

Modeling the biodiversity crisis: the roles of space and market structure in bioeconomic modeling.

Modéliser la crise de la biodiversité : les rôles de l'espace et de la structure de marché dans la modélisation bioéconomique

Thèse de doctorat de l'Université Paris-Saclay

École doctorale n°512 Agriculture, Alimentation, Biologie, Environnement et Santé (ABIES)
Spécialité de doctorat: Sciences Economiques

Graduate school: Biosphera

Thèse préparée à l'AgroParisTech, sous la direction de Lauriane MOUYSSET (Chargée de Recherche CNRS (CIRED)), le co-encadrement de Christopher COSTELLO (Professor of Resource Economics, University of California, Santa Barbara (emLab))

Thèse présentée et soutenue à Nogent sur Marne, le 26 novembre 2024, par

Simon Jean

Composition du Jury

Name LASTNAME Statut, Établissement (Unité de recherche)	Président
Name LASTNAME Grade, University/Lab (research unit)	Rapporteur
Name LASTNAME Grade, University/Lab (research unit)	Rapporteur
Name LASTNAME Grade, University/Lab (research unit)	Examinateuse
Name LASTNAME Grade, University/Lab (research unit)	Examinateuse

Direction de la thèse

Lauriane MOUYSSET Chargée de Recherche CNRS (CIRED)	Directrice de thèse
Christopher COSTELLO Professor of Resource Economics, University of California, Santa Barbara (emLab)	Co-directeur de thèse

Dedication

Acknowledgements

- Encadrement : Lauriane, Chris
- CIRED :
 - Jeunes : Romain, Thibault, Lucas, Baptiste, Valentin, Auriane, Mai Thi, Charles x2, PY le petit Ange, Luc, Louis, Soline, Morgane, Clément, Bertille
 - Titus : Romain, Aurélie, Tarik, Philippe, Antoine, Laurent
 - Support: Estelle, Carine, Valérie
- Emlab :
 - Jeunes : Costello Lab, Jacob, Vincent, Julia, Roberto, Flavio, Dario, Camilo
 - Life : Seth and Ferdi, Ian
 - Work: Tracey, Heather, Lauren, Sarah, Danielle, BreakTime Crew
- Mes amis, mon équipe : Léo, Jeanne, Minha, Clara, Juliette, LV, Joco, Cam, Charlotte, Thomas
- Mes autres sangs: Augustin, Louis, Aymeric, Clara à New York, Eloise
- Mon père et ma mère, mes frères et soeurs, David, Kate, Alice et Mila pour leur accueil à NY aussi, Mambo et Julie pour m'avoir fait une maison en Californie, mes grands parents, mon cousin Paul pour m'avoir martyrisé, la GymSu en entier, Julie à Louvain;
- Merci du fond du coeur, en entier, pour tout et plus, pour une thèse qui pris 3 ans, deux continents, tant d'amis, de bières, de bêtises et de fêtes.

Contents

1 Bioeconomic models for terrestrial social ecological system management : a review	3
2 Little downside and substantial gains result from farming of <i>Totoaba Macdonaldi</i>	5
2.1 Introduction	6
2.2 Methods	8
2.2.1 The Poaching Model	9
2.2.2 The Farming Model	11
2.2.3 Demand	11
2.3 Results	11
2.3.1 Totoaba stock under monopoly is sensitive to cost structure	11
2.3.2 Farming produces conservation benefits	12
2.3.3 An effective policy space for farming.	14
2.4 Conclusion	16
2.5 Contributions and data availability	17
2.A A theoretical model of poachers, traders, and farmers	18
2.A.1 Entry in the fishery and poaching supply	18
2.A.2 Traders as vertical monopolists, without farming	19
2.A.3 Captive breeding, imperfect competition and conservation	19
2.A.4 Extensions	23
2.A.5 Appendices	26
3 The wildland connectivity dilemma : a graph theoretical computational approach	37
3.1 Introduction	39
3.2 Methods	40
3.2.1 Theoretical model	40
3.2.2 Landscape indicators	42
3.2.3 Computational experiments	43
3.3 Results	43
3.3.1 Steady states	43
3.3.2 Wildfire risk reduction and habitat connectivity in steady state landscapes .	44
3.3.3 Properties of steady state landscapes: surface, fragmentation, and diversity	44
3.3.4 Spatial allocation of optimal management at the steady-state landscape cycle	45
3.4 Discussion	45
3.4.1 Confirmation and generalization of existing results	45
3.4.2 Caveats and methodological perspectives	46
3.4.3 Conclusion and policy relevance	47
3.5 Declaration	48
3.5.1 Acknowledgments	48
3.5.2 Data availability	48
3.5.3 Author affiliation	48

3.5.4	Competing interests	48
3.5.5	Contribution	48
3.A	Appendix	49
A	Theoretical model	49
B	Landscape indicators	50
3.B	Figures	52
4	Fences	59

List of Figures

2.1	Evolution of totoaba population and catch over time	7
2.2	Schematic of equilibrium points under different poaching harvest functions	8
2.3	Schematic of monopoly and duopoly market structures	9
2.4	Equilibrium points for wild totoaba stock under different market structures with (left) a linear quadratic cost structure, and (right) a quadratic cost structure.	12
2.5	Sensitivity of equilibrium points to cost structure for wild totoaba stock	13
2.6	Economic and ecological performance of different market regimes	14
2.7	Interaction between substitutability and demand under duopolistic competition	15
2.A.1	Steady state outcomes when multiple traders and multiple farmers are considered (an oligopoly) in the quantity adjustment scenario.	25
2.A.2	Steady-state equilibrium for the wild stock of <i>Totoaba macdonaldi</i> in the ‘extended cartel’ scenario, where the vertical monopoly takes over the ownership of farming operations	27
2.A.3	Evolution of the threshold v to compare vertical monopoly and price setting (Bertrand) harvest functions	30
2.A.4	Percent change in steady state population across scenarios, following the joint evolution of illegal transaction and farming costs	30
2.A.5	Von Bertalanffy Growth curves for wild and farmed <i>Totoaba macdonaldi</i> under different growing conditions	31
3.B.1	Illustration of the habitat and fuel graphs for $n = 3$	52
3.B.2	Number of cycles as a function of biodiversity habitat and budget	53
3.B.3	Production possibility frontier between constraint (as a % of maximum biodiversity sustainable in landscape) and wildfire risk for various budgets, and landscape size	54
3.B.4	Most represented cycles for each biodiversity constraint level, for various budget and landscapes 3×3 , and 4×4 (95% CI shaded)	55
3.B.5	Assessment: surface, components of high-risk graph (95% CI shaded)	56
3.B.6	Assessment: diversity (95% CI shaded)	57
3.B.7	Treatment allocation : number, location	58

List of Tables

2.A.6 Supporting information for the calculation of the <i>Totoaba macdonaldi</i> poaching cost parameters (W_1 and W_2)	32
2.A.7 Regression output for the linear demand estimation calculated by regressing price data on catch data.	33
2.A.8 Supporting information for the calculation of the <i>Totoaba macdonaldi</i> farming cost parameter (v)	34
2.A.9 Summary of <i>Totoaba macdonaldi</i> ecological and market parameters for model calibration	35
2.A.10 Summary of the key functions in the model	36

Introduction

What should I say in the general introduction of my thesis?

- Une page qui résume : le déclin de la biodiversité, l'économie bioéconomique, la contribution méthodologique de la thèse, et les résultats nouveaux.
- Introduction générale thématique : le déclin de la biodiversité / 3-5 pages avec des graphiques.
 - Le déclin de la biodiversité de façon générale
 - Par écosystème et taxon : on parlera alors de poissons, d'espèces invasives, de forêts
 - En rajoutant un petit quelquechose sur le changement climatique
 - Les causes sont à chercher du côté des hommes
- D'où la nécessité d'une approche par les sciences sociales, et l'économie peut y apporter beaucoup, elle l'a déjà fait : social ecological systems, JEEM.
- Pourquoi l'économie et la modélisation bioéconomique?
 - Un problème d'externalité global, de biens publics, de valeurs d'options, d'informations e.g. tous les problèmes spécifiques à l'économie de l'environnement
 - Un outil permettant la modélisation et la prospective: c'est à dire la description, sur base axiomatique, des comportements, à la fois individuels, non coopératifs, et de politique publique.
 - Qui permet d'articuler grâce à un langage partagé différents champs de connaissance, notamment de dialoguer avec les sciences du vivant
 - Nonobstant les contributions
 - Il existe des champs d'application inexplorés, ou des questions importantes non résolues : bien lister ici les dimensions : marché, espace, politique publique/gestion privée.
- **On a donc deux éléments bien identifiés dans la recherche et les questions qu'ils posent** : l'espace et la structure de marché.
- La structure de marché :
 - Question bien vieille dans la littérature sur les ressources naturelles: Solow, Hotelling etc
 - Moins bien tranchée sur la question des ressources renouvelables : rhinos (AER), foresterie etc ...
 - On l'étudie donc dans un cas précis, c'est le chapitre 2
- La question de l'espace :

- L'espace est une dimension importante à prendre en compte, car il conditionne l'exploitation des ressources, autant que comment les décisions doivent être prises : ce qui change (Sanchirico et Costello)
- C'est un challenge qui pose des questions de politique publique : la gestion de l'espace, dans un contexte de fragmentation, de prévention des risques etc est cruciale
- Il faut aussi comprendre comment l'espace, et les processus écologiques qu'il construit, sont formés par les individus [à raffiner] c'est le chapitre 3
- Bien mentionner que l'intégralité des données, codes etc sont disponibles gratuitement et librement.

What's left to do

- List contributions from the literature review, and find a way to put more into it.
- Find references and graphs from the institutions to document biodiversity loss -> IPBES?

Summary of publications and conferences

Chapter 1 : Bioeconomic Models for Terrestrial Social Ecological System Management : a Review, with L. Mouysset

[Replication code](#) and [data](#) are freely accessible

- Published in *International Review of Environmental and Resource Economics* - DOI : 10.1561/101.00000131
- Presentations :
 - European Association of Environmental and Resource Economists (EAERE) Annual Conference, Rimini, 2022
 - ABIES Doctoral Days - Best Poster Award, 2022

Chapter 2: Little downside and substantial gains result from farming of *Totoaba MacDonaldi*, with J. Lawson (co-first author), A. Steinkruger, M. Castellanos-Rico, G.M. Goto, M.A. Cisneros-Mata, E. Aceves Bueno, M.M. Warham, A.M. Sachs and S.D. Gaines

[Replication code](#) and [data](#) are freely accessible.

- Under review at *NPJ Ocean Sustainability*
- Presentations:
 - BIOECON Network Annual Conference, University of Santiago de Compostela, 2023
 - Trade and the Environment, Paris Saclay Applied Economics, 2023
 - European Association of Environmental and Resource Economists Annual Conference, University of Leuven, 2024

Chapter 3 : The Wildland Connectivity Dilemma : a Graph Theoretical Computational Approach, with L. Mouysset

[Replication code](#) and [data](#) are freely accessible

- Working paper
- Presentations :
 - BINGO Seminar, CIRED, 2023
 - Interdisciplinary PhD in Sustainable Development, Columbia University, 2023

Chapter 4 : Fences - the Economics of Movement in Mobile Public Bads

[Replication code](#) and [data](#) are freely accessible

- Working paper
- Presentations :
 - French Association of Environmental and Resource Economists, Université Savoie Mont-Blanc, 2024
 - Parisian PhD Seminar in Environmental Economics, Nogent sur Marne, 2024
 - CIRED Internal Seminar, 2024

Chapter 1

Bioeconomic models for terrestrial social ecological system management : a review

This article [was published](#) in the International Review of Environmental and Resource Economics with Lauriane Mouysset. Data and code are publicly available - DOI 10.1561/101.00000131

Chapter 2

Little downside and substantial gains result from farming of *Totoaba* *Macdonaldi*

This article is under review at *NPJ Ocean Sustainability* and is joint work with Julia M. Lawson (co-first author), Andrew Steinkruger, Miguel Castellanos-Rico, Garrett M. Goto, Miguel A. Cisneros-Mata, Erendira Aceves Bueno, Matthew M. Warham, Adam M. Sachs and Steven D. Gaines

Abstract

Illegal wildlife trade poses a growing threat to species globally. Where bans or policy instruments have failed, conservation farming has been considered, which aims to reduce illegal poaching by “flooding the market” with farmed product. However, predicting if farming will succeed necessitates a holistic understanding of how supply and demand interact and how markets will respond. Poaching and illegal trade for totoaba (*Totoaba macdonaldi*), currently dominated by a Mexican monopolist cartel, has continued unabated despite half a century of prohibitions on international trade and domestic fishing. We investigate if farming can reduce poaching and support a healthy wild population by extending a flexible bioeconomic model of a three-stage illegal supply chain: poachers sell to traders (i.e., middlemen or cartels) who sell to end-markets. While we show under the monopolist a large stable wild population is maintained, this outcome is sensitive to cost parameters. Introducing farming decreases poaching by 29% or increases poaching by 6%, and results are robust to changes in cost parameters. Our results upend previous assertions that certain strategic responses will undermine conservation efforts and always result in population collapse. Furthermore, our quantitative framework can be adapted to evaluate conservation farming for other species and market structures.

2.1 Introduction

Illegal wildlife trade is a multi-billion dollar industry that drives biodiversity loss through unsustainable harvest ([T Sas-Rolfes et al., 2019](#)), spreads zoonotic disease ([Bell et al., 2004](#)), and threatens animal welfare([Baker et al., 2013](#)). The Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) provides a regulatory framework that aims to ensure that international trade of wild animals and plants does not threaten their survival. Yet, for many species, regulatory interventions such as trade bans and controls have failed, and illegal trade in black markets continues to flourish ([Challender and MacMillan, 2014](#); [Challender et al., 2015a](#)). In such instances, supply-side interventions such as conservation farming can theoretically bolster conservation by “flooding the market” with farmed products, leading to reduced market prices and lower poaching incentives ([Gentry et al., 2019](#); [Phelps et al., 2014](#); [Tensen, 2016](#)). Supply-side interventions have occasionally succeeded at reducing poaching and recovering wild populations – e.g., vicuña and spotted cat ([IUCN, 2000](#); [Sahley et al., 2007](#)) – but they have also failed – e.g., green python, African elephant ([Lyons and Natusch, 2011](#); [Hsiang and Sekar, 2016](#)). Uncertainty around conservation outcomes from market-based approaches has led to continued reliance on trade bans and controls that are often ineffective at reducing poaching.

Determining whether farming will succeed or fail requires a holistic understanding of a specific illegal wildlife market¹, including the interplay between market conditions and ecological criteria ([Challender et al., 2015b](#)). Studies have pointed to a common set of farming pitfalls. Species with slow individual growth rates and low fecundity are often unable to grow supply quickly enough to displace illegal products. Further, if poaching is very inexpensive, it is impossible for farming to undercut prices ^{6,8} – e.g., dried seahorses are ‘free’ to poach when retained as bycatch ([Lawson et al., 2017](#)). Demand-side concerns are focused on substitutability between farmed and wild products. Consumers of wildlife for medicinal or conspicuous purposes often prefer wild products for greater perceived potency or associated social status ([Dutton et al., 2011](#); [Gratwicke et al., 2008](#); [Fabinyi, 2012](#)). Here, we develop a quantitative framework that comprehensively considers all these pitfalls while accounting for detailed species-specific and market information.

Another critical factor in driving the success or failure of farming is market structure: illegal markets are often characterized by imperfect competition – where an individual trader or a small number of traders (i.e., middlemen, cartels, gangs, or other criminal organizations) dominate illegal trade and exert significant control over market prices. A bioeconomic model that predicts how imperfectly competitive markets will respond to competition from farming was developed almost two decades ago ([Bulte and Damania, 2005](#); [Damania and Bulte, 2007](#)). Predicted strategic responses depend on how a trader chooses to compete with farming. If a trader responds by price setting (an aggressive response where the trader tries to undercut farmed prices and take market shares), then poaching pressure will increase and can lead to the collapse of the wild population. On the other hand, if traders respond by quantity adjustment (a mutually beneficial response where the trader competes on the amount of output produced, letting market prices adjust), poaching pressure is reduced and wild populations have the possibility to increase. This model has been widely used to both justify ([Biggs et al., 2013](#); [Abbott and van Kooten, 2011](#)) and discourage ([Tensen, 2016](#)) prospective farming initiatives. The authors of the original bioeconomic model concluded that farming is a perilous coin toss ([Bulte and Damania, 2005](#); [Damania and Bulte, 2007](#)). Here, we expand upon this model and reach a different conclusion: that farming can maintain large, stable wild population sizes that are robust to changes in cost structure under both types of competition. Furthermore, quantity adjustment yields substantial decreases in poaching and is the more likely response because prices and profits are higher than under price setting ([Singh and Vives, 1984](#)).

We explore the biological and economic performance of conservation farming for totoaba swim bladder in the context of illegal poaching and trade under different market conditions (Froehlich et al., 2017). Specifically, we examine the evolution of poaching and wild totoaba biomass, as well as prices and profits for different economic actors. The lifecycle for totoaba has been successfully closed in aquaculture, and the species is currently farmed in Mexico for domestic meat production. Totoaba is endemic to Mexico's Gulf of California and is threatened by a lucrative illegal international trade for its large swim bladder (C4ADS, 2017; env, 2019, 2016) . A single totoaba swim bladder can sell for up to \$80,000 USD per kilogram in Chinese end markets, where it is purchased for special occasions, gifting, and speculative investment (ElephantActionLeague, 2018; Sadovy de Mitcheson et al., 2019; Martínez and Alonso, 2021). For nearly half a century international trade for totoaba has been prohibited, and the legal totoaba commercial fishery has been closed. However, illegal fishing and trade continue and are controlled primarily by a single criminal organization (a cartel) that will likely respond strategically to farming (Damania and Bulte, 2007; Felbab Brown, 2022)

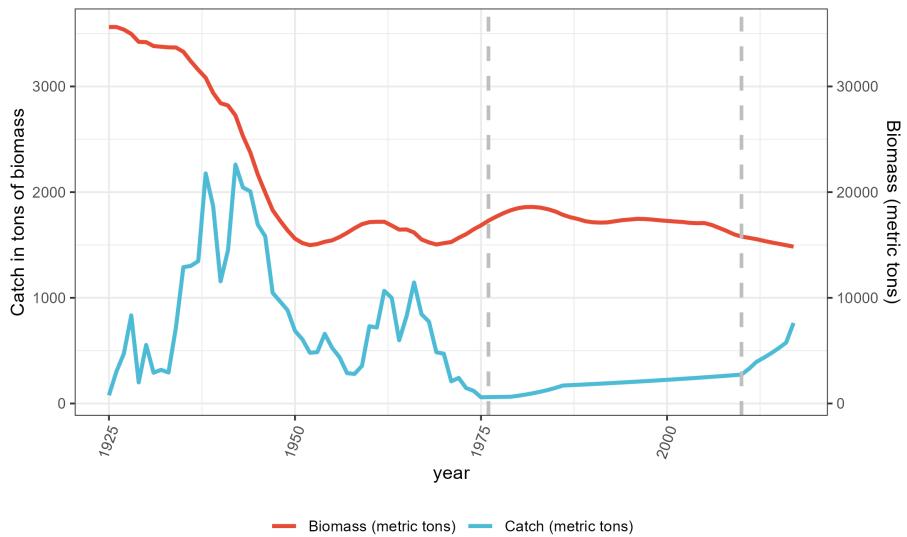


Figure 2.1: Evolution of totoaba population and catch over time

Dashed lines represent listing as CITES Appendix II species, and cartel takeover, respectively

There is an urgent need to reduce poaching for totoaba, as the vaquita (*Phocoena sinus*), a porpoise also endemic to the upper Gulf of California, is caught as bycatch in gillnets used to catch totoaba. The vaquita is on the brink of extinction as there are now fewer than fifteen individuals remaining (Rojas-Bracho et al., 2022). Furthermore, illegal trade has had negative social welfare consequences, as cartels are increasingly extorting Mexican fishing communities (Felbab Brown, 2022). Despite Mexico's attempts to stop totoaba poaching through various enforcement mechanisms, the country recently received wildlife trade sanctions for taking inadequate action (Rojas-Bracho and Reeves, 2013; CITES, 2023). Conservation farming presents a legal alternative to reduce illegal fishing by manipulating market structure.

We assemble and leverage a unique wealth of information on the totoaba stock, poaching sector, and farming sector to estimate the effects of market structure on poaching harvest and stock biomass. We focus on the market structure that best characterizes the totoaba trade – a vertical monopoly where a single monopolist trader controls the entire supply chain – and evaluate how this trader will respond strategically to competition from farming. We also show how to identify an effective policy space, where all supply, demand, and market structure parameters align to ensure that conservation farming will reduce poaching. Our results challenge long-standing model

conclusions ([Bulte and Damania, 2005](#); [Damania and Bulte, 2007](#)), thereby disrupting widely-held beliefs about the impacts of conservation farming. In particular, previous studies cautioned that when a trader responds to farming through price setting, the wild population always declines dramatically. In contrast, we find that for totoaba, price setting can maintain a stable and large population given that as the population size decreases, fishing costs increase. To ensure low retail prices, traders must limit the price they pay to poachers and maintain a viable wild population.

2.2 Methods

We examine the effect of market structure and competition on poaching a population of wild animals using the logistic growth function (Figure 2.2). The poaching harvest function intersects with population growth producing stable and unstable equilibria. If poaching pressure is high relative to population growth (i.e, when demand is large, or poaching costs are low), a single stable equilibrium point is observed with a low wild abundance (an overharvested population). In the opposite scenario, where poaching pressure is low relative to population growth (i.e, when demand is small, or poaching costs are prohibitive), a single stable equilibrium point is observed with a high wild abundance (a healthy population). Between these extremes, two or three potential equilibria can emerge, with uncertain results that depend on the initial size of the population: a large initial population will result in a high abundance equilibrium point, and a small initial population will result in a low abundance equilibrium point.

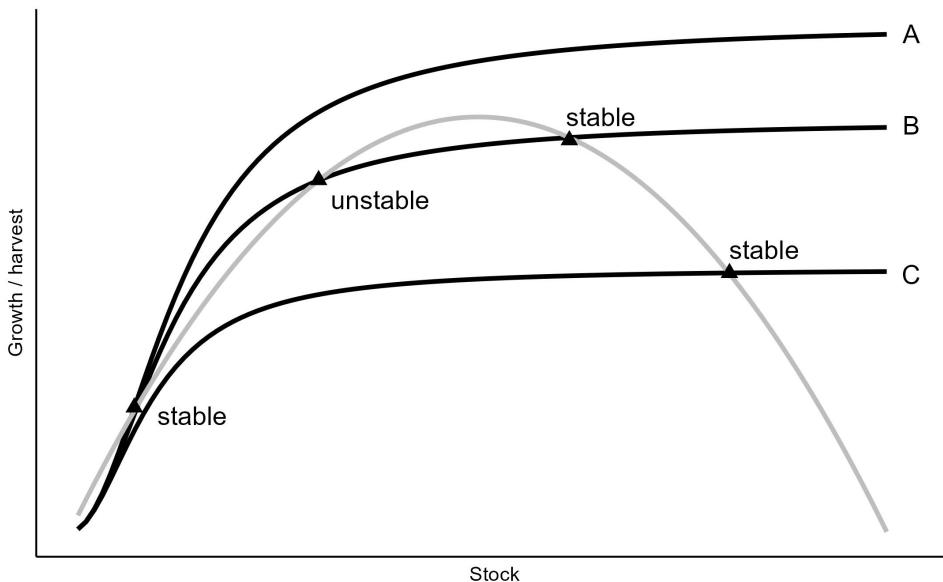


Figure 2.2: Schematic of equilibrium points under different poaching harvest functions

Logistic growth function (light gray) showing equilibria points resulting from three hypothetical poaching harvest functions (black). (A) a single low stable equilibrium point; (B) uncertain outcome, three interior equilibria two of which are stable and one unstable and separating. The long run equilibrium point will depend on the initial size of the population. A large initial population will result in a high abundance equilibrium point, and a small initial population will result in a low abundance equilibrium point; (C) a single high stable equilibrium point.

To assess expectations for totoaba, we first calculate equilibrium points for the stock in the absence of conservation farming under vertical monopolistic conditions (hereafter referred to as monopolistic conditions for ease) (Figure 2.3). A single trader exists in a single location where he is the sole buyer, typical of endemic species such as totoaba ([Wyatt et al., 2020](#); [Martinez-Alvarado and Martinez, 2018](#)). The trader sells poached harvest on an end market where prices and quantities can be manipulated.

Next, we add conservation farming to the monopolistic market structure, creating a duopolistic

market (Figure 2.3). We calculate equilibrium points for the totoaba stock if a monopolistic trader responds to conservation farming either in a way that is (a) mutually beneficial by quantity adjustment or (b) aggressive by price setting. From a policy assessment perspective, any scenario where poached harvest produces a single high stable equilibrium point, and the monopolist cartel loses income, presents clear conservation and social welfare benefits.

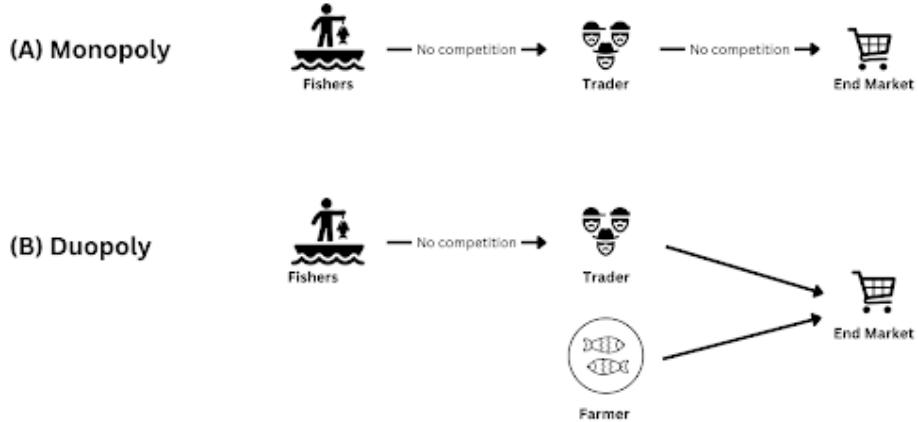


Figure 2.3: Schematic of monopoly and duopoly market structures

(A) monopolistic conditions, where fishers sell to a single trader where they are the sole buyer. This single trader sells poached harvest on an end market where they can manipulate prices and quantities. (B) Next, we add duopoly with farming: A monopolistic trader responds to conservation farming either in a way that is mutually beneficial by quantity adjustment or aggressive by price setting.

Here we briefly discuss our methods with an emphasis on the empirical application. Information on our theoretical conclusions from the bioeconomic model we revisited, lemmas and proofs can be found in the Appendix, section 2.A.5. Table 2.A.10 summarizes all the functions of the model.

2.2.1 The Poaching Model

The growth of the fish stock follows a logistic curve and the stock is poached following a Gordon-Schaefer production model. Totoaba population growth parameters were obtained from the 2017 stock assessment, where the carrying capacity (K) was 20,226 mt, and the stock biomass in 2017 was 14,844 mt ([Cisneros-Mata, 2020](#)). The intrinsic rate of population increase (r) was predicted using the *FishLife* package in R, which estimates growth parameters using totoaba-specific life history data from *FishBase* ([Thorson et al., 2017](#)). The growth equation is :

$$g(x) = rx \left(1 - \frac{x}{K}\right) \quad (2.1)$$

using a predicted r of 0.20. We do not consider potential effects of hyperstability of the stock resulting from poaching on seasonal spawning aggregations ([Erisman et al., 2011](#)) or age structure.

Poachers optimally determine their effort to maximize their profit, with constant catchability σ , and stock biomass, x , obtained from the 2017 stock assessment⁴⁶, and a linear quadratic cost of effort function, E . The poaching equation is $q = \sigma x E$ where $\sigma = 0.00002$.

Poachers are faced with a linear quadratic cost function $C(E) = W_1 E + W_2 E^2$. We calculated two poaching cost parameters W_1 (the linear coefficient of the cost function) and W_2 (the quadratic coefficient of the cost function) by (a) estimating total and average annual operating costs of the fishing fleet using semi-structured interviews conducted by the authors of this study; and (b) calibrating a linear quadratic cost function that matches historical data and predicts future cost

evolution.

We conducted semi-structured interviews in the upper Gulf of California with two fishing cooperative leaders and four fishers in July and August 2018. These interviews informed annual poaching costs: food and fuel, labor, gear replacement, and bribes paid to fisheries officials. The fishery operates over six months with a variable number of active vessels, monthly fishing days, and sets per day ([Cisneros-Mata, 2020](#)). Poaching costs also include annual fleet-wide costs related to gear confiscations, vessel replacement, and fines, adapted and extrapolated from a summary of law enforcement actions provided by Mexico ([noa, 2018](#)). The cost per fishing trip was estimated to be \$5,051.26 during the low season (January and June), \$8,385.34 during the mid-season (February and May), and \$14,386.7 during the high season (March and April) (supplementary table 2.A.6). In our analysis we reconstructed a linear quadratic cost function with cumulative effort. We considered effort in each season cumulative with effort in less intense seasons. We used a low-season average cost for effort levels between 0 and low-season effort; for effort levels between low-season effort and cumulated low and mid-season efforts, we used a mid-season average cost.

We estimated the corresponding poaching cost parameters to match the observed average cost and modeled marginal costs at historical levels (resulting in cost parameters $W_1 = 12,200$ & $W_2 = 0.57$). Our low sample size precludes a robust statistical estimation of these cost parameters, e.g., of the historical cost function and of the evolution of costs if the fishery were to increase. To account for this uncertainty, we run a sensitivity analysis on two dimensions of costs. First, we use different estimates for the average cost and reconstructed total costs, ranging from -10% to +30% of our high season average cost estimates. Second, we test weights for the linear and quadratic costs, ranging from a purely linear cost ($W_1 = 14,386.7$, $W_2 = 0$) to a purely quadratic cost ($W_1 = 0$; $W_2 = 3,74$).

The resulting poaching profit function is calculated as follows:

$$\Pi = p\sigma xE - W_1E - W_2E^2 \quad (2.2)$$

Traders operate on the end market, taking prices as given (competitive scenario) or determining prices (monopolistic scenario) to maximize profits. Traders face a linear demand function. We estimate a linear demand function by regressing price data on estimated catch from 2014 to 201746, yielding the equation $p(q) = \alpha - \beta q$ where the intercept, α , is \$1,625,837 USD and the slope coefficient, β , is \$1,563.75 USD (see supplementary table 2.A.7). Price data were obtained from available literature that provided estimated weight and value of totoaba maw seizures 24,26,50,51. In addition to the literature review, valuable insights were obtained through personal communication with Wild Aid Investigators (pers. comm. Anonymous Wild Aid Investigators, 2018) as well as with local fishers and cooperative leaders in the upper Gulf of California, as previously described. The information shared by investigators and stakeholders was aggregated with the existing data from the literature. To ensure consistency and comparability, we standardized the weight measurements to grams and the currency values to US dollars. We assume that annual catch reaches the market during the same year, i.e, there is no stockpiling. As data are notoriously difficult to acquire for illegal trade, we pool observations and estimate a stationary demand function (supplementary table 2.A.7).

Traders buy totoaba from poachers at **price** s (USD/metric ton). The price paid to poachers balances demand from traders and supply to poachers. It decreases as the population increases, as fishing becomes less demanding. Traders also pay a unit transaction cost c (USD/metric ton), which we conservatively estimated to be zero. At a minimum this unit transaction cost includes transport (land and air travel), and payment to two or three 'runners' who carry up to ten swim bladders each (pers. comm. Anonymous Wild Aid Investigators, 2018). We know through anec-

dotal evidence that unit transaction costs c are likely large ([ElephantActionLeague, 2018](#)). However, due to scarce evidence, we used a value of $c = 0$ thus adopting a conservative strategy.

2.2.2 The Farming Model

We use a linear profit model for aquaculture and estimate a unit farming production cost parameter v (USD/metric ton) using annual operational costs (labor, feed, vessel fuel, facility and administrative fees), as well as annual maintenance of pens (including cleaning) and vessels, using information provided by existing aquaculture facilities (supplementary table 2.A.8). Population growth rates differ in the wild and in captivity. Using captive growth rates obtained from personal communication with totoaba aquaculture producers, we consider harvestable size to be between 4.5 and 5 years old (an adult weight of 21.43 – 27.2 kg), associated with a swim bladder size between 417 – 529 g (supplementary figure 2.A.5). A minimum farmed harvestable size of 4.5 years closely corresponds to the mean swim bladder size (500 g) and estimated adult totoaba size (25.7 kg), as reported in surveys of individuals harvested in the wild ([Cisneros-Mata, 2020](#)). We considered this to be the size at which farmed totoaba would be competitive with the average wild-caught totoaba. We assume that aquaculture operates on a homogenous rotation ([Faustmann, 1849](#); [Mitra and Wan, 1986](#)). The implications of this assumption are discussed in the appendix 2.A.3.1. We compute the farming cost per metric ton as the capitalized sum of annual costs over 4.5 years at a 10% interest rate.

2.2.3 Demand

We use a linear demand function in the case of the vertical monopoly, estimated using price and catch data from 2014 to 2017 (see table 2.A.7), such that $p^w = \alpha^w - \beta^w q^w$. Upon the introduction of aquaculture, following [Singh and Vives \(1984\)](#) and [Damania and Bulte \(2007\)](#), we include a substitutability parameter γ , which measures the imperfect substitutability between farmed and wild products in the linear demand functions. When farmed products are introduced, the linear demand function is modified such that $p^i(q^i, q^k) = \alpha_i - \beta_i q^i + \gamma q^k$ where q^i and q^f indicate the supply from the wild (w) and the farmed supply (f). This demand system emerges from a linear quadratic utility function in Supplementary Text (section 1.3.2). When demand intercept α_i s are equal, and own price effect $\beta_i = \beta_j = \gamma$ are equal, products are perfect substitutes. When demand intercepts are equal, but own price effects differ ($\beta_i \neq \beta_j$), then $\frac{\gamma^2}{\beta_i \beta_j}$ denotes the degree of product substitutability. At present, there has been no stated preference investigation for wild and farmed totoaba swim bladders in Chinese end-markets, although we know that the end-market economic value for fish maw is determined by taxon, size, and thickness of swim bladder. Investigative work in Mexico reports that it is challenging to distinguish between wild and farmed specimens ([ElephantActionLeague, 2018](#)). Therefore, we assume high substitutability (75% product substitutability) and check for smaller substitutability values in our sensitivity analysis (Figure 2.7) (see supplementary table 2.A.9 for a list of parameters).

2.3 Results

2.3.1 Totoaba stock under monopoly is sensitive to cost structure

We revisit and expand upon a bioeconomic model developed nearly two decades ago which differentiates between poachers and traders and develops a three-stage game ([Bulte and Damania, 2005](#); [Damania and Bulte, 2007](#)). The totoaba is an endemic species that is illegally traded by a single trader, a monopolist, who dominates the market. This is the market structure that best

characterizes the present consolidated totoaba trade ([Felbab Brown, 2022](#)). In this setting, a single monopolist trader restricts the supply of wildlife products to consumers, leading to increased prices and profits for the monopolist.

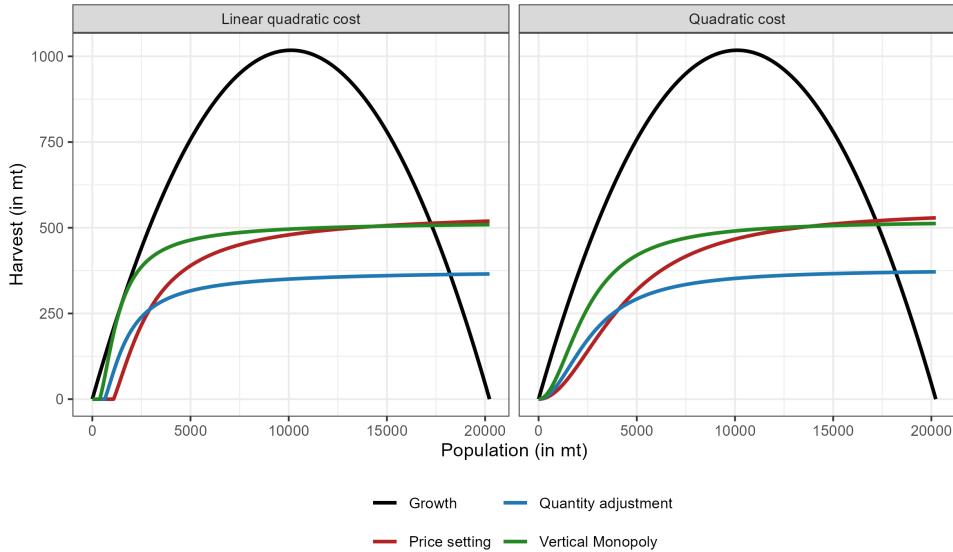


Figure 2.4: Equilibrium points for wild totoaba stock under different market structures with (left) a linear quadratic cost structure, and (right) a quadratic cost structure.

Logistic growth function (black) for *Totoaba macdonaldi* wild stock biomass with intersecting colored lines representing different market structures and competitive responses. Harvest under the status quo vertical monopoly is represented by the green curve. When conservation farming is added to the monopoly scenario the trader can respond either in a mutually beneficial way by adjusting the quantity supplied given a market price (quantity adjustment, in blue). Alternatively, the trader can respond aggressively and try to set a price that undercuts the price of farmed products, resulting in increased poaching (price setting, in red)

We initially calculate equilibrium points for totoaba assuming a quadratic cost structure, consistent with the original model, before calculating equilibrium points under a linear quadratic cost structure (Figure 2.4). Under the quadratic cost structure used in the original bioeconomic model, the totoaba wild stock biomass remains at a high steady-state equilibrium of 17,259 mt. However, we expand upon the quadratic cost structure, introducing a linear quadratic cost structure to account for energy costs associated with fishing. A linear quadratic cost structure more accurately represents new poachers being recruited to the fishery as fishing opportunities increase ([Péreau et al., 2012; Clark, 2007](#)).

We find that under monopoly the linear quadratic cost structure is sensitive to cost parameter specifications, where relatively small changes in cost parameters can cause multiple steady states to emerge (Figure 2.5). If an increase in poaching comes at a small cost increase compared to historical average costs, the aggregate cost is close to linear (e.g. $W_2 = 0.47$) and below, compared to baseline $W_2 = 0.57$. In this case, a low steady-state equilibrium of 1,106 mt, an unstable intermediate equilibrium arises at 1,842 mt and a high stable steady-state equilibrium of 17,277 mt in the vertical monopoly. Our model uses the best available information on the totoaba fishery, but uncertainty surrounding the projected evolution of fishery-wide poaching costs warrants a cautious assessment of monopoly performances: while it could maintain a healthy population, it can also lead to stock collapse.

2.3.2 Farming produces conservation benefits

While our results show that totoaba stock may remain healthy under the current monopolistic market conditions, these results are sensitive to changes in poaching costs (Figure 2.5). Therefore,

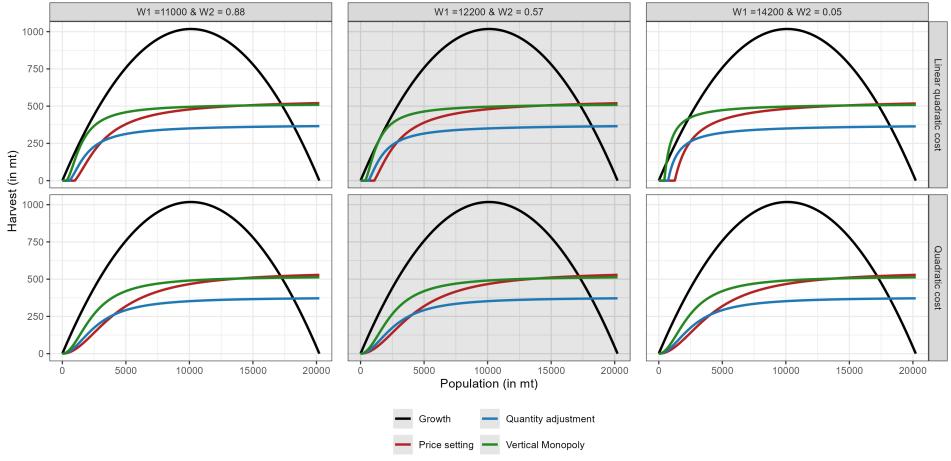


Figure 2.5: Sensitivity of equilibrium points to cost structure for wild totoaba stock

Logistic growth function (black) for *Totoaba macdonaldi* wild stock biomass with intersecting colored lines representing different market structures and competitive responses. Harvest under the status quo vertical monopoly is represented by the green curve. When conservation farming is added to the monopoly scenario the trader can respond either in a mutually beneficial way by adjusting the quantity supplied given a market price (quantity adjustment, in blue). Alternatively, the trader can respond aggressively and try to set a price that undercuts the price of farmed products, resulting in increased poaching (price setting, in red). Cost parameters W_1 and W_2 correspond to the linear quadratic cost structure. In the top panel, equilibria are displayed for the linear quadratic cost, on the bottom, for a quadratic cost. On the left panel, the quadratic component is large, and vertical monopoly maintains a healthy stock. Center panel highlights the baseline scenario. In the right panel, the cost structure is close to linear. In this case, the vertical monopoly may lead to drastic stock decline.

we ask if conservation farming can improve upon the status quo by producing a robust single high stable equilibrium point and reduced cartel profits.

We add conservation farming to the monopolist model and now have two ‘firms’ – a trader and a farmer – competing on a duopolistic market. When farming supplies legal product to end-market consumers, the demand for illegal product will fall, assuming that wild and farmed products are substitutable (an assumption we explore later). The monopolist trader can respond to competition in two ways: a mutually beneficial way by adjusting the quantity supplied given a market price (quantity adjustment), or alternatively, an aggressive way that tries to select a price that undercuts the price of farmed products (price setting). In both scenarios, the trader and farmer choose a quantity supplied simultaneously, without knowing how the other will respond. Illegal markets are almost always characterized as competing through quantity adjustment (Poret, 2009; Flores, 2016). Under the assumption that products are substitutable, it is more profitable – and therefore more likely – for both firms to compete through quantity adjustment (Singh and Vives, 1984). When goods are substitutes, if both firms restrict the quantities supplied, they both enjoy higher prices. If they flood the market, prices and profits collapse. In the case of totoaba, we find that if traders respond through quantity adjustment under the linear quadratic cost structure, then the wild stock biomass increases by 5.45% (compared to a monopoly) to a steady state equilibrium of 18,220 mt, or to 90% of carrying capacity (Table 2.6). This represents a reduction in poaching harvest of 28.27% and \$195.16 million USD of annual lost profit to the trader.

Even if traders respond aggressively through price setting, considered a less likely response (Singh and Vives, 1984) a single high equilibrium emerges (Figure 2.4). Price setting is considered a much less likely response to competition because the trader would face steep profit losses. Under the high steady-state equilibrium with the linear quadratic cost structure, wild stock biomass decreases by 0.24% relative to monopoly, to a steady-state equilibrium of 17,235 mt, or to 85% of carrying capacity (Table 2.6). Although the high steady-state reflects a relatively small increase in poaching harvest by 5.85%, it would result in \$313.84 million USD of annual lost profit to the cartel, making this strategy unlikely.

Scenario	Poached harvest (in mt)	Farmed harvest (in mt)	Steady state population (in mt)	Illegal profit (in million USD)	Farming profit (in million USD)	Fishing profit (in million USD)	Aggregate profit (in million USD)	Illegal profit change (in million USD)	Variation in ss. pop.	Poaching change (%)
Vertical Monopoly	507.04	0.00	17277.0	402.02	0.00	1.22	403.24	0.00	0%	0%
Quantity adjustment	363.71	333.60	18220.5	206.87	174.03	0.57	381.46	-195.16	5.46%	-28.27%
Price setting	536.70	430.05	17235.0	88.18	58.74	3.57	150.49	-313.84	-0.24%	5.85%

Bioeconomic performance

Figure 2.6: Economic and ecological performance of different market regimes

Our current specifications for totoaba show that price setting leads to a slight increase in poaching pressure, however, we argue that price setting does not universally lead to increased poaching pressure, challenging a key conclusion from the original bioeconomic model ([Bulte and Damania, 2005](#); [Damania and Bulte, 2007](#)). Farming puts an upper bound on the price traders can pay to poachers in order to remain competitive. When the cost of farming becomes lower than the combined cost of poaching and trading, price-setting competition does not inevitably result in the overexploitation of the wild stock. This is because when farming costs are low, traders have an incentive to maintain large stocks by poaching less to remain competitive with farmers. This limits the price paid to poachers. On the other hand, when farming costs are large, traders have an incentive to poach more, paying a larger price to poachers while remaining competitive with the farming sector. In the case of totoaba, species specific traits and market characteristics result in a slight increase in poaching in the price setting scenario. However, if the carrying capacity were smaller, or demand lower, the price-setting equilibrium would result in conservation benefits.

While we focus on the effect of conservation farming on a monopolistic market structure, given that this scenario best represents the totoaba fishery today, the effect of conservation farming on market structures can be explored in different contexts. We model alternative market structures, including scenarios with multiple competing traders or multiple competing farmers, and find that if the number of farmers exceeds the number of traders, poaching levels will decline (supplementary figure 2.A.1). Additionally, if farming is taken over by monopolists, we find that poaching is reduced and the wild population increases (supplementary figure 2.A.2).

2.3.3 An effective policy space for farming.

Our analysis provides a quantitative framework that can identify an effective policy space where all supply, demand, and market structure parameters align to ensure that conservation farming will reduce poaching, improving greatly on the original bioeconomic model and the limitations of binary qualitative approaches ([Phelps et al., 2014](#); [Tensen, 2016](#); [Bulte and Damania, 2005](#); [Damania and Bulte, 2007](#); [Challender et al., 2019](#)). This bioeconomic model allows researchers to quantify: (a) how much cheaper farming must be relative to poaching to be competitive; (b) how much of a demand increase can be absorbed by farming; and (c) how substitutable must wildlife products be for farmed products to displace wild products. Critically, we also explore how the interaction between these factors may affect outcomes. We explore how sensitive the results are for totoaba, offering general and totoaba-specific policy solutions to help ensure that conservation farming remains in the effective policy space.

We find that the cost of conservation farming for totoaba can be high and still competitive with poaching, but this is contingent on the cost for traders also being high (supplementary fig-

ure 2.A.4). Traders inherently rely on poachers to obtain totoaba, and if farming is expensive this forces traders to pay poachers higher prices. If traders compete with poachers under the more likely quantity adjustment response, the population remains healthy, even increasing by nearly 6% from the monopoly steady state. However, if traders compete with farmers by price setting, the low prices paid to poachers can incentivize poachers to increase fishing pressure in order to maintain payouts. This can lead to a decrease in the wild population biomass modestly by 0.24% from the monopoly steady state. Policymakers can support farming success by subsidizing farming to keep the cost low while maintaining enforcement to keep the cost of poaching high (for totoaba this includes marine patrols, fisheries closures, and gillnet bans). To mitigate the possibility of stock decline under the less-likely price-setting response, we identify that maintaining conservation farming below \$77,339 USD per mt of totoaba (amounting to a 14% subsidy on unit production cost) will prevent any increase in poaching pressure under either competitive response, assuming no effect of law enforcement in our baseline model.

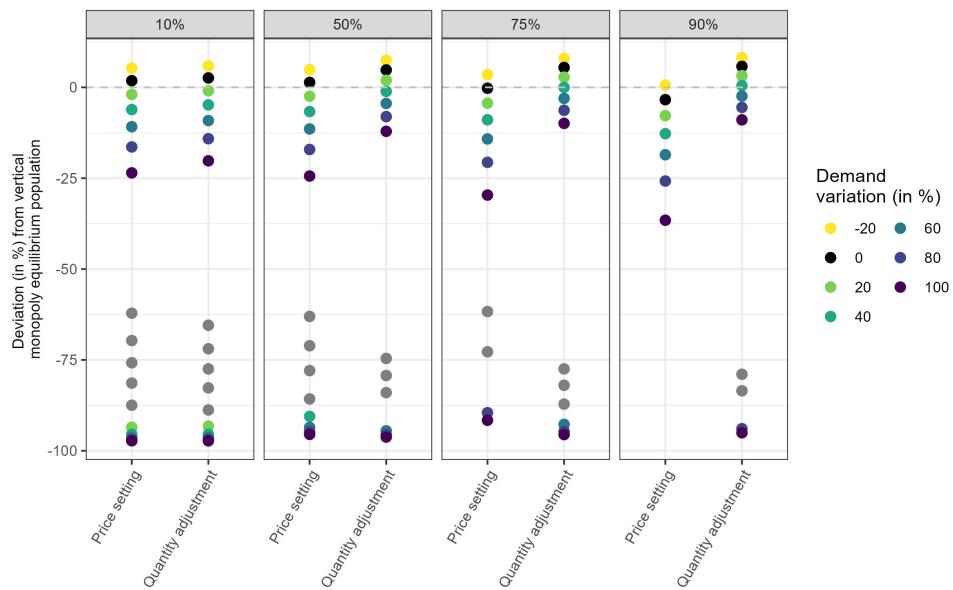


Figure 2.7: Interaction between substitutability and demand under duopolistic competition

Each panel represents a different substitutability between farmed and wild product: large (90% substitutability), baseline (75%), medium (50% substitutability), and low (10% substitutability). Our baseline results are in the 75% substitutability case, with zero demand variation (black dots). When conservation farming is added to the monopoly scenario the trader can respond aggressively and try to set a price that undercuts the price of farmed products (price setting), alternatively the trader can respond in a mutually beneficial way by adjusting the quantity supplied given a market price (quantity adjustment). We simulate a change in end-market demand ranging from a reduction in demand by 20% to an increase in demand up to 100%, in increments of 20%. One, two, or three potential equilibria can emerge. Where three equilibrium points emerge, we color only the high and low stable equilibria (unstable equilibria are indicated in gray). The dotted horizontal lines indicate the status quo monopoly equilibrium population (in the absence of conservation farming). Points closer to 0 represent a high stable equilibrium point, whereas points closer to -100 represent a population collapse stable equilibrium point.

Our results confirm that high substitutability is critical to conservation farming success and leads to larger conservation benefits in the quantity-setting equilibrium, under the assumption that demand remains stable (Figure 2.7). Fish swim bladders have a wide variety of uses and values, and it is possible that farmed totoaba swim bladders may enter into these different product streams ([Sadovy de Mitcheson et al., 2019](#)). In the case of no substitutability, two separate, non-competitive markets emerge. In this scenario the status quo is maintained, both firms set high prices, and traders continue to operate as a monopoly because farmed product does not

compete with wild product. At the other extreme, in the case of perfect substitutability, consumers prefer the cheaper option without any preference of source. This increases the intensity of potential price-setting competition between firms and further depletes the stock in this case. To comply with CITES captive breeding guidelines totoaba must be identified as farmed (CITES, 2019), and distinguishing between products to meet regulatory obligations can artificially lower substitutability. Outcomes vary under intermediate states of substitutability. For low to medium substitutability (i.e., 10 – 50%) traders and farmers are still likely to limit quantity: undercutting a competitor would yield significant profit losses. For high substitutability (i.e., 90%) there is an incentive to compete for market control either by price setting or quantity adjustment, which reflects our main results.

The value of totoaba swim bladder is tied to rarity, and while demand evolution is an open empirical question, we test the sensitivity of our results to simultaneous changes in demand and substitutability (Figure 2.7). Totoaba swim bladder purchases are ‘conspicuous consumption,’ luxury products commonly purchased for social status and speculative investing by wealthy consumers (Sadovy de Mitcheson et al., 2019; Veblen, 2023). A decrease in swim bladder price resulting from conservation farming may actually undermine the desirability of totoaba swim bladders in Chinese end markets, given that the high monetary value is linked to high social status (Jenkins, 2016). However, some increase in demand may be expected if a legal product becomes available, as law-abiding consumers will be more likely to purchase wildlife products when those products are traded and purchased legally (Phelps et al., 2014). Under our high substitutability assumption (75%), competition through quantity adjustment can withstand a 40% increase in demand, whereas competition through price setting is not robust to demand increases. For price setting, a demand increase of 40% would cause the equilibrium population to decrease by 10% from the monopoly status quo, increasing poaching by 216 mt.

There is a much higher threat to the wild population if demand increases under low to medium substitutability (i.e. 10 – 50%), given that this additional demand cannot be fully met by farmed product (Figure 2.7). In the best-case and most likely scenario, medium substitutability (50%) can meet a 20% increase in demand if competition occurs through quantity adjustment, although uncertain outcomes (e.g. high and low steady states) start to emerge if demand increases by 60% or more. In the worst-case scenario, if competition occurs through price setting and products have medium substitutability (50%), any increase in demand reduces the wild population from the status quo. While increases in demand of 20 – 40% still produce a single high equilibrium point, the population size is lower than under monopoly. Furthermore, if demand increases beyond 80%, uncertain outcomes emerge, with the wild population either stabilizing at a high equilibrium point (14,322 mt in the price setting scenario; 15,886 mt in the quantity adjustment scenario) or being pushed to a low equilibrium point (ranging from 763 mt in the quantity adjustment scenario; 909 mt in the price setting scenario). We recommend that stated preference investigations on wild versus farmed product should be undertaken in Chinese end-markets and that these investigations include questions focused on perceived social status benefit and legality (Hinsley and ‘t Sas-Rolfes, 2020).

2.4 Conclusion

Our results show that conservation farming presents a potentially high reward intervention. If traders respond to competition from farming by quantity adjustment, the wild totoaba stock is predicted to increase by 5.45% relative to the status quo monopoly, to a high stable biomass of 18,220 mt (90% of carrying capacity). In addition to improving the totoaba wild stock, this quantity adjustment response will decrease poaching by 28.27% relative to the status quo. If traders

respond by price setting, the wild stock biomass decreases by less than 1% to a high stable biomass of 17,235 mt (85% of carrying capacity). Economic theory concludes that quantity adjustment is the more likely outcome because restricting quantities allows both farmers and traders to collect higher profits (Singh and Vives, 1984). Conservation farming presents a more robust outcome to the status quo monopoly market structure (where a single trader dominates the market), as the wild totoaba reaches either a low or high stable equilibrium biomass depending on the poaching cost structure. We find that if products have high substitutability they are more likely to maintain a high stable equilibrium. Further, under a quantity adjustment response, highly substitutable products can better maintain this high stable equilibrium for demand increases up to 40%. Our results are sensitive to changes in substitutability and increases in demand, therefore we encourage a thorough understanding of end-market demand before implementing conservation farming for totoaba.

We revive an existing bioeconomic model and reach different and optimistic conclusions about the potential for conservation farming to reduce poaching and maintain a healthy wild population. We provide a novel framework to objectively assess the potential effects of farming by grounding our analysis in detailed species ecology and market data. Furthermore, our approach provides a rigorous alternative to existing qualitative frameworks that are unable to analyze the interaction between multiple variables. While our analysis focuses on totoaba, the bioeconomic model is flexible and can be applied more broadly to other species and contexts to examine the effect of conservation farming on a wild population.

Acknowledgements

We thank Mark Buntaine, Chris Costello, and Lauriane Mouysset for helpful comments and feedback on the manuscript, as well as members of the Costello research group. J.M.L acknowledges funding from the Daniel and Dianne Vapnek Fisheries Management Fellowship, the Schmidt Family Foundation Research Accelerator Award as well as from the National Sciences and Engineering Research Council of Canada (NSERC) Postgraduate Scholarship. M.C.-R. acknowledges funding from the Latin American Fisheries Fellowship.

2.5 Contributions and data availability

J.M.L., S.J., M.C-R., G.M.G., M.A.C-M., E.A-B, and S.D.G. contributed to writing the manuscript. J.M.L., S.J., A.S., and S.D.G. contributed to study conception and design. All authors contributed to data acquisition and analysis. All authors approve of the submitted manuscript.

The data that support the findings of this study are available [here](#)

The code used for this study is publicly available on [Github](#) and [archived here](#)

2.A A theoretical model of poachers, traders, and farmers

Our framework follows [Damania and Bulte \(2007\)](#), with a poaching cost structure adapted to fisheries. The model develops a three-stage dynamic, game theoretic, bioeconomic model. The value chain for poached animal products comprises poachers, middlemen traders, and end markets. As a small number of actors characterizes many wildlife markets, the model features a vertical monopoly and looks at the consequences on wildlife population stocks of the introduction of a farmed substitute. In this setting, farmers compete on end markets with traders in quantity and price. In the original model, price competition unambiguously results in larger harvests than in the vertical monopoly case. Therefore, while quantity competition reduces poaching, the threat of a population collapse in the price-setting case should warrant a cautious approach to conservation farming. We argue that this conclusion is erroneous, as the intricacies of imperfect substitutability and market dynamics have not been properly accounted for in the original model. As a matter of fact, standard economic intuition regarding price-setting competition in the homogeneous goods case does not directly apply here, as fishing costs rise as the stock decreases, limiting the ability of the trader to flood the market. We show that scenarios exist where any type of competition unambiguously leads to positive conservation outcomes, i.e., reduced poaching and larger steady-state stocks. We amend the original results and use this model for simulation.

First, poachers illegally harvest wildlife resources. Second, they sell their catch to a monopsonistic buyer. Third, the buyer sells catches on a monopolistic market, which is not accessible to poachers. We label this value chain ‘vertical monopoly’ as a reference case. We then look at the impact of introducing a competitor on the end market, the farming sector.

2.A.1 Entry in the fishery and poaching supply

We denote the fishing effort by E , which is measured in the number of vessel trips. Entry in the poaching sector, \dot{E} , is a function of payoff and an adjustment parameter. Harvest, q , follows the Gordon-Schaefer dynamic biomass model $q = \sigma x E$, with σ the (stock-independent) catchability coefficient, and E , effort. The payoff is determined by the price paid to the poachers s minus the cost of effort. We adopt a disaggregated view of the fishery, and consider increasing marginal costs of effort, as individuals have to be attracted from other activities with increasing opportunity costs. To account for energy costs, we derive a modified version of this model using a linear-quadratic cost function (see [37, 53]). Entry happens as long as the profit of the marginal poacher is positive :

$$\dot{E} = \eta \frac{d\Pi}{dE} = \eta \frac{d}{dE} [sq - W_1 * E - W_2 E^2] \quad (3)$$

The resource stock biomass x follows a logistic growth curve and is harvested. Overall, the dynamics are:

$$\dot{x} = g(x) - q = rx \left(1 - \frac{x}{K}\right) - \sigma x E \quad (4)$$

Where r is the intrinsic population growth rate, and K is the carrying capacity.

Fishermen enter the fishery as long as the marginal profit from selling to traders along the vertical value chain is positive. As the resource is in open access from the fishermen poachers maximize their instantaneous profit with respect to effort. The optimal effort and aggregate supply of poached fish is:

$$\frac{d\Pi}{dE} = 0 \Rightarrow E^* = \max \left(0, \frac{s\sigma x - W_1}{2W_2} \right) \quad (5)$$

$$\Rightarrow q^* = \max \left(0, \frac{s\sigma^2 x^2 - W_1 \sigma x}{2W_2} \right) \quad (6)$$

Given the linear quadratic nature of the costs, there is no effort or catch for low stock levels and/or low prices. Effort and catch increase with the price paid to poachers, s .

2.A.2 Traders as vertical monopolists, without farming

We introduce a trader who has market power on the end-market (monopoly) and on the primary market, making it a “vertical monopoly”. The trader has to set price s on the primary market to clear the poaching market. On the end market, we assume the trader faces a linear inverse demand :

$$P^m = \alpha^m - \beta^m q^W \quad (7)$$

Trading an illegal commodity incurs transaction costs c . Hence, the monopoly profit can be written as :

$$\Pi^m = (\alpha^m - \beta^m q^W - c - s)q^W \quad (8)$$

The optimal level of output is :

$$q_m^{\tilde{W}} = \frac{\alpha^m - c - s}{2\beta^m} \quad (9)$$

Using the poachers’ supply, it must be that in equilibrium, the supply of the monopolist trader equals the supply of the poachers. The price paid to poachers s balances supply and demand (consistent with equation 13 in [Damania and Bulte \(2007\)](#)). Substituting s^* into equation 9 yields the quantities of poached product in the vertical monopoly scenario :

$$\text{Price paid to poachers : } s_m^*(x) = \frac{W_2(\alpha_m - c) + \beta^m(W_1\sigma x)}{\sigma^2 x^2 \beta^m + W_2} \quad (10)$$

$$\text{Poaching : } q_m^*(x) = \frac{\sigma^2 x^2 (\alpha_m - c) - W_1\sigma x}{2(\sigma^2 x^2 \beta^m + W_2)} \quad (11)$$

First, note that equation 11 is consistent with equation 14 in [Damania and Bulte \(2007\)](#), as the limiting case where $W_1 = 0$ and $W_2 = W$.

2.A.3 Captive breeding, imperfect competition and conservation

In this part of the model, a farmer can grow and sell totoaba. The theoretical model focuses on the duopolistic competition between the two actors on the end market for totoaba. As products are strategic substitutes, it is natural to investigate the case where Cournot competition arises. Indeed, when products are substitutes, each firm tries to maximize its residual demand (25). Nonetheless, given the asymmetric nature of costs, we also investigate Bertrand competition, as [Damania and Bulte \(2007\)](#).

2.A.3.1 Introducing aquaculture

The aquaculture farm needs to determine the optimal harvest age, based on the intrinsic growth rate in the pen, and expected prices. A sizeable literature has shown that rotation time is invariant to market structure in forestry applications ([Faustmann, 1849](#); [Mitra and Wan, 1986](#)) although quantities can be modified. The optimal rotation literature confirms the existence of a Faustmann rotation, where a set of T^* pens are equally distributed among each age class (1 pen per age class until T^*). While it is arguably unrealistic to expect this structure for an inherited forest, it is reasonable to assume that a farm would *ex-ante* determine this rotation period given the expected price schedule over time. We assume that the aquaculture farm aims at producing a product that is as similar as possible from a biophysical stand-point and thus determines T^* . As we consider a stationary demand function, one can write the farming problem as a linear profit maximization

problem, where the unit cost of production equals the capitalized sum of annual average variable costs over T^* periods. Therefore, we assume that an aquaculture firm can raise totoaba at cost v and sell it to the market:

$$\Pi^F = (P^F - v)q^F \quad (12)$$

With v the unit cost per ton of totoaba, corresponding to the capitalized sum of annual costs.

2.A.3.2 Utility maximization and demand functions

Upon the introduction of farmed goods, the inverse demand functions change. We use a model consistent with (Singh and Vives, 1984), where a representative consumer maximizes a quadratic and strictly concave utility function subject to prices:

$$\max_{q^W, q^F} V = \alpha^W q^W + \alpha^F q^F - \left(\frac{\beta^W(q^W)^2 + 2\gamma q^W q^F + \beta^F(q^F)^2}{2} \right) - p^W q^W - p^F q^F \quad (13)$$

Two inverse demand functions emerge, that the traders and farmers face :

$$P^W = \alpha^W - \beta^W q^W - \gamma q^F \quad (14)$$

$$P^F = \alpha^F - \beta^F q^F - \gamma q^W \quad (15)$$

Where W, F refers to wild and farmed. We assume $\gamma > 0$ e.g that goods are substitutes. When $\alpha_W = \alpha^F$ and $\beta^W = \beta^F = \gamma$, the goods are perfect substitutes. When $\alpha^F = \alpha^W$, but $\beta^F \neq \gamma$ or $\beta^W \neq \gamma$, $\frac{\gamma^2}{\beta^W \beta^F}$ measures the degree of product differentiation.

Rearrange the initial inverse demand functions into direct demand functions:

$$q^W = a^W - b^W P^W + e P^F \quad (16)$$

$$q^F = a^F - b^F P^F + e P^W \quad (17)$$

With $a^i = \frac{\alpha^i \beta^j - \alpha^j \gamma}{\beta^i \beta^j - \gamma^2}$, $b^i = \frac{\beta^j}{\beta^i \beta^j - \gamma^2}$ and $e = \frac{\gamma}{\beta^i \beta^j - \gamma^2}$

2.A.3.3 Cournot competition in the retail market

Assume that the two firms compete by setting their quantities. We solve the multi-stage game using backward induction. First, we derive the supply function resulting from Cournot competition. Second, we find the price paid to poachers so that the quantities supplied by the traders on the end market equal the quantities supplied by poachers.

Taking the inverse demand functions and plugging them into the profit functions:

$$\begin{aligned} \Pi^F &= (\alpha^F - \beta^F q^F - \gamma q^W - v)q^F \\ \Pi^W &= (\alpha^W - \beta^W q^W - \gamma q^F - s - c)q^W \end{aligned}$$

In a Cournot equilibrium, each firm takes its competitor's quantity as given, and picks optimal reaction functions.

Solving for the Nash equilibrium using reaction functions, each firm supplies:

$$q_c^W = \frac{2\beta^F(\alpha^W - (s + c)) - \gamma(\alpha^W - v)}{4\beta^W\beta^F - \gamma^2} \quad (18)$$

$$q_c^F = \frac{2\beta^W(\alpha^F - v) - \gamma(\alpha^W - s - c)}{4\beta^W\beta^F - \gamma^2} \quad (19)$$

Now, we find the equilibrium price paid to poachers for each unit of totoaba $s_C^*(x)$ by equating \tilde{q}_c^W and q_c^W , and find the Nash equilibrium supply functions.

In the Cournot equilibrium:

$$\text{Price paid to poachers: } s_C^*(x) = \frac{2W_2(2\beta^F(\alpha^W - c) - \gamma(\alpha^F - v)) + W_1\sigma x(4\beta^F\beta^W - \gamma^2)}{4W_2\beta^F + \sigma^2 x^2(4\beta^F\beta^W - \gamma^2)} \quad (20)$$

$$\text{Poaching : } q_C^{W*}(x) = \frac{\sigma^2 x^2(2\beta^F(\alpha^W - c) - \gamma(\alpha^F - v)) - 2\beta^F W_1 \sigma x}{4W_2\beta^F + \sigma^2 x^2(4\beta^W\beta^F - \gamma^2)} \quad (21)$$

First, including a linear component for energy in the poaching cost significantly raises the price paid to poachers (when $W_1 > 0$). Second, poaching decreases with the degree of substitutability between farmed and wild products (γ), and increases with the production cost of farmed products v . On the other hand, it increases with demand for the wild product α^W . For low stock values, poaching can be null since the production costs increase as stocks diminish. In the polar quadratic cost case (e.g. $W_1 = 0$), our results differ from [Damania and Bulte \(2007\)](#) by a magnitude effect. Nonetheless, the results stand :

Lemma 1: *Assume the market is large, i.e., the residual demand for large stock levels is large enough. For any given wildlife stock, poaching levels in equilibrium with captive breeding will be lower than those without captive breeding, if the introduction of captive-bred animal products has no impact on the parameters of the original inverse demand function for wild animal products.*

See Appendix 2.A.5.1. for proof of Lemma 1

2.A.3.4 Bertrand competition in the retail market

Interior solution: the two firms compete by setting their prices. This section investigates a potential interior equilibrium, where both producers operate on the market.

Using demand functions instead of inverse demand functions:

$$q^F = a^F - b^F P^F + e P^W$$

$$q^W = a^W - b^W P^W + e P^F$$

With $a^i = \frac{\alpha^i \beta^j - \alpha^j \gamma}{\beta^i \beta^j - \gamma^2}$, $b^i = \frac{\beta^j}{\beta^i \beta^j - \gamma^2}$ and $e = \frac{\gamma}{\beta^i \beta^j - \gamma^2}$

Firms set their prices. The Bertrand profit equations are :

$$\Pi^F = (P^F - v)q^F = (P^F - v)(a^F - b^F P^F + e P^W)$$

$$\Pi^W = (P^W - (s + c))q^W = (P^W - (s + c))(a^W - b^W P^W + e P^F)$$

Solving for the reaction functions :

$$r^F(P^W) = \frac{a^F + b^F v + e P^W}{2b^F} \quad (22)$$

$$r^W(P^F) = \frac{a^W + b^W(s + c) + e P^F}{2b^W} \quad (23)$$

Finding the interior solution for the Nash Equilibrium :

$$P_B^F = \frac{2b^W(a^F + vb^F) + e(a^W + b^W(s + c))}{4b^Fb^W - e^2}$$

$$P_B^W = \frac{2b^F(a^W + b^W(s + c)) + e(a^F + vb^F)}{4b^Fb^W - e^2}$$

The equilibrium price paid to poachers is determined by equating the quantity supplied by the trader in Bertrand duopoly and the quantity supplied by the poachers and yields the quantity supplied yields :

In the **Bertrand equilibrium** :

$$\text{Price paid to poachers } s_B^*(x) = \frac{2W_2b^W[b^F(2a^W + ev) + ea^F + c(e^2 - 2b^Wb^F)] + W_1\sigma x(4b^Fb^W - e^2)}{\sigma^2x^2(4b^Fb^W - e^2) + 2W_2b^W(2b^Fb^W - e^2)} \quad (24)$$

$$\text{Poaching : } q_B^{W*}(x) = \frac{b^W[\sigma^2x^2(b^F(2a^W + ev) + ea^F + c(e^2 - 2b^Wb^F)) - W_1\sigma x(2b^Fb^W - e^2)]}{2Wb^W(2b^Wb^F - e^2) + (4b^Fb^W - e^2)\sigma^2x^2} \quad (25)$$

We amend the original results from [Damania and Bulte \(2007\)](#) with the concurring Lemma 2:

Lemma 2: *With Bertrand competition, if the introduction of captive-bred products has no impact on the parameters of the demand function for wild animal products, poaching levels with captive breeding are ambiguous. The driver of the equilibrium is the cost ratio between aquaculture and the illegal poaching sector, i.e. v and $c + s(x)$*

- For relatively low ratio values (i.e. $c + s(x) >> v$), poaching is unambiguously lower than without captive breeding for any given wildlife stock
- For intermediate ratio values, poaching is larger (for $x < \tilde{x}$), then lower (for $x > \tilde{x}$), than without captive breeding (with \tilde{x} such that $q_B^{W*} = q_m^W$)
- For large values of unit farming costs, poaching is unambiguously larger than without captive breeding for any wildlife stocks

See appendix 2.A.5.2 for proof of Lemma 2.

Our results significantly differ from [Damania and Bulte \(2007\)](#), as Bertrand competition does not unambiguously lead to more extraction. Indeed, poaching functions are ambiguously ranked, and the final location of the steady state depends on the species intrinsic growth rate r and carrying capacity K .

With low farming costs, traders have an incentive to maintain large stocks. As the price paid to poachers is inversely related to the size of the stock, low harvest maintains large stocks and thus limits the price paid to poachers. Given its operational costs, it is the only way for the trader to remain competitive with the farming sector. On the other hand, when farming costs are large, the traders are incentivized to harvest more, as they can afford to pay a larger price to poachers while remaining competitive with the farming sector.

Corner solution: in a perfectly substitutable framework, a corner solution emerges if one firm has a lower marginal cost than the other: if farmed and wild animal products were perfect substitutes and farmed products unambiguously cheaper to produce, poaching would cease. In the

context of imperfectly substitutable goods, this result is challenged. For poaching to cease, it must be that :

$$v = -\frac{1}{e}(2(a^W - cb^W) - \frac{1}{b^F}(ce + a^F)) \quad (26)$$

In our setup, the marginal cost of production for farming would need to be **negative** for poaching to stop¹. Moreover, as substitutability increases, this cost lowers. The relative cost of trading poached goods plays a minor role.

2.A.3.5 Steady state equilibria

Given the inverted U-shape of the logistic growth function, several steady-state equilibria can arise. First, if the *harvest function* (that is increasing and concave) is *steeper* than the growth function at low stock levels, there can be (i) no equilibrium if the harvest at $K/2$ is larger than the growth rate, (ii) one bifurcation point (tangent harvest and growth functions at $K/2$, and (iii) two equilibria, with one stable and one unstable. If the *growth function* is *steeper* than the growth function at low stock levels, there can be (i) a single equilibrium, (ii) a bifurcation point and an equilibrium, (iii) three interior equilibrium, with only two being stable (see figure 1 for an illustration)

2.A.4 Extensions

2.A.4.1 An oligopoly model

We extend our model to gauge the impact of the number of traders and farmers. We denote by \mathcal{I} the set of individual traders $i \in \mathcal{I}$ and by \mathcal{J} the set of individual farmers $j \in \mathcal{J}$. The demand functions are :

$$P_k^W = \alpha^W - \beta^W \sum_{i \in \mathcal{I}} q_i^W - \gamma \sum_{j \in \mathcal{J}} q_j^F \quad (27)$$

$$P_l^F = \alpha^F - \beta^F \sum_{j \in \mathcal{J}} q_j^F - \gamma \sum_{i \in \mathcal{I}} q_i^W \quad (28)$$

Cournot oligopoly Each farmer and trader maximizes profits by taking as given its competitors' quantity commitments. We assume traders and farmers are homogeneous, i.e for each type of producer, costs are identical :

$$i, j \in \mathcal{I}, i \neq j, c_i = c_j = c$$

$$k, l \in \mathcal{J}, k \neq l, v_k = v_l = v$$

Assuming that $\text{card}(\mathcal{I}) = N$ and $\text{card}(\mathcal{J}) = M$, the profit functions for each farmer and trader can be written as :

$$\Pi_i^W = \left(\alpha^W - \beta^W(N-1)q_i^W - \beta^W q_i^W - \gamma M q^F - s - c \right) q_i^W \quad (29)$$

$$\Pi_k^F = \left(\alpha^F - \beta^F(M-1)q_k^F - \beta^F q_k^F - \gamma N q^W - v \right) q_k^F \quad (30)$$

Where q_i^W denotes the quantities sold by all other traders different from trader k (and q_i^F for farmers different from farmer i). Given that all players in each type are identical cost-wise, the

¹If consumers enjoy a numeraire good, they must receive compensation to consume the farmed good such that they increase their numeraire consumption to make up for the imperfectly substitutable nature of the farmed good.

reaction functions are :

$$\forall i, j \in \mathcal{I} : q_i^W = q_j^W = q^W = \frac{\alpha^W - (s + c) - \gamma M q^F}{(N + 1)\beta^W} \quad (31)$$

$$\forall k, l \in \mathcal{J} : q_k^F = q_l^F = q^F = \frac{\alpha^F - v - \gamma N q^W}{(M + 1)\beta^F} \quad (32)$$

The **Cournot-Nash equilibrium** is :

$$\text{Poaching} : q_{\text{Cournot}}^W = \frac{\beta^F(M + 1)(\alpha^W - (s + c)) - \gamma M(\alpha^F - v)}{\beta^W \beta^F (M + 1)(N + 1) - \gamma^2 NM} \quad (33)$$

$$\text{Farming} : q_{\text{Cournot}}^F = \frac{\beta^W(N + 1)(\alpha^F - v) - \gamma N(\alpha^W - (s + c))}{\beta^W \beta^F (M + 1)(N + 1) - \gamma^2 NM} \quad (34)$$

(35)

The primary market (between poachers and traders) must clear, and $s(x)$ equates supply and demand:

$$Nq_{\text{Cournot}}^W = q^W \quad (36)$$

$$\iff s^{C^*}(x) = \frac{2W_2N[\beta^F(M + 1)(\alpha^W - c) - \gamma M(\alpha^F - v)] + W_1\sigma x(\beta^F\beta^W(M + 1)(N + 1) - \gamma^2 NM)}{\sigma^2 x^2[\beta^F\beta^W(M + 1)(N + 1) - \gamma^2 NM] + 2W_2N(M + 1)\beta^F} \quad (37)$$

Solving for the equilibrium quantity, the quantity supplied on the market by individual traders is :

$$q_{\text{Cournot}}^W = \frac{\sigma^2 x^2 [\beta^F(M + 1)(\alpha^W - c) - \gamma M(\alpha^F - v)] - \sigma x W_1 N (M + 1) \beta^F}{\sigma^2 x^2 (\beta^F \beta^W (M + 1) (N + 1) - \gamma^2 NM) + 2W_2 N (M + 1) \beta^F} \quad (38)$$

In our case study, when $c = 0$, it shows that when the number of farmers is larger than the number of traders, the introduction of farming generates larger steady-state stocks. An interesting perspective is when there remains 1 sole trader, and the number of farmers increases: in this case, poaching is drastically cut down, as shown in Figure 2.A.1. When the number of traders is larger than the number of farmers, steady-state stocks decrease. In our context, when the number of traders is limited, increasing the number of farming facilities is a safe way to guarantee conservation outcomes.

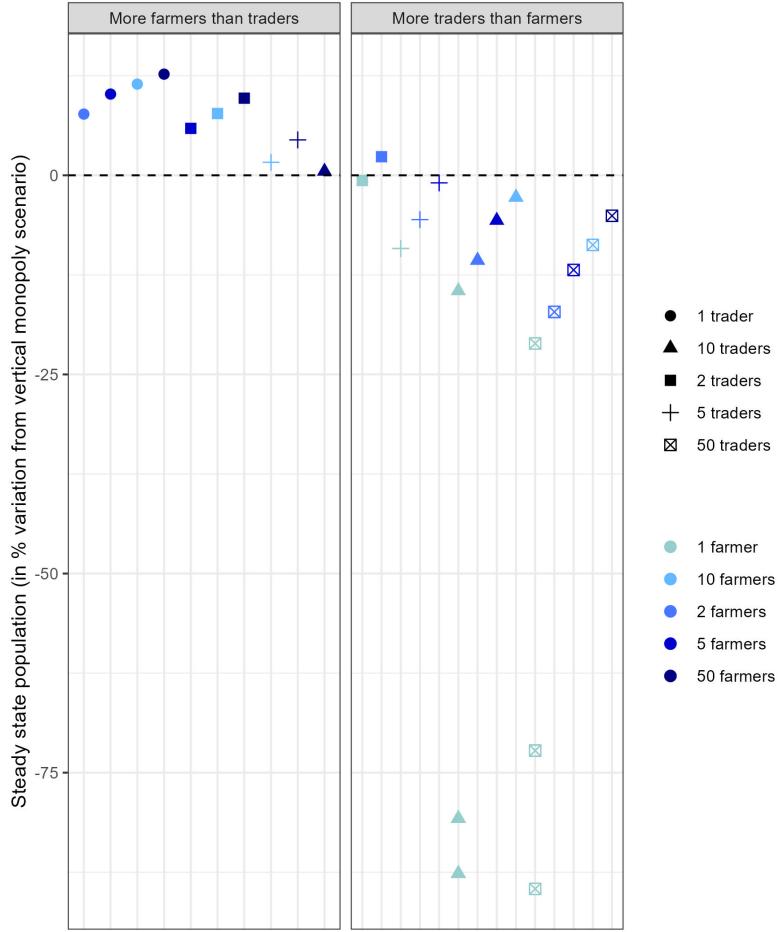


Figure 2.A.1: Steady state outcomes when multiple traders and multiple farmers are considered (an oligopoly) in the quantity adjustment scenario.

The left panel shows the steady state of the wild *Totoaba macdonaldi* population when there are more farmers than traders. The right panel shows the steady state of the wild population when there are more traders than farmers

Bertrand oligopoly Using the same notations as previously, the demand functions can be written as :

$$\forall i \in \mathcal{I} : q_i^W = q^W = \frac{1}{N}(a^W - b^W P^W - eP^F) \quad (39)$$

$$\forall j \in \mathcal{J} : q_j^F = q^F = \frac{1}{M}(a^F - b^F P^F - eP^W) \quad (40)$$

Using these demand functions and solving for the reaction functions in each case yields :

$$r^F(P^W) = \frac{a^F + b^F v + eP^W}{2b^F} \quad (41)$$

$$r^W(P^F) = \frac{a^W + b^W(s + c) + eP^F}{2b^W} \quad (42)$$

These reaction functions are the same as in the duopoly case (see eq. 23). This result shows that aggregate production is invariant to the number of farmers or traders as long as both are present on the market. Moreover, the individual production for traders is $\frac{1}{N}q_B^W$ and $\frac{1}{M}q_B^F$ with q_B^W and q_B^F referring to the duopoly equilibrium quantities for poached and farmed productions. In a Bertrand equilibrium, irrespective of the number of players, price-setting competition pushes the price to its minimum such that both firms still operate (given that traders have a stock-dependent production cost). Increased competition in the form of more players cannot push the prices fur-

ther down. Therefore, aggregate output remains the same and individual production is divided among players.

This result further contradicts the results in [Damania and Bulte \(2007\)](#), as the authors find that increasing the number of players in a Bertrand set-up has detrimental effects on the steady-state stock. We find no effect, consistent with the theory and intuition.

2.A.4.2 Trader take over of the aquaculture sector

In this section, we look at the 'extended cartel' scenario, where the vertical monopoly takes over the ownership of the aquaculture firm.

To gain intuition, assume poached and farmed products are perfect substitutes. On the one hand, the vertical monopoly has two production technologies: poaching (with a variable marginal cost, as the price paid to poachers depends on the population stock) and farming (with a constant marginal cost). In this case, the vertical monopoly equates the marginal costs across production units; that is, it buys a poached product to poachers up until the marginal cost of an extra poached unit equates to that of a farmed unit. In this case, if the marginal cost of farming is lower than market prices absent farming, then poaching goes down. Notice that the only way for traders to limit the price paid to poachers is to maintain a healthy stock. Therefore, the new equilibrium population stock is larger than the initial stock, and poaching is lower.

Now consider the case at stake, where products are imperfect substitutes. In this case, the extended cartel does not only equate marginal costs, as marginal revenues diverge across products. We use the following model to investigate the resulting equilibrium. Let the profit of the extended cartel be:

$$\Pi(q^F, q^W) = (\alpha^W - \beta^W q^W - \gamma q^F - (s + c))q^W + (\alpha^F - \beta^F q^F - \gamma q^W - v)q^F \quad (43)$$

The extended cartel maximizes its profit with respect to the poached and farmed products. The poached production it sells on end markets is :

$$q^W = \frac{\sigma^2 x^2 (\beta^F (\alpha^W - c) - \gamma (\alpha^F - v)) - W_1 \beta^F \sigma x}{2(\beta^F W + \sigma^2 x^2 (\beta^F \beta^W - \gamma^2))} \quad (44)$$

Figure 2.A.2 shows that if the 'extended cartel' scenario arises, poaching goes down, and the steady-state population increases.

2.A.5 Appendices

2.A.5.1 Lemma 1 : content and proof

Assume $\alpha^W = \alpha^m$ and $\beta^m = \beta^W$, i.e., that the demand faced by the monopolist is the same as in the duopolistic case. Comparing monopoly and Cournot harvest functions:

$$\begin{aligned} q_m^W &\geq q_c^W \\ \Rightarrow v &\leq \bar{v} = \alpha^F - \frac{\gamma(\alpha^m - c)\sigma^2 x^2 - W_1 \sigma x}{2\beta^m \sigma^2 x^2 + 2W} \end{aligned}$$

First, look at when $x \rightarrow 0$:

$$\lim_{x \rightarrow 0} \bar{v} = \alpha^F$$

This requires that farming costs are lower than the choke price for consumers on their market. This condition is necessary for a farm competitor to enter the market.

Second, acknowledge that the second part of the equation is weakly decreasing, but non-increasing.

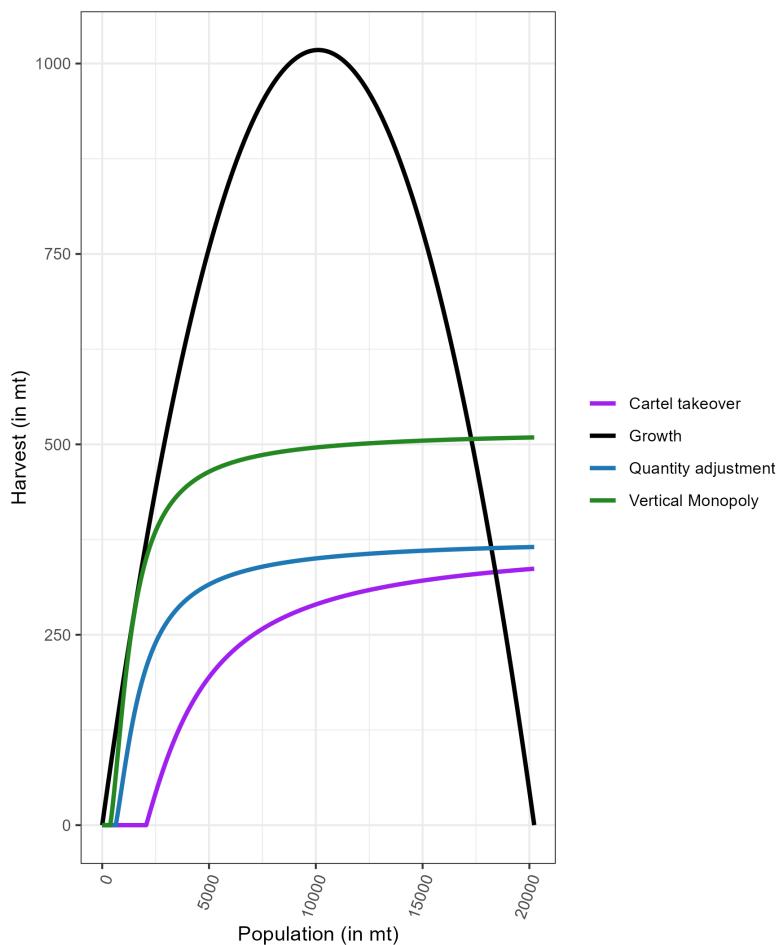


Figure 2.A.2: Steady-state equilibrium for the wild stock of *Totoaba macdonaldi* in the ‘extended cartel’ scenario, where the vertical monopoly takes over the ownership of farming operations

Assuming the carrying capacity goes to infinity, it is limited by :

$$\lim_{x \rightarrow \infty} \bar{v} = \alpha^F - \gamma \frac{(\alpha^m - c)}{2\beta^m}$$

As fish abundance increases, the price paid to poachers decreases, as there is less scarcity. From equation (20), when $x \rightarrow \infty$, the price paid to poachers drops to 0. Moreover, notice that the last term in parenthesis is equation (9) for $s = 0$. Therefore, it means that the residual willingness to pay, when the poachers behave like a monopoly and $x \rightarrow \infty$, is larger than the unit cost of farming.

If the market is truly duopolistic, in the sense that the poachers could not manage the stock such that they depress demand so much as to kick their competitor out of the market, then Cournot competition unambiguously leads to lower poaching levels than a monopoly does.

2.A.5.2 Lemma 2

Assume that the demand parameters are unchanged by the introduction of farmed substitutes, that is to say $\alpha^W = \alpha^m$ and $\beta^W = \beta^m$, and use the definition of the coefficients for the direct demand function:

$$\begin{aligned} a^j &= \frac{\alpha^j \beta^i - \alpha^i \gamma}{\beta^j \beta^i - \gamma^2}; & b^j &= \frac{\beta^i}{\beta^j \beta^i - \gamma^2} \\ a^m &= \frac{\alpha^m}{\beta^m}; & b^m &= \frac{1}{\beta^m} \end{aligned}$$

For $i, j \in \{W, F\}$ and m the monopoly case. To establish Lemma 2, we compare q_B^W and q_m^W . Equation (11) can be rewritten as :

$$q^m(a^m, b^m) = \frac{\sigma^2 x^2 (a^m - b^m c) - b^m W_1 \sigma x}{2\sigma^2 x^2 + 2W b^W}$$

Therefore:

$$\begin{aligned} q_m^W &\geq q_B^W \\ \Rightarrow v &\leq \frac{a^m - b^m c}{b^W b^F e} \left[\frac{2W_2 b^W (2b^F b^W - e^2) + (4b^F b^W - e^2) \sigma^2 x^2}{2\sigma^2 x^2 + 2b^m W_2} \right] - \frac{W_1 \sigma x [(4b^F b^W - e^2)(b^m - b^W) + e^2 b^W]}{b^W b^F e (2\sigma^2 x^2 + 2b^m W_2)} \\ &\quad - \frac{e a^F + c(e^2 - 2b^W b^F) + 2b^F a^W}{b^F e} \end{aligned}$$

Notice that this equation can be reframed as :

$$F(x|c) \geq v \text{ where } F(x|c) = \Phi \frac{\eta + \mu x^2}{\theta + \nu x^2} - \frac{\kappa x}{\omega x^2 + \epsilon} - \zeta$$

And :

$$\begin{aligned} \Phi &= \frac{a^m - b^m c}{b^W b^F e}, \quad \eta = 2W_2 b^W (2b^W b^F - e^2), \quad \mu = (4b^W b^F - e^2) \sigma^2, \\ \theta &= 2W_2 b^m, \quad \nu = 2\sigma^2, \quad \zeta = (e a^F + c(e^2 - 2b^W b^F) + 2b^F a^W) \end{aligned}$$

$$\begin{aligned} \kappa &= \frac{W_1 \sigma [(4b^F b^W - e^2)(b^m - b^W) + e^2 b^W]}{b^F * e}, \\ \omega &= 2b^W b^F e \sigma^2 \text{ and } \epsilon = 2b^m b^W b^F e W_2 \end{aligned}$$

Analysis of $\Phi_{\theta+\nu x^2}^{\eta+\mu x^2}$: if $\mu\theta - \nu\eta < 0$, the first component of $F(x|c)$ is decreasing:

$$(4b^W b^F - e^2)b^m - 2(b^W b^F - e^2)b^W < 0$$

$$\iff \frac{\gamma^2}{\beta^m(\beta^W \beta^F - \gamma^2)^3} [\beta^m \beta^F + \gamma^2 - 4\beta^F \beta^W] < 0$$

Under the assumption that $\beta^m = \beta^W = \beta^F = \beta$, it is clear that

$$\frac{\gamma^2}{\beta(\beta^2 - \gamma^2)}(\gamma^2 - 3\beta^2) < 0$$

as $\gamma < \beta$. Therefore, $\Phi_{\theta+\nu x^2}^{\eta+\mu x^2}$ is *decreasing* $\forall x$

Analysis of $\frac{\kappa x}{\omega x^2 + \epsilon}$: the second component of $F(x|c)$ is increasing for $x \leq \sqrt{\frac{\epsilon}{\omega}}$, and decreasing after, since $x \in \mathbb{R}^+$. Noticing that $\kappa < 0$:

- For $x \in [0, \frac{1}{\sigma}\sqrt{W_2 b^m}]$, $\frac{\kappa x}{\omega x^2 + \epsilon}$ is negative and decreasing
- For $x > \frac{1}{\sigma}\sqrt{W_2 b^m}$, $\frac{\kappa x}{\omega x^2 + \epsilon}$ is negative and increasing

Conclusion Overall, $F(x|c)$ is such that :

- For $x \leq \frac{1}{\sigma}\sqrt{W_2 b^m}$, the first component is decreasing, while the second component is increasing
- For $x \geq \frac{1}{\sigma}\sqrt{W_2 b^m}$, the first component is decreasing and the second component is decreasing

Hence, $F(x|c)$ is bounded above by $\max(F(0|c), F(\frac{1}{\sigma}\sqrt{W_2 b^m}|c))$, and bounded below by $F(K|c)$ where K is the system carrying capacity. Therefore:

1. If $v < F(K|c)$, then Bertrand harvest is always lower than monopoly harvest
2. If $F(K|c) < v < F(0|c)$, then Bertrand harvest starts by being lower than in the monopoly case, but gets larger for large stock values.
3. Eventually, if $F(0|c) < v$, then Bertrand harvest is always larger than in the monopoly case

Figure 2.A.3 illustrates this lemma with our parameter specification.

Corner equilibrium: for a corner solution to emerge, it must be that $q_B^{w*} = 0$,

$$v = v(x) = \frac{W_1(2b^F b^W - e^2)}{\sigma x b^F e} - \frac{2b^F a^W + e a^F + c(e^2 - 2b^W b^F)}{b^F e} \quad (45)$$

Equation 45 shows that for low stock values, costs can still be positive and poaching disappear. However, to ensure that poaching is *never* beneficial in the Bertrand equilibrium, it must be that $v = \min v(x) = -\frac{2b^F a^W + e a^F + c(e^2 - 2b^W b^F)}{b^F e v}$. In this case, the subsidy rate is so high that production is always beneficial for the farmer, and prices are too low for the trader to compete. In our baseline specification, this would amount to $v = -720,855$ USD.

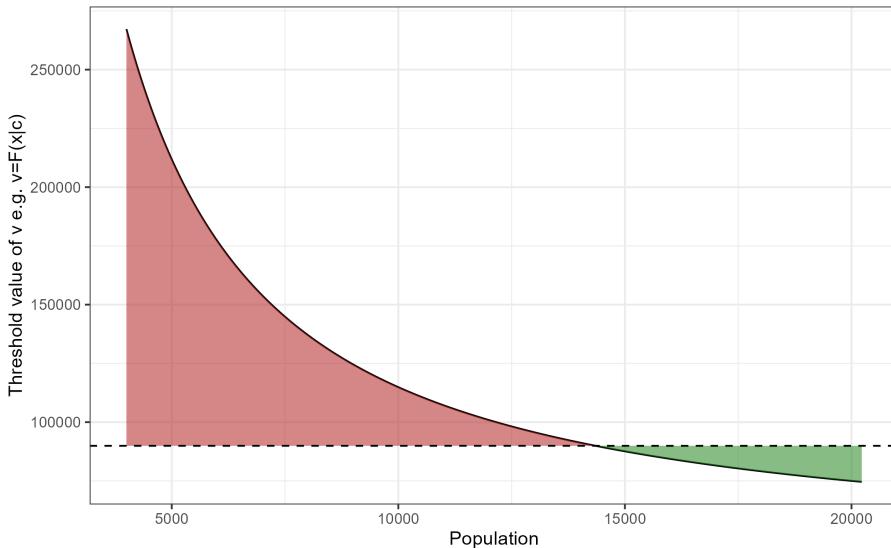


Figure 2.A.3: Evolution of the threshold v to compare vertical monopoly and price setting (Bertrand) harvest functions

In green, vertical monopoly harvests more than in the price setting equilibrium. For larger population values, in red, price setting leads to more harvest than the vertical monopoly. This illustrates our main specification and property 2 above.

Supplementary Figures and Tables

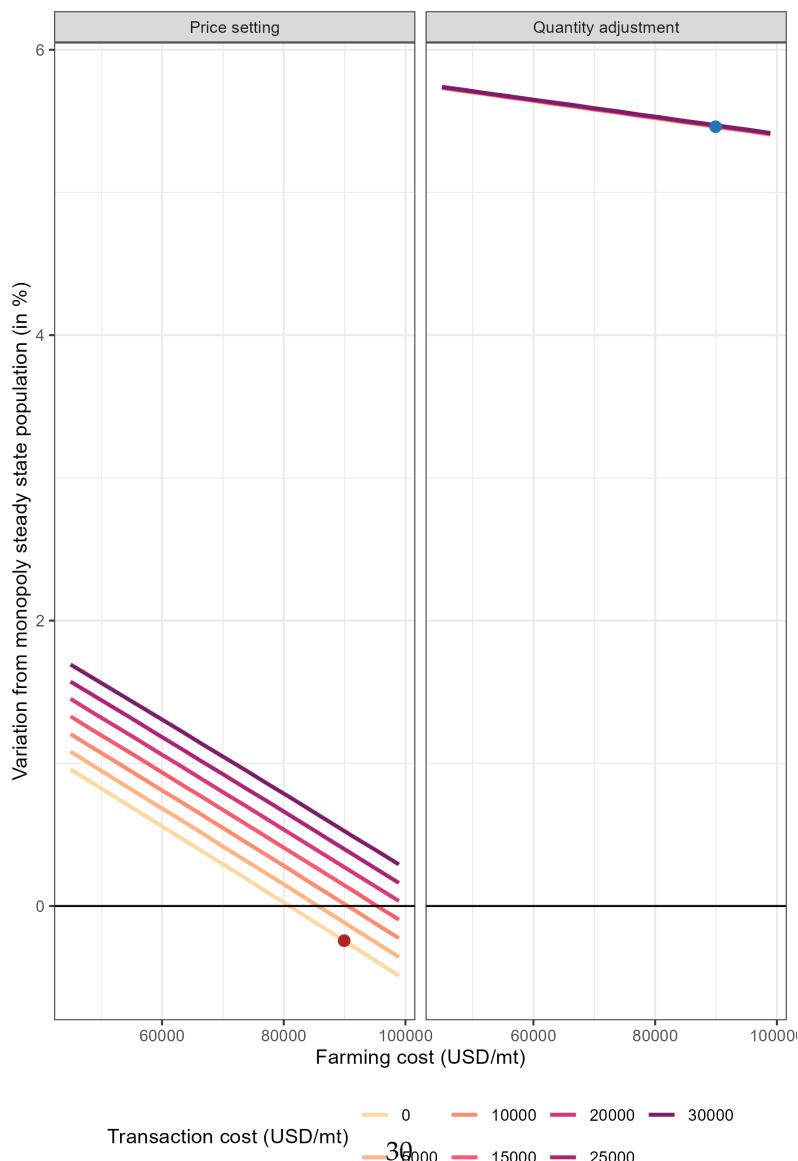


Figure 2.A.4: Percent change in steady state population across scenarios, following the joint evolution of

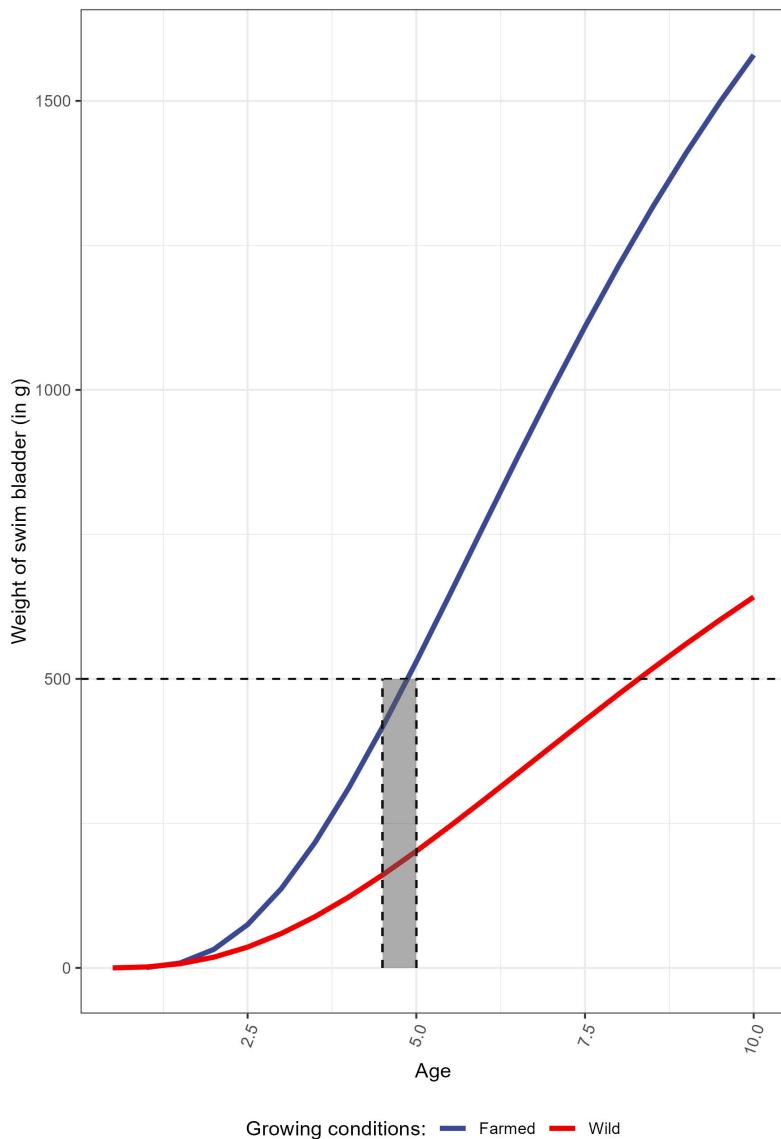


Figure 2.A.5: Von Bertalanffy Growth curves for wild and farmed *Totoaba macdonaldi* under different growing conditions

Gray box indicates the range of ages that possess a 500 gram swim bladder. The wild individual growth curve was calibrated with information from the stock assessment, while the farmed individual growth curve was calibrated using

Variable	Low Season	Mid Season	High Season	Source
Vessels	5	20	50	Cisneros-Mata (2020)
Days per month	4	12	14	Cisneros-Mata (2020)
Total fleet days year	20	240	700	Cisneros-Mata (2020)
Food fuel day	525	525	525	Semi-Structured Interviews
Totoaba gearset	2	3	6	Cisneros-Mata (2020)
Gear loss day	0.5	0.5	0.5	Semi-Structured Interviews
Gearset vessel per day	2	3	3	Cisneros-Mata (2020)
Gear replacement	1600	1600	1600	Semi-Structured Interviews
Bribes/year	600	7200	21000	Semi-Structured Interviews
Average cost (per vessel day)	8385.34	14386.69	5051.26	Authors' calculation

Table 2.A.6: Supporting information for the calculation of the *Totoaba macdonaldi* poaching cost parameters (W_1 and W_2)

The methods section details how and when semi-structured interviews were conducted.

<i>Dependent variable:</i>	
Price	
Catch	-1,563.752** (725.985)
Constant	1,625,837.000*** (406,789.500)
Observations	45
R ²	0.097
Adjusted R ²	0.076
Residual Std. Error	431,737.700 (df = 43)
F Statistic	4.640** (df = 1; 43)

Note: *p<0.1; **p<0.05; ***p<0.01

Table 2.A.7: Regression output for the linear demand estimation calculated by regressing price data on catch data.

Data were obtained from the available literature that provided estimated weight and value of *Totoaba macdonaldi* maw seizures on estimated *Totoaba macdonaldi* catch from 2014 to 2017 obtained from a recent stock assessment. The methods section details where information was obtained from.

Variable	Value	Source
Sphere	1.00	Earth Ocean Farm Video, 2022
Capacity per sphere (t)	144.00	Earth Ocean Farm Video, 2022
<i>In \$USD</i>		
Maintenance year	12500.00	Felipe Ramirez, InnovaSea, 2018
Cleaning year	5000.00	Felipe Ramirez, InnovaSea, 2018
Vessel maintenance/year	10000.00	Tyler Korte, BlueOcean Mariculture, 2018; Fernando Cavalin, Earth Ocean Farms, 2018
Fuel year	25122.50	Author's Calculations
Feed	312480.00	Tyler Korte, BlueOcean Mariculture, 2018
Labor	1580000.00	Authors' calculations
Facility lease	150000.00	Cygnus Ocean Farms, 2017
Admin.	50000.00	Cygnus Ocean Farms, 2017
Operational costs	2145102.50	Authors' calculations
Operational costs (per t & year)	14896.55	Authors' calculations

Table 2.A.8: Supporting information for the calculation of the *Totoaba macdonaldi* farming cost parameter (v)

Annual cost estimates were obtained from informants and converted to \$USD. Capacity of each farming pen was obtained from Earth Ocean Farms, and an annual cost 706 per tonne of totoaba was calibrated using personal communications with totoaba aquaculture producers.

Parameter	Value	Concept	Units
α	1,625,836.98	Demand model : intercept	USD
β	1,563.75	Demand model : coefficient	USD/metric ton of biomass
γ	1,354.25	Demand model : substitutable good coefficient	USD/metric ton of biomass
r	0.20	Intrinsic growth rate	unitless
K	20,226.00	Carrying capacity (in metric tons)	metric tons of biomass
σ	2×10^{-5}	Catchability	% of biomass/vessel trip
$AvgCost$	14,386.69	Average cost per vessel trip at historical value	USD/vessel trip
W	3.75	Quadratic cost parameter - Quadratic cost function	USD vessel trip ⁻²
W_1	12200.00	Linear cost parameter - Linear quadratic cost function	USD/vessel trip
W_2	0.57	Quadratic cost parameter - Linear quadratic cost function	USD vessel trip ⁻²
v	89929.92	Unit cost of farming	USD/metric ton of biomass
i_r	0.10	Interest rate	%
Age	4.50	Age of farmed totoaba	Years
c	0.00	Unit cost of trading	USD/ metric ton of biomass

Table 2.A.9: Summary of *Totoaba macdonaldi* ecological and market parameters for model calibration

The methods section details where information was obtained to estimate each parameter, as well as relevant equations.

Concept	Formula	Reference
<i>Fishery</i>		
Growth	$\dot{x} = rx(1 - \frac{x}{K}) - \sigma xE$	eq. 4
<i>Poaching</i>	s is price paid to poachers	
Harvest technology	$q = \sigma xE$	
Profit	$\Pi = s \times (\sigma xE) - W_1 E - W_2 E^2$	
Poached harvest	$q^W = \frac{s\sigma^2 x - W_1}{2W_2}$	eq. 6
<hr/>		
<i>Vertical monopoly scenario</i>		
Demand	$P^m = \alpha^m - \beta^m q$	eq. 7
Profit	$\Pi^m = (P^m - s - c)q$	eq. 8
Supply on end market	$q_m^*(x) = \frac{\sigma^2 x^2 (\alpha_m - c) - W_1 \sigma x}{2(\sigma^2 x^2 \beta^m + W_2)}$	eq. 11
<hr/>		
<i>Duopoly</i>		
Aquaculture profit	$\Pi^F = (P^F - v)q^F$	eq. 12
Demand for imperfect substitutes	$P^W = \alpha^W - \beta^W q^W - \gamma q^F$	eq. 14
	$P^F = \alpha^F - \beta^F q^F - \gamma q^W$	eq. 15
Quantity adjustment (Cournot) supply	$q_C^{W*}(x) = \frac{\sigma^2 x^2 (2\beta^F (\alpha^W - c) - \gamma (\alpha^F - v)) - 2\beta^F W_1 \sigma x}{4W_2 \beta^F + \sigma^2 x^2 (4\beta^W \beta^F - \gamma^2)}$	eq. 21
Price setting (Bertrand) supply	$q_B^{W*}(x) = \frac{b^W [\sigma^2 x^2 (b^F (2a^W + ev) + ea^F + c(e^2 - 2b^W b^F)) - W_1 \sigma x (2b^F b^W - e^2)]}{2Wb^W (2b^W b^F - e^2) + (4b^F b^W - e^2) \sigma^2 x^2}$	eq. 25

Table 2.A.10: Summary of the key functions in the model

For model conclusions, the plotted functions are growth, vertical monopoly end market supply (q^m), quantity adjustment end market supply (q_C^W) and price setting end market supply (q_B^W)

Chapter 3

The wildland connectivity dilemma : a graph theoretical computational approach

Abstract

Background: Fuel treatment operations help to mitigate the spread and severity of wild-fires in numerous ecosystems. As they aim at fragmenting the fire landscape, they also fragment wildlife habitat. This poses a dilemma for land managers, in the form of a trade-off between lowering wildfire patch connectivity and maintaining wildlife habitat connectivity. Previous studies have investigated the spatial allocation of fuel treatments over time, mostly without specific care devoted to biodiversity, in a variety of case studies. However, they lack generality and an interpretative framework. We use dynamic programming and graph theory on every possible theoretical landscape configuration to gain a general understanding of the allocation of treatments over space and time and the corresponding landscape properties with various habitat connectivity targets.

Results: Our results show that all initial landscapes converge to steady-state landscape cycles. Moreover, we show that there exist optimal trajectories that significantly reduce wildfire risk while safeguarding habitat connectivity. As the policy budget increases, more risk reduction is achieved, albeit with a decreasing marginal efficiency, and more steady-state cycles emerge. As habitat targets increase, increasing the budget is of no effect, and risk increases, while the number of steady-state cycles decreases. Landscapes are less risky, more fragmented, and diverse when the budget is large and biodiversity targets are low, while they are more compact and less diverse when the opposite is true. Treatment allocation follows graph centrality measures, and central cells are treated first. When the budget increases, fewer central cells (i.e. edge patches) are treated as well. When biodiversity targets increase, central cells are no longer treated as they decrease habitat connectivity. Treatment is reshuffled to the edges of the landscape.

Conclusion: Computational experiments generalize existing results. Using graph theory, general insights can be gained, and help managers faced with multiple objectives in forested landscapes. From a policy perspective, in the face of climate change, increasing treatment budgets should be a priority to avoid increasing damages. A key guideline is treating a variety of seral stages to create landscape diversity, mitigate risk and guarantee the connectivity of wildlife habitat.

Keywords : Fuel treatment, connectivity, wildfire risk, wildlife habitat, spatial optimization, graph theory

3.1 Introduction

Hazardous and intense wildfires threaten forest resilience and can cause ecosystem shifts (Coop et al., 2020). Moreover, intense wildfires cause human damages, in the form of direct asset losses: in 2018, wildfires in California have caused \$ 27 billion (Wang et al., 2021). Indirect costs are also of concern, especially related to wildfire smoke (increase in PM 2.5 concentrations have important health impacts (Burke et al., 2023; Heft-Neal et al., 2023), recreation values are affected in the US, amounting to \$USD 2.3 billion (Gellman et al., 2023)). Aside from directly measurable costs, they also cause dramatic impacts on biodiversity across taxa (Wintle et al., 2020). Global warming affects water supply and fuel moisture (Jolly et al., 2015; Abatzoglou and Williams, 2016; Ruf-fault et al., 2018), and is projected to increase the frequency, severity, and magnitude of wildfires (Wasserman and Mueller, 2023). Additionally, large wildfires are of importance in the face of climate change releasing a lot of greenhouse gas and reducing the atmospheric carbon sinks (Zheng et al., 2023; Sweeney et al., 2023). Recent wildfire events in California (since 2018), in Australia (2019-2020), and in Europe (France, Portugal, Greece in 2022) have epitomized these trends.

In numerous regions, such as conifer forests in California (Vaillant et al., 2009; Kalies and Yocom Kent, 2016; Low et al., 2023), eucalypt forests in South Western Australia (Burrows and McCaw, 2013; Boer et al., 2009; Florec et al., 2020), southern Europe (Fernandes et al., 2013), evidence shows that fuel treatments (e.g. prescribed burns, mechanical thinning and managed wildfires), can mitigate wildfire intensity and spread. Land management agencies have historically implemented these policies in Australia (Burrows and McCaw, 2013), Europe, and the United States (and are projected to ramp up, for example under the Infrastructure Investment and Jobs Act of 2021 in the US). Understanding the spatial allocation of treatments, as climate change impacts negatively both costs and feasibility, is a major driver of policy success (Williams et al., 2017; Florec et al., 2020). **Idea:** land is public, and massive externality/costs + free rider problem + information requirements warrant a public policy approach to this issue. One of the drivers of its success is space.

By changing the structure of the landscape, fuel management operations also affect the structure of biodiversity habitat, notably, its structural connectivity (Taylor et al., 1993). Maintaining habitat connectivity, through wildlife corridors, landscape links, and ecoducts (Turner, 2005; Turner and Gardner, 2015), is instrumental in mitigating the biodiversity crisis. Species richness and diversity are intimately linked to landscape connectivity (Olds et al., 2012; Tian et al., 2017; Velázquez et al., 2019) and are necessary to maintain ecosystems in the future. The impact of fuel treatments on biodiversity remains a debated topic. Evidence suggests that maintaining a variety of vegetation types and ages on a patchy landscape maintains a 'fire mosaic' (Sitters et al., 2015) (e.g. landscape level variations in habitat types that provide habitat to an ecological community) or that fuel treatment can be beneficial to wildlife (Saab et al., 2022; Loeb and Blakey, 2021) and even restore local populations (Templeton et al., 2011). On the other hand, treating at too high a frequency may be detrimental to biodiversity (Bradshaw et al., 2018). Overall, implementing fuel treatment challenges the connectivity of wildlife habitat. In this context, understanding the trade-offs between risk reduction and biodiversity conservation, as well as the spatial patterns of operations that could reconcile the two objectives is key. In this study, we investigate the spatial allocation of fuel treatments to optimally reduce wildfire risks while maintaining biodiversity habitat.

A substantial literature has applied optimization techniques to tackle the spatial allocation of fuel treatments. Analytical (Finney, 2001), simulation-based (Finney, 2007; Rytwinski and Crowe, 2010) or mixed-integer programming techniques (Wei et al., 2008) have solved the allocation of treatments in a static framework. Given the dynamic nature of fuel growth, studies based on

mixed-integer dynamic programming ([Wei et al., 2008](#); [Minas et al., 2014](#); [Rachmawati et al., 2015, 2016](#)) have studied the temporal and spatial allocation of fuel treatments on real and simulated landscapes. While they solve the spatial treatment allocation problem in forests, these articles fail to acknowledge the multiple uses and objectives land planners have to consider, such as habitat conservation. Several articles have devoted their attention to the spatial allocation of treatments while conserving habitat, and investigated the trade-offs between risk reduction and biodiversity conservation, using spatial heuristics ([Calkin et al., 2005](#); [Lehmkuhl et al., 2007](#)) and linear programming ([Williams et al., 2017](#); [Rachmawati et al., 2018](#)). Most of the existing literature focuses on case studies and lacks a general interpretative framework to generalize its results. Graph theory offers a toolbox suited to analyze the properties of connected patches of land with varying characteristics, and has extensively been applied in landscape ecology ([Urban and Keitt, 2001](#); [Minor and Urban, 2008](#); [Rayfield et al., 2016](#)). Recent research focusing on the allocation of fuel treatments has leveraged tools from graph theory ([Matsypura et al., 2018](#); [Pais et al., 2021a](#)). Reconciling habitat and wildfire risk mitigation using graph theory is a recent research endeavor ([Rachmawati et al., 2018](#); [Yemshanov et al., 2022](#)) and has focused on specific case studies.

In this article, we leverage graph theory on an exhaustive set of theoretical landscapes to study the general patterns of treatment allocation emerging from a multi-objective, dynamic, and integer landscape management problem, governed by connectivity. We analyze all the landscape configurations resulting from a 20-period planning horizon, for regular grid landscapes, in a graph theoretical perspective. In doing so, we examine the fuel treatment patterns resulting from all the range of habitat connectivity, in order to characterize long-term landscape properties. We characterize the landscapes using a range of ecological indicators and find general mechanisms and guiding principles applicable to a broad class of settings, to guide decision-makers and foster new efficient multi-objective graph theory algorithms.

Our contributions are several. First, we provide a spatial framework to understand the trade-offs between wildfire risk reduction and biodiversity conservation. Using graph theory, we derive general principles regarding the spatial characteristics of landscapes and treatments from an exhaustive set of theoretical landscapes to guide policymakers as well as future research in heuristics to reconcile conflicting land-based phenomena. Eventually, we characterize the risk and biodiversity profiles consistent with a changing climate, where windows of opportunity are shorter and costs of treatment larger, and the associated spatialized treatments.

3.2 Methods

3.2.1 Theoretical model

We consider theoretical landscapes represented by a regular grid of $n \times n$ cells with a forest seral stage succession module. We use a stylized representation of the link between vegetation age, habitat, and wildfire risk. We denote by A_t the set of equal, standardized area cells in the theoretical landscape of dimension $n \times n$ (hereafter referred to as being of size = n) in period t . Each cell a_i at time t is characterized by a seral stage: absent, young, or old. At each time step, it changes stage until it is in the 'old' stage, where it remains. Upon treatment, a cell's seral stage is set to 'absent' (see equation A.1 in appendix A).

A cell offers wildlife habitat once it is 'mature' (eg seral stage is at least 'young'), i.e, when the time elapsed since the last burn reaches the maturity threshold (eq. A.2). We assume that habitat quality is uniformly distributed among habitat patches and that neighboring cells are reachable, conditional on being 'mature'. After the wildlife habitat maturity threshold, a cell can turn at critical risk of wildfire during a 'normal' hot season. We assume an Olsen-type model

of flammability (Olson, 1963; McCarthy et al., 2001), where age is the main predictor. Therefore, after the ‘high fuel load’ threshold is crossed, the cell is regarded as ‘high risk’ from then on, until treatment suppresses this risk (eq. A.3).

We define cells to be connected if (i) they are within an 8-cell neighborhood and (ii) share the same status. Regarding biodiversity, we focus on general characteristics related to landscape structural connectivity rather than functional connectivity, as we are agnostic about effective species (Fahrig et al., 2011). We assume that species are able to disperse from one patch to another, and that habitat quality is uniformly distributed conditional on habitat being available. We consider the wildfire risk through the lens of potential spread, which is only driven by fuel. Consistent with the literature (see Peterson et al. (2009), Pais et al. (2021b); Gonzalez-Olabarria et al. (2023)), a wildfire can spread in any direction, conditional on neighbor cells with high risk. However, if surrounding cells do not display high risk, fire does not spread.

We use a network structure to apprehend the landscapes. We transform A_t the set of cells constituting the landscape into graphs G_t whose vertices V_t (or nodes) are the cells in the landscape, and edges E_t represent the connections between cells. We partition the landscape in two graphs, G_{B_t} and G_{F_t} , each describing the network of mature habitat and risky patches (see fig. 1 for a representation). Landscape ecology has long used numerous, theoretically grounded indicators to analyze landscapes (Urban and Keitt, 2001; Minor and Urban, 2008). We use a global connectivity indicator that satisfies Pascual-Hortal and Saura (2006) criteria, grounded in graph theory, that offer a reformulation of Rachmawati et al. (2016) (see Appendix A.3).

We define the global connectivity index of habitat and risky patches in landscape $A(t)$ as:

$$H_i(A(t)) = \text{card}(V_{i_t}) + 2 \times \text{card}(E_{i_t}) \text{ with } i \in \{B, F\} \quad (3.1)$$

This indicator considers that a habitat patch is connected to itself (i.e, within a habitat patch, there is no barrier) and whether it is connected to other patches. It implies lower connectivity when the distance between patches increases, attains its maximum value when a single habitat patch covers the whole landscape, indicates lower connectivity as the habitat is progressively more fragmented, considers negative the loss of a connected or isolated patch, and detects as more important the loss of bigger patches, of key and less important steppingstone patches.

To manage the expected damages resulting from wildfires, the land planner can decide to undertake specific treatments, in the form of a combination of controlled burns and/or mechanical thinnings. Upon treatment, we assume that vegetation age in the cell is reset to ‘absent’: the wildfire risk vanishes, but so does the habitat and its connection to surrounding cells. Given the tension between maintaining habitat and reducing wildfire risk, the land planner aims to minimize a deterministic measure of connectivity of the high fuel loads in the landscape while maintaining a given level of biodiversity habitat connectivity under a budget constraint, over a planning horizon of length T . For the sake of the analysis, we focus on two layers of complexity over time and space: risk connectivity and biodiversity habitat. We do not consider heterogeneity in the economic costs or benefits (i.e, homogeneous treatment costs and no patch-specific asset to protect). The framework is however amenable to such a prioritization. We also assume that the budget cannot be banked, and has to be utilized in each period, consistent with operational rules. Moreover, as the budget is constrained in each period, the measure of risk is bounded and the planning horizon is finite, we rule out discounting and assume each generation matters as much to the social planner.

The optimization problem is :

$$\min_x \left[\sum_{t=1}^T H_F(A(t)) \right] \quad (3.2)$$

Such that:

$$A_i(t+1) = \min((A_i(t) + 1)(1 - x_i(t)), 2), \quad t = 1, \dots, T, \quad \forall i \in C \quad (3.3)$$

$$H_B(A(t)) \geq Biod, \quad t = 1, \dots, T \quad (3.4)$$

$$\sum_i x_i(t) \leq Budget, \quad t = 1, \dots, T \quad (3.5)$$

$$A(0) \text{ given} \quad (3.6)$$

$$x(t) \in \{0, 1\}^{n^2} \quad (3.7)$$

We abstract from decision-making in a risky environment, as it has been extensively described in economics and decision theory ([Mouysset et al., 2013](#)). Moreover, we mimic the role of risk aversion by varying the level of habitat connectivity constraint the decision maker chooses. We solve the dynamic, integer program of the landscape manager using dynamic programming. Dynamic programming provides a temporal decomposition of the initial problem defined over T periods, into T simpler problems, as it relies on the 'optimality principle'¹. Second, it provides feedback controls which are known to be more adaptive especially if shocks occur or uncertainties affect the states or the dynamics of the system. The outputs of the method are both the optimal policies $x_j^*(t, A)$, i.e, the sequence of optimal controlled burns, and the optimal states $A_j^*(t, A_0)$ resulting from the optimal policies and the initial conditions

We solve the land planner's problem for every possible initial condition, thus giving rise to general conclusions on the properties of landscapes and treatments emerging from this problem, under various budget scenarios to account for climate change.

3.2.2 Landscape indicators

To characterize the managed landscapes, we mobilize several indicators from landscape ecology and graph theory (see appendix B). First, we account for the risky and habitat areas in the landscape (eq. B.1). Second, to assess landscape connectivity/fragmentation and diversity in the context of fire mosaics ([Bradstock et al., 2005](#)), we use our connectivity metric (eq. 3.1), the number of components e.g. the number of maximal connected subgraphs within the graph, that is not connected to other vertices (eq. B.2) for the risky cells graph, as well as the corresponding areas. To specifically assess landscape diversity, we use the Simpson index ([Simpson, 1949](#)) on seral stages (eq. B.3)². However, the Simpson index does not account for the diversity of spatial patterns: a checkered landscape with two seral stages would be as diverse as a landscape with two large patches for each seral stage, according to the Simpson index. Therefore, we use the landscape shape index (eq. B.4), a normalized ratio between the perimeter of biodiversity habitat and its area ([Patton, 1975](#); [McGarigal and Marks, 1995](#)). To disentangle the correlated effects of perimeter and area that affect the landscape shape index, we use a land type heterogeneity index, that averages the probability that, for each cell, neighbors in the 4 cardinal directions share the same land types (eq. B). The index ranges between 0, when the land type is the same across the whole landscape, to 1, in a checkered landscape. The index assesses whether the landscape

¹"An optimal policy has the property that whatever the initial state and initial decision are, the remaining decisions must constitute an optimal policy with regard to the state resulting from the first decision". (See [Bellman \(1957\)](#), Chap. III.3., p.83)"

²Similar results can be found with the Shannon index ([Shannon, 1948](#)). To avoid issues related to degenerate values and logarithms, we focus on the Simpson index.

is a mosaic (Bradstock et al., 2005), and if it displays structural diversity, conducive to diverse communities and functional diversity.

3.2.3 Computational experiments

Our problem can be viewed as a critical node detection problem, i.e. a problem of locating the nodes that best degrade connectivity metrics (Arulselvan et al., 2009). Problems of the critical node class are computationally difficult (e.g. NP - Hard) in a single graph (Arulselvan et al., 2009; Matsypura et al., 2018). Efficient heuristics to find near-optimal solutions exist and leverage perturbations around local solutions (Arulselvan et al., 2009; Zhou and Hao, 2017). Our problem is a constrained, integer optimization problem that constrains not only the set of nodes to be removed but also metrics relative to a larger graph structure (e.g. supergraph of risky patches), biodiversity habitat. For this reason, existing heuristics may not perform well on our problem. Moreover, the complexity of our combinatorial problem increases with landscape size and vegetation age class exponentially, displaying the ‘curse of dimensionality’ (Bellman, 1957). Therefore, we limit ourselves to studying all the initial conditions in landscapes of size $n = 3$ and 4 . While this formulation appears simplifying, it encapsulates the main mechanisms displayed in similar models (Rachmawati et al., 2016, 2018). It allows us to solve the problem for the whole set of initial conditions, for the whole range of biodiversity habitat connectivity constraint values, over 20 years. In our analysis, we consider a range of budget values for treatment costs normalized to 1. As common in the literature, we can express the budget as a share of land being treated ranging from 5% to 44% of the surface area. These values encompass historical and projected policies in Australia (Burrows and McCaw, 2013), the United States (Office, 2019) and Southern Europe (Fernandes et al., 2013).

Of all the 3^n^2 initial conditions landscapes, we only keep landscapes that are unique up to a permutation³. This results in a sharp reduction of landscapes to consider, from 19,683 initial conditions to 2861 unique initial landscapes for $n = 3$, and from 43,046,721 initial to 5,398,082 unique initial landscapes for $n = 4$. We focus on exact optimal solutions for all the initial conditions of these small-scale landscapes and implement our own solution algorithm in Python 3.9.13. Data and code are publicly available.

3.3 Results

3.3.1 Steady states

Our simulations show that 100% of the initial landscapes converge in finite time towards a steady state solution, that minimizes wildfire risk while satisfying budgetary and habitat connectivity requirements. Steady states are landscape cycles with finite periods. Analyzing the steady-state cycles (and the unique landscapes that form them) drastically reduces the set of landscapes to analyze: they represent 2% (resp. 0.001%) of the initial landscapes of size $n = 3$ (resp. $n = 4$). Our model highlights the convergence of landscapes towards types that can be managed to deliver several objectives. As landscape size increases, the number of steady state landscape cycles increases, but the power of convergence increases as well (e.g. ratio between initial configurations and effective steady state landscapes): from 19 683 initial landscapes when $n = 3$, 51 steady states emerge and from 43 046 721 initial landscapes when $n = 4$, at most 95 diverse steady-state landscapes emerge. Focusing on steady states makes all the more sense as landscape size increases.

³That is to say, landscape A is included in the set of initial conditions \mathcal{I} if and only if for any element B in \mathcal{I} , A is not a permutation (eg can be obtained through rotations or symmetries) of B

Eventually, figure 3.B.2 shows that conditional on data availability on every patch, the more the decision maker wants to conserve biodiversity, the fewer steady-state landscapes she has to consider. An increase in the habitat requirement reduces the room for maneuver. Indeed, budget acts as a complexifying factor: the larger the budget (relative to costs), the larger the set of steady-states to consider. Aiming for relatively large habitat connectivity reduces the set of viable strategies to be considered and can more efficiently guide policy.

3.3.2 Wildfire risk reduction and habitat connectivity in steady state landscapes

Figure 3.B.3 shows the wildfire risk reductions and habitat requirements normalized by their respective maximum values for landscapes of size $n = 3$ and 4 . The maximum value for both risk and habitat corresponds to a landscape covered in 'old' vegetation, which we take to be the counterfactual. Randomly assigned treatments do generate risk reductions but are not cost nor habitat-efficient. Following our spatial optimization procedure, it is clear that implementing fuel treatment reduces wildfire risk while supporting biodiversity habitat. Figure 3.B.3 shows that these two objectives come as a trade-off, albeit moderate: indeed, increasing habitat requirements increases the remaining risk, but there are combinations that can satisfy large habitat connectivity and risk reductions. Budget is a key factor in risk reduction, as it relaxes the trade-off between the two objectives: increasing the budget reduces the wildfire risk while maintaining a range of biodiversity constraints. When habitat constraints are large, however, the marginal effect of budget is limited, and a larger remaining risk needs to be accepted. For example, with a budget of 25% of land to be treated (with landscape size $n = 4$), and no habitat constraint, risk can be reduced up to 80% compared to the counterfactual scenario. However, when the habitat constraint is at 60%, only 70% of risk reduction can be achieved. Moreover, this risk reduction can be achieved with a lower budget. Conversely, as the costs of treatment increase, for a stable budget, the remaining risk increases sharply, and factoring in habitat requirements in the decision-making is not necessary for targets below 80%.

3.3.3 Properties of steady state landscapes: surface, fragmentation, and diversity

Figure 3.B.4 displays, for each class, the most frequent steady-state cycle for landscapes of size 3 and 4 for each biodiversity target. Figure 3.B.5 shows the indicators relative to the surface and components of the high-risk graph and figure 3.B.6 shows the indicators related to diversity, both for landscapes of size $n = 3$ and 4 , averaged over all the steady-state landscape cycles.

Previous results show that budget increases risk reduction, conditional on habitat connectivity constraint being low. Focusing on zones A and A' of the panels of figure 3.B.5 shows that risk reduction primarily comes from a reduced surface (panels 3.B.5a and 3.B.5b), and an increase in the number of components, i.e. disconnected high-risk patches (panels 3.B.5e and 3.B.5f). Overall, the high-risk area is reduced and the number of components increases, thus resulting in smaller largest high-risk component area (panels e and f). As more connected habitat area needs to be protected, the high-risk surface increases (fig. 3.B.4 panels 3.B.5a and 3.B.5b) and the number of high-risk components drastically reduces. The landscapes collapse to the same dominant structure (fig. 3.B.4), where the high-risk area is (almost) maximal and there is one large, well-connected component. Overall, landscapes are riskier but also feature larger, better-connected biodiversity habitat. For large budgets (e.g. 3 and 4), these effects are non-trivial: the number of components (weakly) increases first, small components either disappear or increase in size (see figure 3.B.4 for budget 4

in panels A' , B' and C'), risky patches are reallocated to connect separated components before the high-risk surface increases.

Landscape diversity unambiguously increases with the budget (panels 3.B.6a, 3.B.6b, sections A and A'). As more units are treated, the evenness of seral stages increases in the landscapes. When the habitat objective is low, the spatial diversity of landscapes increases with the budget (panels 3.B.6c, 3.B.6d): even though the relative area of habitat decreases with the budget, the shape of habitat is more irregular, and the landscape is more of a mosaic. In this context, cells with a ‘young’ seral stage act as stepping stones and corridors between high-risk habitat patches. When habitat objectives increase, diversity collapses both quantitatively and qualitatively (fig. 3.B.6). The Simpson index collapses from panels A (resp. A') to G (resp. F'), as land types gradually homogenize (see fig. 3.B.4 for an illustration) across all budgets. Moreover, landscapes form less of a mosaic, and are more clumpy, as displayed by the LSI and Land type heterogeneity index. Overall, for large habitat targets, landscapes tend to homogenize and to be better connected, although less quantitatively and qualitatively diverse.

Results are consistent across landscape sizes while they display more variability for size $n = 3$, as border effects play a larger role.

3.3.4 Spatial allocation of optimal management at the steady-state landscape cycle

Figures 3.B.7a and 3.B.7b display the number of fuel treatments in the steady-state cycles, for various budgets and habitat connectivity constraints. Treatment allocation follows the evolution of the high-risk area (fig 3.B.5a and 3.B.5b): the larger the budget, the larger the treated area, the budget constraint is always satiated. However, when biodiversity targets increase, the budget constraint is no longer satiated.

Figures 3.B.7c and 3.B.7d display the average spatial location of treatments in the steady state cycles. The darker the cell, the higher the frequency of treatment. First, not all cells are equally treated. For low levels of biodiversity constraint, panels A and A' of figures 3.B.7c and 3.B.7d show that central cells are primarily treated, and when the budget increases, cells on the edges get treated, while corner cells are never treated. In the context of critical node detection, when the ecological requirements are low, the high-risk graph is primarily considered, and nodes with the most cost-efficient risk reduction, i.e., with the largest degree are targeted. Once the most connected cells are treated, lower-degree cells get treated.

When habitat constraints increase, several effects come at play. Not only does the number of treatments decrease, but the spatial allocation also changes. For example, in panels A and B for budgets 3 and 4, panels C and D for budget 2 and panels E and F for budget 1 in figure 3.B.7c, the number of treatment remains the same but is spatially reallocated to lower degree nodes. Treatments are spatially reallocated before being reduced. In this context, as the relative weight of the habitat graph increases, treating the most cost-efficient risk-reducing nodes also degrades habitat connectivity. Therefore, as habitat targets increase, edge and corner (e.g. low degree nodes) are being treated and habitat connectivity is maintained.

3.4 Discussion

3.4.1 Confirmation and generalization of existing results

Our analysis of the exhaustive set of initial conditions for small-scale landscapes confirms existing results in the literature. We argue that they bring robust evidence and complement the existing literature to derive general conclusions.

Our model encompasses 3 seral stages and 1 composite vegetation type and proves the convergence of every initial condition to a steady state cycle, irrespective of the initial configuration. We extend [Minas et al. \(2014\)](#) that find convergence patterns for *homogeneous* landscapes only, i.e, landscapes where the initial vegetation age is uniformly distributed. We show that in the event of environmental perturbations that do not disrupt ecosystem dynamics, an appropriate policy can recover the previous equilibrium risk and habitat. We hypothesize that as long as the risk/seral-stage relationship reaches a plateau for every vegetation type on the landscape, convergence should be observed.

Our results display a concave production possibility frontier (PPF) between wildfire risk reduction and habitat connectivity, consistent with PFF literature ([Arthaud and Rose, 1996](#); [Calkin et al., 2005](#)). Our results also confirm that trading one objective for the other is not as efficient as increasing the policy budget to reconcile objectives. We show that increasing the policy budget nonetheless has diminishing returns for risk reduction, as highlighted by [Wei et al. \(2008\)](#); [Yemshanov et al. \(2021\)](#) and [Pais et al. \(2021b\)](#).

Our study yields clear results in terms of landscape ecology, leveraging concepts from landscape ecology, and highlighting the spatial mechanisms underlying the shape of PPF. We show that treatment allocation targets the most central nodes first and then focuses on less connected nodes (e.g cells closer to the border of the landscape) when habitat goals are low. In doing so, we do find general treatment allocation principles where previous studies on larger landscapes could not ([Minas et al., 2014](#); [Rachmawati et al., 2016](#)), generalize smaller scale ([Konoshima et al., 2008](#)) and case study specific ([Yemshanov et al., 2021](#); [Pais et al., 2021a](#)) results.

Leveraging a dynamic integer programming, graph theoretic framework on small-scale landscapes, we show that cell-level metrics help formalize and understand the drivers of treatment allocation and rationalize existing results. Furthermore, we show that while prioritization approaches based on a graph theoretic framing fare very well in an unrestricted set-up, including biodiversity habitat targets augments the problem's complexity. We generalize case studies ([Yemshanov et al., 2022](#)) and show less central high-risk nodes need to be targeted to achieve risk reduction and safeguard biodiversity habitat.

3.4.2 Caveats and methodological perspectives

Our analysis tackles the exhaustive set of landscapes of size $n = 3$ and 4 . Our approach allows us to study the steady-state patterns emerging from any initial condition, replicates existing results in larger landscapes, and sheds light on the mechanisms underlying the wildland dilemma. Increasing landscape size is incompatible with this approach, as we would run into a dimensionality curse ([Bellman, 1957](#)). To conserve our exhaustive approach, different proof mechanisms would be required. Nonetheless, if landscape size is of the essence for actual policy recommendation, so are other layers of information such as habitat quality, treatment costs, and values at risk heterogeneity. These other layers would reduce the computational burden, and we believe our results, targeting the most cost-efficient, risk-reducing, and habitat-conserving strategies, would still apply.

In our model, we use a simple relationship to characterize the link between the seral stage, habitat formation for a single species, and wildfire risk and severity. This choice is motivated by the existence of a lower bound for a fire return interval and drives our ability to adopt our exhaustive approach. Increasing the number of seral stages would help to complexify the relationships governing habitat formation and wildfire risk and severity: in some ecosystems, wildfire risk and severity may be higher for young vegetation than for older and may not be linear ([Taylor et al., 2014](#)). On the other hand, some species may require old-growth forests to survive, not 'young' forests, and old-growth forests may also be more fire-resilient ([Lesmeister et al., 2021](#)). As the

number of seral stage augments, convergence towards steady-state landscape cycles would take longer, but we hypothesize it would still occur. Moreover, as long as wildfire risk and habitat quality are in conflict, a trade-off would govern treatment allocation. Multiple seral stages may be targeted for fuel treatment, depending on their location and properties, but we claim the general mechanism would still apply: in a graph weighted for different risk and habitat properties, centrality and connectivity would still guide treatment allocation.

We implicitly assume that focusing on a given species' habitat would also provide habitat for a variety of species and be conducive to functional diversity. However, this does not imply that all species would benefit from maintaining a given habitat type ([Saab et al., 2022](#)). Moreover, the lack of structural diversity may cause the trophic web of the targeted species to collapse. Therefore, management objectives should include structural diversity. In this case, landscapes could not satisfy extreme habitat connectivity targets and diversity targets. For intermediate goals, however, we claim that treatment allocation would still aim at fragmenting the landscape, and node centrality and connectivity would still govern allocation.

Eventually, we chose to abstract from a stochastic ignition process affecting the landscape. As a thought experiment, imagine a Bernoulli-distributed, high-risk area independent probability of ignition in each period. If part of the landscape ignites, all that remains is the unburnt habitat, while if not, all habitat remains. A decision-maker faced with maximizing the expected payoff in this scenario would solve the reciprocal of our problem. On the one hand, she has to ensure that the high-risk cells in the landscape are not 'too' connected, to maximize the remaining habitat in the event of a wildfire. On the other hand, she wants to maximize connectivity for wildlife when there is no wildfire. As a result, the trade-off she faces, and the resulting spatial allocation of treatment would be the same. The stochastic nature of ignition may change the steady state cycles, but convergence would not be impossible. If the probability of wildfire increases, she focuses more on maintaining a 'young' seral stage over the landscape. In this setting, increasing the probability of ignition would act as a decrease in our habitat target as well as an increase in the budget available for policy. With our model, we are able to disentangle these two effects and understand how each constraint would play. We claim we match with actual policy, where the budget is not fully endogenously determined.

3.4.3 Conclusion and policy relevance

While there is a *dilemma* for land managers between lowering wildfire risk and severity and maintaining species habitat connectivity, reconciling the two objectives is not a dead end. This is an important result for land planners as biodiversity habitat targets are gradually included in policy agendas (for example, the recent pledge by the participants to the Conference of Parties on Biodiversity in Montreal to preserve 30% of land and oceans by 2030 for biodiversity⁴). It shows that if policymakers can commit to a given budget over time, these biodiversity targets can be reached and a management cycle that minimizes wildfire risk can be implemented in wildlands. Moreover, as steady-state cycles are reached, the uncertainty over future land uses is resolved while achieving policy goals.

In the face of climate change, treatment costs are expected to increase ([Kupfer et al., 2020](#)). The decreasing marginal efficiency of budget to reduce risk highlights that as climate change increases the costs of treatments, risk, and damages will increase at an increasing rate, unless the budget is changed accordingly.

Our analysis shows that budget should be determined by factoring a careful, *ex-ante* analysis of treatment costs, the policy maker's risk aversion towards a measure of wildfire risk and severity,

⁴See Target 2 in the [Keunming-Montreal Global Diversity Framework, 2022](#)

and ecological preferences. Indeed, low budget-to-cost ratios are incompatible with high risk and severity aversions and/or large ecological requirements.

As wildfires and biodiversity habitat destruction are challenges in the face of global warming, finding policy guidance tools is of the essence. Many studies focus on specific case studies or limited ranges of potential initial conditions. We develop a simplified ecological model of habitat and wildfire connectivity to guide policymakers in the form of general principles. Reducing wildfire risk and accommodating wildlife habitat is possible with carefully designed policies, where budget plays a key role. However, it is impossible to achieve drastic risk reduction without harming biodiversity habitat. General principles of treatment allocation in the landscape are derived, and the concepts of graph theory provide an operational toolbox to understand the underlying mechanisms. Landscape patches that display high wildfire risk seral stages and are well connected to other patches should be treated first. When habitat targets are included, tackling lower-risk patches is of the essence to maintain habitat connectivity.

Our article summarizes and generalizes how policies should be implemented, both in terms of budgets and spatial allocation, to protect and enhance ecosystem health.

3.5 Declaration

3.5.1 Acknowledgments

This research was conducted while SJ was on leave at the Environmental Markets Lab, UC Santa Barbara. We acknowledge support from the Center for Scientific Computing from the CNSI, MRL: an NSF MRSEC (DMR-1720256) and NSF CNS- 1725797, at UC Santa Barbara. Moreover, the authors are grateful to the editor and X anonymous referees, as well as participants to the Columbia Interdisciplinary PhD Workshop in Sustainable Development and the BINGO group at CIRED for their valuable comments.

3.5.2 Data availability

Given its size, steady-state cycle data is available upon request from the authors. Code for replication is available at https://github.com/sim-jean/Landscape_connectivity_dilemma

3.5.3 Author affiliation

CIRED, Ecole des Ponts, AgroParisTech, EHESS, CIRAD, CNRS, Université Paris-Saclay, Nogent-sur-Marne, France

3.5.4 Competing interests

The authors declare no conflict of interest.

3.5.5 Contribution

LM designed the study, SJ ran the computational experiment, SJ and LM analyzed the results and wrote the manuscript.

3.A Appendix

A Theoretical model

A.1 Vegetation dynamics

In cell i at time t , vegetation ages $A_i(t)$ evolves according to the following :

$$A_i(t+1) = (A_i(t) + 1)(1 - x_i(t)), t \in \{0, 1, \dots, T\}, \forall i \in C \quad (\text{A.1})$$

Where $x_i(t) \in \{0, 1\}$ is a binary variable, representing the treatment status of cell i at time t . Correspondingly, the age vector across the landscape is $A(t) = \{A_i(t)\}_{i \in C}$.

A.2 Mature habitat and risky patch designation

Cell i is labeled 'mature' to host wildlife in year t as:

$$\text{Mature}_i(A(t)) = \begin{cases} 1 & \text{if } A_i(t) \geq m \\ 0 & \text{otherwise} \end{cases} \quad (\text{A.2})$$

Where m is the 'mature' threshold. Correspondingly, the vector of mature cells across the landscape is $\text{Mature}(A(t)) = \{\text{Mature}_i(A(t))\}_{i \in C}$

Similarly, cell i is labeled as 'high fuel load' in year t as:

$$\text{High}_i(A(t)) = \begin{cases} 1 & \text{if } A_i(t) \geq d \\ 0 & \text{otherwise} \end{cases} \quad (\text{A.3})$$

Where d is the 'high fuel load' threshold. Correspondingly, the vector of high fuel load cells across the landscape is $\text{High}(A(t)) = \{\text{High}_i(A(t))\}_{i \in C}$

We assume that the maturity threshold is crossed before the high risk threshold, i.e $m < d$.

A.3 Global connectivity index and graph theory

Let a grided landscape of size n , where for each cell a_i in the set of cells A in the landscape, one defines Φ_i the set of cells connected to cell i (i.e, cells share the same status and can only be in the 8-direction direct neighborhood). Moreover, let Q_{ij} be a binary variable such that $Q_{ij} = 1$ if cells a_i and a_j are connected, 0 otherwise. [Minas et al. \(2014\)](#) define the following connectivity metric over a landscape:

$$\sum_{i \in C} \sum_{j \in \Phi_i} Q_{ij} \quad (\text{A.4})$$

Now view the landscape as a graph G , with vertices V and edges E such that $G(V, E)$. For the proof, assume that Y is a binary vector such that $Y_i = 1$ if cell i is 'high risk' and 0 otherwise, and that we focus on the 'high risk' graph on the landscape. The argument is identical in the case of mature habitat.

In graph theory, an adjacency matrix \mathcal{K} for an undirected graph is a binary, symmetric, square matrix of dimension $\text{card}(V)^2$ where $k_{ij} = 1$ if vertices i and j are connected, 0 otherwise. In our

context, it is clear that $k_{ij} = Q_{ij}$. Equation A.4 can be reformulated as :

$$Y' \mathcal{K} Y = \sum_j \left(Y_j \sum_i Y_i k_{ij} \right) = \sum_j \left(Y_j \left(Y_j k_{jj} + \sum_{i \neq j} Y_i k_{ij} \right) \right)$$

Given the symmetric nature of \mathcal{K} , $\forall i \neq j$, $k_{ij} = k_{ji}$. Each cell is connected to itself so $k_{jj} = 1$. $Y_i \in \{0, 1\}$ i.e $Y_i^2 \in \{0, 1\}$:

$$\begin{aligned} Y' \mathcal{K} Y &= \sum_j \left(Y_j^2 + \sum_{i \neq j} Y_i Y_j k_{ij} \right) \\ &= \sum_j Y_j + 2 \sum_{j < i} \left(\sum_{i \neq j} Y_j Y_i a_{ij} \right) \end{aligned}$$

The first sum is the number of cells either 'mature' or 'high risk', i.e, the cardinal of the nodes of the 'high risk' graph e.g $\text{card}(V)$. In the second sum, $\sum_{i \neq j} Y_j Y_i a_{ij}$ is the number of connections of cell i to cell j , as the product $Y_i Y_j a_{ij} = 1$ if and only if cell i and j share the same status ($Y_i = Y_j$) and are in the 8-cell neighborhood ($a_{ij} = 1$). By definition, the sum of the number of connections of each cell to other cells is $\text{card}(E)$. Hence, for a set of cells C , reformulated in terms of graph theory :

$$\sum_{i \in C} \sum_{j \in \Phi_i} Q_{ij} = \text{card}(V) + 2\text{card}(E) \quad (\text{A.5})$$

A.4 Dynamic programming equation

The Bellman equation links current and future payoffs in a recurring fashion.

$$V(t, A) = \min_{x \in \{0,1\}^{n^2}} (H(A) + V(t+1, A_t + 1)) \quad (\text{A.6})$$

subject to constraints (3.3), (3.5), (3.4) and (3.7).

We use backward induction given by the final value $V(T, A) = H(A)$ to dynamically solve the program.

B Landscape indicators

Area We use the number of vertices (nodes) for both subgraphs and take into account cell dimensions:

$$\text{Area}(G_i) = \text{card}(V_i) \text{ for } i \in \{B, F\} \quad (\text{B.1})$$

Number of components

$$\#\text{components}_i = \text{card}(\text{Maximal connected subgraphs of } G_i \text{ for } i \in \{B, F\}) \quad (\text{B.2})$$

Simpson diversity index: let p_i be the proportion of land type i in the landscape. The Simpson diversity index is :

$$SIDI = 1 - \sum_i p_i^2 \quad (\text{B.3})$$

Landscape shape index: following McGarigal and Marks (1995), the adapted LSI index from Patton (1975) in a raster landscape is:

$$LSI = \frac{0.25 \times perimeter(G)}{n} \quad (B.4)$$

Where $perimeter(G)$ is the perimeter of the cells comprised in the graph as vertices.

Land Type Heterogeneity Index: let d_{ij} be a binary variable such that $d_{ij} = 1$ if patch i and j share the same land type. Define \mathcal{J} as the set of neighbors in 4 directions (north, south, east, west) of cell i ⁵. The land type heterogeneity index is :

$$LTH = \frac{1}{N} \sum_{i=1}^N \left(\frac{\sum_{j \in \mathcal{J}_i} d_{ij}}{card(\mathcal{J}_i)} \right) \quad (B.5)$$

⁵The set \mathcal{J}_i varies with cell i to account for edge effects

3.B Figures

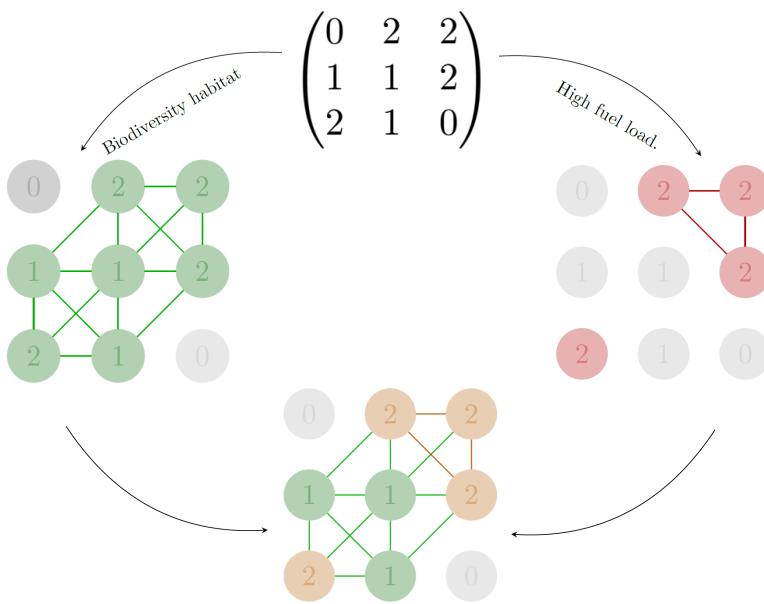


Figure 3.B.1: Illustration of the habitat and fuel graphs for $n = 3$

In this graph, green cells support biodiversity habitat only, while red cells display high risk.

The high risk graph has two components (top right corner with 3 nodes, and bottom left corner with 1 node), while the biodiversity habitat graph only has one.

Cells for which the value is 0 are not considered as nodes for both graphs, and are thus not connected to the rest of the graphs. In the end, because high fuel load cells also support biodiversity habitat, the landscape can be represented as the overlap between the two graphs, where orange cells are high fuel load and also support biodiversity habitat.

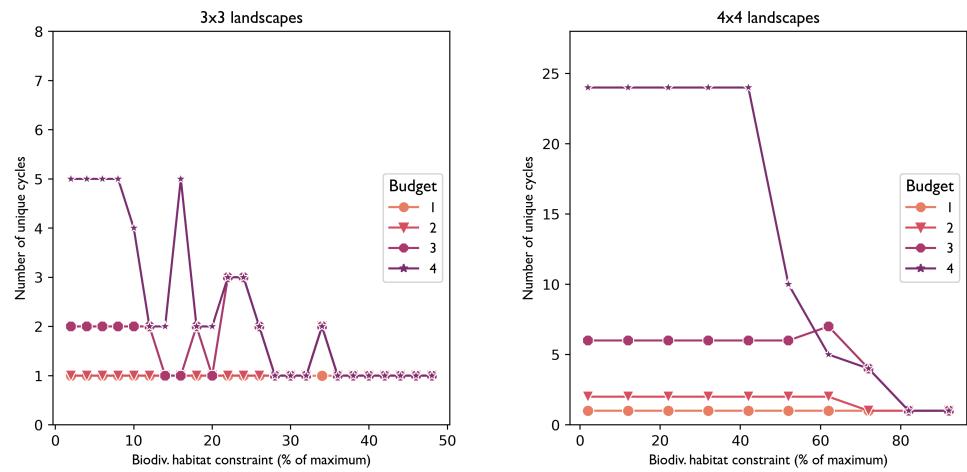


Figure 3.B.2: Number of cycles as a function of biodiversity habitat and budget

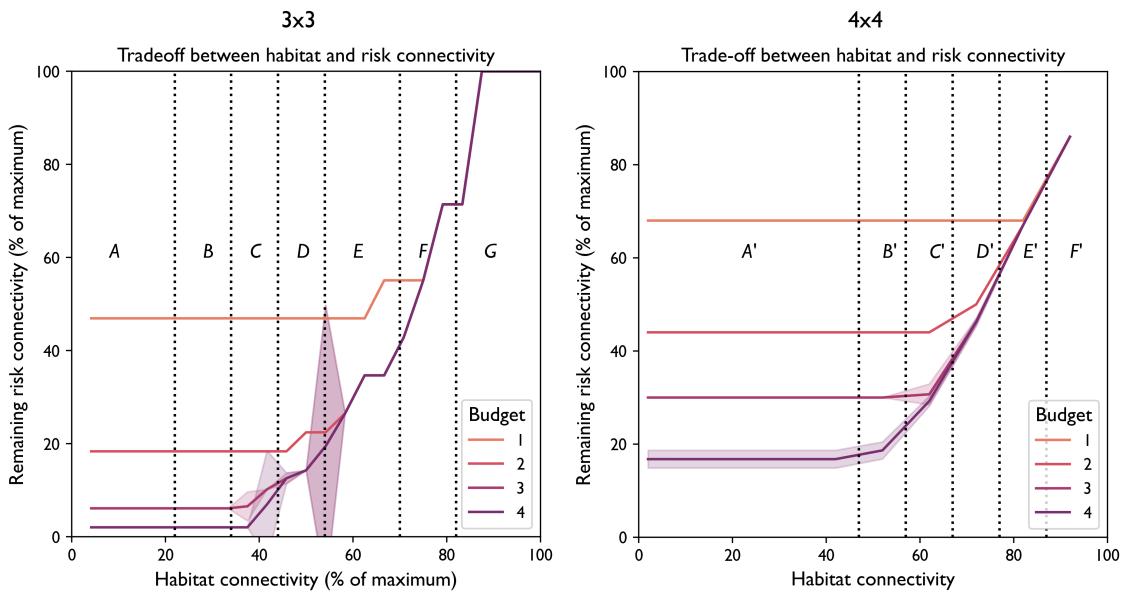


Figure 3.B.3: Production possibility frontier between constraint (as a % of maximum biodiversity sustainable in landscape) and wildfire risk for various budgets, and landscape size

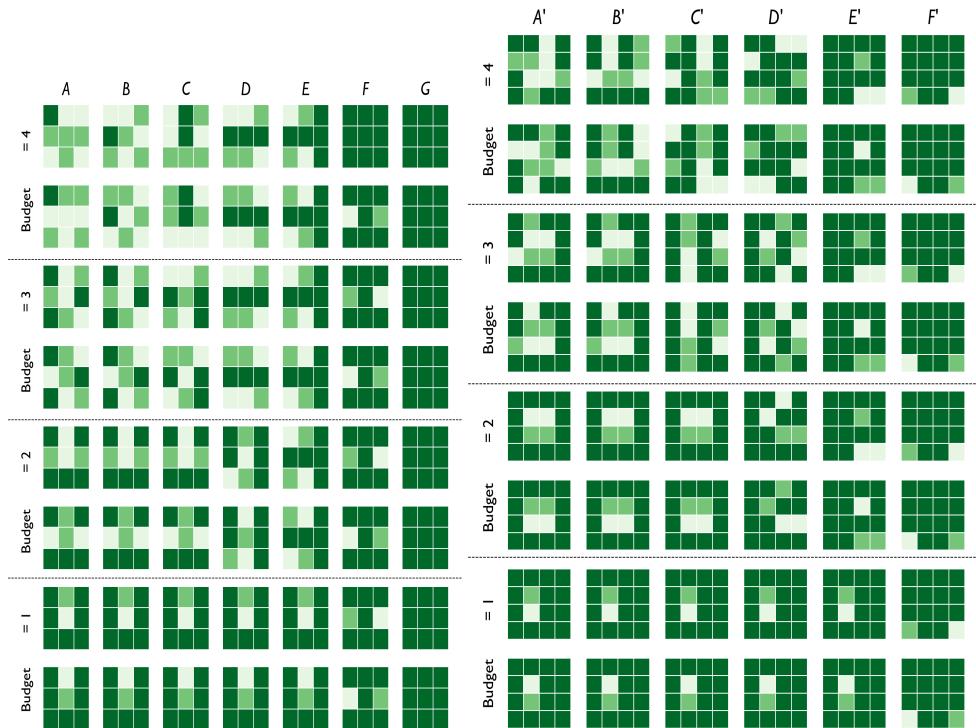


Figure 3.B.4: Most represented cycles for each biodiversity constraint level, for various budget and landscapes 3×3 , and 4×4 (95% CI shaded)

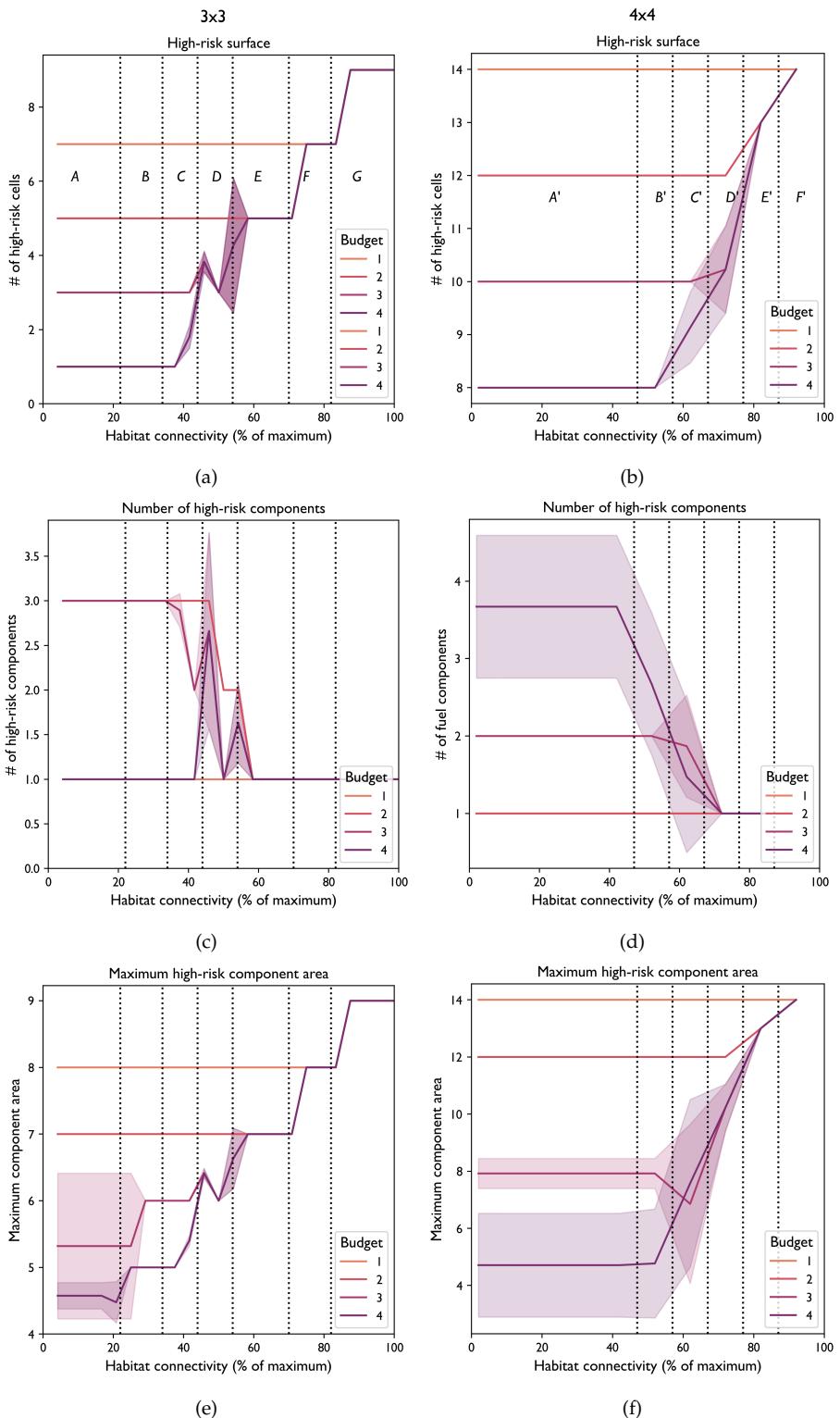


Figure 3.B.5: Assessment: surface, components of high-risk graph (95% CI shaded)

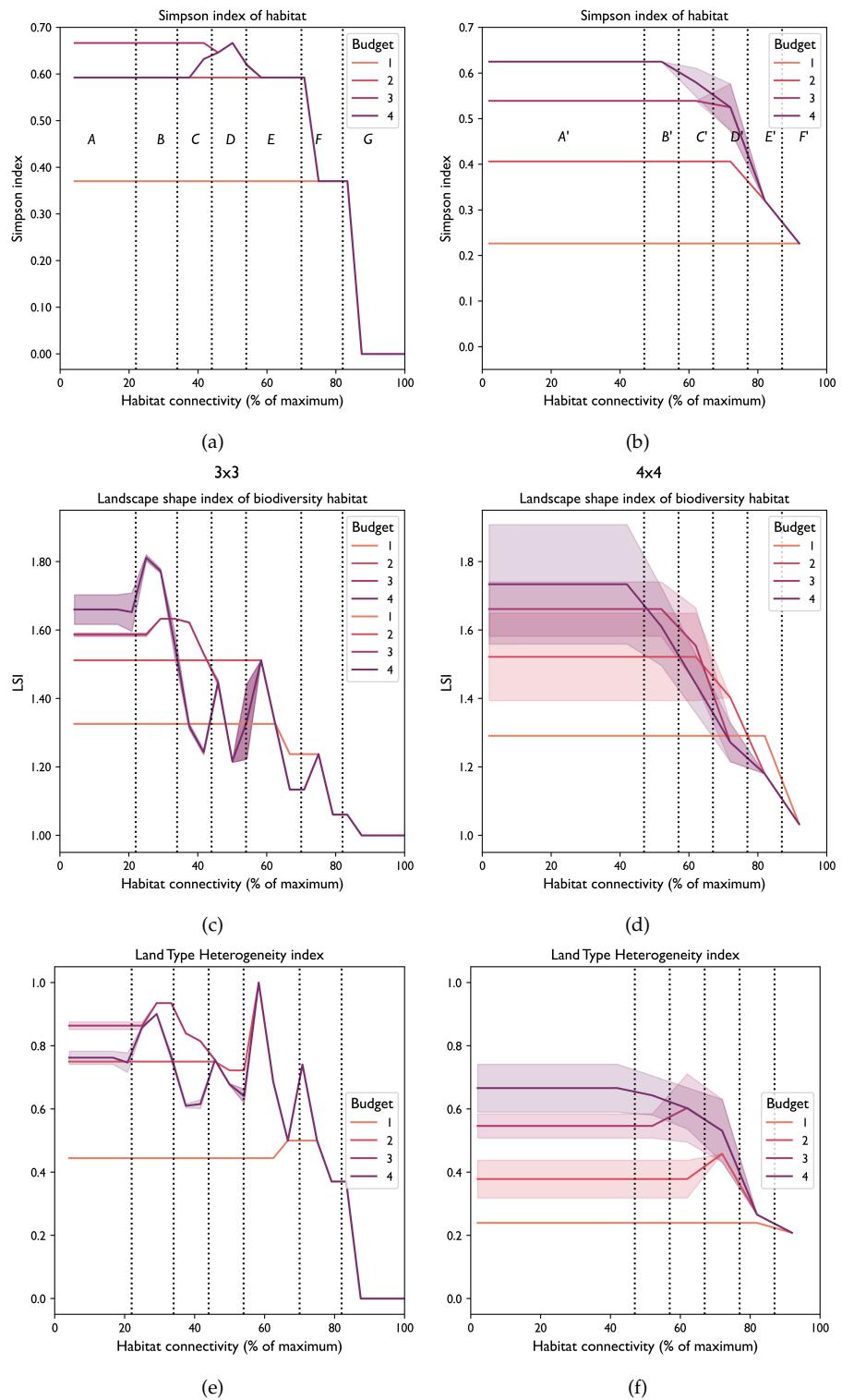


Figure 3.B.6: Assessment: diversity (95% CI shaded)

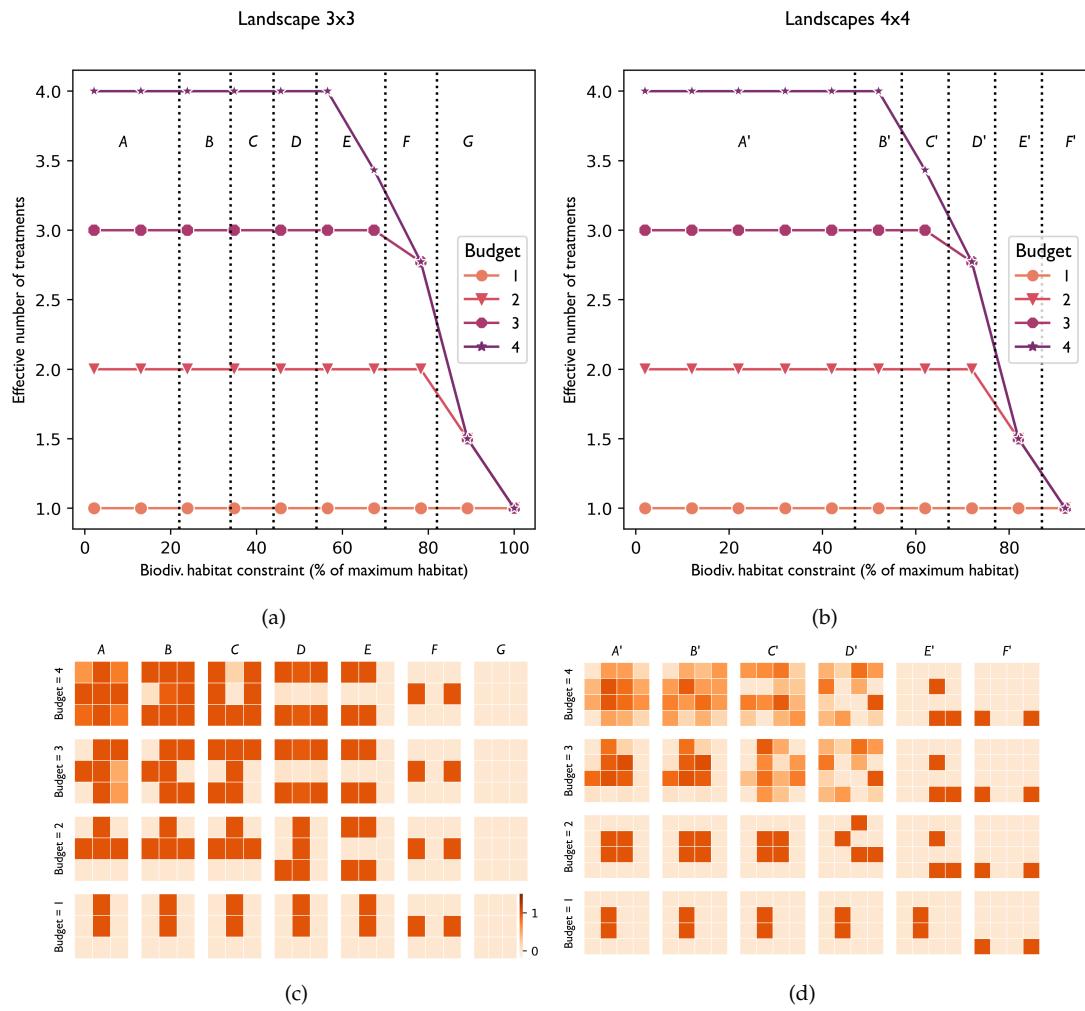


Figure 3.B.7: Treatment allocation : number, location

Chapter 4

Fences

Bibliography

- Collateral Damage : How illegal trade in totoaba swim bladders is driving the vaquita to extinction. Technical report, Environmental Investigation Agency, 2016.
- Species Specific matters : Totoaba (Totoaba Macdonaldi) Report of Mexico. Technical report, CITES, 2018.
- CITES's Last Chance : Stop the illegal totoaba trade to save the vaquita - Briefing to the 18th Conference of the Parties to CITES and the 71st Standing Committee. Technical report, Environmental Investigation Agency, 2019.
- J. T. Abatzoglou and A. P. Williams. Impact of anthropogenic climate change on wildfire across western us forests. *Proceedings of the National Academy of Sciences*, 113(42):11770–11775, 2016. doi: 10.1073/pnas.1607171113. URL <https://www.pnas.org/doi/abs/10.1073/pnas.1607171113>.
- B. Abbott and G. C. van Kooten. Can domestication of wildlife lead to conservation? The economics of tiger farming in China. *Ecological Economics*, 70(4):721–728, Feb. 2011. ISSN 0921-8009. doi: 10.1016/j.ecolecon.2010.11.006. URL <https://www.sciencedirect.com/science/article/pii/S0921800910004593>.
- G. J. Arthaud and D. W. Rose. A methodology for estimating production possibility frontiers for wildlife habitat and timber value at the landscape level. *Canadian Journal of Forest Research*, 26(12):2191–2200, Dec. 1996. ISSN 0045-5067. doi: 10.1139/x26-248. URL <https://cdnsciencepub.com/doi/10.1139/x26-248>. Publisher: NRC Research Press.
- A. Arulselvan, C. W. Commander, L. Elefteriadou, and P. M. Pardalos. Detecting critical nodes in sparse graphs. *Computers & Operations Research*, 36(7):2193–2200, 2009. ISSN 0305-0548. doi: <https://doi.org/10.1016/j.cor.2008.08.016>. URL <https://www.sciencedirect.com/science/article/pii/S0305054808001494>.
- S. E. Baker, R. Cain, F. van Kesteren, Z. A. Zommers, N. D'Cruze, and D. W. Macdonald. Rough Trade: Animal Welfare in the Global Wildlife Trade. *BioScience*, 63(12):928–938, Dec. 2013. ISSN 0006-3568. doi: 10.1525/bio.2013.63.12.6. URL <https://doi.org/10.1525/bio.2013.63.12.6>.
- D. Bell, S. Roberton, and P. R. Hunter. Animal origins of SARS coronavirus: possible links with the international trade in small carnivores. *Philosophical Transactions of the Royal Society B: Biological Sciences*, 359(1447):1107–1114, July 2004. ISSN 0962-8436. doi: 10.1098/rstb.2004.1492. URL <https://www.ncbi.nlm.nih.gov/pmc/articles/PMC1693393/>.
- R. Bellman. *Dynamic Programming*. Princeton University Press, 1957.
- D. Biggs, F. Courchamp, R. Martin, and H. P. Possingham. Legal Trade of Africa's Rhino Horns. *Science*, 339(6123):1038–1039, Mar. 2013. doi: 10.1126/science.1229998. URL <https://www.science.org/doi/10.1126/science.1229998>. Publisher: American Association for the Advancement of Science.

- M. M. Boer, R. J. Sadler, R. S. Wittkuhn, L. McCaw, and P. F. Grierson. Long-term impacts of prescribed burning on regional extent and incidence of wildfires—Evidence from 50 years of active fire management in SW Australian forests. *Forest Ecology and Management*, 259(1):132–142, 2009. ISSN 0378-1127. doi: <https://doi.org/10.1016/j.foreco.2009.10.005>. URL <https://www.sciencedirect.com/science/article/pii/S0378112709007294>.
- S. D. Bradshaw, K. Dixon, H. Lambers, A. Cross, J. Bailey, and S. Hopper. Understanding the long-term impact of prescribed burning in mediterranean-climate biodiversity hotspots, with a focus on south-western australia. *International Journal of Wildland Fire*, 27, 2018.
- R. A. Bradstock, M. Bedward, A. M. Gill, J. S. Cohn, R. A. Bradstock, M. Bedward, A. M. Gill, and J. S. Cohn. Which mosaic? A landscape ecological approach for evaluating interactions between fire regimes, habitat and animals. *Wildlife Research*, 32(5):409–423, Aug. 2005. ISSN 1448-5494, 1448-5494. doi: 10.1071/WR02114. URL <https://www.publish.csiro.au/wr/WR02114>.
- E. H. Bulte and R. Damania. An Economic Assessment of Wildlife Farming and Conservation. *Conservation Biology*, 19(4):1222–1233, 2005. ISSN 0888-8892. URL <https://www.jstor.org/stable/3591307>. Publisher: [Wiley, Society for Conservation Biology].
- M. Burke, M. L. Childs, B. De la Cuesta, M. Qiu, J. Li, C. F. Gould, S. Heft-Neal, and M. Wara. Wildfire Influence on Recent US Pollution Trends, Jan. 2023. URL <https://www.nber.org/papers/w30882>.
- N. Burrows and L. McCaw. Prescribed burning in southwestern australian forests. *Frontiers in Ecology and the Environment*, 11(s1):e25–e34, 2013. doi: <https://doi.org/10.1890/120356>. URL <https://esajournals.onlinelibrary.wiley.com/doi/abs/10.1890/120356>.
- C4ADS. Hooked : How Demand for a Protected Fish Lined the Pockets of Mexican Cartels & Sunk the Future of an Endangered Porpoise Species. Technical report, C4ADS, 2017.
- D. E. Calkin, S. S. Hummel, and J. K. Agee. Modeling trade-offs between fire threat reduction and late-seral forest structure. *Canadian Journal of Forest Research*, 35(11):2562–2574, Nov. 2005. ISSN 0045-5067. doi: 10.1139/x05-177. URL <https://cdnsciencepub.com/doi/10.1139/x05-177>. Publisher: NRC Research Press.
- D. W. S. Challender and D. C. MacMillan. Poaching is more than an Enforcement Problem. *Conservation Letters*, 7(5):484–494, 2014. ISSN 1755-263X. doi: 10.1111/conl.12082. URL <https://onlinelibrary.wiley.com/doi/abs/10.1111/conl.12082>. _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1111/conl.12082>.
- D. W. S. Challender, S. R. Harrop, and D. C. MacMillan. Towards informed and multi-faceted wildlife trade interventions. *Global Ecology and Conservation*, 3:129–148, Jan. 2015a. ISSN 2351-9894. doi: 10.1016/j.gecco.2014.11.010. URL <https://www.sciencedirect.com/science/article/pii/S2351989414000791>.
- D. W. S. Challender, S. R. Harrop, and D. C. MacMillan. Understanding markets to conserve trade-threatened species in CITES. *Biological Conservation*, 187:249–259, July 2015b. ISSN 0006-3207. doi: 10.1016/j.biocon.2015.04.015. URL <https://www.sciencedirect.com/science/article/pii/S0006320715001603>.
- D. W. S. Challender, M. Sas-Rolfes, G. W. J. Ades, J. S. C. Chin, N. Ching-Min Sun, J. I. Chong, E. Connelly, L. Hywood, S. Luz, R. K. Mohapatra, P. de Ornellas, K. Parker, D. W. Pietersen, S. I. Roberton, G. Semiadi, D. Shaw, C. R. Shepherd, P. Thomson, Y. Wang, L. Wicker, S. B.

- Wu, and H. C. Nash. Evaluating the feasibility of pangolin farming and its potential conservation impact. *Global Ecology and Conservation*, 20:e00714, Oct. 2019. ISSN 2351-9894. doi: 10.1016/j.gecco.2019.e00714. URL <https://www.sciencedirect.com/science/article/pii/S2351989419301544>.
- M. A. Cisneros-Mata. Evaluacion de la poblacion de Totoaba Macdonaldi. Technical report, IN-APESCA, 2020.
- CITES. Additional Information Regarding the Registration of the Operation "Earth Ocean Farms S. de R.L de C.V." breeding Totoaba Macdonaldi (submitted by Mexico). 2019.
- CITES. Notification to the Parties : compliance action plan of Mexico on Totoaba (Totoaba Macdonaldi), 2023.
- C. W. Clark. *The Worldwide Crisis in Fisheries: Economic Models and Human Behavior*. Cambridge University Press, Cambridge, 2007. ISBN 978-0-521-84005-7. doi: 10.1017/CBO9780511617966. URL <https://www.cambridge.org/core/books/worldwide-crisis-in-fisheries/0726A744452A8E61CEE20A090F5CE9B7>.
- J. D. Coop, S. A. Parks, C. S. Stevens-Rumann, S. D. Crausbay, P. E. Higuera, M. D. Hurteau, A. Tepley, E. Whitman, T. Assal, B. M. Collins, K. T. Davis, S. Dobrowski, D. A. Falk, P. J. Fornwalt, P. Z. Fulé, B. J. Harvey, V. R. Kane, C. E. Littlefield, E. Q. Margolis, M. North, M.-A. Parisien, S. Prichard, and K. C. Rodman. Wildfire-Driven Forest Conversion in Western North American Landscapes. *BioScience*, 70(8):659–673, Aug. 2020. ISSN 0006-3568. doi: 10.1093/biosci/biaa061. URL <https://doi.org/10.1093/biosci/biaa061>.
- R. Damania and E. H. Bulte. The economics of wildlife farming and endangered species conservation. *Ecological Economics*, 62(3):461–472, May 2007. ISSN 0921-8009. doi: 10.1016/j.ecolecon.2006.07.007. URL <https://www.sciencedirect.com/science/article/pii/S0921800906003417>.
- A. J. Dutton, C. Hepburn, and D. W. Macdonald. A Stated Preference Investigation into the Chinese Demand for Farmed vs. Wild Bear Bile. *PLOS ONE*, 6(7):e21243, July 2011. ISSN 1932-6203. doi: 10.1371/journal.pone.0021243. URL <https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0021243>. Publisher: Public Library of Science.
- ElephantActionLeague. Operation Fake Gold : the Totoaba Supply Chain - From Mexico's Totoaba Cartels to China's Totoaba Maw Wholesalres - An illegal Trade Killing the Vaquita. Technical report, Elephant Action League, 2018.
- B. E. Erisman, L. G. Allen, J. T. Claisse, D. J. Pondella, E. F. Miller, and J. H. Murray. The illusion of plenty: hyperstability masks collapses in two recreational fisheries that target fish spawning aggregations. *Canadian Journal of Fisheries and Aquatic Sciences*, 68(10):1705–1716, Oct. 2011. ISSN 0706-652X. doi: 10.1139/f2011-090. URL <https://cdnsciencepub.com/doi/full/10.1139/f2011-090>. Publisher: NRC Research Press.
- M. Fabinyi. Historical, cultural and social perspectives on luxury seafood consumption in China. *Environmental Conservation*, 39(1):83–92, Mar. 2012. ISSN 1469-4387, 0376-8929. doi: 10.1017/S0376892911000609. URL <https://www.cambridge.org/core/journals/environmental-conservation/article/abs/historical-cultural-and-social-perspectives-on-luxury-seafood-consumption-in-china/742483C9475BD76E5A17BCC2651F1BC3>.

- L. Fahrig, J. Baudry, L. Brotons, F. G. Burel, T. O. Crist, R. J. Fuller, C. Sirami, G. M. Siriwardena, and J.-L. Martin. Functional landscape heterogeneity and animal biodiversity in agricultural landscapes. *Ecology Letters*, 14(2):101–112, 2011. doi: <https://doi.org.inshs.bib.cnrs.fr/10.1111/j.1461-0248.2010.01559.x>. URL <https://onlinelibrary-wiley-com.inshs.bib.cnrs.fr/doi/abs/10.1111/j.1461-0248.2010.01559.x>.
- M. Faustmann. Berechnung des Werthes, welchen Waldboden, sowie noch nicht haubare Holzbestände fur die Waldwirthschaft besitzen [Calculation of the value which forest land and immature stands possess for forestry]. *Allgemeine Fotst- und Jagd-Zeitung*, 25:441–455, 1849. URL <https://cir.nii.ac.jp/crid/1571135650773047168>.
- V. Felbab Brown. Organized Crime is Taking Over Mexican Fisheries. *Brookings Institute*, 2022.
- P. M. Fernandes, G. M. Davies, D. Ascoli, C. Fernández, F. Moreira, E. Rigolot, C. R. Stoof, J. A. Vega, and D. Molina. Prescribed burning in southern europe: developing fire management in a dynamic landscape. *Frontiers in Ecology and the Environment*, 11(s1):e4–e14, 2013. doi: <https://doi.org/10.1890/120298>. URL <https://esajournals.onlinelibrary.wiley.com/doi/abs/10.1890/120298>.
- M. A. Finney. Design of Regular Landscape Fuel Treatment Patterns for Modifying Fire Growth and Behavior. *Forest Science*, 47(2):219–228, May 2001. ISSN 0015-749X. doi: 10.1093/forestscience/47.2.219. URL <https://doi.org/10.1093/forestscience/47.2.219>.
- M. A. Finney. A computational method for optimising fuel treatment locations. *International Journal of Wildland Fire*, 16(6):702–711, Dec. 2007. ISSN 1448-5516, 1448-5516. doi: 10.1071/WF06063. URL <https://www.publish.csiro.au/wf/WF06063>. Publisher: CSIRO PUBLISHING.
- V. Florec, M. Burton, D. Pannell, J. Kelso, and G. Milne. Where to prescribe burn: the costs and benefits of prescribed burning close to houses. *International Journal of Wildland Fire*, (29), 2020.
- D. Flores. Violence and law enforcement in markets for illegal goods. *International Review of Law and Economics*, 48:77–87, Oct. 2016. ISSN 0144-8188. doi: 10.1016/j.irle.2016.10.002. URL <https://www.sciencedirect.com/science/article/pii/S0144818816300564>.
- H. E. Froehlich, R. R. Gentry, and B. S. Halpern. Conservation aquaculture: Shifting the narrative and paradigm of aquaculture’s role in resource management. *Biological Conservation*, 215:162–168, Nov. 2017. ISSN 0006-3207. doi: 10.1016/j.biocon.2017.09.012. URL <https://www.sciencedirect.com/science/article/pii/S0006320717307565>.
- J. Gellman, M. Walls, and M. Wibbenmeyer. Welfare losses from wildfire smoke: evidence from daily outdoor recreation data. *Resources for the Future Working Papers*, 2023.
- R. R. Gentry, S. D. Gaines, J. S. Gabe, and S. E. Lester. Looking to aquatic species for conservation farming success. *Conservation Letters*, 12(6):e12681, 2019. ISSN 1755-263X. doi: 10.1111/conl.12681. URL <https://onlinelibrary.wiley.com/doi/abs/10.1111/conl.12681>. _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1111/conl.12681>.
- J. R. Gonzalez-Olabarria, J. Carrasco, C. Pais, J. Garcia-Gonzalo, D. Palacios-Meneses, R. Mahaluf-Recasens, O. Porkhum, and A. Weintraub. A fire spread simulator to support tactical management decisions for Mediterranean landscapes. *Frontiers in Forests and Global Change*, 6, 2023. ISSN 2624-893X. URL <https://www.frontiersin.org/articles/10.3389/ffgc.2023.1071484>.

- B. Gratzwicke, J. Mills, A. Dutton, G. Gabriel, B. Long, J. Seidensticker, B. Wright, W. You, and L. Zhang. Attitudes Toward Consumption and Conservation of Tigers in China. *PLOS ONE*, 3(7):e2544, July 2008. ISSN 1932-6203. doi: 10.1371/journal.pone.0002544. URL <https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0002544>. Publisher: Public Library of Science.
- S. Heft-Neal, C. F. Gould, M. Childs, M. V. Kiang, K. Nadeau, M. Duggan, E. Bendavid, and M. Burke. Behavior Mediates the Health Effects of Extreme Wildfire Smoke Events, Feb. 2023. URL <https://www.nber.org/papers/w30969>.
- A. Hinsley and M. 't Sas-Rolfes. Wild assumptions? Questioning simplistic narratives about consumer preferences for wildlife products. *People and Nature*, 2(4):972–979, 2020. ISSN 2575-8314. doi: 10.1002/pan3.10099. URL <https://onlinelibrary.wiley.com/doi/abs/10.1002/pan3.10099>. _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1002/pan3.10099>.
- S. Hsiang and N. Sekar. Does Legalization Reduce Black Market Activity? Evidence from a Global Ivory Experiment and Elephant Poaching Data, June 2016. URL <https://www.nber.org/papers/w22314>.
- IUCN. The World Conservation Union on the Effectiveness of Trade Measures Contained in the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). Technical report, Environmental Investigation Agency, 2000.
- D. Jinkins. Conspicuous consumption in the United States and China. *Journal of Economic Behavior & Organization*, 127:115–132, July 2016. ISSN 0167-2681. doi: 10.1016/j.jebo.2016.03.018. URL <https://www.sciencedirect.com/science/article/pii/S0167268116300312>.
- W. M. Jolly, M. A. Cochrane, P. H. Freeborn, Z. A. Holden, T. J. Brown, G. J. Williamson, and D. M. J. S. Bowman. Climate-induced variations in global wildfire danger from 1979 to 2013. *Nature Communications*, 6(1):7537, July 2015. ISSN 2041-1723. doi: 10.1038/ncomms8537. URL <https://doi.org/10.1038/ncomms8537>.
- E. L. Kalies and L. L. Yocom Kent. Tamm review: Are fuel treatments effective at achieving ecological and social objectives? a systematic review. *Forest Ecology and Management*, 375:84–95, 2016. ISSN 0378-1127. doi: <https://doi.org/10.1016/j.foreco.2016.05.021>. URL <https://www.sciencedirect.com/science/article/pii/S0378112716302626>.
- M. Konoshima, C. A. Montgomery, H. J. Albers, and J. L. Arthur. Spatial-Endogenous Fire Risk and Efficient Fuel Management and Timber Harvest. *Land Economics*, 84(3):449–468, Aug. 2008. ISSN 0023-7639, 1543-8325. doi: 10.3368/le.84.3.449. URL <https://le.uwpress.org/content/84/3/449>. Publisher: University of Wisconsin Press Section: Articles.
- J. Kupfer, A. Terando, P. Gao, C. Teske, and J. Hiers. Climate change projected to reduce prescribed burning opportunities in the south-eastern united states. *International Journal of Wildland Fire*, 29, 2020.
- J. M. Lawson, S. J. Foster, and A. C. J. Vincent. Low Bycatch Rates Add Up to Big Numbers for a Genus of Small Fishes. *Fisheries*, 42(1):19–33, Jan. 2017. ISSN 0363-2415. doi: 10.1080/03632415.2017.1259944. URL <https://doi.org/10.1080/03632415.2017.1259944>. Publisher: Taylor & Francis _eprint: <https://doi.org/10.1080/03632415.2017.1259944>.
- J. F. Lehmkuhl, M. Kennedy, E. D. Ford, P. H. Singleton, W. L. Gaines, and R. L. Lind. Seeing the forest for the fuel: Integrating ecological values and fuels management. *Forest Ecology and Management*, 246(1):73–80, 2007. ISSN 0378-1127. doi: <https://doi.org/10.1016/j.foreco.2007.03.071>. URL <https://www.sciencedirect.com/science/article/pii/S0378112707002800>.

- D. B. Lesmeister, R. J. Davis, S. G. Sovorn, and Z. Yang. Northern spotted owl nesting forests as fire refugia: a 30-year synthesis of large wildfires. *Fire Ecology*, 17(1):32, Nov. 2021. ISSN 1933-9747. doi: 10.1186/s42408-021-00118-z. URL <https://doi.org/10.1186/s42408-021-00118-z>.
- S. C. Loeb and R. V. Blakey. Bats and fire: a global review. *Fire Ecology*, 17(1):29, Nov. 2021. ISSN 1933-9747. doi: 10.1186/s42408-021-00109-0. URL <https://doi.org/10.1186/s42408-021-00109-0>.
- K. E. Low, J. J. Battles, R. E. Tompkins, C. P. Dillingham, S. L. Stephens, and B. M. Collins. Shaded fuel breaks create wildfire-resilient forest stands: lessons from a long-term study in the Sierra Nevada. *Fire Ecology*, 19(1):29, May 2023. ISSN 1933-9747. doi: 10.1186/s42408-023-00187-2. URL <https://doi.org/10.1186/s42408-023-00187-2>.
- J. A. Lyons and D. J. D. Natusch. Wildlife laundering through breeding farms: Illegal harvest, population declines and a means of regulating the trade of green pythons (*Morelia viridis*) from Indonesia. *Biological Conservation*, 144(12):3073–3081, Dec. 2011. ISSN 0006-3207. doi: 10.1016/j.biocon.2011.10.002. URL <https://www.sciencedirect.com/science/article/pii/S0006320711003685>.
- I. Martinez-Alvarado and E. Martinez. Trafficking of Totoaba Maw. In *Green crime in Mexico : a collection of case studies*. Palgrave macmillan edition, 2018.
- I. A. Martínez and A. I. Alonso. Mexican organized crime and the illegal trade in totoaba maw. *Trends in Organized Crime*, 24(4):526–546, Dec. 2021. ISSN 1936-4830. doi: 10.1007/s12117-021-09436-9. URL <https://doi.org/10.1007/s12117-021-09436-9>.
- D. Matsypura, O. A. Prokopyev, and A. Zahar. Wildfire fuel management: Network-based models and optimization of prescribed burning. *European Journal of Operational Research*, 264(2):774–796, Jan. 2018. ISSN 03772217. doi: 10.1016/j.ejor.2017.06.050. URL <https://linkinghub.elsevier.com/retrieve/pii/S037722171730591X>.
- M. A. McCarthy, A. M. Gill, and R. A. Bradstock. Theoretical fire-interval distributions. *International Journal of Wildland Fire*, 10(1):73–77, 2001. URL <https://doi.org/10.1071/WF01013>.
- K. McGarigal and B. J. Marks. FRAGSTATS: spatial pattern analysis program for quantifying landscape structure. Technical report, 1995. URL <https://doi.org/10.2737%2Fpnw-gtr-351>.
- J. Minas, J. W. Hearne, and D. L. Martell. A spatial optimisation model for multi-period landscape level fuel management to mitigate wildfire impacts. *European Journal of Operational Research*, 232(2):412–422, Jan. 2014. ISSN 0377-2217. doi: 10.1016/j.ejor.2013.07.026. URL <https://www.sciencedirect.com/science/article/pii/S0377221713006073>. Publisher: North-Holland.
- E. S. Minor and D. L. Urban. A Graph-Theory Framework for Evaluating Landscape Connectivity and Conservation Planning. *Conservation Biology*, 22(2):297–307, 2008. ISSN 1523-1739. doi: 10.1111/j.1523-1739.2007.00871.x. URL <http://onlinelibrary.wiley.com/doi/abs/10.1111/j.1523-1739.2007.00871.x>. _eprint: <https://conbio.onlinelibrary.wiley.com/doi/pdf/10.1111/j.1523-1739.2007.00871.x>.
- T. Mitra and H. Y. Wan. On the faustmann solution to the forest management problem. *Journal of Economic Theory*, 40(2):229–249, Dec. 1986. ISSN 0022-0531. doi: 10.1016/0022-0531(86)90073-6. URL <https://www.sciencedirect.com/science/article/pii/0022053186900736>.
- L. Mouysset, L. Doyen, and F. Jiguet. How does economic risk aversion affect biodiversity? *Ecological Applications*, 23(1):96–109, 2013. ISSN 10510761. URL <http://www.jstor.org/stable/23440820>.

- U. S. G. A. Office. Wildland fire - federal agencies' efforts to reduce wildland fuels and lower risk to communities and ecosystems. Technical report, United States Government Accountability Office, 2019.
- A. D. Olds, K. A. Pitt, P. S. Maxwell, and R. M. Connolly. Synergistic effects of reserves and connectivity on ecological resilience. *Journal of Applied Ecology*, 49(6):1195–1203, 2012. doi: <https://doi.org/10.1111/jpe.12002>. URL <https://besjournals.onlinelibrary.wiley.com/doi/abs/10.1111/jpe.12002>.
- J. S. Olson. Energy storage and the balance of producers and decomposers in ecological systems. *Ecology*, 44(2):322–331, 1963. doi: <https://doi.org.inshs.bib.cnrs.fr/10.2307/1932179>. URL <https://esajournals-onlinelibrary-wiley-com.inshs.bib.cnrs.fr/doi/abs/10.2307/1932179>.
- C. Pais, J. Carrasco, P. Elimbi Moudio, and Z.-J. M. Shen. Downstream protection value: Detecting critical zones for effective fuel-treatment under wildfire risk. *Computers & Operations Research*, 131, July 2021a. ISSN 0305-0548. doi: 10.1016/j.cor.2021.105252. URL <https://www.sciencedirect.com/science/article/pii/S0305054821000447>.
- C. Pais, J. Carrasco, D. L. Martell, A. Weintraub, and D. L. Woodruff. Cell2Fire: A Cell-Based Forest Fire Growth Model to Support Strategic Landscape Management Planning. *Frontiers in Forests and Global Change*, 4, 2021b. ISSN 2624-893X. URL <https://www.frontiersin.org/articles/10.3389/ffgc.2021.692706>.
- L. Pascual-Hortal and S. Saura. Comparison and development of new graph-based landscape connectivity indices: towards the prioritization of habitat patches and corridors for conservation. *Landscape Ecology*, 21(7):959–967, Oct. 2006. ISSN 0921-2973, 1572-9761. doi: 10.1007/s10980-006-0013-z. URL <http://link.springer.com/10.1007/s10980-006-0013-z>.
- D. R. Patton. A Diversity Index for Quantifying Habitat "Edge". *Wildlife Society Bulletin (1973-2006)*, 3(4):171–173, 1975. ISSN 0091-7648. URL <https://www.jstor.org/stable/3781151>. Publisher: [Wiley, Wildlife Society].
- S. H. Peterson, M. E. Morais, J. M. Carlson, P. E. Dennison, D. A. Roberts, M. A. Moritz, and D. R. Weise. Using HFire for spatial modeling of fire in shrublands. Technical report, 2009. URL <https://doi.org/10.2737%2Fpsw-rp-259>.
- J. Phelps, L. R. Carrasco, and E. L. Webb. A Framework for Assessing Supply-Side Wildlife Conservation. *Conservation Biology*, 28(1):244–257, 2014. ISSN 1523-1739. doi: 10.1111/cobi.12160. URL <https://onlinelibrary.wiley.com/doi/abs/10.1111/cobi.12160>. _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1111/cobi.12160>.
- S. Poret. An optimal anti-drug law enforcement policy. *International Review of Law and Economics*, 29(3):221–228, Sept. 2009. ISSN 0144-8188. doi: 10.1016/j.irle.2009.01.005. URL <https://www.sciencedirect.com/science/article/pii/S0144818809000064>.
- J. C. Péreau, L. Doyen, L. R. Little, and O. Thébaud. The triple bottom line: Meeting ecological, economic and social goals with individual transferable quotas. *Journal of Environmental Economics and Management*, 63(3):419–434, May 2012. ISSN 0095-0696. doi: 10.1016/j.jeem.2012.01.001. URL <https://www.sciencedirect.com/science/article/pii/S0095069612000022>.
- R. Rachmawati, M. Ozlen, K. J. Reinke, and J. W. Hearne. A model for solving the prescribed burn planning problem. *SpringerPlus*, 4(1):630, Oct. 2015. ISSN 2193-1801. doi: 10.1186/s40064-015-1418-4. URL <https://doi.org/10.1186/s40064-015-1418-4>.

- R. Rachmawati, M. Ozlen, K. J. Reinke, and J. W. Hearne. An optimisation approach for fuel treatment planning to break the connectivity of high-risk regions. *Forest Ecology and Management*, 368:94–104, May 2016. ISSN 0378-1127. doi: 10.1016/j.foreco.2016.03.014. URL <https://www.sciencedirect.com/science/article/pii/S0378112716300731>.
- R. Rachmawati, M. Ozlen, J. Hearne, and K. Reinke. Fuel treatment planning: Fragmenting high fuel load areas while maintaining availability and connectivity of faunal habitat. *Applied Mathematical Modelling*, 54:298–310, Feb. 2018. ISSN 0307-904X. doi: 10.1016/j.apm.2017.09.045. URL <https://www.sciencedirect.com/science/article/pii/S0307904X17306066>.
- B. Rayfield, D. Pelletier, M. Dumitru, J. A. Cardille, and A. Gonzalez. Multipurpose habitat networks for short-range and long-range connectivity: a new method combining graph and circuit connectivity. *Methods in Ecology and Evolution*, 7(2):222–231, Feb. 2016. ISSN 2041-210X. URL <https://onlinelibrary.wiley.com/doi/abs/10.1111/2041-210X.12470>. Publisher: John Wiley & Sons, Ltd.
- L. Rojas-Bracho and R. R. Reeves. Vaquitas and gillnets: Mexico’s ultimate cetacean conservation challenge. *Endangered Species Research*, 21(1):77–87, July 2013. ISSN 1863-5407, 1613-4796. doi: 10.3354/esr00501. URL <https://www.int-res.com/abstracts/esr/v21/n1/p77-87/>.
- L. Rojas-Bracho, B. Taylor, C. G. Booth, L. Thomas, A. Jaramillo-Legorreta, E. Nieto-García, G. Cárdenas Hinojosa, J. Barlow, S. L. Mesnick, T. Gerrodette, P. Olson, A. Henry, H. Rizo, E. Hidalgo-Pla, and A. Bonilla-Garzón. More vaquita porpoises survive than expected. *Endangered Species Research*, July 2022. ISSN 1863-5407. doi: 10.3354/esr01197. URL <https://research-repository.st-andrews.ac.uk/handle/10023/25767>. Accepted: 2022-08-03T11:30:03Z.
- J. Ruffault, T. Curt, N. K. Martin-StPaul, V. Moron, and R. M. Trigo. Extreme wildfire events are linked to global-change-type droughts in the northern Mediterranean. *Natural Hazards and Earth System Sciences*, 18(3):847–856, Mar. 2018. ISSN 1561-8633. doi: 10.5194/nhess-18-847-2018. URL <https://nhess.copernicus.org/articles/18/847/2018/>. Publisher: Copernicus GmbH.
- A. Rytwinski and K. A. Crowe. A simulation-optimization model for selecting the location of fuel-breaks to minimize expected losses from forest fires. *Forest Ecology and Management*, 260 (1):1–11, June 2010. ISSN 0378-1127. doi: 10.1016/j.foreco.2010.03.013. URL <https://www.sciencedirect.com/science/article/pii/S0378112710001593>.
- V. A. Saab, Q. R. Latif, W. M. Block, and J. G. Dudley. Short-term benefits of prescribed fire to bird communities of dry forests. *Fire Ecology*, 18(1):4, Apr. 2022. ISSN 1933-9747. doi: 10.1186/s42408-022-00130-x. URL <https://doi.org/10.1186/s42408-022-00130-x>.
- Y. Sadovy de Mitcheson, A. W.-l. To, N. W. Wong, H. Y. Kwan, and W. S. Bud. Emerging from the murk: threats, challenges and opportunities for the global swim bladder trade. *Reviews in Fish Biology and Fisheries*, 29(4):809–835, Dec. 2019. ISSN 1573-5184. doi: 10.1007/s11160-019-09585-9. URL <https://doi.org/10.1007/s11160-019-09585-9>.
- C. T. Sahley, J. T. Vargas, and J. S. Valdivia. Biological sustainability of live shearing of vicuña in Peru. *Conservation Biology: The Journal of the Society for Conservation Biology*, 21(1):98–105, Feb. 2007. ISSN 0888-8892. doi: 10.1111/j.1523-1739.2006.00558.x.
- C. E. Shannon. A mathematical theory of communication. *Bell System Technical Journal*, 27 (3):379–423, 1948. doi: <https://doi.org/10.1002/j.1538-7305.1948.tb01338.x>. URL <https://onlinelibrary.wiley.com/doi/abs/10.1002/j.1538-7305.1948.tb01338.x>.

- E. H. Simpson. Measurement of Diversity. *Nature*, 163(4148):688–688, Apr. 1949. ISSN 1476-4687. doi: 10.1038/163688a0. URL <https://doi.org/10.1038/163688a0>.
- N. Singh and X. Vives. Price and Quantity Competition in a Differentiated Duopoly. *The RAND Journal of Economics*, 15(4):546–554, 1984. ISSN 0741-6261. doi: 10.2307/2555525. URL <https://www.jstor.org/stable/2555525>. Publisher: [RAND Corporation, Wiley].
- H. Sitters, J. D. Stefano, F. J. Christie, P. Sunnucks, and A. York. Bird diversity increases after patchy prescribed fire: implications from a before-after control-impact study. *International Journal of Wildland Fire*, 24:690–701, 2015. URL <https://api.semanticscholar.org/CorpusID:129053082>.
- K. Sweeney, R. Dittrich, S. Moffat, C. Power, and J. D. Kline. Estimating the economic value of carbon losses from wildfires using publicly available data sources: Eagle Creek Fire, Oregon 2017. *Fire Ecology*, 19(1):55, Sept. 2023. ISSN 1933-9747. doi: 10.1186/s42408-023-00206-2. URL <https://doi.org/10.1186/s42408-023-00206-2>.
- C. Taylor, M. A. McCarty, and D. B. Lindenmayer. Nonlinear effects of stand age on fire severity. *Conservation Letters*, (7), 2014. doi: doi:10.1111/conl.12122.
- P. D. Taylor, L. Fahrig, K. Henein, and G. Merriam. Connectivity is a vital element of landscape structure. *Oikos*, 68(3):571–573, 1993. ISSN 00301299, 16000706. URL <http://www.jstor.org/stable/3544927>.
- A. R. Templeton, H. Brazeal, and J. L. Neuwald. The transition from isolated patches to a metapopulation in the eastern collared lizard in response to prescribed fires. *Ecology*, 92(9):1736–1747, 2011. doi: <https://doi.org/10.1890/10-1994.1>. URL <https://esajournals.onlinelibrary.wiley.com/doi/abs/10.1890/10-1994.1>.
- L. Tensen. Under what circumstances can wildlife farming benefit species conservation? *Global Ecology and Conservation*, 6:286–298, Apr. 2016. ISSN 2351-9894. doi: 10.1016/j.gecco.2016.03.007. URL <https://www.sciencedirect.com/science/article/pii/S2351989415300421>.
- J. T. Thorson, S. B. Munch, J. M. Cope, and J. Gao. Predicting life history parameters for all fishes worldwide. *Ecological Applications*, 27(8):2262–2276, 2017. ISSN 1939-5582. doi: 10.1002/eap.1606. URL <https://onlinelibrary.wiley.com/doi/abs/10.1002/eap.1606>. _eprint: <https://onlinelibrary.wiley.com/doi/pdf/10.1002/eap.1606>.
- Y. Tian, Y. Liu, C. Y. Jim, and H. Song. Assessing Structural Connectivity of Urban Green Spaces in Metropolitan Hong Kong. *Sustainability*, 9(9):1653, Sept. 2017. ISSN 2071-1050. doi: 10.3390/su9091653. URL <https://www.mdpi.com/2071-1050/9/9/1653>. Number: 9.
- M. G. Turner. Landscape ecology: What is the state of the science? *Annual Review of Ecology, Evolution, and Systematics*, 36(1):319–344, 2005. doi: 10.1146/annurev.ecolsys.36.102003.152614. URL <https://doi.org/10.1146/annurev.ecolsys.36.102003.152614>.
- M. G. Turner and R. H. Gardner. *Landscape Ecology in Theory and Practice*. Springer New York, 2015.
- D. Urban and T. Keitt. Landscape Connectivity: A Graph-Theoretic Perspective. *Ecology*, 82(5):1205–1218, 2001. ISSN 1939-9170. doi: 10.1890/0012-9658(2001)082[1205:LCAGTP]2.0.CO;2. URL <https://onlinelibrary.wiley.com/doi/abs/10.1890/0012-9658%282001%29082%5B1205%3ALCAGTP%5D2.0.CO%3B2>.

- N. Vaillant, J. A. Fites-Kaufman, and S. L. Stephens. Effectiveness of prescribed fire as a fuel treatment in californian coniferous forests. *International Journal of Wildland Fire*, (18), 2009.
- T. Veblen. *The Theory of the Leisure Class: An Economic Study in the Evolution of Institutions. The Original 1889 Scripture of the Work*. Independently published, May 2023. ISBN 9798394370342.
- J. Velázquez, J. Gutiérrez, A. García-Abril, A. Hernando, M. Aparicio, and B. Sánchez. Structural connectivity as an indicator of species richness and landscape diversity in Castilla y León (Spain). *Forest Ecology and Management*, 432:286–297, 2019. ISSN 0378-1127. doi: <https://doi.org/10.1016/j.foreco.2018.09.035>. URL <https://www.sciencedirect.com/science/article/pii/S0378112718312957>.
- D. Wang, D. Guan, S. Zhu, M. M. Kinnon, G. Geng, Q. Zhang, H. Zheng, T. Lei, S. Shao, P. Gong, and S. J. Davis. Economic footprint of California wildfires in 2018. *Nature Sustainability*, 4(3):252–260, Mar. 2021. ISSN 2398-9629. doi: 10.1038/s41893-020-00646-7. URL <https://doi.org/10.1038/s41893-020-00646-7>.
- T. N. Wasserman and S. E. Mueller. Climate influences on future fire severity: a synthesis of climate-fire interactions and impacts on fire regimes, high-severity fire, and forests in the western United States. *Fire Ecology*, 19(1):43, July 2023. ISSN 1933-9747. doi: 10.1186/s42408-023-00200-8. URL <https://doi.org/10.1186/s42408-023-00200-8>.
- Y. Wei, D. Rideout, and A. Kirsch. An optimization model for locating fuel treatments across a landscape to reduce expected fire losses. *Canadian Journal of Forest Research*, 38(4):868–877, Apr. 2008. ISSN 0045-5067. doi: 10.1139/X07-162. URL <https://cdnsciencepub.com/doi/10.1139/X07-162>. Publisher: NRC Research Press.
- B. A. Williams, L. P. Shoo, K. A. Wilson, and H. L. Beyer. Optimising the spatial planning of prescribed burns to achieve multiple objectives in a fire-dependent ecosystem. *Journal of Applied Ecology*, 54(6):1699–1709, 2017. doi: <https://doi.org/10.1111/1365-2664.12920>. URL <https://besjournals.onlinelibrary.wiley.com/doi/abs/10.1111/1365-2664.12920>.
- B. A. Wintle, S. Legge, and J. C. Woinarski. After the megafires: What next for australian wildlife? *Trends in Ecology & Evolution*, 35(9):753–757, 2020. ISSN 0169-5347. doi: <https://doi.org/10.1016/j.tree.2020.06.009>. URL <https://www.sciencedirect.com/science/article/pii/S0169534720301713>.
- T. Wyatt, D. van Uhm, and A. Nurse. Differentiating criminal networks in the illegal wildlife trade: organized, corporate and disorganized crime. *Trends in Organized Crime*, 23(4):350–366, Dec. 2020. ISSN 1936-4830. doi: 10.1007/s12117-020-09385-9. URL <https://doi.org/10.1007/s12117-020-09385-9>.
- D. Yemshanov, N. Liu, D. K. Thompson, M.-A. Parisien, Q. E. Barber, F. H. Koch, and J. Reimer. Detecting critical nodes in forest landscape networks to reduce wildfire spread. *PLOS ONE*, 16(10):e0258060, Oct. 2021. ISSN 1932-6203. doi: 10.1371/journal.pone.0258060. URL <https://journals.plos.org/plosone/article?id=10.1371/journal.pone.0258060>. Publisher: Public Library of Science.
- D. Yemshanov, R. G. Haight, N. Liu, R. Rempel, F. H. Koch, and A. Rodgers. Exploring the tradeoffs among forest planning, roads and wildlife corridors: a new approach. *Optimization Letters*, 16(3):747–788, Apr. 2022. ISSN 1862-4472, 1862-4480. doi: 10.1007/s11590-021-01745-w. URL <https://link.springer.com/10.1007/s11590-021-01745-w>.

- B. Zheng, P. Ciais, F. Chevallier, H. Yang, J. G. Canadell, Y. Chen, I. R. van der Velde, I. Aben, E. Chuvieco, S. J. Davis, M. Deeter, C. Hong, Y. Kong, H. Li, H. Li, X. Lin, K. He, and Q. Zhang. Record-high CO₂ emissions from boreal fires in 2021. *Science*, 379(6635):912–917, Mar. 2023. doi: 10.1126/science.adc0805. URL <https://www.science.org/doi/10.1126/science.adc0805>. Publisher: American Association for the Advancement of Science.
- Y. Zhou and J.-K. Hao. A fast heuristic algorithm for the critical node problem. In *Proceedings of the Genetic and Evolutionary Computation Conference Companion*, GECCO '17, page 121–122, New York, NY, USA, 2017. Association for Computing Machinery. ISBN 9781450349390. doi: 10.1145/3067695.3075993. URL <https://doi.org/10.1145/3067695.3075993>.
- M. 'T Sas-Rolfes, D. W. Challender, A. Hinsley, D. Veríssimo, and E. Milner-Gulland. Illegal Wildlife Trade: Scale, Processes, and Governance. *Annual Review of Environment and Resources*, 44(1):201–228, Oct. 2019. ISSN 1543-5938, 1545-2050. doi: 10.1146/annurev-environ-101718-033253. URL <https://www.annualreviews.org/doi/10.1146/annurev-environ-101718-033253>.

Mots clés: 3 à 6 mots clés

Résumé: Lorem ipsum dolor sit amet, consectetur adipiscing elit. Ut purus elit, vestibulum ut, placerat ac, adipiscing vitae, felis. Curabitur dictum gravida mauris. Nam arcu libero, nonummy eget, consectetur id, vulputate a, magna. Donec vehicula augue eu neque. Pellentesque habitant morbi tristique senectus et netus et malesuada fames ac turpis egestas. Mauris ut leo. Cras viverra metus rhoncus sem. Nulla et lectus vestibulum urna fringilla ultrices. Phasellus eu tellus sit amet tortor gravida placerat. Integer sapien est, iaculis in, pretium quis, viverra ac, nunc. Praesent eget sem vel leo ultrices bibendum. Aenean faucibus. Morbi dolor nulla, malesuada eu, pulvinar at, mollis ac, nulla. Curabitur auctor semper nulla. Donec varius orci eget risus. Duis nibh mi, congue eu, accumsan eleifend, sagittis quis, diam. Duis eget orci sit amet orci dignissim rutrum.

Nam dui ligula, fringilla a, euismod sodales, sollicitudin vel, wisi. Morbi auctor lorem non justo. Nam lacus libero, pretium at, lobortis vitae, ultricies et, tellus. Donec aliquet, tortor sed accumsan bibendum, erat ligula ali-

quet magna, vitae ornare odio metus a mi. Morbi ac orci et nisl hendrerit mollis. Suspendisse ut massa. Cras nec ante. Pellentesque a nulla. Cum sociis natoque penatibus et magnis dis parturient montes, nascetur ridiculus mus. Aliquam tincidunt urna. Nulla ullamcorper vestibulum turpis. Pellentesque cursus luctus mauris.

Nulla malesuada porttitor diam. Donec felis erat, congue non, volutpat at, tincidunt tristique, libero. Vivamus viverra fermentum felis. Donec nonummy pellentesque ante. Phasellus adipiscing semper elit. Proin fermentum massa ac quam. Sed diam turpis, molestie vitae, placerat a, molestie nec, leo. Maeccenas lacinia. Nam ipsum ligula, eleifend at, accumsan nec, suscipit a, ipsum. Morbi blandit ligula feugiat magna. Nunc eleifend consequat lorem. Sed lacinia nulla vitae enim. Pellentesque tincidunt purus vel magna. Integer non enim. Praesent euismod nunc eu purus. Donec bibendum quam in tellus. Nullam cursus pulvinar lectus. Donec et mi. Nam vulputate metus eu enim. Vestibulum pellentesque felis eu massa.

Title: TITLE (English)**Keywords:** 3 - 6 keywords

Abstract: Lorem ipsum dolor sit amet, consectetur adipiscing elit. Ut purus elit, vestibulum ut, placerat ac, adipiscing vitae, felis. Curabitur dictum gravida mauris. Nam arcu libero, nonummy eget, consectetur id, vulputate a, magna. Donec vehicula augue eu neque. Pellentesque habitant morbi tristique senectus et netus et malesuada fames ac turpis egestas. Mauris ut leo. Cras viverra metus rhoncus sem. Nulla et lectus vestibulum urna fringilla ultrices. Phasellus eu tellus sit amet tortor gravida placerat. Integer sapien est, iaculis in, pretium quis, viverra ac, nunc. Praesent eget sem vel leo ultrices bibendum. Aenean faucibus. Morbi dolor nulla, malesuada eu, pulvinar at, mollis ac, nulla. Curabitur auctor semper nulla. Donec varius orci eget risus. Duis nibh mi, congue eu, accumsan eleifend, sagittis quis, diam. Duis eget orci sit amet orci dignissim rutrum.

Nam dui ligula, fringilla a, euismod sodales, sollicitudin vel, wisi. Morbi auctor lorem non justo. Nam lacus libero, pretium at, lobortis vitae, ultricies et, tellus. Donec aliquet, tortor sed accumsan bibendum, erat ligula ali-

quet magna, vitae ornare odio metus a mi. Morbi ac orci et nisl hendrerit mollis. Suspendisse ut massa. Cras nec ante. Pellentesque a nulla. Cum sociis natoque penatibus et magnis dis parturient montes, nascetur ridiculus mus. Aliquam tincidunt urna. Nulla ullamcorper vestibulum turpis. Pellentesque cursus luctus mauris.

Nulla malesuada porttitor diam. Donec felis erat, congue non, volutpat at, tincidunt tristique, libero. Vivamus viverra fermentum felis. Donec nonummy pellentesque ante. Phasellus adipiscing semper elit. Proin fermentum massa ac quam. Sed diam turpis, molestie vitae, placerat a, molestie nec, leo. Maeccenas lacinia. Nam ipsum ligula, eleifend at, accumsan nec, suscipit a, ipsum. Morbi blandit ligula feugiat magna. Nunc eleifend consequat lorem. Sed lacinia nulla vitae enim. Pellentesque tincidunt purus vel magna. Integer non enim. Praesent euismod nunc eu purus. Donec bibendum quam in tellus. Nullam cursus pulvinar lectus. Donec et mi. Nam vulputate metus eu enim. Vestibulum pellentesque felis eu massa.

