Environmental policy for ecosystem services and biodiversity

Preferences for fish conservation and instruments for forest policy

Brian Danley
Centre for Environmental and Resource Economics (CERE)
Faculty of Forest Sciences
Department of Forest Economics
Umeå

Doctoral thesis
Swedish University of Agricultural Sciences
Umeå 2019

Acta Universitatis agriculturae Sueciae

2019:4

ISSN 1652-6880
ISBN (print version) 978-91-7760-326-9
ISBN (electronic version) 978-91-7760-327-6

Print: SLU Service/Repro, Uppsala 2019

Environmental policy for ecosystem services and biodiversity: preferences for fish conservation and instruments for forest policy

Abstract

Threats to ecosystem services and biodiversity are some of the most important contemporary policy problems facing the planet. Consisting of four papers, the motivation for this thesis is threefold: to understand the different ways of defining the benefits humans receive from the environment, how the public may be able to understand more nuanced scientific aspects of biodiversity, and how forest policy instruments are used to encourage family forest owners to safeguard ecosystem services and biodiversity on their properties.

Paper I describes the different ways in which ecosystem services are uniquely defined in the broad ecosystem services literature. Paper II investigates how respondents to a stated preferences survey value a little-known species with an important biodiversity characteristic relative to a familiar species of lesser biodiversity importance. The interpretation of Paper II is that using multiple flagship species targeted toward different members of the general public may be a way to use unfamiliar yet ecologically important species for public outreach purposes. Papers III and IV investigate how family forest owners in Sweden interact with and think about forest policy instruments aimed at preserving biodiversity on their forest properties.

The result of Paper I shows that conceptual ecosystem services definitions exist on a spectrum with some definitions being more characteristic of natural sciences, some more characteristic of economics, and some existing between the natural science and economics definitions. Paper II confirms the appeal of an unfamiliar species with a unique biodiversity characteristic relative to a more familiar species is not limited by geographic distance, but the appeal of such biodiversity may include areas close to its habitat. Paper III shows Swedish family forest owners who are interested in taking environmental efforts on their properties and stay out of forest stewardship certification may be more skeptical of state action to secure biodiversity in Swedish forests. Finally, Paper IV shows the benefits of targeting different kinds of instruments to different kinds of ownership objectives may be limited.

Findings from this thesis may contribute to better environmental policy by improving conservation public outreach efforts and by clarifying the opportunities and challenges of engaging family forest owners in achieving public policy goals.

Keywords: conservation, flagship species, endemic species, forest policy, environmental responsibility

Author's address: Brian Danley, SLU, Department of Forest Economics, 901 83 Umeå, Sweden

Dedication

To my parents for always encouraging my academic efforts and to my wife for being my partner in the journey.

I think nature's imagination is so much greater than man's… she's never going to let us relax.

Richard Feynman

Contents

List of publications

This thesis is based on the work contained in the following papers, referred to by Roman numerals in the text:

I Danley, B., Widmark, C. (2016). Evaluating conceptual definitions of ecosystem services and their implications." *Ecological Economics*, 126, 132-138.

II Danley, B., Sandorf, E.D., Campbell, D. (2018). Paying for biodiversity or familiarity? Investigating distance decay and relative preferences for fish conservation. (manuscript)

III Danley, B. (2018). Skepticism of state action in forest certification and voluntary set-asides: a Swedish example with two environmental offsetting options. *Scandinavian Journal of Forest Research*, 33(7), 695-707.

IV Danley, B. (2018). Forest owner objectives typologies: instruments for each owner type or instruments for most owner types? (manuscript)

Papers I and III are reproduced in compliance with publishing agreements.

The contribution of authors to the co-authored papers included in this thesis was as follows:

I Camilla Widmark had the idea to look at ecosystem service definitions and assisted with writing the manuscript. I conducted the review of the literature, developed the typology of definitions, and wrote the text.

II Danny Campbell had the idea to look at distance decay, assisted with model choice, provided the mapping tools, and helped write the study area and methods sections. Erlend Dancke Sandorf assisted with model choice and estimation, results interpretation, and helped write the methods section. I developed the research question, estimated the model, presented the results, and wrote the majority of the text.

1 Introduction

This thesis spans several topics and methods, but its central concern is environmental policy framed by ecosystem services and biodiversity. Ecosystem services and biodiversity have become prominent features in discussions of contemporary environmental science and policy. Paper I sets the context for environmental policy study at a broad, conceptual level and can be used to frame many contemporary issues, including the contents of this thesis. Paper II is an example of using non-market valuation techniques to estimate the benefits society receives from what are often termed 'cultural ecosystem services' (MESAB 2005) from the existence of fish biodiversity. Papers III & IV investigate forest policy instruments that are aimed to secure a host of implicit ecosystem services (cultural, provisioning, regulating, and supporting) from Swedish forests by preventing trees from being cut.

The Convention on Biological Diversity is the most prominent policy document uniting the empirical papers (CBD 1992). The Convention on Biological Diversity is the most significant international recognition of the importance of biodiversity within species, between species, the diversity of Earth's ecosystems, and by implication the diversity of Earth's ecosystem services. Central to the Convention on Biological Diversity is the idea that conserving biodiversity is compatible with the exploitation of natural resources if it maintains biodiversity stocks and is done in a socially equitable way, i.e. consistent with the tenants of sustainable development. Article 7 of the Convention on Biological Diversity calls for signatory states to conduct environmental monitoring and assessment, which is facilitated via ecosystem services definitions – concepts which are reviewed in Paper I. Article 14 encourages impact assessments of conservation policies, which may include non-market benefits of biodiversity. Paper II investigates non-market benefits of biodiversity using stated preferences methods. The Convention on Biological Diversity also encourages the adoption of "economically and socially sound measures that act as incentives for the conservation and sustainable use of components of biological diversity," implying that countries should diversify the kinds of policy instruments they use and avoid strict command-and-control regulation if appropriate (CBD 1992, Article 11).

Voluntary instruments such as forest certification, contracts for forest conservation with family forest owners, and encouraging landowners to exceed official forest conservation guidelines by leaving voluntary set-asides under the principle of ‘freedom with responsibility’ are examples of such incentives in Swedish forest policy presented in Papers III and IV.

With the Convention on Biological Diversity as the starting point, the policy contexts branch off into biodiversity in marine ecosystems and biodiversity in forest ecosystems. European Union policy influences both contexts. Marine biodiversity serves as an empirical example of environmental policy, while forest biodiversity concerns habitat protection of land. The latter is connected to the EU consensus on Sustainable Forest Management, and expressed in Sweden’s national Sustainable Forests (*Levande Skogar*) goals.

This first half of the thesis is organized to link together the various topics of the four individual papers, explain their policy contexts, and describe the various methods applied in the thesis. The next section will explain how ecosystem services models and definitions, evaluated in Paper I, describe the ways Papers II, III, and IV study the beneficial relationships between human society and the environment. Section 3 introduces the literature review of ecosystem services and why the stated preferences valuation of unfamiliar biodiversity in Paper II addresses a key challenge in the ecosystem services research agenda. To introduce the topics of Papers III and IV, section 4 presents the context of Swedish forest policy and explains the importance of family forest owners in environmental policy goals. Section 5 presents each of the four different theories and methods used in the Papers I-IV in the same order as they are applied in the papers. Due to length restrictions in Paper IV itself, particular attention is given to Paper IV’s method in Section 5. Section 6 summarizes the key research questions, methods, and results from Papers I-IV. I outline the contributions each paper makes to its respective literature and possibilities for future research in Section 7. The two surveys used in Papers II, III, and IV are presented in the appendix.

2 An ecosystem services organization of the thesis

The diverse topics in this thesis are united by various themes from ecosystem services terminology I evaluate in Paper I, which serve as mental models for understanding humanity's relationship with nature. As I claim in Paper I, the foundational theories and frameworks of ecosystem services are inclusive enough to incorporate almost any aspect of nature having a direct or indirect connection to human wellbeing. This section will illustrate the connections between the different topics in this thesis using the two most common conceptual ecosystem service models that describe the relationship between nature and human wellbeing: the service cascade model, and the four categories of ecosystem services from the Millenium Ecosystem Assessment. To demonstrate how Paper I adds a useful dimension to conceptual models of ecosystem services, I also organize the topics of Papers II, III, and IV based on the typology of ecosystem services definitions from Paper I. Placing Papers II, III, and IV into these three different mental models shows how the seemingly diverse issues of non-market valuation, policy instruments, and family forest owner opinions address different issues on the spectrum of the human-nature relationship that must be addressed for successful environmental policy.

First, any ecosystem services framework needs to make some kind of distinction between the physical components of nature (structure), the functioning or interaction between those components (process or function), and nature's resulting contribution to human welfare (benefit or benefit-providing service) (Danley and Widmark 2016). A model of nature's structure, processes, and benefits as a service cascade is one way of describing how the intersection of human wellbeing and nature is studied in this thesis, as illustrated in Figure 1. The cascade model, made famous by authors such as Potschin & Haines-Young (2016), describes the human-nature relationship as a linear flow of goods and services from nature's physical ecosystems to its service-providing functions, to the benefits humans receive from nature.

Economic valuation of nature, such as that produced in Paper II, is inherently a measure of the benefits people derive from a specific part of the environment at a given point in time (e.g. Boyd and Banzhaf 2007). Economic valuations of nature can subsequently be used to inform policy interventions to improve the structures, processes, and service flows from the environment, as shown by the arrow at the bottom of Figure 1 connecting benefits and values back to policy interventions. The forest policy analysis in Papers III & IV are studies of different policy interventions to influence the biophysical structure of forest ecosystems. Because the forest ownership objectives analysed in Paper IV imply there is some kind of benefit to forest management, the ownership objectives typology from Paper IV is an indirect evaluation of forest ecosystem services and benefits as assessed by family forest owners. The connection to forest ecosystem benefits in Paper IV is therefore represented by a dotted arrow compared to the solid arrow for the direct analysis of public policy instruments to maintain forest ecosystems (structures) in Paper IV and ecosystem services and benefits in Paper II.

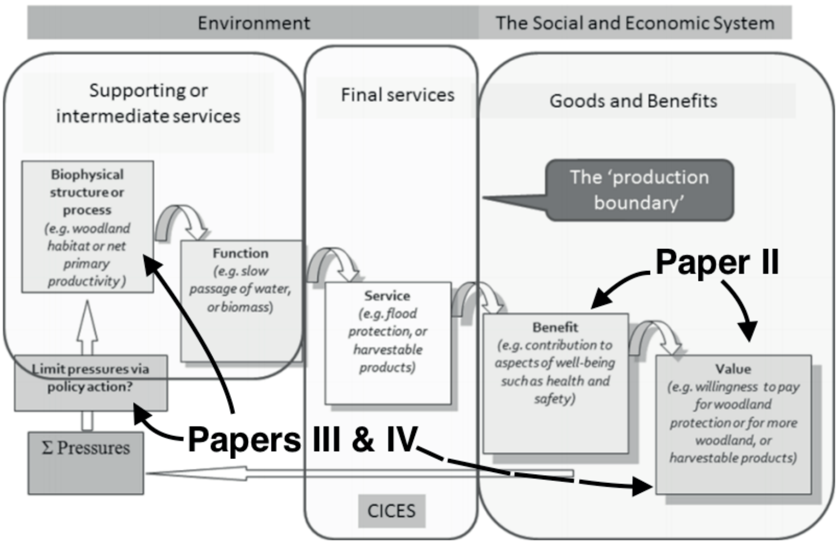


Figure 1. The service cascade model. Adapted from Potschin and Haines-Young 2016. Arrows are added to illustrate the different aspects of the service flow spectrum studied in Papers II, III, and IV.

Second, the four broad categories of ecosystem services articulated in the Millennium Ecosystem Assessment (MESAB 2005) can also describe the direct or indirect aspects of the environment studied in the empirical papers. The four different categories of ecosystem services are one of the most common ways of describing different kinds of benefits humans receive from the environment and is illustrated in Figure 2. Economic values generated from the stated preferences survey in Paper II are best described as cultural ecosystem service values from a unique assembly of fish species in a particular marine catchment. Since the fish species in Paper II are only fished recreationally and not commercially, the existence of and recreational value from conserving the five species in Paper II can be considered as primarily non-provisioning, cultural ecosystem services. The five fish species certainly have a supporting and regulating role in their respective ecosystem, but the economic values derived from the stated preferences survey do not fully capture these intermediate yet important ecosystem services. The policy instruments in Papers III and IV are aimed at securing the full range of supporting, regulating, cultural, and provisioning ecosystem services from Swedish forests for the benefit of the Swedish public in general. Policy instruments in Papers III and IV enhance supporting, regulating, cultural ecosystem services, and some non-timber forest products while reducing some of the timber provisioning ecosystem services from commercial forest management. Family forest owner opinions of forest ecosystem services generally concern the private benefits generated from provisioning and cultural ecosystem services of their forest properties but also their perceptions of the importance of biodiversity in general, which implicitly includes all four kinds of ecosystem services.

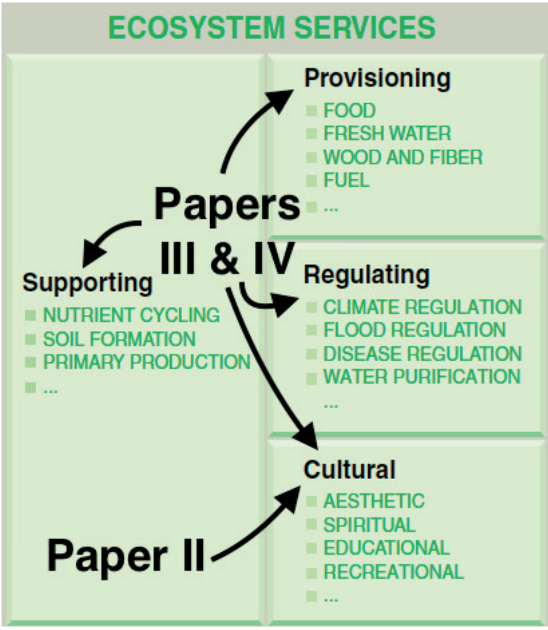
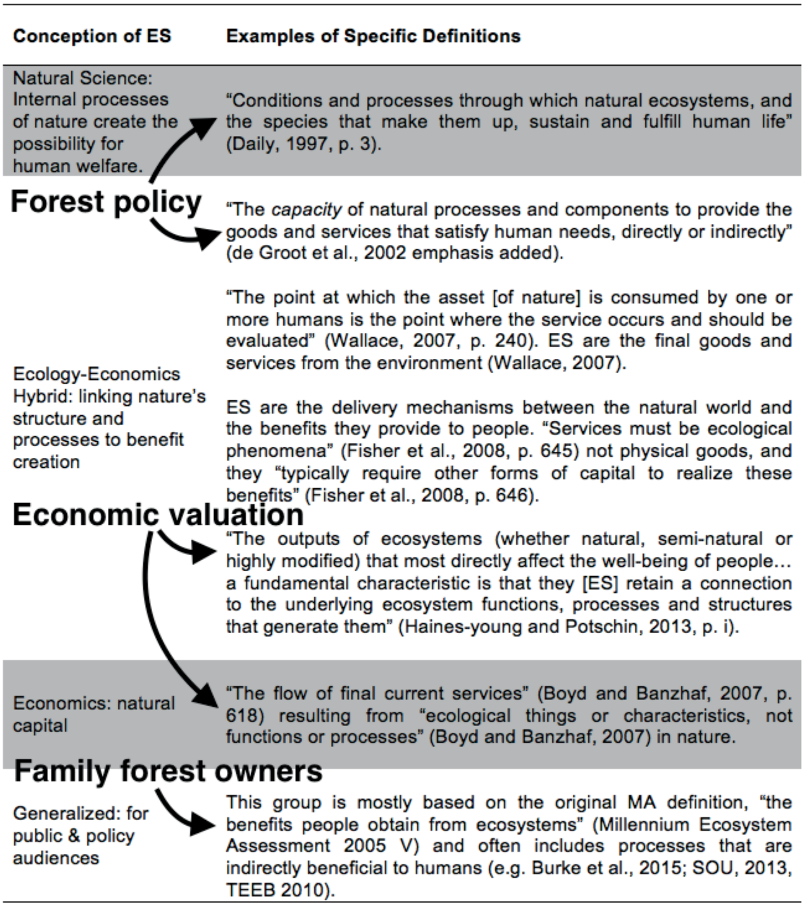


Figure 2. Four types of ecosystem services. Adapted from MESAB 2005. Arrows are added to indicate which of the four ecosystem services are studied in Papers II, III, and IV.

Ecosystem services are not explicitly defined in Papers II, III, and IV, but the different definitions of ecosystem services I identify in Paper I describe how the benefits of nature are approached by economic valuation, Swedish forest policy, and the opinions of family forest owners in the thesis. Figure 3 illustrates which definitions of ecosystem services match to the different central features of Papers II, III, and IV. The forest policy instruments in Papers III and IV aim to keep some of Sweden's productive forests from being felled, thereby enabling the conditions and processes of standing forests to provide goods and services that satisfy human needs. Because the ecosystem benefits of Swedish forest policy instruments are mostly implicit, the definitions of ecosystem services that are characteristic of natural science describe how forest ecosystem services are addressed in the Swedish Forestry Model.

Of course, ecosystem service definitions more consistent with environmental and natural resource economics describe how non-market valuation defines ecosystem services. The monetary values generated in Paper II are a subjective assessment of the service flow resulting from a feature of the environment that directly affects the well-being of people: the existence of five fish species in a specific ecosystem. Swedish family forest owners' opinions of various forest policy instruments and their own ownership objectives assessment concern not only the benefits of forest ecosystems, but also owners' opinions of biodiversity in general. Since the family forest owner opinions I investigate concern a variety of aspects, presumably ranging from the structures and processes inherent in the concept of biodiversity as well as the benefits of forest ecosystems, the generalized definition of ecosystem services based on the Millennium Ecosystem Assessment best matches how I investigate this stakeholder group's relationship with the environment.



Conception of ES	Examples of Specific Definitions
Natural Science: Internal processes of nature create the possibility for human welfare.	"Conditions and processes through which natural ecosystems, and the species that make them up, sustain and fulfill human life" (Daily, 1997, p. 3).
Ecology-Economics Hybrid: linking nature's structure and processes to benefit creation	"The *capacity* of natural processes and components to provide the goods and services that satisfy human needs, directly or indirectly" (de Groot et al., 2002 emphasis added).
	"The point at which the asset [of nature] is consumed by one or more humans is the point where the service occurs and should be evaluated" (Wallace, 2007, p. 240). ES are the final goods and services from the environment (Wallace, 2007).
	ES are the delivery mechanisms between the natural world and the benefits they provide to people. "Services must be ecological phenomena" (Fisher et al., 2008, p. 645) not physical goods, and they "typically require other forms of capital to realize these benefits" (Fisher et al., 2008, p. 646).
	"The outputs of ecosystems (whether natural, semi-natural or highly modified) that most directly affect the well-being of people… a fundamental characteristic is that they [ES] retain a connection to the underlying ecosystem functions, processes and structures that generate them" (Haines-young and Potschin, 2013, p. i).
Economics: natural capital	"The flow of final current services" (Boyd and Banzhaf, 2007, p. 618) resulting from "ecological things or characteristics, not functions or processes" (Boyd and Banzhaf, 2007) in nature.
Generalized: for public & policy audiences	This group is mostly based on the original MA definition, "the benefits people obtain from ecosystems" (Millennium Ecosystem Assessment 2005 V) and often includes processes that are indirectly beneficial to humans (e.g. Burke et al., 2015; SOU, 2013, TEEB 2010).

Figure 3. Ecosystem services definitions and the central features of this thesis. Adapted from Danley and Widmark 2016. Arrows are added to show the aspects of ecosystem services and biodiversity addressed by forest policy instruments, economic valuation, and the opinions of family forest owners.

3 Introduction to ecosystem services and public preferences

3.1 From concepts to economic valuation

Papers I & II are investigations into the terminology used to organize and communicate environmental science and policy and how the public may understand ecosystem services and biodiversity. Paper I describes the different meanings of ecosystem services in the scientific literature along with a commentary on how to interpret ecosystem services as a broader feature of environmental science. This review is an examination of how the conceptual definitions of ecosystem services simultaneously describe nature's benefits to humans as well as how these definitions represent different conceptual approaches to studying the environment. The ambition of Paper I was to determine which aspects of the environment meet the criteria for being labelled an ecosystem service so that I could subsequently focus on those aspects in my thesis. Instead of finding a single criterion for how to identify ecosystem services for environmental policy, four different types of criteria emerge from the vast ecosystem services literature.

So much has been written about ecosystem services from a critical perspective (see review in Schröter et al. 2014), explaining its history in economics (e.g. Gómez-Baggethun et al. 2010), and how ecosystem services should be continuously re-imagined as an evolving concept (Reyers et al. 2012), it is surprising a comparison of the different conceptual definitions of ecosystem services did not already exist. Ecosystem services is a buzzword, but it is also used in ways that ascribe concrete meaning to aspects of the environment that specify certain structures, processes, and benefits from nature as ecosystem services. Classification systems such as the Common Classification of Ecosystem Services version 4.3 (Haines-Young and Potschin 2011) organize many aspects of the environment for practical purposes and are not exclusively concerned with isolating ecosystem services beyond assigning them a categorical designation in an overall classification standard. The contribution of Paper I is to show that despite the similarity in what aspects of the environment are identified as ecosystem services across the academic literature, the "central definitions [of ecosystem services] can reasonably be interpreted as distinct from each other" (Danley and Widmark 2016, p 134) and to describe what these differences are.

As Paper I argues, the foundational literature of ecosystem services intended to make a broadly applicable and persuasive case for biodiversity conservation, but it lacks a theoretical completeness needed to consistently identify exactly what ecosystem services are separate from other aspects of nature (Lele et al. 2013). Significant controversy exists over the importance of consistency and clarity in ecosystem services science, particularly since much of the field aims to inform policy decisions. A good example is the multi-decade debate started by the Costanza et al. (1997) valuation of every ecosystem service on Earth, which has been hailed for raising public awareness despite consistent criticism from many economists that the methodology is deeply flawed (e.g. Toman 1998). Nonetheless, public awareness of ecosystem and biodiversity conservation and sustainability seems to have increased in recent decades. Established in the wake of the UN Rio Earth summit of 1992, subsequent national conservation and sustainable development policies pursuant to the Convention on Biological Diversity represent important progress on global ecosystem services and biodiversity issues. The recent passage of the Paris Climate Agreement, however, stands as a reminder that ecosystem services and biodiversity have been relatively less successful in generating policy attention relative to climate issues (Geijzendorffer et al. 2017). With limited budgets and attention spans, the large spectrum of ecosystem services and biodiversity issues must compete along with other issues, not only environmental matters, for public attention.

The scientific community has invested substantial effort in public outreach and education campaigns concerning biodiversity issues (e.g. Novacek 2008), but how public interest in environmental issues can translate into appreciation of nuanced biodiversity *per se* has long been a debated issue (Sagoff 1988; Mckinley et al. 2017; Young et al. 2014). For example, how might the lay public actually ascribe economic value to ecosystems and species with important biodiversity characteristics that they have never heard of and will probably never experience if there are a host of other issues on which they can focus their attention? In Paper II, I explore an applied version of this question by examining the relative value survey respondents place on the existence of fish species with different familiarity and biodiversity characteristics. Paper II is an investigation of how the public's general interest in environmental concerns may or may not translate into an appreciation of more complex, nuanced scientific aspects of ecosystem services and biodiversity.

3.2 Unfamiliar biodiversity in marine ecosystems as an example

The investigation into relative public preferences for unfamiliar species with biodiversity characteristics in Paper II comes from an interview survey conducted in Northern Ireland and the Republic of Ireland in 2007. The study site itself, the Lough Melvin catchment, has been studied relatively frequently compared to other catchments partially because the catchment exists in both Northern Ireland and the Republic of Ireland and is therefore of political importance. Being conducted well before anticipations of Great Britain's exit from the European Union, the context of the survey in Paper II is influenced by a number of European Union policies concerning marine and biodiversity issues. First, Lough Melvin is a Special Area of Conservation under the EU Habitats Directive (Kelly et al. 2012), making it a part of the Natura 2000 network of protected areas. In addition to the Habitats Directive, the European Union's Water Framework Directive and Marine Strategy Framework Directive both aim to ensure 'good ecological status' of freshwater and coastal marine areas and explicitly integrate economics into water management. Both directives institute natural river basin districts as the units of management for marine areas and call for public participation in policy. Non-market valuation and cost-benefit analysis are two of the tools that member states are encouraged to use in their management of marine areas under the Water Framework Directive and Marine Strategy Framework Directive (European Commission 2016).

Only a minority of water basins in Ireland are, however, on pace to meet 'good ecological status' targets as per the Water Framework Directive (Robins et al. 2017). Despite relatively good ecological conditions in Lough Melvin around the time of this study, phosphorus loadings from agriculture in particular were an increasing cause for concern (Campbell and Foy 2008). In addition to its political importance, the Lough Melvin catchment is important for conservation purposes because it is a unique ecosystem, even if the species investigated in Paper II are not of particularly threatened status. The within species genetic diversity of three kinds of Lough Melvin brown trout in particular make the catchment of special conservation concern based on the ecological argument that "conservation measures should be based on local populations rather than solely on evolutionary lineages or defined taxa" (McKeown et al. 2010, p 343).

While the cost of conservation is often paid from public budgets or the opportunity cost of forgone economic activity, the benefits of ecosystem and biodiversity conservation are generally not fully realized in existing markets. The value of certain cultural ecosystem services from species conservation, particularly the non-use or existence value the general public ascribes to various species, often needs to be estimated using non-market valuation techniques (Brouwer et al. 2013). Estimating economic values of, among other things, biodiversity conservation by asking members of the public to state their economic preferences enables conservation benefits to be put in commensurate terms with conservation costs using cost-benefit analysis (Johansson and Kriström 2015). Stated preferences surveys, such as the one analysed in Paper II, are therefore a way of actively including the public in policy decision making as encouraged in the Convention on Biological Diversity and the Water Framework Directive.

With a particular focus on relative and not absolute values in Paper II, the results my co-authors and I report have a different policy application than classic cost-benefit analysis: which species may be good candidates to select as a flagship species for outreach and education purposes. Flagship species are used as easily recognizable symbols to "increase public awareness of conservation issues and rally support for the protection of [specific] species" (Favreau et al. 2006, p 3,951). Some in the conservation community have been calling for a greater diversity of species to be used for conservation outreach purposes and point to the The International Union for Conservation of Nature's Red List of Threatened Species (IUCN) 'climate change flagship fleet' as an example (e.g. Barua et al. 2011). Methods used in Paper II are one way to explore using more than one flagship species for a given conservation initiative by targeting different members of the general public with different flagship species.

4 Introduction to Swedish forest policy

4.1 Biodiversity in forests as an example

Papers III and IV investigate biodiversity policy in forests in a substantially more applied and detailed way relative to Papers I and II. The broad forest goals outlined at the EU and Swedish levels set the policy backdrop for the two papers about family forest owners (Papers III & IV). No formal EU directives or regulations exist specifically for forest management or forest conservation. Natura 2000's network of protected areas includes almost one quarter of Europe's forest cover (European Commission 2015), arguably making Natura 2000 a *defacto* EU policy on forest conservation and protection. National parks and nature reserves cover 86% of Sweden's land-based Natura 2000 obligations with a high degree of overlap between these two forms of protection and Natura 2000 areas (Hedeklint and Höjer 2017). With a great deal of Natura 2000 obligations satisfied, forest policy in Sweden generally falls under the non-legally binding EU consensus on 'Sustainable Forest Management' as expressed in the EU Forest Strategy. The current EU Forest Strategy, adopted in 2013, states that member states should take a comprehensive approach in their national forest policies that integrates EU policies on topics such as employment and rural development, bioenergy and bioeconomy, climate change policy, and conservation and biodiversity policy (European Commission 2013).

With no EU level directives or regulations strictly steering Sweden's forest policy, domestic policy drives on-the-ground practice concerning forest management. Revisions to the Forestry Act in 1993, which removed the majority of strict regulatory control over forestry issues, state that equal considerations should be given both to public environmental concerns as well as production concerns in forest management (Bush 2010). The notion of equal consideration of environmental and production concerns has been criticized as an ideal that is not reflected in the reality of Swedish forestry by most measures, such as the proportions of forest in production and the proportion in conservation (e.g. Lindahl et al. 2015). Practically, environmental forest policy is articulated in the national-level environmental quality goals of Sustainable Forests (*Levande Skogar*). Sustainable Forests includes a host of specific objectives, including the preservation of existing forest biodiversity, habitat connectivity for biodiversity, and the preservation of cultural artifacts and recreational values (Swedish Environmental Protection Agency 2018a). The most recent evaluation of these environmental objectives shows Sweden is not on pace to reach the 2020 goals set out in Sustainable Forests. The Swedish Forest Agency's suggested response to these deficiencies includes improving environmental considerations and increasing the protection of forests with high ecological values (Swedish Environmental Protection Agency 2017a).

While forests cover almost 70% of Sweden's surface area, due largely to the legal exclusion of forests on national parks and any forest with less than 1 cubic meter of tree growth per hectare per year from productive forest management, productive forests constitute approximately 55% of Sweden's landscape (SFA 2014). Because forests that are not officially protected or legally excluded from commercial management in Sweden are almost exclusively managed using a clear-cut system, forest conservation policy operates in two main contexts in Sweden: either setting forest aside from production for ecological and social reasons, or mitigating the negative environmental impacts of clear-cut forestry. Of Sweden's 23.4 million hectares of productive forest, an estimated 200,000 hectares are final felled on a yearly basis with a further 620,000 hectares being thinned or cleaned (Nilsson 2015). Usually having one of the highest annual harvest volumes in Europe, similar to Finland and Germany, approximately 85 million cubic meters of timber are harvested from Swedish forests on average each year (Nilsson 2015), which accounted for 11% of Sweden's export value in 2013 (SFA 2014). The challenge of meeting the Sustainable Forests targets, much less making environmental and production concerns equal, given such a large-scale intensity of timber production in Sweden means that mitigating the environmental harms of forestry operations is an urgent national priority. Slightly more than half of all productive forests are owned by individuals and families in Sweden (SFA 2014), which means policy instruments to ensure forest management measures meet environmental standards on family owned forests are crucial to half of Sweden's productive forests.

The papers in this thesis specifically look at families and individuals who own forested land in Sweden, called 'individual owners' (*enskilda ägare)* in the Swedish statistics. This ownership class consists of individual persons, or groups of people, who own forested land as opposed to corporations or public entities that own forested land. In the two forest policy papers, I use the terminology 'non-industrial private forest owners,' abbreviated as NIPF owners, simply because this is the most widely used term to talk about this class of Swedish forest owners in the academic literature. An alternative terminology would be 'family forest owners,' abbreviated as FFOs, which is used in contexts such as the United States National Woodland Owner Survey (e.g. Butler 2016). As I mention in Paper IV, the high frequency of commercial forestry operations on this category of forests implies these lands are well integrated into industrial forest management, making the term 'non-industrial' somewhat misleading. On the other hand, many forests in Sweden are now owned by members of multiple households, with the typical example being siblings who have inherited their parents' land and own it together. Accordingly, a more fitting label in the Swedish context might be 'extended family forest owners.' For the purposes of my thesis, both terms are interchangeable. Since the term 'family forest owners' makes clear that this ownership category consists of non-corporate persons who own forests, I refer to this group as family forest owners in the remainder of this thesis introduction.

4.2 Forest policy instruments (the so-called Swedish Forestry Model)

The forest policy instruments I investigate in this thesis are essentially a collection of efforts to mitigate the environmental harms of Sweden's predominately clear-cut forest management system. This collection of forest protection policy instruments and guidelines is often referred to as the Swedish Forestry Model (KSLA 2009) and primarily consists of formally protected areas, voluntarily set-aside areas, and retention structures left in felled areas. This constellation of instruments set in the backdrop of Sweden's deregulated forest policy emerged after revisions to the Forestry Act in 1993 (Lindahl et al. 2015). In the simplest terms, protected areas and voluntary set-asides aim to prevent felling on some of the most ecologically and socially valuable forest in Sweden by excluding them from clear-cutting while the retention structures left during felling aim to mitigate the most negative consequences of clear-cutting. Figure 4 organizes selected instruments from the Swedish Forestry Model for ease of reference.

Formally protected areas form the core of Sweden's forest protection but also include other landscapes and marine areas. Production forestry is expressly excluded on national parks, nature reserves, habitat protection areas, and nature conservation agreements. According to Statistics Sweden in 2017, 1.2% of Sweden's surface area is protected by national parks, which are concentrated in the northern part of the country. Nature reserves cover the most land area of any formal protection status by far at 3,888,516 hectares (Statistics Sweden 2017). Two additional forms of protection cover a smaller area than nature reserves and offer landowners the ability to retain ownership of the property and many rights of use, such as hunting and fishing: Habitat protection areas and nature conservation agreements (Widman 2015). Habitat protection areas are permanent conservation contracts in which, like nature reserves, owners are compensated for the full reduction in property value from the contract plus a 25% supplement. Nature conservation agreements compensate landowners up to 60% of their property value for contracts of up to 50 years (Swedish EPA 2017).

overlap

State policy	**Description**	**Scale of application**
Formal protection by the state	National parks, nature reserves, habitat protection areas, nature conservation agreements	Small, approx 4% of sub-alpine productive forest (ca. 940,000 ha)
Voluntary set-asides	Private contributions. Areas generally between 0.5 and 20 hectares chosen according to forest stewardship certification standards or at the owner's discretion	Small, between 3 and 5% of sub-alpine productive forest (ca. 700,000- 1,175,000 ha)
Clear-cut forestry with retention structures (general considerations)	Retention standards during felling: leave some standing trees, avoid soil damage in riparian areas, protect cultural artifacts from structural harm, leave some deadwood	All managed forests (approx 23,400,000 ha)
Private mechanism	**Description**	**Scale of application**
Forest stewardship certification	Voluntary enrollment of forest property into Forest Stewardship Council (FSC) or Endorsement of Forest Certification (PEFC)	Large, all private company owned forests, state company owned forests, and various other publically owned forests are certified. An unknown percentage of family owned forests are certified.

Figure 4. Selected instruments in the Swedish Forestry Model. Forest stewardship certification overlaps with voluntary set-asides and retention standards for clear-cutting.

The amount of formally protected forest in Sweden is usually divided into forests that are close to the mountainous region bordering Norway or at relatively high elevations (alpine forests) and forests that are below this boundary. The regions indicated as 'region 1' in Figure 5 shows the areas of Sweden considered as alpine forests in which more than half of all forests are formally protected. Forests with relatively lower growth rates and relatively high nature value dominate this area (Statistics Sweden 2017). In forests below the alpine boundary, 2% of forest is protected via nature reserves and habitat protection areas. A further 1% of the forest area is protected by time-limited nature conservation agreements and 1% of land is currently being converted to nature reserves.[1] The long-term goal for protected forests below the alpine forest boundary is 10%. Forests officially protected from commercial operations currently comprise less than half of this target (4%) in non-alpine areas. The remaining area target not covered by formal protection instruments (6%) needs to be supplemented with voluntary set-asides.

[1] Some of the area currently being converted to nature reserves comes from compensating mostly companies that own these areas with productive forest via a company called ESAB, which was specifically created for this purpose. It is not yet clear if the land protected through the ESAB program was entirely composed of previously voluntarily set-aside forest and therefore represents only a change in protection status from voluntarily protected to officially protected.

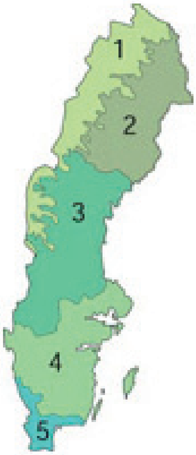


Figure 5. Different forest biomes in Sweden. 1= Alpine, 2=Northern Boreal, 3= Southern Boreal, 4= Boreonemoral, 5= Nemoral. Source Swedish Environmental Protection Agency 2018b

I explain voluntary set-asides in Paper III, but it bears repeating here to provide an introduction to the Swedish Forestry Model as a whole. All forest landowners in Sweden are encouraged to voluntarily set aside parts of their productive forests for conservation purposes. Voluntary set-asides should be at least 0.5 hectares of contiguous, productive forest with one or a combination of high nature value, cultural significance, or social value. The landowner should set the area aside without compensation and any management measures taken in the voluntary set-aside should not damage the natural, cultural, or social value on the stand. The status of the stand should be recorded in a forest management plan (SFA 2012). As of 2016, the total area of forests voluntarily set aside from production is estimated to be 1.2 million hectares, although this is admittedly an over-estimation. The Sustainable Forests goal is that voluntary set-asides will comprise 1.45 million hectares of productive forest by 2020 (SFA 2017). The actual total amount of voluntary set-asides is uncertain since set-aside areas are self-reported by forest owners. Forest companies publish information on their voluntary set asides (Skogs Industrierna 2018)[2] but the vast majority of set-asides on family owned forestland are private information. Depending on how much of the 1% of forest currently being converted to nature reserves were previously voluntarily set-asides, a further 1-2% (approx. 235,000-500,000 hectares) of non-alpine forest needs to be set aside from production to meet the 10% target.

[2] An English version of the map of voluntary set-asides by forest companies can be found at https://www.forestindustries.se/forest-industry/sustainable-development/voluntary-set-aside-forests-in-sweden/map/

Explaining forest voluntary set-asides in Sweden requires a reference to the Forest Stewardship Council (FSC) and the Programme for the Endorsement of Forest Certification (PEFC).[3] Both certification standards use respective national forest legislation to create their country-specific guidelines, but both standards operate completely independent from Swedish government oversight (Lister 2011). Despite FSC and PEFC's status as private, market institutions, their requirement that at least 5% of productive forestland should be set aside from production on all certified forests is an essential mechanism compelling forest owners to leave voluntary set-asides (Johansson 2013). Currently, forest certification standards provide more explicit criteria for how voluntary set asides should be chosen compared to information officially provided by the state (Brukas and Sallnäs 2012). Both FSC and PEFC prohibit cutting on areas that meet the criteria of a Woodland Key Habitat (FSC Sweden 2010; PEFC Sweden 2012), which presumably means that all Key Habitats on certified lands are voluntarily set-aides but the reverse is not necessarily true. According to FSC standards, forest stands that have been sold to the Swedish state or are under contract for nature conservation cannot be considered as voluntary set-asides (FSC Sweden 2010). Although all land-owning forest companies are certified and the vast majority of the forest industry is certified, it is not truly known how much forest owned by individual people and families in Sweden is certified. One estimate put the approximate percentage of individuals and families owning certified forests in Sweden at 17% (Johansson and Lidestav 2011), but this figure was estimated about five years before the survey in this thesis was conducted.

Forest stewardship certification through FSC and PEFC also overlaps with the third pillar of the Swedish Forest Model for environmental harm mitigation: leaving retention structures (*generell hänsyn*, directly translated as 'general considerations') during felling. As I mention in Paper III and IV, the retention practices include leaving some standing trees, leaving some deadwood, avoiding damage to soil in riparian areas, and protecting various cultural features from structural harm, such as old building foundations (Hysing 2009). Landowners are legally obligated to take these measures on any felled forest in Sweden and they are not compensated for the opportunity cost of doing so, which is probably low (e.g. Carlén et al. 1999). Retention structures have increased structural diversity in the Swedish forest landscape (e.g. Kruys et al. 2013; Simonsson et al. 2016), and seem to mitigate some of the more serious consequences of clear felling on affected biota (Gustafsson, Kouki, and Sverdrup-Thygeson 2010).

Important to note is that many practical decisions regarding what to leave and where to physically drive the forest harvesting machinery seem to be made by forestry contractors (Lindroos, Lidestav, and Nordfjell 2005). While forest

[3] An English translation of the FSC standard can be found at the following address. https://se.fsc.org/preview.fsc-forest-management-standard-for-sweden.a-772.pdf The PEFC standard is only available in Swedish.

companies may select and signify certain trees to be retained before felling, it seems that families and individuals who own forests may have a small role, if any, in selecting retention structures relative to contractors who do the actual cutting (e.g. Hogl et al. 2005). Despite landowners' legal obligation to see that retention structures are left during felling, no research exists on the direct question of how much family forest owners are involved in selecting retention structures during clear cutting. The Swedish state has largely left the implementation of retention structures as an issue for the forestry sector to solve as a part of their 'sectorial responsibility' (*sektorsansvaret*) to contribute to environmental goals in the Swedish Forest Model (Swedish Forest Agency 2016).

Forest stewardship certification through FSC and PEFC are designed to play an important role in securing a quantity and quality of forest retention structures that is beyond what is strictly required in Swedish law. Particularly for measures that are difficult or prohibitively expensive to monitor, such as retention structures and voluntary set-asides, forest certification is meant to secure good environmental practices based on the voluntary willingness of owners of certified properties (Romero et al. 2013). Due to the central role of FSC and PEFC certification, the 'soft governance' of the Swedish Forestry Model (Carlsson 2017) is defined by official state policy, the private mechanism of market-driven forest certification, and voluntary efforts on the part of non-certified actors in the Swedish forest sector. Complicating the dual public and private structure of the Swedish Forest Model, timber from certified forests carries a price premium with at least some timber buyers in Sweden, such as the forest owners association Södra Skogsägarna and the forest company Holmen (Villalobos, Coria, and Nordén 2018). Because of the price premium forest owners can expect from FSC or PEFC certification, the financial motivation for certifying a forest property is clear while the environmental stewardship motivation for certification is more ambiguous.

Given the incentives for certification in Sweden, it should be expected that Johansson and Lidestav (2011) found felling activity in areas with higher percentages of family owned forestland under FSC and PEFC certification to be more intense compared to other areas. Relying on FSC and PEFC certification to enhance environmental harm-mitigation efforts in Swedish forestry is to hope that forest owners who demonstrate a particular interest in profit from their forests will voluntarily restrain their commercial activities for environmental reasons. Sweden's heavy reliance on volunteerism and market-driven certification in its forest policy contribute to what Lindahl et al. (2015) identify as an 'implementation deficit' in the current Swedish Forestry Model.

4.3 Family forest owners as actors in the Swedish Forestry Model

Most research about the responsibility of private actors for achieving forest protection goals in Sweden focus on corporate actors (e.g. Simonsson et al. 2016) or critical evaluations of governance processes involving many different actors (e.g. Wallin 2017), but the view of family forest owners on the responsibility for nature protection delegated to them by Swedish law is sparse (e.g. Widman 2016; Löfmarck et al. 2017). Sweden's reliance on volunteerism from its family forest owners is one example of involving private stakeholders in public forest policy, a global trend which has increased in recent decades (Gregersen and Contreras 2010). With rising incomes and awareness of environmental issues in western countries, family forest owners may be increasingly able and willing to manage their properties based on environmental and social ideals. Commensurately, public demands to bring a broader array of forest ecosystem services under deliberate state governance have increased significantly in recent decades, particularly in Sweden (Mårald et al. 2017).

Family forest owner willingness to manage their lands in ways that intentionally or unintentionally provide more public ecosystem services from their forests may be an opportunity in which public and private interests can be brought into closer alignment. The challenge for forest policy researchers and practitioners in many industrialized countries is exactly how to formulate and implement policy that links public demands for non-timber ecosystem services with family forest owners who could be mandated, nudged, or otherwise persuaded to provide the desired services (Cubbage, Harou, and Sills 2007). The Swedish Forestry Model is one example of how landowners, including family forest owners, can be incorporated into public policy in a way that attempts to elicit cooperation and buy-in to policy objectives and methods of policy implementation.

Papers III and IV address family forest owners as actors in the Swedish Forestry Model who are heavily influenced by what I see as an omnipresent yet under-appreciated feature of the Swedish Forestry Model: freedom with responsibility. Sweden, not unlike other northern European countries, often has strong norms to follow laws that may already allow a relatively wide interpretation of what the law actually entails (e.g. Bernitz 2013). It is natural to assume that Swedish family forest owners would feel a relatively strong moral or normative pressure to follow regulations or guidelines from the state concerning management of their forest properties. The fact that Swedish family forest owners can generally be expected to follow norms of environmentally responsible forestry practices is not terribly different from what may be said about family forest owners in, for example, other Nordic countries. What is unique is how the state and the Swedish forest sector, which includes family forest owners, have come to an implicit agreement that the forest sector acting on normative pressure to do what is in the public interest substitutes for the detailed state regulation of forestry repealed in the Forest Act of 1993 (Lister 2011; Lindahl et al. 2015).

Two phrases are commonly used in reference to the self-regulatory expectations on the forest sector: 'sectorial responsibility' (*sektorsansvaret*) and 'freedom with responsibility' (*frihet under ansvar*), which carry the same or similar meaning. For the sake of brevity, I will use the phrase 'freedom with responsibility' to refer to the concept as it pertains to Swedish family forest owners. The phrase 'freedom with responsibility' is never actually mentioned in the Swedish Forestry Act, or any other official forestry legislation for that matter. The concept is only implicit as far as legislation goes, but is discussed as if it is official policy by actors in the forest sector (e.g. Löfmarck et al. 2017), government reports (Riksrevisionen 2018), and the Swedish Forest Agency itself (Swedish Forest Agency 2017). The Swedish Forest Agency's webpage about 'freedom with responsibility' explains that in order to achieve Sweden's dual production and environmental forestry goals, forest owners and managers must do "substantially more than what the law requires" (*betydligt mer än vad lagen kräver*) as it pertains to environmental harm mitigation efforts.

What Sweden has done with encouraging family forest owners to voluntarily exceed loosely enforced guidelines under the principle of 'freedom with responsibility' is to tacitly adopt explicitly normative language and then plan on normatively motivated actions to contribute an essential percentage of forest protection. I argue that 'freedom with responsibility' is important beyond its surface rhetorical appearance and represents a tool of public policy, what Bemelmans-Videc et al. (2003) call a 'public sermon,' that encourages people to do what is in the public interest for other than purely self-interested reasons. Löfmarck (et al. 2017) explore the different ways in which family forest owners understand their various responsibilities as landowners in interactions with forest industry service providers, but do not set the this responsibility in context as a policy instrument to encourage voluntary efforts.

To translate the meaning of 'freedom with responsibility' for an international forest policy audience, I argue this normative concept is an attempt to incorporate what Aldo Leopold famously articulated in his essay 'Land Ethic' (Leopold 1949) as a tool of statecraft. Leopold's idea is that landowners often have ideals of environmental stewardship in mind when managing their properties and envisions a future in which forest management will be driven by the recognition of the intrinsic value of nature. 'Freedom with responsibility' is used a tool of statecraft to encourage all Swedish landowners to contribute substantially more to environmental efforts than what the law requires, which is, by design, necessary to achieve public environmental objectives. Accordingly, Papers III and IV approach Swedish family forest owners' experiences and opinions of the various policy instruments in the Swedish Forestry Model as stakeholders who are "simultaneously private agents maximizing utility from their forest stands as well as public partners with the state" (Danley 2018, p 699).

5 Theory, methods, and materials

Four different methodologies are used in this thesis. This section introduces each methodology in the order in which they appear in Papers I-IV. The conceptual framework implied in Papers III and IV is also explained before presenting the methods of those two papers. Due to length limitations in Paper IV itself, the methodology used in Paper IV is presented in relatively more extensive detail in Section 5.5. A description of the two surveys used for Papers II-IV follows the theoretical and methodological sections.

5.1 Literature review method (Paper I)

The guiding research question framing my reading of the literature was "What are the different criteria that define what ecosystem services are as natural phenomena in a way that uniquely excludes other aspects of nature?" To that end, the sampling method can be described as 'maximum variance sampling' (Patton 2002) since I derived insights by searching for literature that exhibited the greatest differences in how ecosystem services are uniquely and exclusively identified in nature. This method entailed a broad reading of the ecosystem services literature with particular attention to the more frequently cited review and conceptual literature.

The typology of conceptual ecosystem services definitions is the result of reaching 'data saturation' by collecting different definitions until reading additional material added "little in terms of further themes, insights, perspectives or information" (Suri 2011, p72) on how to identify ecosystem services. Paper I is not an exhaustive review of all ecosystem services definitions, but it seeks to describe the different kinds of definitions that can be found in the academic and policy literature. The Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) conceptual framework contests the idea of ecosystem services in general, and is the most important example of conceptual notions of ecosystem services that lie outside the scope of Paper I's typology (Borie and Hulme 2015). Like the IPBES conceptual framework, many critiques of ecosystem services object to anthropocentric or commodification-of-nature implications that are sometimes associated with ecosystem services terminology (e.g. Schröter et al. 2014; Silvertown 2015; Van Hecken et al. 2018). Because most critiques of ecosystem services do not advocate alternative conceptual definitions of how to identify ecosystem services but instead comment on the inherent logic associated with the concept, these critiques are not directly relevant for Paper I's typology.

5.2 Stated preferences theory, method, and application (Paper II)

Respondents to the discrete choice experiment analysed in Paper II are assumed to choose between different conservation alternatives based on which alternative maximizes his or her welfare, or utility. Random utility theory asserts that individuals make utility-maximizing decisions based on parameters that are known to themselves, but are not directly observable to the researcher (McFadden 1973). Due to the lack of actual choices made in incomplete or non-existing markets for goods such as biodiversity, the utility individuals derive from many environmental goods and services must be estimated using stated preferences surveys. Discrete choice experiments are one stated preference method in which individuals choose between different outcomes with varying choice attributes, which include a positive cost associated with some of the outcomes. In discrete choice modelling, individuals' utility for a composite good is comprised of the qualitative characteristics of the good (Lancaster 1966). In this case, the five different fish species in the survey and the cost of conservation are the qualitative characteristics of the good that survey respondents were asked to value. Discrete choice experiments, by design, are well suited to make comparisons between different attributes of particular policy programs (Ben-Akiva and Lerman 1985), which is the main focus of Paper II.

The multinomial logit model is the standard econometric tool for discrete choice modelling, but it has a variety of limitations, such as the assumption that all individuals have homogenous preferences for qualitative characteristics of the economic good (Train 2009). The latent class model is one way to relax this restriction of preference homogeneity by allowing for a degree of heterogeneity among respondents based on observable individual-specific characteristics. In a latent class multinomial logit model, preferences of all individuals are assumed to belong to one of several different classes in which preferences are homogenous within classes and heterogeneous between classes. Other, more complicated ways of incorporating heterogeneity into discrete choice analysis exist, such as the random parameter logit and the latent class mixed multinomial logit model (e.g. Greene and Hensher 2013). The degree of heterogeneity provided by a latent class model is, however, sufficient to address the main research question of heterogeneous preferences for potential flagship species in these survey data. The k-means cluster from Paper IV is another way of assigning individuals to discrete latent classes based on individual-specific characteristics.

A main component of Paper II is distance decay in environmental valuation, which has traditionally been investigated to determine the area and population over which to aggregate benefits of public projects for cost-benefit analysis (e.g. Pate and Loomis 1997; Hanley et al. 2003). Distance decay is said to occur if the economic value ascribed to environmental goods and services reduces, or decays, with increasing Euclidean distance from the environmental good or service. In recent years, applications of distance decay in environmental valuation have expanded beyond defining the geographical range of public project beneficiaries. Looking specifically at environmental sites with recreational use value, Schaafsma et al. (2013) find the close proximity of substitute recreational sites increases the effect of distance decay on stated willingness-to-pay. Other studies evaluating distance decay have looked for spatial patchiness in willingness-to-pay values (e.g. Johnston and Ramachandran 2014), although the economic meaning of such spatial clustering is not always made clear. In general, the economic value of environmental goods and services with use value is expected to decrease with increasing distance from the object of valuation (e.g. Campbell et al. 2009), although the distance decay of non-use value in general is more ambiguous (Concu 2007). For example, some goods with important non-use values such as landscapes with iconic or national status and conservation habitat restoration frequently exhibit no distance decay and may even increase in value further away from the location of the good (Loomis 2000; Rolfe and Windle 2012; Giraud et al. 2010).

One intuitive reason for an absence of distance decay in valuation of iconic landscapes or the conservation of well-known species is because there may be no close substitutes for such goods even at large geographic scales (Bateman et al. 2006). What has not been specifically addressed is if the frequent lack of distance decay in valuation of species conservation (e.g. Wallmo and Lew 2016) occurs due to the biodiversity characteristics of various species or because the appeal of qualities such as familiarity or charisma are insensitive to distance. To my knowledge, Paper II is the first study that attempts to distinguish between the effects of distance on biodiversity value and familiarity value and the first study focusing on distance decay in relative, not absolute, valuation for species conservation. Literature on the policy uses of stated preference valuation for environmental issues, such as distance decay, generally pertain to absolute willingness-to-pay values and cost-benefit analysis (Freeman III, Herriges, and Kling 2014). For example, absolute willingness-to-pay valuation may be used to determine if and how much of an environmental good to provide via public policy programs. The suggested use of the economic valuation study in Paper II is direct public outreach based on which of the choice attributes is relatively most preferred by respondents living relatively closer to or farther away from the study site.

Latent class analysis of discrete choice experiments identifies a limited number of preference types that can be directly interpreted as different market segments and should accordingly respond to different types of marketing messages. One recent suggestion for using latent class analysis is to assist in identifying various species as potential symbols of conservation efforts, or flagship species, among different types of conservation-minded individuals in the general public (Veríssimo, Pongiluppi, et al. 2013; Veríssimo, Fraser, et al. 2013). The prospect of being able to target members of the public who would respond relatively stronger to a nuanced message of biodiversity based on geographical characteristics while using a more traditional flagship species for other members of the public may be a valuable tool for 'conservation marketing' by governments and environmental organizations (Wright et al. 2015). From another standpoint, using the results of stated preferences valuation studies for public conservation outreach allows for a somewhat rare opportunity to validate the results of stated preferences surveys, which could possibly offer insights on how to improve survey methods.

5.3 Family forest owners in a principal-agent framework (Papers III and IV)

My approach to analysis of family forest owners as participants in Swedish forest policy is inspired by what is called the 'principal-agent problem.' Readers with training in economics may take the basic tenets of this analytical framework for granted, but since the intended audience of Papers III and IV includes researchers with a non-economics background, I will briefly outline the problem. An agent is a party contracted to act on behalf of another party, designated as a principal, in some specific matter. Perhaps the most important assumption of this model is that agents' (or potential agents') interests differ from the interests of the principal. The importance of some difference of interest between the principal and the agent cannot be overstated in comparing this conceptual approach in studying family forest owners to, for example, a service logic in which environmental value is co-created with family forest owners (e.g. Matthies et al. 2016). In order to attract and motivate agents to best serve the objectives of a given principal, incentives are employed to reward or punish agents based on observable outcomes from the agent's behavior. A classic question in game theory literature is how to design incentives to maximize the probability that agents are self-interested in acting in the principal's best interest while minimizing the cost of the incentive to the principal (Grossman and Hart 1983).

Two other key dynamics of the principal-agent problem are the observability of the agent's actions by the principal and how to infer the type of potential agent to select for contracting purposes. Cases in which the effort of an agent is imperfectly observable creates a situation called 'moral hazard' in which agents may be rewarded or punished based on an outcome that does not perfectly reflect the efforts of the agent. Cases in which the type of agent (i.e. honest or dishonest) is imperfectly observable when the principal selects an agent are called 'adverse selection' (Cvitanic and Zhang 2013). A simplified figure explaining the basic components of the principle-agent problem is shown in Figure 6.

In Papers III and IV, as well as in this thesis introduction, the principal is the Swedish state acting on behalf of the Swedish public. The interest of the Swedish state is to ensure a socially desirable flow of all non-timber forest ecosystem services from family owned forests as per the Sustainable Forests goals. Family forest owners are agents who are encouraged to take certain actions and avoid others using the public policy instruments of the Swedish Forestry Model as incentives. Although the interests of family forest owners to manage their properties in environmentally and socially responsible ways means they may be somewhat willing to manage their forests in the interests of the state, the basic assumption is there is still a gap between what owners do and what the state wants. In this mental model, policy instruments exist to motivate family forest owners to take additional environmental management efforts that serve the public interest.

It is costly to observe the quality of family forest owner voluntary set-asides and forest retention structures after final felling, making it difficult to know the effort that each owner makes and to motivate additional actions with targeted incentives. Freedom with responsibility gives an unenforceable obligation to family forest owners as a collective of contracting agents to assist in closing the gap in forest protection goals. Family forest owners are also not rewarded for voluntary set-asides, which means observability of forest owner effort is problematic but there may not be a clear problem of moral hazard. Family forest owners are also recruited into forest certification, which may attract owners more inclined to felling (Johansson and Lidestav 2011), but may exhibit no positive or adverse selection based on the quality of forest retention structures (Villalobos, Coria, and Nordén 2018). In the research in this thesis, I approach family forest owners as agents who interact with and have opinions about efforts made by the state, the principal, to persuade them to take actions that are in the public interest but not necessarily in the interest of owners (agents) themselves.

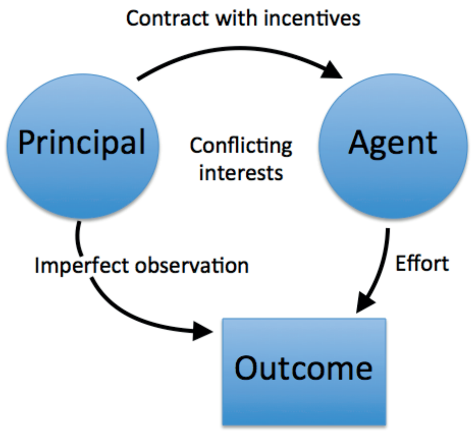


Figure 6. Basic features of a principal-agent framework used to approach family forest owners and policy instruments in Papers III and IV.

5.4 Certification and set-asides as a selection problem (Paper III)

Given the costs of obtaining forest certification and the market benefits it entails (e.g. Villalobos et al. 2018) it is reasonable to assume that set-asides are but one of a host of decisions family forest owners consider when deciding whether or not to certify their properties. Including set-asides as an automatic feature in the larger decision of whether or not to enroll in forest certification is similar to a behavioral nudge in which the default is set to the socially desirable alternative (Thaler and Sunstein 2008), but in this case there should be no way to opt-out of the default according to certification standards. The promise of forest certification is to provide both financial benefits to the landowner as well as secure environmental best practice in forest management (Rametsteiner and Simula 2003), so participation in certification cannot be directly considered as a charitable act, such as joining an environmental organization (e.g. Hossain and Lamb 2012). It is clear that participation in FSC and PEFC certification in Sweden is a signal of family forest owners' financial motivations (Johansson and Lidestav 2011), but it is not clear what participation in certification signals about owners' environmental motivations.

If forest income motivations dominate environmental motivations, there may be adverse selection into certification. In the adverse selection case, FSC and PEFC environmental guidelines act as a restraint on what would otherwise be more environmentally damaging forest practices without certification. Therefore, adverse selection implies that forest owners who are inclined toward relatively more environmentally harmful forestry practices are the owners who tend to benefit from the certified timber price premium. If environmental motivations dominate income motivations in forest certification selection, then FSC and PEFC certification are directing the price premium towards owners who are more inclined toward environmentally friendly forest management. If this is the case, it calls into question the additionality of forest certification's environmental benefit (i.e. owners may have used the same environmentally friendly management measures even without certification.) Empirically, Villalobos et al. (2018) find environmental conditions post felling and the proportion of set-asides are neither positively nor negatively affected by certification, which is inconclusive concerning desirable or adverse selection. If, on the other hand, forest industry recruitment of family forest owners into certification is the dominant explanation for which owners select to certify their properties, then there is no reason to expect desirable or adverse selection. If this effect dominates, then whether or not the various actors in the forest industry overseeing forest management plans and practical forestry operations are following sustainable management practices according to certification standards is the most important factor.

Regardless of remaining questions about how to consider forest stewardship certification on family owned forests, certification functions as a selection mechanism that overlaps with 'freedom with responsibility' appeals for owners to leave voluntary set-asides. Figure 7 describes three possible forest set-aside outcomes given the overlapping voluntary mechanisms family forest owners face, as modelled in Paper III. Since selection (or recruitment) into forest certification is presumably related to the presence of voluntary set-asides on family owned forests, I employ a selection model to account for the relatedness of the three discrete outcomes. Selection models are one way of handling estimation problems in which the dependent variable is unobserved for some respondents. In paper III, I am unable to observe which family forest owners with certified forests would have independently decided to make a set-aside if they were not certified.

Certified	VSA (certification standard)	0	
Not Certified	No VSA	1	0
Not Certified	VSA (unspecified standard)	1	1

Figure 7. Three discrete outcomes modelled via a bivariate probit model with selection (i.e. a binary Heckman selection model). VSA stands for 'voluntary set-aside.'

Selection models have been applied extensively in certain subfields of economics, with an estimation of the reservation wage for labor force participation being the first application (Heckman 1979). Van De Ven and Van Praag (1981) first developed the bivariate probit model with selection in a study of health insurance deductibles to "get rid of an annoying feature" of some missing data in their survey (p. 237). In a bivariate probit model with selection, the selection and the outcome equations are related to each other by the correlation of their respective error terms. In other words, significant correlation between the error terms of the selection equation (certification status) and outcome equation (the presence of set-asides on non-certified properties) imply the existence of unobserved variables that explain both selection and outcome. To paraphrase a long debate in the health and labor economics literature, the appeal of selection models comes at the cost of imposing the estimated variance from the selection equation onto the variance estimation of the independent variables in the outcome equation (Puhani 2000).[4]

There is no definitive way to decide if a selection model is appropriate relative to alternative specifications, but there are at least two reasons to prefer a selection model for investigating family forest owner beliefs, certification, and set-asides. One reason is that beliefs concerning responsibility and environmental efforts can be *a priori* assumed relevant for both the certification decision and the voluntary set-aside decision in combination as opposed to only the outcome (set-aside) decision (Dow and Norton 2003). If there was a clear, logical reason to believe that family forest owner beliefs about responsibility and environmental efforts on their forests had a relevant interpretation for voluntary set-asides and not for certification then a two-part OLS may be a desirable alternative (e.g. Madden 2008). The second reason to prefer a selection model for investigating owner beliefs, certification, and set asides is that non-belief variables can be used as 'exclusion restrictions' in explaining the selection equation while being excluded from the outcome equation (Madden 2008). In Paper III, the variables strongly explaining selection are forest property location and membership in a forest owners association on certification status (i.e. the exclusion restriction variables) and forest size, while the belief variables are only significant in the outcome equation. With exclusion restriction variables (i.e. non-belief variables) having the strongest significance in the selection equation, concerns of multicollinearity impacting the belief variables of interest are mitigated (Puhani 2000). Therefore, a case-specific assessment of whether a selection model is appropriate for exploring Paper III's main research question meets two of Dow and Norton's (2003) three criteria for employing a selection model. Dow and

[4] Without presenting the entireity of the mathmatical formulation here, exactly how estimated variance from the selection equation is imposed in the outcome equation estimation can be found in equation 18 in Van De Ven and Van Praag (1981).

Norton's 2003 third criterion requires an evaluation of the inverse Mills ratio, which does not exist for the bivariate probit selection model.

5.5 Creating family forest owner clusters (Paper IV)

Designing or selecting policy instruments that are appropriate for their intended audience is a popular topic not only in the behavioral literature on the effect of incentives on individuals (see section 3.5), but also in applied policy and public administration, such as the Smart Regulation framework (Gunningham, Grabosky, and Sinclair 1998). Paper IV tests an idea that different types of family forest owners will prefer, or match to, different types of policy instruments that are intuitively consistent with their ownership objectives. In a review of family forest owner typologies in Europe, Ficko et al. (2017) conclude that "except for the concept of consumer segmentation originally coming from marketing research… there have been little efforts to find other theoretical foundation for making PFO [private forest owner] typologies" (p. 9). As I write in Paper IV, "the theoretical foundation for catering policy instruments based on landowner objectives typologies is often implicit in the literature" (p 4). I think it reasonable to assume for practical purposes that the different owner types identified by typology methods should somehow correspond to family forest owners' latent value orientation toward their properties, which should be observable in other matters of family forest owner opinion (e.g. Karppinen 2000). Still, the ambiguous theoretical foundation for the use of family forest owner typologies is a weakness in the literature (Ficko et al. 2017).

Methodologically, Paper IV looks for correlations between owner types and policy instrument types and does not make causal claims that family forest owner objectives types cause policy instrument opinions. Based on results from Paper III, I caution readers that the policy instrument type that most strongly correlates with one of the owner objectives types, forest stewardship certification, may have more to do with non-ownership objectives characteristics that make some owners more likely to be recruited into certification. Although the methods used for Papers II and III are discussed in some detail in the papers themselves, the methodology for creating forest owner typologies is only briefly mentioned in Paper IV and warrants further discussion. Furthermore, I have some critical remarks to make on the standard application of the methods from Paper IV.

Principal component analysis and factor analysis are established methods to reduce the dimensionality of data and facilitate interpretation of relationships in the non-reduced data (Dunteman 1989). Both principle components and factors are essentially linear combinations of non-reduced variables that are created to explain the maximum amount of variance or correlation in the data. The following equations represent the linear combinations of p variables produced by an unrotated principle component analysis that retains as many components as variables (i.e. a standard principle component analysis). X is a vector of p variables and α_n is a vector of p coefficients (or loadings) α_{n1}, α_{n2}, …α_{np}, produced from the principal component analysis (adapted from Jolliffe 2002).

Equation 1

$$\alpha_1 X = \alpha_{11}x_1 + \alpha_{12}x_2 + \ldots + \alpha_{1p}x_p = \sum_{j=1}^{p} \alpha_{1j}x_j$$

$$\alpha_2 X = \alpha_{21}x_1 + \alpha_{22}x_2 + \ldots + \alpha_{2p}x_p = \sum_{j=1}^{p} \alpha_{2j}x_j$$

$$etc. \ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots$$

$$\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots\ldots$$

$$\alpha_p X = \alpha_{p1}x_1 + \alpha_{p2}x_2 + \ldots + \alpha_{pp}x_p = \sum_{j=1}^{p} \alpha_{pj}x_j$$

Note that residuals do not appear in these expressions. Principal components are generated in such a way as to maximize the explained variance in the variance/correlation matrix using the first principle component, $\alpha_1 X$, and subsequent principal components maximize the remaining variance/correlation so that the correlation between all principal components is zero (i.e. the principle components are orthogonal to each other). Since variables in Paper IV are normalized to have a mean of zero and a standard deviation of one before the principal component analysis is conducted (as is common in the family forest owner literature), the data reduction analysis used in Paper IV explains the correlation between the normalized variables (Jolliffe 2002). Residuals will exist if fewer factors than the number of original variables, p, are retained for analysis, which is typically the case in applications of principal component analysis.

A plethora of alternatives are available for the exact specification of principal component or factor analysis. Since the purpose of Paper IV is to test a common assumption, I applied the most common method to create a forest owner typology: Varixmax (orthogonally) rotated principal component analysis with a k-means cluster performed on resulting, individual-specific principle component scores. Since rotation of principle components is so similar to introducing a latent variable structure like factor analysis, it is often argued that a rotated principle component is essentially a factor analysis (Abdi and Williams 2010), although the distinction is not important for this application. Also in accordance with standard practice in the family forest owner literature, I retain only the first m principle components with clearly interpretable loadings (Ficko et al. 2017) such that $m < p$. The remaining ($p \times m$) matrix of principle components is multiplied by an ($m \times m$) orthogonal matrix that is chosen based on the Varimax criterion to increase the interpretability of each component. Note the ($p \times m$) matrix is simply the transpose of the reduced number of principle components from Equation 1 above. Like most rotation criteria, Varimax seeks to increase the interpretability of components by increasing the loadings of variables having a high correlation with respective components and reducing the loadings of variables having low correlation with respective components to minimize the number of variables with intermediate (i.e. close to 0.5) loadings on each component (Jolliffe 2002).

Since Varimax rotation retains the orthogonal relationships between principal components, resulting measures of family forest ownership objectives are not correlated with each other. From a behavioral interpretation standpoint, orthogonality is unappealing in describing relationships between family forest ownership objectives. For example, orthogonality in the factor scores means that recreational and traditional objectives are completely unrelated, which is unlikely to be the case. An alternative would be to allow correlation between principle components using an oblique rotation (Abdi and Williams 2010). Because the objective of rotating principal components is to facilitate interpretation of the data reduction, I strongly encourage others to explore oblique rotation to explain family forest ownership objectives. Had Paper IV been a methods paper about describing family forest ownership objectives, then I would have explored this option in the text, but this methodological issue was outside the scope of investigation.[5]

Also following the most commonly used clustering method, I performed a k-means cluster on individual-specific Varimax rotated component scores. Component scores (also called factor scores) for each respective component are derived by multiplying the rows from the original data matrix by their respective vector of coefficients (loadings) (Dunteman 1989). Component scores are constructed to be mean zero with standard deviation one (Jolliffe 2002), which means the collection of individual-specific component scores will be multivariate normally distributed with dimension m. In non-technical terms, the mathematical expectation of multivariate normal distributions is to exhibit the highest density at the mean of the distribution (in this case with m=3, the expected mean is $\{0,0,0\}$) and dissipate in density from the origin (Gut 2009). In other words, the only expected discrete tendency in trivariate normal component score space is central tendency to the mean of $\{0,0,0\}$. Figure 8

[5] A k-means cluster on the full sample of orthogonally rotated factor scores produces forest owner clusters with qualitatively different charactersitics. Most notably, the 'Family' cluster disappears and a cluster than can be described as 'Passive' owners appears. This difference occurs because the quasi-spherical distribution of orthogonal component scores becomes quasi-eliptical (oblique) with correlation between the component scores. Accordingly, the correlation structure becomes important in the clustering outcomes.

visualizes the spread of the three-dimensional individual-specific principle component scores in two-dimensional space.

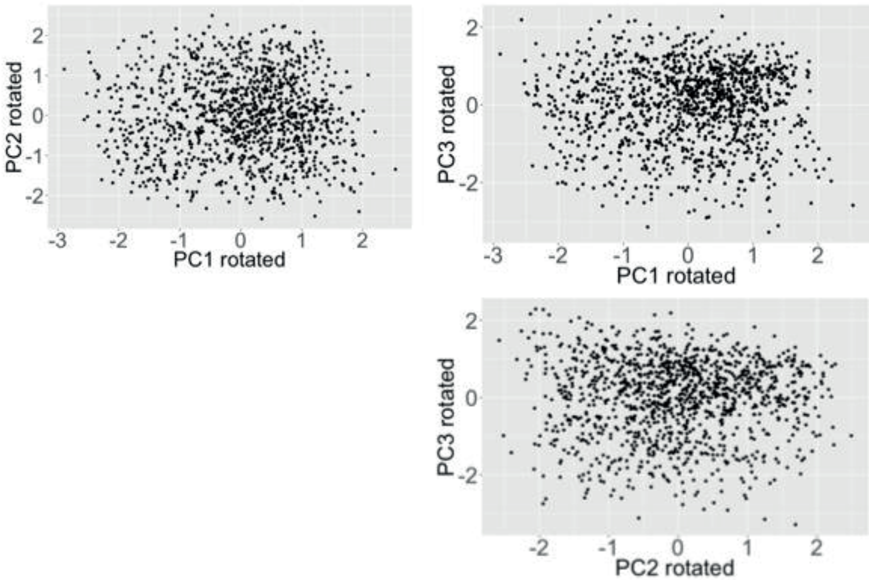


Figure 8. Distributions of individual-specific principle component scores used for discrete clustering created using ggplot2 (Wickham 2016). ‘PC1 rotated’ represents the scores from the first principal component with ‘PC2’ and ‘PC3 rotated’ being scores for component 2 and 3, respectively.

Figure 8 shows there is little intuitive justification for the existence of discrete groupings in this component score space, although discrete clustering algorithms will still produce discrete clusters. In fact, clustering data based on principle component scores produces no statistical advantages compared to clustering based on original variables other than allowing for a visual inspection of the data to search for naturally occurring, discrete clusters (Jolliffe 2002). As I discuss in footnote 5 of Paper IV, it is standard practice in the family forest owner typology literature to cluster family forest owners based on retained principle component scores that capture somewhere between 40 and 60% of the cumulative variance in the original data. Therefore, the typology I produce in Paper IV and many typologies in the family forest owner literature exclude a large amount of information from the non-reduced data. An alternative would be to run a cluster on the original, non-reduced data and interpret resulting clusters based on the original variables. I have found no discussion on why principle component analysis is done before clustering in the family forest owner typology literature, but I suspect its main purpose is parsimony in explaining objectives and comparability of those objectives across studies that ask landowners different questions in different countries.

Again, solving this methodological issue is not the purpose of Paper IV, but it suffices to say that describing family forest ownership objectives with a few parsimonious components sacrifices a substantial amount of information; around half in most cases in the family forest owner typology literature.

A standard k-means clustering algorithm was applied to find a pre-determined number of clusters in $m=3$ dimensional component score space. Using the Euclidian distance between individual-specific principle component scores as the distance metric, the k-means algorithm minimizes the sum of the squared error over all k clusters (i.e. within-cluster sum of squares) as follows (Jain 2010):

Equation 2

$$J(C) = \sum_{k=1}^{K} \sum_{x_i \in c_k} \|x_i - \mu_k\|^2$$

where x_i are the 3 dimensional component scores to be assigned to K different clusters, denoted by $C = \{c_1,\ k= 1, \ldots\ c_K,\ k=K\}$. The mean of each cluster is represented by μ_k and J is the objective function to minimize. K is chosen by a heuristic criterion, in this case the interpretability and comparability of cluster characteristics with other typologies in the family forest owner literature. Because any given iteration of the k-means algorithm converges to a local optimum given the starting values of μ_k and assignment of each x_i to various clusters in C (Marina 2006), fifty randomized vectors of starting values were used to confirm results from the built-in kmeans function in R (R Core Team 2018) produced a globally optimal clustering of survey respondents. Mean scores of each cluster, μ_k, represent how respondents answered forest ownership objectives questions relative to other owners and not in absolute terms. For example, the 'Profit oriented' cluster identified in Paper IV tended to ascribe higher importance to pecuniary (financial) objectives and lower importance to recreational and traditional objectives relative to other owners and not necessarily in absolute terms.

Given the distribution of individual-specific component scores, x_i, shown in Figure 8, the boundaries between clusters from the k-means algorithm are almost guaranteed to be arbitrary. The weighting of the normalized principle component scores and weighted partitioning of the sample serves a dual purpose: one is to see if two groups that are underrepresented among the population of all Swedish family forest owners due to sampling based on properties makes a difference in results. The other purpose is to provide a clustering outcome that can be expected to have different boundaries between clusters. Making small changes to the principle component analysis and which observations are included in the clustering is a straightforward and common way of checking if clustering results are robust (Everitt et al. 2011). In Paper IV, I created three different cluster assignments among respondents, not to check the internal validity of the method, but to test the external validity of somewhat different clustering solutions against forest policy instrument experiences and opinions. Despite the critiques I have made of the standard family forest owner typology methods, typologies created using variations of the standard method still showed no better matching with opinions and experiences of forest policy instrument types than what I find in Paper IV. Nonetheless, it is important to explore variations of the standard methods to describe family forest ownership objectives based on which variations best fit the specific research question of investigation.

5.6 Data (the surveys - Papers II, III, and IV)

Data from two surveys are analysed in this thesis. For Paper II, stated preferences data were examined from a survey gathered to estimate the existence value of a select subgroup of fish species in the Lough Melvin Catchment in Ireland. The survey was conducted via in-person interviews in 2007, well before anticipations of Great Britain's exit from the European Union. The population of interest was the adult population of the Republic of Ireland and Northern Ireland. The study adopted a stratified random sample to reflect the geographic distribution of the adult population, the approximate rural/urban split, the approximate socio-economic status of the regional population, and the approximate gender and age profile of the populations within both jurisdictions.

The survey involved numerous rounds of design and testing before interviews were conducted to confirm that respondents could meaningfully interpret the attribute levels. A series of six focus group discussions were held consisting of meetings in the Lough Melvin Catchment area and four discussion meetings in other areas of Northern Ireland and the Republic of Ireland. Focus group participants included several stakeholder groups such as farmers, foresters, anglers, and members of the general public. Input from the focus groups and subject matter experts was used to refine the wording, options, and layout of the choice experiment. Furthermore, a pilot survey of over 100 respondents was conducted to gather additional information and test the survey design. Feedback from the focus groups, pilot study, and expert opinions indicated that people found the experiment to be meaningful and credible. The choice experiment was based on an experimental design using an algorithm that minimized the variance of the sum of the marginal willingness to pays (Scarpa and Rose 2008) and invoked Bayesian assumptions informed on estimates from pilot studies (Vermeulen et al. 2011).

The survey contained five different parts. After first inquiring as to basic sociodemographic information, the second part inquired about respondents' attitudes toward the environment and conservation in general. The third was the valuation exercise in which the discrete choice experiment was conducted with the aid of prompt cards. A fourth section asked follow-up questions about the valuation respondents had just performed. The fifth and final section collected various information including respondent income and employment, if they had ever fished in the Lough Melvin catchment, and their general impressions of the survey. Respondents were shown pictures of the Lough Melvin catchment and the fish species included in the discrete choice experiment. Respondents were informed the following information about the fish species in the discrete choice experiment: Sonaghan only exists in Lough Melvin and will go completely extinct if they disappear from the catchment; the population of Arctic char is the last such population in Northern Ireland, with populations of Arctic char in decline across catchments in the Republic of Ireland as well; and Atlantic salmon, Gillaroo and Ferox are present in other catchments across both Northern Ireland and the Republic of Ireland. Finally, respondents were told that all species except the Arctic char have angling potential. Because respondents were told to consider the Sonaghan, Ferox, and Gilleroo as a separate species, they are considered as such for the purposes of this study. The full survey and selected prompt cards, including a sample choice card, are presented in the appendix.

The second survey analysed in Papers III and IV come from a postal mail survey of Swedish family forest owners sent out in December 2014 with a reminder and duplicate survey sent to those who had not yet responded in January of 2015. Survey recipients were selected using a proportionate stratified sampling method based on the county in which their forest property exists (Frayer and Furnival 1999). This yielded a list of 3,000 forest owners with a registered Swedish address and more than 5 hectares of forest, but due to duplicates, 2,987 unique owners received a survey. Forest properties larger than 5 hectares were selected since most of the forest policy instruments in the Swedish Forestry Model mitigate the environmental damages of commercial forestry, and properties should be large enough for commercial management to be feasible. Of the 2,987 recipients receiving a survey, 1,296 were returned with 32 of those being blank. Overall this survey had a response/cooperation rate of 42%, which is comparable to other family forest owner mail surveys (e.g. Arano and Munn 2006; Joshi and Arano 2009; Häyrinen et al. 2014; Kumer and Štrumbelj 2017). A small percentage of respondents (4%) took the option to complete an online version of the survey, which was offered to help increase the response rate. Investigating differences among respondents who answered the online survey shows no statistically significant survey mode effect based on the following main variables: certification status, voluntary set-asides, policy opinions, and forest property characteristics.

Sweden is unique in the possibility it offers for sampling landowners. A database with detailed information on all registered persons in Sweden contains detailed socio-economic information on all individuals in the country who own a forested property (Haugen, Karlsson, and Westin 2016). The database allows researchers to know some characteristics of the population of all Swedish family forest owners, such as owners' age distribution. The survey used in Papers III and IV sampled forest owners based on forest property, which is not the same as sampling based on the population of all owners since many forest properties are owned by multiple people. Figure 9 compares survey respondents to both the targeted sample of owners (survey recipients), which are collected based on forest property, and the population of all family forest owners in Sweden as reported by both the the Swedish Forest Agency (2014), and Haugen et al. (2016). Because forest properties can have more than one owner, it is possible to have a sample with no detectable response bias based on geographic and demographic differences between survey respondents and the targeted sample of all survey recipients, but still not be representative of all family forest owners.

Chi-squared difference of proportions tests on all variables available for comparison show no detectable non-response bias based on county, region, forest size, or percentage of sole owners. Comparing all survey respondents to the population of all Swedish family forest owners shows the data under-represent female forest owners and forest owners under the age of 65. Since older individuals and men are more likely to be sole owners of Swedish forests (Lidestav 2010) and respondents were sampled based on forest properties, the under-representation of these two groups is expected. The sample of family forest owners used in Papers III and IV can be considered representative of a sample of forest owners based on forest properties, but not of all Swedish family forest owners.

	Respondents	**Survey recipients**	**All Swedish NIPF owners***
Region in Sweden			
South	0.44	0.41	0.42
Middle	0.26	0.28	0.30
North	0.31	0.31	0.27
Sole owners	0.37	0.39	0.33
Forest size (avg. hectares)	53.5	53.4	0.48
Female	0.24	na.	0.38
Over 65	0.46	na.	0.30

*source SFA 2014 & Haugen et al 2016

Figure 9. Variables used for non-response analysis. Comparing shares of respondents, all owners receiving a survey (non-respondents and respondents), and all Swedish NIPF owners.

The family forest owner survey consisted of six parts. The first part contained questions about respondents' forest properties including certification status, if the owners were members in forest owners associations, and forest ownership objectives questions. Second, respondents were asked about their opinions concerning Swedish forest policy in general and the nature conservation actions that had been taken on their properties. A third section asked owners about the sources of information they prefer to use when considering various aspects of their properties. Next, respondents were asked a series of questions about future possibilities for conservation-related actions and alternative management methods. Finally, respondents were asked for their sociodemographic information, including the percentage of their income that comes from their forest properties over a five-year period. A translated version of the family forest owner survey follows the stated preferences survey in the appendix.

6 Overview of the papers

“Evaluating conceptual definitions of ecosystem services and their implications”

‘Ecosystem services’ is a phrase with many meanings, yet few studies have primarily focused on comparing different definitions of the term. Ecosystem services are now generally used in identifying an appropriately wide range of environmental variables for policy and management as well as better understanding the benefits provided by those aspects of the environment, but the term is also a designation assigned to various aspects of nature. This study describes the different ways in which ecosystem services are uniquely defined in the broad ecosystem services literature to create a typology of different kinds of conceptual definitions. First, the foundational ecosystem service literature is briefly presented to argue that the term was first developed to advocate for the importance of conserving the Earth’s ecosystems and biodiversity. Given the public policy agenda of the earliest ecosystem services literature, theoretical consistency and clarity for exactly what aspects of nature qualify as ecosystem services was not a main priority.

Some ecosystem services frameworks and organizational classification systems do provide consistency for organizing environmental science and policy to a certain extent. The categories of supporting, regulating, provisioning, and cultural ecosystem services is an example of a widely used ecosystem services framework, but this framework does not define what aspects of nature qualify to be placed into these four categories. The most frequently cited conceptual definitions to identify ecosystem services in a way that uniquely excludes other aspects of nature are, in some ways, reasonably different from each other. Conceptual ecosystem services definitions exist on a spectrum with some definitions being more characteristic of natural sciences, some more characteristic of economics, and some existing between the natural science and economics definitions.

Definitions that are more similar to natural science tend to emphasize the importance of conditions and processes that make human welfare possible or generate the capacity for humans to benefit from the environment. Ecology-economics hybrid definitions make a finer distinction between which aspects of nature's structures (physical ecosystems), processes (ecological phenomenon), and benefits to humans can be called an ecosystem service and in what context. Some definitions see ecosystem services as the last distinct process or function of nature to deliver a benefit to human beings. Others employ the concept of 'intermediate services' to distinguish ecological processes that do not directly benefit humans from final ecosystem services. Conceptual definitions from an economics tradition tend to measure the benefits from the environment as a proxy of final service flows, even if the service flow itself is a biophysical structure or process. Finally, the generalized definition of ecosystem services as both the benefits people obtain from ecosystems and the indirectly beneficial aspects of the environment is often used for public or policy audiences. The generalized definition can conflate a number of more nuanced distinctions of what is and is not an ecosystem service.

The paper concludes by observing that despite the success of ecosystem services science, there are still some ambiguities and contradictions in what can actually be identified as an ecosystem service. Since the term 'ecosystem services' was created to be as inclusive as possible in describing how humanity benefits from the environment, using ecosystem services as a terminology may not uniquely identify novel aspects of the environment for science and policy.

Paying for biodiversity of familiarity? Investigating distance decay and relative preferences for fish conservation.

The public's interest in conservation but often limited scientific knowledge of ecosystems and biodiversity may pose a challenge for how to use public opinion for environmental policy. Easily recognizable species are often used as so-called 'flagship' species to raise awareness and funding for conservation action, but this practice has been criticized for neglecting low-profile species with important biodiversity status. How the public ascribes economic value to species with important scientific characteristics but are previously unknown to them given the presence of more traditionally familiar flagship species is important since much of the world's important biodiversity is often unknown to the public. One component of biodiversity is the geographic distribution of where species live, with species that live in only one habitat being endemic to that particular habitat. This study investigates how respondents to a discrete choice experiment ascribe value to the conservation of five different fish species with one species being non-endemic to the study area and familiar to most respondents while another, much lesser-known species, is endemic to the study area.

This paper uses stated preferences data asking respondents in Northern Ireland and the Republic of Ireland to value the existence of five different fish species in the Lough Melvin catchment, which straddles the border between Northern Ireland and the Republic of Ireland. Using a latent class model, we investigate possible distance decay effects in which species respondents prioritize for economic valuation. The Euclidean distance, in kilometers, between the study site and the location of each respondent's primary residence is included as a covariate in the latent class membership function. Other variables describing the location of respondents' primary residences are also included in the latent class membership function along with an indicator variable if the respondent is a recreational angler.

Results show that among the two classes with high willingness to pay for conservation, one class has a stronger relative preference for the familiar, non-endemic species while another class has the strongest relative preference for the non-familiar species endemic to the study area. There is no significant monotonic distance decay in which species respondents tend to prioritize for valuation. In other words, the appeal of an unfamiliar species with a unique biodiversity characteristic relative to a more traditional choice for a flagship species is not limited by geographic distance. Using conditional estimates for the marginal rate of substitution between the unfamiliar endemic species and the familiar non-endemic species, one of the areas with a high relative preference for the endemic species exists within 50 kilometers of the endemic habitat, Lough Melvin. Using a Kriging prediction of the importance respondents said they ascribed to the endemic species in the valuation exercise also confirmed the geographic appeal of the endemic species is not geographically limited, but seems to include areas close the Lough Melvin. The interpretation of these results is that using multiple flagship species targeted toward different members of the general public may be a way to include unfamiliar yet ecologically important species for public outreach purposes. Adding novelty to the literature, this study shows how individuals who live relatively close to unfamiliar biodiversity may be among those who are more likely to value such biodiversity higher relative to familiar substitute species.

Skepticism of state action in forest certification and voluntary set-asides: a Swedish example with two environmental offsetting options

Non-industrial private forest owners (family forest owners) in Sweden are encouraged to mitigate environmental damages from forestry on their properties under a principle of "freedom with responsibility," although the level of mitigation is generally left to the owners' discretion. One voluntary measure private forest owners are encouraged to take is setting aside part of their productive forests for conservation. The purpose of this paper is to evaluate how non-industrial private forest owner beliefs concerning both their own and the Swedish state's responsibility for nature protection differ among owners of certified forests, who automatically leave a set-aside, and those who have stayed out of forest certification but have decided to leave a set-aside. A Heckman selection bivariate probit model is used to explore the effects of forest property characteristics and forest owner beliefs concerning responsibility on the relatedness of the certification and set-aside decisions. Results show that the more a respondent believes the state is responsible for fulfilling environmental goals compared to private forest owners, the less likely it is that an owner of a non-certified forest will leave a set-aside for conservation. Beliefs about responsibility do not, however, differ among owners of certified and non-certified forests. From a policy perspective, Swedish government agencies may have difficulty steering specific measures takes by private forest owners who are interested in conservation but have stayed out of forest certification regimes.

Forest owner objectives typologies: instruments for each owner type or instruments for most owner types?

The extensive literature on non-industrial private forest owner typologies often assumes that different kinds of landowner types will respond to different policy instruments according to shared forest ownership objectives. Although forest owner typologies using principal component analysis and subsequent k means clustering techniques are prolific, the surprisingly little empirical work linking forest owner objectives with forest owner opinions or experiences of different policy instruments shows ambiguous support for recommended targeting efforts. This study uses some standard tools of analysis for private forest owner objectives on a survey of Swedish forest owner opinions concerning various forest conservation policy instruments.

Four assertions from the literature are identified as to how different non-industrial private forest owner types should respond to different policy instrument types. First, owners ascribing a high priority to financial (or pecuniary) objectives are more likely to respond positively to economic instruments. Next, owners giving a high priority to non-pecuniary objectives or ascribe a high priority to all objectives are more likely to respond positively to policy instruments or take voluntary environmental measures. Third, owners who give low priority to all ownership objectives are less likely to respond to any particular instrument or take voluntary measures. Finally, all non-industrial private forest owners are inclined to disapprove of prescriptive regulations. Based on these assertions and the specific policy environment for private forest owners in Sweden, 8 related hypotheses are developed to evaluate if owner types are more or less likely to hold opinions of various forest policy instruments according to the four assertions.

Results show some significant but overall weak relationships between ownership objectives and Sweden’s command and control green tree retention measures, participation in voluntary forest stewardship certification, acceptance of a hypothetical financial incentive, and overall interest in taking more environmentally beneficial forest management measures. These results suggest the benefits of targeting different kinds of instruments to different kinds of ownership objectives may be limited, but it is advisable to design any given policy instrument to be compatible with multiple ownership objectives.

7 Contributions and future studies

This thesis spans two main topics, but the uniting theme is environmental policy for ecosystem services and biodiversity. The first two papers are investigations into how environmental science is organized and studied and how the general public may be able to prioritize some of the more nuanced aspects of ecosystem services and biodiversity given their often limited prior familiarity with such biodiversity. The last two papers evaluate how family forest owners think about their normative responsibility for nature protection, their ownership objectives, and how they interact with the Swedish state's efforts to secure national forest conservation goals. This final section will discuss some contributions this thesis makes to the literature and suggest further topics for research.

Paper I gives a practical guide to understanding how ecosystem services are generally defined in the vast ecosystem services literature. Like any other term, ecosystem services do not need to be carefully defined in every study that mentions them. However, when claims are made about the state of knowledge concerning ecosystem services or their novelty among other features of environmental science, clarity and specificity in definitions are important. Economics and numerous natural science disciplines have evolved in parallel to continuously developing ecosystem services language, which means ecosystem services organizational frameworks may offer competing definitions of ecosystem services in some cases. To the point that ecosystem services language facilitates the interface of environmental science, policy, and the general public, it should be embraced. The organizational features of ecosystem services that hinder clarity and communication may instead be worth revising or discarding. One example of inconsistency in ecosystem services terminology is the often contradictory ways of defining 'intermediate ecosystem services,' which Potschin-Young et al. (2017) argue is so problematic that the term should be discarded entirely.

Future studies could include a discourse analysis of how the human-nature relationship is constructed in ecosystem services definitions, classification systems, and the broader literature. Given the initial use of ecosystem services terminology to warn of impending catastrophe and inspire conservation action, some of the unconscious narratives that are reproduced in ecosystem services literature are worth exploring. Also, implicit in discussions of how to define ecosystem services is the notion that the public should be able to appreciate how human wellbeing depends on the complexities of ecosystems and biodiversity and live sustainably based on that appreciation. A key challenge I see in engaging the public in ecosystem and biodiversity issues is how to present a message based on sound science that neither overcomplicates the issues, leading to a decrease in effective outreach, nor romanticizes nature in a way that is incompatible with science. Further investigations into how well the public may be able to appreciate the nuances of ecosystems and biodiversity and take action accordingly, such as Paper II, are therefore needed.

Paper II finds that when members of the general public are asked to put an economic value on fish biodiversity in a particular ecosystem, a sizable subgroup gives higher priority to an unfamiliar species with an important biodiversity characteristic relative to a species that is more familiar. From the literature on stated preferences and distance decay, it appears that people everywhere care about non-use conservation value regardless of where that conservation is located. Only a small subset of the Earth's biota has the advantage of being immediately recognizable and perhaps even eliciting a positive emotional response from the public. Yet popular species are not necessarily species worth prioritizing for conservation.

Using stated preferences survey results for direct public outreach and conservation marketing, as Paper II suggests, is a potential area for future research. Based on the results of Paper II, it is possible to imagine testing different logos of a conservation program in different geographic areas with one logo emphasizing a familiar species and the other emphasizing a little-known, yet important, biodiversity characteristic. To increase the effectiveness of using stated preferences valuation for practical conservation marketing in general, those employing such methods can borrow suggestions from the literature on making valuation more useful in real-world applications (e.g. Nursey-Bray et al. 2014). For example, involving stakeholder groups in the formation, execution, and implementation of using stated preferences results for policy purposes has been associated with greater policy success (e.g. Börger et al. 2014; Waite et al. 2015). Also, the behavioral economics literature on valuation can assist in fine-tuning the details of different conservation framing narratives and presentation techniques for conservation marketing (see review in Freeman III et al. 2014).

The forest policy papers contribute to the literature by clarifying the role of family forest owners in the Swedish Forestry Model and how they interact with the Swedish state's efforts to achieve the goals outlined in the Swedish policy document Sustainable Forests. The Paris Climate Agreement is possibly the most prominent example of using voluntary commitments to achieve collective goals and may signal the increasing importance of voluntary environmental efforts to meet global challenges. Some problems, however, may be more easily solved by volunteerism than others. Swedish forest policy's heavy reliance on volunteerism to reach some of its environmental goals offers an alternative model for how to incorporate landowners' voluntary efforts into state policy, but it also presents some challenges that are important for other countries to consider if they want to imitate Sweden's example.

Paper III is the first evaluation of how family forest owners engage with the overlapping voluntary alternatives to contribute voluntary set-asides toward the country's forest conservation goals. The assertion that 'freedom with responsibility' makes all family forest owners partners with the state in forest service provisioning is also novel. The results of Paper III raise the question of how to coordinate at a large scale the efforts of pro-environmental forest owners who are uninterested in private forest certification or state efforts at forest conservation environmental efforts. Questions of what coordination mechanisms for nature conservation are preferred by family forest owners who choose to stay out of forest certification is a possibility for future research.

There is also more research to be done concerning how environmental management measures are actually implemented in interactions between family forest owners and service providers in the forest industry. As far as the role of certification in securing environmental best practices on family owned forests, Villalobos et al. (2018) raises serious concerns about the central role of certification in Swedish forest policy. In short, not only do Villalobos et al (2018) find the majority of felled stands on family owned forests to not meet the minimum requirements for retention standards (64%), but they also find no significant positive effect of certification status on the amount of environmentally important areas preserved after felling, the number of trees and high stumps left after felling, or the share of set-aside land.

Since Villalobos et al. (2018) find no biodiversity benefits of forest certification on family owned forests, further qualitative research is needed on how measures such as set-asides, retention structures, and other environmental efforts are carried out on family owned forests in practice. For instance, what kinds of nature consideration alternatives are presented to family forest owners and by which service providers? Do forest management plans written for family forest owners by forestry professionals tend to include set-asides by default? To what extent are certification standards used for commercial operations on non-certified family forests as a matter of good practice by the forest industry? With almost all forest industries being certified in Sweden and a limited number of contractors to do the actual felling, it is important to establish exactly how forest management on non-certified family forest owned lands differs from that on certified lands in practice.[6]

Paper IV contributes to the forest policy literature by summarizing what kinds of family forest owner types should be more or less interested in what kinds of policy instruments by attempting to match different owner types with multiple policy instruments. The results cast some doubt on the intuitively appealing idea that family forest owners will be more effectively engaged in

[6] Masters of forestry students studying at SLU Umeå take note of a potential thesis topic!

environmental forest management by public policy instruments targeted to their ownership objectives. Instead, family forest owners as a group tend to either accept or reject any given forest policy instrument in the Swedish Forestry Model for nature considerations.

Future studies based on family forest owners and public policy instrument matching can take three possible forms. One option is to run an experiment explicitly and deliberately targeting different policy instrument types to different types of forest owners. All applications in the family forest owner literature, including my own, assume different types of forest policy instruments implicitly target different owner types. For example, forest owners interested in financial objectives are always assumed to be interested in economic instruments even though efforts have not been made to actually target economic owner types with economic instruments. A second alternative is to use more data about family forest owners to predict which public policy instruments appeal to which forest owners. The 'more data' alternative may be of interest to some researchers, but adding more variables will necessarily give less parsimonious insights for public policy recommendations. A third and final alternative is to investigate family forest ownership objectives as endogenous variables that change over the course of an owner's forest management history and are influenced by interactions with forest service providers and policy implementation efforts. The third alternative is inspired by life course history theory (e.g. Butler et al. 2017) and would require at least a two time period panel in which to begin exploring family forest owner attitudes, forest industry service providers, and public policy instruments as co-evolving relationships. From a public policy standpoint, a time series analysis could reveal medium or long-term effects of public policy and forest industry activities and show opportunities to nudge the evolution of family forest ownership objectives toward the public interest.

Acknowledgements

First of all I would like to thank the people at the Department of Forest Economics for their support during my PhD studies. I wish to specifically thank my primary supervisor, Camilla Widmark, for introducing me to Swedish forest policy, graciously allowing me to use the survey data she collected for two of my papers, and involving me in teaching the Swedish and International Forest Policy course. In the Forest Policy course I had the pleasure to learn about forestry practices in England, Germany, Austria, Slovakia, and Portugal along with Camilla and numerous Masters of Forestry students. My view of Swedish forestry issues would not be same without this wealth of international context and comparisons. I wish to thank Professors Danny Campbell and Nick Hanley for an invaluable study visit at the University of Stirling. I am extremely grateful for the practical competency in programming discrete choice models I gained from Danny during my time at Stirling. I also wish to thank Bengt Kriström, another of my supervisors, for his insightful comments and for facilitating numerous opportunities that have enhanced my academic skill set.

Others in the CERE network have lended their assistance in numerous ways. I wish to thank Erlend Dancke Sandorf for helping to facilitate my study visit in Stirling, assisting with my non-market valuation paper, and for his continued friendship. I want to specifically thank Francisco Aguilar for his help on my forest policy research, introducing me to a broader forest policy research network, and for his continued friendship. Others have read my research and offered helpful advice, including but not limited to Chandra Kiran Krishnamurthy, Amin Karimu, Tommy Lundgren, Lars Persson, and Jörgen Rudolphi.

I also want to extend tremendous thanks to my many peers. Foremost of all is my wife, Therese Danley. She assisted me in formulating, revising, and/or presenting results in three of my four papers as well as revising Sections 1-7 of this thesis. If I can write an introduction and background that Therese thinks is good, then I know I am on the right track. Also, Therese's help with data presentation has saved my papers and presentations from many embarrassingly bad tables and figures. I am grateful for the support of the other PhD students at the Department of Forest Economics: Sandra Schusser, Jinggang Guo, Mattia Luigi Ratti, Oben Bayrak, and Philip Kofi Adom, among others. Other peers from CERE include Alejandro Eguez, Golnaz Amjadi, Amiro Melkamu, and Victoria Ayubu. Some of you have gone through courses and exams with me, most of you have listened to my research and helped make it better, but all of you have made my experience as a PhD student better in some way. A special thanks goes to Sandra Schusser, Jinggang Guo, Alejandro Eguez, and Golnaz Amjadi as the fellow students who have had the most significant impact on my time as a PhD student.

I am grateful to members of the 2017, and 2018 SLU forest faculty PhD student council for their friendship and service attitude. The members of the council taught me that with enough action-oriented PhD students working together we could shape a better student experience at SLU, even in such a large bureaucracy. I also wish to thank those who were involved in establishing the joint research school Ecology and Society (ECOS), which started as a discussion between several fellow students after a PhD council meeting.

Last but not least I would like to thank the Masters of Forestry students who went on the international forest policy trips between 2014 and 2018. Witnessing these students learn about forestry outside their home country gave me a special insight on forestry in Sweden and some of its neighbors.

Brian Danley
December 13, 2018

References

Abdi, Hevré, and Lynne J Williams. 2010. “Principal Component Analysis.” *Wiley Interdisciplinary Review: Computational Statistics* 2 (4): 433–59. doi:10.1002/wics.101.

Arano, Kathryn G., and Ian A. Munn. 2006. “Evaluating Forest Management Intensity: A Comparison among Major Forest Landowner Types.” *Forest Policy and Economics* 9 (3): 237–48. doi:10.1016/j.forpol.2005.07.011.

Barua, Maan, Meredith Root-Bernstein, Richard J. Ladle, and Paul Jepson. 2011. “Defining Flagship Uses Is Critical for Flagship Selection: A Critique of the IUCN Climate Change Flagship Fleet.” *Ambio* 40 (4): 431–35. doi:10.1002/joc.4774.

Bateman, Ian J, Brett H Day, Stavros Georgiou, and Iain Lake. 2006. “The Aggregation of Environmental Benefit Values : Welfare Measures , Distance Decay and Total WTP ☆.” *Ecological Economics* 60: 450–60. doi:10.1016/j.ecolecon.2006.04.003.

Bemelmans-Videc, Marie-Louise, Ray C. Rist, and Evert Vedung. 2003. *Carrots, Sticks & Sermons: Policy Instruments and Their Evaluation*. New Brunswick, London: Transaction Publishers.

Ben-Akiva, Moshe, and Steven R. Lerman. 1985. *Discrete Choice Analysis: Theory and Application to Travel Demand*. Cambridge, MA: MIT Press.

Bernitz, Ulf. 2013. “Commercial Norms and Soft Law.” *Scandinavian Studies in Law* 58: 29–43.

Börger, Tobias, Nicola J Beaumont, Linwood Pendleton, Kevin J Boyle, Philip Cooper, Stephen Fletcher, Tim Haab, et al. 2014. “Incorporating Ecosystem Services in Marine Planning: The Role of Valuation.” *Marine Policy* 46: 161–70. doi:https://doi.org/10.1016/j.marpol.2014.01.019.

Borie, M, and Mike Hulme. 2015. “Framing Global Biodiversity: IPBES between Mother Earth and Ecosystem Services.” *Environmental Science & Policy* 54: 487–96. doi:10.1016/j.envsci.2015.05.009.

Boyd, James, and Spencer Banzhaf. 2007. “What Are Ecosystem Services? The Need for Standardized Environmental Accounting Units.” *Ecological Economics* 63: 616–26. doi:10.1016/j.ecolecon.2007.01.002.

Brouwer, Roy, Luke Brander, Onno Kuik, Elissaios Papyrakis, and Ian Bateman.

2013. “A Synthesis of Approaches to Assess and Value Ecosystem Services in the EU in the Context of TEEB.” *TEEB Follow-up Study for Europe*, no. May: 144.
Brukas, Vilis, and Ola Sallnäs. 2012. “Forest Management Plan as a Policy Instrument: Carrot, Stick or Sermon?” *Land Use Policy* 29 (3): 605–13. doi:10.1016/j.landusepol.2011.10.003.
Bush, Terrence. 2010. “Biodiversity and Sectoral Responsibility in the Development of Swedish Forestry Policy, 1988-1993.” *Scandinavian Journal of History* 35 (4): 471–98. doi:10.1080/03468755.2010.528249.
Butler, B. 2016. “Family Forest Ownerships of the United States, 2013: Findings from the U.S. Forest Service’s National Woodland Owner Survey (Draft)” 114.
Butler, Sarah M., Brett J. Butler, and Marla Markowski-Lindsay. 2017. “Family Forest Owner Characteristics Shaped by Life Cycle, Cohort, and Period Effects.” *Small-Scale Forestry* 16 (1): 1–18. doi:10.1007/s11842-016-9333-2.
Campbell, Danny, W George Hutchinson, and Riccardo Scarpa. 2009. “Using Choice Experiments to Explore the Spatial Distribution of Willingness to Pay for Rural Landscape Improvements.” *Environment and Planning A* 41: 97–112. doi:10.1068/a4038.
Campbell, Emer, and Bob Foy. 2008. “Executive Summary of Lough Melvin Catchment Management Plan.” Northern Regional Fisheries Board.
Carlén, O, L Mattsson, O Atlegrim, and K Sjöberg. 1999. “Cost Efficiency in Pursuing Environmental Objectives in Forestry.” *Journal of Environmental Management* 55 (2): 111–25. doi:http://dx.doi.org/10.1006/jema.1998.0247.
Carlsson, Julia. 2017. “Participatory Scenario Analysis in Forest Resource Management: Exploring Methods and Governance Challenges from a Rural Landscape Perspective.” Swedish University of Agricultural Sciences, Faculty of Forest Sciences.
CBD. 1992. “Convention on Biological Diversity.”
Concu, Giovanni B. 2007. “Are Non-Use Values Distance-Independent? Identifying the Market Area Using a Choice Modelling Experiment.” M05_6. Brisbane, Australia.
Costanza, Robert, Ralph Arge, Rudolf De Groot, Stephen Farberk, Monica Grasso, Bruce Hannon, Karin Limburg, et al. 1997. “The Value of the World ’ s Ecosystem Services and Natural Capital.” *Nature* 387 (May): 253–60. doi:10.1038/387253a0.
Cubbage, Frederick, Patrice Harou, and Erin Sills. 2007. “Policy Instruments to Enhance Multi-Functional Forest Management.” *Forest Policy and Economics* 9 (7): 833–51. doi:10.1016/j.forpol.2006.03.010.
Cvitanic, Jakša, and Jianfeng Zhang. 2013. *Contract Theory in Continuous-Time Models*. Berlin, Heidelberg: Springer-Verlag. doi:https://doi.org/10.1007/978-3-642-14200-0.
Danley, Brian. 2018. “Skepticism of State Action in Forest Certification and Voluntary Set-Asides: A Swedish Example with Two Environmental

Offsetting Options." *Scandinavian Journal of Forest Research* 7581: 1–13. doi:10.1080/02827581.2018.1479442.
Danley, Brian, and Camilla Widmark. 2016. "Evaluating Conceptual de Fi Nitions of Ecosystem Services and Their Implications." *Ecological Economics* 126: 132–38. doi:10.1016/j.ecolecon.2016.04.003.
Dow, William H, and Edward C Norton. 2003. "Choosing Between and Interpreting the Heckit and Two-Part Models for Corner Solutions." *Health Services & Outcomes Research Methodology* 4: 5–18.
Dunteman, George H. 1989. *Principal Components Analysis*. Newbury, CA: Sage Publications.
European Commission. 2013. "A New EU Forest Strategy: For Forests and the Forest-Based Sector." Brussels. https://eur-lex.europa.eu/resource.html?uri=cellar:21b27c38-21fb-11e3-8d1c-01aa75ed71a1.0022.01/DOC_1&format=PDF.
European Commission. 2015. "Natura 2000 and Forests. Part I - II. Technical Report 2015 - 088." doi:10.2779/699873.
European Commission. 2016. "Introduction to the New EU Water Framework Directive." Accessed: Dec 16, 2018. http://ec.europa.eu/environment/water/water-framework/info/intro_en.htm.
Everitt, Brian S., Sabine Landau, Morven Leese, and Daniel Stahl. 2011. *Cluster Analysis*. 5th ed. Kings College London: Wiley.
Favreau, Jorie M., C. Ashton Drew, George R. Hess, Matthew J. Rubino, Frank H. Koch, and Katherine A. Eschelbach. 2006. "Recommendations for Assessing the Effectiveness of Surrogate Species Approaches." *Biodiversity and Conservation* 15 (12): 3949–69. doi:10.1007/s10531-005-2631-1.
Ficko, Andrej, Gun Lidestav, Aine NıDhubháin, Heimo Karppinen, Ivana Zivojinovic, and Kerstin Westin. 2017. "European Private Forest Owner Typologies : A Review of Methods and Use." *Forest Policy and Economics*, no. September. doi:10.1016/j.forpol.2017.09.010.
Frayer, W.E., and George M. Furnival. 1999. "Forest Survey Sampling Designs: A History." *Journal of Forestry* 97 (12): 4–10.
Freeman III, A. Myrick, Joseph A. Herriges, and Catherine L. Kling. 2014. *The Measurement of Environmental and Resource Values. Theory and Methods*. 3rd Editio. New York: Resources for the Future Press.
Geijzendorffer, Ilse R, Astrid J A Van Teeffelen, Hilary Allison, Daniela Braun, Katherine Horgan, Maitane Iturrate-garcia, Maria Joa, Shoko Sakai, and Debra Zuppinger-dingley. 2017. "How Can Global Conventions for Biodiversity and Ecosystem Services Guide Local Conservation Actions ?" *Current Opinion in Environmental Sustainability* 29: 145–50. doi:https://doi.org/10.1016/j.cosust.2017.12.011.
Giraud, Kelly, Branka Valcic, Geographic Embedded, Samples Case, Study Of, Alaskan Steller, S E A Lion, and C Taylor. 2010. "Willingness-to-Pay Estimates and Geographic Embedded Samples: Case Study of Alaskan Steller Sea Lion." *Journal of International Wildlife Law and Policy* 7 (1–2): 57–72. doi:10.1080/13880290490480167.

Gómez-Baggethun, Erik, Rudolf de Groot, Pedro L. Lomas, and Carlos Montes. 2010. "The History of Ecosystem Services in Economic Theory and Practice: From Early Notions to Markets and Payment Schemes." *Ecological Economics* 69: 1209–18. doi:10.1016/j.ecolecon.2009.11.007.

Greene, William H., and David A. Hensher. 2013. "Revealing Additional Dimensions of Preference Heterogeneity in a Latent Class Mixed Multinomial Logit Model." *Applied Economics* 45 (14): 1897–1902. doi:10.1080/00036846.2011.650325.

Gregersen, Hans, and Arnoldo Contreras. 2010. "Rethinking Forest Regulations: From Simple Rules to Systems to Promote Best Practices and Compliance." Washington, DC. http://www.rightsandresources.org/wp-content/exported-pdf/forestpoliciesfullreport23june2011finalnewpartners.pdf.

Grossman, Stanford J., and Oliver D. Hart. 1983. "An Analysis of the Principal-Agent Problem." *Econometrica* 51 (1): 7–45. doi:10.2307/1912246.

Gunningham, Neil, Peter Grabosky, and Darren Sinclair. 1998. *Smart Regulation: Designing Environmental Policy*. Oxford: Clarendon press.

Gustafsson, Lena, Jari Kouki, and Anne Sverdrup-Thygeson. 2010. "Tree Retention as a Conservation Measure in Clear-Cut Forests of Northern Europe: A Review of Ecological Consequences." *Scandinavian Journal of Forest Research* 25 (4): 295–308. doi:10.1080/02827581.2010.497495.

Gut, Allan. 2009. *An Intermediate Course in Probability*. Second Edi. Dordrecht Heidelberg, London, New York: Springer.

Haines-Young, R, and Marion Potschin. 2011. "Common International Classification of Ecosystem Services (CICES): 2011 Update." *Expert Meeting on Ecosystem Accounts …*, 1–17. http://www.cices.eu/wp-content/uploads/2009/11/CICES_Update_Nov2011.pdf.

Hanley, Nick, Felix Schla, and James Spurgeon. 2003. "Aggregating the Benefits of Environmental Improvements : Distance-Decay Functions for Use and Non-Use Values." *Journal of Environmental Management* 68: 297–304. doi:10.1016/S0301-4797(03)00084-7.

Haugen, Katarina, Svante Karlsson, and Kerstin Westin. 2016. "New Forest Owners: Change and Continuity in the Characteristics of Swedish Non-Industrial Private Forest Owners (NIPF Owners) 1990–2010." *Small-Scale Forestry*. doi:10.1007/s11842-016-9338-x.

Häyrinen, Liina, Osmo Mattila, Sami Berghäll, and Anne Toppinen. 2014. "Changing Objectives of Non-Industrial Private Forest Ownership: A Confirmatory Approach to Measurement Model Testing." *Canadian Journal of Forest Research* 44 (4): 290–300. doi:10.1139/cjfr-2013-0211.

Heckman, James J. 1979. "Sample Selection Bias as a Specification Error." *Econometrica*, no. 47: 153–62.

Hedeklint, Karin, and Olle Höjer. 2017. "Protected Nature 2017 (Skyddad Natur 2017-12-31)." https://www.scb.se/contentassets/e419dfae78ef4162a2b4b2c4bd4ef4e4/mi0603_2017a01_sm_mi41sm1801.pdf.

Hogl, Karl, Michael Pregernig, and Gerhard Weiss. 2005. "What Is New about

New Forest Owners ? A Typology of Private Forest Ownership in Austria." *Small-Scale Forest Economics, Management and Policy* 4 (3): 325–42. doi:10.1007/s11842-005-0020-y.
Hossain, Belayet, and Laura Lamb. 2012. "The Dynamics of Environmental Giving in Canada: Evidence of Rising Demand for Environmental Quality ?" *Economic Papers: The Economic Society of Australia* 31 (2): 265–73. doi:10.1111/j.1759-3441.2012.00177.x.
Hysing, Erik. 2009. "From Government to Governance? A Comparison of Environmental Governing in Swedish Forestry and Transport." *Governance* 22 (4): 647–72. doi:10.1111/j.1468-0491.2009.01457.x.
Jain, Anil K. 2010. "Data Clustering: 50 Years beyond K-Means." *Pattern Recognition Letters* 31 (8). Elsevier B.V.: 651–66. doi:10.1016/j.patrec.2009.09.011.
Johansson, Johanna. 2013. *Constructing and Contesting the Legitimacy of Private Forest Governance*. Department of Political Science, Umeå University.
Johansson, Johanna, and Gun Lidestav. 2011. "Can Voluntary Standards Regulate Forestry? - Assessing the Environmental Impacts of Forest Certification in Sweden." *Forest Policy and Economics* 13 (3): 191–98. doi:10.1016/j.forpol.2010.11.004.
Johansson, Per-Olov, and Bengt Kriström. 2015. *Cost-Benefit Analysis for Project Appraisal*. Cambridge, UK: Cambridge University Press.
Johnston, Robert J., and Mahesh Ramachandran. 2014. "Modeling Spatial Patchiness and Hot Spots in Stated Preference Willingness to Pay." *Environmental and Resource Economics* 59 (3): 363–87. doi:10.1007/s10640-013-9731-2.
Jolliffe, Ian. 2002. *Principle Component Analysis, Second Edition*. Second. New York: Springer.
Joshi, Sudiksha, and Kathryn G. Arano. 2009. "Determinants of Private Forest Management Decisions: A Study on West Virginia NIPF Landowners." *Forest Policy and Economics* 11 (2): 132–39. doi:10.1016/j.forpol.2008.10.005.
Karppinen, Heimo. 2000. "Forest Values and the Objectives of Forest Ownership." University of Helsinki.
Kelly, Fiona L, Lynda Connor, Emma Morrissey, Ciara Wogerbauer, Ronan Matson, and Rory Feeney. 2012. "Water Framework Directive Fish Stock Survey of Lough Melvin, July 2011." Dublin, Ireland.
Kruys, Nic, Jonas Fridman, Frank G??tmark, Per Simonsson, and Lena Gustafsson. 2013. "Retaining Trees for Conservation at Clearcutting Has Increased Structural Diversity in Young Swedish Production Forests." *Forest Ecology and Management* 304: 312–21. doi:10.1016/j.foreco.2013.05.018.
Kumer, Peter, and Erik Štrumbelj. 2017. "Clustering-Based Typology and Analysis of Private Small-Scale Forest Owners in Slovenia." *Forest Policy and Economics* 80: 116–24. doi:10.1016/j.forpol.2017.03.014.
Lancaster, Kelvin. 1966. "A New Approach to Consumer Theory." *Journal of Political Economy.* 74 (2): 132–57. doi:10.1086/226550.

Lele, S, O Springate-Baginski, R Lakerveld, D Deb, and P Dash. 2013. “Ecosystem Services: Origins, Contributions, Pitfalls, and Alternatives.” *Conservation & Society* 11 (4): 343–58. doi:10.4103/0972-4923.125752.

Leopold, Aldo. 1949. *A Sand Country Almanac, and Sketches Here and There*. New York: Oxford University Press.

Lidestav, Gun. 2010. “In Competition with a Brother: Women’s Inheritance Positions in Contemporary Swedish Family Forestry.” *Scandinavian Journal of Forest Research* 25 (sup9): 14–24. doi:10.1080/02827581.2010.506781.

Lindahl, Karin Beland, Anna Stèns, Camilla Sandström, Johanna Johansson, Rolf Lidskog, Thomas Ranius, and Jean Michel Roberge. 2015. “The Swedish Forestry Model: More of Everything?” *Forest Policy and Economics* 77: 186–99. doi:10.1016/j.forpol.2015.10.012.

Lindroos, Ola, Gun Lidestav, and Tomas Nordfjell. 2005. “Swedish Non-Industrial Private Forest Owners – a Survey of Self-Employment and Equipment Investments.” *Small-Scale Forest Economics, Management and Policy* 4 (4): 409–26.

Lister, Jane. 2011. *Corporate Social Responsibility and the State: International Approaches to Forest Co-Regulation*. Vancouver, Toronto. UBC Press.

Löfmarck, Erik, Ylva Uggla, and Rolf Lidskog. 2017. “Forest Policy and Economics Freedom with What? Interpretations of "responsibility” in Swedish Forestry Practice.” *Forest Policy and Economics* 75: 34–40. doi:10.1016/j.forpol.2016.12.004.

Loomis, John B. 2000. “Vertically Summing Public Good Demand Curves : An Empirical Comparison of Economic versus Political Jurisdictions.” *Land Economics* 76 (2): 312–21.

Madden, David. 2008. “Sample Selection versus Two-Part Models Revisited: The Case of Female Smoking and Drinking.” *Journal of Health Economics* 27: 300–307. doi:10.1016/j.jhealeco.2007.07.001.

Mårald, Erland, Camilla Sandström, Annika Nordin, and contributing authors. 2017. *Forest Governance and Management across Time: Developing a New Forest Social Contract*. 1st ed. New York: Routledge.

Marina, Meilă. 2006. “The Uniqueness of a Good Optimum for K-Means.” In *ICML 2006 Proceedings of the 23rd International Conference on Machine Learning*, 625–32. Pittsburgh.

Matthies, Brent D, Dalia D Amato, Sami Bergh, Tommi Ekholm, Hans Fredrik Hoen, Jani Holopainen, Jaana E. Korhonen, et al. 2016. “An Ecosystem Service-Dominant Logic? Integrating the Ecosystem Service Approach and the Service-Dominant Logic.” *Journal of Cleaner Production* 124: 51–64. doi:10.1016/j.jclepro.2016.02.109.

McFadden, Daniel. 1973. *Conditional Logit Analysis of Qualitative Choice Behavior*. Edited by P. Zarembka. New York: Academic Press.

McKeown, N J, R A Hynes, R A Duguid, A Ferguson, and P.A. Prodöhl. 2010. “Phylogeographic Structure of Brown Trout Salmo Trutta in Britain and Ireland : Glacial Refugia , Postglacial Colonization and Origins of Sympatric Populations.” *Journal of Fish Biology* 76: 319–47. doi:10.1111/j.1095-

8649.2009.02490.x.
Mckinley, Duncan C, Abe J Miller-rushing, Heidi L Ballard, Rick Bonney, Hutch Brown, Susan C Cook-patton, Daniel M Evans, et al. 2017. “Citizen Science Can Improve Conservation Science , Natural Resource Management , and Environmental Protection.” *Biological Conservation* 208. Elsevier Ltd: 15–28. doi:10.1016/j.biocon.2016.05.015.
MESAB. 2005. “Living Beyond Our Means: Natural Assets and Human Well-Being". Millenium Ecosystem Assesement Board. www.millenniumassessment.org/.
Nilsson. 2015. “*Skogsdata* Forest Data 2015,” 152. http://pub.epsilon.slu.se/12626/17/nilsson_p_cory_n_151007.pdf.
Novacek, M. J. 2008. “Engaging the Public in Biodiversity Issues.” *Proceedings of the National Academy of Sciences* 105 (Supplement 1): 11571–78. doi:10.1073/pnas.0802599105.
Nursey-Bray, Melissa J, Joanna Vince, Michael Scott, Marcus Haward, Kevin O Toole, Tim Smith, Nick Harvey, and Beverley Clarke. 2014. “ScienceDirect Science into Policy? Discourse, Coastal Management and Knowledge.” *Environmental Science & Policy* 38: 107–19. doi:https://doi.org/10.1016/j.envsci.2013.10.010.
Pate, Jennifer, and John Loomis. 1997. “The Effect of Distance on Willingness to Pay Values : A Case Study of Wetlands and Salmon in California.” *Ecological Economics* 20: 199–207.
Patton, Michael Quinn. 2002. *Qualitative Research & Evaluation Methods*. 3rd editio. London: Sage Publications.
Potschin-Young, Marion, Balint Czúcz, Camino Liquete, Joachim Maes, Graciela M Rusch, and Roy Haines-Young. 2017. “Intermediate Ecosystem Services: An Empty Concept?” *Ecosystem Services* 27: 124–26. doi:https://doi.org/10.1016/j.ecoser.2017.09.001.
Potschin, Marion, and Roy Haines-Young. 2016. “Defining and Measuring Ecosystem Services.” In *Routledge Handbook of Ecosystem Services*, edited by Marion Potschin, Roy Haines-Young, Robert Fish, and R. Kerry Turner, 25–41. New York: Taylor & Francis.
Puhani, Patrick A. 2000. “THE HECKMAN CORRECTION FOR SAMPLE SELECTION AND ITS CRITIQUE.” *Journal of Economic Surveys* 14 (1): 53–68.
R Core Team. 2018. “R: A Language and Environment for Statistical Computing. R Core Team.” Vienna, Austria: R Foundation for Statistical Computing.
Rametsteiner, Ewald, and Markku Simula. 2003. “Forest Certification — an Instrument to Promote Sustainable Forest Management?” *Journal of Environmental Management* 67: 87–98. doi:10.1016/S0301-4797(02)00191-3.
Reyers, Belinda, Stephen Polasky, Heather Tallis, Harold A Mooney, and Anne Larigauderie. 2012. “Finding Common Ground for Biodiversity and Ecosystem Services.” *BioScience* 62 (5): 503–7. doi:10.1525/bio.2012.62.5.12.

Riksrevisionen (Swedish National Audit Office). 2018. “The Protection of Valuable Forest (Skyddet Av Värdefull Skog).” Stockholm. https://www.riksrevisionen.se/rapporter/granskningsrapporter/2018/skyddet-av-vardefull-skog.html.

Robins, L, T P Burt, L J Bracken, J Boardman, and D B A Thompson. 2017. “Environmental Science & Policy Making Water Policy Work in the United Kingdom : A Case Study of Practical Approaches to Strengthening Complex, Multi-Tiered Systems of Water Governance.” *Environmental Science & Policy* 71. Elsevier Ltd: 41–55. doi:http://dx.doi.org/10.1016/j.envsci.2017.01.008.

Rolfe, John, and Jill Windle. 2012. “Distance Decay Functions for Iconic Assets : Assessing National Values to Protect the Health of the Great Barrier Reef in Australia.” *Environmental and Resource Economics* 53: 347–65. doi:10.1007/s10640-012-9565-3.

Romero, Claudia, Francis E. Putz, Manuel R. Guariguata, Erin O. Sills, Paolo O. Cerutti, and Guillaume Lescuyer. 2013. “An Overview of Current Knowledge about the Impacts of Forest Management Certification: A Proposed Framework for Its Evaluation.”

Royal Swedish Academy of Agriculture and Forestry (KSLA). 2009. “The Swedish Forestry Model.” Stockholm.

Sagoff, Mark. 1988. “Some Problems With Environmental Economics.” *Environmental Ethics* 10 (1): 55–74.

Scarpa, Riccardo, and John M Rose. 2008. “Design Efficiency for Non-Market Valuation with Choice Modelling : How to Measure It , What to Report and Why.” *The Australian Journal of Agricultural and Resource Economics* 52: 253–82. doi:10.1111/j.1467-8489.2007.00436.x.

Schaafsma, M., R. Brouwer, A. Gilbert, J. van den Bergh, and A. Wagtendonk. 2013. “Estimation of Distance-Decay Functions to Account for Substitution and Spatial Heterogeneity in Stated Preference Research.” *Land Economics* 89 (3): 514–37. doi:10.3368/le.89.3.514.

Schröter, Matthias, Emma H van der Zanden, Alexander P E van Oudenhoven, Roy P Remme, Hector M Serna-Chavez, Rudolf S de Groot, and Paul Opdam. 2014. “Ecosystem Services as a Contested Concept: A Synthesis of Critique and Counter-Arguments.” *Conservation Letters* 00 (0): n/a-n/a. doi:10.1111/conl.12091.

SFA, Swedish Forest Agency. 2012. “Forestry’s Voluntary Set-Asides / Skogsbrukets Frivilliga Avästtningar.”

SFA, Swedish Forest Agency. 2014. “Swedish Statistical Yearbook of Forestry 2014.” Jönköping.

SFA, Swedish Forest Agency. 2017. “Area of Forest Voluntary Set-Asides Mapped / Arealen Frivilliga Avsättningar Av Skog Kartlagd,” March 30. https://via.tt.se/pressmeddelande/arealen-frivilliga-avsattningar-av-skog-kartlagd?publisherId=415163&releaseId=1259452.

Swedish FSC standard for forest certification including SLIMF indicators. 2010, Uppsala. https://se.fsc.org/preview.fsc-forest-management-standard-for-

sweden.a-772.pdf
Swedish PEFC forest standard / Svensk PEFC Skogsstandard. 2012. Stockholm. http://pefc.se/wp-content/uploads/2010/11/n-pefc%20swe%20002%20-%20svensk%20pefc%20skogsstandard%20120801.pdf
Silvertown, Jonathan. 2015. "Have Ecosystem Services Been Oversold?" *Trends in AEcology and Evolution* 30 (11): 641–48. doi:10.1016/j.tree.2015.08.007.
Simonsson, Per, Lars Östlund, and Lena Gustafsson. 2016. "Conservation Values of Certified-Driven Voluntary Forest Set-Asides." *Forest Ecology and Management* 375: 249–58. doi:10.1016/j.foreco.2016.05.039.
Skogs Industrierna. 2018. "Map with Voluntary Set-aside Forest." https://www.forestindustries.se/forest-industry/sustainable-development/voluntary-set-aside-forests-in-sweden/map/
Statistics Sweden. 2017. "Protected Nature in Sweden 2017." *Protected Nature in Sweden*. https://www.scb.se/en/finding-statistics/statistics-by-subject-area/environment/land-use/protected-nature/pong/statistical-news/protected-nature-2017/.
Suri, Harsh. 2011. "Purposeful Sampling in Qualitative Research Synthesis." *Qualitative Research Journal* 11 (2): 63–75.
Swedish Environmental Protection Agency. 2017a. "Environmental Goals: Yearly Follow-up of Sweden's National Environmental Goals 2017. (Miljömålen: Årlig Uppföljning Av Sveriges Nationella Miljömål 2017)." https://www.miljomal.se/Global/24_las_mer/rapporter/malansvariga_myndigheter/2017/au2017.pdf.
Swedish Environmental Protection Agency. 2017b. "Nature Conservation Agreements / Naturvårdsavtal." 2017.
Swedish Environmental Protection Agency. 2018a. "Specifications for Sustainable Forests." http://www.swedishepa.se/Environmental-objectives-and-cooperation/Swedens-environmental-objectives/The-national-environmental-objectives/Sustainable-Forests/Specifications-for-Sustainable-Forests/.
Swedish Environmental Protection Agency.2018b. "Valuable State Owned Forests (Skyddsvärda Statliga Skogar)." https://www.naturvardsverket.se/Stod-i-miljoarbetet/Vagledningar/Skyddad-natur/Skyddsvarda-statliga-skogar/.
Swedish Forest Agency. 2016. "Core Issues for Good Environmental Considerations (Målbilder För God Miljöhänsyn)." Jönköping. https://www.skogsstyrelsen.se/malbilder.
Swedish Forest Agency. 2017. "Freedom with Responsibility (Frihet under Ansvar)." Accessed: November 17, 2018. https://www.skogsstyrelsen.se/aga-skog/du-och-din-skog/frihet-under-ansvar/.
"Swedish FSC Standard for Forest Certification Including SLIMF Indicators." 2010. Uppsala. https://se.fsc.org/preview.fsc-forest-management-standard-for-sweden.a-772.pdf.
"Swedish PEFC Forest Standard / Svensk PEFC Skogsstandard." 2012. Stockholm. http://pefc.se/wp-content/uploads/2010/11/n-pefc swe 002 - svensk pefc skogsstandard 120801.pdf.
Thaler, Richard H., and Cass R. Sunstein. 2008. *Nudge: Improving Decisions about*

Health, Wealth, and Happiness. New Haven & London: Yale University Press.
Toman, Michael. 1998. "Why Not to Calculate the Value of the World's Ecosystem Services and Natural Capital." *Ecological Economics* 25: 57–60.
Train, Kenneth. 2009. *Discrete Choice Methods with Simulation*. New York: Cambridge University Press.
Van De Ven, Wynand, and Bernard Van Praag. 1981. "The Demand for Deductibles in Private Health Insurance: A Probit Model with Sample Selection." *Journal of Econometrics*, no. 17: 229–52.
Van Hecken, Gert, Vijay Kolinjivadi, Catherine Windey, Pamela McElwee, Elizabeth Shapiro-Garza, Frédéric Huybrechs, and Johan Bastiaensen. 2018. "Silencing Agency in Payments for Ecosystem Services (PES) by Essentializing a Neoliberal 'Monster' Into Being: A Response to Fletcher & Büscher's 'PES Conceit.'" *Ecological Economics* 144 (October 2017). Elsevier: 314–18. doi:10.1016/j.ecolecon.2017.10.023.
Veríssimo, Diogo, Iain Fraser, Weber Girão, Alberto A Campos, Robert J Smith, and Douglas C Macmillan. 2013. "Evaluating Conservation Flagships and Flagship Fleets." *Conservation Letters* 7 (3): 263–70. doi:10.1111/conl.12070.
Veríssimo, Diogo, Tatiana Pongiluppi, Maria Cintia M Santos, Pedro F Develey, Iain Fraser, Robert J Smith, and Douglas C Macmilan. 2013. "Using a Systematic Approach to Select Flagship Species for Bird Conservation." *Conservation Biology* 28 (1): 269–77. doi:10.1111/cobi.12142.
Vermeulen, Bart, Peter Goos, Riccardo Scarpa, and Martina Vandebroek. 2011. "Bayesian Conjoint Choice Designs for Measuring Willingness to Pay." *Environmental and Resource Economics* 48: 129–49. doi:10.1007/s10640-010-9401-6.
Villalobos, Laura, Jessica Coria, and Anna Nordén. 2018. "Has Forest Certification Reduced Forest Degradation in Sweden ?" *Land Economics* 93 (3): 390–112.
Waite, Richard, Benjamin Kushner, Megan Jungwiwattanaporn, Erin Gray, and Lauretta Burke. 2015. "Use of Coastal Economic Valuation in Decision Making in the Caribbean: Enabling Conditions and Lessons Learned." *Ecosystem Services* 11: 45–55. doi:https://doi.org/10.1016/j.ecoser.2014.07.010.
Wallin, Ida. 2017. "Forest Management and Governance in Sweden." Swedish University of Agricultural Sciences.
Wallmo, Kristy, and Daniel K Lew. 2016. "A Comparison of Regional and National Values for Recovering Threatened and Endangered Marine Species in the United States." *Journal of Environmental Management* 179: 38–46.
Wickham, Hadley. 2016. *Ggplot2: Elegant Graphics for Data Analysis*. Springer-Verlag New York.
Widman, U. 2015. "Shared Responsibility for Forest Protection?" *Forest Policy and Economics* 50: 220–27. doi:10.1016/j.forpol.2014.10.003.
Widman, Ulrica. 2016. "Protecting Forests through Partnerships." Umeå University.

Wright, Andrew J, Diogo Veríssimo, Kathleen Pilfold, E C M Parsons, Kimberly Ventre, Jenny Cousins, Rebecca Jefferson, Heather Koldewey, Fiona Llewellyn, and Emma Mckinley. 2015. "Ocean & Coastal Management Competitive Outreach in the 21st Century : Why We Need Conservation Marketing." *Ocean & Coastal Management* 115: 41–48.

Young, Juliette C, Kerry A Waylen, Simo Sarkki, Steve Albon, Ian Bainbridge, Estelle Balian, James Davidson, et al. 2014. "Improving the Science-Policy Dialogue to Meet the Challenges of Biodiversity Conservation : Having Conversations Rather than Talking at One-Another." *Biodiversity Conservation* 23: 387–404. doi:10.1007/s10531-013-0607-0.