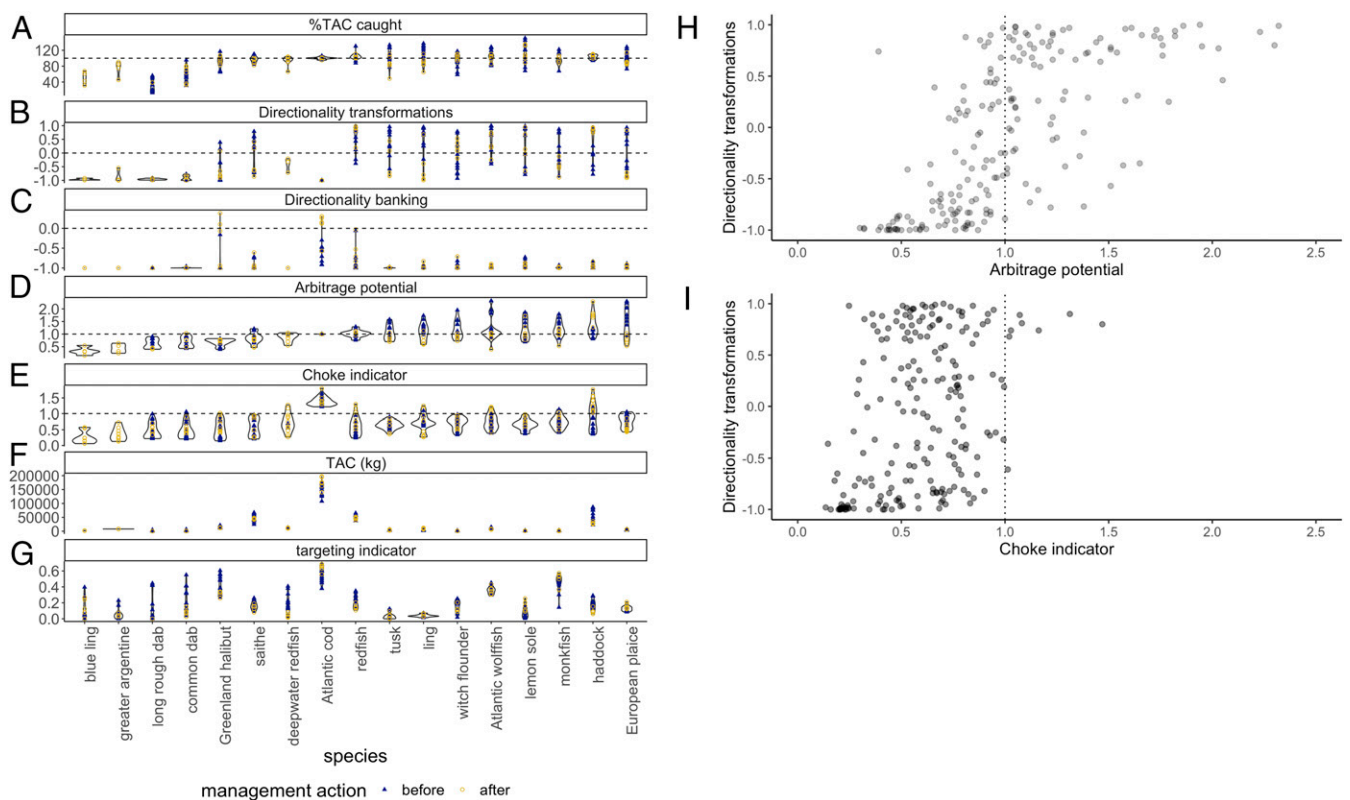


Fig. 2. Variables and indicators for the Icelandic catch-quota balancing system. (A) %TAC caught; (B) directionality of transformations: positive values indicate transformations into the species and negative values indicate transformations out of the species; (C) directionality of banking: positive values indicate borrowing from next year and negative values indicate saving to next year; (D) arbitrage potential: a value <1 means that it would be cheaper to lease quota in the corresponding species and then transform into a basket of the other species rather than lease quota for the basket directly; a value >1 implies the inverse; (E) choke indicator: >1 indicates that the cost of leasing quota exceeds landed value, net of fishermen's catch share; (F) TAC in kilos of gutted fish; (G) targeting indicator; (H) directionality of transformations as a function of arbitrage potential (excluding Atlantic cod); and (I) directionality of transformations as a function of choke indicator (excluding Atlantic cod). The blue triangles indicate observations before the rule change in 2011/2012, which limited flexibility in transformation usage, while the yellow points indicate observations after the rule change. Species are organized from lowest to highest mean values for *arbitrage potential*. Note that a few species have fewer observations as they were added later to the species transformation system (blue ling, greater argentine, and deep-sea redfish). For each species in A–G, the violin plot indicating the data frequency of distribution is also plotted.

and *choke indicator* act together, with different fishers possibly responding differently to ecological and economic signals. The way this could be explained is that, for example, a fishing company may run out of redfish quota and is forced to pay a high price, while another fishing company may be using species transformations to cover their redfish catch while simultaneously leasing out redfish quota. Moreover, including the cost of fuel is an important assumption when calculating the choke indicator as we assume that fuel is expended on the species mix and not for target species only; our results, however, are largely robust to this assumption as shown in a sensitivity analysis (*SI Appendix*). It could be that the low number of choke observations for small biomass species in Iceland are related to the asymmetry of the transformation limits (Fig. 1); these limits are hardly ever met by individual vessels for small biomass species but more frequently so for haddock and redfish especially after the management changes in 2011/2012 (*SI Appendix*, Table S8).

Atlantic cod may be the ultimate choke species in the Icelandic demersal quota system as we found that the average cod quota lease price exceeded estimated marginal value (average ex-vessel price adjusted for crew share, quota fee, and fuel cost) in all years. Moreover, the catch-quota balance for cod is nearly perfectly aligned for the majority of vessels (*SI Appendix*, Fig. S2). Atlantic cod is by far the most abundant demersal species and contributes the majority of the catch value of the Icelandic demersal fleet, so that each company would need to own some

cod quota to run a demersal fishing operation, but it is the only species for which quota cannot be increased via transformation. We found that the vast majority of demersal trips and catch contain Atlantic cod (*SI Appendix*, Fig. S4) and Atlantic cod is at times actively avoided by vessels in the Icelandic fleet (21). Positive species transformations for many species will thus be limited due to the choke effect of cod, and the choke effect of cod may explain the high level of quota saving observed for many species, while borrowing is observed for cod (Fig. 2C). Ultimately, if cod quota is exhausted, then there is no incentive to transform into species for which the amount of cod is the limiting factor. This is an observation that needs to be considered if fisheries managers consider translating mechanisms from the Icelandic context to other fisheries (4), especially in an ecosystem where such a large economically important species is absent.

The results also show that directionality of transformations is predictable from TAC, which could be because larger quantities of fish may be cheaper to process and distribute (Table 1). We find that transformations tend to reduce catch of low TAC species and increase catch of high TAC species, for example redfish and haddock (*SI Appendix*, Fig. S3). Since the legal limits are more constraining for transformations into high TAC species (Fig. 1), the tendency to transform into high TAC species reduces the ecological risks associated with species transformation. However, a small negative effect for TAC is shown for six of the single species models, which may indicate companies needed to rely less

Table 1. Directionality of transformations model with fractional logit estimates of the contribution of each of the predictor variables, SEs, z values, and probabilities

Predictor	Estimate	SE	z value	Pr(> z)
Arbitrage potential	1.47	0.21	7.14	<0.001
Choke indicator (dummy variable)	-0.17	0.44	0.38	0.70
TAC	0.99	0.29	3.41	<0.001
Targeting indicator	-0.09	0.14	-0.63	0.53
Targeting indicator * arbitrage potential	-0.21	0.15	-1.35	0.18
Cox and Snell's $R^2 = 0.27$				
Nagelkerke's $R^2 = 0.58$				

It can be observed that *arbitrage potential* and total allowable catch (TAC) are the most important predictors of transformation directionality with a positive effect. P-values significant at the < 0.05 level are printed in bold. Predictor variables that were included are as follows (continuous variables are indicated as c; dummy variables as d; ranges are specified): 1) *arbitrage potential* (lease price over CE conversion ratio, normalized; c {95.7; 176.8}); 2) *choke indicator* (lease price rises above ex-vessel price plus marginal costs; d {choke observation, no choke observation}); 3) TAC (c {176; 86980}); 4) targeting indicator, percent catch of a species for which a species is two-thirds of the catch (c {0;1}).

on balancing mechanisms in high TAC years for those particular species, possibly indicating a general quota shortage in lower TAC years.

Contrary to our expectations, the interaction between the *targeting indicator* and *arbitrage potential* had no effect in predicting directionality of transformations (Table 1), but it did show a small positive effect in the multispecies vessel-level model and in five of the species' vessel-level models (Fig. 3). Some species (e.g., European plaice, redfish, and lemon sole) are thus predicted to have increased catches when both the *targeting indicator* and *arbitrage potential* have high values, indicating that for those species arbitrage opportunities increase when the species is more targetable. However, we also found a negative interaction effect in five of the single-species vessel-level models, which may be caused by increased lease prices due to choke effects rather than *arbitrage potential*. If species are highly targetable, they may also be easier to avoid. In a scenario where lease prices rise due to choking effects, the ability to avoid such species can result in lower catches.

Larger companies rely more on the transformation system and possibly make more use of the potential for arbitrage as there was a small positive effect of total demersal quota holdings in the multispecies vessel-level model and in 6 out of 14 single-species vessel-level models (Fig. 3). Moreover, boats with a small boat permit type (using either only hook and line gear or smaller than 10 gross tonnage) had less catch per quota than larger boats for most species, indicating more missed fishing opportunities, which is also demonstrated by larger amounts of unused quota for this fleet segment (*SI Appendix, Fig. S5*), as well as less transformation and banking activity in several species (*SI Appendix, Fig. S6*). Only for Greenland halibut is the small boat permit predicted to have more catch per quota on average, this is because Greenland halibut is a major source species for the larger boat permit (*SI Appendix, Fig. S6*).

The management action in 2011/2012 resulted in negative changes in catch–quota balance for the sink species Atlantic wolffish, ling, and monkfish (Fig. 3), as well as a large positive change for Greenland halibut, which acted as a main source species. Therefore, the management action appears to have been effective across a variety of species. This is also reflected in the fact that large TAC overages became less common after the management action, with the only overages above 10% of TAC occurring for lemon sole (Fig. 24).

Several of our results have important policy implications. First, the arbitrage incentive that arises from species quota transformation ratios that are not aligned with quota markets should be considered when fisheries managers consider the implementation of such mechanisms e.g., in the context of the common fisheries policy in the European Union (4). Such incentives could result in systematic overfishing especially in cases where a highly constraining factor/species such as the Atlantic cod in the Icelandic case is absent. Second, we showed that fishers tend to save quota rather than borrow from the next year, but that companies at the same time use species transformations to cover catch in the same species as is saved. This is possible because balancing is done at the vessel and not at the company level. Simple policy changes could be 1) to allow companies to use species transformations to cover catches only if they have already borrowed the maximum amount from the next year, and 2) to balance catch to quota at the company level rather than the vessel level. In this way, a large amount of species transformations could have been avoided. For instance, 53.3% of positive haddock transformations could have been avoided if balancing was done at the company level or if banking was prioritized over transformations (*SI Appendix, Table S6*). In addition, the limit for transformation into each species is based on total vessel quota across species (Fig. 1). This design feature is particularly risky for profitable small biomass species as total CE holdings can be several times their TAC; it would thus be prudent to add a species-specific limit for positive transformations, as is already the case in Iceland for negative transformations.

Beyond fisheries, ITQ balancing mechanisms such as those studied here could be a template for new approaches to sustainable governance that respect multiple interconnected planetary boundaries to resource utilization and pollution, while recognizing the potential for marginal trade-offs to improve cost effectiveness (22). This approach, which may be described as “flexibility within limits,” allows for partial substitutability between different forms of natural capital and can therefore be viewed as a compromise between strong and weak forms of sustainability (23, 24).

In conclusion, with the recent modifications to the catch–quota balancing system in 2011/2012 and additional slight adjustments, catch–quota balancing mechanisms could balance socioeconomic benefits for fishers harvesting uncertain and interconnected natural resources with ecological risks of overexploitation. Our conclusions, however, are very much bound to the Icelandic context where one highly abundant and strictly managed stock, Atlantic cod, may drive much of the observed behavior. We advise managers to consider this important role of cod when considering application of the Icelandic catch–quota balancing system to other ecosystems. Other mixed-fisheries ITQ systems may have a similar ubiquitous and economically important species (12, 25) and could benefit from Iceland's experiences with the balancing system. Arbitrage opportunities were nonetheless observed, which in the absence of restraining factors could result in ecological risks, especially for valuable low biomass species.

Materials and Methods

We obtained data on catches, quota, and lease values and company characteristics from the Fishery Directorate (www.fiskistofa.is/) (26) and ex-vessel prices from Statistics Iceland (<https://hagstofa.is/>) (27).

The *targeting indicator* was calculated by computing the fraction of catch for each species where the species was at least two-thirds of the catch. As an indication of company size, we summarized the companies' holdings in all demersal species multiplied by the respective species' CE value.

The directionality index was predicted using a fractional logit model (28) and species-level predictors using the following equation:

$$D_{s,t} = 2 * (E_{s,t} + P_{s,t} + M_t + F_t + \varepsilon_t) - 0.5 \quad [2]$$

where $D_{s,t}$ is the mean predicted directionality at time t for species s , $E_{s,t}$ is a matrix of ecological fixed effects (*targeting indicator* and TAC), $P_{s,t}$ is a

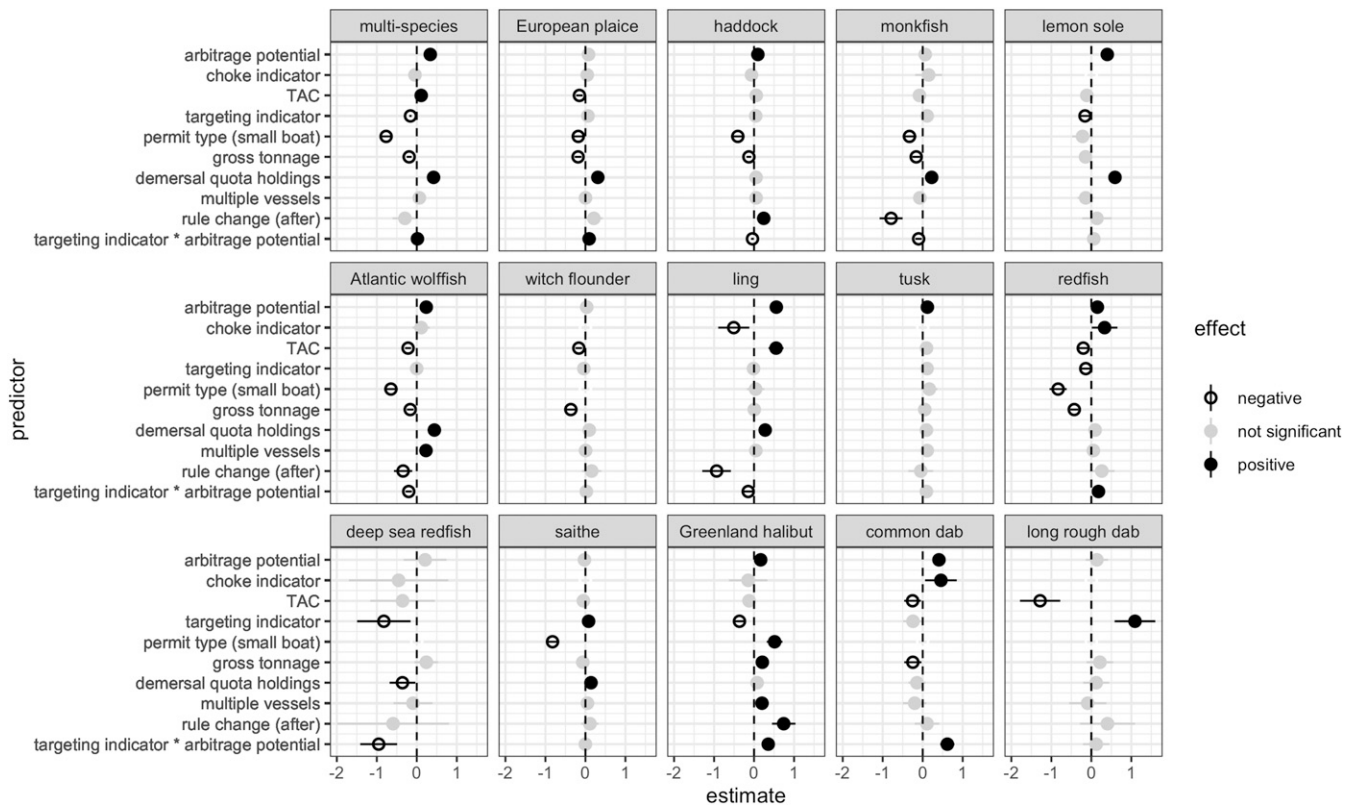


Fig. 3. Effect size for predictors predicting catch/quota in multispecies and single-species models, except Atlantic cod and a few species with too few observations. Effect size is represented by the location of the dots in the estimate values range along with 95% CIs. When CIs cross the dotted line at 0, the predictor variable is considered not significant (note that because there is a large amount of data underlying the figure the CI is often very narrow and not visible in the figure). The exponent of the effect size is the predicted increase in catch/quota for each unit increase in the predictor variable (e.g., on average for all species the ratio of catch/quota is predicted to increase with 1.4 with an increase of 1 in arbitrage potential, all else being equal). Predictor variables that were included are as follows (continuous variables are indicated as c, dummy variables as d, ranges are specified): 1) *arbitrage potential* (lease price over CE conversion ratio, normalized; c {95.7; 176.8}); 2) *choke indicator* (lease price rises above ex-vessel price plus marginal costs; d {choke observation, no choke observation}); 3) total allowable catch (TAC) (c {176; 86980}); 4) targeting indicator, percent catch of a species for which a species is two-thirds of the catch (c {0;1}); 5) a variable that represents the two main fleet segments (d {small boats with passive gear, larger boats with mostly active gear}); 6) the gross tonnage of the vessel (c {1; 7682}); 7) the amount of demersal quota held by the company operating the vessel (c {0;40568493}); 8) whether the company operating the vessels has multiple vessels (d {single vessel, multiple vessels}); 9) the management adjustments (rule change) (d {2011/2012 and prior, after 2011/2012}); as well as 10) the interaction effect for the *targeting indicator* and the *arbitrage potential*. The small-boat permit variable is not included for species that were not caught by small boats, and the choke indicator is not included for species that were never indicated as a choke species. Observations with negative allowed catch amounts (caused by borrowing and having only a small amount or no allocated quota) were excluded, representing 0.4% of catches of the Icelandic demersal fleet.

matrix of economic time- and species-specific fixed effects (*choke indicator* and *arbitrage potential*), and F_t is a dummy variable for the fishing year. We used the Newey–West estimator to calculate SEs, which is robust in the presence of autocorrelation. We chose to use a fractional logit as the directionality values are bounded between -1 and 1 ; to meet the requirements for the fractional logit model, we divided directionality values by 2 and added 0.5 so that values occur on a continuous interval of 0 to 1.

The individual level models were set up using the following equation assuming a gamma distribution and using a log link:

$$\mu_{i,s,t} = Q * E^{E_{s,t} + P_{s,t} + S_{i,t} + R_i + R_s + \epsilon_{i,s,t}} \quad [3]$$

where $\mu_{i,s,t}$ is the predicted mean catch of vessel i at time t in species s , $Q_{i,s,t}$ is quota of vessel i at time t in species s , $S_{i,t}$ is a matrix of vessel and time-fixed effects, R_i are the vessel random effects, and R_s are the species random effects. The model is offset by the amount of quota, and therefore predicts the ratio between mean predicted catch and quota (μ/Q). Autocorrelation in the time-series was controlled for using a first-order autoregressive model. In all

models, we standardized predictor variables to have a mean of 0 and a SD of 1.

Data Availability. Anonymized data have been deposited in GitHub, <https://github.com/maartje-oostdijk/quota-balancing>.

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Supporting Information

The Icelandic demersal fisheries

The Icelandic demersal fishery may be considered mixed with 18 demersal species subject to quota management, the largest by volume being cod, haddock, redfish, saithe and Greenland halibut (Figure S1). Cod is the dominant species, accounting for 54% of overall landings volume in demersal quota-based fisheries in 2017, and is almost always present in landings by demersal vessels. Cod prices are generally higher than for other demersal species, meaning that the importance of cod is even greater in value terms (57% in 2017). There is significant variation in species mixtures across fleet segments; redfish and saithe are mainly caught by the larger trawlers and accounted 54% freezer-trawler landings in 2017 while cod, haddock and other demersal species are more important to the onshore fleet.

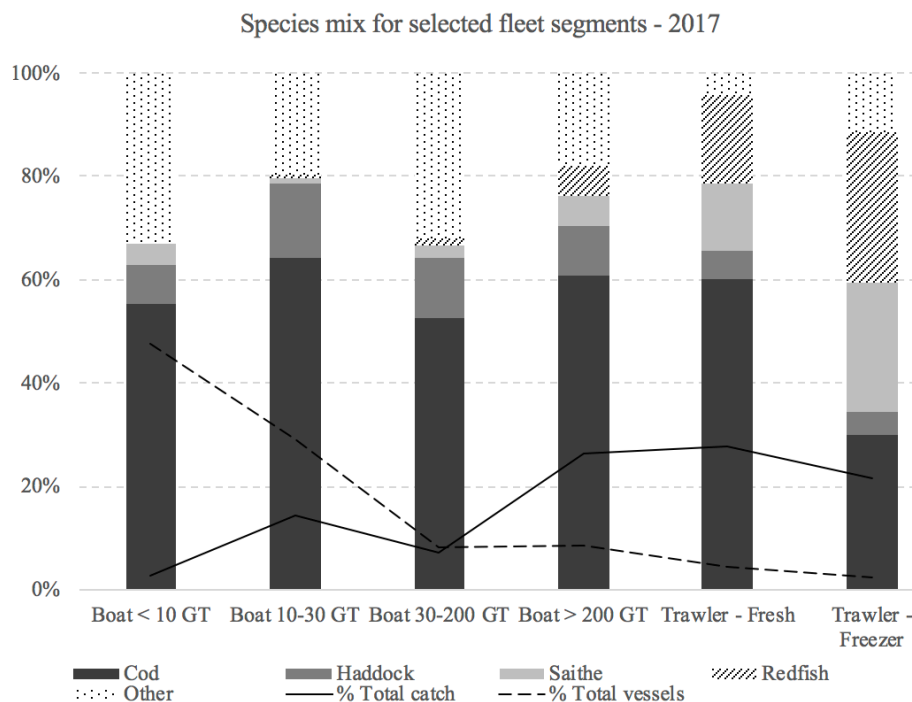


Figure S1. Species mixture for selected fleet segments in 2017 based on weight (Data source: Statistics Iceland, 2018; <https://hagstofa.is/>). Excluded fleet segments relate to coastal fishing, which is not quota-based and generally only composes 2% of the catch, and pelagic vessels. Percent landings and vessels within each segment are also shown as a share of the segments combined.

The Icelandic catch-quota balancing system

Catch-quota balancing in Iceland dates back to the origins of the demersal ITQ system in the 1980s. The regulations have been amended several times to cover more species and tighten the limits on balancing activity. The current regulations provide for three main balancing mechanisms: (i) quota transfers between fishing years in the same species, (ii) quota transformations between species in the same year, and (iii) sale of unwanted catch to the DoF, all subject to limits (Table S1, Figure 1). Catch which is not covered by quota or permitted by these mechanisms is classified as excess catch which may lead to imposition of a fine, while quota which is not allocated to catch, transformed or banked is unused and effectively lost.

Quota are registered to fishing boats and the balancing regulations therefore operate at the level of vessels rather than companies. As a result, a multi-vessel company may simultaneously

engage in different balancing behaviour which might appear unnecessary when considering the company's overall quota and catch. For instance, a company might transform into a species on one vessel while electing to save unused quota in the same species on another vessel.

Table S1: Catch-quota balancing mechanisms for demersal species in the Icelandic ITQ system

Mechanism	Regulations	Limits
Transfers between fishing years in the same species.	Excess quota can be “banked” for use the following fishing year. Quota shortfall can be “borrowed” from the following fishing year.	15% of allocated quota. 5% of allocated quota.
Transformation between species In the same fishing year.	Excess quota in demersal species can be applied to quota shortfalls in other demersal species. The quota conversion ratio between species is based on their relative historic ex vessel prices (“Cod Equivalent” or CE) as set annually by the Directorate of Fisheries.	<ul style="list-style-type: none"> - Transformations out of any species < 30% of allocated quota in that species. - Transformations into each species < 1.5% of the total allocated quota for all demersal species (in CE). - Total transformations < 5% of the total allocated quota for all demersal species (in CE). - Transformations into cod are not permitted.
Surrender to DoF	Undesired catch can be sold at auction with 80% of realised value allocated to a fisheries fund run by the government. The remaining 20% is allocated to the vessel owner and crew. This catch is then excluded from vessel landings for quota purposes.	5% of actual catch by species, applied on a quarterly basis.

Changes in regulation: The above regulations applied over the entire period analysed (fishing years 2001/2-2016/17), with three exceptions. Firstly, the banking limit was initially 20% but increased to 33% in 2008/9 and was then reduced to 15% in 2009/10. Secondly, the limit on the total amount of positive transformations (i.e. enable catch to exceed quota) was reduced from 2.0% to 1.5% in 2011/12. Finally, the 30% limit on the amount of negative transformations (out of a species) was introduced in 2011/12; prior to this, a vessel was permitted to transform its entire quota for any species to cover catch in other species, subject to the positive transformation limits. The government bill proposing the 30% limit included an observation by the government that up to 80% of quota in some species was used for transformation.

The conversion ratio used for transforming between species is based on the “Cod Equivalent” value of each species set annually by the Directorate of Fisheries. The CE value is calculated as the average ex vessel price per gutted kilogram over the preceding year ending 30 April, relative to cod. The conversion ratio between quota in different species is effectively determined by the ratio of their historic market prices and may therefore not reflect their current relative value, it also does not include the price of fishing.

The asymmetry of the limits is a noticeable feature of the Icelandic balancing system: the limit on the amount of positive transformation (into species) is the same absolute amount (in cod equivalent tonnes) for all species regardless of their biomass. Setting this limit at 1.5% means that it is only of practical relevance to the most commercially important species i.e. haddock,

redfish & saithe, as hitting this limit would translate to no more than roughly doubling the TACs of these relatively large stocks, which in some cases could be feasible if desired by the industry. For low-biomass species, on the other hand, this limit translates to a multiple of several times that of the TAC, meaning that it is likely unachievable by the industry anyway, or if it is achievable, then the DoF would need to intervene in order to prevent significant overfishing, thereby negating the need to have any limitation in these instances. Secondly, and by the same token, the negative transformation limit of 30% only has a minor effect on transformations in reality. Transformation of 30% away from cod would cause a flood of available quota to other species, but this is unlikely to happen due to economic reasons.

Table S2: Percentage of catch surrendered for demersal species in the Icelandic ITQ system between 2002-17.

Year/Species	Atlantic cod	Haddock	Saithe	Redfish	Ling	Tusk	Atlantic wolffish	Monkfish	Other species
2002	0.3%	0.2%	0.1%	0.1%	0.1%	0.0%	0.0%	0.1%	0.1%
2003	0.7%	0.9%	0.0%	0.2%	0.4%	0.6%	0.0%	0.3%	0.2%
2004	0.8%	0.0%	0.1%	0.0%	2.8%	1.7%	0.1%	0.5%	0.2%
2005	0.9%	0.1%	0.1%	0.0%	1.2%	2.8%	0.0%	0.3%	0.1%
2006	0.8%	0.1%	0.0%	0.0%	0.2%	1.3%	0.5%	0.3%	0.1%
2007	0.9%	0.1%	0.0%	0.0%	0.2%	0.7%	1.6%	0.3%	0.1%
2008	2.2%	0.0%	0.0%	0.1%	0.2%	0.4%	0.9%	0.2%	0.0%
2009	2.4%	0.0%	0.0%	0.4%	0.1%	0.6%	1.1%	0.2%	0.0%
2010	2.3%	0.2%	0.2%	0.0%	0.2%	2.0%	0.6%	0.0%	0.3%
2011	1.3%	0.5%	0.2%	0.4%	1.8%	1.2%	0.4%	1.8%	0.1%
2012	1.0%	0.1%	0.1%	0.3%	1.0%	0.5%	0.5%	0.6%	0.0%
2013	0.7%	2.6%	0.1%	0.3%	1.2%	0.4%	1.1%	0.5%	0.1%
2014	0.6%	2.7%	0.1%	0.2%	0.3%	0.1%	0.9%	0.2%	0.1%
2015	0.5%	0.2%	0.0%	0.1%	0.0%	1.9%	0.8%	0.1%	0.1%
2016	0.6%	1.2%	0.0%	0.2%	0.0%	3.4%	0.0%	0.0%	0.2%
2017	0.5%	0.1%	0.0%	0.0%	0.0%	2.1%	0.0%	0.1%	0.1%
Total	0.9%	0.4%	0.0%	0.1%	0.5%	1.1%	0.5%	0.4%	0.1%

Table S3: Volume of catch (Gutted weight - 1000 tonnes) surrendered for demersal species in the Icelandic ITQ system between 2002-17.

Year/Species	Atlantic cod	Haddock	Saithe	Redfish	Ling	Tusk	Atlantic wolffish	Monkfish	Other species
2002	0.48	0.09	0.02	0.04	0.00	0.00	0.00	0.00	0.03
2003	1.10	0.39	0.01	0.12	0.01	0.02	0.00	0.00	0.08
2004	1.32	0.01	0.05	0.02	0.07	0.05	0.01	0.01	0.07
2005	1.53	0.07	0.04	0.01	0.04	0.09	0.00	0.01	0.04
2006	1.25	0.08	0.01	0.01	0.01	0.04	0.06	0.01	0.03
2007	1.37	0.05	0.01	0.00	0.01	0.03	0.17	0.01	0.02
2008	2.34	0.02	0.01	0.06	0.01	0.02	0.10	0.00	0.01

2009	3.26	0.01	0.01	0.18	0.01	0.03	0.12	0.01	0.01
2010	2.85	0.09	0.07	0.02	0.01	0.10	0.06	0.00	0.06
2011	1.75	0.21	0.06	0.16	0.11	0.06	0.04	0.04	0.05
2012	1.38	0.03	0.05	0.12	0.07	0.03	0.05	0.02	0.00
2013	1.14	0.76	0.03	0.13	0.11	0.02	0.08	0.01	0.03
2014	0.99	0.79	0.02	0.08	0.03	0.00	0.05	0.00	0.04
2015	0.88	0.06	0.00	0.06	0.00	0.06	0.05	0.00	0.03
2016	1.10	0.34	0.00	0.08	0.01	0.09	0.00	0.00	0.06
2017	0.94	0.03	0.00	0.02	0.00	0.06	0.00	0.00	0.04

Supplementary methods section

Indicators

Nine predictor variables were used relating to fishing vessel (i) or species and time (s,t) vessel characteristics (Table S4). All continuous variables were standardized to have a mean of 0 and standard deviation of 1.

Arbitrage potential: This quantity represents the lease price of a species normalised for its cod equivalent value, to allow for the differences in cost of conversion under transformation:

$$A_s = \frac{L_s}{CE_s} \quad (1)$$

Where A_s is the adjusted lease price for a given species, L_s is the lease price for a given species and CE_s is the CE value of a given species.

The quantity can be compared across species in a given year to gauge the potential for transformation-based arbitrage. Parity between two species implies there is no opportunity for arbitrage since the cost of leasing quota in species A is equivalent to the cost of leasing quota in species B and then transforming that quota into quota for species A. Quota lease prices are based on gutted weight and were therefore converted to whole weight, upon which ex-vessel prices are based, by multiplying values with the standard gutting ratios set by the directorate of fisheries.

Choke indicator: Holland (2013)¹ suggested that lease prices in excess of ex-vessel prices might be expected for choke species on the basis that the quota lease price of the choke species will reflect not only its value but also the value of companion species which could not otherwise be caught. For each species and fishing year, we calculated the ratio of the average quota lease price to average ex-vessel price. Ex-vessel prices were adjusted for crew share (24.5%) and other variable costs such as costs for fuel and the quota fee in order to better capture the marginal value of fish caught to the quota holder.

Total allowable catch (TAC): We also included the relative TAC as a possible driver as it is an indication of biomass. Species with higher biomass are more likely to be targets than low biomass species⁵.

Targeting indicator: We created a potential index of targetability for each species, based on the percentage of the species that was caught on trips where the species accounted for at least two thirds of total trip catch and could therefore be classified as the target species. The value of this index is expected to be higher for species that have a high biomass or aggregate closely (for instance during the spawning season). As TAC broadly reflects relative biomass, we tested for correlation between this indicator and TAC, with the results being within reasonable bounds ($R^2 = 0.28$).

Gross tonnage: We also included two vessel level characteristics; gross tonnage and permit type.

Permit type: There are many different permit types in the Icelandic demersal fishery, but the most important distinction is between smaller boats (<30 gross tonnage) which are only allowed to use hook and line gear, and larger boats that are likely to use trawling gear. Boats

that are smaller than 10 gross tonnage always belong to this permit type, also if they use different gear types (e.g. gillnets). Temporary and permanent trade between these two market segments is restricted; quota are allowed to flow from the large boat permit type to the smaller types but not vice versa.

Company size indicator: We included the total demersal quota holdings of the company in cod equivalence. It is the total demersal quota holdings that inform the limit of species transformation usage (Table S1).

Multiple vessels:

If companies have multiple vessels they have more leeway to use the system in a systematic manner. A company can for instance intentionally create a deficit in a species by leasing out quota to another vessel within the company and use the species transformation system to cover the catch.

Rule changes: The series of rule changes that took place in 2011/2012 is implemented as a dummy variable (before/after rule changes) in the models. The rule changes are described in more detail above in appendix A.

Random effects:

We added the following random effects to the model: the fishing year and an identifier for each vessel. The vessel level identifier is included because several observations are taken from each vessel (longitudinally) which are not independent. Similarly, observations within the same year are expected to be more similar as observations across years and therefore non-independent. We therefore grouped the observations at the level of these two important variables. In the single multi-species model also the species were added as a random effect as species biological characteristics and economic characteristics other than investigated with the fixed effects described above were expected to determine a large proportion of the variation (i.e. observations within a species are not independent).

Table S4. Predictor variables tested in mixed effects models.

Predictor variable	Unit	Levels (raw)
arbitrage potential (by species)	Unitless	Continuous variable, Min = -95.7, Max = 176.8
choke indicator (by species)	Unitless	Dummy variable, 1 = choke, 0 = no choke
TAC (by species)	Tons	Continuous variable > 0. Min = 176, Max = 86980
targeting indicator (by species)	Unitless	Continuous variable with range {0,1}.
permit type (by vessel)	Unitless	2 levels: - small boat - large boat
gross tonnage	Tons	Continuous variable with range {1, 7682}

demersal quota holdings (by company)	CE tons	Continuous variable => 0. Min = 0, Max = 40568493
multiple vessels (by company)	Unitless	Categorical, yes or no.
rule change	Unitless	Before / After 2011/2012
species (random)	Unitless	15 species
period (random)	Year	2001/2 -2016/7
vessel ID (random)	Unitless	Dummy variable of 1824 different vessels.

Fit of Gamma distributions

We tested for the fit of gamma distributions and log-normal distributions (for each species-year combination) using a Kolmogorov-Smirnov test. In the case of a better fit for the log-normal distribution we would still be able to use the gamma-distribution since Firth (1988) worked out that model estimates for data that follows a log-normal distribution are more reliable using the gamma distribution and non-transformed data. Only few species-year (26 of 220) did not follow either the log-normal or gamma distribution, but deviations as judged by visual inspection were only small and judged to probably not bias the estimates.

Total catch, quota and balancing activity by species for the period 2001/2-2015/16

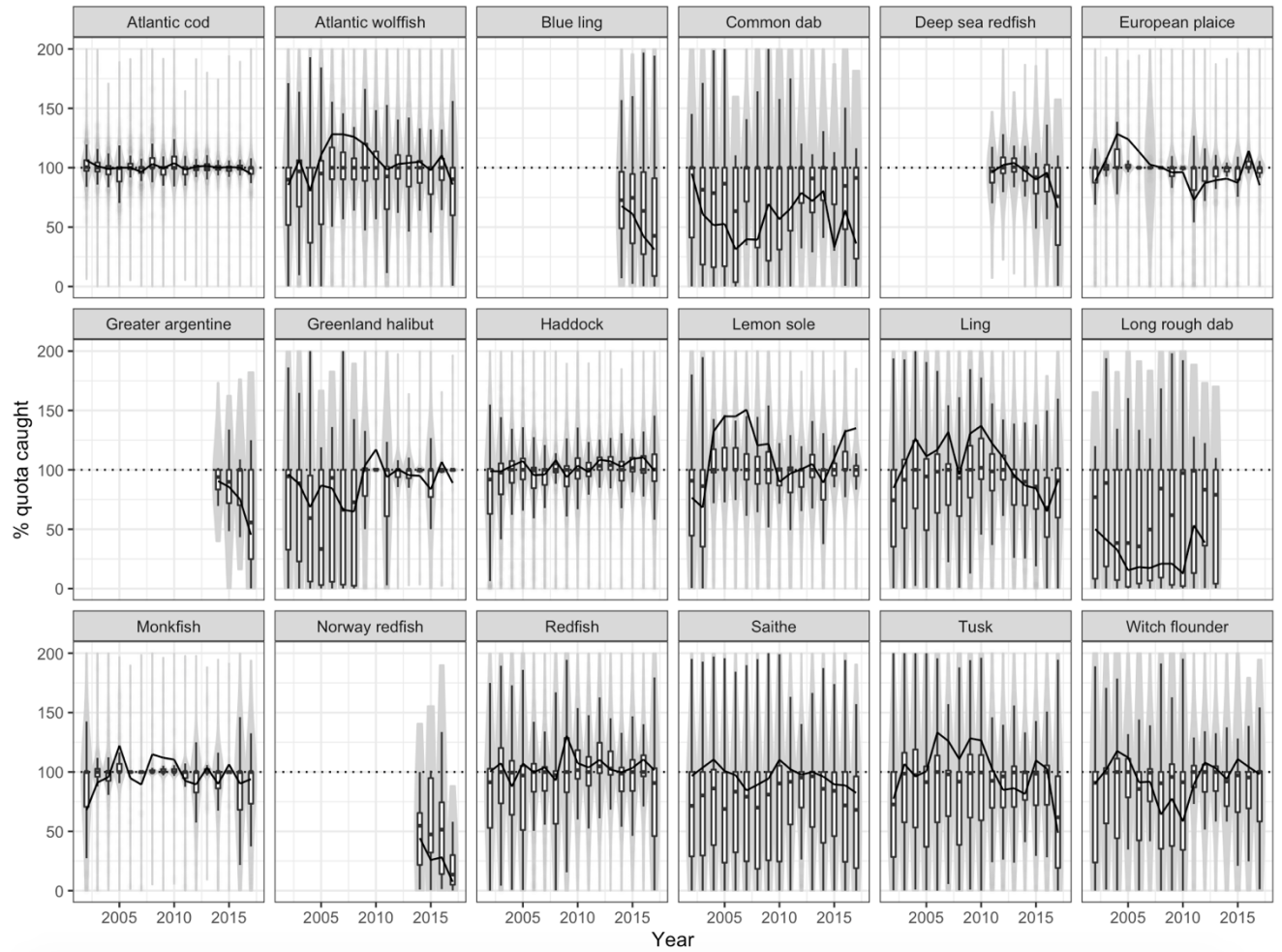
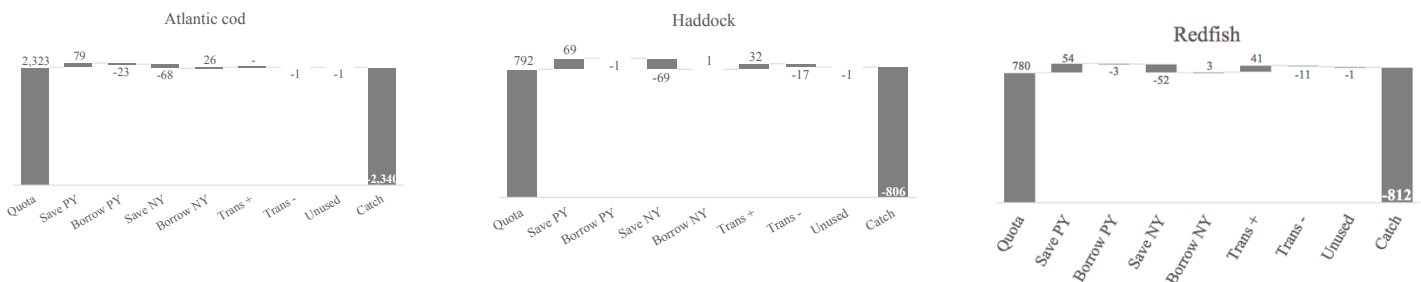


Figure S2: % total allowable catch (TAC) filled (black continuous line, at fleet level) and % allocated and leased quota used (violin and box-plots, at vessel level). It can be seen from the figure that there is a large display of variability in catch compared to quota for the different vessels in the Icelandic demersal fleet. This variability is notably lower for some species, especially Atlantic cod has almost perfect matching, while European plaice and monkfish are also relatively well aligned.



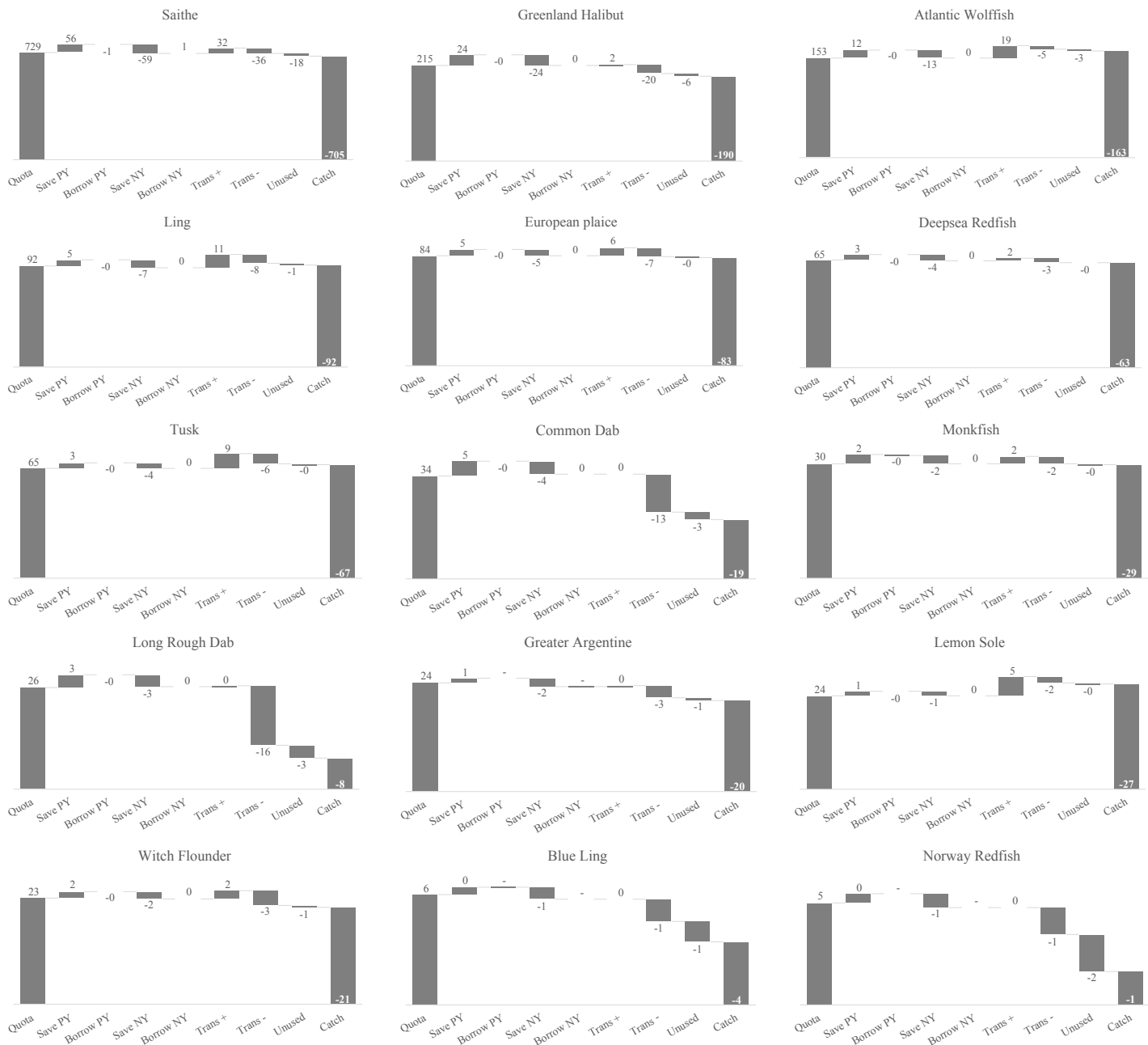


Figure S3: Bridge charts displaying cumulative allocated quota, catch, unused quota and use of balancing mechanisms per species for the whole period studied (2001-2017). Charts are sorted from highest to lowest allocated catch. “Save PY” is the quota saved from the previous year for use in the current year and therefore increases the amount of fish that can be caught, while “Borrow PY” refers to the amount borrowed in the previous year from the current year and reducing the amount of fish that can be caught. “Save NY” and “Borrow NY” are the corresponding terms relating to the next year. “Trans +” and “Trans -” refer to positive and negative transformations. It can be seen from the figure that species with lower allocated quota (e.g. common dab, long rough dab, greater argentine, blue ling, Norway redfish) have mainly negative transformations, while species with larger amounts of allocated quota (notably redfish and haddock) tend to have more positive than negative transformations. There are also some examples of the opposite: Greenland halibut has a relatively large amount of quota but has mostly negative transformations while lemon sole is a small biomass species with a relatively large amount of positive species transformations compared to negative.

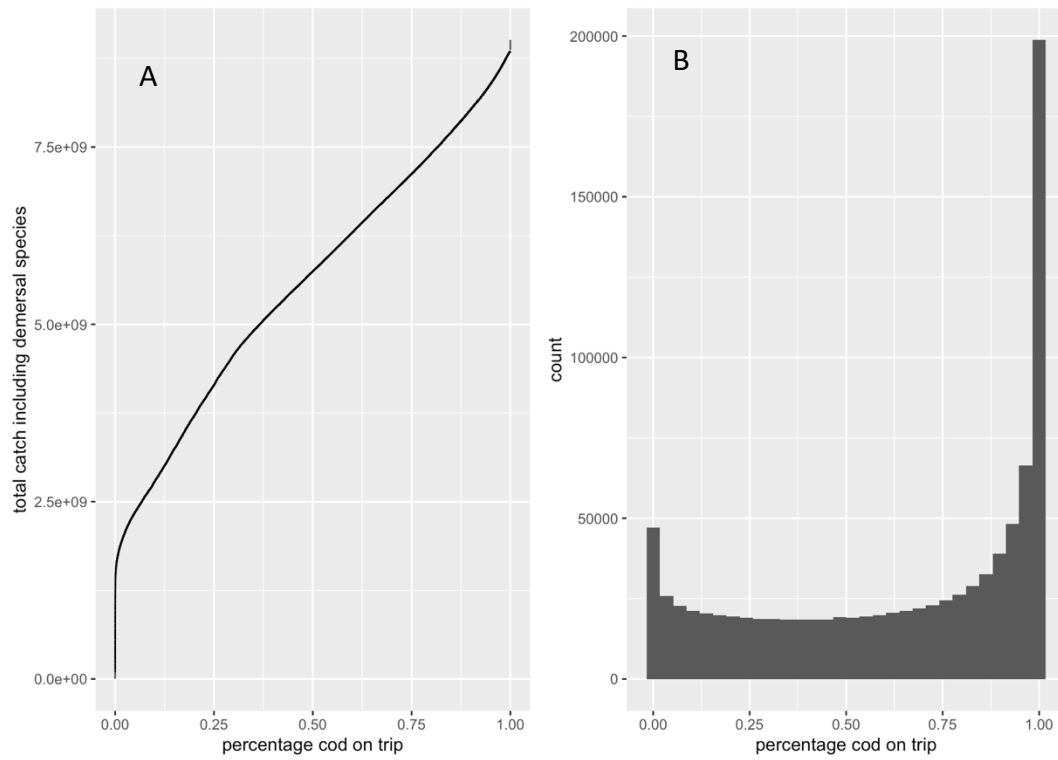


Figure S4. Percentage Atlantic cod per trip: A) percentage cod per trip compared to cumulative catch over the whole time period, the largest volume of catch is caught on trips where Atlantic cod is present. B) Histogram of cod percentage per trip, Atlantic cod is part of the species mix in the vast majority of trips.

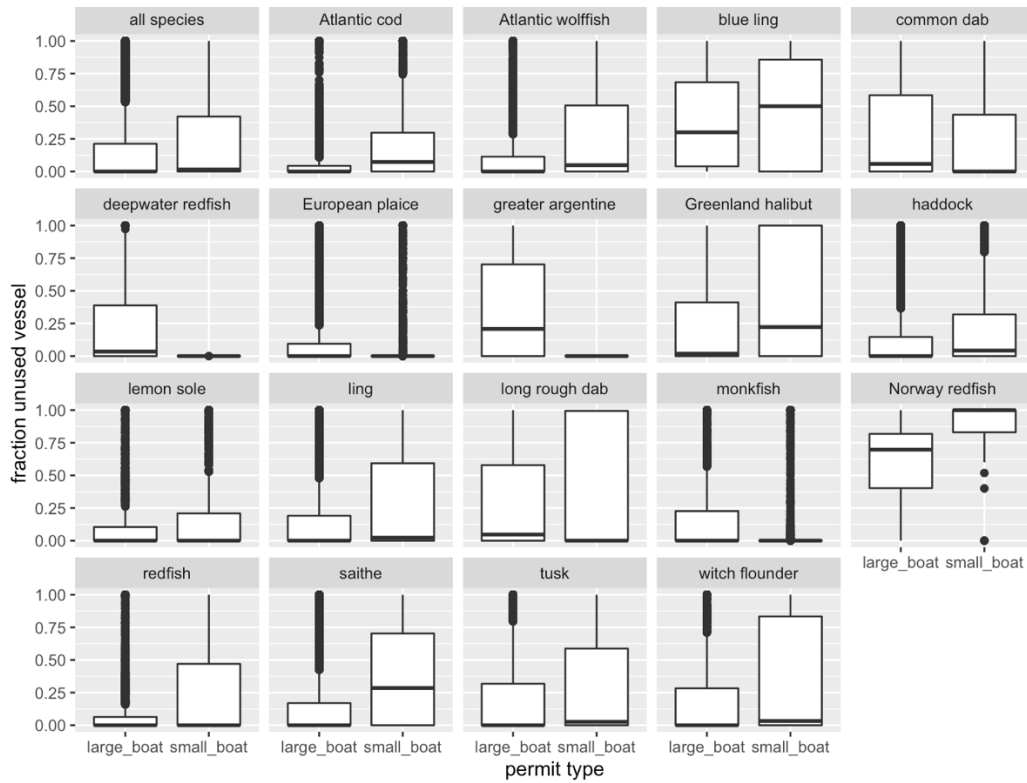


Figure S5. Fraction unused quota by permit type, for all demersal species combined (first panel) and all demersal species individually.

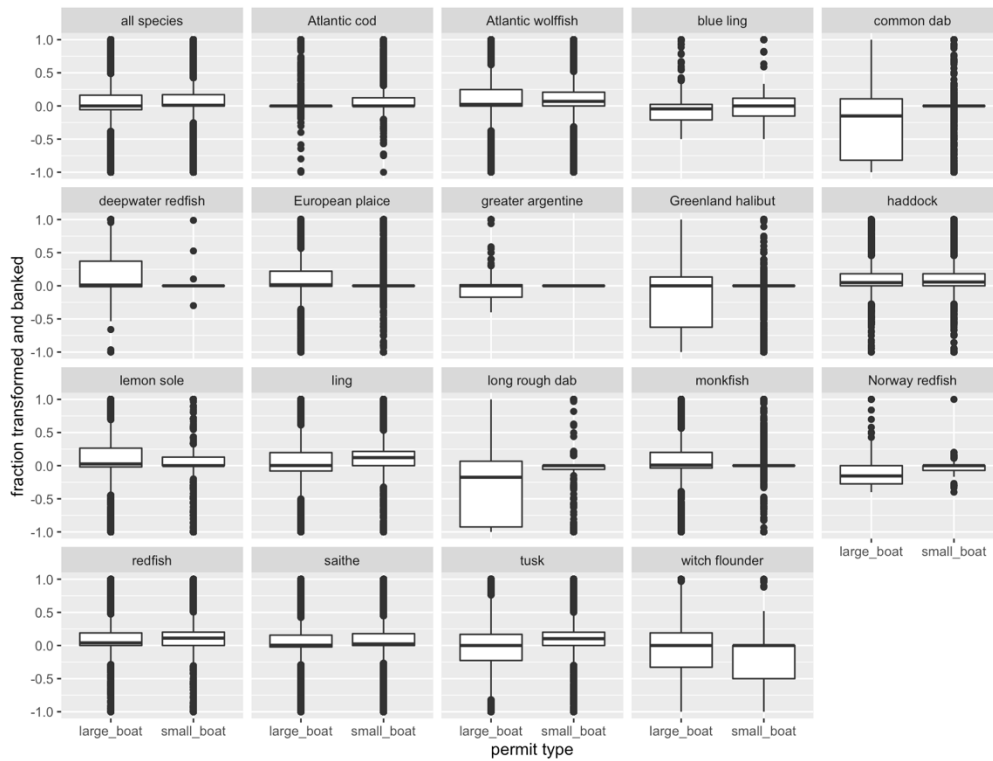


Figure S6. Fraction banked and transformed quota by permit type, for all demersal species combined (first panel) and all demersal species individually.

Interactions between banking, leasing and species transformations

Table S5. Total species transformation volume coinciding with offsetting leases over 2001/2-2016/17.

Species	(1) POSITIVE TRANSFORMATIONS (into the species)				(2) NEGATIVE TRANSFORMATIONS (out of the species)			
	All vessels	Only leasing out	Offset volume	% Total	All vessels	Only leasing in	Offset volume	% Total
Angler	2	1	1	27%	-	2	1	44%
Greater Argentine	197	103	47	24%	-	4,589	2,812	55%
Blue Ling	26	0	0	2%	-	1,744	1,056	46%
Common Dab	255	30	30	12%	-	15,964	11,385	62%
Cod	-	-	-	-	-	2,170	1,211	53%
Deepsea Redfish	1,800	294	239	13%	-	4,222	2,536	54%
Haddock	35,469	15,779	14,302	40%	-	17,272	8,511	47%
Greenland Halibut	3,273	930	815	25%	-	19,771	11,793	54%
Ling	11,507	4,437	3,049	26%	-	8,757	5,447	49%
Long Rough Dab	237	14	11	5%	-	17,478	14,072	70%
Lemon Sole	5,532	2,125	1,413	26%	-	1,812	1,141	49%
Plaice	7,508	2,041	1,464	19%	-	8,080	6,002	67%
Redfish	46,592	18,829	16,522	35%	-	11,466	6,105	48%
Saithe	33,717	11,328	10,174	30%	-	39,571	22,964	50%
Small Redfish	7	-	-	0%	-	1,660	915	41%
Tusk	9,164	2,791	1,894	21%	-	6,638	4,010	43%
Witch Flounder	1,956	451	282	14%	-	3,274	1,876	51%
Atlantic Wolffish	19,246	9,550	6,556	34%	-	7,008	4,050	54%
Grand Total	178,349	69,366	57,299	32%	-	173,451	106,882	54%

Table S6. Species transformations that could have been avoided if catch-quota needed to be matched first at the company level before using the species transformation system:

<i>Species</i>	<i>% positive transformation while borrowing from next year</i>	<i>% net transformed positive transformation while saving to next year</i>
<i>monkfish</i>	9.26	0.00
<i>Atlantic wolffish</i>	12.38	19.63
<i>Blue ling</i>	18.14	0.00
<i>Common dab</i>	0.07	0.00
<i>Deep sea redfish</i>	20.02	0.00
<i>European plaice</i>	7.66	0.00
<i>Greater Argentine</i>	0.39	0.00
<i>Greenland halibut</i>	20.84	0.00
<i>Haddock</i>	27.68	53.31
<i>Lemon sole</i>	5.41	8.01
<i>Ling</i>	7.67	31.90
<i>Long rough dab</i>	1.28	0.00
<i>Norway redfish</i>	0.01	0.00
<i>Redfish</i>	18.51	24.52
<i>Saithe</i>	21.70	0.00
<i>Tusk</i>	9.27	33.85
<i>Witch flounder</i>	4.51	0.00

Sensitivity analysis; excluding fuel costs from fixed costs in calculating the choke indicator

Compared to the more conservative choke indicator including fuel cost, excluding the cost of fuel in the indicator reduces the amount of choke observations substantially. While including fuel costs we found 27 choke observations of 9 species as well as Atlantic cod for all years studied (a total of 43 choke observations). Excluding the cost of fuel only 9 choke-observations remain of 5 species as well as Atlantic cod for all years studied (a total of 25 choke observations) (Figure S7).

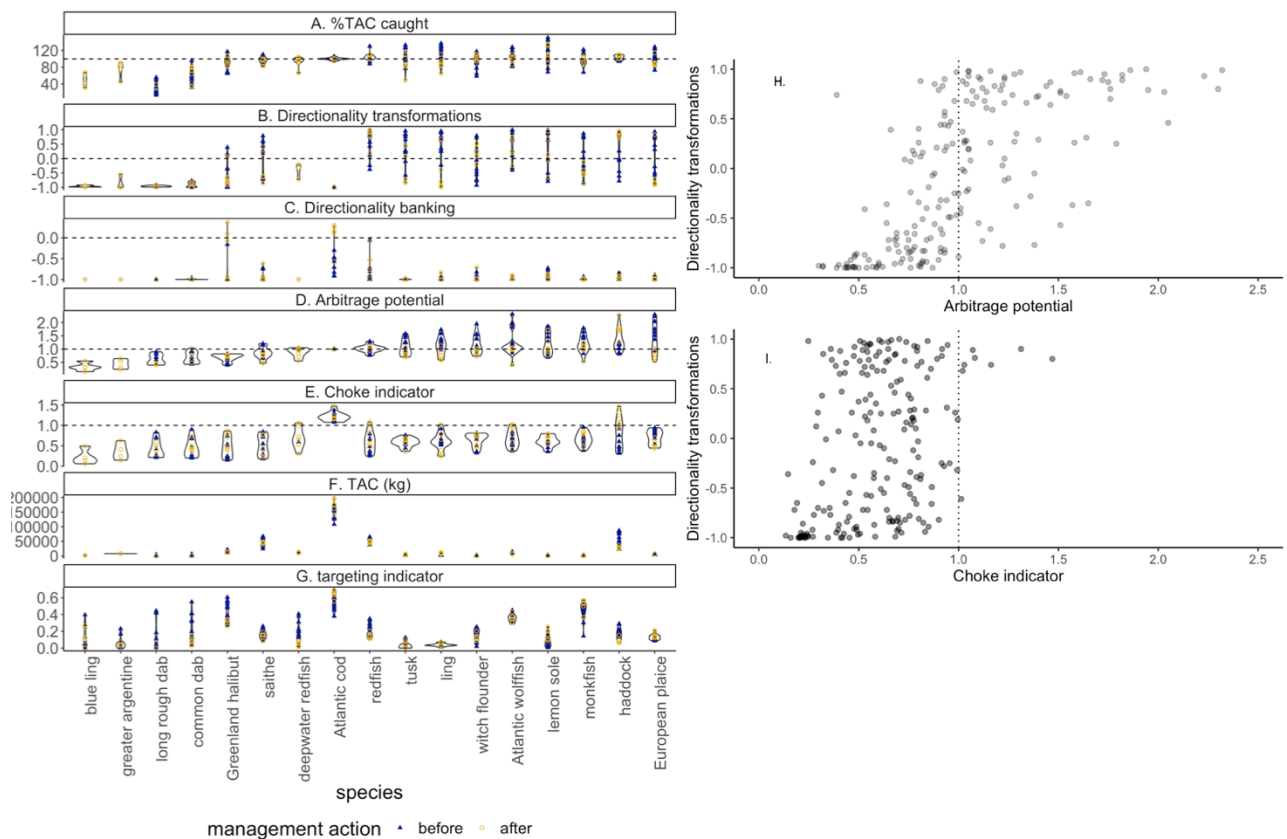


Figure S7: Variables and indicators for the Icelandic catch-quota balancing system. (A) %TAC caught; (B) Directionality of transformations: positive values indicate transformations into the species and negative values indicate transformations out of the species; (C) Directionality of Banking: positive values indicate borrowing from next year and negative values indicate saving to next year; (D) Arbitrage potential: a value < 1 means that it would be cheaper to lease quota in the corresponding species and then transform into a basket of the other species rather than lease quota for the basket directly, a value > 1 implies the inverse; (E) Choke indicator: > 1 indicates that the cost of leasing quota exceeds landed value, net of fishermen's catch share; (F) TAC in kilos of gutted fish; (G) Targeting indicator; (H) Directionality of transformations as a function of arbitrage potential (excluding Atlantic cod); and (I) Directionality of transformations as a function of choke indicator (excluding Atlantic cod). Blue triangles indicate observations before the rule change in 2011/2012 which limited flexibility in transformation usage, while yellow points indicate observations after the rule change. Species are organized from lowest to highest mean values for *arbitrage potential*. Note that a few species have fewer observations as they were added later to the species transformation system (blue ling, greater argentine and deep-sea redfish). For each species in A-G the violin plot indicating the data frequency of distribution is also plotted.

Directionality model: while the coefficient for the choke effect is different it is still not significant, the significant effects in the model are thus similar and the major conclusions that we discuss in the text still hold under the alternative assumption.

Table S7: Directionality of transformations model with fractional logit estimates of the contribution of each of the predictor variables, standard errors, z values and probabilities. It can be observed that *arbitrage potential* and TAC are the most important predictors of transformation directionality with a positive effect. Asterisks represent significance levels, * at the 0.05 level ** at the 0.01 level and *** at the <0.01 level. Predictor variables that were included are (continuous variables are indicated as c, dummy variables as d, ranges are specified) **1.** *arbitrage potential* (lease price over cod equivalent conversion ratio, normalised; c {95.7; 176.8}), **2.** *choke indicator* (lease price rises above ex-vessel price plus marginal costs, excluding fuel); d {choke observation, no choke observation}), **3.** total allowable catch (TAC) (c {176; 86980}) **4.** Targeting indicator, % catch of a species for which a species is 2/3 of the catch (c {0;1})

Predictor	Estimate	standard error	z value	Pr(> z)
Arbitrage potential	1.43	0.20	7.14	< 0.001***
Choke indicator	0.77	1.35	0.58	0.56
Total allowable catch (TAC)	0.95	0.29	3.23	0.001***
Targeting indicator	-0.09	0.15	-0.62	0.54
Targeting indicator * arbitrage potential	-0.18	0.15	-1.22	0.22
Cox & Snell's R ² = 0.40				
Nagelkerke's R ² = 0.52				

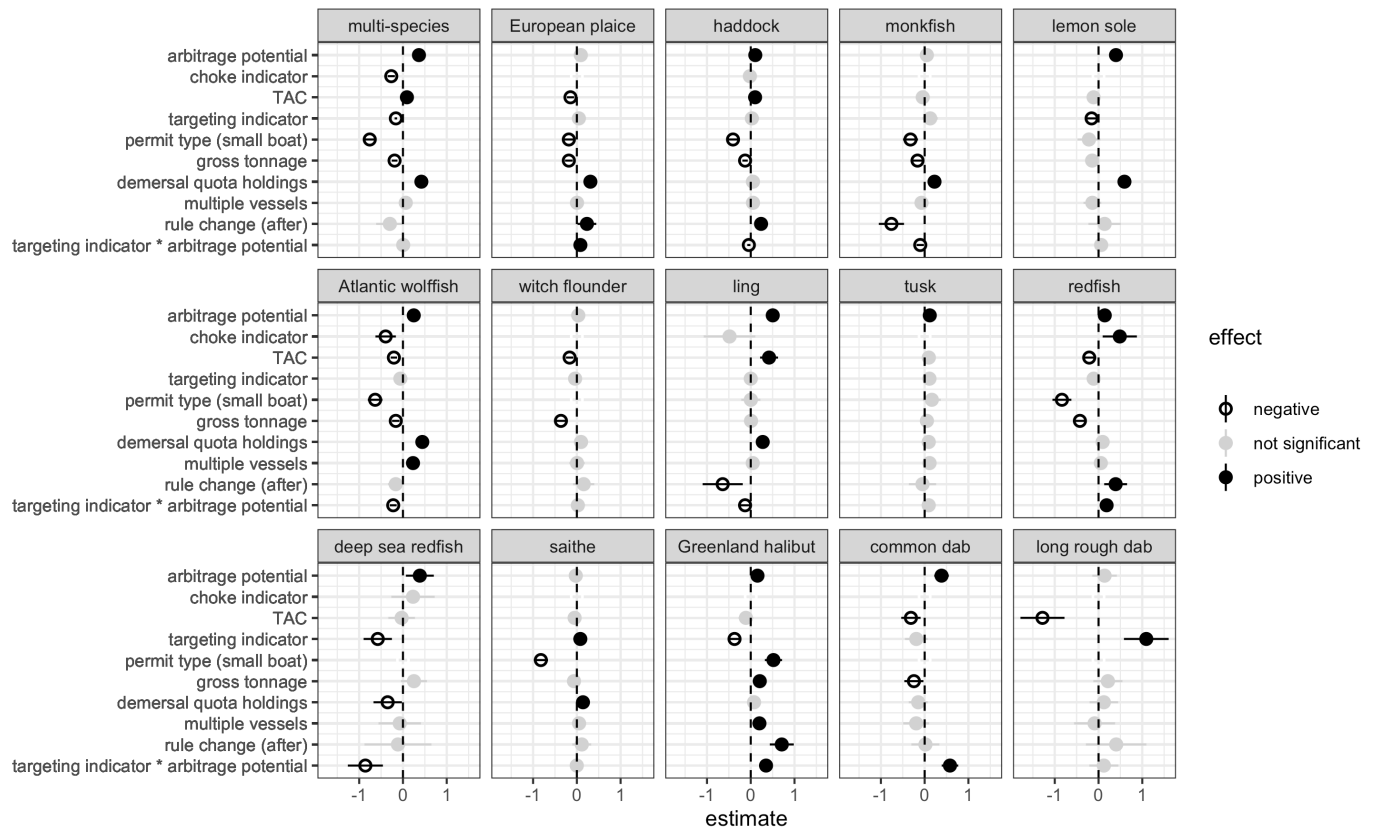


Figure S8: Effect size for predictors predicting catch/quota in multi-species and single species models calculating the choke indicator excluding fuel costs. Atlantic cod and a few species with too few observations were excluded. Effect size is represented by the location of the dots in the estimate values range' along with 95% confidence intervals. When confidence intervals cross the dotted line at 0, the predictor variable is considered not significant (note that because there is a large amount of data underlying the figure the CI is often very narrow and not visible in the figure). The exponent of the effect size is the predicted increase in catch/quota for each unit increase in the predictor variable (e.g. on average for all species the ratio of catch:quota is predicted to increase with 1.4 with an increase of 1 in arbitration potential all else being equal). Predictor variables that were included are (continuous variables are indicated as c, dummy variables as d, ranges are specified) **1.** *arbitrage potential* (lease price over cod equivalent conversion ratio, normalised; c {95.7; 176.8}), **2.** *choke indicator* (lease price rises above ex-vessel price plus marginal costs, excluding fuel); d {choke observation, no choke observation}), **3.** total allowable catch (TAC) (c {176; 86980}), **4.** Targeting indicator, % catch of a species for which a species is 2/3 of the catch (c {0;1}), **5.** a variable that represents the two main fleet segments (d {small boats with passive gear, larger boats with mostly active gear}), **6.** the gross tonnage of the vessel (c {1; 7682}), **7.** the amount of demersal quota held by the company operating the vessel (c {0;40568493}), **8.** whether the company operating the vessels has multiple vessels (d {single vessel, multiple vessels}), **9.** the management adjustments (rule change) (d {2011/2012 and prior, after 2011/2012}) as well as **10.** the interaction effect for the *targeting indicator* and the *arbitrage potential*. The small-boat permit variable is not included for species that were not caught by small boats and the choke indicator is not included for species that were never indicated as a choke species. Observations with negative allowed catch amounts (caused by borrowing and having only a small amount or no allocated quota) were excluded, representing 0.4% of catches of the Icelandic demersal fleet.

Vessel-level models: The choke indicator becomes significant and negative when excluding the cost of fuel in the multi-species vessel level model, while in the model which considers the cost of fuel when calculating the choke indicator, the coefficient was negative but not

significant. The small and significantly positive effect for the interaction between targeting and arbitrage is no longer significant in the model with less choke observations.

Due to the smaller amount of observations of choke effects under the alternative interpretation of fuel costs, the choke effect is modeled in fewer single species models. i.e. it's no longer included in the models for Monkfish, Greenland halibut, Common dab, and European plaice. The single species models with choke effects look largely similar except for the model for Atlantic wolffish: the coefficient for the choke effect is now negative and significant when excluding the cost of fuel and for ling the reverse is true, while in the model including the cost of fuel the choke indicator was negative and significant, in the model excluding the cost of fuel the coefficient is negative but no longer significant.

All choke observations that do not incorporate the fuel price when calculating the choke indicator occur after the rule change. Lease prices tend to be higher for more commercially interesting species and transformation into these species is lower after the rule change as is reflected in the negative coefficient for the rule change for a few of these single species models (e.g. Atlantic wolffish, haddock). The lower amount of transformation into such species with generally higher lease prices may be reflected in the negative coefficient for the choke indicator in the multi-species model.

From the analysis in this appendix and the analysis presented in the main text we conclude that choke effects may be present in the Icelandic system but that they account for a small minority of the observations for all species years, except for haddock in recent years and Atlantic cod in all years. This is also reflected in the fact that directionality of banking is mainly negative except for a few species-years. The fact that there are so few cases in Iceland where average lease price rises steeply above ex-vessel price (except for Atlantic cod) could possibly in part be explained by the species transformation system itself. Small biomass species are unlikely to become choke species as there are few cases in which the maximum amount of transformations is ever used (since this maximum is limited by the total amount demersal quota holdings, see Figure 1 in the main text). Larger biomass species are more likely to become choke species simply because the transformation rules are more restricting and quota could become sparse this is also reflected in the percentage of positive transformations for which the limits are met as presented in table S8, this is only a substantial amount of the observations for Haddock and Redfish and mainly after the management changes in 2011/12).

Moreover, (despite trying to calculate an indicator of targeting) distinguishing between target and bycatch species is a difficult issue in multi-species fisheries. It is very likely that fishers rather catch/target mixes of species and that they are uncertain a priori about the exact amount of species that will be caught, with some species becoming choke species as discussed in ². This is reflected in the fact that most choke observations are in fact in higher biomass species such as Haddock and Redfish (and Atlantic cod!) which are the main contributors to the demersal catch value after Atlantic cod³. This is not unique to the Icelandic system, it's more frequently the case that important commercial species that are important targeted species end up being choke species^{4,5}. We therefore chose to include the fuel cost when calculating the choke indicator in the main body of the text, but we do acknowledge that including the cost of fuel is an important assumption.

Table S8: Transformation activity of vessels relative to limits. The upper rows relates to the period 2000/1-2010/11 and the lower rows to the period 2011/12-2016/17. Vessels are assessed as being at the transformation limit if within 1% of the negative limit or 0.1% of the positive limit.



Paper III

Structure and evolution of cod quota markets networks in Iceland over times of financial volatility

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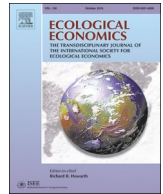
Author contributions: MO conceived of the research idea, lead the data collection and carried out the analysis, MO wrote the article with help of MJS, SA and PJW.



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Structure and evolution of cod quota market networks in Iceland over times of financial volatility



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ABSTRACT

The quota market is the instrument through which fishing operations under an individual transferable quota (ITQ) system achieve greater efficiency. It allows fishing companies to optimally configure their quota portfolios to their catches. Globally, fisheries corresponding to ~25% of landings have adopted ITQ systems. However, there is surprisingly little empirical information on quota markets functioning. Here we study the development of quota share and lease markets and assess market activity and functioning for the cod fishery in Iceland. We use a social network analysis to assess changes in four Icelandic quota markets, distinguished by boat size (large versus small) and permanence of transfers (leases versus shares). The quota market for permanent trades in small-boat quota shows a sharp increase in trade and network connectivity between 2004 and 2006, resulting in a high rate of quota concentration. The quota markets for permanent quota shares were the most fragmented and sparse during the years of the financial crash in Iceland and never regain the same activity. Our results suggest that quota systems evolve towards a consolidated state and that their markets are not entirely resilient to financial instability. We also found some evidence that better-connected traders could sell quota at higher prices in the lease markets, though price dispersion was generally low.

1. Introduction

Exploitation of fisheries resources is a prime example of the tragedy of the commons, due to the “race to fish” (Birkenbach et al., 2017) as well as overfishing (Worm et al., 2006; Berkes et al., 2006). The introduction of property rights is often proposed as a solution to common-pool resource problems, as such rights clarify the boundaries of ownership and rules of access to resources, thereby reducing competition and incentivizing long-term stewardship of the individually owned resource (Schlager and Ostrom, 1992). For marine resources, the most common approach for implementing property rights is the introduction of fishing quota that are allocated to individuals, boats, or communities (EDF, 2018). In 2013, nearly 200 rights-based fisheries management programs existed worldwide and were implemented in 40 countries, affecting > 500 different marine species (EDF, 2018). In approximately 80% of those systems, the quotas were transferable (Carothers and Chambers, 2012). Studies on quota systems (including ITQ systems)

showed benefits that included decreased fishing mortality (Melnychuk et al., 2016) and a decrease in the race to fish (Birkenbach et al., 2017).

Considerable efficiency gains have been shown to arise from the implementation of an ITQ system, the main driver of which has been decreased harvesting costs as the fleet size decreases (Grafton, 1996). The fleet size decreases when less profitable harvesters choose to leave the industry and quota consolidate with the remaining harvesters (Asche et al., 2008). Consolidation can occur on two levels: accumulation of quota onto fewer vessels and accumulation of quota into the hands of fewer holders (often also due to mergers of companies). The exit of the less profitable harvesters from the fishery can reduce overcapacity, a necessary step towards economic sustainability in fisheries (Branch, 2009). However, quota consolidation can also occur in a fleet where overcapacity is no longer a problem, caused by other economic benefits such as profitability and the resilience gained by owning a diversity of fishing quota (Agnarsson et al., 2016; Holland et al., 2017). Although consolidation is expected to be fastest soon after a quota

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system has been introduced as many fishers decide to leave the industry, in some cases, consolidation continues long after this initial period (Agnarsson et al., 2016; Chambers and Carothers, 2017). It can be perceived as generating negative societal consequences as well, as shown for 39 fisheries in Iceland (Agnarsson et al., 2016; Edvardsson et al., 2018; Pálsson and Helgason, 1995), Canadian fisheries (Pinkerton and Edwards, 2009), and US fisheries (Himes-cornell and Hoelting, 2015; McCay et al., 1995), especially where consolidation has led to decreased participation of small-boat fishers, decreased employment, and flow of quota away from remote rural areas with little other employment (Chambers and Carothers, 2017).

1.1. Quota consolidation and market functioning

Although quota consolidation is an expected and intended consequence of introducing an ITQ system, indicating that in some cases it can reflect a well-functioning and efficient quota market, many of the criticisms of ITQ systems arise from a combination of market inefficiencies and the common practice of grandfathering of quota rights (i.e., only current harvesters receive quota allocations at no cost). The main justification for implementing transferable quotas in many fisheries is that through trading and leasing, quota should flow to the most efficient operators, a common assumption underlying the implementation of many market-based management mechanisms for renewable resources, such as water markets (Loch et al., 2018) and carbon markets (Starkey, 2012). In a fully efficient quota market, there would be symmetric information among participants and there would be no market distortions such as speculative behavior (Matthiasson, 2014) and transaction costs should not be prohibitively high (Innes et al., 2014; Loch et al., 2018). The quota price in an efficient market should reflect future revenue and incorporate the expectation of changes in future profitability. Following this theory there should be a relationship between stock productivity and both permanent quota sales and temporary quota lease prices (Arnason, 2005; Batstone and Sharp, 2003; Newell et al., 2005). Quota prices are therefore critical for informing operators on decisions of increasing or reducing their ownership of quota shares as well as shorter term decisions on leasing quota within a fishing season. The efficient quota market could then allow for the optimal distribution of quota among the most cost-effective operators (Newell et al., 2005).

However, evidence to support efficient market functioning or reveal inefficiencies is sparse. Empirical studies on the functioning of quota markets remain relatively few (Newell et al., 2005; Holland, 2013;) and most focus on the seasonal leasing of quota (Pinkerton and Edwards, 2010; van Putten et al., 2011; van Putten and Gardner, 2010). Newell et al. (2005) show an active market for both quota sales and leases and a decrease of price dispersion over time, indicating a well-functioning quota market in New Zealand. Price dispersion, according to efficient market theory, should go down over time as traders learn what a reasonable price for quota is. Moreover, especially in quota lease markets the fact that traders are both selling and buying should accelerate this process (Ropicki and Larkin, 2014). On the other hand, due to the often decentralized nature of quota markets price dispersion can be expected to be higher than in more central commodity markets (Ropicki and Larkin, 2014). Grainger and Costello (2012) found that the security of property rights is reflected in the price of quota, in New Zealand, USA and Canada. Other studies show that quota markets do not always function as expected (Pinkerton and Edwards, 2009), and have known market inefficiencies, namely: (1) large wealth effects from the original grandfathering of quota that prevent new fishers from entering the market due to a lack of capital (Pinkerton and Edwards, 2009); (2) informal functioning, causing greater trade among owners that have personal connections and causing asymmetric information held by buyers and sellers (Chambers and Carothers, 2017; Dobeson, 2018); (3) distortion caused, for example, by speculative investments (BC Halibut fishery, Pinkerton and Edwards, 2009); and (4) likely high transaction

costs as large amounts of quota remain inactive (for example in a Great Barrier Reef fin-fish fishery, Innes et al., 2014).

These examples illustrate the need to better understand the performance of trade markets for quota shares to evaluate whether the ITQ management system is adequately delivering its intended goals. Moreover, financial instability has not been investigated as a challenge to the performance of an ITQ system. For example, the financial crisis has been earlier described as having an impact on Icelandic companies' ability to trade quota, mainly because there was, prior to the crisis, a great willingness of the banks to provide loans to Icelandic fishing companies since the early 2000s and loans were provided for other industry spending using fishing quota as a collateral (e.g., Matthiasson, 2012; Gunnlaugsson and Saevaldsson, 2016).

In this paper, we investigated how the largest quota markets for shares and leases in Iceland function, by exploiting a large data set of quota trades spanning a decade (2004–2016). We also looked into the effects of financial volatility on the trade activity. The analysis was performed for two separate markets, large-boat quota versus small-boat quota, the latter of which operates in relation to a more recently introduced quota system (1991 versus 2001). To do so, network analysis was used. In the context of quota markets, network analyses have been used to study connections between processors and quota owners in quota lease markets (van Putten et al., 2011) and the relation between the amount of fish landed and quota owned (Innes et al., 2014). Van Putten et al., (2011) described the occurrence of “broker nodes” in a lease quota network where traders use so called brokers to lease out their quota for them, thereby increasing lease prices. Ropicki and Larkin (2014) regressed network features such as degree and centrality to price dispersion in a quota lease market and found that more central nodes in the network leased quota in at lower prices. However, these studies did not concern quota share markets and often drivers for quota trade were not investigated, with the exception of Leon et al. (2015) who found that the amount of allocated quota was a better prediction for the likelihood of a fisher buying quota than catch per unit effort in the Tasmanian rock lobster fishery. Since ITQ systems hold the highest costs of management per fishing boat (Beddington et al., 2007), understanding the performance of such systems is an important step in evaluating the fisheries management system overall.

1.2. The cod ITQ system in Iceland

The development of the lease and share markets for cod (*Gadus morhua*) in the Icelandic Exclusive Economic Zone were analyzed over a time of financial volatility between 2004 and 2016. Iceland was one of the first countries that implemented an individual quota program. Quotas were introduced in 1984 and were made fully transferable in 1990 (Pálsson and Helgason, 1995). This study focused on cod quota, as cod is so abundant in Icelandic waters that any demersal fishery would have to own or lease some cod quota to cover its catches (Pálsson and Helgason, 1995). Iceland has two main quota markets: a market for temporary leases and a market of permanent transfers of quota shares. Quota shares are the owners right to a percentage of each year's total allowable catch (TAC) in each fishery, and these shares are registered per boat. Once the TAC has been set, the harvest rights for the fishery in question are calculated by multiplying the boat's quota share, defined as a percentage, with the TAC. Leases are individual quota transfers by weight (kg), valid for a fishing year. These markets are further split in two, as quota from the small-boat fleet cannot flow to the “industrial” part of the fleet. The quota system for the small-boat fleet was phased in between 2001 and 2004; earlier, small boats were managed by an effort system. Currently, there are two different types of general fishing permits, which define the small- versus large-boat fleet segments. Small-boat quotas may currently only be utilized by boats smaller than a) 30 GRT that only use hand-line or long-line or b) 10 GRT that have no gear restrictions. Large boat can use any type of gear and are generally composed of vessels larger than 30 GRT. In 2013 the legal size of boat

in the small-boat segment increased from 15 GRT to the current 30 GRT. The markets for quota are informal and quota can be freely traded by individuals registered in Iceland owning a fishing vessel within each permit type. However, the Fisheries Directorate keeps track of quota transfers and concentration limits (e.g. no > 12% of cod quota in the large-boat quota market or 4% in the small-boat market can be held by an individual company). Large- and small- boat markets are also separated by the restriction that large boats may not buy or lease quota from small boats, although vice versa is permitted. Small-boat holders often rely on brokers for quota shares and leases (Dobeson, 2018) (for an example of such a broker and up-to-date prices in the small-boat system see the following link: http://kmrosa.is/Default.asp?Sid_Id=3851&tld=99&Tre_Rod=&qsr). In 2009, a new non-ITQ small-boat handline season was instituted called “coastal fishing” (“strandveiðar”) in an effort to offer access for newcomers to fishing, partly in response to rulings by the United Nations Human Rights Committee on the social equity problems of the privatization of fisheries resources in Iceland (case 1306/2004). The “coastal fishing” operates under an effort-based system in the summer months (May–August, and is allocated 2% of the cod TAC) many of the small ITQ boats currently also fish with a “coastal fishing” permit (Chambers and Carothers, 2017).

Though there is a need for quantitative analysis on these matters, the evolution of quota markets in Iceland has not yet been well studied. As an exception, a few assessments have been done regarding early stages of implementation of the Icelandic ITQ system (Eythórssón, 2000, (Matthiasson and Klemensson, 2004). Eythórssón (2000) investigated consolidation in the earlier years of the large-boat quota system. Matthiasson and Klemensson studied the differences between quota prices that were directly traded on the quota share market and quota prices that were implicit when whole firms were being bought (indirect quota trade). Recent research has shown several inefficiencies, namely speculative investment in quota shares (Gunnlaugsson and Saevaldsson, 2016) (Gunnlaugsson and Saevaldsson, 2016) as well as asymmetric information held by traders (Dobeson, 2018). In this study, we hypothesized that consolidation would be observed in the initial stage of the implementation of the quota system for the small-boat fleet due to a reduction in profitability associated with the switch to output controls, as the small-boat fleet was catching more than its allocated share of the TAC under the prior effort-based system (Haraldsson, 2008). As the large-boat fleet ITQ system was first implemented in the 1980s and made fully transferable in 1991, it was expected to have been consolidated at the start of our analysis (Agnarsson et al., 2016). Drivers for quota trade as examined by the ERG models were thus expected to relate more to efficiency than consolidation in the large-boat market, while we expected strong significant drivers for consolidation in the small-boat market. We also expected to find reduced trading as an impact of the financial crisis in 2007–2008, due to the decrease in fishing companies' abilities to finance new quota purchases (Gunnlaugsson and Saevaldsson, 2016). Moreover, we expected to find a negative relation between price dispersion and nodes with a more central position in the network for quota that was leased in and the reverse for quota that was leased out (i.e. buyers with more connections were expected to buy at lower prices and sellers with more connections were expected to sell at higher prices).

2. Research methods

2.1. Data collection and pre-processing

In Iceland, all transfers of quota must be registered by vessel on the website of the Directorate of Fisheries (DOF; www.fiskistofa.is) since the fishing year 2000–2001, and these data sets are publicly available. In this analysis, quota shares (%) and leases (kg) from 2000 to 2016 were summed by registered company, using boat registration matched to company ownership records from the DOF website. Quota shares and leases were analyzed separately as they reflect separate markets. We

acquired data on lease prices from DOF for a subset of our data (2004–2009); these data are not publicly available. Price information of permanent quota share transfers are not registered by DOF.

2.2. Cod quota market dynamics

To describe the quantity of trade activity in the cod quota system we calculated descriptive statistics of the share and lease trades for cod quota. Descriptive statistics included for each owner the: 1) frequency of trades, 2) net percentage of quota owned that was traded among owners, and 3) gross percentage of quota owned that was traded among owners. The net percentage traded was calculated by summing over all incoming and outgoing trades for each company holding quota, where an outgoing trade was calculated as negative and an incoming trade was positive. The sum of these absolute amounts divided by two is representative of the amount of quota that actually changed hands within that year (gross percentage).

2.3. Distribution of quota holdings

To describe changes in the distribution of quota holdings over time we calculated Lorenz curves (Lorenz, 1905) and Gini coefficients (Gini, 2012) for all years in which no changes in regulations took place regarding the small-boat and large-boat quota markets. Lorenz curves compare the quota distribution to a completely equal distribution of quota holdings. The Gini coefficient measures the distance between the completely equal distribution on the Lorenz curve and the actual distribution. A value of zero thus shows a completely equal distribution and a value of one shows a maximally concentrated distribution (all quota owned by a single individual).

2.4. Cod quota market network analysis

The quota market was modeled as a weighted directed network. Incoming trades indicated that quota was bought during an individual transaction and the outgoing trades indicated that quota was sold, while the weights reflected the amount of quota traded. Networks were created within years for each market, so that network statistics listed below could be calculated each year and used to describe the changes in the market structure over time.

2.4.1. How are trade connections per quota holder distributed?

We measured the number of incoming and outgoing trades (i.e., ‘ties’ in network terminology) per quota holder. These we refer to as indegree and outdegree respectively. Degree is one of the most robust measures for the importance of a node in a network (Fuller et al., 2017) (Table 1). To test whether the probability of a trade forming conforms to a scalable distribution, we fitted power-law, exponential and log-normal relationships to the in- and out-degree distributions, using methods developed by Clauset et al. (2009). This method combines goodness-of-fit tests based on the Kolmogorov–Smirnov statistic and maximum likelihood ratio tests to assess how likely it is that the observed data points were drawn from these distributions. This was done separately for indegrees and outdegrees. Such scalable distributions could be indicative of network processes such as popularity and the formation of hubs, producing a market that is not easily accessed by all individuals (Innes et al., 2014; van Putten et al., 2011). As an alternative we also fitted the log-normal distribution and measured deviations from normality using skewness and kurtosis.

2.4.2. Modularity of the quota trading networks: are some quota holders better connected than others?

Cohesiveness of the network was calculated using the fraction of the largest component (“Giant component”, Table 1) (Kim, 2013). A component is defined as a (set of) node(s) (i.e. quota holders in our case), that are connected within the network. The Giant component is the one

Table 1
Descriptive network indicators that we calculated for the small and large boat lease and share networks.

Network statistic	Formula	Description
Degree	$D_i = \sum_j A_{ij}$ where D is the degree for node i and A are the (unweighted) number of trades between the node i and another node j .	Sum of amount of incoming (buying or leasing in quota) and outgoing trades (selling or leasing out quota)
Giant component	$G = k/N$ where k is the number of nodes in the giant component and N is the total number of nodes in the network.	The fraction of nodes that are in the largest isolated component of the network.
Modularity	$Q = \left(\frac{1}{2m}\right) * \sum (A_{ij} - k_i * k_j / 2m)$ where m is the total of the weights in the graph, A_{ij} is a particular edge between node i and j in the network (More formally, the element of the A adjacency matrix in row i and column j), k_i is the degree of i , k_j is the degree of j , c_i is the type (or component) of i , c_j that of j , the sum goes over all i and j pairs of vertices. Edge weights are considered as the element of the A adjacency matrix, and k_i is the sum of weights of adjacent edges for vertex i .	The modularity of a graph with respect to some division (or node types) measures how separated the different node types are from each other. The algorithm assumes that communities in the network will be more connected among each other (Csárdi and Nepusz, 2006).
Global clustering coefficient	$C = \frac{c}{Z}$ where c is all closed triplets and Z is all triplets open and closed.	Proportion in the network of the two nearest neighbors of a node that are also nearest neighbors of each other.
Density	$D = \frac{E}{N(N-1)}$ where E is the number of all edges and N is the number of nodes in the network.	The ratio of the number of edges E (trades in our case) to the number of possible edges in a network with N nodes.
Reciprocity	$R = \frac{r}{E}$ where r is the number of reciprocal edges and E is the number of all edges.	The ratio of reciprocal edges in the graph to all edges.
Betweenness centrality	$g(v) = \sum \left(\frac{S_{st}(v)}{S_{st}} \right)$ where S_{st} is the number of shortest paths from node s to node t and $S_{st}(v)$ is the number of those paths that pass-through v .	The number of shortest paths going through a node. Measures a node relative importance/centrality in the network Csárdi and Nepusz, 2006).

that contains the largest fraction of quota holders. Therefore, connectivity is high where the Giant component spans a large portion of the whole network. As another metric of cohesiveness, we calculated modularity. A more modular network is composed of distinct groups rather than continuous relationships among quota holders (Clauset et al., 2004). We expected the lease network to be more connected and less modular than the share network, as leasing occurs more frequently than share transactions in most quota markets studied (Innes et al., 2014; van Putten et al., 2011; van Putten and Gardner, 2010). Therefore, we expected to find, especially in years of increased trade in the lease markets, a Giant Component that spanned a high percentage of quota holders and a low metric of modularity.

Two other metrics that measure the cohesiveness of the network are the network clustering coefficient and density (Gephart and Pace, 2015). The clustering coefficient measures the number of closed triangles in a network (i.e., the proportion of edges between two nodes that are both trading with a third node, Table 1). This measure is often high in social networks, and was used here to assess the tendency of groups forming (Barabasi, 2009): i.e., friends of friends are more likely to be friends. This would in our case also indicate cohesiveness. Density is the total number of actual trade routes used (i.e. edges in network terminology) divided by the total number of possible trade connections, the latter of which is simply the number of nodes N times $N-1$ (Table 1). A higher value for both clustering coefficient and density would indicate a more densely connected network. We also calculated reciprocity which is the proportion of ties in the network that is mutual. Descriptive network characteristics were calculated in RStudio (R Core Team, 2015) using the package igraph (Csardi and Nepusz, 2006).

2.4.3. Do more connected traders in the market get better prices on the lease markets?






For a small subset of lease market networks (2004–2009), we had price information connected to individual trades. Following Ropicki and Larkin, 2014 we regressed two of our (unweighted) network metrics against price dispersion for individual trades, using an ordinary least square regression to test if more important nodes in the network leased quota in for lower prices and leased quota out for higher prices. Price dispersion was calculated by the percentage deviation of actual price for the trade with a predicted price as fitted by a polynomial

regression (Ropicki and Larkin, 2014). The chosen metrics were a) closeness centrality (Table 1) and b) a dummy variable for broker nodes. We defined a broker node in the lease network as a node that had more than three incoming edges as well as outgoing edges and more than ten edges in total. We added the quantity of the trade (in kilos) as a control variable in the regression, as we assumed that trades with larger quantities would have lower prices. We scaled all continuous predictors by subtracting the mean and dividing by the standard deviation. We removed trades that obviously represented barter trades from this analysis (i.e. near zero pricing) (Ropicki and Larkin (2014)).

2.5. Exponential random graph modeling of quota share networks

To test which factors were significant for the establishment of trades in the quota share and lease markets we used Exponential Random Graph (ERG) modeling. ERG modeling is used here to help quantify the effect of characteristics of quota holders on trade formation (i.e., characteristics of ‘nodes’ on ‘edges’ in network terminology) (Kolaczyk and Csárdi, 2014; Fischer and Jasný, 2017). In our ERG model the number of trades between two quota holders can be viewed as a value in the response variable of a regression model, and the predictor variables would be internal network characteristics of the quota holders (e.g., reciprocity as described above) as well as external characteristics (e.g., vessel size). The approach is divided into two steps. In the first step, we search for a well-fitting statistical model for the empirical quota market network. This means assessing which potential predictors are significant and their effect on the overall response (number of trades in the network, i.e. presence or absence of edges between nodes). The second step uses simulations to evaluate the robustness of the model selected in step one, and how well the resulting model is able to reproduce the structural characteristics of the empirical network (Hunter et al., 2009). The outcome of the ERG modeling is the log-odds for a set of parameters, similar to those of a logistic regression. The value of the log-odds characterizes the strength and the direction (\pm) of the influence of a parameter on the likelihood of trade-formation, which is unweighted, i.e. zero or one. We conducted this analysis for the fishing years 2004–2005 (after ITQ implementation and before financial crisis), 2008–2009 (during financial crisis), and 2012–2013

Table 2
Explanation of variables used in specifying the ERG model.

Variable name		Definition
Edges	E	Parameter that specifies the exact number of edges (density) in the simulated network
Reciprocity		Tendency for edges to be mutual
Geometrically weighted indegree		A form of preferential attachment, i.e. tendency to receive additional incoming edges if node already has many edges
Geometrically weighted outdegree		A form of preferential attachment, i.e. tendency to receive additional outgoing edges if node already has many edges
Indegree centrality		Tendency to receive more incoming edges based on the specified node attributes
Homophily		Tendency to connect to nodes which have similar attributes

(after financial crisis) to see if driving factors change over time and whether there was an effect of the financial crisis.

We set the network for the ERG modeling to represent trade between vessels. A variety of internal network characteristics of the quota holders were tested, including homophily, heterophily, and an ‘edges’ term. Homophily is the tendency for nodes with similar characteristics to be more likely to interact (Fischer and Jasny, 2017; Prell et al., 2017). On the contrary, heterophily is the tendency for nodes with dissimilar characteristics to be more likely to interact (Prell et al., 2017) (Table 2). The “edges” term indicates the density of the network (Hunter et al., 2009), as described previously network density are the number of connections as a fraction of the total number of possible connections (Table 1). In a dense network the likelihood of an edge existing between two nodes would be higher than in a sparser network. We also included other possible tendencies of trade formation that are known to shape social networks (Snijders et al., 2010), including a) reciprocity, i.e. the tendency for mutual ties (companies that sell quota to a certain company would be more likely to buy quota from that same company and vice versa), b) geometrically weighted in- and outdegrees (GW indegree and outdegree), where more weight is given to low degree nodes, i.e. a form of preferential attachment (companies that have many trades have a higher chance of new incoming/outgoing trades). A positive GW indegree indicates dispersion, while the negative GW indegree indicates anti-centrality. Goodness of fit statistics were used to select for the above tendencies that shape social networks.

The characteristics described above are endogenous to the network (i.e. they are explained by the network itself). ERG modeling was also used to test whether external covariates might explain the dynamics of the quota share markets, including tests for whether a) quota owners with large amounts of allocated quota were more likely to trade with quota owners with smaller amounts of allocated quota, b) quota was more likely to flow between small and larger vessels, c) there was an effect of allocated quota on indegree, d) there was an effect of gross tonnage (GRT) on indegree, and e) whether a greater mismatch between actual catch and allocated quota (only for catches that were higher than quota) in one year led to greater indegree the following year. Because gross tonnage and allocations were highly correlated we regressed allocations against gross tonnage and used the residuals to get an allocation effect that was free from the effect of the vessel size. Case (e) tests whether trade leads to better catch-quota matching (as intended): if allocations were inadequate to meet the amount of fish caught (i.e., high mismatch) in one year, then it would be expected that a fishing company would buy quota to cover the mismatch the next year. Matching was indicated by a normalized ratio on a scale from -0.5 to $0.5(\text{catch} / (\text{quota} + \text{catch}) - 0.5)$, but all instances of catching less than the quota (< 0) were set to 0. See Table 2 for a full list of the specified covariates and network processes that we included in the ERG modeling. For the quota lease markets, the same endogenous characteristics were used, but slightly different covariates were tested and reported in Appendix E.

For each of the 33 total candidate models applied to the different markets (small-boat and large-boat) as well as the different periods (2004–2005, 2008–2009 and 2012–2013), we calculated Akaike’s Information Criterion, AIC. AIC represent the relative strengths of evidence for a particular model given the set of models, where the lowest AIC score represents the best model (which should be the most parsimonious model i.e. the model that gives the best explanation with fewer parameters). Further information on model selection can be found in Appendix D. ERG models were made in R using the package statnet (Handcock et al., 2003). Significance levels were set at 0.05, 0.01 and 0.001.

3. Results

3.1. Cod quota market dynamics

The lease market was more active by about an order of magnitude than the share market (Fig. 1a–b). On average, the lease market had 4250 leases per year while the quota share market had only 301

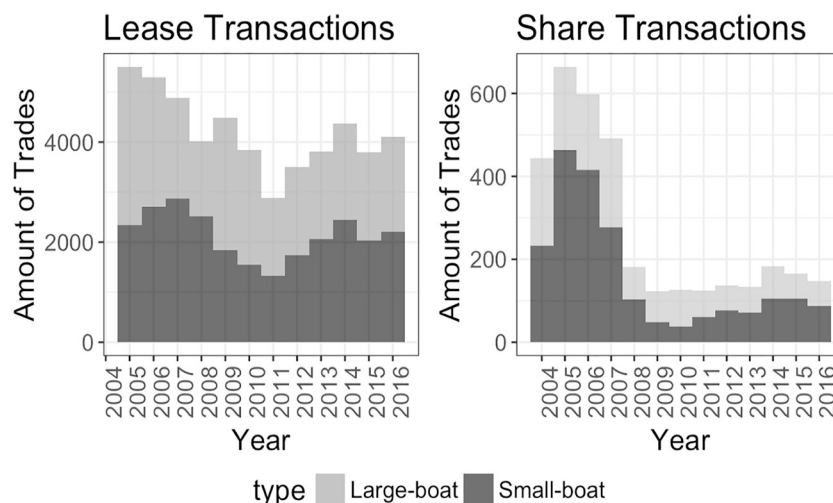


Fig. 1. Statistics of cod market dynamics a) amount of individual trades in lease transactions for large and small boat fleets, b) amount of individual trades in quota share transactions for large and small boat fleets. Note that y-axes differ and the histogram is stacked.

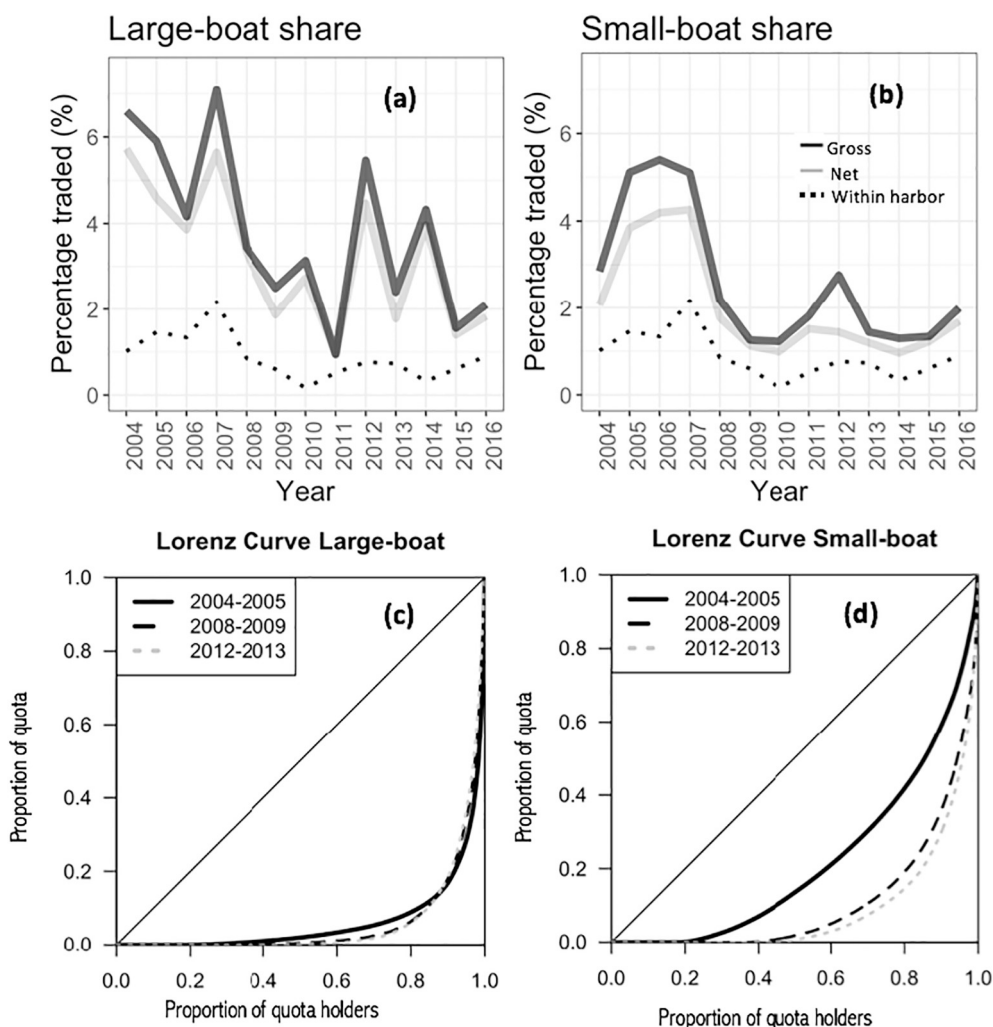


Fig. 2. Statistics of cod market dynamics distinguishing gross and net trades of cod quota as well as Lorenz curves showing the distribution of quota holdings: a) percentage share traded in large-boat quota market, b) percentage share traded in small-boat quota market, c) and d) Lorenz curves for the large-boat and small-boat quota holders, respectively.

transactions. The number of lease transactions decreased over the study period for the large-boat quota market (by ~40% in the large-boat, from 3106 to 1850; by ~3% in the small-boat, from 2240 to 2175). The quota share market showed a steep increase in number of transactions for the small-boat system in the first years of the study period (Fig. 1b); this activity died down after 2005, and steeply decreased after 2007.

A small amount of quota was bought and sold in the same year by the same owner. This is visible because the gross trades (black line) were higher than the net trades (grey line) in Fig. 2(a–b). The gross trade represents the total turnover of quota in that year and the net trade represents the trade balance for all fishers/fishing companies in that fishing year. There is a smaller amount of trade occurring between vessels in the same harbor than between vessels registered in different harbors (dotted line in Fig. 2(a–b)). At the start of our time series, quota ownership was already relatively concentrated (Gini = 0.56 for the small-boat and 0.87 for the large-boat quota market). Over time, the small-boat market quota became consolidated into the hands of fewer owners while the large-boat market there was very little changed in quota concentration, despite an active quota market. This change in ownership in quota is displayed in the Lorenz curves in Fig. 2: the 1:1 line represents even quota ownership and over time the small-boat quota market showed a growing deviation from this line while the large-boat market was already relatively concentrated in 2004–2005. In the fishing year of 2012–2013 the Gini was 0.82 for the small-boat and

0.88 for the large-boat quota market.

In the small-boat market the number of registered accounts also decreased by 11%, from 495 in the fishing year of 2004–2005 to 441 in the fishing year of 2012–2013 (Fig. A4), and more fishers in the small-boat market became “lease-dependent”. In the fishing year 2004–2005, 15% ($n = 80$) of the registered accounts did not have quota allocated but still caught cod by leasing quota; this doubled to 33% ($n = 151$) by 2012–2013. However, on average each of these fishers in 2004–2005 used on average 26 tonnes of leased cod quota (0.016 % of allocations) while this was reduced to 13 tonnes on average in 2012–2013 (0.008% of allocations). Seventy-seven percent of these companies in 2012–2013 were also registered under a license of “coastal fishing” ($n = 116$) as well, indicating that this increase in the number of lease-dependent boats accompanied by a reduction in average quota quantity leased was likely a result of the introduction of the coastal fishing system in Iceland.

3.2. Cod quota market network analysis

3.2.1. Node level metrics

The degree distributions in Fig. 3 generally show a higher cumulative frequency of high indegrees than high outdegrees across quota holders. This was true both in the lease and in the permanent share networks and indicates that there was a small set of owners that bought

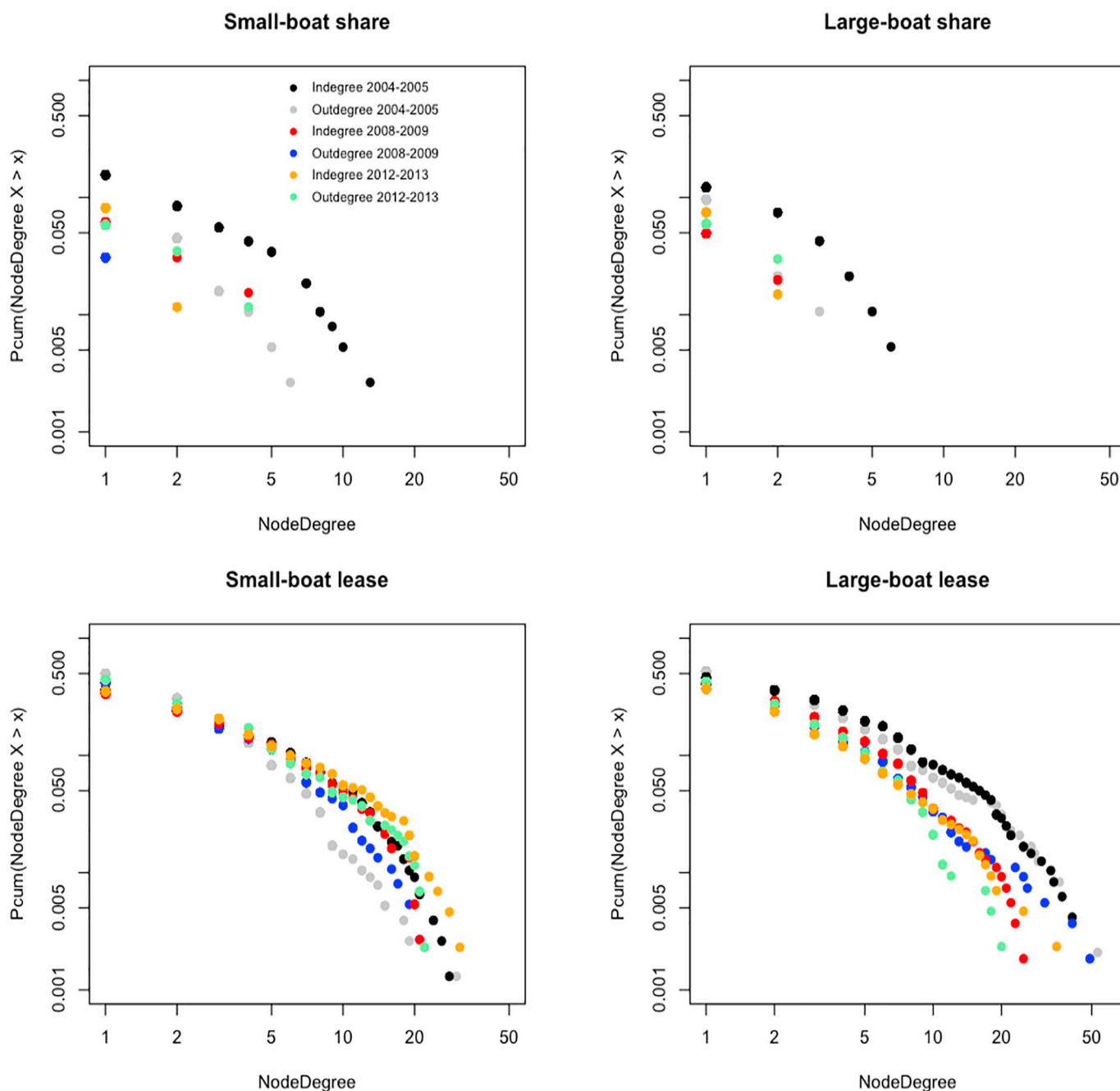


Fig. 3. Cumulative incoming and outgoing degree distributions for: a) the small-boat quota market for share transfers b) the large-boat quota market for share transfers c) the small-boat lease market d) the large-boat lease market.

quota frequently within the year, while individual owners did not sell quota as frequently. We found right skewness towards higher degrees in all distributions (data measured by Skewness and Kurtosis are presented in Table B1).

For 9 out of the 24 distributions in Fig. 3 we found some support for power law fits ($p < 0.10$; see Table B2 in the Supplementary material). All of these were distributions from quota share markets. We found that there was insufficient evidence to distinguish between the power law and log-normal distribution types (log-likelihood ratios can be found in Table B3). Though, for all possible power-law distributions we could rule out an exponential distribution (Table B3).

In Fig. 4, traders are highlighted that have both in- and outgoing trades as well as five or more trades in the same fishing year. We consider these highly active quota holders to reflect “broker nodes,”

which were more frequent in the peak year of trade (2004–2005). The “broker nodes” decreased with a decrease in the amount of trade (Fig. 4).

3.2.2. Cohesiveness

For the lease networks, almost all nodes were in the Giant component (Fig. 5b), showing a very cohesive network in which virtually every company was indirectly tied to all the other companies. This fraction is smaller for the quota share networks, which became more fragmented after 2008. Modularity also differed between the quota lease and share markets: the lease market showed a very cohesive network (modularity < 0.31), whereas the share markets showed high modularity (> 0.61). This modularity increased in later years when the networks were very fragmented (0.72–0.96). The number of traders

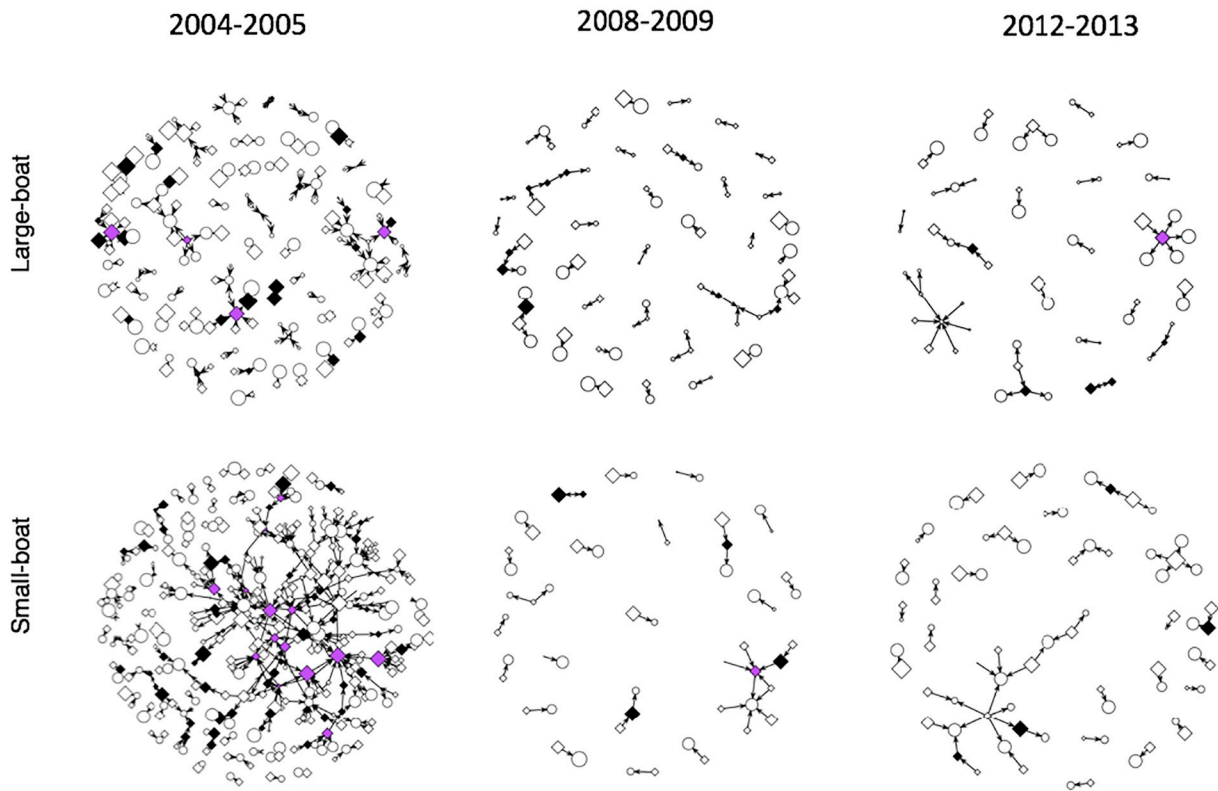


Fig. 4. Owner networks of the small-boat quota share market for permanent shares for fishing years 2004–2005, 2008–2009 and 2012–2013. Upper panels show the large-boat networks and lower panels show the small-boat networks. Nodes are scaled to the amount of quota owned. Open diamond nodes have outgoing trades, open circle nodes have incoming trades, black colored nodes have both incoming and outgoing trades and purple nodes have both incoming and outgoing trades as well as five or more total trades in the same fishing year (“broker nodes”). (For interpretation of the references to colour in this figure legend, the reader is referred to the web version of this article.)

decreased over time in the quota lease market. Both for the lease and share markets clustering coefficients and network density approached zero, though for both share markets clustering was slightly higher in the years after 2008 (Fig. 5c).

3.2.3. The influence of network statistics on price dispersion

Price dispersion was rather low in both networks with an absolute mean of 1.95% in the large-boat market (maximum dispersion: 39%, minimum dispersion: –40%) and an absolute mean of 3.95% in the

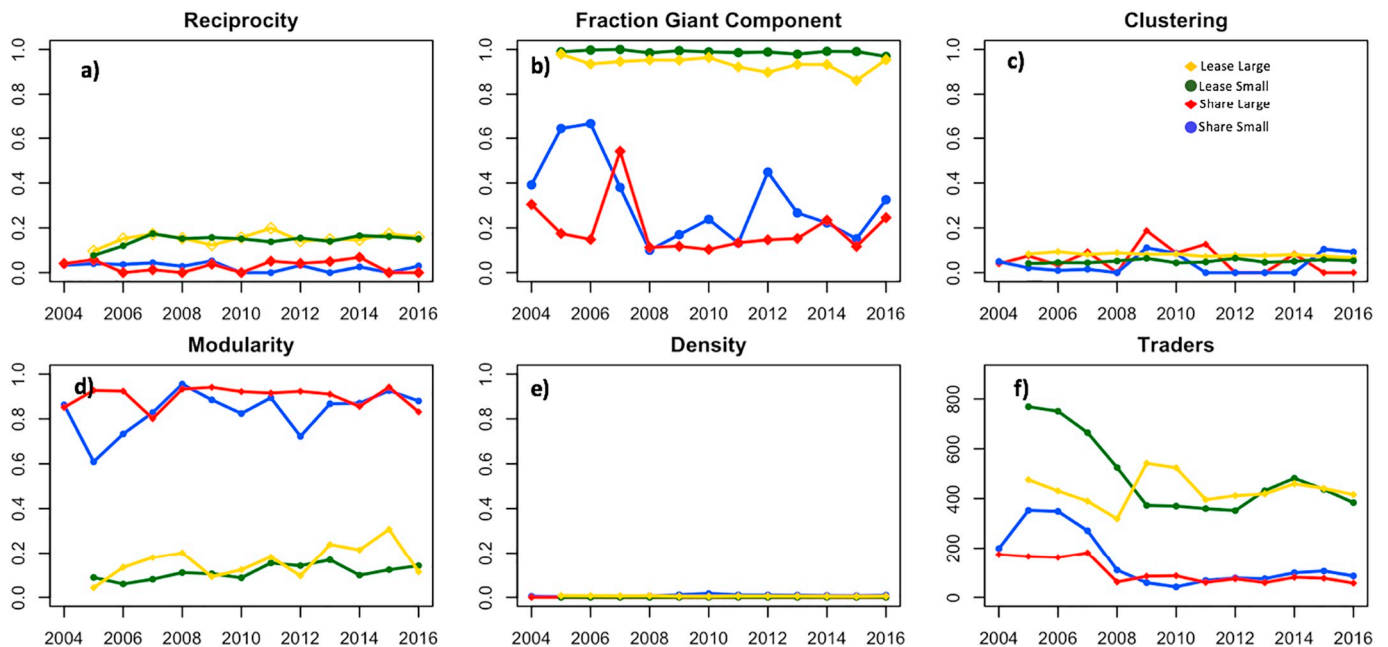


Fig. 5. Changes over time in the quota markets of the following network indicators: a) reciprocity, b) fraction of the network's giant component, c) clustering coefficients, d) modularity, e) network density and f) amount of traders.

Table 3
Estimation results of ordinary leased square regressions of network characteristics and price dispersion (2004–2009).

Small-boat lease market					Large-boat lease market		
Direction	Variable	Coef.	SE	$P > t $	Coef.	SE	$P > t $
Leasing in	Quantity	$-2.84e^{-03}$	$6.14e^{-04}$	$< 0.001^{***}$	$-9.48e^{-04}$	$3.11e^{-04}$	0.00235^{**}
	Broker	$-1.53e^{-03}$	$6.47e^{-04}$	0.0184^*	$4.75e^{-04}$	$3.27e^{-04}$	0.14567
	Betweenness	$-2.65e^{-04}$	$6.46e^{-04}$	0.682	$-2.66e^{-04}$	$3.27e^{-04}$	0.41571
Leasing out	Quantity	$-2.39e^{-07}$	$1.78e^{-08}$	$< 0.001^{***}$	$-2.62e^{-08}$	$3.11e^{-09}$	$< 0.001^{***}$
	Broker	$4.84e^{-03}$	$5.06e^{-04}$	$< 0.001^{***}$	$1.14e^{-03}$	$2.98e^{-04}$	$< 0.001^{***}$
	Betweenness	$-2.87e^{-03}$	$2.51e^{-04}$	$< 0.001^{***}$	$6.51e^{-04}$	$1.27e^{-04}$	$< 0.001^{***}$

small-boat market (maximum dispersion: 75%, minimum dispersion: –61%). There were slight indications that network position mattered for the prices obtained in the lease-markets. Broker nodes got significantly higher prices for quota that they leased out in both the small- and large-boat (Table 3) as well as central nodes as measured by betweenness centrality in the large-boat network, although surprisingly we found the opposite result for betweenness centrality in the small-boat network. Results for prices regarding the buying of lease quota were less significant, only for broker nodes in the small-boat lease market we found the expected negative effect on price dispersion.

3.2.4. Exponential random graph modeling of quota share networks

Results of the model selection can be found in Appendix D coefficients of the models with the lowest AICs can be found in Table 4. The results of the lease market ERG models are reported in Appendix E. The networks for quota share trading were sparse as indicated by a large and negative coefficient for the edges term (Table 4) (i.e. there are few trade routes compared to all possible trade routes in the network, similar to density (Fig. 5e)). For the 2008–2009 and 2012–2013 small-boat networks, ERG models with a covariate for the allocation effect on indegree had lower AIC scores than other models, while the most parsimonious 2004–2005 small-boat network, as well as several large-boat networks, also included covariates related to GRT. In 2004–2005, a significant positive impact of reciprocity was found for both networks and a significant negative GW indegree was found for small boats, indicating preferential trading with quota holders that have higher degrees (i.e., a popularity effect). The geometrically weighted (GW) indegree and outdegree were positive and significant in most other years, indicating dispersion. The matching ratio (see Appendix D) did not

Table 4
Exponential random graph model results for the likelihood of trades established between quota holders in share markets. Edges, reciprocity, GW indegree and outdegree are endogenous effects while the node indegree effects (GRT and allocated quota), heterophily (GRT and allocated quota) relate to the exogenous covariates effects on trade-formation.

Predictor variables		2004–2005	2008–2009	2012–2013
Small-boat network trade connections	Edges	$-10.11 (0.43)^{***}$	$-7.11 (0.81)^{***}$	$-5.63(0.44)^{***}$
	Reciprocity	$1.89 (0.58)^{**}$	$1.73 (0.89)$	–
	GW indegree	$-0.50 (0.19)^*$	$1.30 (0.63)^*$	$1.23 (0.50)^*$
	GW out degree	$1.17 (0.18)^{***}$	$2.45 (0.77)^{**}$	$0.25 (0.43)$
	Node indegree effect: ln (GRT)	$2.06 (0.20)^{***}$		
	Node indegree effect: allocated quota	$-1.26 (0.19)^{***}$	$1.61 (0.48)^{***}$	$1.28 (0.44)^{**}$
	Heterophily: allocated quota	$0.50 (0.19)^{**}$		
	Heterophily: ln(GRT)	$-0.28 (0.19)$		
Large-boat network trade connections	Edges	$-7.21 (0.38)^{***}$	$-6.65 (0.66)^{***}$	$-5.52 (0.63)^{***}$
	Reciprocity	$2.59 (0.51)^{***}$	–	–
	GW indegree	$-0.09 (0.27)$	$1.74 (0.54)^{**}$	$1.04 (0.51)^*$
	GW out degree	$1.76 (0.30)^{***}$	$1.95 (0.58)^{***}$	$0.70 (0.49)^{***}$
	Node indegree effect: ln(GRT)	$-0.23 (0.06)^{***}$		$0.28 (0.11)$
	Node indegree effect: allocated quota	$-0.13 (0.07)$	$0.32 (0.13)^*$	
	Heterophily: allocated quota			
	Heterophily: ln(GRT)	$-0.23 (0.06)^{***}$	$-0.47 (0.12)^{***}$	$-0.58 (0.14)$

Significance level at: * $p < 0.05$; ** $p < 0.01$; *** $p < 0.001$. Values are estimated log-odds ratios for the tested predictor variable. Standard errors are in parentheses. Models with (–) did not converge. Blanks indicate the coefficient was excluded from the most parsimonious model according to our model selection procedure (see Appendix D).

appear to be a strong explanatory factor.

We also found significant effects of covariates on the probability of a trade route existing between owners. We found a positive significant effect of GRT and a negative effect of quota allocation on indegree in the small-boat share network of 2004–2005 (Table 4). However, for the two other periods, indegree in the small-boat network was better explained by a positive effect of allocated quota. We also found some significant negative effects of heterophily on GRT in the large-boat network, suggesting that owners with larger boats tended trade with owners with smaller boats in those years (i.e. the large-boat and small-boat networks in 2004–2005 and 2012–2013). Summary statistics showed that in each of the year quota tended to flow towards larger boats and owners which had more quota allocated (Fig. C1 and C2).

3.3. Effects of the financial crisis in quota market networks

The descriptive network statistics indicated a likely impact of the financial crisis in the years 2008–2009. The number of quota holders active in the quota markets (traders, in Fig. 5f) came almost to a halt, reducing trading activity in and after the crisis year (2008–2009). Share networks were less cohesive/more fragmented after 2008, as is indicated by decreasing fractions of the Giant component and increasing modularities prior to 2008 in contrast with more stable values post-2008 (Fig. 5b and d). There also was a small increase in clustering (i.e. neighboring quota holders also forming trade routes).

4. Discussion

The quota market is the instrument through which a more efficient

fishery is gained. As within an ITQ system, more efficient owners/operators buy quota from less efficient ones which may then leave the industry, thereby reducing overcapacity. However, efficiency gains depend on the underlying assumption of free and efficiently operating quota markets, for which there is very little empirical evidence. Here, we provide an empirical evaluation of the cod quota market in Iceland using share and lease trade data within small- and large-boat markets. This study demonstrated a number of important trends in trade networks. First, the lease markets were more connected than the share market, as expected. However, unexpectedly, the small-boat lease market also interacted with other management systems (i.e., the coastal fishing system), to support a new fleet segment. Second, the small-boat fleet demonstrated an initial rapid consolidation, but not the large-boat fleet. This pattern corroborates the results of Agnarsson et al. (2016), who suggested that the large-boat market was already rather consolidated at the onset of both studies due to its decade-earlier transition into an ITQ system. Third, there were some indications of the importance of personal ties for trade formation which could indicate a market inefficiency (e.g., through asymmetric information among traders). Fourth, trading in the quota markets decreased steeply in the crisis years and did not regain the same activity in the time-period studied. This decrease could be related to speculative distortion, another market inefficiency. Fifth, there were several small but significant relations between network position and price-dispersion.

More frequent trade took place on the lease market for small boats than on that market for large boats. This could be explained by a large amount of cost-free intracompany trades among several boats owned by the same company (Fig. A2). We also found that a small new segment of small-boat quota holders has likely developed after the introduction of coastal fishing in 2009. Since 2004/2005, more registered small-boat companies/fishers have become lease-dependent (i.e., own no quota) in the small-boat lease market, and in the most recent market analyzed, 77% of lease-dependent boats likewise participate in the effort-based coastal fishing system. Because coastal fishing is restricted to the summer months, it is possible that fishers who made costly investments for coastal fishing, for instance by buying a boat and jig-machines (Chambers et al., 2017; Chambers and Carothers, 2017), needed to supplement their income by leasing quota and thus extending the season by several months before and after the coastal fishing months. As a result, the mean amount of quota leased by a boat simultaneously decreased by 50%, likely as a result of coastal fishing vessels needing to lease smaller total amounts quota annually to maintain profitability, in comparison with vessels who were entirely lease-dependent before the coastal fishing system was introduced. This interpretation is in agreement with Chambers and Carothers (2017) who found that small-boat fishers generally perceive that it is difficult or impossible to make a year-round income outside of the ITQ system. However, a longitudinal analysis would be necessary to distinguish whether the new fleet segment is composed of new entrants or previous participants of the small-boat fleet.

During the initial wave of quota share trade in the small-boat system shortly after the implementation of the ITQ system in 2001–2004, consolidation was rapid (Gini index changed from 0.56 to 0.78 from 2004 to 2009). These results are in line with previous reporting of increasing consolidation within the quota system in Icelandic fisheries soon after its implementation (Agnarsson et al., 2016; Pálsson and Helgason, 1995). In- and outdegree distributions were also in agreement with a network that was consolidating. As fewer companies bought quota, quota became concentrated in the hands of a small group of companies who continued to buy, resulting in a higher indegree. In the small-boat lease market the distributions were similar, suggesting that there were quota holders that repetitively leased quota from many other fishers/fishing companies. This could indicate a general shortage of cod quota (Matthiasson, 2012) or the need to lease quota several times in a fishing year to avoid fines.

ITQ systems tend to have more quota concentrated on larger boats

(Marvin, 1992) and accumulation of allocated quota to companies with more allocated quota is a driving mechanism for consolidation (a form of “the rich getting richer”) that has been observed earlier in the Icelandic quota system (Agnarsson et al., 2016; Pálsson and Helgason, 1995). The boats gross tonnage (GRT) in our ERG models had a strong explanatory power, but only in 2004–2005 and mainly for the small-boat market. In the initial years after introduction of the small-boat quota system, this trend may be explained by the switch to output controls with initial share allocations proportionate to fishing history, which would have reduced each vessel's landings proportionately. Those that chose to remain fishing rather than exit the fishery would need to buy enough quota to remain profitable, and for larger boats, this happens to be a larger absolute amount of quota shares needed to remain profitable which might have been sourced from a larger amount of transactions. The strong explanatory power of GRT in the small-boat market in 2004–2005 has likely also to do with this very recent introduction of the ITQ system in that market. It is also supported by the negative effect of allocation (after removing the GRT effect) during the same year, indicating that for those small-boat fishers who had relatively little initial quota allocation yet still wished to remain in the system, a greater investment in cod quota was needed to stay profitable. The previous year catch to quota ratio of the vessel (matching, see Appendix D) did not appear to be a strong driver of trade as the covariate was never included in any of the most parsimonious models, suggesting that, at least for cod, quota trade does not primarily occur to adjust quota levels to actual catch levels. In this study no effect of GRT was found for the small boat ERGMs of 2008–2009 and 2012–2013, several years after the initial allocation. However, there were positive effects of allocation on indegree for the small-boat markets of 2008–2009 and 2012–2013 and the large-boat market in 2008–2009, which means that quota consolidated in fishing companies with more allocated quota rather than on larger vessels in those years.

Previous studies associated low clustering to companies using brokers for trading quota (Innes et al., 2014; van Putten et al., 2011), something that is known to occur in the Icelandic quota market as well (Sanchirico et al., 2006). Even though there are some transaction costs associated with the use of a broker, fishers in Iceland tend to prefer the use of a broker to get a better price compared to selling or leasing through their own networks (Chambers and Carothers, 2017). The occurrence of “brokers” in networks has been observed in several empirical studies in quota lease markets, such as the great barrier reef fin-fish fishery in Queensland, Australia and the rock lobster lease market in Tasmania, Australia (Innes et al., 2014; van Putten et al., 2011) and have even been included in complex fisheries simulation models (Fulton et al., 2014).

However, other network characteristics such as high reciprocity (in the lease markets around 20% of ties was reciprocal) and modularity do point to the importance of personal ties in the quota share networks. For example, a possible reason we observed for the significant reciprocity is that some companies collaborated by moving quota temporarily between boats, possibly to avoid quota to be taken away when it is not fished for > 50% for two years in a row (Fiskistofa.is).

In both quota trade networks, some effect of the financial crash was likely, as a lesser percentage of quota was being traded and trades were less frequent after this period. Although it is difficult to attribute causality to the crisis, as we could not directly test for it, the steep decrease in trade visible in both large- and small-boat markets suggests that it was more likely the result of a common factor, rather than routine decreases in activity observed as consolidation progresses after the initial years of implementation (Newell et al., 2005). Moreover, there is evidence that the crisis had an impact on fishing companies' finances (Gunnlaugsson and Saevaldsson, 2016), which are likely to impact trade patterns. Trade activity in small-boat quota shares does not regain its pre-crisis level, probably because the system is very consolidated in the later years and pre-crisis trade activity may have been inflated due to speculative trading. Given the increase in the quota

share price starting in 2000 (Fig. A1), speculative trading would not be surprising. The pre-crisis price increase of quota could be a reason for the high investment, as an initial increase in price and the expectation of further price increase can positively impact demand (attract new buyers) in asset markets such as is the case for housing markets (Case and Shiller, 2003). The latter was indicated by Dobeson (2018) who interviewed small-scale fishers in Iceland about the transition to the quota system, describing financialisation and speculative trading in the small-boat system. However, we cannot fully explain why the quota price seems to decrease with a delay of a year after the trading in the quota market has steeply decreased. Though some of the fishers might have gotten into debt because of the decrease in quota value, some of the negative effects of the crisis for the Icelandic fishing companies were counteracted by an increase in export value after the decrease of the value of Icelandic Krona (Gunnlaugsson and Saevaldsson, 2016). This is probably also reflected in the lease prices (Fig. A1 c); lease prices strongly increased after the crisis years. Interestingly clustering increased after the crisis years in the quota share market, possibly indicating that personal connections became more important in the markets right after the financial crash.

Sales price was positively impacted by a node's higher centrality and whether or not a node was characterized as a broker in both the small- and large-boat lease networks. We only found a small significant negative effect in the small-boat quota lease markets and no significant effects of price dispersion on quota that was leased in in the large-boat lease market. Ropicki and Larkin (2014) found a stronger negative correlation: quota lease traders with larger information-sharing networks leased quota in at lower prices. Ropicki and Larkin (2014) thus concluded that fishers with stronger network positions had greater negotiating power in the quota lease market for red snapper in the Gulf of Mexico. There is some indication that this is the case for the cod quota market in Iceland, though it is considerably smaller than the effects found by Ropicki and Larkin (2014). In other studies of quota lease markets, broker fishing companies have been shown to profit by leasing quota at a higher price than it was bought (van Putten et al., 2011). The effect for brokers that we found, though significant and in the expected direction, was small and price dispersion was so low that this could hardly seem an explanation for the occurrence of brokers in the cod quota lease markets.

Our results may be affected by some confounding factors: first, we were not able to filter out barter trades (i.e. trading one species for another species or set of species) (Holland, 2013) because in the Icelandic system these are not separately registered, and thus a significant portion of our data might be barter data. One would expect to see barter trade as the market pushes for more efficient operations (Holland, 2013). Barter trades are beyond the scope of this study but an estimate of the extent of barter trade in Iceland could be done by comparing net company changes in quota ownership over all species owned (Byrne, C., personal communication, April 2017). Second, our analysis is based solely on company records, not considering the fact that individuals in the fishery may have own several companies with separate quota accounts. Nonetheless, we gained useful insights on the quota market development using network analysis, though our study would have benefited from price information connected to individual trades, mainly in the share market and a longer time series of (average) share prices (Fig. A1).

5. Conclusion

There are indications of market inefficiencies in the quota market for trade in quota shares and leases: the quota share and lease prices are private between fishing companies, and thus asymmetric information exists, personal ties are important in the markets as our results show that reciprocity is significant in shaping trade relations, market concentration is quite high and the price and quantity of trade seem to show a boom/bust pattern that may indicate speculative distortion,

though a longer time series of quota share prices would be needed to be certain of such a pattern. We found that drivers associated with consolidation (gross tonnage of vessels or allocated quota to the vessel owner) were more important factors shaping trade relations than the need to match quota to catches. We believe that the assumption that the quota is flowing freely to the most efficient companies should be further investigated. In the absence of public information on quota share prices, assuming efficiency seems a stretch, social ties and how they shape quota markets need more attention.

Conflicts of interest

The authors declare no conflict of interest.

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Appendix A-E. Supplementary data

Supplementary data to this article can be found online at <https://doi.org/10.1016/j.ecolecon.2019.01.035>.

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Appendix A: background statistics fishing industry Iceland and additional trade statistics for the lease and share quota markets for large and small boats

We used complementary monthly quota price data published by the Central Bank of Iceland (CBI), which is to our knowledge the only source that describes price development of cod quota from 1997 until 2009.



Fig. A1: a) Cod quota price development, price is steeply increasing until 2008. b) net profitability for the fleet, a clear dip in profitability is visible at year 2008, but this soon recovers in 2009 c) Lease price of cod quota, lease prices are high during the years post 2009 when the currency (ISK) dropped in value and export prices strongly increased. The standard deviation is also higher in the crisis years as indicated by the shaded error bars. This is likely due to large fluctuations in the currency.

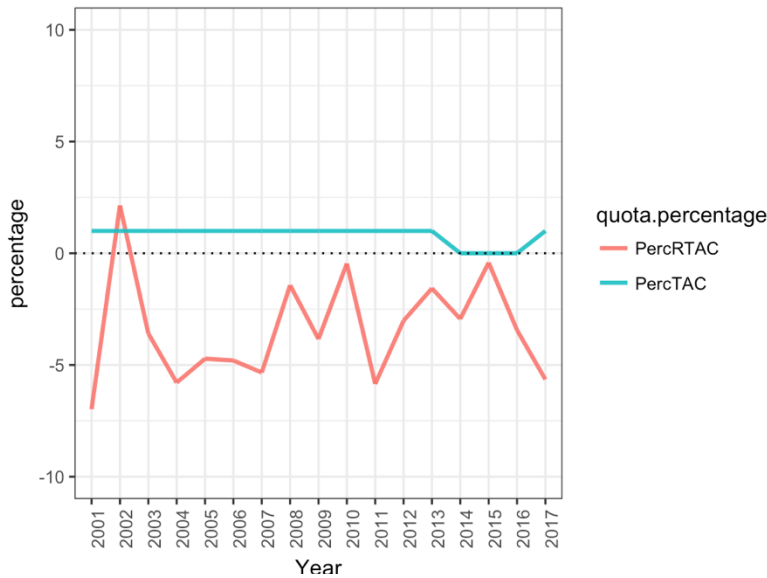


Fig. A2 Percentage of TAC landed (PercTAC) and percentage of TAC as compared to the TAC recommended by the marine research institute (<https://www.hafogvatn.is/>).

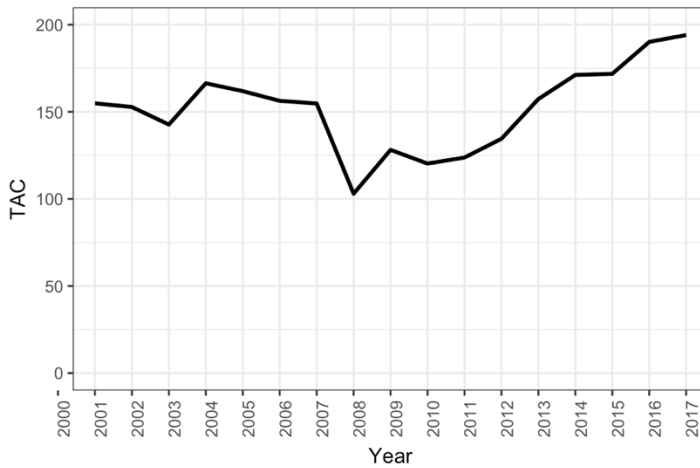


Fig. A3 TAC levels in tonnes of cod.

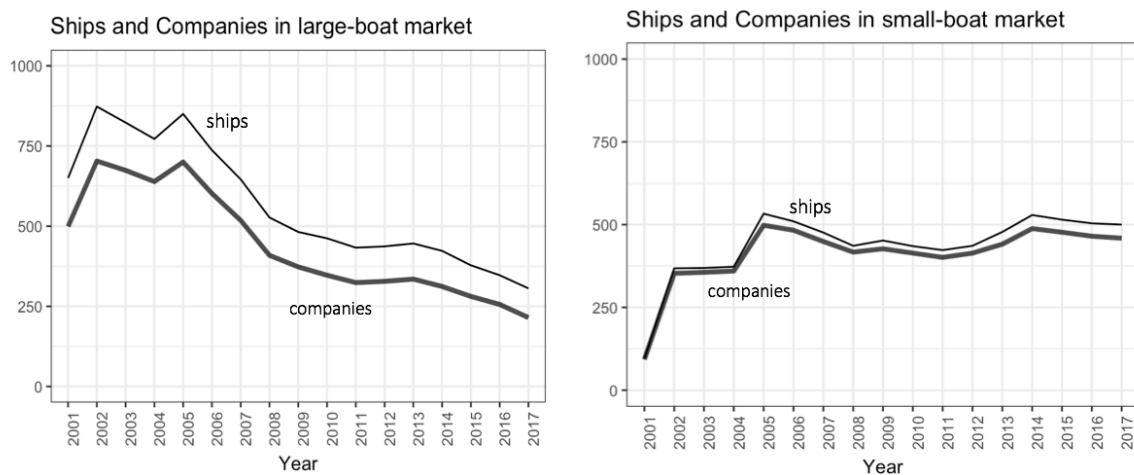


Fig. A4 Number of registered companies under an a) large-boat license and b) small-boat license.

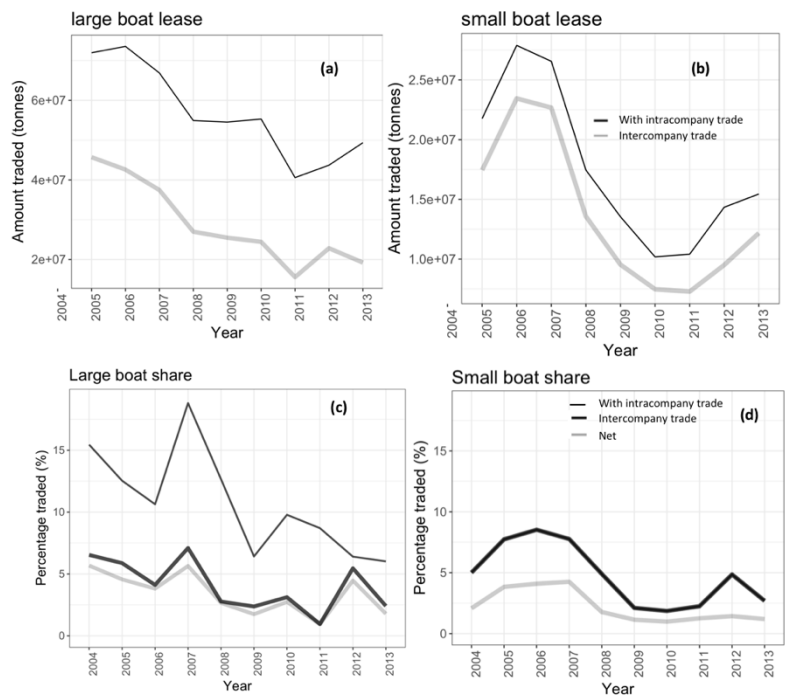


Fig. A5. Statistics of cod market dynamics distinguishing intracompany, intercompany and net trades of cod quota: a) total volume (tonnes of cod) in large-boat lease quota market, b) total volume (tonnes of cod) in small-boat lease quota market, c) percentage of cod quota in the quota share market for large boats and d) percentage of cod quota in the quota share market for small boats

Appendix B: Additional node-level metrics for the quota share and lease markets for large and small boats

Table B1: Skewness and Kurtosis values for in and out degree distributions in the quota share and lease markets

Market	Year	In/out-degree	Skewness	Kurtosis
Share SB	2004-2005	Indegree	3.91	19.67
		Outdegree	3.67	28.36
	2008-2009	Indegree	2.67	9.43
		Outdegree	0.21	-0.99
	2012-2013	Indegree	1.45	3.84
		Outdegree	3.02	12.29
Share LB	2004-2005	Indegree	2.52	15.02
		Outdegree	1.26	31.06
	2008-2009	Indegree	1.24	33.58
		Outdegree	0.58	12.78
	2012-2013	Indegree	3.27	13.60
		Outdegree	1.24	10.85
Lease LB	2004-2005	Indegree	3.37	15.02
		Outdegree	4.24	30.06
	2008-2009	Indegree	4.61	33.58
		Outdegree	3.15	12.78
	2012-2013	Indegree	3.41	13.60
		Outdegree	3.08	10.85
Lease SB	2004-2005	Indegree	3.33	13.75
		Outdegree	4.82	30.89
	2008-2009	Indegree	3.52	17.88
		Outdegree	6.12	49.83
	2012-2013	Indegree	4.87	32.83
		Outdegree	3.17	15.26

Table B2: Power law fits to in and out degree distributions. Significant p-values are printed bold

Small Share	boat	xmin	alpha	p	Gof
ideg5		2	2.38	0.551	0.021
odeg5		2	3.5	0.008	0.035
ideg9		2	2.6	0.431	0.0397
odeg9		No fit (not enough different outdegrees)			
ideg13		2	3.5	0.581	0.023
odeg13		4	3.5	0.15	0.047

Large share	boat				
ideg5		3	3.43	0.219	0.037
odeg5		2	3.5	0.299	0.019
ideg9		2	3.5	0.881	0.015
odeg9 -No fit (not enough different outdegrees)					
ideg13		2	3.36	0.77	0.023
odeg13		2	3.48	0.79	0.0225
Small Lease	boat				
ideg5		12	3.5	0.00	0.093
odeg5		4	3.05	0.00	0.14
ideg9		6	2.7	0.00	0.13
odeg9		2	2.2	0.00	0.07
ideg13		4	2.32	0.00	0.08
odeg13		5	2.82	0.00	0.10
Large Lease	boat				
ideg5		7	2.66	0.00	0.14
odeg5		4	2.29	0.00	0.
ideg9		8	3.4	0.00	0.103
odeg9		7	2.9	0.00	0.085
ideg13		6	2.75	0.00	0.089
odeg13		5	3.06	0.00	0.13

Table B3: Comparisons with alternative distributions. Positive log-likelihood values in combination with sufficiently low p-values indicate that the power-law distribution is favored over the alternative distribution.

	Distribution:	Log normal		Exponential	
		Log likelihood	p	Log likelihood	p
Small-boat share	Ideg 2005	0.01	0.74	122.68	3.31e-12
	Ideg 2009	Not enough information to fit log normal		36.13	0.0064
	Ideg 2013	1.39	0.75	57.25	0.00019
	Odeg 2013	0.20	0.91	51.33	0.0015
Large boat share	Ideg 2005	0.0026	0.89	68.33	5.18 e-06
	Odeg 2005	0.00054	0.94	148.75	1.94 e-09
	Ideg 2009	0.0026	0.90	74.00	0.00019
	Ideg 2013	0.00018	0.98	38.70	0.0027
	Odeg 2013	6.13	0.45	49.54	0.0018

Appendix C. Summary statistics ERGM and goodness-of-fit diagnostics ERGM for quota share markets

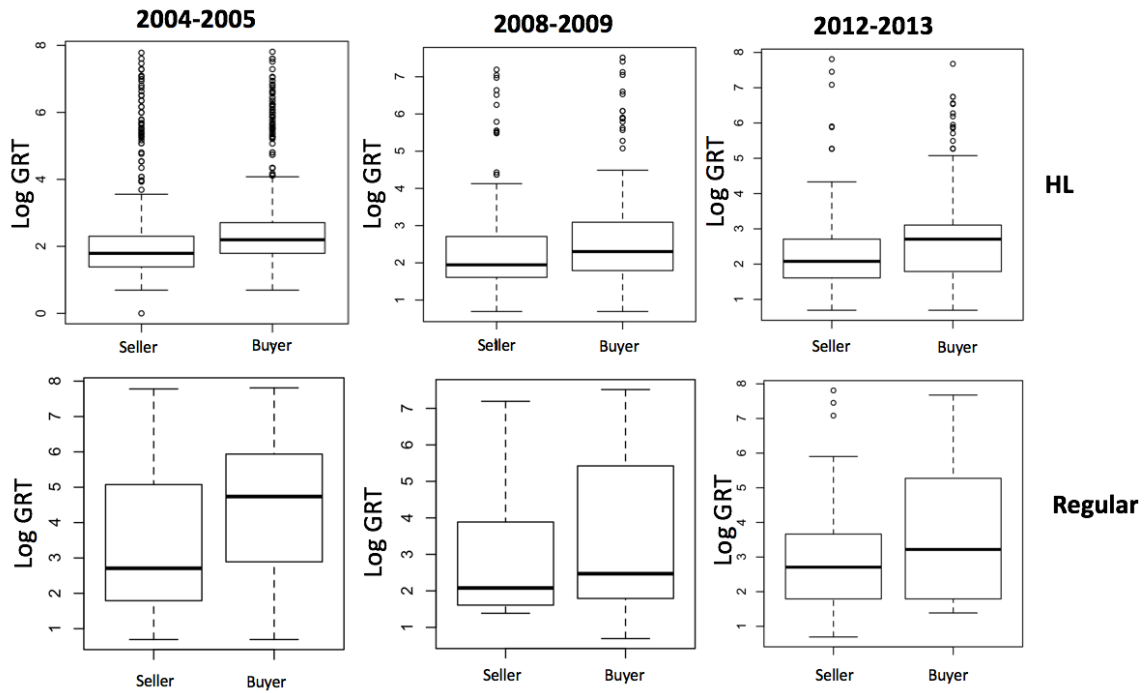


Figure C1: Log transformed GRT for buying and selling vessels in small-boat and Large-boat quota markets

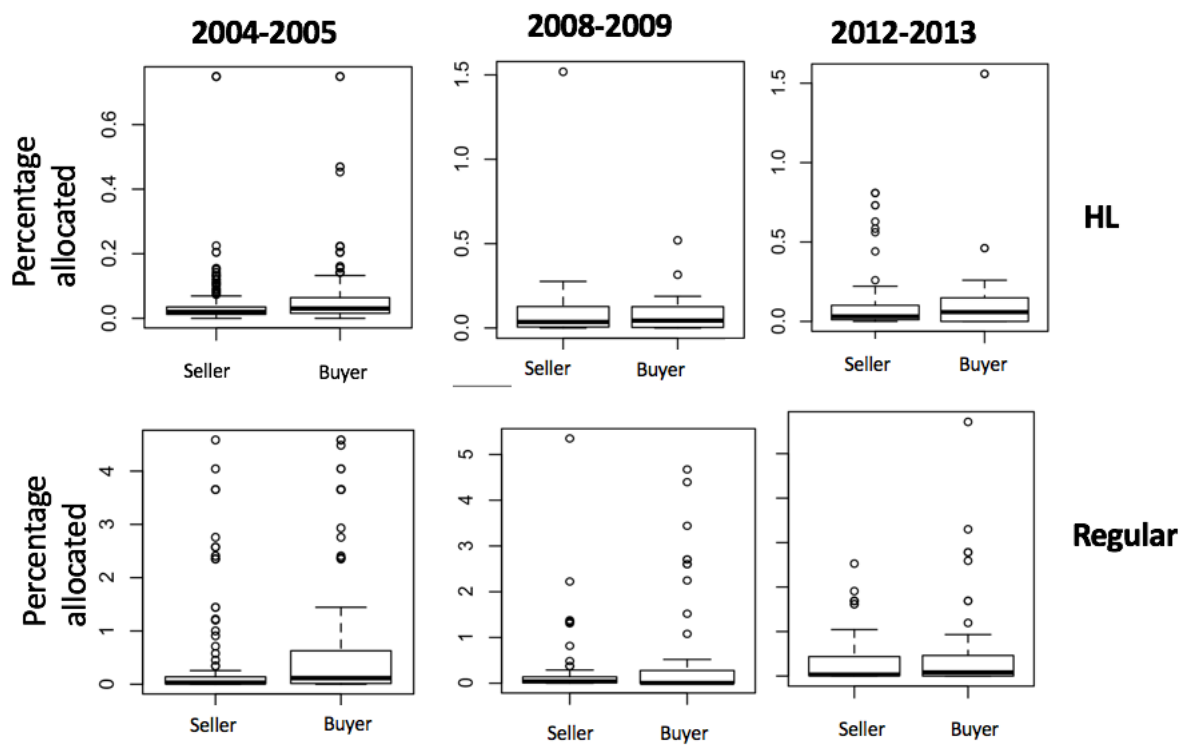


Figure C2: Percentage allocated for buying and selling owners in small-boat and Large-boat quota markets

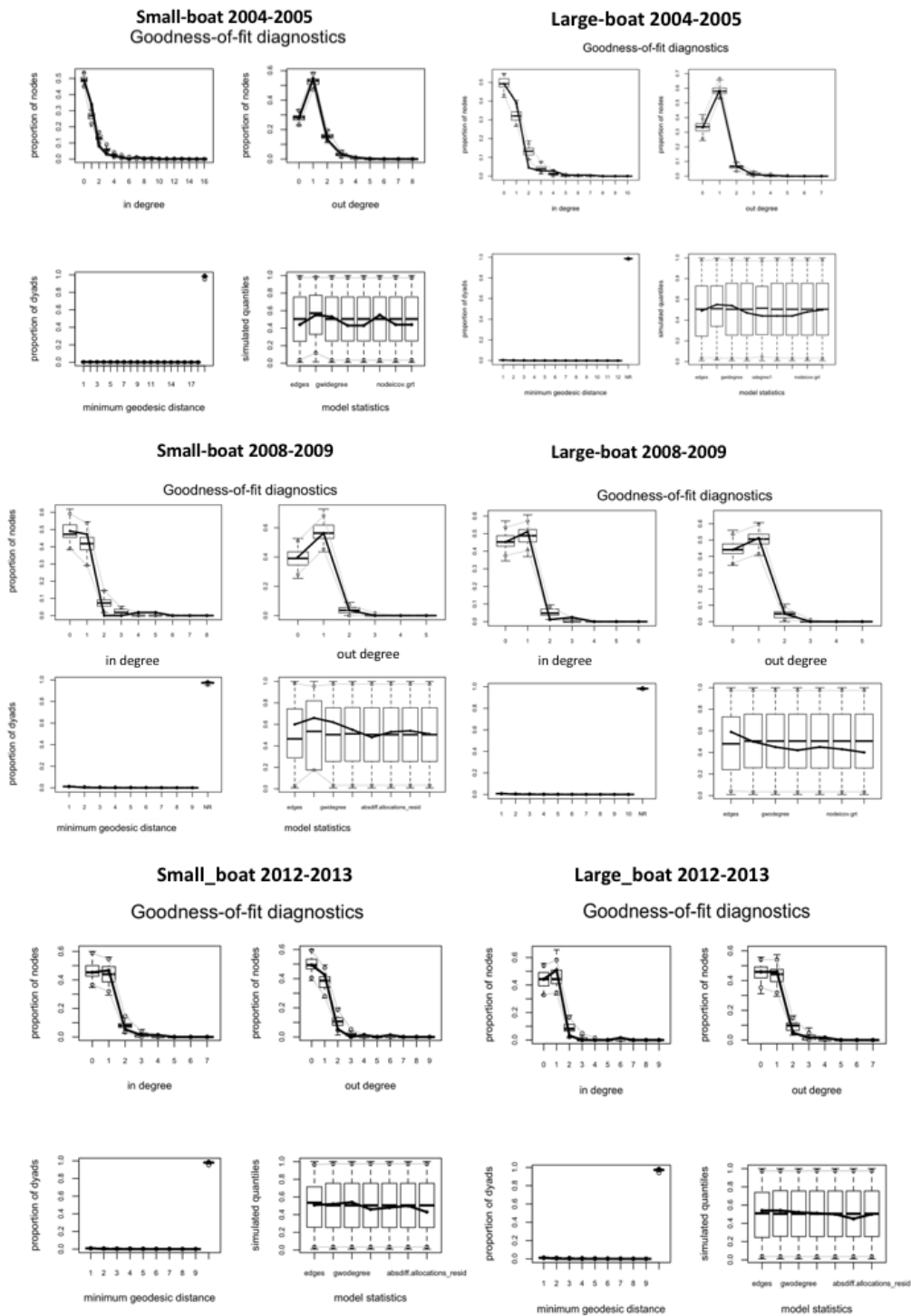


Figure C3: Goodness of fit diagnostics of the ERGM models for the small and large-boat markets for the years: 2004-2005, 2008-2009 and 2012-2013

Appendix D: Model selection ERG models

Table D 1. Model selection results for the small- and large-boat quota markets for quota shares (2004-2005). Header abbreviations are: k , number of parameters; AIC, Akaike Information Criterion; Δ AIC, difference in AIC with that of the lowest value across models.

Model	Fishing year 2004-2005				
	Small boat			Large boat	
	k	AIC	Δ AIC	AIC	Δ AIC
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: Matching	5	4531	10	1727	2
Nullmodel + Heterophily: grt	1	4641	120	1754	29
null model + Heterophily: grt + Node indegree effect: grt	2	4567	46	1727	2
null model + Heterophily: grt + Node indegree effect: grt+ Node indegree effect: Allocated quota	3	4531	10	1725	0
null model + Node indegree effect: Allocated quota	1	4623	102	1757	32
null model + Heterophily: Allocated quota + Node indegree effect: Allocated quota	2	4621	100	1751	26
null model +Node indegree effect: grt + Heterophily: Allocated quota	2	4568	47	1733	8
null model + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: grt	3	4528	7	1734	9
null model +Node indegree effect: grt	1	4567	46	1742	17
null model +Heterophily: Allocated quota	1	4643	122	1758	33
null model + Heterophily: grt + Heterophily: Allocated quota	1	4642	121	1756	31
null model + Node indegree effect: grt + Node indegree effect: Allocated quota	2	4531	10	1741	16
null model + Heterophily: grt + Node indegree effect: Allocated quota	2	4617	96	1749	24
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota	3	4569	48	1727	2
null model + Heterophily: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota	3	4618	97	1750	25
Nullmodel + Heterophily: grt + Node indegree effect: Matching	2	4617	96	1751	26
null model + Heterophily: grt + Node indegree effect: grt +Node indegree effect: Matching	3	4556	35	1727	2
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota +Node indegree effect: Matching	4	4525	4	1726	1
null model + Node indegree effect: Allocated quota +Node indegree effect: Matching	2	4611	90	1754	29
null model + Heterophily: Allocated quota + Node indegree effect: Allocated quota +Node indegree effect: Matching	3	4607	86	1750	25
null model +Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: Matching	3	4556	35	1734	9
null model +Node indegree effect: Allocated quota +Heterophily: Allocated quota+ Node indegree effect: Matching +Node indegree effect: grt	3	4522	1	1735	10
null model +Node indegree effect: grt + Node indegree effect: Matching	2	4554	33	1741	16
null model +Heterophily: Allocated quota + Node indegree effect: Matching	2	4617	96	1755	30
null model + Heterophily: grt + Heterophily: Allocated quota + Node indegree effect: Matching	3	4618	97	1753	28
null model + Node indegree effect: grt + Node indegree effect: Allocated quota +Node indegree effect: Matching	3	4523	2	1741	16
null model + Heterophily: grt + Node indegree effect: Allocated quota +Node indegree effect: Matching	3	4605	84	1747	22
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota+Node indegree effect: Matching	4	4558	37	1728	3
null model + Heterophily: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota + Node indegree effect: Matching	4	4607	86	1749	24
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Matching	3	4556	35	1727	2
null model + Node indegree effect: Matching	1	4630	109	1756	31
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota	4	4521	0	1726	1
Null model		4644	123	1759	34
Null model with only density		4733	212	1807	82

Table D 2. Model selection results for the small- and large-boat quota markets for quota shares (2008-2009). Header abbreviations are: k , number of parameters; AIC, Akaike Information Criterion; ΔAIC , difference in AIC with that of the lowest value across models.

Model	Fishing year 2008-2009				
	Small boat				Large boat
	k	AIC	ΔAIC	AIC	ΔAIC
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: Matching	5	363	4.3	569	5.2
Nullmodel + Heterophily: grt	1	371.8	13.1	567.5	3.7
null model + Heterophily: grt + Node indegree effect: grt	2	364.6	5.9	564.5	0.7
null model + Heterophily: grt + Node indegree effect: grt+ Node indegree effect: Allocated quota	3	361.4	2.7	565.2	1.4
null model + Node indegree effect: Allocated quota	1	358.7	0	582.5	18.7
null model + Heterophily: Allocated quota + Node indegree effect: Allocated quota	2	360.7	2	574.7	10.9
null model +Node indegree effect: grt + Heterophily: Allocated quota	2	365.2	6.5	578.6	14.8
null model + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: grt	3	362.1	3.4	576.6	12.8
null model +Node indegree effect: grt	1	363.3	4.6	583.9	20.1
null model +Heterophily: Allocated quota	1	372.6	13.9	579.1	15.3
null model + Heterophily: grt + Heterophily: Allocated quota	1	373.7	15	569.3	5.5
null model + Node indegree effect: grt + Node indegree effect: Allocated quota	2	360.1	1.4	584.4	20.6
null model + Heterophily: grt + Node indegree effect: Allocated quota	2	360.1	1.4	563.8	0
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota	3	366.3	7.6	566.3	2.5
null model + Heterophily: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota	3	361.1	2.4	565.6	1.8
Nullmodel + Heterophily: grt + Node indegree effect: Matching	2	370.3	11.6	569.2	5.6
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Matching	3	366.3	7.6	566.4	2.6
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota +Node indegree effect: Matching	4	363	4.3	567.2	3.5
null model + Node indegree effect: Allocated quota +Node indegree effect: Matching	2	360.5	1.8	584.6	20.7
null model + Heterophily: Allocated quota + Node indegree effect: Allocated quota +Node indegree effect: Matching	3	362.5	3.8	576.7	12.8
null model +Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: Matching	3	366.9	8.2	580.5	16.8
null model +Node indegree effect: Allocated quota +Heterophily: Allocated quota + Node indegree effect: Matching	3	363.7	5	578.6	14.8
null model +Node indegree effect: grt + Node indegree effect: Matching	2	364.9	6.2	585.9	22.1
null model +Heterophily: Allocated quota + Node indegree effect: Matching	2	370.9	12.2	581.2	17.3
null model + Heterophily: grt + Heterophily: Allocated quota + Node indegree effect: Matching	3	372	13.3	571.3	7.5
null model + Node indegree effect: grt + Node indegree effect: Allocated quota +Node indegree effect: Matching	3	361.7	3	586.4	22.6
null model + Heterophily: grt + Node indegree effect: Allocated quota +Node indegree effect: Matching	3	361.9	3.2	565.8	1.9
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: Matching	4	367.9	9.2	568.2	4.5
null model + Heterophily: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota + Node indegree effect: Matching	4	362.9	4.2	567.7	3.8

null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Matching	3	366.3	7.6	566.4	2.6
null model + Node indegree effect: Matching	1	369.1	10.4	584.7	20.8
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota	4	362	3.3	567.1	3.3
Null model		371	12.3	582.7	18.9
Null model with only density		382.5	23.8	605.2	41.4

Table D 3. Model selection results for the small- and large-boat quota markets for quota shares (2012-2013). Header abbreviations are: k , number of parameters; AIC, Akaike Information Criterion; ΔAIC , difference in AIC with that of the lowest value across models.

Model	Fishing year 2012-2013				
	Small boat			Large boat	
	k	AIC	ΔAIC	AIC	ΔAIC
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: Matching	5	570.2	5.8	434.7	4.9
Nullmodel + Heterophily: grt	1	574.4	10	434.6	4.8
null model + Heterophily: grt + Node indegree effect: grt	2	571.5	7.1	429.8	0
null model + Heterophily: grt +Node indegree effect: grt+ Node indegree effect: Allocated quota	3	567.7	3.3	431	1.2
null model + Node indegree effect: Allocated quota	1	564.4	0	448.8	19
null model + Heterophily: Allocated quota + Node indegree effect: Allocated quota	2	566.4	2	442.8	13
null model +Node indegree effect: grt + Heterophily: Allocated quota	2	571.5	7.1	445.1	15.3
null model + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: grt	3	568	3.6	444.6	14.8
null model +Node indegree effect: grt	1	569.5	5.1	449.9	20.1
null model +Heterophily: Allocated quota	1	574.4	10	447	17.2
null model + Heterophily: grt + Heterophily: Allocated quota	1	576.4	12	436.4	6.6
null model + Node indegree effect: grt + Node indegree effect: Allocated quota	2	566.1	1.7	450.7	20.9
null model + Heterophily: grt + Node indegree effect: Allocated quota	2	566.1	1.7	430.5	0.7
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota	3	573.5	9.1	431.7	1.9
null model + Heterophily: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota	3	567.8	3.4	432.5	2.7
nullmodel+ Heterophily: grt + Node indegree effect: Matching	2	576.4	12	433.3	3.5
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Matching	3	572.4	8	430.7	0.9
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota +Node indegree effect: Matching	4	568.9	4.5	432	2.2
null model + Node indegree effect: Allocated quota +Node indegree effect: Matching	2	569.1	4.7	449.8	20
null model + Heterophily: Allocated quota + Node indegree effect: Allocated quota +Node indegree effect: Matching	3	567.5	3.1	443.8	14
null model +Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: Matching	3	572.3	7.9	445.9	16.1
null model +Node indegree effect: Allocated quota +Heterophily: Allocated quota + Node indegree effect: Matching	3	569.5	5.1	445.7	15.9
null model +Node indegree effect: grt + Node indegree effect: Matching	2	570.4	6	450.9	21.1
null model +Heterophily: Allocated quota + Node indegree effect: Matching	2	576.4	12	446.2	16.4
null model + Heterophily: grt + Heterophily: Allocated quota + Node indegree effect: Matching	3	578.4	14	435.3	5.5
null model + Node indegree effect: grt + Node indegree effect: Allocated quota +Node indegree effect: Matching	3	567.4	3	451.8	22
null model + Heterophily: grt + Node indegree effect: Allocated quota +Node indegree effect: Matching	3	567.2	2.8	431.1	1.3
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: Matching	4	574.3	9.9	432.7	2.9
null model + Heterophily: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota + Node indegree effect: Matching	4	568.8	4.4	433.1	3.3
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Matching	3	572.4	8	430.7	0.9
null model + Node indegree effect: Matching	1	574.5	10.1	449.9	20.1
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota	4	569	4.6	432.8	3
Null model		572.5	8.1	450	20.2
Null model with only density		572.5	8.1	451.9	22.1

Appendix E: ERG model selection and results for the quota lease markets

For the quota lease markets we included tests for whether a) quota owners with large amounts of allocated quota were more likely to trade with quota owners with smaller amounts of allocated quota, b) quota was more likely to flow between small and larger vessels c) there was an effect of allocated quota on indegree, d) there was an effect of gross tonnage (GRT) on indegree, e) there was an effect of allocated quota on outdegree, f) there was an effect of gross tonnage (GRT) on outdegree.

Table E 1. Model selection results for the small boat quota markets for quota leases (2004-2005). The models for the large-boat quota market did not converge. Header abbreviations are: k , number of parameters; AIC, Akaike Information Criterion; ΔAIC , difference in AIC with that of the lowest value across models.

Model	Fishing year 2004-2005		
	Small boat		
	k	AIC	ΔAIC
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: Matching	5	23098	583
Nullmodel + Heterophily: grt	1	22699	184
null model + Heterophily: grt + Node indegree effect: grt	2	22681	166
null model + Heterophily: grt + Node indegree effect: grt+ Node indegree effect: Allocated quota	3	22847	332
null model + Node indegree effect: Allocated quota	1	22846	331
null model + Heterophily: Allocated quota + Node indegree effect: Allocated quota	2	22701	186
null model +Node indegree effect: grt + Heterophily: Allocated quota	2	22682	167
null model + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: grt	3	22699	184
null model +Node indegree effect: grt	1	23104	589
null model +Heterophily: Allocated quota	1	23105	590
null model + Heterophily: grt + Heterophily: Allocated quota	1	22679	164
null model + Node indegree effect: grt + Node indegree effect: Allocated quota	2	22845	330
null model + Heterophily: grt + Node indegree effect: Allocated quota	2	22699	184
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota	3	22846	331
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota	4	22670	155
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: ln (GRT) + Node outdegree effect: Allocated quota	6	22515	0
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: ln (GRT)	5	22653	138
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: Allocated quota	5	22664	149
Null model		23139	624
Null model with only density		24034	1519

Table E 2. Model selection results for the small- and large-boat quota markets for quota leases (2008-2009). Header abbreviations are: k , number of parameters; AIC, Akaike Information Criterion; ΔAIC , difference in AIC with that of the lowest value across models.

Model	Fishing year 2008-2009				
	Small boat				Large boat
	k	AIC	ΔAIC	AIC	ΔAIC
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: Matching	5	10375	424	17721	569
Nullmodel + Heterophily: grt	1	10102	151	17473	321
null model + Heterophily: grt + Node indegree effect: grt	2	10093	142	17444	292
null model + Heterophily: grt + Node indegree effect: grt+ Node indegree effect: Allocated quota	3	10125	174	17692	540
null model + Node indegree effect: Allocated quota	1	10122	171	17542	390
null model + Heterophily: Allocated quota + Node indegree effect: Allocated quota	2	10097	146	17649	497
null model +Node indegree effect: grt + Heterophily: Allocated quota	2	10090	139	17545	393
null model + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: grt	3	10101	150	17738	586
null model +Node indegree effect: grt	1	10360	409	17782	630
null model +Heterophily: Allocated quota	1	10338	387	17721	569
null model + Heterophily: grt + Heterophily: Allocated quota	1	10093	142	17693	541
null model + Node indegree effect: grt + Node indegree effect: Allocated quota	2	10126	175	17472	320
null model + Heterophily: grt + Node indegree effect: Allocated quota	2	10080	129	17473	321
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota	3	10106	155	17470	318
null model + Heterophily: grt + Node indegree effect: Allocated quota +Heterophily: Allocated quota	3	10102	151	17473	321
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: ln (GRT)	4	10072	121	17441	289
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: Allocated quota	6	9951	0	17152	0
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: ln (GRT)	5	10003	52	17206	54
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: Allocated quota	5	10066	115	17418	66
Null model		10380	429	17801	649
Null model with only density		10962	1011	18450	1298

Table E 3. Model selection results for the small- and large-boat quota markets for quota leases (2012-2013). Header abbreviations are: k , number of parameters; AIC, Akaike Information Criterion; ΔAIC , difference in AIC with that of the lowest value across models.

Model	Fishing year 2012-2013				
	Small boat			Large boat	
	k	AIC	ΔAIC	AIC	ΔAIC
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: Matching	5	13154	553	11997	375
Nullmodel + Heterophily: grt	1	12829	228	11691	69
null model + Heterophily: grt + Node indegree effect: grt	2	12832	231	11690	68
null model + Heterophily: grt + Node indegree effect: grt + Node indegree effect: Allocated quota	3	12943	342	12008	386
null model + Node indegree effect: Allocated quota	1	12942	341	11952	330
null model + Heterophily: Allocated quota + Node indegree effect: Allocated quota	2	12840	239	11849	227
null model + Node indegree effect: grt + Heterophily: Allocated quota	2	12842	241	11851	229
null model + Node indegree effect: Allocated quota + Heterophily: Allocated quota + Node indegree effect: grt	3	12839	238	11934	312
null model + Node indegree effect: grt	1	12831	230	12049	427
null model + Heterophily: Allocated quota	1	13271	670	11982	360
null model + Heterophily: grt + Heterophily: Allocated quota	1	12840	239	11933	311
null model + Node indegree effect: grt + Node indegree effect: Allocated quota	2	12943	342	11860	238
null model + Heterophily: grt + Node indegree effect: Allocated quota	2	12813	212	11693	71
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota	3	12931	330	11860	238
null model + Heterophily: grt + Node indegree effect: Allocated quota + Heterophily: Allocated quota	3	12830	229	11691	69
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: ln (GRT)	4	12817	216	11689	67
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: Allocated quota	6	12601	0	11622	0
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: ln (GRT)	5	12653	52	11622	0
null model + Heterophily: grt + Node indegree effect: grt + Heterophily: Allocated quota + Node indegree effect: allocated quota + Node outdegree effect: Allocated quota	5	12663	62	11632	10
Null model		No convergence		12058	436
Null model with only density		13823	1222	12563	941

Table E4: Exponential Random Graph model results for the likelihood of trades established between quota holders in lease markets. Edges, reciprocity, GW indegree and outdegree are endogenous effects while the node indegree effects (GRT and allocated quota), heterophily (GRT and allocated quota) relate to the exogenous co-variates effects on trade-formation.

Predictor Variables		2004-2005	2008-2009	2012-2013	
Small-boat network trade connections	Edges	-10.72 (0.27) ***	-8.88 (0.34) ***	-8.62 (0.23) ***	
	Reciprocity				
		2.89 (0.15) ***	3.01 (0.16) ***	2.78 (0.14) ***	
	GW indegree	-2.15 (0.13) ***	-1.72 (0.19) ***	-1.64 (0.17) ***	
	GW out degree	0.11 (0.14)	-0.46 (0.18) ***	-1.03 (0.16) ***	
	Node indegree effect: ln (GRT)	1.34 (0.11) ***	0.52 (0.12) ***	0.67 (0.09) ***	
	Node indegree effect: Allocated quota	-0.44 (0.11) ***	0.55 (0.13) ***	0.20 (0.09) *	
	Heterophily: Allocated quota	0.23 (0.01) *	0.24 (0.11) *	0.19 (0.08) *	
	Heterophily: ln(GRT)	-0.20 (0.01) *	0.04 (0.10)	-0.12 (0.07)	
	Node outdegree effect: ln (GRT)	1.53 (0.11) ***	1.28 (0.11) ***	1.10 (0.07) ***	
	Node outdegree effect: Allocated quota	-1.55 (0.12) ***	-0.88 (0.11) ***	-0.61 (0.08) ***	
	Large-boat network trade connections	Edges		-5.72 (0.10) ***	-5.94 (0.11) ***
		Reciprocity		3.04 (0.14) ***	3.02 (0.16) ***
		GW indegree		-1.76 (0.13) ***	-1.28 (0.15) ***
GW out degree			-0.91(0.13) ***	-0.94 (0.15) ***	
Node indegree effect: ln(GRT)			-0.05 (0.09)	0.21 (0.002) ***	
Node indegree effect: Allocated quota			0.62(0.10) ***	-0.06 (0.03) *	
Heterophily: Allocated quota			-0.26 (0.10) *	-0.09 (0.03) ***	
Heterophily: ln(GRT)			-1.16(0.12) ***	-0.26 (0.02) ***	
Node outdegree effect: ln(GRT)			1.03 ***	0.17 (0.02) ***	
Node outdegree effect: Allocated quota			-0.82 ***	-0.04 (0.03)	

Significance level at: *p<0.05; **p<0.01; ***p<0.001. Values are estimated log-odds ratios for the tested predictor variable. Standard errors are in parentheses. Models with (-) did not converge. Blanks mean that this coefficient was not in the most parsimonious model according to our model selection procedure (see table E1-E3).

The full model had consistently the lowest AIC's (Table E1-E3), model results are reported in table E4. Similar to the networks for quota share the lease market networks were sparse as indicated by a large and negative coefficient for the edges term (Table E4) (i.e. there are few trade routes compared to all possible trade routes in the network, similar to density (Figure 5e)). In all lease market networks, a significant impact of reciprocity was found. All lease networks had a negative GW indegree, indicating preferential trading with quota holders that have higher degrees (i.e., a popularity effect). This is in contrast to the share networks where few popularity effects were present. In all lease networks except for the small-boat network of 2008-2009 smaller boats tend to trade with larger boats as is indicated by the negative and significant heterophily coefficient. In most lease networks the indegree effect for GRT was positive (2004-2005 and 2012-2013 small-boat networks and 2008-2009 large- and small-boat networks) indicating that larger boats had more incoming leases. However, in all networks the outdegree effect for GRT was also positive so larger boats also leased more quota indicating that these larger boats were overall more active on the lease markets. Surprisingly however the

outdegree effect of allocated quota was negative, which might be caused by the stronger GRT effect. The indegree effect of allocated quota was less consistent, negative in some networks (the small-boat 2004-2005 network and the large-boat 2012-2013 network) while positive in the other networks.

Paper IV



Quota Trade and potential measures of the Transitional Gains Trap - A quantitative analysis of the Icelandic ITQ system

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(in review Marine Resource Economics)

Author contributions: CB conceived of the idea and designed the study. MO collected the data and carried out part of the analysis. CB wrote the first draft and MO, SA and BD helped with the writing.

The Transitional Gains Trap in grandfathered ITQ systems

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ABSTRACT

Trade in grandfathered fishing quota has been argued to inevitably lead to a Transitional Gains Trap whereby revocation of quota initially gifted becomes politically impossible as the original beneficiaries are eventually succeeded by harvesters who have paid for their quota and would become loss-making in the event of revocation. This paper proposes a quantitative measure of the trap which reflects not only the potential loss faced by individual harvesters due to revocation of purchased quota but also the offsetting effect on these harvesters of accumulated profits due to the initial grandfathered allocations and subsequent investment in quota. Illustrative calculations for Iceland's ITQ system suggest that the extent of the trap varied considerably by species but remained below 60% under a range of assumptions. Finally, it is shown that the proposed measure may decrease over time, suggesting that the trap may not necessarily be a long-term political barrier to reform.

Introduction

The Individual Transferable Quota (ITQ) approach to fisheries management is widespread, affecting more than 500 different species in 40 countries by 2013 (EDF, 2013), and has been promoted as a way to prevent overfishing and improve economic efficiency (Arnason, 2012; Grimm et al., 2012; Costello et al., 2008; Costello et al., 2010; Wilen, 2005; Newell et al., 2005). A key challenge in introducing ITQ's is initial allocation, an issue recognised by the early architects of rights-based fishing (Scott, 1988; Christy, 1973). The most common method is grandfathering whereby incumbents at the time of introduction are gifted quota based on fishing history (Lynham, 2014). This approach has been justified on the basis that it secures fishing industry cooperation (Lynham, 2014; Grainger & Costello, 2016) but has faced challenges from other stakeholders including fishermen, processors, local communities and society at large due to concerns about the impact on employment, barriers to entry and the allocation of resource rent (Grainger & Parker, 2013; Gretarsson, 2011; Pinkerton & Edwards, 2009; Matthiasson, 2008; Huppert, 2005; Eythorsson, 2000; Matulich et al., 1996; Squires et al., 1995). In some ITQ fisheries, controversy has persisted long after the initial grandfathering decision, exacerbated by perceptions of windfall profits and culminating in calls to reallocate

the rent associated with quota (Bromley, 2015; Macinko, 2014; Bromley and Macinko, 2007; Clark, 2006).

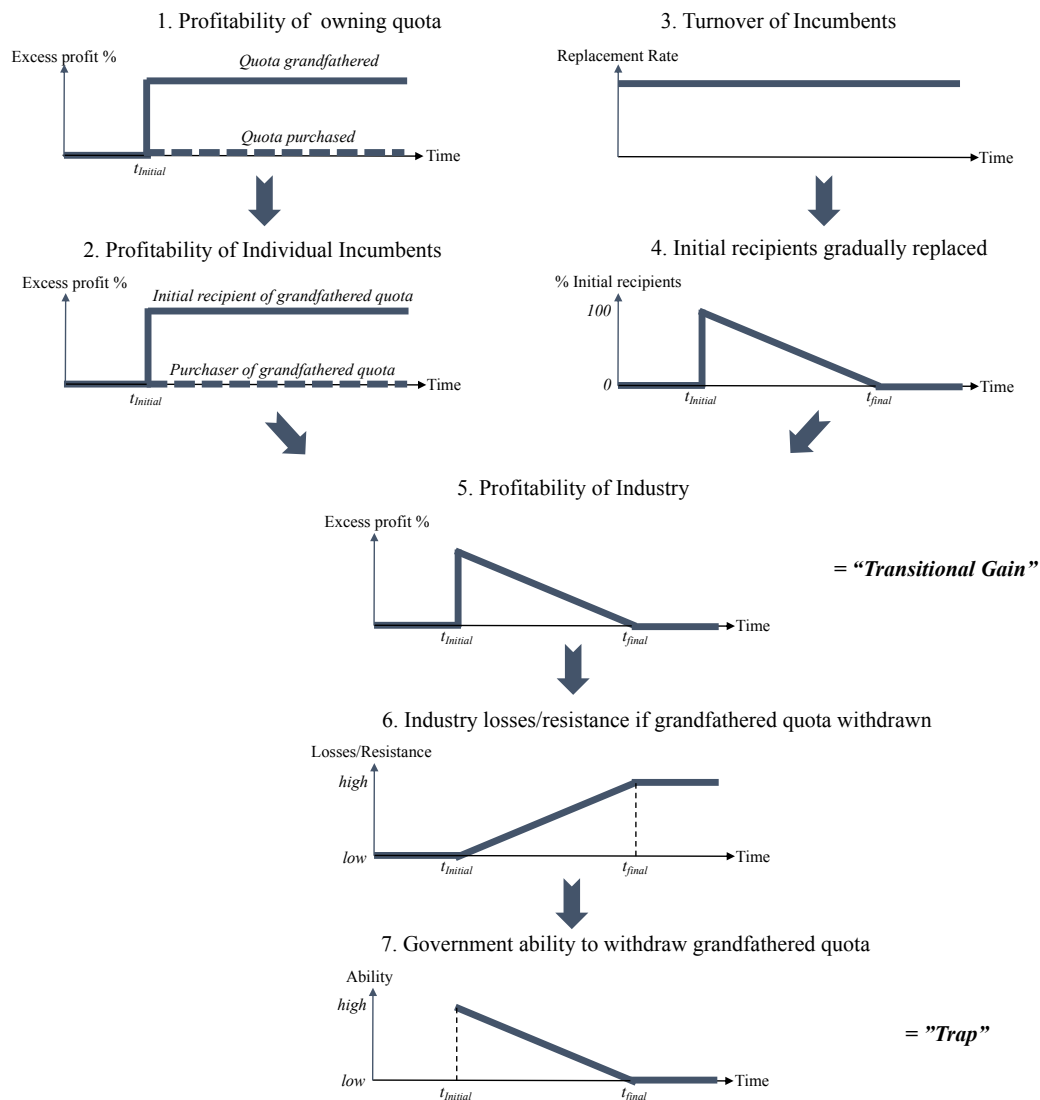


Figure 1. Stylised representation of the Transitional Gains Trap. Quota is grandfathered at time $t_{initial}$ and the trap is complete by time t_{final} .

The Transitional Gains Trap (Tullock, 1975) has been cited as a political barrier to reforming grandfathered ITQ systems (Copes, 1979, 1986; Flaaten et al., 1995, 2017). It states that when a government supports an industry, profits only increase temporarily before returning to normal levels, at which point it becomes politically impossible to withdraw the support. Tullock (1975) proposed several mechanisms by which this could happen but, in the context of grandfathered ITQ, the distinction has been made between the high profitability of owning quota received due to grandfathering which generate windfall profits and the low profitability of owning quota purchased at market prices which do not therefore generate excess profits (Fig. 1 panel 1). The

profitability of individual industry incumbents reflects the extent to which they purchased their quota; an incumbent which holds purely grandfathered quota which was not paid for will enjoy high excess profits while an incumbent with purely purchased quota will not earn any excess profits from their quota (Fig 1. panel 2). The distinction is also made here between the profitability of owning quota and the profitability of incumbents because the latter can own a mixture of grandfathered and purchased quota and therefore exhibit an intermediate level of profitability, although Tullock did not make this point in his original paper. Over time, the initial recipients of grandfathered quota are replaced by later generations that purchase quota from the initial recipients at market prices and do not earn excess profits (Fig 1. panels 3-4). Eventually, all industry participants only earn a normal rate of profit, despite continuing to receive resource rent by virtue of holding quota, and overall industry profits return to normal levels so that the excess profits gained are only transitional (Fig 1. panel 5). As a result, the industry is faced with significant losses in the event of withdrawal of grandfathered quota, increasing industry resistance to reform of quota allocation to the point that the government is trapped (Fig 1. panels 6-7).

The main policy implication of the Transitional Gains Trap is the gradual loss of flexibility to reverse legislation (Acemoglu & Robinson, 2019). For policymakers contemplating the introduction of grandfathered ITQ, this may make more flexible alternatives such as fixed term quota or quota auctions appear relatively attractive¹. For policymakers considering reform of existing grandfathered ITQ systems and reallocation of the associated rent, the TGT raises an empirical question - what flexibility remains? More specifically, to what extent have the initial recipients already cashed in windfall gains and been succeeded by companies who only earn a normal rate of return?

Despite the potential policy relevance of TGT, we have been unable to identify any quantitative studies in the fisheries literature and only identified one beyond fisheries, which examined voter disposition to property rights reform but did not quantify the extent of the trap (Ferrell, 2019). The purpose of the current paper is to address this gap by proposing a measure of TGT based on the potential loss faced by purchasers of quota in the event of revocation. The proposed TGT measure is applied to the Icelandic ITQ system from its introduction in 1991 to

¹ According to Fischel (1995) the reversibility of legislation is a “crucial means of ensuring that bad laws have short lives”. See also North (1993) for a discussion of adaptive efficiency and the need for flexible institutions.

2016. It should be emphasised that the analysis relies on broad-based assumptions, does not consider changes in the ownership of the companies registered as ultimate owners of quota, and focuses exclusively on permanent quota shares, ignoring the lease market. For these reasons, the results for the Icelandic ITQ system should be considered illustrative.

The Icelandic ITQ System

The origin of Iceland's current ITQ system is the Fisheries Management Act of 1990 which introduced permanent, transferable and divisible quota (Agnarsson et al., 2016). Although individual fishing quota existed prior to this time, they co-existed with other systems, including effort-based allocation and trade was subject to restrictions and negligible (Eythorsson, 1996, 2000; National Economics Institute, 1999; Arnason, 2005). The current ITQ system initially comprised 19 quota types, corresponding to distinct species and, in the case of shellfish, specific regions (Table 1). The system has subsequently been extended and 49 distinct ITQ types had been employed by 2016. Quota are transferable as either permanent shares (specified as a percentage of TAC allocated to the ITQ system) or annual leases (specified in tonnes). The ITQ system has also been adapted to incorporate fleet segments such as small hook-and-line vessels which previously operated under permits, catch or effort restrictions (Matthiasson & Agnarsson, 2010). In these instances, a portion of overall quota has been reallocated from existing owners to the additional fleet segment. It should be noted that the first article of the Fisheries Management Act states that Icelandic fish stocks are the common property of the nation and that harvest rights conferred under the act are not irrevocable and do not constitute ownership rights (Matthiasson, 2012, Althingi; 2020).

Table 1. ITQ type by year of introduction (Source: Directorate of Fisheries).

ITQ Type	Year introduced	ITQ Type	Year introduced
Cod	1991	Long Rough Dab	1997
Haddock	1991	Cod - Norway	1999
Saithe	1991	Cod - Russia	1999
Redfish	1991	Lemon Sole	1999
Greenland Halibut	1991	Ocean Redfish - Inside EEZ	2000
Plaice	1991	Ling	2001
Atlantic Herring	1991	Tusk	2001
Capelin	1991	Angler	2001
Lobster	1991	Blue Whiting	2002
Shrimp – 13 regions	1991 - 2014	Atlanto Scandian Herring	2002
Scallop – 7 regions	1991 - 2002	Atlanto Scandian Herring - Norway	2002
Atlantic Wolffish	1996	Deepwater Redfish	2010
Witch Flounder	1996	Blue Ling	2013
Oceanic Redfish - outside EEZ	1997	Greater Argentine	2013
Common Dab	1997	Norway Redfish	2013
		Haddock - Russia	2016

The amount of quota initially allocated to fishing vessels has mainly been based on catch history (Runolfsson & Arnason, 2001). For example, the fishing quota for demersal species were allocated to vessels based on their landings during the previous three years. There are exceptions; herring quota were divided equally between vessels while capelin quota were partly allocated based on hold capacity (Matthiasson, 2012). However, the common feature of these different allocation methods is that quota were initially gifted by the government to incumbent fishing companies, that is “grandfathered”, rather than being sold.

Quota are attached to vessels and are therefore the harvesting rights of the vessel owners. Quota trade can arise directly, when quota are transferred between vessels owned by different companies, or indirectly, when the vessel or its owner are sold to a different company and the quota automatically follow (Fig. 2). It should be noted that transfers may also occur between vessels owned by the same company, in which case there is no trade. Ownership of quota may change when vessel owners merge, even though the amount of quota effectively held by the original owners may not change. Both quota transfers and changes in vessel ownership are registered with the Directorate of Fisheries (“DoF”), a government agency which administers

the fisheries management system (www.fiskistofa.is) making it possible to track changes in quota held by specific vessels and the associated vessel owners.

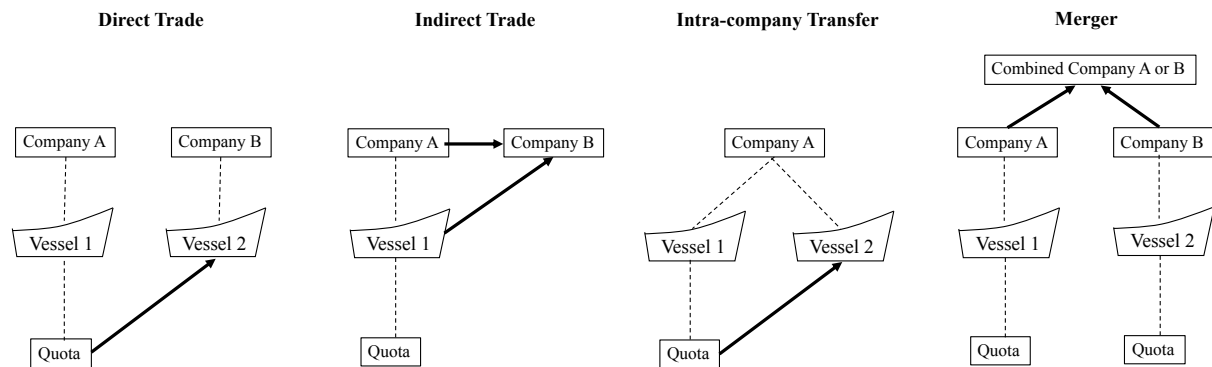


Figure 2. Different quota transfer mechanisms.

The Icelandic ITQ system has been credited with improving industry performance (Gunnlaugsson & Saevaldsson, 2016; Knutsson et al., 2016; Arnason, 2005) but has also met resistance, one of the principal issues being initial allocation (Kokorsch et al., 2015; Matthiasson, 2012; Benediktsson & Karlsdottir, 2011; Eggertsson, 2009). There have been legal challenges from fishermen who did not receive grandfathered quota, as well as communities where quota have been sold to other regions at a cost to the local economy (Kokorsch et al., 2015; Gretarsson, 2011; Benediktsson & Karlsdottir, 2011). Grandfathering has also resulted in political pressure to reallocate resource rent from the industry to the Icelandic nation as owner of the country's marine resources (Matthiasson 2012; Matthiasson, 2008). Despite the introduction of a profit-based quota fee in 2002 and subsequent increases, some political parties have pushed for more radical reform including phased withdrawal and auction of quota (Matthiasson, 2012; Frettatiminn, 2016; Morgunbladid, 2016), as has occurred in the Faroe Islands (Marter, 2018). During a 2010 Icelandic government review of options to reform the ITQ system (Icelandic Ministry of Fisheries & Agriculture, 2010), the Transitional Gains Trap was cited as a barrier to reallocation of quota. In the supporting analysis, the annual volume of quota changing ownership for three major demersal species (cod, haddock and saithe) was used to establish the ratio of quota retained each year. By compounding this annual retention ratio over 1991-2008, it was estimated that only 17% to 24% of quota in these species had been retained by the original recipients (Kristofersson, 2010).

Theoretical Framework of Transitional Gains Trap

Tullock's original paper (1975) does not contain a formal model from which a measure of TGT can be derived. This section therefore suggests a simple model and then offers a trade-based measure of TGT intended to fit the characteristics of ITQ fisheries. The model assumptions are briefly discussed but this study does not attempt to validate them; its aim is limited to developing a measure of TGT consistent with Tullock's original paper and presenting illustrative calculations for the Icelandic ITQ system.

Simple TGT model

Tullock introduces the Transitional Gains Trap concept by posing a question - why is it that government programs to privilege particular industries fail to improve profitability in the long run and yet they are not withdrawn? While there are some ambiguities in Tullock's paper, one interpretation² of his response can be summarized as follows: (a) incumbents when a privilege is introduced benefit from it without any cost but are inevitably replaced over time by successors who pay, directly or indirectly, to enjoy the privilege; (b) successors will resist withdrawal of the privilege to the extent that they have paid for it and would suffer below-normal returns; and (c) when the whole industry resists withdrawal, it becomes politically impossible. Tullock does not consider the extent of the trap during the transition period but it seems reasonable to assume that that the trap can be partial reflecting, for example, the likelihood that the privilege will continue or the government effort required to withdraw it.

The above assumptions suggest a model based on three aggregate variables: (a) the extent to which incumbents have paid for a privilege; (b) the combined resistance of these incumbents to withdrawal of the privilege; and (c) the degree to which this resistance prevents the government from revoking the privilege. The most straightforward reading of Tullock's original paper would involve the direction of causation from (a) to (b) to (c) with all variables increasing over time, although these restrictions are not necessary. A general model (Fig. 3, left-hand panel) could feature feedback effects, for example the incentive for quota owners to trade quota in order to accelerate TGT and reduce the risk of revocation. Exogenous politico-

² This interpretation is viewed as most consistent with the presentation of TGT in the fisheries economics literature and also with much of the argumentation in Tullock's original paper. However, Tullock's position is not entirely clear. For example, in his discussion on taxi medallions, he states that surviving original owners have opportunity costs on which they will only receive normal returns, effectively eliminating the distinction between initial recipients and successors. In this alternative interpretation, the transition is not driven by trade but rather by a gradual adjustment of capital values. Of course, these different interpretations are not mutually exclusive.

economic factors such as the political system, reliance on the fishing industry and access to finance may also be important. Finally, the functional forms of these relationships could be non-linear (for example, due to economies of scale in lobbying) and subject to discontinuities (for example, due to a dominant core group of quota owners³).

In order to simplify the analysis and focus on the role of trade, a stylized model is adopted here in which the above variables take values between 0% and 100%, and respond linearly in the direction of causation described above (Fig. 3, right hand panel). According to this model, the extent to which industry incumbents have paid for a privilege also indicates the extent of TGT; when 100% of incumbents have paid for a privilege the extent of the trap is then 100%. These simplifications do not preclude extension of the results presented in this paper using more sophisticated functional forms.

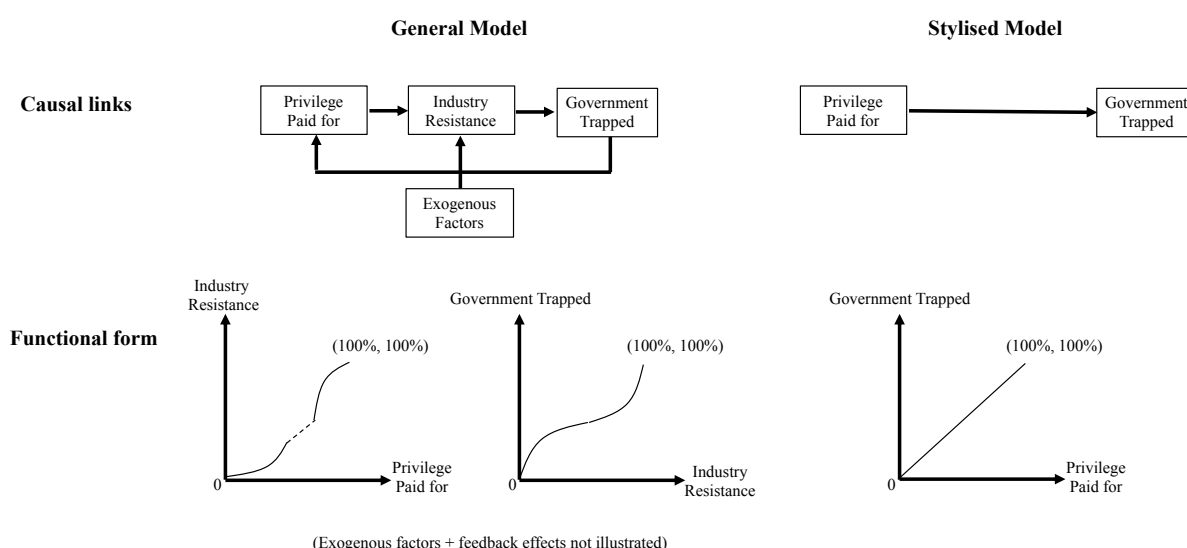


Figure 3. General and stylized models of TGT.

A number of observations are in order regarding the basis for the above model. Perhaps of greatest importance is the idea that sunk costs matter; successors resist withdrawal because they have incurred costs while the original recipients have not. Although Tullock discusses the concept of opportunity cost, it is clear that he is concerned with the actual cost incurred in purchasing quota - this is the basis for distinguishing between the initial recipients and their

³ In Iceland, the 2020-21 board of the main trade association, Seafood Iceland, comprised 20 individuals of which 18 represented fishing companies together accounting for 58% of quota. (Seafood Iceland, 2020; DoF, 2020).

successor and therefore fundamental to the existence of a Transitional Gains Trap. Viewed from the perspective of opportunity cost, this distinction disappears and with it the whole concept of TGT. Although this assumption is at odds with the standard focus on opportunity cost (Mankiw & Taylor, 2017), there is evidence to support it in certain contexts. For example, the behavioural economics literature describes how consumers can be influenced by sunk costs and discount opportunity costs (Thaler, 1980). There is also evidence that sunk costs can influence corporate decision-making, for example due to the associated debt and risk of bankruptcy (Flaaten, 2010; Roth et al., 2015). Cherry et al. (2002) show how the outcome of dictator games can depend on how the wealth was initially acquired, arguing that earned wealth is less likely to be redistributed since it is perceived as legitimate. Interestingly, this perception appears to attenuate the desire for redistribution on the part of players not initially receiving wealth (Oxoby & Spraggon, 2008), suggesting that the voter disposition towards reform of quota allocation may be influenced by TGT. The practical relevance of historic cost has been commented on in the case of Canadian dairy industry quota reform where assistance to farmers was based on original cost (Barrichello et al., 2009; Trebilcock, 2014).

Tullock illustrates TGT by drawing on examples involving both people and firms and, while the above model is applicable to both groups, it is important to recognize some important differences. For example, the potential for generational succession to drive TGT (Copes, 1986; Flaaten et al., 2015) is clearly relevant to individuals but need not apply to companies which can persist indefinitely. Additionally, companies are likely to use different methods of influencing government, for example by way of lobbying, compared to individuals who can vote. The above model abstracts from issues which a more general model would address but are beyond the scope of this paper. The focus on successors paying for a privilege ignores the possibility that initial beneficiaries may also incur costs in order to secure the privilege in the first place for example by racing for catch history (Lynham, 2014), or that successor willingness to pay may be attenuated by the perceived risk of subsequent withdrawal (Grainger & Costello, 2014). The above model also does not consider what motivates government and industry decisions and the resulting scope for strategic behaviour (Holcombe, 2018).

TGT model applied to grandfathered ITQ

In applying the above model to grandfathered ITQ, it is helpful to consider some distinguishing features of fishing quota compared to other privileges - emphasis is placed here on divisibility and multiple quota types within a single ITQ system. These properties allow harvesters to vary

in the extent of their quota holdings and to simultaneously hold both retained grandfathered quota and purchased quota, potentially in different quota types. In such cases, only the amount purchased represents an investment that contributes to TGT in the above model. This means that companies that have bought quota can nonetheless vary in how much they contribute to TGT.

The ability to simultaneously hold grandfathered and purchased quota also has dynamic implications for the potential loss faced by a harvester in the event of revocation. In particular, the cost of purchased quota may be offset over time by windfall profits from retaining or selling grandfathered quota. For example, if a company purchases quota in one species using proceeds from the sale of grandfathered quota in another species, it can be argued that the company's overall net investment in quota is zero. The impact of resource rent is more complicated; where it is accumulated due to retained grandfathered quota, this can also be viewed as windfall profits that mitigate a company's overall potential losses. However, resource rent due to purchased quota must be weighed against the incremental financing cost - the overall effect on net investment may be positive or negative.

Proposed TGT Measure for ITQ fisheries

The above observations suggest a TGT measure for ITQ fisheries which reflects the net investment cost incurred by each harvester, as set out below. The TGT measure at the end of period T is defined in (1). The first term normalizes the measure with P_T referring to the price of one percentage share of quota at the end of period T . This normalization facilitates comparison across quota types of different value and also allows interpretation of the measure as the percentage of outstanding quota needed to compensate all companies facing a potential loss. The second term in (1) sums cumulative net investment cost across all N companies by the end of period T , subject to an upper limit for each company equal to the value of quota held on the basis that this acts as a ceiling on the resources that company would dedicate to protecting its quota, with $\%Q_{own_{i,T}}$ referring to the quota share held by company i at time T . This limit also ensures that the TGT measure does not exceed 100% since a company's contribution will not exceed its quota share. The condition in (2) requires that only incumbents with positive net cumulative investment costs contribute towards the TGT measure, reflecting the TGT focus on incurred cost rather than opportunity cost. $Net\ Cost_{i,T}$ is defined in (3) with $\%Q_{buy_{i,t}}$ and $\%Q_{sell_{i,t}}$ referring to the quota percentage bought and sold respectively by company i during period t . The amount of quota held is defined in (4) and reflects not only

quota trade but also quota allocated, $\%Qallocated_{i,t}$. Allocations comprise primarily quota initially grandfathered but can also be reallocation of quota, for example when a new fleet segment is incorporated into the ITQ system and dilutes pre-existing quota shares. $Rent_t$ refers to the resource rent attributable to one percent of quota share in period t and r is the cost of capital.

$$\%TGT_T = \frac{1}{P_T} \cdot \sum_{i=1}^N \text{Minimum} (\text{Net Cost}_{i,T} , P_T \cdot \%Qown_{i,T}) \quad (1)$$

$$\forall i : \text{Net Cost}_{i,T} > 0 \quad (2)$$

where, (3)

$$\text{Net Cost}_{i,T} = \sum_{t=1}^T (P_t \cdot \%Qbuy_{i,t} - P_t \cdot \%Qsell_{i,t} - Rent_t \cdot \%Qown_{i,t}) \cdot (1 + r)^{T-t}$$

$$\%Qown_{i,T} = \sum_{t=1}^T (\%Qallocated_{i,t} + \%Qbuy_{i,t} - \%Qsell_{i,t}) \quad (4)$$

The proposed measure is defined for a single quota type but can readily be adapted to provide an aggregate measure of TGT for a fishery with multiple quota types. In this case, the quota share owned by company i is calculated as the total value of quota owned by the company relative to the total value of quota for the industry, P_t becomes the aggregate price of 1% of quota for all quota types and the quantity in (3) indicates the aggregate net investment by company i across all types. An advantage of the aggregate measure is that it takes account of situations where a company has realized excess profits for some quota types but losses for others. Where broad-based reform of an ITQ system is contemplated, such an aggregate measure may be more relevant.

An interesting property of the proposed measure is that it can decline over time if part of a company's quota was grandfathered because accumulation of resource rent from the grandfathered quota can gradually offset the cumulative net investment cost of purchased quota. This is illustrated in Fig. 4 which shows how accumulation of resource rent affects the net investment of companies with equal amounts of grandfathered and purchased quota. The

net investment of company (A) which received grandfathered quota is initially zero and becomes increasingly negative (i.e. surplus) as resource rent is accumulated and compounds in value. The net investment of company (B) with purchased quota is initially the purchase price of the quota. This quota yields the same resource rent as grandfathered quota but also attracts a cost of capital. For simplicity of exposition, these are assumed to cancel out so that the Net Investment is constant, although this need not be the case⁴. Company (C), which holds both types of quota, will follow an intermediate trajectory with its net investment initially being positive but falling and eventually becoming negative. At the point of transition labelled “Breakeven”, the cumulative value of rent already received from grandfathered quota equals the net cost of purchased quota and the company no longer contributes to TGT. The higher the ratio of grandfathered to purchased quota, the earlier a harvester will reach this point. Although this illustration is highly stylized in that it does not allow for ongoing trade, it is argued below that such an approach may be useful in a particular scenario described further below.

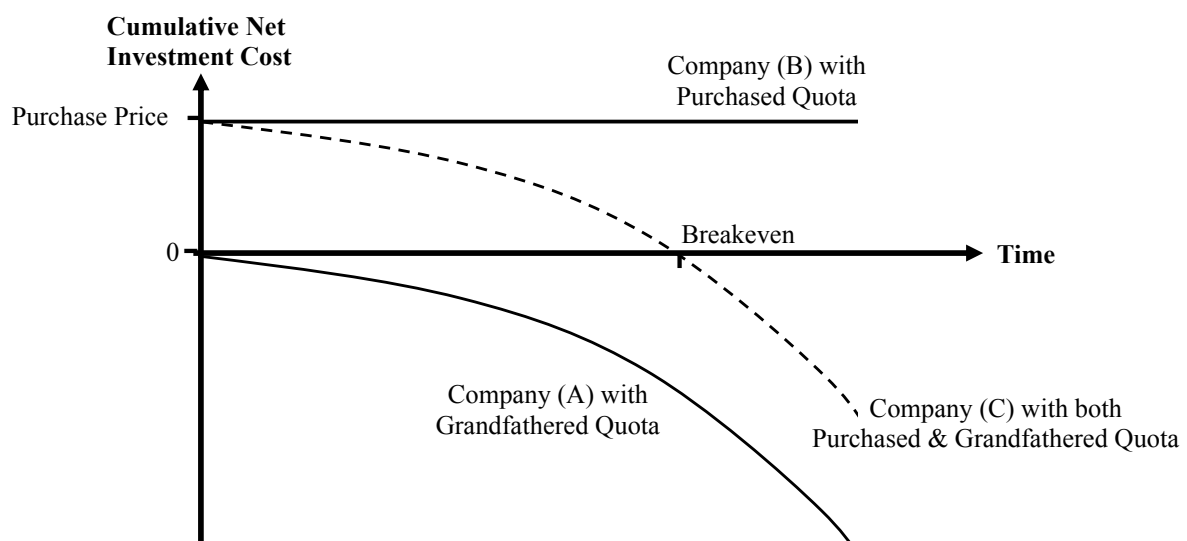


Figure 4. Effect of rent accumulation on cumulative net investment cost of quota holders.

A convenient feature of the proposed measure is that, under simple assumptions of a fixed quota price and no resource rent or cost of capital, it reduces to the percentage of quota no longer held by the original recipients, referred to here as cumulative net trade or $\%CNT_T$ in (5).

⁴ For example, the perceived risk of revocation may reduce quota prices and result in resource rent exceeding the cost of capital. Equally, if TACs are unexpectedly cut then resource rent may fail to cover the cost of capital.

This quantity is consistent with TGT references which emphasise how much quota has been sold by the initial recipients (Copes 1986; Kristofersson, 2010).

$$\%CNT_T = \sum_{i=1}^N \sum_{t=1}^T (\%Qbuy_{i,t} - \%Qsell_{i,t}) \quad \forall i \sum_{t=1}^T (\%Qbuy_{i,t} - \%Qsell_{i,t}) > 0 \quad (5)$$

TGT forecast scenario

In general, forecasting the proposed TGT measure would be problematic due to the need for assumptions, particularly regarding company-level quota trade. However, there is a scenario in which future quota trade should be irrelevant; when a government credibly commits to revoke quota at a specified future date T' after the date of announcement T . Provided the commitment is credible, subsequent quota prices should only reflect the value of resource rent up to the point of scheduled revocation and trade after the announcement should not influence losses due to revocation. In this scenario, the evolution of TGT depends only on each company's net cost at the time of announcement and the ongoing impact of resource rent and the cost of capital, as shown in (6). Such analysis may be helpful to policy makers evaluating the feasibility of quota reform under different time frames and illustrative calculations are therefore presented in the current paper.

(6)

$$Net\ Cost_{i,T'} = Net\ Cost_{i,T} \cdot (1+r)^{T'-T} - \sum_{t=T}^{T'-T} (Rent_t \cdot \%Qown_{i,T}) \cdot (1+r)^{T'-t}$$

Methodology

This study calculates the proposed TGT measure in (1) and cumulative net trade in (5) for 16 quota types in the Icelandic ITQ system over the period 1991-2016. The aggregate TGT measure is also calculated over the same period. As discussed above, the aggregate measure allows a company's net investment cost in one quota type to be offset by accumulated profits in another quota type. The impact of this offsetting can be elucidated by comparing the aggregate TGT measure to weighted average TGT for the same quota types, provided the weightings are consistent i.e. based on quota value. Weighted average TGT is therefore calculated over the period 1991-2016 for comparison to the aggregate TGT measure.

Calculating the proposed TGT measure requires estimates of each company's cost of capital and resource rent as they are not directly observable. The cumulative compound nature of the proposed measure means that errors in these estimates may significantly affect the results. Sensitivity of the results is therefore examined by comparing the aggregate TGT measure across scenarios in which each assumption is varied in turn.

Finally, the aggregate TGT measure is forecast for 20 years, assuming no quota trade. Quota prices are held at 2016 levels and resource rent is assumed to equal the product of quota price and the cost of capital, for consistency. The implied time for each company to break even (referred to as "Breakeven" in Fig. 4) is also calculated by determining the time at which the Net Cost for that company as defined in (6) reaches zero. An expression for the breakeven time for company i can then be derived algebraically and is defined here as BE_i with time notation dropped for simplicity of exposition (all values are at the beginning of the forecast period).

$$BE_i = \frac{\ln(Rent \cdot \%Qown_i / (Rent \cdot \%Qown_i - r \cdot Net\ Cost_i))}{\ln(1 + r)} \quad (7)$$

As can be seen, the condition for a company to reach breakeven is that forecast resource rent received exceeds the cost of capital associated with its net cost of investment. The assumption that quota yield is equal to the cost of capital reduces (7) to the expression in (8), in which a company's cumulative net investment must be less than the value of its quota holding if it is to break even and stop contributing to TGT.

$$BE_i = \frac{\ln(1 / (1 - (Net\ Cost_i / P \cdot \%Qown_i)))}{\ln(1 + r)} \quad (8)$$

Treatment of merged companies

Ownership changes resulting from mergers may lead to overestimation of TGT to the extent that they do not increase the owners' potential losses from revocation. The Icelandic harvesting sector has undergone significant merger activity (Runolfsson 2000, Jonsdottir & Knutsson 2009) suggesting that this bias may materially affect the results. In order to allow for this, clusters of companies identified by the authors as merging together during the period are each consolidated and treated as a single company. It should be emphasised that this approach is an approximation which may lead to underestimation of TGT to the extent that a potential

loss faced by one of the merging companies prior to merger is netted off against accumulated profits of another company. It should be noted that each cluster is treated as a single entity over the entire period until 2016 regardless of when mergers occurred during the period, in order to simplify the analysis. As a result, the TGT measure for years prior to 2016 will be overstated to the extent that it reflects mergers that occurred later. Another limitation is that available data is not exhaustive regarding merger activity and some mergers may not be accounted for, particularly if smaller companies are involved. Correcting for these issues is challenging and involves looking through the corporate veil at the position of underlying shareholders which is beyond the scope of the current study. Instead, each merger cluster is consolidated in the base case and the impact is highlighted in sensitivity analysis where clusters are not consolidated.

Data

The main data comprised individual vessel quota holdings and owner details which were combined to derive annual quota holdings and trade, direct and indirect, by owner between 1991-2016. These data were complimented by quota share price and resource rent data. Quota holdings were obtained from DoF for 16 quota types accounting for an average of 84% of harvesting revenue over the period and involving 2,949 vessels. Fishing years normally begin on 1st September and quota holdings are based on this date except for the first fishing year which commenced on 31st January 1991 and lasted only 8 months. Vessel ownership details from late 1992 until 2016 were obtained from DoF while details for the first fishing year were extracted from the 1991 vessel register (Sailing Institute, 1991). Vessel ownership details were not available for the beginning of fishing years 1992-93 and were assumed to be unchanged from 1991. Data regarding mergers were obtained from Runolfsson (2000) and Gudmundsdottir (2018) and supplemented by online press searches using keywords for “merger” and names of the 50 largest harvesters.

Quota share price were obtained from reports by the National Economics Institute (1999), Institute of Economic Studies (2010) and Integra Consulting (2017) and from the Central Bank of Iceland (personal communication, 4 October 2019), as well three newspaper articles published in 2014-15. The Integra Consulting report (2017) was based on estimated values rather than reported prices. Where prices were expressed per kilogram, they were converted to share prices using TAC allocations. For some species-year combinations, price data were not available and were therefore interpolated where adjacent values were available or otherwise

held constant (Table 2). All values are expressed in real terms with reported values deflated using the retail price index published by Statistics Iceland.

Table 2. Price per 1% quota share in millions of Icelandic krona (2016 prices). Underlined values are interpolated where adjacent values are available and otherwise held constant.

Species	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000	2001	2002	2003	2004	2005	2006	2007	2008	2009	2010	2011	2012	2013	2014	2015	2016	
Cod	<u>1,005</u>	1,005	729	554	792	1,139	1,892	2,667	3,091	3,474	2,499	<u>2,936</u>	<u>3,373</u>	3,810	4,149	5,325	7,623	6,390	2,434	2,678	3,004	<u>3,544</u>	<u>4,083</u>	4,623	4,945	<u>4,945</u>	
Haddock	<u>176</u>	176	169	111	99	125	206	281	235	292	270	<u>329</u>	<u>388</u>	447	594	1,001	1,252	<u>1,139</u>	<u>1,026</u>	<u>913</u>	<u>800</u>	<u>687</u>	<u>574</u>	461	<u>461</u>	<u>461</u>	
Saithe	<u>138</u>	138	135	108	84	77	84	119	145	169	160	<u>189</u>	<u>218</u>	247	312	344	405	<u>410</u>	<u>415</u>	<u>420</u>	<u>425</u>	<u>430</u>	<u>436</u>	441	<u>441</u>	<u>441</u>	
Redfish	<u>209</u>	209	217	187	158	237	402	524	493	258	371	<u>361</u>	<u>351</u>	341	334	292	367	<u>375</u>	<u>384</u>	<u>393</u>	<u>401</u>	<u>410</u>	<u>418</u>	427	<u>427</u>	<u>427</u>	
Ling											<u>9</u>	<u>9</u>	<u>9</u>	9	11	14	14	<u>32</u>	<u>50</u>	<u>68</u>	<u>87</u>	<u>105</u>	<u>123</u>	141	<u>141</u>	<u>141</u>	
Atl. Wolffish						<u>30</u>	30	38	36	37	38	<u>57</u>	<u>75</u>	94	71	95	129	<u>122</u>	<u>116</u>	<u>109</u>	<u>102</u>	<u>95</u>	<u>88</u>	81	<u>81</u>	<u>81</u>	
Angler fish											20	<u>20</u>	<u>20</u>	20	30	83	72	<u>65</u>	<u>58</u>	<u>52</u>	<u>45</u>	<u>38</u>	<u>31</u>	25	<u>25</u>	<u>25</u>	
Gr. Argentine																							<u>35</u>	35	<u>35</u>	<u>35</u>	
GrInd. Halibut	<u>96</u>	96	108	105	106	72	88	<u>99</u>	109	135	290	<u>276</u>	<u>261</u>	247	149	141	133	<u>141</u>	<u>150</u>	<u>158</u>	<u>166</u>	<u>174</u>	<u>182</u>	191	<u>191</u>	<u>191</u>	
Plaice	<u>36</u>	36	39	35	35	35	42	42	45	45	54	<u>55</u>	<u>56</u>	57	82	106	138	<u>127</u>	<u>115</u>	<u>104</u>	<u>93</u>	<u>82</u>	<u>71</u>	60	<u>60</u>	<u>60</u>	
Lemon Sole	<u>0</u>	0	0	0	0	0	0	0	0	0	9	10	<u>11</u>	<u>12</u>	14	19	21	39	<u>35</u>	<u>32</u>	<u>29</u>	<u>25</u>	<u>22</u>	<u>19</u>	16	<u>16</u>	<u>16</u>
Atl. Herring	<u>33</u>	33	32	29	40	113	197	<u>162</u>	127	172	172	<u>188</u>	<u>204</u>	<u>220</u>	<u>236</u>	<u>252</u>	<u>268</u>	<u>284</u>	<u>300</u>	<u>316</u>	<u>332</u>	<u>348</u>	<u>364</u>	380	<u>380</u>	<u>380</u>	
Capelin	<u>146</u>	<u>146</u>	<u>146</u>	146	138	166	376	<u>496</u>	617	562	<u>567</u>	<u>571</u>	<u>575</u>	<u>579</u>	<u>583</u>	<u>587</u>	<u>591</u>	<u>595</u>	<u>599</u>	<u>603</u>	<u>608</u>	<u>612</u>	<u>616</u>	620	<u>620</u>	<u>620</u>	
Blue whiting											<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	<u>249</u>	249	<u>249</u>	<u>249</u>	
A-S herring											<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	<u>303</u>	303	<u>303</u>	<u>303</u>	
Lobster	<u>36</u>	36	39	31	29	23	23	<u>24</u>	24	34	45	<u>48</u>	<u>51</u>	<u>53</u>	<u>56</u>	<u>59</u>	<u>62</u>	<u>65</u>	<u>67</u>	<u>70</u>	<u>73</u>	<u>76</u>	<u>79</u>	81	<u>81</u>	<u>81</u>	
TOTAL	1,876	1,876	1,615	1,306	1,483	2,018	3,341	4,452	4,932	5,187	4,504	5,600	6,145	6,690	7,178	8,871	11,644	10,333	6,299	6,465	6,713	7,175	7,671	8,132	8,453	8,453	

The appropriate method of estimating resource rent from financial data has been debated (Hannesson, 2017; Nielsen et al., 2017; Jensen et al., 2019). Economic profits from harvesting have been suggested as a proxy (Lindner, 1992; Asche et al., 2008; Gunnlaugsson & Agnarsson, 2019; Jensen et al., 2019) and are reported annually by Statistics Iceland for the entire industry based on EBITDA less the cost of tangible capital (vessels and gear). However, this approach can be problematic. Firstly, it has been claimed that harvesting profits in Iceland have been partly shifted to processing by transfer pricing within vertically integrated companies (Flaaten et al., 2017), resulting in underestimation. It has also been pointed out that economic profits include other forms of rent such as infra-marginal and market power, resulting in overestimation. Finally, economic profit for mixed fisheries like Iceland's must be allocated to individual species.

Lease prices have also been proposed as an indicator of resource rent (Matthiasson, 2008) and have the advantage that they capture the marginal value of quota and are available by species although Hannesson (2017) cautions that they may overstate resource rent to the extent that they do not fully reflect fixed costs and Asche et al. (2008) suggest there is evidence of this in Iceland. In the current study, alternative estimates of resource rent are presented with the high and low alternatives (lease prices and harvesting economic profit) used in sensitivity analysis and the remaining alternative (economic profit from harvesting and processing combined)

adopted as the base case (Fig. 5). Average annual lease prices between 2000-16 were obtained from the DoF website and from National Economics Institute (1999) and Central Bank of Iceland (personal communication) for earlier years. Lease prices are reported per kilogram and converted to quota share values using corresponding TAC levels.

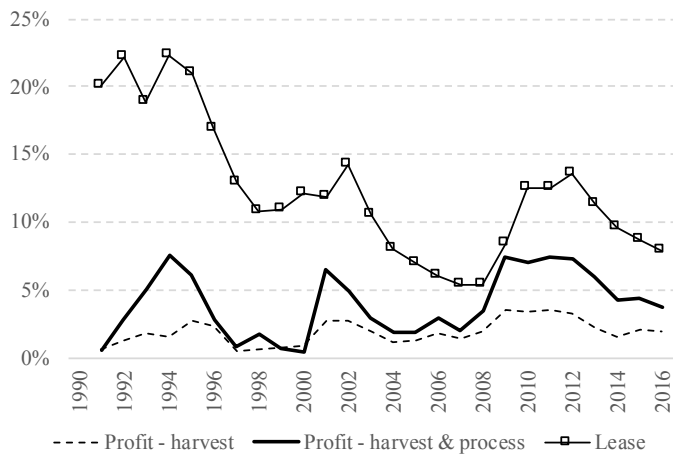


Fig 5. Resource rent relative quota price based on reported economic profits and lease prices.

The cost of capital is not directly observable and was assumed to be a real return of 6%. This is the same assumption used by Statistics Iceland in calculating economic profits and slightly lower than the 6.5% average return on capital calculated by Flaaten et al. (2017) for the Icelandic harvesting sector between 2009-13. Sensitivity analysis is undertaken using alternative values of 4% and 8%. It should be noted that the reported economic profits used to estimate resource rent are not adjusted when the cost of capital is sensitised due to data limitations. In practice, a higher cost of capital would result in lower economic profits. As a result, the cost of capital sensitivity analysis will understate the impact on the TGT measure.

Results

The TGT measure was between 3% for greater argentine and 55% for lobster at the end of fishing year 2016 (Fig. 4) under base case assumptions i.e. resource rent based on economic profits from harvesting and processing, 6% cost of capital and clusters of merged companies treated as single entities. The average TGT value across analysed species was 23% by 2016 while cumulative net trade averaged 61%, indicating that the majority of quota was no longer held by the original beneficiaries. While cumulative net trade generally increased over time, the TGT measure declined in several cases, particularly after 2008 and for pelagic species.

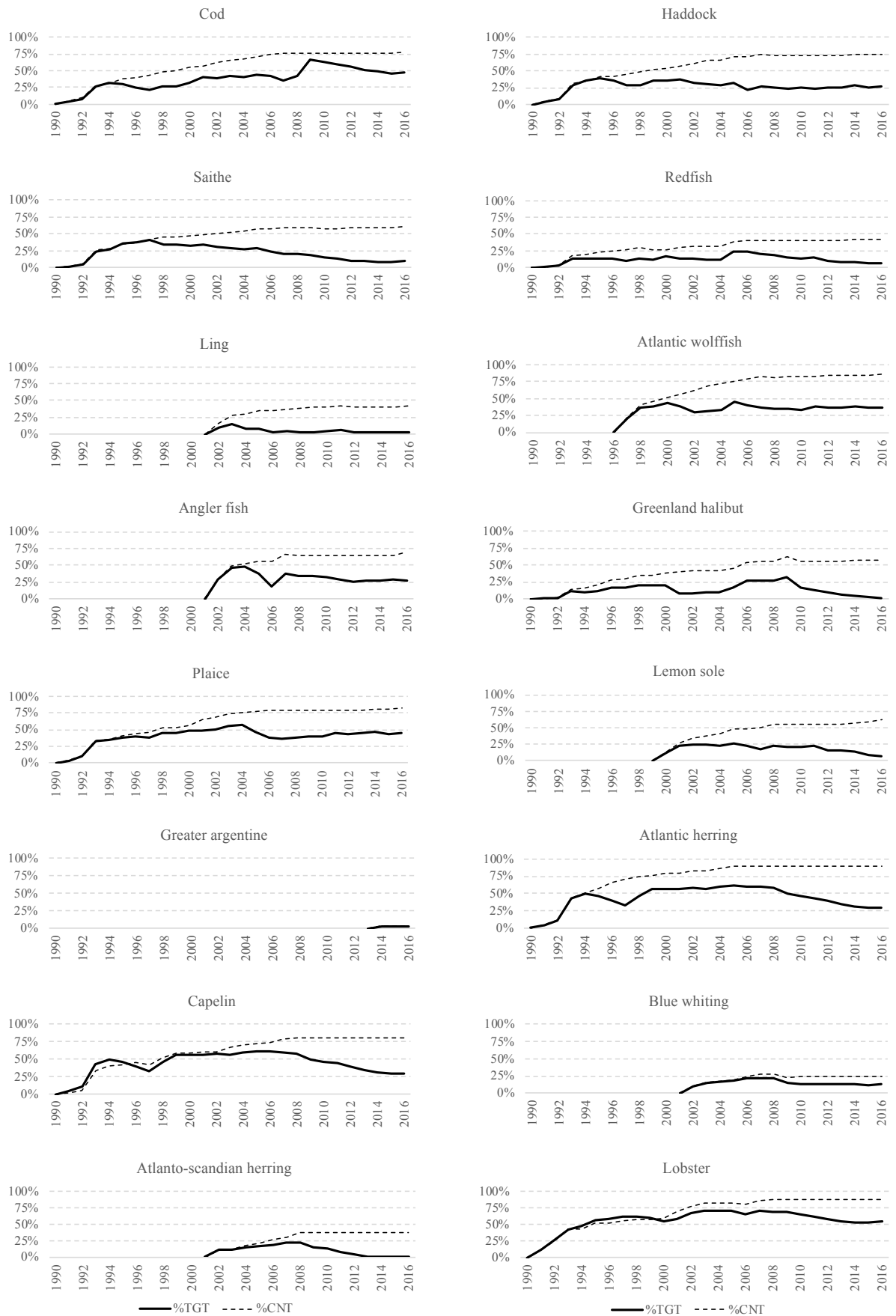


Fig 6. %TGT and %CNT by species for the years 1991 to 2016.

The TGT measure aggregated across the 16 quota types rose rapidly to 26% by 1995 and peaked at 38% in 2009 after which it fell gradually to 29% by 2016 (Fig. 5). The corresponding weighted average TGT was 7% higher or 36% by the end of 2016.

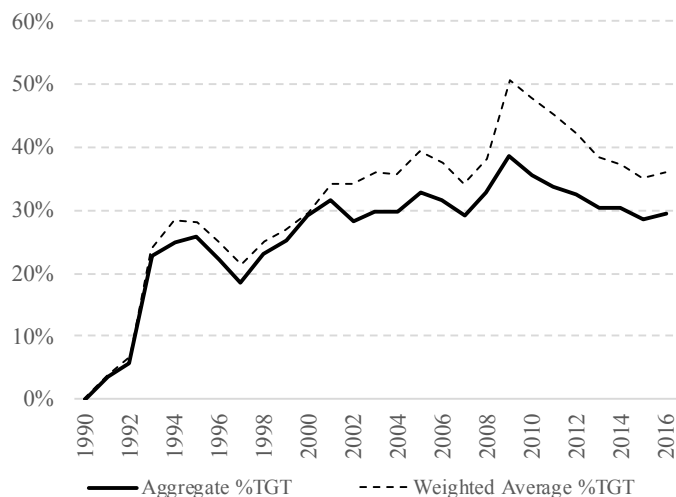


Fig 7. Aggregate %TGT and Weighted average %TGT for the years 1991 to 2016.

Sensitivity analysis highlighted the importance of the assumptions regarding resource rent, treatment of merged companies and cost of capital. Assuming a lower resource rent based on harvesting economic profits increased the aggregate TGT measure in 2016 by 16% to 45% while the alternative, higher resource rent based on average lease prices resulted in a lower aggregate TGT of 10% by 2016 (Table 4). If clusters of companies which merge over the period are not each combined into a single entity, then the aggregate TGT measure increases to 40% by 2016. Adjusting the assumed cost of capital up or down by 2% resulted in aggregate TGT of 35% and 23% respectively. The average TGT measure across all scenarios was 34%.

Table 3. Sensitivity analysis of aggregate TGT measure in 2016. “*” indicates the base case.

	Merged company treatment:					
	Combine as single entity			No adjustments		
	Cost of capital:			Cost of capital:		
	4%	6%	8%	4%	6%	8%
Resource Rent:						
EP - harvesting	38%	45%	55%	53%	63%	72%
EP - harvesting + processing	23%	29% *	35%	31%	40%	48%
Lease prices	8%	10%	13%	13%	15%	19%

Forecast aggregate TGT assuming no further trade, constant quota prices and resource rent consistent with a 6% cost of capital, declined steadily to 19% after 20 years (Fig. 7). The combined quota of companies forecast to be at or above break-even increased from 52% at the beginning of the forecast period to 76% by year 20.

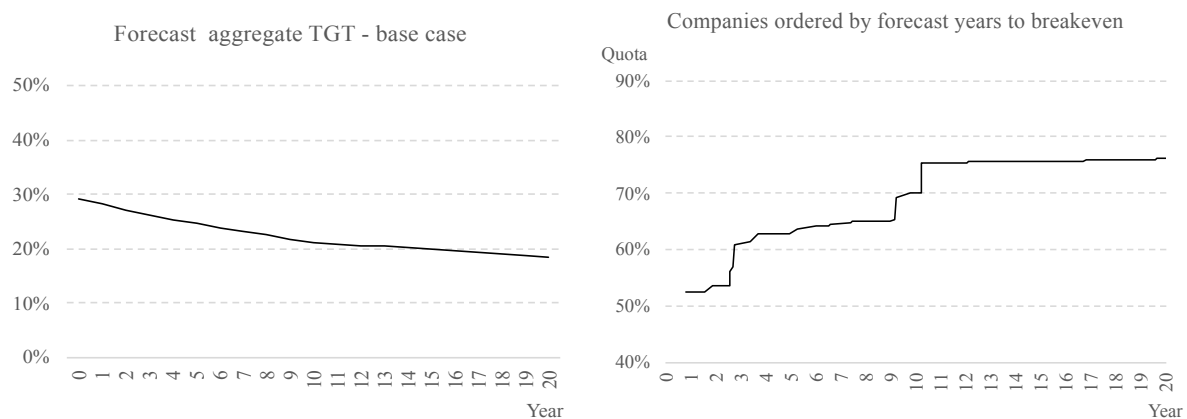


Fig 8. Forecast aggregate %TGT for 20 years from 2016 assuming no trade. Left panel shows forecast measure and right panel shows quota holdings of companies at or above break-even.

Discussion

The results shows that the TGT measure at species level averaged 23% by 2016 and generally remained below 60% under base case assumptions with only 5% of observations exceeding this threshold and primarily relating to lobster. The aggregate TGT measure initially peaked at 38% in 2009 and subsequently fell to 29% by the end of the period. Sensitivity analysis indicated that the results were strongly influenced by methodological assumptions regarding resource rent and treatment of merged companies, particularly in the later years. Finally, forecast TGT declined steadily from 29% in 2016 to 21% after ten years when ignoring the effects of further trade.

The most comparable analysis found in the literature was by Kristofersson (2010) who estimated that 17% of cod quota and 24% of haddock and saithe quota were still in the hands of the original 1991 recipients by the end of fishing year 2009. These estimates imply cumulative net trade for these species of 83% and 76% respectively which are in close agreement with the results of the current study (86%, 84% and 74% respectively, prior to adjusting for merger activity by the end of 2009) despite Kristofersson (2010) following a different methodology based on compounding annual trade levels. Yandle & Dewees (2008) refer to analysis of the New Zealand quota system which found that by 1996, 41% of quota

was no longer held by the original recipients. Over the corresponding ten-year period for the Icelandic quota system (1991-2001), the average ratio for the nine species analysed in the current study and allocated quota in 1991 (Table 1) was 44% prior to adjusting for merger activity and 33% after adjusting. The similarity in results is perhaps not surprising given that the two systems share several attributes; they introduced quota at a similar time, are both geographically isolated and comprise a combination of offshore fisheries exploited by a small number of vertically integrated companies and onshore fisheries involving many small independently owned vessels (Yandle & Dewees, 2008; Byrne et al., 2020).

It is noticeable that cumulative net trade plateaus in the late 2000's for many species, although the level at which it stabilizes varies greatly (Fig. 4). This finding contrasts with the perception that quota trade by succeeding generations inevitably leads to replacement of the initial quota recipients but is consistent with the idea of quota-owning companies of indefinite lifespan, as discussed in the introduction, particularly in the context of a consolidating industry into which entry may be more challenging. It is intriguing that the timing of this stabilization is similar across species, regardless of when they were grandfathered. It seems likely that this pattern is partly due to the 2008 collapse of the Icelandic financial system which had previously facilitated significant merger and acquisition activity (Matthiasson, 2012). Species grandfathered later generally plateaued at lower levels of cumulative net trade (for example ling, lemon sole, atlanto-scandian herring, blue whiting) which is to be expected given that the relevant fleet segments had already partially consolidated by the time these quota were introduced.

TGT by species generally tracks cumulative net trade until the mid 1990's and peaks for several species between 2005-10 or around the same time as cumulative net trade plateaus (Fig. 6). This pattern is consistent with the expected cumulative effect of resource rent gradually eroding individual companies' net investment in quota (Fig. 4). For some species, such as cod, the peaking of the proposed TGT measure is quite pronounced and appears to be partly due to normalization using quota prices which fall dramatically in 2009. The overall TGT measure follows a similar pattern as for individual species, although the influence of the more valuable quota types is evident, with the decline in 1997 coinciding with dramatic increases in the price of the three most valuable quota types at that time (cod, redfish, capelin – see Table 2.) and the converse occurring in 2009-10 for cod. It is reiterated, however, that the price assumptions underlying these results rely heavily on interpolation due to data limitations.

The difference between aggregate and weighted average TGT indicates the effect of pooling each quota owner's cumulative investment and profits across quota types. This effect reaches 7% by 2016 or just over 1/6 of total net investment by companies prior to netting off across species. While evaluation of the relative contribution of offsetting sales and resource rent is beyond the scope of this study, there is some circumstantial evidence that company concentration of quota portfolios contributed to this result. This can be seen by comparing the weighted average TGT to aggregate TGT under a scenario assuming zero resource rent, zero cost of capital and quota prices fixed at 2016 levels. Under these assumptions, the weighted average TGT reduces to a weighted average CNT, with the weightings based on 2016 quota prices TGT while the aggregate TGT becomes aggregate CNT, but allowing for offsetting trade across species. The difference remains at 7% by 2016, suggesting that companies have been concentrating their quota portfolios in order to build scale and specialize in certain species while divesting quota in others, and is consistent with Newell et al. (2005) who found evidence of this behaviour in New Zealand's ITQ system.

Sensitivity analysis highlights the importance of the assumptions tested. Relative to the base case result of 29%, the greatest variation in the aggregate TGT measure was due to changing the resource rent basis; using lease prices yielded a result of 10% while assuming economic profits from harvesting yielded a result of 45% (Table 3). The variation due to changing cost of capital was less significant (6% higher or lower) although, as mentioned above, this sensitivity will be an underestimate since the economic profits are not sensitized⁵. The sensitivity to assumptions regarding resource rent and cost of capital are to be expected given the cumulative compound nature of the proposed TGT measure. In practice, they might be expected to offset each other since, for given quota prices, higher resource rent will tend to be accompanied by higher cost of capital, thus making the most extreme scenarios (high resource rent combined with low cost of capital or, conversely, low resource rent combined with high cost of capital) less likely. Excluding these four scenarios leaves fourteen scenarios of which only two involve an overall TGT measure above 50% (Table 3). However, the main conclusion

⁵ The impact of this underestimation can be roughly calculated in the scenario where resource rent is based on harvesting economic profit because the underlying capital value has been published by Statistics Iceland since 2002 (the corresponding values for processing are not published). A 2% capital charge equates on average to 13% of harvesting economic profits for these years. Allowing for the same change in harvesting economic profits of the entire period analysed, 1991-2016, decreases the TGT measure assuming 4% cost of capital from 38% to 36% and increases the TGT measure assuming 8% cost of capital from 55% to 58% (Table 3.)

to be drawn from the sensitivity analysis is that reliable and accurate market-based assumptions are crucial to estimating the extent of TGT using the measure proposed in this paper.

As expected, the TGT measure is forecast to decline in the absence of further quota trade as harvesters whose quota holdings are of greater value than their net cumulative investment accumulate resource rent in excess of the cost of capital. Forecast TGT begins to level off in year ten, suggesting that there is a rump of harvesters accounting for 19% of industry quota which will not breakeven in the medium term. This result is intriguing given that new entry would might be expected to be limited following introduction of quota due to industry consolidation. An alternative explanation is that initial recipients of grandfathered quota purchased additional quota at inflated prices, for example during the early 2000's when the Icelandic financial sector was liberalized (Matthiasson, 2012). The sharp decline in quota prices following the collapse of the financial sector (Table 2) may have caused capital losses for these companies sufficient to wipe out any accumulated profits from grandfathered quota. As previously discussed, the assumption of no quota trade is unrealistic but can be relaxed in the event of a credible announcement to revoke quota at a future date. To the extent that any announcement is in practice not completely credible i.e. a government can change its mind or be replaced in elections, then the subsequent trade may contribute to TGT because purchasers will attribute some value to the possibility that quota purchased will persist after the announced date of revocation.

Policy implications

It is important to economic development that governments have the freedom to reverse policies (Fischel, 1995), including those which confer privileges on particular sectors (Acemoglu & Robinson, 2019). The standard description of TGT suggests inevitability; as time passes and more quota are traded the TGT deepens until all incumbents have paid market price for their quota after which revocation is impossible since it will impose below normal profit levels on the entire industry. The proposed measure TGT and illustrative results of the current paper challenge this description. When considering not only how much companies invest in quota but also the offsetting effects of divestment and resource rent in excess of the cost of capital, it is shown that the resulting measure of TGT can decline over time. Illustrative calculations across a range of scenarios for Iceland's ITQ system over the period 1991-2016 suggest that the extent of TGT in the Icelandic quota system is far from complete after 26 years. Although the methodology and data employed in this analysis are both subject to limitations, they

demonstrate that the TGT can be examined empirically and should not be taken for granted as a barrier to reform.

These limitations themselves arguably have an additional implication – the importance of transparency in quota markets. While, the Icelandic quota leasing market is active and prices are published (albeit they may still be affected by trade which is not arms-length), permanent quota prices are not reported and the profitability of harvesting is difficult to establish due the high level of vertical integration. This lack of transparency makes it more difficult for stakeholders to have an informed debate about the performance of the current quota system or options for reform and reallocation of the resource rent extracted from Iceland’s marine resources. Finally, to the extent that policy makers are interested to investigate the potential impact of revocation, the current paper suggests a method to forecast the development of TGT. Provided any decision to revoke quota is credibly announced, it is shown that the proposed TGT measure can be forecast without the need for assumptions about future trade. Such forecasts may be useful for identifying companies which have invested most in quota and evaluating the extent to which compensation in the event of revocation may be appropriate. While the forecast presented highlights the benefit of a longer time frame in terms of lower forecast TGT, in practice any government would need to balance against this the lower credibility of decisions over longer time frames and the discounting effect on any fiscal benefits from auction proceeds.

Methodological limitations

In addition to the question of opportunity cost previously discussed, there are at least two potential criticisms of the approach to estimating TGT described in this paper which need to be addressed; (1) the cost of revocation to a company is greater than the value of quota; and (2) it does not capture indirect trade due to transactions occurring at a higher ownership level than the ultimate vessel owner registered by DoF, for example due to trade in its shares. The validity of the first criticism, that companies face additional costs due to quota revocation, depends on what then happens to the quota. The implicit assumption in this paper is that the purpose of revocation is rent capture rather than closure of the fishery; revoked quota would therefore be sold and incumbents would continue to fish as before. Clearly, if revocation were undertaken with other goals in mind, this assumption would be questionable.

The second issue, indirect trade due to transactions which do not result in a change in the identity of the parent fishing company, could have a significant impact on the analysis. An investor purchasing shares in the registered parent company of a vessel could be viewed as contributing to TGT but would not be captured in the measure presented in this paper. Equally, a merger of quota-owning companies could result in vessels and quota being combined into one company and appearing as indirect trade even though the amount of quota effectively owned by each shareholder may not have changed. Analysing these types of transactions presents difficulties due to data limitations. Information regarding shareholders is included in company annual reports but companies are only required by law to cover the largest ten while information regarding corporate transactions is even more limited because Icelandic harvesting companies are today, with one exception, privately held and therefore not subject to the same disclosure requirements faced by listed companies. This discussion also raises broader questions about whether TGT should be viewed as a phenomenon at the level of companies or the people who own them. While companies lobby governments, it is people who elect them and it may therefore be reasonable to consider how shareholders and voters more generally are affected by TGT. However, following this approach to its logical conclusion, it could be argued that individuals are only affected temporarily and the quota inherited by their successors represent uncompensated transfers (Buchanan, 1983) which ought not to contribute to the trap. From this perspective, generational transition effectively nullifies the Transitional Gains Trap and its perpetuation may rely on successive generations continuing to invest afresh in quota, although this raises the further question whether such cost should be offset by any inheritance.

Conclusion

The nature and extent of the Transitional Gains Traps in ITQ fisheries is likely to feature increasingly in debate about the appropriate allocation of rent from nationally-owned marine resources. This paper has argued that it is possible to quantify the extent of the trap in terms of the loss faced by incumbents in the event of quota revocation, taking account of any offsetting profits attributable to ownership of quota. When the extent of TGT is evaluated on this basis, illustrative calculations for the Icelandic ITQ system over the period 1991-2016 indicate that the trap is only partial and has in declined in recent years. While the analysis in this paper relies on several simplifications and broad assumptions, it is hoped that the approach and results presented are of sufficient interest to policy makers to justify further examination of the relevance of the Transitional Gains Trap argument to debate about reform of grandfathered ITQ systems.

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Paper V

Risk assessment for key socio-economic and ecological species in a sub-arctic marine ecosystem under combined ocean acidification and warming

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Under Review in Ecosystems

Author contributions: MO conceived of the research idea, carried out literature research and ran modeling scenarios with the help of ES. MO wrote the first draft of the manuscript and ES and MJS helped with writing.

Risk assessment for key socio-economic and ecological species in a sub-arctic marine ecosystem under combined ocean acidification and warming

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Abstract

The Arctic may be particularly vulnerable to the consequences of both ocean acidification (OA) and global warming, given the faster pace of warming and acidification. Here, we use the Atlantis ecosystem model to assess how the trophic network of marine fishes and invertebrates in the Icelandic waters is responding to the combined pressures of OA and warming. We develop an approach which allows us to focus on species of economic (catch-value), social (number of participants in fisheries), or ecological (keystone species) importance. We parameterize the model with literature-determined ranges of sensitivity to OA and warming for different species and functional groups in the Icelandic waters. We found divergent species responses to warming and acidification levels; (mainly) planktonic groups and forage fish benefited while (mainly) benthic groups and predatory fish decreased under warming and acidification scenarios. Assuming conservative harvest rates for the largest catch-value species, Atlantic cod, we see that the population is projected to remain stable under even the harshest acidification and warming scenario. Further, for the scenarios where the model projects reductions in biomass of Atlantic cod, other species in the ecosystem increase, likely due to a reduction in competition and predation. These results highlight the interdependencies of multiple global change drivers and their cascading effects on trophic organization, and the supply of an important species from a socio-economic perspective in the Icelandic fisheries.

Keywords: Risk, warming, acidification, network analysis, ecosystem model

1. Introduction

Globally, ocean acidification is predicted to increase mortality, impact growth and survival of many marine species (Falkenberg et al., 2018), and likely re-shaping marine ecosystems in the near future. Increasing CO₂ levels in the atmosphere due to anthropogenic activity increase the amount of CO₂ dissolved in oceans, lowering pH levels (IPCC, 2014). Over the last 100 years ocean pH has decreased by 0.1 units, which corresponds to a 30% increase in acidity, and another drop of 0.3 is predicted to occur in the next 80 years (IPCC, 2014). Ocean acidification is affecting pH levels in Arctic and sub-Arctic regions at a faster rate than elsewhere due to low calcium carbon saturation state, increased fresh water input and higher per unit CO₂ absorption due to the colder temperatures (Falkenberg et al., 2018). Not all organisms are equally likely to be affected by ocean acidification. Species with shells of aragonite or calciferous exoskeletons are most vulnerable to acidification, while species lacking these traits may become increasingly dominant due to competitive exclusion (Olsen et al., 2018). Recent studies also provide evidence that lower pH is affecting the larval stage of commercially important fish species, making them highly sensitive to further decreases in pH levels (Frommel et al., 2011). These projected changes in ocean pH levels may be negatively synergistic with the emergence of novel species assemblages and ecosystems in marine systems driven by climate change (Pinsky et al., 2013), and may therefore have a yet unknown effect on productivity as well as affect people that directly depend on these ecosystems for their traditions and livelihoods.

Ocean acidification may have lasting and cascading effects on the food web, although these are still uncertain. A recent meta-analysis on species responses to ocean acidification in the gulf of California showed that one third of over four hundred studies found strong responses of species and trophic cascades to ocean acidification (Busch and McElhany, 2016). Bermudez et al. (2015) found that diatoms contained fewer fatty acids under increased CO₂ levels (750 µatm), which directly translated into a decrease in fatty acids and the caloric content of zooplankton. Copepods are a key diet species to many marine predators, which as a calcifying organism could be negatively affected by acidification (Cripps et al., 2015). However, the results from mesocosm experiments in natural ecosystems are slightly more optimistic than those from the lab studies. Leu et al. (2013), for instance, did not find a detrimental effect of

increased CO₂ on fatty acids and Niehoff et al. (2013) found that effects were likely dampened by community interactions.

Climate change induced increases in temperature have resulted in the ocean having stored more than 90% of the global increase in energy in the climate system (IPCC, 2014), causing a re-shaping of marine ecosystems (Cheung et al., 2011). Shifts in primary productivity could alter productivity in higher trophic levels (Cheung et al., 2011), and ocean warming could alter phenology and cause mismatches in time-sensitive events (Cheung et al., 2011; Pankhurst and Munday, 2011; Sumaila et al., 2011), inhibition of reproduction (Pankhurst and Munday, 2011), migration to lower depth or higher latitudes (Dulvy et al., 2008), and decreases in aerobic performance due to lower oxygen levels in non-mobile species (Pörtner and Knust, 2007). Warming and ocean acidification can also act synergistically in their impact on marine populations as warming can make organisms more sensitive to changes in pH levels (Lischka and Riebesell, 2012) and can impact the flows of nutrients in the marine system by altering chemical processes (Chen et al., 2015). Such synergistic responses have been examined with the Atlantis end-to-end ecosystem model, and were found to have negative effects on biomass when fishing was considered (Griffith et al., 2012). The Atlantis model has been used before to address ocean acidification in various ecosystems, but none of the current applications includes direct ocean acidification impacts on recruitment or mortality of vertebrates (Hodgson et al., 2018; Marshall et al., 2017; Weijerman et al., 2015). Also impacts of ocean warming have previously been studied with the Atlantis model, projecting negative effects of warming on key species in the southern Benguela upwelling (Ortega-Cisneros et al., 2018) and both additive and synergistic negative effects on the Northeast US ecosystem under scenarios of warming and top predator removal (Nye et al., 2013).

Concurrent effects of acidification and ocean warming may also affect the livelihoods and economies of countries greatly dependent on fisheries, such as for example Iceland (Oostdijk et al., 2020). The Icelandic marine ecosystem is already altered due to both ocean acidification and marine ecosystem warming due to climate change. In Iceland, the influx of warm water from the Atlantic, has caused changes in groundfish diversity and assemblage structure by an increase in “warm water species” (Stefansdottir et al., 2010). Previous research has shown that life history strategies of fish change in warmer waters; younger fishes grow faster, while older fish grow slower and ultimately body size will be less large (Brunel and Dickey-collas, 2010).

Moreover, faster growing species are more responsive to changes in temperature (Free et al., 2019). The impact of temperature changes on fish productivity, however, depends on the temperature of a population's environment in relation to its specific temperature niche (Free et al., 2019). Atlantic cod (*Gadus morhua*) and Atlantic herring (*Clupea harengus*) populations at the cooler end of their niche rather benefited from warming, while populations at the warmer end of their niche had a decrease in productivity (Free et al., 2019). Further, two recent studies showed that Atlantic cod larvae exhibited tissue damage due to decreases in pH (Frommel et al., 2011) and decreased recruitment by 8 to 24 percent compared to baseline levels (Stiasny et al., 2016). Another species that is very important in Icelandic fisheries, haddock (*Melanogrammus aeglefinus*), could also be affected by ocean acidification as a large part of its diet is comprised of benthic organisms (Sturludottir et al., 2018). Overall, Arctic (including Iceland) socio-economics have been projected to benefit from the increase in productivity due to warming and to experience a negative impact of ocean acidification, with a positive net effect (Lam et al., 2016).

In this paper, we model the responses of marine food webs to ocean acidification and rising ocean temperatures to determine ecological-economic implications of global change impacts on marine food webs. We use the Atlantis end-to-end ecosystem model adapted to the Icelandic marine ecosystem to conduct simulations of different scenarios of OA and warming and assess their isolated and combined effect on the marine food web. We use network metrics to select key-species for the Icelandic food web and also select species with high socio-economic importance. We parametrize the model with reported sensitivities to warming and acidification in the literature, and then use the model to determine possible impacts of combined warming and acidification scenarios. We expect that our analysis will allow us to identify possible changes in the food web as well as which important fishing sectors may be hit by combined acidification and warming.

2. Methods

2.1. Study system

Ecosystem: Iceland is located in the sub-arctic, but the waters south off the island are relatively warm because of the Atlantic meridional overturning circulation (AMOC), while the northern waters are characterized by polar waters off the Greenland current (Astthorsson et al., 2007). This results in highly variable conditions and ecosystem productivity. The main spawning

grounds of the largest fish populations are in the more productive and warm water to the south while the nursery grounds are in the colder water to the north (Astthorsson et al., 2007).

Socio-economics: The Icelandic ITQ system comprises several fisheries (fisheries with the highest catch-values are listed in Table 1). Atlantic cod is fished by the largest number of participants in the Icelandic fisheries (360 companies in 2015, Table 1) and represents almost half of the catch-value (Statistics Iceland, 2015). Haddock is the second largest contributor to catch-value in Icelandic fisheries (Statistics Iceland, 2015). Per sector, the demersal sector has the most participants, with greater participation on species like Atlantic cod and haddock. On the other hand, the pelagic sector in Iceland is very concentrated with only a few vessels participating (Byrne et al., 2020). This sector is highly efficient and profitable and thus contributes to much of the revenues created by Icelandic fisheries (Nielsen et al., 2017). Also, lobster and shrimp fishing have relatively low numbers of participants. Similarly, there are very little landings and vessels participating in shellfish fisheries in Iceland. Given this distribution of catch value and participants in the Icelandic ITQ system, the greatest potential negative impacts of climate change would take place in the demersal sector. While negative climate impacts on benthic groups are likely to only be causing indirect socio-economic effects.

Table 1. Percentage catch value and participating companies and boats in the Icelandic ITQ system in 2015. Atlantic cod has the largest number of unique participants (companies) and a large share of these have boats with either a small boat or hook and line permit.

Species	Catch value of major species %	Number of unique companies	Regular quota boats	Hook and line boats
Atlantic cod	41%	360	254	288
Golden redfish	9%	267	207	230
Capelin (plus roe)	9%	< 23	22	0
Haddock	8%	274	232	248
Atlantic mackerel	7%	-	-	-
Greenland halibut	7%	76	178	52
Saithe	6%	297	225	268
Blue whiting	4%	< 27	26	0
Herring	2%	< 24	23	0
Northern shrimp	2%	< 25	24	0
Lobster	1%	< 11	10	0

2.2. The Atlantis ecosystem model

In this study we used the Atlantis ecosystem model adapted to the trophic relationships in Icelandic waters. The Atlantis model is a whole ecosystem model built on an oceanographic model and includes all major marine functional groups in the Icelandic exclusive economic zone (EEZ). The oceanographic part of the model contains 51 three-dimensional spatial boxes that exchange water flows, salinity levels, and temperature. The oceanographic data were adapted from a hydrodynamic model developed by (Logemann et al., 2013). The ecological model contains 52 functional groups, where vertebrates are generally modeled with a higher level of detail than invertebrates (i.e. some vertebrates are modeled as separate species, while other species with less detailed assessments are modeled as functional groups). In Sturludottir et al. (2018), vertebrate groups have age structure and recruitment was modeled using the Beverton-Holt function, while invertebrates and plankton groups were simple biomass pools. Sturludottir et al. (2018) provide a detailed description of the Icelandic Atlantis model as well as a sensitivity analysis.

2.3 Scenario development

For this study we used the Atlantis model to test the effects of different scenarios, namely OA and warming. First, we repeat the oceanographic time-series (temperature, salinity and water fluxes) nine times for the last ten years of the run period (2002-2012) to create a baseline scenario until the year 2100. We then ran the model for each of the scenarios as described in sections 2.2., 2.3., 2.4. and 2.5, using a parameter set retrieved from the literature (section 2.6.). Fishing mortality and recruitment series were kept constant after 2012.

2.4. Scenario development – warming

Warming in Iceland has been more intense than the global average, a phenomenon known as Arctic Amplification. By 2050, the predicted median warming for the region surrounding Iceland is 1.34–2.10°C depending on the Representative Concentration Pathway (RCP) scenario, and 1.50–4.10°C for the last decade of the 21st century (IPCC, 2014). It is however still uncertain what will happen to air and water temperatures in Iceland due to the unpredictability of natural fluctuations and possible changes in the Atlantic meridional overturning circulation (AMOC) related to anthropogenic forcing; for example, some models and empirical evidence show a slowing down of the AMOC by freshening of the Arctic waters (Sévellec et al., 2017). The AMOC has been shown to be likely slowing down since 2004 and

this has lowered the amount of heat stored in the deeper ocean, causing accelerated warming in the atmosphere and sea surface temperatures (SST) (Chen and Tung, 2018).

Due to the considerable uncertainty regarding the level of warming, as well as the level of CO₂ in the atmosphere by the end of the century, we used three temperature scenarios and two acidification scenarios (based on pCO₂) to project possible changes for the Icelandic marine ecosystem and fisheries. These should not be read as projections/predictions, but more as ways of understanding combined effects of ocean warming and acidification, and how this could possibly impact Icelandic fisheries. The RCP 4.5 scenario is deemed one of the most probable global scenarios due to the current trajectory of anthropogenic emissions (International Energy Agency, 2019), however considerable uncertainty remains regarding the actual level of warming as important global environmental tipping points in the climate system could rapidly increase warming (Lenton et al., 2019). RCP 4.5 suggests a median warming of 2.9°C of the sea for the waters around Iceland (Canada/Greenland/Iceland region) for 2065 and a median level of ocean warming of 3.7 °C by 2100, and slightly lower amounts of warming for the air (IPCC, 2014). We modeled the impact of a temperature increase of 2°C, 3°C and 4°C on the Icelandic marine ecosystem, modeled as a sudden increase in temperature after 2012 (since this was the last year in the oceanographic model of Logemann et al. (2013)). These scenarios were combined with scenarios for OA as described below for a total of 12 scenarios.

2.5. Scenario development – ocean acidification

A drop in pH between -0.25 and -0.3 is expected in the RCP 4.5 scenario (IPCC, 2014). We developed two scenarios for ocean acidification, namely a moderate and a severe acidification scenario. For the moderate scenario we reduced (i) growth of benthic groups by 20%, and (ii) cod recruitment by 20%. For the severe acidification scenario, we reduced (i) growth of benthic groups by 30%, and (ii) cod recruitment by 30%. These are relatively small reductions, as studies have suggested that recruitment may be reduced by 76% to 92% for Arctic cod (Stiasny et al., 2016). The reason why we chose a less extreme estimate is that some cod populations' larval stages are robust to OA (Frommel et al., 2013). Moreover, studies with species with faster generation times show that species can adapt over several generations and that lethal effects are reduced (Thor and Dupont, 2015). Figure 1 shows how combined ocean and acidification could differently impact species in a hypothetical food web. This illustrates some of our scenarios, namely temperature increases with or without the two types of marine group

changes for the moderate OA scenario (6 scenarios) and temperature increases with or without marine group changes for the severe acidification scenario (6 scenarios).

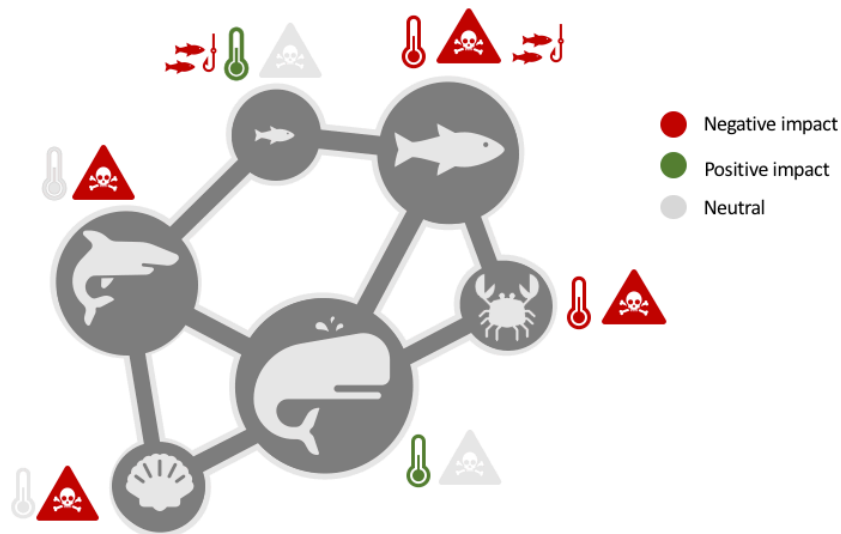


Figure 1. A hypothetical food web with different effects of ocean acidification or warming on species. Species are directly exposed to pressures of ocean acidification (triangle icons), warming (thermometer icons) and fishing (hook icons) but not all species are equally affected by the same drivers and species will likely experience indirect effects due to impacts on prey species or changes in competition and predation.

2.6. Model parameters

Species in focus: We determined which species/functional groups are most important from a socio-economic perspective as well as keystone ecological species/functional groups. We then classified species/functional groups in three categories; *socio-economic* for species that were not found to be keystone ecological species but were important from a socio-economic perspective, *ecological* if species were indicated as keystone ecological species but were not important from a socio-economic perspective, and *socio-ecological* for those species that were indicated as both ecological keystone species and important from a socio-economic perspective. For the selected species/species groups, we conducted a literature review on effects of ocean acidification and ocean temperature warming to obtain a range of values for the parameters necessary for the Atlantis model.

Species in focus, socio-economic indicators: We obtained catch-value and the number of fishery participants for marine stocks in Iceland where they were available. Data on fisheries participants was retrieved from the Directorate of Fisheries (<http://www.fiskistofa.is/>) and data

on the percentage catch value of major species was retrieved from Statistics Iceland (<https://hagstofa.is/>).

Species in focus, network indicators: We assessed ecological importance of a species in the food web by using three network indicators that measure feeding interactions. The Google page rank indicator (designed to rank webpages in order of importance by other websites that link to that website, directly and indirectly (Avrachenkov and Litvak, 2006)) quantifies key species as those where any change in biomass can impact the largest number of other species via both the direct and indirect predator and prey interactions (Allesina and Pascual, 2009). Indegree centrality quantifies the relative importance of a species as by how many predators depend on that species as a prey species (Chen et al., 2008). Finally, we used the centrality measure betweenness, which indicates how many species are directly and indirectly connected through that species or functional group (McDonald-Madden et al., 2016). We ranked species or functional groups from high to low importance and selected those with the 10 highest scores for at least one of the network indicators.

Ocean acidification effects on species: We reviewed the existing literature on ocean acidification at northern latitudes. We searched the literature using the Google scholar search engine and used as search terms: “ocean acidification” and “species name”, or “ocean acidification” and “family name”, or “ocean acidification” and “functional group name”. We also studied the references in the papers we retrieved in the first search, and included those as additional sources if relevant. We retrieved information on magnitude and direction of OA on species population parameters to be used in our modeling exercise (see paragraph 2.5 for how the impact of OA was implemented). For species whose responses to OA were mixed we did not model the impacts of OA on these species or groups (Figure 2, Table S1).

Temperature effects on species: We also searched the literature for evidence of the relationship between temperature and recruitment, and the between temperature and growth. Again, we searched the literature using the Google scholar search engine, using as search terms: “temperature range” or “global warming effects” and “species name”, or “temperature range” or “global warming effects” and “family name”, or “temperature range” or “global warming effects” and “functional group name”. We used the information retrieved from the literature to determine the parameter ranges to parameterize the Atlantis model (Figure 2, Table S1). If we

found temperature optima or niches, these were used to parameterize the temperature optima in Atlantis as was done in (Griffith et al., 2012). If this information was not available, rather than temperature increasing or reducing species growth, temperature is only impacting species through respiration. Although recognized that species are likely already shifting geographical distributions due to ocean warming (Campana et al., 2020; Pinsky et al., 2013), we did not account for spatial shifts in species distributions in this work.

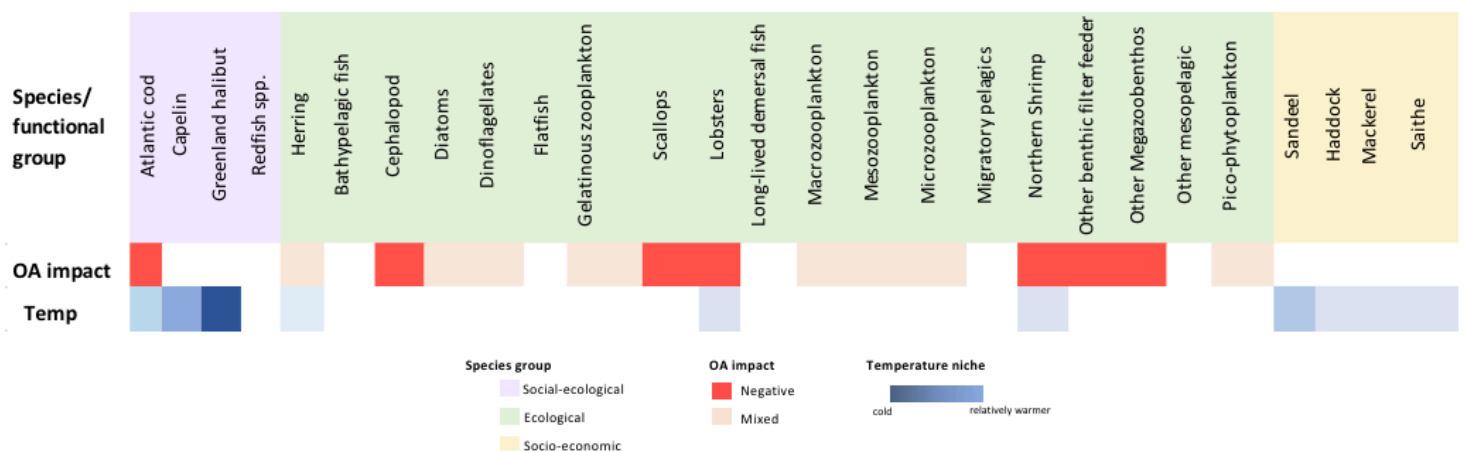


Figure 2. Summary figure of literature review outcomes of impacts of Ocean Acidification and temperature niches for species and functional groups. Boxes that are not colored indicate that not enough information was available to parameterize the model. Species/functional groups are colored by their classification based on ecological and/or socio-economic importance. Detailed descriptions of impacts and temperature niches can be found in Table S1 in the supplementary material.

3. Results

Selecting key-stone species

We found that species at the lower trophic levels had the highest values in almost all three network indicators used for species selection. Several zoo-planktonic groups were indicated as keystone species by the Google page rank indicator (i.e. micro-zooplankton, meso-zooplankton macro-zooplankton and gelatinous zooplankton, Figure 3A). Higher values of this indicator are given to species where any change in biomass could impact the largest number of other species via both the direct and indirect predator and prey interactions. Many other species however, had high betweenness centrality (Figure 3B), but a relatively low Google page rank value. Flatfish had the highest betweenness centrality, which indicates that this functional group is fundamental to the flows between species, i.e. many of the shortest paths in the network go through this functional group. Several other fish species, mesopelagic fish, Greenland halibut (*Reinhardtius hippoglossoides*), Atlantic cod and redbfish, also had relatively high betweenness values (Figure 3B). Several planktonic groups (macro-zooplankton, meso-zooplankton), several benthic groups (Other mega-zooplankton, northern shrimp) and forage fish (i.e. capelin

(*Mallotus villosus*) and sandeel fish showed the highest indegree values, the indicator used to find key prey species in the network (Figure 3C).

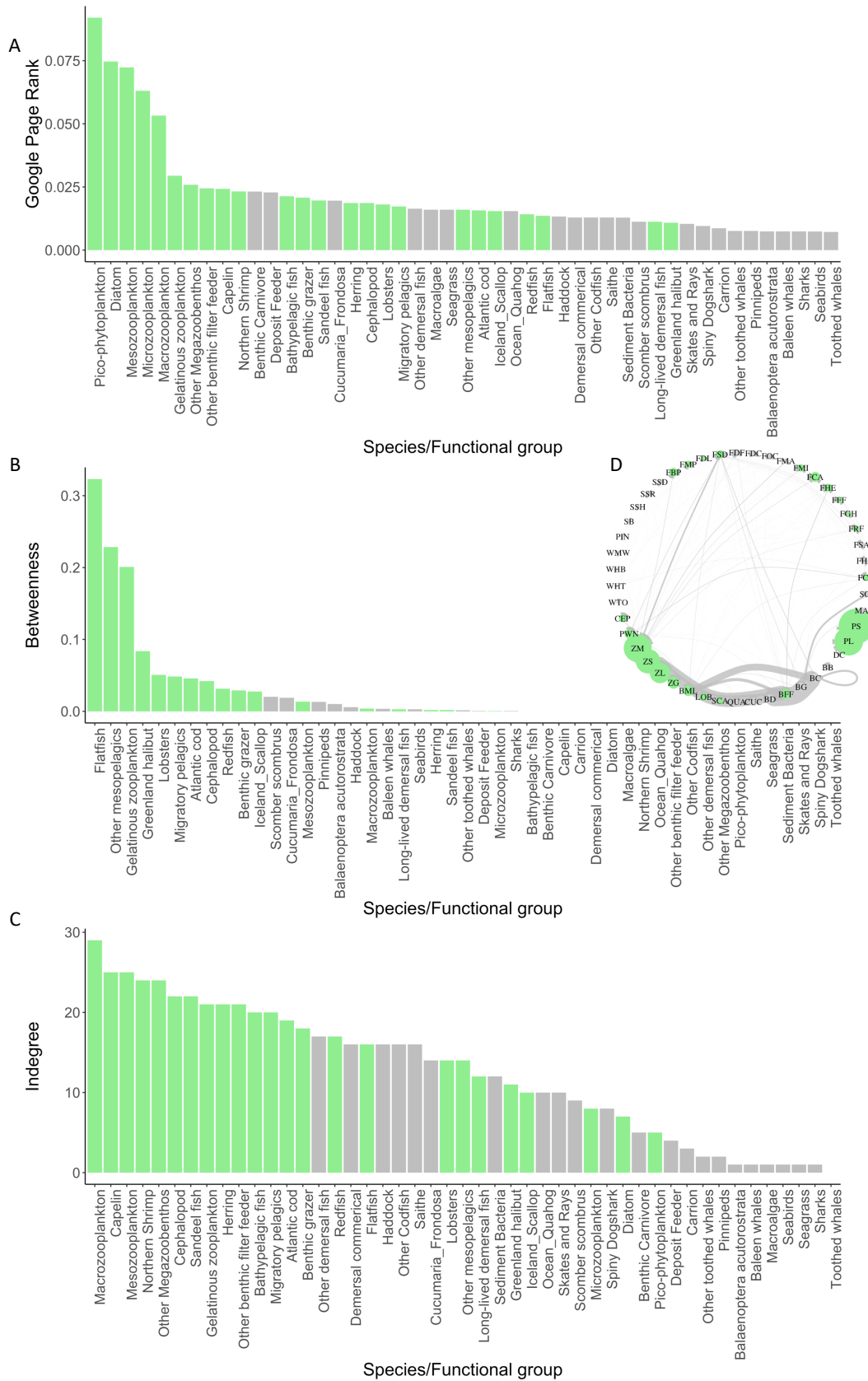


Figure 3. Species /functional groups in Atlantis and network indicator scores indicating importance in the food web. Species/functional groups are displayed from high to low scores left to right; A) Species ranked by Google Page Rank, B) species ranked by Betweenness centrality and C) species ranked by indegree. Light green colored species/functional groups are those that are selected as key-stone species in the food web by at least one of the three indicators. D) The food web in the Atlantis model, nodes are sized by the google page-rank indicator, the Edges (flows between species) are sized by biomass flows (Table S2 contains the species codes used in this figure and the functional groups they represent).

Warming scenarios

Percentage differences between biomass levels in the “baseline” scenario and the warming scenarios became larger with higher temperature increases (Figure 4). We found several positive impacts of warming scenarios on fish biomass, some of high relevance. For instance, herring was projected to increase by almost 100% under 3 °C warming and up to 170% with 4 °C warming, and capelin was also projected to increase under 3 and 4 °C warming, but with broader ranges, by 38% and 226% respectively. The increase in capelin in the warming scenarios is a quite unrealistic result given the low optimal temperature for capelin (between -1 and 7 °C Table S1) was used to parameterize the model. Because of this, we believe this increase in capelin is more likely due to decreases in predation (e.g. by Atlantic cod), rather than directly due to warming. Also, the increase in primary productivity could have impacted capelin biomass, i.e. diatoms and pico-phytoplankton were found to have extreme increases in biomass under all warming scenarios (Table S3, Figure S1). Long-lived demersal fish, mackerel (*Scomber scombrus*), flatfishes and lobsters (*Homarus spp.*) were consistently projected to be negatively impacted in the 3 °C and 4 °C warming scenarios (Figure 4), as well as several zooplankton groups (micro-zooplankton, gelatinous zooplankton and meso-zooplankton).

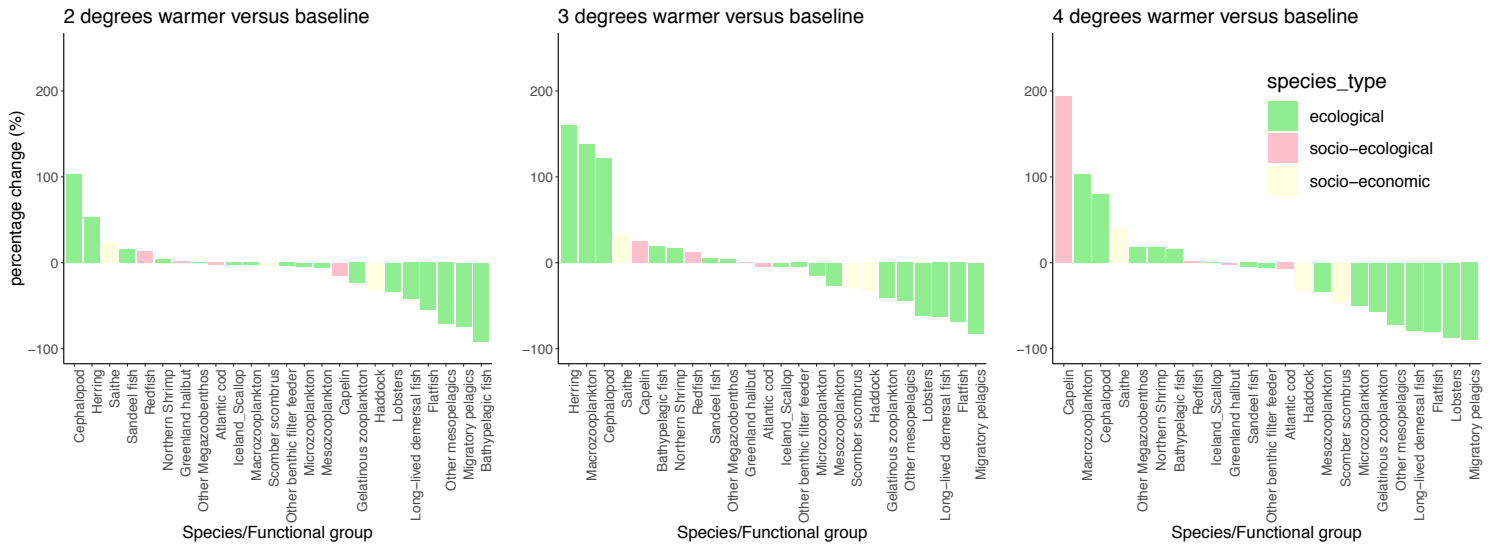


Figure 4. Percentage change in the key functional groups under 3 different scenarios of global warming versus a baseline scenario with no warming. Species or functional groups are colored by the categories established in the literature review (species/groups mainly important for their position in the food web (ecological), species/groups important for both their position in the food web and for Icelandic fisheries (social-ecological), and species that are important for the fisheries but were not indicated as key species in the food web (socio-economic). A few species with changes higher than 250% are shown in Table S3.

The species with highest socio-economic importance, Atlantic cod and haddock, were projected to decrease under warming of 3 and 4 degrees (Figure 5), but the decrease was not very large (-5% and -9% for Atlantic cod and -7% and -8% for Haddock, in the 3 °C and 4 °C warming scenarios respectively).

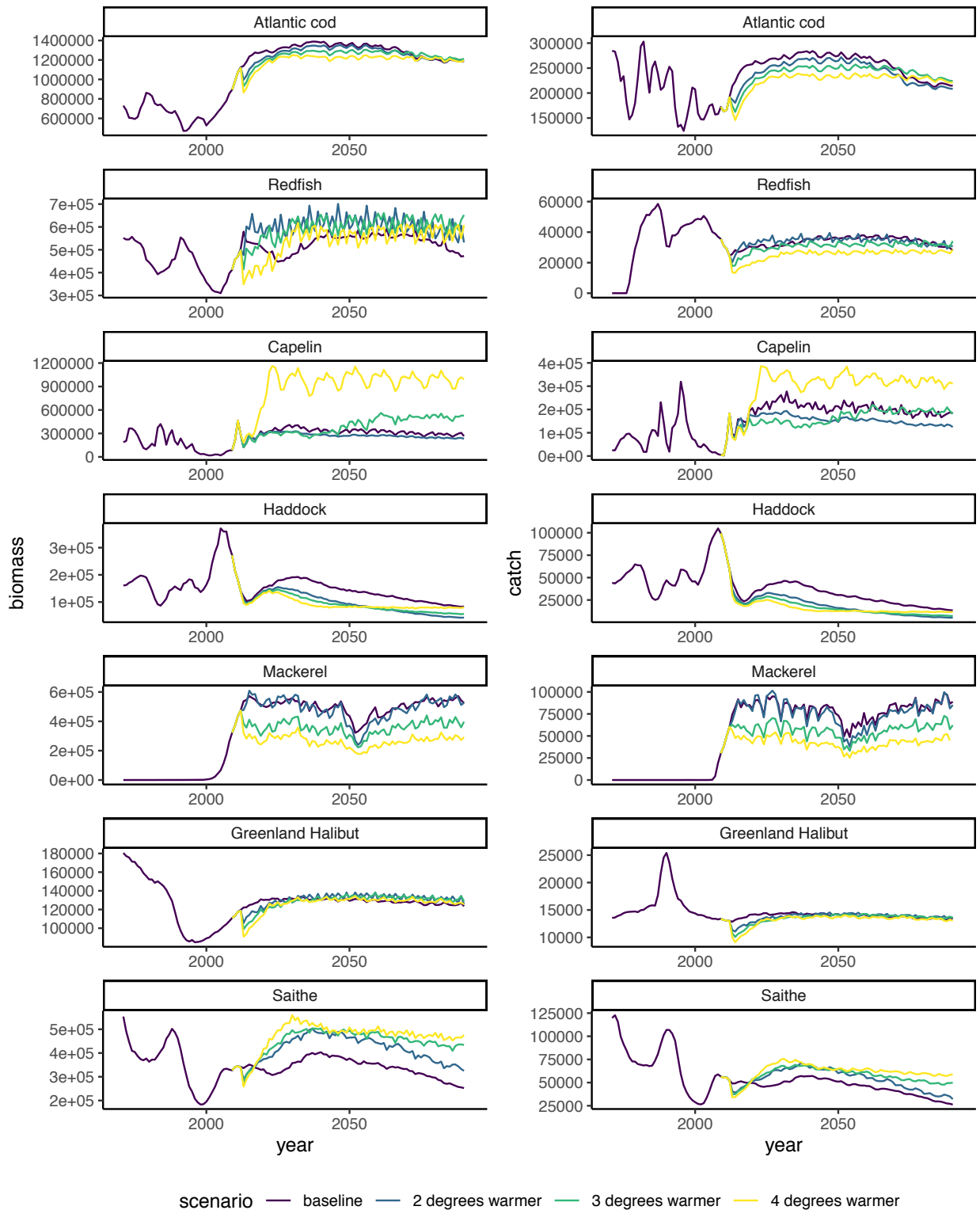


Figure 5. Biomass (left panels) and catch (right panels) of the main Icelandic fisheries (i.e. those with 5% or more of Icelandic catch-value as described in Table 1) under the baseline scenario and three different global change scenarios. Note that the y-axes are on very different scales.

Combined effects of warming and Ocean Acidification on marine ecosystem functioning

We find that with combined warming and ocean acidification changes in biomass in the key-species/functional groups were very similar to the scenarios where only the impact of 2 °C warming was studied (Figure 4 and Figure 6), and in general changes in biomass due to OA impacts only were much smaller. This suggests that the model is much more sensitive to changes in temperature than the changes in recruitment and growth implemented for the OA scenarios.

Starting with OA alone, we find the largest reduction in diatom biomass, although diatom biomass showed very large fluctuations over time (Figure S1). Atlantic cod showed the second largest reduction in biomass under both OA scenarios, with on average a -6% and -11% reduction in biomass (Figure 6). Macro-zooplankton, flatfish and herring biomass, on the other hand, increased between 6% and 18% due to the changes in benthic growth and Atlantic cod recruitment (Figure 6).

The combined scenarios were most impactful for Atlantic cod biomass, which decreased in all scenarios and exhibited the largest decrease (-13% on average over 2013 to 2100) with 4°C warming and a reduction in cod recruitment of 30% (Figure 6). Flatfish, migratory pelagics, other meso-pelagics, lobster and long-lived demersal fish were also very strongly affected, with the largest decreases found for lobster (-90%) and migratory pelagics (-91%) under 4°C of warming and 30% decrease in growth/recruitment of certain species/functional groups. On the other hand, herring biomass was projected to increase in all scenarios, with the largest increase (190%) with 4°C warming and 30% reduction in growth of invertebrates and cod recruitment. Capelin biomass increased the most with 4°C warming and 20% reduction in cod recruitment and benthic growth rates (203% on average over 2013 to 2100). Further, in some cases warming and OA had antagonistic effects. For example, cephalopod biomass increased with warming alone, and this increase was smaller when OA was added. Macro- and micro-zooplankton biomass was also projected to increase with 2 °C warming and acidification, but decrease under 2°C warming without OA.

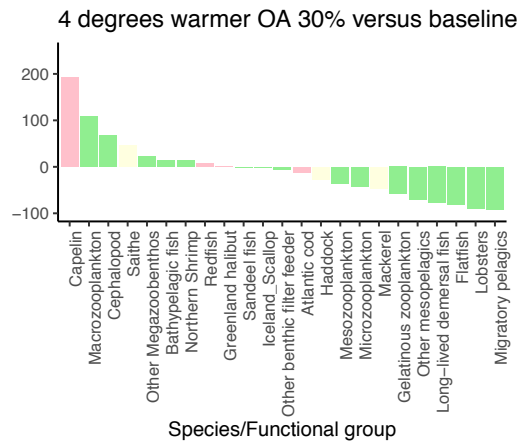
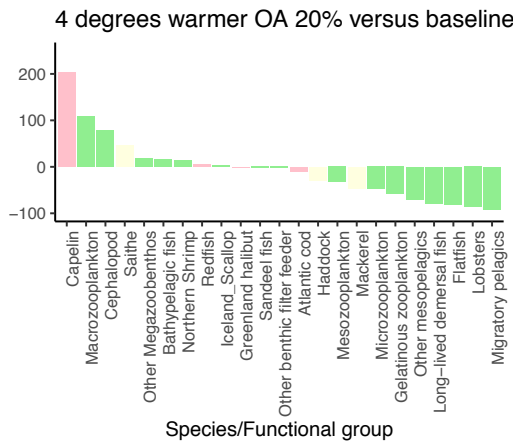
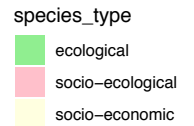
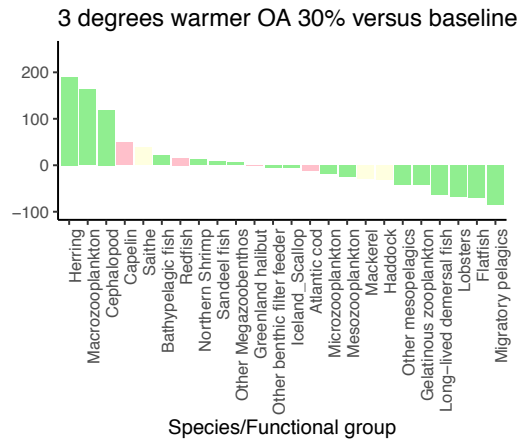
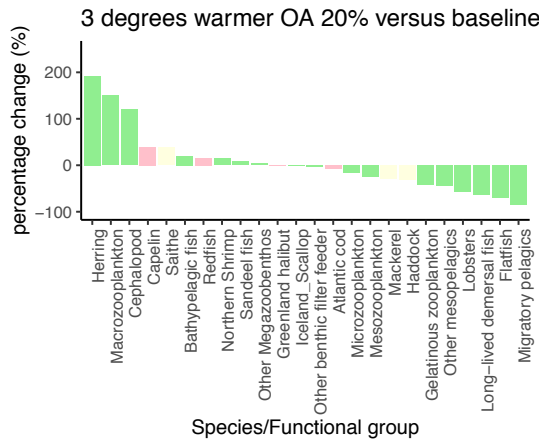
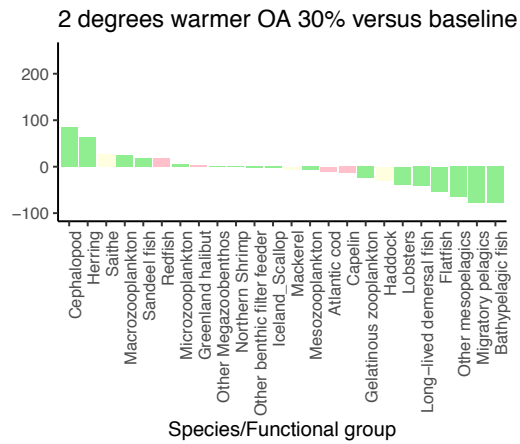
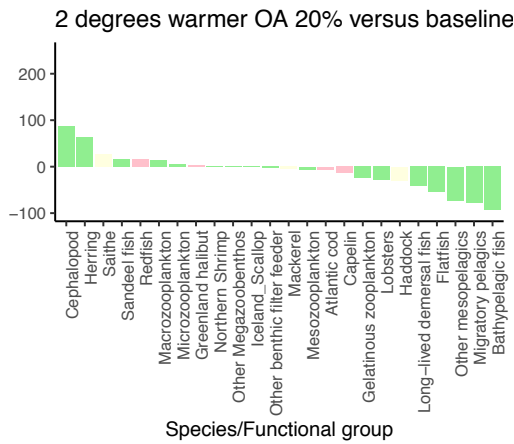
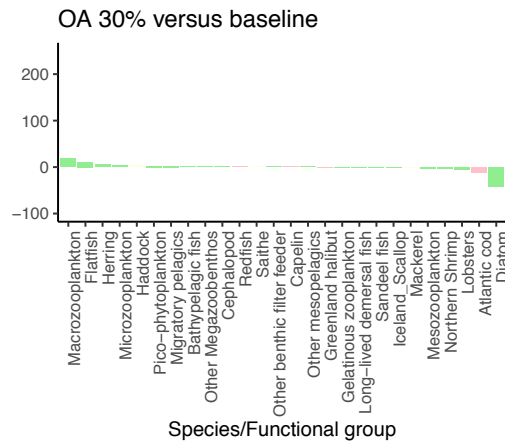
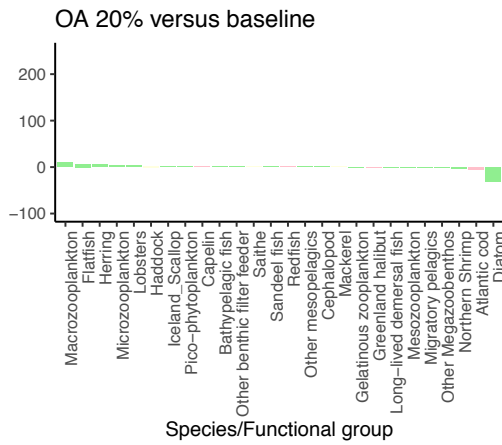


Figure 6. Percentage change in the key functional groups under 8 different scenarios of global change (ocean acidification with a 20% or 30% reduction in growth or recruitment and different warming scenarios (2, 3, and 4 °C of warming)) versus a baseline scenario with no warming and no acidification. Species or functional groups are colored by the categories established in the literature review (species/groups mainly important for their position in the food web (ecological), species/groups important for both their position in the food web and for Icelandic fisheries (social-ecological), and species that are important for the fisheries but were not indicated as key species in the food web. A few species with changes higher than 250% are shown in Table S3.

Although the biomass of the Atlantic cod stock was surely impacted by the forced reduction in recruitment, biomass levels seemed to stabilize in all scenarios at levels higher than present, while in the baseline scenario biomass increased further (Figure 7). The model did not forecast a collapse or a strong decrease in biomass compared to current levels, biomass of Atlantic cod has been increasing steadily since the reduction of harvest rates in the early 2000's. When this harvest rate is kept constant, the Atlantis model predicted a rather stable biomass and catches, but reduced biomass of haddock compared to the present (Figure 7). Also, mackerel, a newly important species in terms of catch-value was projected to have lower biomass and catch levels under scenarios of increased warming and acidification, while saithe and redfish were projected to have increased biomass levels (Figure 7).

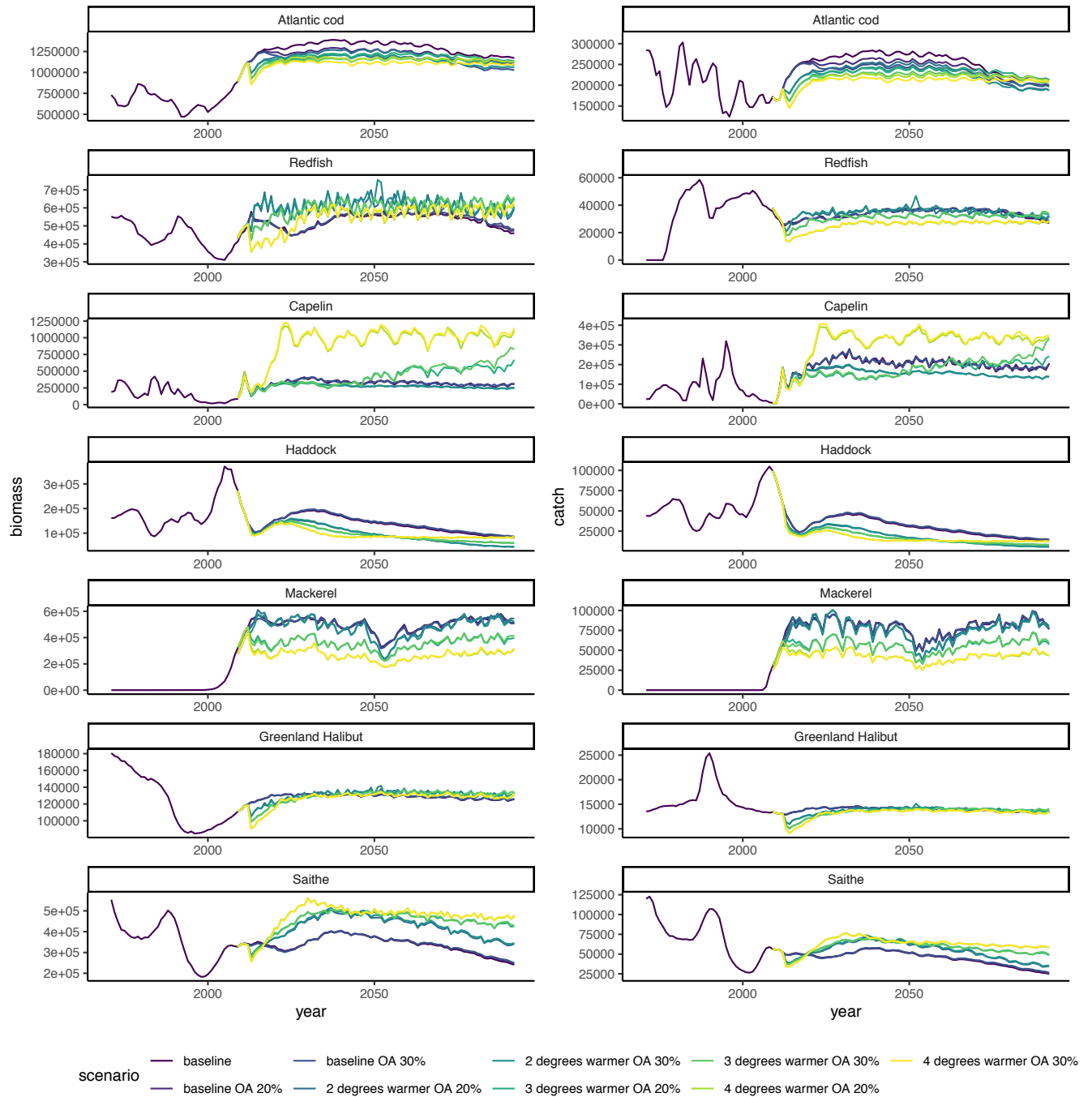


Figure 7. Biomass (left panels) and catches (right panels) of the main Icelandic fisheries (5% or more of Icelandic catch-value in Table 1, organized from highest to lower catch values) in Atlantis under the baseline scenario and eight different global change scenarios. Note that the y-axes are on very different scales.

4. Discussion

We set up to model the responses of marine food webs to ocean acidification and rising ocean temperatures using the Atlantis whole ecosystem model adapted to the Icelandic system. Our results showed an expected reshaping of the Icelandic marine food web under different scenarios of global change, namely ocean acidification and warming. Overall, lower trophic

levels such as planktonic groups benefited more from warming scenarios than higher trophic levels. Shifts in important predator species such as Atlantic cod had important implications for the species that they feed on, e.g. the increase in capelin under a decrease of Atlantic cod biomass and an increase in primary productivity. Overall, the Icelandic Atlantis model predicted increases in certain species groups under warming and acidification and decreases in others, and the increases tended to be larger in functional groups of lower trophic levels (first producers and first consumers).

We found that the Icelandic implementation of the Atlantis model is more sensitive to changes in temperature than those of OA. Modeling impacts of OA by reducing growth of benthic groups is a common approach with the Atlantis model (Marshall et al., 2017; Olsen et al., 2018), but since many uncertainties remain regarding the impacts of OA and possible cascading effects through the food web, the relatively small impact found in our results should be interpreted with caution. Important indirect effects have been found previously when studying OA impacts in the Californian current, for instance by decreased biomass levels of groundfish feeding on benthic groups affected by OA (Marshall et al., 2017). Olsen et al. (2018) found predominantly negative effects from OA across a suite of Atlantis models representing 8 different ecosystems, but similar to our findings, the authors also found positive effects for instance through reduced competition for benthic groups that were not modeled to be directly affected by OA (e.g., amphipods, isopods). There is still much uncertainty regarding ocean acidification impacts on fish stocks and in many published experiments the impacts of recruitment have not been measured by rearing adults in more acid conditions (Frommel et al., 2014; Stiasny et al., 2016).

One of the arguably most important species in the food web, Atlantic cod, showed some surprising modeling results. While Atlantic cod biomass was projected to be lower under the most severe scenario modeled (a 4°C warming scenario and a 30% reduction in Atlantic cod recruitment due to acidification), biomass levels of Atlantic cod were still projected to be higher than those today under the current conservative harvest rate. Atlantic cod in Icelandic water is not as close to the upper end of its thermal niche as, for instance, the Arctic cod populations in Norwegian waters (Drinkwater, 2005; Hänsel et al., 2020). Modeling combined warming and acidification impacts on Arctic cod in the Barents Sea predicted a severe decline in recruitment (Koenigstein et al., 2018) and the risk of the collapse of the commercial fishery

(Hänsel et al., 2020), but the modeled effects of acidification were also more severe (i.e. reduced recruitment to 24.5% of current levels; Hänsel et al., 2020). Atlantic cod is fished by almost all companies in Iceland and since two stocks of the same species in Norway and the western Baltic are projected to have reduced recruitment under OA (Stiasny et al., 2016) experimental studies on the Icelandic stocks are needed. Atlantic cod is also likely to be a main choke species in the Icelandic demersal fishery, any biomass changes in cod will thus very likely also impact the fishing of other demersal populations (Oostdijk et al., 2020).

There were four zooplankton groups among the selected important ecological functional groups in the Atlantis model. The fact that impacts of OA resulted in mixed responses in lab experiments and mesocosm experiments combined with the importance of these species in the food web stresses the need for more research efforts. Research also needs to focus on reproduction of the same experiments as experimental conditions could result in different outcomes of studies. OA effects on zooplankton are not conclusive yet, while severe negative effects were found for krill (Cooper et al., 2016; McLaskey et al., 2016) and copepods (Thor and Dupont, 2015), which are both main components of Icelandic zooplankton. Strong effects on krill and copepods as have been found in some of the studies, which may decrease ecosystem-level productivity and thus negatively impact fisheries, but this is still largely uncertain as other studies find weaker or no significant effects on plankton communities (Falkenberg et al., 2018). A recent series of mesocosm experiments, however, found that mainly functional groups at an intermediate trophic level (first consumers, i.e. ascidians, sponges and copepods) reduced under combined warming and acidification (Nagelkerken et al., 2020) while species at the lowest trophic levels (first producers, i.e. phytoplankton, algae) and secondary consumers increased in biomass. We found that the functional group macrozooplankton increased under scenarios of warming and warming and OA and that mesozooplankton and gelatinous zooplankton decreased under the same scenarios, however we did not consider direct impacts of OA on those groups. In future study it would thus be important to model the impacts of combined warming and acidification on zooplankton more realistically. It is important to note that in general, the network indicators that we used to select key ecological groups were biased towards lower trophic level groups, and important predator species that impact the food web by top-down control may have been missed in our selection process. If we would have used the Google page rank indicator alone for species selection as suggested in McDonald-Madden et al. (2016) the emphasis on lower trophic web species would

have been even stronger, suggesting that a combination of indicators may work better to select important species in the food web.

At this point in time we have very little information on the response of Icelandic species and stocks. Moreover, the Icelandic Atlantis model was set up with much more data on large commercial stocks. In this sense the Icelandic Atlantis model is not very different from other Atlantis models (Kaplan et al., 2012; Marshall et al., 2017), as simply more information is available on those species and benthic organisms are usually modeled as biomass pools (Marshall et al., 2017), and the modeling of OA impacts is therefore only impacting growth while in reality different aspects of a species life history can be affected (e.g. the larval stage, or growth in adult stages due to differences in calcification rates etc.). Another major limitation of our modeling approach is that we did not account for spatial shifts in species distributions due to increases in temperature, which is expected (to cause and already is causing) major re-distributions of fish biomass (Fulton, 2011). However, models for larger areas are probably better suited for this approach as the Atlantis model only includes the areas around Iceland and cannot consider new introductions of species due to range-shifts in an endogenous fashion. Including species movements because of temperature would then only result in species losses but not in gains which is unlikely given the current poleward shift of several species-distributions (Campana et al., 2020).

5. Conclusion

We found moderate effects on catch-levels of the main commercially important species in the Icelandic marine food web, under different scenarios of warming and acidification. However, large uncertainties remain regarding the sensitivities of species for decreasing pH levels (i.e. this has not been empirically studied for species in the Icelandic marine ecosystem), warming and their combined effects. Since zooplankton groups were indicated as key-stone ecological groups in the food web more experimental research on combined warming and acidification on these groups is needed for the Icelandic marine ecosystem to gauge possible cascading effects on the ecosystem. We do not know the full extent of threats that climate change poses to fisheries, but combined OA and warming will re-shape ecosystems and it is important that both economic and social implications will be investigated. We know that poorer fishers will likely be hit harder by climate change than bigger companies, which can more easily adapt and switch fishery for instance (Fulton, 2011) or access quota markets (Oostdijk et al., 2019). Moreover,

compared to the large-boat fleet, crew and captains on the smaller boats earn about half as much and are thus possibly more vulnerable if climate change effects turn out to be negative (Nielsen et al, 2017). It is an open question as to how do governments will deal with ecosystem shifts and if equitable outcomes will be considered when determining who will benefit from the new opportunities to fish.

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Supplementary information

Temperature niche and sensitivity for ocean acidification – model parameterization

Optimal temperatures for the species and functional groups for which this information was available differed widely and were also different for juvenile versus adult life stages (Table S1). For many species optimal temperature ranges are at the high end compared to the average temperature of Icelandic waters, which ranges from approximately 6-9°C at a depth of 70 meters and is lower at lower bottom depths. However, species such as capelin and sandeel have lower temperature preferences and occur at lower depths, suggesting that these species may be more vulnerable to future warming. Important species or functional groups which are very likely to be negatively impacted by ocean acidification are: Atlantic cod, cephalopods, benthic filter feeders (containing echinoderms and brittle stars), lobsters, Northern shrimp and other megazoobenthos (Table S1).

Table S1 Key species and functional groups in the Icelandic marine ecosystem and ocean acidification and warming effects. Pink cells show species that are both important because of socio-economics and were found to be key-stone species by at least one of the three network metrics. The green cells show species that were found to be key-stone species by at least one of the three network metrics, but do not contribute directly in the sense of many fisheries participants or significant catch-value. Yellow cells indicate species that were not found to be important ecological species by any of the three network indicators used but are important regarding numbers of participants and catch value (Table 1).

Species/functional group	OA direction effect (effect)	Optimal temperature?
Atlantic cod (<i>Gadus morhua</i>)	Negative on recruitment (doubling of daily larval mortality rates & 24% decrease in recruitment under ~1100 μ atm) (Frommel et al., 2011; Stiasny et al., 2019, 2016)	<i>Growth</i> : 10-15 °C, higher temperature for juveniles as for adult fish, no Atlantic cod stocks at bottom temp > 12°C <i>Recruitment</i> : non-linear, at bottom temperatures increases with bottom temp. below 5°C, recruitment decreases above bottom temperatures of 8.5°C (between those values a plateau) Drinkwater, 2005; Benjamin and Fredou, 1999 Thermal niche: -1.5 – 19 °C
Capelin (<i>Mallotus villosus</i>)	NA	Icelandic stock: feeding in waters -0.5-6 °C, spawning 2-7 °C (Rose, 2005), typically found in waters -1 to 6°C. Large shifts in distribution with changes in temperature.
Greenland halibut (<i>Reinhardtius hippoglossoides</i>)	Of the family: <i>Hippoglossus hippoglossus</i> negative on growth (Gräns et al., 2014)	Increased growth but increased density dependent mortality found for juveniles (Sünksen et al., 2010)– may cancel each other out Temperature preference 1.3 to 2.7°C (occur usually at great depth, mean depth 1048 (+/- 112 m). (Peklova et al., 2012)
Redfish spp.	NA	NA
Herring (<i>Clupea harengus</i>)	Mixed (organ damage larvae but neutral effect in mesocosm due to increase primary productivity) (Frommel et al., 2014; Sswat et al., 2018)	Thermal nice 8-12 °C (Macdonald, 2019) (based on only 5 adult individuals in the waters around Iceland) (Macdonald, 2019). For Baltic herring: viable hatch highest at 7-13 °C (Peck et al., 2012)
Bathypelagic fish	NA	Warming is much slower in deeper water, but the species living in the deep sea may be especially vulnerable as they are adapted to stable climates (Levin et al., 2019; Levin and Le Bris, 2015). But similarly, the amount warming in the deep sea is still uncertain due to uncertainties regarding thermal mixing and currents, we thus chose not to explicitly model warming effects for Bathypelagic fish.

Species/functional group	OA direction effect (effect)	Optimal temperature?
Cephalopod	Negative sub-lethal effects on energy budget, reduced size (Falkenberg et al., 2018; Spady et al., 2014; Kaplan et al., 2013; Sigwart et al., 2016)	Shifts in species occurring (Golikov et al., 2013), but functional group is too broad to model such detailed community changes.
Diatoms	Mixed (Falkenberg et al., 2018; Rossoll et al., 2012)	Possible shifts in species composition (e.g. larger species found with warming in study by Sett et al., (2018) as well as (Boyd, 2019) for southern polar ocean, many different optimal temperatures for different species, modulated by iron deficiency. – - not modeled
Dinoflagellates	Mixed some species may benefit some species may have decreased calcification rates (Eberlein et al., 2014; Falkenberg et al., 2018; Van de Waal et al., 2013)	As arctic and near-arctic waters get warmer the environment becomes more suitable for different species (Okolodkov and Dodge, 1996). Decline in Dinoflagellates abundance with warming SST and increased windy conditions in summer in the Southern Ocean (Hinder et al., 2012). – not modeled.
Flatfish	No species in Icelandic EEZ studied, Negative for flatfish larvae studied, e.g. Pimentel et al. (2015) studied combined warming and acidification. Skeletal abnormalities, albeit at very high CO ₂ concentrations (pCO ₂ ~1600µatm). No estimate of impact on recruitment given.	European plaice (<i>Pleuronectes platessa</i>) juveniles: 3–18°C; eggs: 5-7°C; settlers 3-6 °C (Petitgas et al., 2013) No other species that occur in Icelandic EEZ individually studied. The default option for temperature curves in Atlantis is used.
Gelatinous zooplankton	Mixed (Falkenberg et al., 2018; Niehoff et al., 2013)	NA
Scallops	Negative (Falkenberg et al., 2018; Goethel et al., 2017; Iglukowska et al., 2017; Schalkhausser et al., 2013)	NA
Lobsters	Negative, morphological changes in long-term study (Agnalt et al., 2013; Falkenberg et al., 2018; Small et al., 2016)	Effect on fecundity probably positive, with annual instead of biannual spawning for Norway lobster (at average temperature of 11 °C) (Eiriksson, 2014).
Long-lived demersal fish	NA	NA
Macrozooplankton	Mixed (Falkenberg et al., 2018; Venello et al., 2018; Yang et al., 2018)	NA
Mesozooplankton	Mixed (Falkenberg et al., 2018; Niehoff et al., 2013)	NA
Microzooplankton	Mixed (Falkenberg et al., 2018)	NA
Migratory pelagics	NA	NA
Northern Shrimp	Negative (Chemel et al., 2020; Dupont et al., 2014; Falkenberg et al., 2018) but relatively small effect for OA found in (Arnberg et al., 2013)	Optimal temperature around 9 °C for population in colder waters (Ouellet et al., 2017).

Species/functional group	OA direction effect (effect)	Optimal temperature?
Other benthic filter feeder	Negative (Dell'Acqua et al., 2019; Falkenberg et al., 2018)	NA
Other Megazoobenthos	Negative (Falkenberg et al., 2018; Hale et al., 2011)	NA
Other mesopelagics	NA	NA
Pico-phytoplankton	Mixed	NA
Sandeel	NA	Warming sea temperatures in the North Sea have been linked to declines in sandeel recruitment (Arnott & Ruxton 2002) and long-term changes in sandeel sizes (Frederiksen <i>et al.</i> 2011). Decline in Icelandic sandeel population (not caused by fishing) and probably a poleward shift (Vigfusdottir et al., 2013). Thus, optimal temperature is probably situated around an average 7 °C SST.
Haddock (<i>Melanogrammus aeglefinus</i>)	NA	Juveniles: “The growth rate and scope for growth were maximum at 12°C, whereas growth efficiency was greatest (26.0 to 32.2%) at temperatures between 5 and 8°C” (Peck et al., 2003).
Mackerel (<i>Scomber scombrus</i>)	NA	Most suitable temperature for egg development ~ range from 12 to 13°C (SST) (Ibaibarriaga et al., 2007). Embryo's range 11-21°C, center 16 °C (Magnusson and Destasio, 1997)
Saithe (<i>Pollachius virens</i>)	NA	Eggs: temperatures ranging from 2-17°C, most at temperatures of 5-11°C (depth 50-250m) Larvae: 3-9°C (depth: usually shore to 200m) Juveniles: most 3-12°C Adults: avoid temperatures < 3°C and > 11°C (National Oceanic and Atmospheric administration, 1999)

Table S2: Species /functional group used in the Icelandic Atlantis model codes and species/functional groups belonging to that code.

code	Species/ functional group
FCD	Cod (<i>Gadus morhua</i>)
FHA	Haddock (<i>Melanogrammus aeglefinus</i>)
FSA	Saithe (<i>Pollachius virens</i>)
FRF	Redfish (<i>Sebastes sp</i>)
FGH	Greenland halibut (<i>Reinhardtius hippoglossoides</i>)
FFF	Flatfish
FHE	Herring (<i>Clupea harengus</i>)
FCA	Capelin (<i>Mallotus villosus</i>)
FMI	Blue whiting (<i>Micromesistius poutassou</i>)
FMA	Mackerel (<i>Scomber scombrus</i>)
FOC	Other codfish
FDC	Demersal commercial
FDF	Other demersal fish
FSD	Sandeel fish
FDL	Long lived demersal
FMP	Large pelagic fish
FBP	Small pelagic fish
SSR	Skates
SSD	Small sharks
SSH	Large sharks
SB	Seabird
PIN	Pinniped
WMW	Minke whale (<i>Balaenoptera acutorostrata</i>)
WHB	Baleen whale
WHT	Tooth whale
WTO	Other tooth whale
CEP	Cephalopod
PWN	Shrimp
ZS	Microzooplankton
ZM	Mesozooplankton
ZL	Macrozooplankton
ZG	Gelatinous zooplankton
LOB	Norway lobster
BML	Other megazoobenthos
SCA	Iceland scallop
QUA	Ocean quahog
CUC	Cucumbers
BD	Deposit feeder
BFF	Other benthic filter feeders
BG	Benthic grazer

code	Species/ functional group
BC	Benthic carnivore
BO	Meiobenthos
PL	Diatom
PS	Pico-phytoplankton
MA	Macroalgae
SG	Seagrass
DF	Dinoflagellates
PB	Pelagic bacteria
BB	Sediment bacteria
DL	Labile detritus
DR	Refractory detritus
DC	Carrion

Table S3: Functional groups that increased by more than 250% in global change scenarios compared to the baseline scenario.

scenario	Functional group	Percentage change
2 degrees warmer	Diatom	15231%
2 degrees warmer	Pico-phytoplankton	41629%
3 degrees warmer	Diatom	91457%
3 degrees warmer	Pico-phytoplankton	88854%
4 degrees warmer	Herring	269%
4 degrees warmer	Diatom	135070%
4 degrees warmer	Pico-phytoplankton	78231%
2 degrees warmer OA 20%	Diatom	13980%
2 degrees warmer OA 20%	Pico-phytoplankton	38617%
2 degrees warmer OA 30%	Diatom	14864%
2 degrees warmer OA 30%	Pico-phytoplankton	38022%
3 degrees warmer OA 20%	Diatom	88953%
3 degrees warmer OA 20%	Pico-phytoplankton	91575%
3 degrees warmer OA 30%	Diatom	86160%
3 degrees warmer OA 30%	Pico-phytoplankton	91681%
4 degrees warmer OA 20%	Herring	319%
4 degrees warmer OA 20%	Diatom	134363%
4 degrees warmer OA 20%	Pico-phytoplankton	79710%
4 degrees warmer OA 30%	Herring	302%
4 degrees warmer OA 30%	Diatom	119016%
4 degrees warmer OA 30%	Pico-phytoplankton	73015%

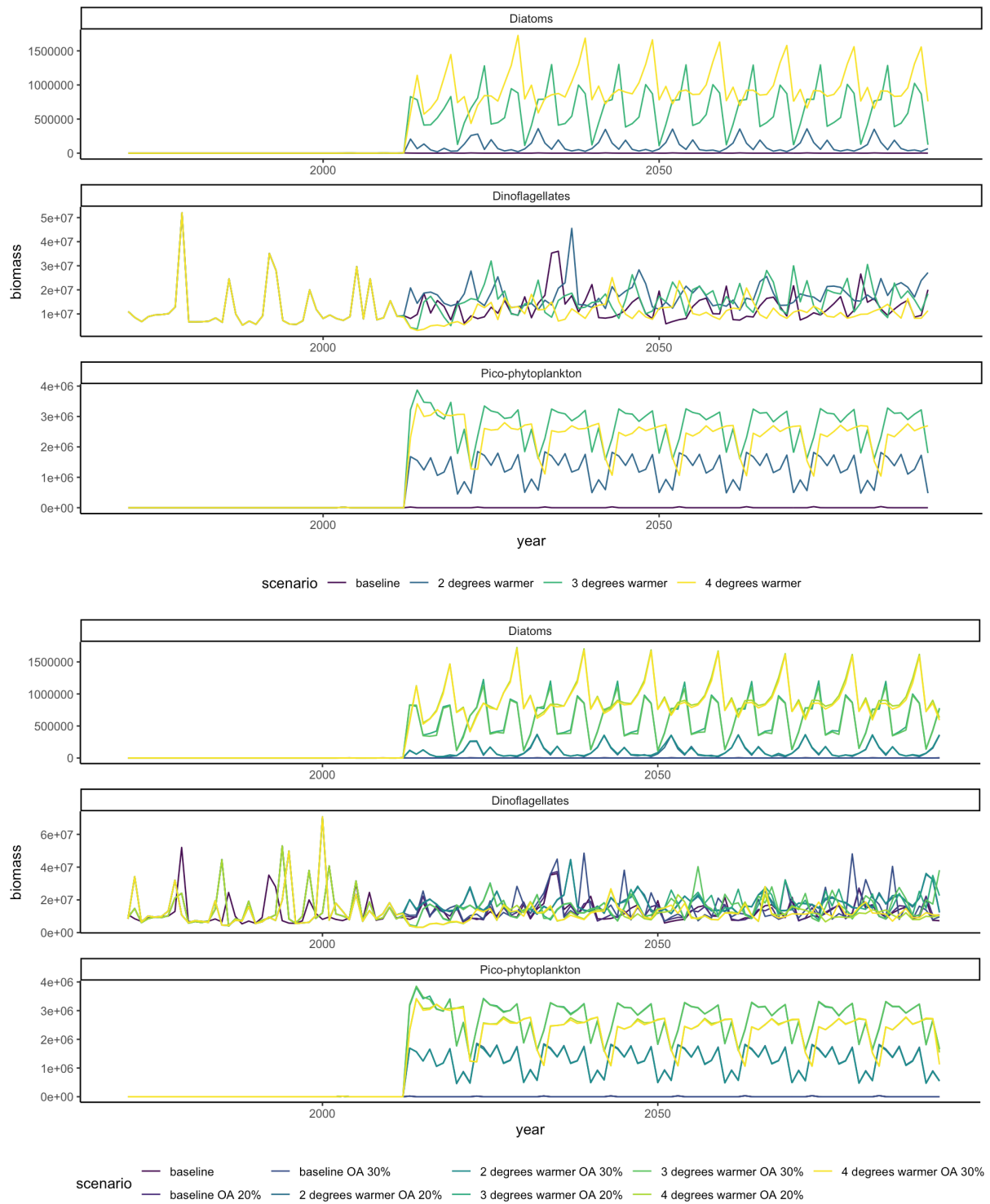


Figure S1: Phytoplanktonic groups in Atlantis under the different warming scenarios (upper panel) and different OA and OA combined with warming scenarios (lower panel).

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