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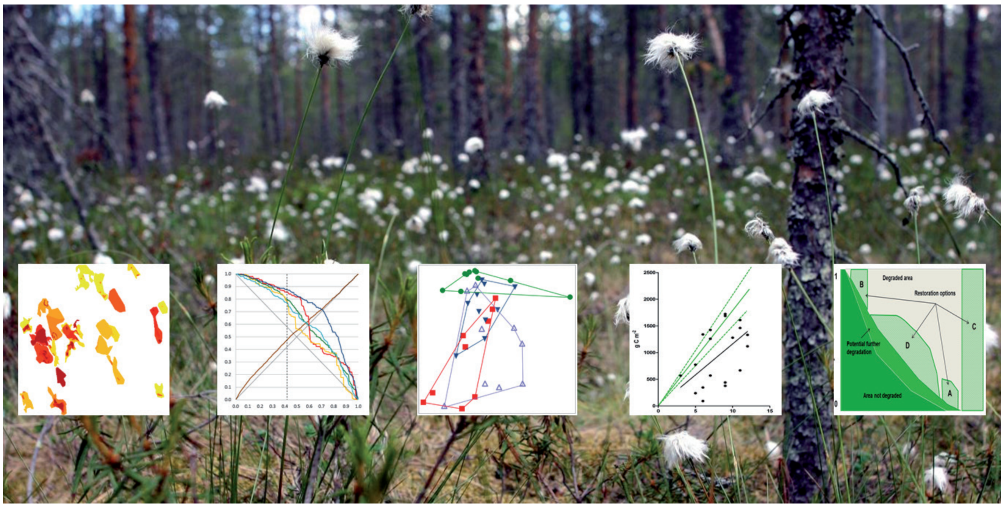
296

Santtu Kareksela

Ecosystem Rescue

When Protection is not Enough



JYVÄSKYLÄN YLIOPISTO

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UNIVERSITY OF JYVÄSKYLÄ

JYVÄSKYLÄ 2015

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Conserve the best, restore the rest.
-Professor Hans Joosten

For Carita, Elli and Unto

ABSTRACT

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Yhteenveto: Ekosysteemien turvaaminen – kun suojelualueet eivät enää riitä
Diss.

Identifying the most profitable trade-offs in resource allocation plays an essential role in modern conservation biology in a finite world. Questions like what, where and how much are put into context in the framework of systematic conservation planning, with the cost-effective allocation of actions like protection or ecological restoration. The question of how much is enough is, however, still a challenge in decision-making in conservation, including both ecological and socio-ethical uncertainty. In this thesis I studied how and to what extent modern methods in conservation planning and the restoration of ecosystems can help us in alleviating the apparent challenges of resource allocation. The studied approaches include ecological loss avoidance in land-use and ecological restoration of degraded ecosystems' structure and function. I also studied the implementation of both spatial conservation planning and allocation of resources in large scale restoration planning. My results show that the loss of biodiversity can be effectively reduced by emphasising ecological loss avoidance when planning general land-use. However, the magnitude of the effects is dependent on the success of the implementation of the prioritization analysis. I discus the results of the present prioritization analysis with the planning authorities who commissioned the analysis, and present observations that can be used to close the gap between high end prioritization analyses and their implementation. I also found that some functions (and services) of an ecosystem can be enhanced already in a short time period post restoration. Furthermore, I found flexibility in the structure-function relationship of the ecosystem as the peat growth function of the studied peatland ecosystems was restored despite seemingly slower recovery of the vegetation. Finally, my results on the allocation of restoration efforts demonstrate an urgent need for the incorporation of the degree of the change in the ecosystem to setting the target for restoration, in order to complement the area based goals. I provide an operational model and a platform for the implementation of a new rationale for the allocation of restoration resources.

Keywords: Cost-efficiency; ecological restoration; ecosystem structure and function; science-implementation gap; spatial conservation prioritization.

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LIST OF ORIGINAL PUBLICATIONS

The thesis is based on the following original papers, which will be referred to in the text by their Roman numerals I-IV.

I Kareksela S., Moilanen A., Tuominen S. & Kotiaho J.S. 2013. Use of inverse spatial conservation prioritization to avoid biological diversity loss outside protected areas. *Conservation Biology* 27: 1294-1303.

II Kareksela S., Moilanen A., Ristaniemi O., Välivaara R. & Kotiaho J.S. 2014. Exposing ecological and economic costs of research-implementation gap and compromises in decision-making. Manuscript.

III Kareksela, S.*, Haapalehto, T.O.*, Juutinen, R., Matilainen, R., Tahvanainen, T., Kotiaho, J.S. 2014. Fighting carbon loss of degraded peatlands by jump-starting ecosystem functioning with ecological restoration. Submitted manuscript.
*Shared first authorship. This manuscript has also been included as a chapter in Tuomas Haapalehto's thesis.

IV Kotiaho J.S., Haapalehto T., Halme P., Kareksela S., Oldén A., Päivinen J. and Moilanen A. 2014. Convention on Biological Diversity and (not) restoring the world. Manuscript.

The table shows the contributions to the original papers by the authors. Smaller contributions are stated in the acknowledgements of the papers.

	I	II	III	IV
Original idea	SK, AM, JSK	SK, AM, JSK	SK, TT, TOH, JSK	JSK, AM, PH, TOH, SK, JP
Data	SK, ST	SK, OR, RV	SK, RJ, TOH, RM	NA
Analyses	SK, AM	SK, AM, JSK, OR, RV	SK, TOH, JSK, TT	JSK, AM, PH, TOH, SK, JP, AO
Writing	SK, AM, JSK	SK, AM, JSK, OR, RV	SK, TOH, JSK, TT, RJ, RM	JSK, AM, PH, TOH, SK, JP, AO

SK = Santtu Kareksela, JSK = Janne S. Kotiaho, AM = Atte Moilanen, ST = Seppo Tuominen, OR = Olli Ristaniemi, RV = Reima Välivaara, TOH = Tuomas Haapalehto, TT = Teemu Tahvanainen, RM = Rose Matilainen, RJ = Riikka Juutinen, PH = Panu Halme, AO = Anna Oldén, JP = Jussi Päivinen

1 INTRODUCTION

1.1 What on earth is eating the Earth?

Many letters on conservation science start with a variation of a statement, which lists ecosystem destruction, habitat fragmentation, climate change, and invasive species as the greatest threats to biodiversity, the function of ecosystems, and ultimately human wellbeing. Indeed, the negative effects of anthropogenic activities are reflected at some level by most ecosystems, with variable outcomes from the large scale destruction of ecosystems (Millennium Ecosystem Assessment 2005) to the homogenization of otherwise seemingly natural communities (Clavel *et al.* 2011). One distinctively different threat, which is not stated as frequently, is the one presented by, for example, Gilbert (2011): the human mind. Even when our knowledge of the global anthropogenic threats is acknowledged we are still hindered by our brain that often seems to be optimized for solving problems quite different from those we face today (Gilbert 2011). It seems that we are not so good at grasping the big picture, in general. This is of course quite understandable: degradation at the level of ecosystems or landscapes can be deviously gradual (Suding and Hobbs 2009), it can take time to realize extinctions (Tilman *et al.* 1994, Kuussaari *et al.* 2009) and sufficient habitat connectivity can be difficult to observe, measure, and interpret (Arponen *et al.* 2011). For example, halting the ongoing loss of biodiversity is frequently considered as a key priority in conservation (CBD 2010). The problem and its possible solution have multiple dimensions. Protecting enough is already too late as we have created an extinction debt (Tilman *et al.* 1994, Hanski 2000), in other words, species will continue to become extinct even in a utopian scenario, where no further habitats are destroyed (Vellend 2006). This extinction debt can only be reversed or settled with the ecological restoration of degraded habitats, thus increasing the area of available habitat for species near to their extinction threshold (Hanski 2000, Hanski and Ovaskainen 2002, Kuussaari *et al.* 2009). However, ecological restoration does not directly prevent degradation elsewhere, leading to complicated resource allocation prioritization between conservation and ecosystem restoration and management actions: Furthermore, with finite resources it seems inevitable that bringing anthropogenic biodiversity loss to a total halt is an insurmountable task. If everything cannot be saved we face a complicated triage: the actions aiming at the salvation of the next most threatened species may not be the actions that lead to the highest biodiversity in a one hundred year time scale (Bottrill *et al.* 2008).

Our inabilities to see the nature of the problem, or more importantly the true nature of the required solution, can make a big difference in how cost-effectively we are able to react to the more direct problems of the modern world. The increasing amount of detailed conservation actions and intervention methods further complicates the decision-making process. Consequently, many approaches and applications in conservation science aim to increase our ability to solve large scale or otherwise complicated problems in decision-making (Margules and pressey 2000, Hobbs *et al.* 2006, Puozols and Moilanen 2013, Dicks et al. 2014). A strong emphasis is now also on our abilities to on the one hand understand and on the other hand to properly deliver the often complicated solutions provided by high-end analyses in conservation planning, in order to prevent possible gaps between research and implementation (Knight *et al.* 2006). Despite the seemingly increased focus of conservation science on spatial conservation planning, also filling the gaps in the knowledge about the specific ecology of the major threats is of major importance. In conservation science these perhaps relatively small details compile into the complicated ecological models found in many decision support tools (Ferrier and Wintle 2009) or provide us with much needed novel perspectives and methods in smaller scale ecological interventions (Lindenmayer *et al.* 2008).

1.2 How much is enough?

As a society we set high demands on land-use that constantly increase with an increasing population (e.g. Foley *et al.* 2005, Millennium Ecosystem Assessment 2005, Polasky *et al.* 2008). The inevitable result is that the area remaining for fully functioning ecosystems and for maintaining landscape level biodiversity is constantly diminishing (Chan *et al.* 2006, Fischer *et al.* 2007) setting an ever-increasing pressure on the intelligent planning of land-use. Intuitively, it feels that one of the most important questions conservation science faces is, how much is enough (Tear *et al.* 2005)? Unfortunately, despite great improvements in linking population viability analysis techniques, connectivity research, biodiversity monitoring methods, and climate change research to conservation planning (e.g. Nicholson *et al.* 2006, Rondinini and Chioza 2010) we are still plagued by an uncertainty about the quantitative adequacy of the protected area networks (Pimm *et al.* 2001, Tear *et al.* 2005). As scientists, we can only produce estimates about the quantity of area and resource allocation, which would be big enough to ensure the long term persistence of populations and ecosystem functioning (Tilman *et al.* 1994, Scott *et al.* 2001, Cabeza and Moilanen 2003, Svancara et al. 2005, Tear et al. 2005). It seems that science has not, and may never have, an accurate answer to the question of how much area is needed to meet the varying goals for the conservation of biodiversity and the functions of ecosystems (e.g. Svancara et al. 2005, Tear et al. 2005, Rondinini and Chiozza 2010). Furthermore, although it certainly seems that the will for increased resource allocation for conservation does exist, at least at political level (CBD 2010), recent studies reveal that large scale degradation of ecosystems and loss of biodiversity are still genuine threats both outside and inside protected areas due to various mechanisms (Butchart *et al.* 2009, Laurance et al. 2012) the most obvious being perhaps climate change (Willis and Bhagwat 2009).

On the other hand the "how much is enough?" as a scientific question has been challenged altogether as a "how much is enough myth" (Wilhere 2008). Wilhere (2008) reasons that the word "enough" makes the question a societal and ethical consideration as we are actually dealing with the tolerable extinction risks or tolerable risk of crossing thresholds in ecosystem functioning. On the other hand, no matter how accurate the ecological knowledge that science can provide for conservation decision-making, the targets set for conservation ultimately always reflect the will of society (Pullin *et al.* 2013). However, at the moment, even when society is comfortable with producing quantitative targets for conservation, scientists can only estimate the probabilities for the success in reaching the targets (Tear *et al.* 2005). The question thus becomes an ethical consideration on what kind of probabilities we can accept and live with. Wilhere (2008) concludes that it is wrong for scientists to claim to be able to answer a question that is ultimately an ethical issue, influenced by the values of society. This can create unrealistic expectations that have already been noted in restoration ecology (Hilderbrand et al. 2005, Hobbs et al. 2011). Thus, we have a two dimensional problem: as a society we do not have (nor ever will have) an unambiguous opinion on the ethical problem of how much "enough" really means (Mermet *et al.* 2013), and as scientists we still cannot provide certainty (nor agree on) on how to best get there, even if an consensus on the interpretation of "enough" were achieved.. Acknowledging these two dimension, however, opens up possibilities for intelligently, and as comprehensively as possible, alleviating the how much is enough problem.

1.3 Alleviating the "how much is enough" problem

1.3.1 Towards better trade-offs – conserve the best, restore the rest?

The question of how (in)sufficient our actions are may be impossible to explicitly define from both ecological and ethical perspectives. None the less, it is still a very important to ask this question. Understanding what is likely to be lost and being able to estimate the economic and ecological costs of the losses and the potential counter actions helps both scientists and society to quantify the possible dimensions of "enough". In other words, weighing the costs, benefits and uncertainty of conservation actions is the only way for society to reach a conclusion, and also the only meaningful approach for scientists to find the best practices for getting there.

The most basic division of methods aiming at conserving biodiversity and sustaining an ecosystem's functions is dividing them into preservative and improving actions. A preservative action in its very basic form means establishing new conservation areas or avoiding impact on the most fragile ecosystems (Cuperus 2001, ten Kate 2004). Improving actions mainly include ecological restoration and the management of ecosystems so that further degradation is halted and the recovery of the system initiated (Hobbs *et al.* 2011, Suding 2011, Halme *et al.* 2013). Although all these approaches can be listed under the concept of intervention ecology (intervening in the degradation of the Earth, e.g. Hobbs 2011) they apply different mechanisms with respect to the loss of biodiversity, meaning that they also apply different pathways to alleviate problems in resource allocation. First, it should be noted that we cannot increase the area of pristine ecosystems by protection. This means that the extinction debt resulting from the degradation of ecosystems and the loss of habitats cannot be reversed or paid off by protection alone. We also need to re-establish lost habitats by restoration of ecosystems (Hanski 2000, Kuussaari 2009). Preventing the increase of our extinction debt alone may well mean protecting nearly everything that has not yet been degraded, and preventing further degradation of vast areas only partially damaged. Totally halting the loss of biodiversity (CBD 2010), in other words to stopping any further decrease and paying off the existing extinction debt, requires the large scale net improvement of ecosystems. The magnitude of the task has now been globally acknowledged: in the Conference of the Parties of the Convention on Biological Diversity (CBD 2010) 194 countries agreed to restore at least 15% of degraded ecosystems by 2020 (Aichi targets). This is, of course, going to be quite costly, to say the least. In Finland, the price tag for restoring 15% of degraded peatlands alone is around 500 000 000 US dollars (see chapter III). Although this price seems to be "enough", we are still unable to provide an ecology-based answer to whether the 1 million restored peatland hectares costing 500 million USD is enough to halt or reverse the loss of biodiversity in peatlands in Finland. This certainly deserves to be called the "how much is enough" problem.

There are, however, widely accepted approaches for alleviating the challenge of the wisest allocation of our resources. These approaches can be roughly divided into three categories: 1) Expanding the networks of protected areas cost-efficiently with the best available options (e.g. Margules and Pressey 2000, Lehtomäki *et al.* 2009), 2) avoiding or reversing unnecessary damage caused to ecosystems by economic land-use (e.g. I, Cuperus *et al.* 2001, ten Kate *et al.* 2004, Polasky *et al.* 2005), and 3) restoring the ecological value of already degraded ecosystems partly or totally (e.g. Hobbs et al. 2011, Suding 2011, Halme *et al.* 2013). Although crossing the borders of these approaches in modern land-use planning is more of a rule than an exception, finding the most profitable trade-offs between allocating resources to damage avoidance, restoration or acquiring land for protection is still complicated. Or even, as Pullin *et al.* (2013) put it, ”identifying the kind of research that is likely to deliver the most useful and cost-effective results is deceptively difficult”.

The research perhaps most committed to tackling the dilemma of resource allocation is the one related to an approach called Systematic Conservation Planning (Margules and Pressey 2000, Margules and Sarkar 2007, Kukkala and Moilanen 2013). In systematic conservation planning one of the main tenets is to reduce harmful opportunism that can lead to unnecessary losses in land-use (Margules and Pressey 2000, Pressey and Bottrill 2008, Kukkala and Moilanen 2013). That is, to create conservation area networks through the systematic evaluation of the outcome by applying concepts like complementarity and cost-effectiveness, and by considering options like protection, restoration and management (e.g. Wilson *et al.* 2009, Kukkala and Moilanen 2013).

Ferrier and Wintle (2009) provide several different ways by which systematic quantitative approaches of systematic conservation planning help to alleviate resource allocation problems. These are for example: offering operational models from planning to implementation, offering complementarity based analyses for the evaluation of the performances of different solution options, and through quantitative and systematic approaches offering informative result evaluation and presentation to avoid gaps between planning and decision-making. In addition, many systematic prioritization approaches allow the incorporation of costs into the analysis enabling a quantitative evaluation of cost-efficiency (e.g. Wilson *et al.* 2009) and, for example, a more realistic replacement-cost analysis (I, Cabeza and Moilanen 2006). This way evaluating economic and ecological trade-offs becomes more reliable, which not only makes resource allocation in decision-making easier, but intuitively also makes the decisions more acceptable by society, when the trade-offs are first quantitatively examined as a function of gained economical goods and lost ecological values (Wilhere 2008, Ferrier and Wintle 2009, Nelson *et al.* 2009). Of course, adding economics to conservation decision-making is a slippery slope with the danger of overlooking the ecological values (e.g. Arponen *et al.* 2010). However, at least at the level of the analyses the benefits of a holistic view seem to be significant (I, Naidoo *et al.* 2006), which is intuitive as most conservation allocation solutions are, in the end, economically limited.

Protected areas are not equal with respect to increasing the landscape and ecosystem level ecological values (biodiversity, ecosystem services etc.). Thus, when faced with limited resources it is not trivial how the areas to be protected are chosen or which restoration methods are implemented. Consequently, a sub discipline in systematic conservation planning, spatial conservation prioritization, has emerged (Ferrier and Wintle 2009, Wilson *et al.* 2009). Spatial prioritization tools, like Zonation (Moilanen *et al.* 2005) or Marxan with Zones (Watts et al. 2009), focus on problems in spatial resource allocation (although not always seen as resource allocation, see Game *et al.* 2013). Spatial conservation prioritization aims to a better design and an effective expansion of reserves. Common approaches are to consider irreplaceability and complementarity of areas when identifying the conservation values of the different parts of the landscape (Sarkar *et al.* 2006, Pressey *et al.* 2007, Wilson *et al.* 2009) and to reduce overall ecological losses by considering where the probability of the loss of ecological values, if not protected, is highest (e.g., Pressey *et al.* 2004, Nicholson *et al.* 2006, Visconti *et al.* 2011). In addition to the spatial analyses, the existing non-spatial methods related to systematic conservation planning focus on a variety of conservation and management actions in differing temporal scenarios, i.e. emphasising what over where (e.g. Pouzols and Moilanen 2013, Mazziotta *et al.* 2014). The "what" and "where" tend to have a trade-off, in which a meaningful consideration of different management options and temporal scenarios in a spatially explicit model becomes computationally very challenging (Possingham *et al.* 2009). Thus, with the current knowledge and tools, prioritization simultaneously in space and time often means reduced scale, data, models and predictability (Possingham *et al.* 2009).

As a summary, there are two major aspects to consider when trying to sustaining functional ecosystems: intelligently decreasing further degradation (chapters I and II in my thesis) and restoring already existing conservation areas when relevant, thus increasing the ecological value of the already protected area (III and IV). I will next concentrate to the possibilities of extending the spatial conservation prioritization outside protected areas to include all land-use and, on the other hand, to our abilities to make the necessary qualitative and quantitative difference on protected areas through ecological restoration.

1.3.2 Avoiding unnecessary ecological losses outside protected areas

Despite the development of modern conservation methods towards cost-effectiveness and complementarity, global level studies show depressing results indicating that biodiversity is still under severe threat (Butchart *et al.* 2010), even within conservation areas (Laurance *et al.* 2012). This is not only a landscape level effect of habitat loss and extinction debt, but also results from an incapability to protect the ecosystem integrity within the borders of a conservation area (Laurance *et al.* 2012). Of course, in addition to local direct anthropogenic threats, large scale abiotic factors such as climate change and increased nitrogen outputs put a huge pressure on conservation area networks to effectively protect biodiversity (e.g. Gareth *et al.* 2006, Willis and Bhagwat 2009).

The inadequacy of conservation area networks and the threats towards already protected areas have resulted in a suggestion for a shift from the protected-not protected dichotomy to more holistic land-use decision-making approaches that cover also the ecological value of the landscape outside protection (e.g. Maiorano *et al.* 2008, Mathur and Sitha 2008, Chazdon *et al.* 2009). A holistic perspective to land-use opens up the possibility to modify the "protect as efficiently as possible" problem formulations into "destroy as little as possible". Avoiding unnecessary environmental and ecological damage should of course be the fundamental guide line for all our actions (Cuperus *et al.* 2001, ten Kate *et al.* 2004). This can be divided roughly into two domains: first, when it is unnecessary to expand the development to include all areas, avoid areas where ecological losses are greatest, and second, avoid actions that cause an unnecessary amount of damage in areas under development. Tools for more holistic conservation planning, integrated to general land-use, have been developed (e.g. Gordon *et al.* 2009, Watts *et al.* 2009, Willis *et al.* 2012). For example Marxan with Zones (Watts *et al.* 2009) can be used to identify areas where some development can be incorporated without compromising the ecological values targeted to those areas (see e.g. Wilson *et al.* 2010). It is thus spatially identifying sets of targeted ecological and economical features that can mutually exist in certain defined areas. Indeed, achieving a functioning co-existence of economic and ecological returns seems a fitting answer to the "where to put things?" question (Polasky *et al.* 2008), as long as economic returns are not overemphasized (Arponen *et al.* 2010). Several case studies have demonstrated the power of the holistic view on land-use prioritization, allowing the identifying profitable sustainable co-existence and directing land-use to the ecologically least vulnerable or valuable areas (I, Ban and Vincent 2009, Klein *et al.* 2010, Weeks *et al.* 2010, Wilson *et al.* 2010, Moilanen *et al.* 2011, Grantham *et al.* 2013). However, as the holistic land-use perspective is relatively young (at least when using modern prioritization methods), further research is still needed to identify ecosystem-specific best practices.

When incorporating the spatial conservation prioritization methods outside the traditional conservation framework the implementation of the results may be more difficult than in the case of e.g. the "straight forward" expansion of protected area networks. The research-implementation gap is a real problem (Knight *et al.* 2008) and several studies deal with operational models for implementing conservation science to practice (Margules and Pressey 2000, Knight *et al.* 2006, 2011, Game *et al.* 2013, Dicks *et al.* 2014, Walsh *et al.* 2014), demonstrating that best practises are both needed and hard to define. If a prioritization analysis truly is cost-effective and complementary, i.e. performs near optimally, then any deviation from the proposed results means shortcomings for the scientists in either including all of the relevant factors to the socio-ecological model of the analysis or in delivering the information all the way to the final decision-making level. Unfortunately, a "writing the wrongs" culture is still not exactly in the hard core of science, despite some progress (see e.g. Redford and Taber 2000, Knight 2006, Game *et al.* 2013). Thus, finding the best practices to close the research-implementation gap remains especially hard as long as the comparison of prioritization analyses and actual outcomes, which is needed to identify and evaluate the reasons of possible shortcomings, is not a commonplace (II). As long as a significant research-implementation gap exists, there will also be an unnecessary gap between the possible and realized benefits of holistic land-use prioritization.

1.3.3 Restoring ecosystem structure and function

Ecological restoration refers to actions aimed at re-establishing a desired state of an ecosystem that, for some reason, has been degraded or lost (Bradshaw 1996, SER 2004). It may include all or part of the systems natural features and dynamics (SER 2004). In layman terms it is the fixing a damaged ecosystem. The science of ecological restoration, restoration ecology, is a relatively young discipline. However, our current interest towards restoring ecosystems has resulted in a major output of case studies, experimental set ups, theoretical modelling, and conceptual discussion. While the broad lines of ecological restoration have been more or less agreed upon (SER 2004), theoretical and conceptual discussions on the aims, and possibilities of restoration are still ongoing (e.g. Hobbs 2001, Hildebrand *et al.* 2005, Clewell and Aronson 2006, Cabin 2007, Giardina *et al.* 2007, Hobbs 2007, Miller and Hobbs 2007, Hobbs *et al.* 2009, Jackson and Hobbs 2009, Perring et al. 2014) and the detailed character of ecological restoration is still further developing (Hobbs *et al.* 2011). The concept of restoring has also raised big philosophical questions for restoration ecology: "Can or should we restore exactly what once existed?" (Hildebrand *et al.* 2005, Jackson and Hobbs 2009, Perring *et al.* 2014) or "Can and should human culture be included?" (Higgs 2005, Clewell and Aronson 2006, Mace 2014). These questions may at first seem arbitrary in a situation where we still struggle to properly close existing ditches in a peatland. However, the philosophy and ethics of ecological restoration are closely related to large scale goals and thus also to the cost-effectiveness of any intervening actions. For example, the frequent (often annual) management needed to prevent the unwanted succession of semi-cultural ecosystems back to a more natural dynamic state intuitively has a massive cumulative price through time. On the other hand, restoration with a detailed historical target in a changed landscape is likely to be costly, if at all possible (Jackson and Hobbs 2009, Hobbs *et al.* 2009).

Ecological restoration and management rightfully attract keen scientific attention also at the ecosystem scale. Ecological restoration alleviates the limited space problem by restoring the ecological value at the ecosystem level of, for example already protected areas that have been partly or totally destroyed. We are indeed living in an era of ecosystem manipulation: while the human population has influenced ecosystems almost everywhere around the Earth (Foley *et al.* 2005, Millennium Ecosystem Assessment 2005), ecological restoration offers approaches aiming at the re-establishment of ecosystems' functions and structures, such as C sequestration and species communities, respectively, in order to respond to the threats of habitat and ecosystem level degradation (Bradshaw 1996, Dobson 1997, Vanha-Majamaa *et al.* 2007, Hobbs *et al.* 2011). From a broader perspective, ecological restoration is a form of ecological intervention (Hobbs *et al.* 2011). In ecological interventions the goal can also be to increase the ecological values of degraded ecosystems without definitively aiming at a specific historic state of the ecosystem (Jackson and Hobbs 2009, Hobbs *et al.* 2011). The results of restoration, like an increase in biodiversity or the revival of populations of individual species, are promising (e.g. Benayas *et al.*, 2009), and new targets for the restoration of ecosystem functions and services (profitable and/or needed ecosystem functions) are being developed (Hobbs and Cramer 2008, Benayas *et al.* 2009, Aerts and Honnay 2011, Bullock *et al.* 2011, Suding 2011). Still, it seems unlikely that restored ecosystems will ever be exactly like their pristine targets, and there are serious doubts concerning the recovery of the functions of ecosystems after restoration (Zedler and Kercher 2005 Benayas *et al.* 2009, Moreno-Mateos *et al.* 2012). The recovery of most of an ecosystem's functions is often thought to be dependent on the recovery of its original structure. However the detailed relationship between ecosystems' functions and different parts and forms of their structures is still far from well understood (Bradshaw 1996, Cortina *et al.* 2006, Cardinale 2012). The general assumption seems to be that ecosystem structures and functions are linked, but with varying strength making detailed predictions on the outcome and possible trade-offs of the restoration difficult (Ehrenfeld and Toth 1997, Aerts and Honnay 2011, Bullock *et al.* 2011, Suding 2011, Halme *et al.* 2013).

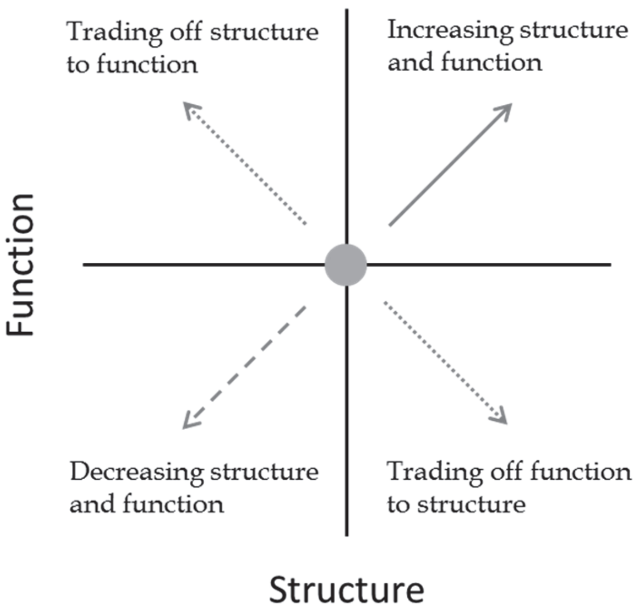

FIGURE 1 The four possibilities of change in an ecosystem in a two-dimensional space of an ecosystem's structure and function. Note that measures of structure and function may differ according to the perspective of the set target. Thus, an increase in structure and function from one perspective may mean their decrease from another perspective (e.g. semi-cultural versus natural state of an ecosystem).

The simplest model for ecosystem structure (e.g. community composition), functions (e.g. carbon sequestration) and ecological restoration is two dimensional (Bradshaw 1996). However, it is often not recognized that within the two-dimensional space of structure and function, every ecosystem has four possibilities or directions to move towards (Fig. 1): both function and structure can increase or decrease, structure can increase while function decreases, and function can increase while structure decreases. With these theoretical possibilities, the set targets for manipulating an ecosystem and the resulting change can be viewed from three different perspectives: 1) the natural ecosystem perspective, where the targeted structure and function are those typical for a natural ecosystem of the focal spatial context 2) the target oriented perspective, where the structure and function are defined by a target and can intentionally differ from the natural ecosystem 3) the total increase perspective, where the target is not a specific structure or function, but an increase in their common components (like biodiversity or carbon sequestration) *per se*.

In options 1 and 2 the measure of structure is similarity compared to the targeted state, natural, semi-cultural or novel. In option 3 structure and function are not relative to the target and both can increase above a "natural level", relevant examples being the ability of certain plantations to sequester CO_2 (Lindenmayer *et al*. 2012) and the effects on biodiversity (at site or landscape level) in many semi-cultural habitats (Middleton *et al.* 2009). In options 2 and 3 already the target includes some level of novelty, whereas in option 1 novelty may arise from an inability to reach the original target. The division is not relevant only for the sake of definition. Indeed, many ecological questions related to systematic conservation planning and evaluation of the success (and further actions) depend on the perspective the target is looked at. For example, the cost of restoration may be mitigated or even completely offset, if we consider that restoration may produce ecosystem services with economic value (Kimmel and Mander 2010, Bullock *et al.* 2011, Law *et al.* 2014). From this perspective the relationship of an ecosystem's functions and the structure needed to provide the functions becomes increasingly important. Notably, the existence of valued ecosystem services and biodiversity are not necessarily naturally spatially correlated (Bullock *et al.* 2011, Lindenmayer *et al.* 2012). If restoration is conducted for the sake of biodiversity it is important to know how much of possibly cost-effective novelty can be applied while still also obtaining the function based ecosystem services (Lindenmayer *et al.* 2008, Seastedt *et al.* 2008, Perring *et al.* 2014), and on the other hand, if allowing novelty can produce a net gain in both structure and function compared to aiming strictly at a natural outcome. Of course, perhaps the most relevant question is: how much site level structure or global biodiversity can we let go in development projects in the first place without running the risk of the costs of lost ecosystem services outweighing the gain of the development (Cardinale *et al.* 2012, Hooper *et al.* 2012, Naeem *et al.* 2012)?

Restoration ecology has in many contexts been praised as the modern approach for alleviating land-use trade-offs by, for example, reducing existing extinction risks (Kuussaari *et al.* 2009) or by compensating net negative effects through offsetting (Suding 2011, Maron *et al.* 2012). Just as all areas do not contribute equally to, for example, the conservation of biodiversity, not every ecological restoration or management action provides equally successful outcomes, nor do all successful outcomes contribute equally to biodiversity and ecosystem service targets. Thus, it is essential to include ecological restoration and management considerations into modern systematic conservation planning (Wilson et al. 2011, Kukkala and Moilanen 2013). In general, systematic conservation planning or ecological decision analysis applies to restoration just as well as to spatial conservation prioritization. However, as is demonstrated above, relevant planning of the allocation of a restoration effort is a seemingly more complicated problem, as it needs to incorporate high-end ecology of ecosystem dynamics (Suding *et al.* 2004, Suding and Hobbs 2008) to reach meaningful targets with a tolerable (un)certainty (Lindenmayer *et al.* 2012, Possinghan *et al.* 2009). While spatial conservation planning is concerned primarily with biodiversity patterns in landscapes (although in an increasingly dynamic manner), restoration planning needs to include an ecosystem- and often site-specific evaluation of what once was, what is, and what will be with a deep ecological understanding of the fact that changes in different features are not independent from each other or linear in time (Suding et al. 2004, Suding and Hobbs 2008, Menz *et al.* 2013). Furthermore, there is no reason to assume that degraded ecosystems are static. Further degradation can be deceivingly slow, but an ecosystem may also have resilience, or an ability to restore "spontaneously" (Prach *et al.* 2001, Prach 2003, Cole et al. 2014) demonstrating how complicated it is to produce a relevant evaluation of the cost-effectiveness of restoring even a single site (Prach and Hobbs 2008, Lake 2013). After all, the effect of a restoration action is not the resulting state of the ecosystem, but the difference between the post-restoration state and the state the system would have changed into if not restored. Needless to say, this kind of evaluation is rarely possible in small or even larger restoration projects (Ruiz-Jaen and Aide 2005, Miller *et al.* 2010, Wortley *et al.* 2013). Still, in individual cases of ecological restoration the aim to restore (or die trying) is usually intent at a level of a spatially defined site. When searching for optimality in planning for multiple sites, estimates on the actual effects or certainty of the outcome are even harder to define (Possingham *et al.* 2009). In a systematic conservation planning process, including restoration considerations, one has to decide whether to protect or protect and restore and between any other possible combinations of total or partial protection and forms of ecosystem manipulation and possible restricted economic land-use (e.g. Banks-Leite 2014). Tools for optimal restoration planning exist (Noss *et al.* 2009, Wilson *et al.* 2011, Pouzols and Moilanen 2013). However, they are naturally computationally challenging and data hungry (Possingham *et al.* 2009, Menz *et al.* 2013). Thus, given the schedule and the magnitude of the task (e.g. Aichi targets in CBD 2010), lower-dimensional and operationally more feasible approaches may be relevant (IV). In addition, a strategy needs to be chosen to decide how the possible trade-offs are perceived. The debate between novelty and naturalness can be intense and especially here the deceptiveness of the human mind (Gilbert 2011), myths related to restoration (Hilderbrand *et al.* 2005) and different social and scientific rationales of restoration (Clewell and Aronson 2006) need to be accepted, acknowledged and finally objectively weighed.

1.4 Boreal peatlands as a study system

The empirical research presented in the four chapters of this thesis is carried out mainly using boreal peatlands as a study system. Boreal peatlands have several characteristic features, which influence their use as a study system from the perspectives of both conservation prioritization and restoration ecology. Peatlands in Finland have been in their primary successional trajectory since the last glacial period. In other words, the dynamics of the system are slow making spatial conservation prioritization easier. In peatlands, what you see is what you get, for a relatively long time. Thus, the static view of the system is quite realistic and decreases uncertainty in analysis results (Possingham *et al.* 2009). Global environmental threats, like climate change (e.g. Willis and Bhagwat 2009), are of course an exception and their effects on boreal peatlands in general, or in Finland specifically, still need further research (Strack 2008).

Peatlands are hydrological entities: different parts of a peatland are connected to each other on varying levels determined by their hydrology. Basically, this means that no major land-use activities (e.g. peat-mining) can usually be carried out without affecting the entire hydrological entity. However many larger, for example partly forestry drained peatlands, can still hold significant biodiversity values, if parts of the peatland have a less degraded hydrology. From a conservation planning perspective this means that the peatland areas should in the end be prioritized as hydrological entities of varying size and shape rather than as fixed pixels. Analytically, the fact that the planning units vary in shape is a minor complication. However, comparing the priority of planning units with varying size may cause confusion, especially if the analysis is complementarity based (personal observation). This demands special consideration in the evaluation and especially in the presentation of the results.

The conditions in peatlands are relatively harsh: the vegetation has to endure partly anoxic and often acidic conditions, which vary as a function of the hydrology of the area (Rydin and Jeglum 2006). The harsh conditions have a directional effect on the peatland community assembly, which shows in a relatively consistent and straight forward general response (Haapalehto 2014) to restoration methods of, for example, returning the original hydrological conditions destroyed by drainage for forestry. However, since the dynamics of peatland ecosystems are slow, species specialized for peatlands may have not been selected for characteristics that promote rapid colonization, which can show as slow post restoration recovery of the original plant diversity, (Haapalehto *et al.* 2011, Hedberg *et al.* 2012).

With the social awareness of climate change, increasing attention is paid to an ecosystems' ability to absorb carbon (Feng 2005, Davidson and Janssens 2006). Compared to other ecosystems, peatlands are the largest reserves of organic carbon in the soil (e.g. Gorham 1991, Kercher 2005, Page *et al.* 2002, 2011): boreal and subarctic peatlands alone comprise *circa* 30 % of the global soil carbon pool (547 Pg C) (Yu, 2011). This feature along with very characteristic species communities make peatlands very interesting from both conservation and restoration perspectives.

1.5 Aims of this thesis

My aim is to investigate how and to what extent modern conservation planning methods and restoration of ecosystems can help us to alleviate the apparent resource allocation challenges. In the first two chapters I study the design and implementation of spatial conservation planning methods in land-use planning outside protected areas. First (I) I develop an inverse use of a spatial conservation prioritization approach and study the possible ecological gains of extending the spatial conservation methods to land-use on non-protected peatland landscapes in the province of central Finland. I will then continue (II) with the peatland zoning prioritization to study how the prioritization analysis (I) was implemented in the end and explore the reasons for the possible differences between land-use allocation suggestions of scientists, planners and in the resulting final land-use zoning. I will also investigate the quantitative ecological costs of the research-implementation gap and suggest how the gap can be exposed and hopefully made smaller.

In the third and fourth parts of the thesis I will investigate ecological restoration as a means towards more effective conservation resource allocation. I first investigate the possibilities to increase the value of degraded ecosystems by ecological restoration. More specifically, I study if the peatland restoration methods, most commonly used in Finland, actually succeed in returning the peatland ecosystems' most characteristic structure and functions. Finally, in chapter four I investigate possibilities to make national to global scale restoration target setting more comprehensive. That is, to include the level of degradation and the effect of restoration to target and goal setting, instead of only considering the total area, on which restoration actions are then applied. More specifically, the aim of my thesis is to answer the following questions:

1. How effectively ecological values can be retained by applying spatial conservation prioritization methods in ecological loss avoidance outside protected areas (I)?
2. How to identify, quantify and demonstrate a research-implementation gap and its effects in conservation planning (II)?

3. Is ecological restoration effective in restoring a peatland ecosystem's most characteristic functions of peat growth and carbon sequestration (III)?
4. Is recovery of the ecosystem structure a prerequisite for recovery of ecosystem functioning (III)?
5. How should large scale restoration resource allocation be effectively planned (IV)?

2 METHODS

2.1 Avoiding biological diversity loss outside protected areas

2.1.1 Biodiversity to peat-mining trade-off in Central-Finland

I illustrate the benefits of considering ecological values in land-use planning outside protected areas with an inverse application of spatial conservation prioritization. I apply a spatial conservation planning method Zonation to a landscape level land use zoning that was prepared in the province of Central Finland. This zoning plan directs spatial allocation of peat-mining areas within the province. The objectives for the spatial allocation of peat-mining are both quantitative and qualitative: In order to satisfy the varying interests of different parties (e.g. stakeholders, peat-mining companies, environmental administrative, regional energy demands), a flexible preliminary target of 6000 - 12000 ha of suitable peat-mining area was to be reserved for peat production. Simultaneously, the peat-mining areas must be allocated so that the ecological losses are minimized.

The focal set of areas consists of 306 partly ditched peatlands (36503 ha) that need to be put in a prioritized order for peat-mining while simultaneously safeguarding the ecological values of the landscape. Conservation areas and other unditched peatland areas of highest ecological value were pre-excluded from the prioritization as they would not be allocated to peat-mining in any case. Thus the prioritization concentrates on areas that are all predetermined by a stakeholder group (peat-mining companies, local energy company, landowners, nature conservation organizations, local and national environmental administrations, and the Finnish Forest and Park Services) to in principle be suitable for peat production. However, the focal peatlands have varying proportion of the total area suitable for peat-mining (peat-mining potential) and they contain variable level of ecological values.

2.1.2 Zonation

I used Zonation method and software (Moilanen *et al.* 2005, 2009, 2012) to perform the land-use prioritization analysis. Using Zonation analysis results one can identify areas with high relative importance for retaining ecological quality and connectivity simultaneously for multiple features in the landscape (Moilanen *et al.* 2005, Kremen *et al.* 2008, Lehtomäki *et al.* 2009, Moilanen et al. 2012, Willis *et al.* 2012).

In Zonation framework the starting assumption is that from conservation perspective it is best to conserve the entire landscape. Then, a priority ranking for the chosen units is developed via iterative minimization of ecological losses. In the analysis, grid cells (ranked units) are iteratively removed (or proposed to be allocated for economic land-use) while minimizing aggregate loss of biodiversity features considering their weights and relative change in their distributions. The operational principle of the analysis can be summarized as maximizing retention of weighted range-size normalized feature richness while retaining a complementarity-based balance across all features (Moilanen et al. 2011). The recorded order of removal is then used to produce a relative ranking.

Zonation suits especially well for holistic land-use planning. The analysis balances heuristically between maintaining different ecological features, habitat abundances, quality, and connectivity. In addition, the case specific prioritization models in Zonation analyses can be set to balance between multiple direct or indirect costs of conservation (e.g. land price or estimated economic gain if the land was in economic use, respectively). Furthermore, Zonation operates without *a priori* set targets, prioritizing the entire investigated landscape, i.e. producing rank to all planning units. Trade-offs between conservation and economic development may be investigated in a more informative and flexible way when the decision where to draw the line does not have to be set *a priori*. Providing the prioritization over whole landscape instead of a dichotomic division of protected or not, enables profound trade-off evaluation to support decision-making.

2.1.3 Data and analysis structure

The biodiversity data of the prioritized areas were mainly gathered by biologists of the Regional Council of Central Finland. For the present analysis, habitat type data were grouped into 18 more robust classes according to the main peatland types and their nutrient levels, including moderately and severely changed peatlands and peat-mining areas as separate habitat types.

I constructed seven analysis variants of increasing complexity by adding new data and analysis features to previous analysis variants (Fig. 2). I built the analysis variants in the following order: I) habitat type, II) + habitat type weight, III) + habitat condition weight, IV) + connectivity interaction with unditched peatland areas over 20 ha, V) + peat-mining potential, VI) + red listed and rare species, and VII) + bird territories (see chapter I for details). I developed the analysis in stages so that I could verify the correctness of input data and analysis setup at each step.

I weighted features according to an expert-based decision about their relative importance (I). Feature weights vary within and between feature groups. I considered some core feature groups (e.g. habitat types) more important than 'supplementary data' (e.g. bird territories), and some features within each feature group more important than others (e.g. nutrient rich fens versus nutrient poor pine mires). The final weight of each ecological feature is defined by the relative importance of the feature within the feature group and by the relative importance of the feature group to other feature groups. I defined cost of the peatland areas as lost "peat-mining potential", measured in units of area suitable for peat-mining. In other words, this cost is an opportunity cost incurred when an area is not available for peat extraction. Grid cells that remain to the end of the ranking are mutually complementary and have generally high weighted feature richness and rarity combined with low peat-mining potential. Focusing on the low-priority end of a Zonation priority ranking (the first grid cells removed) allows implementation of the inverse spatial conservation prioritization principle: identification of areas that have comparatively low ecological values but high utility for other land uses.

I performed the initial prioritization at a 25 x 25 m grid resolution. For the purpose of zoning, I then constructed the final prioritization with entire peatland entities as planning units (306 planning units). I did this because when a peatland is mined, the drainage of the area usually affects the entire peatland, not just the individual grid cells where peat has been extracted. On the other hand, fragmented peat-mining would not be economically cost-efficient either.

In the end I used the feature-specific performance curves to investigate the quantitative trade-off between ecological features and peat-mining potential. I compared the outcome of the full prioritization with a random allocation scenario and with a greedy selection scenario in which peat extraction was maximized without regard to ecological values.

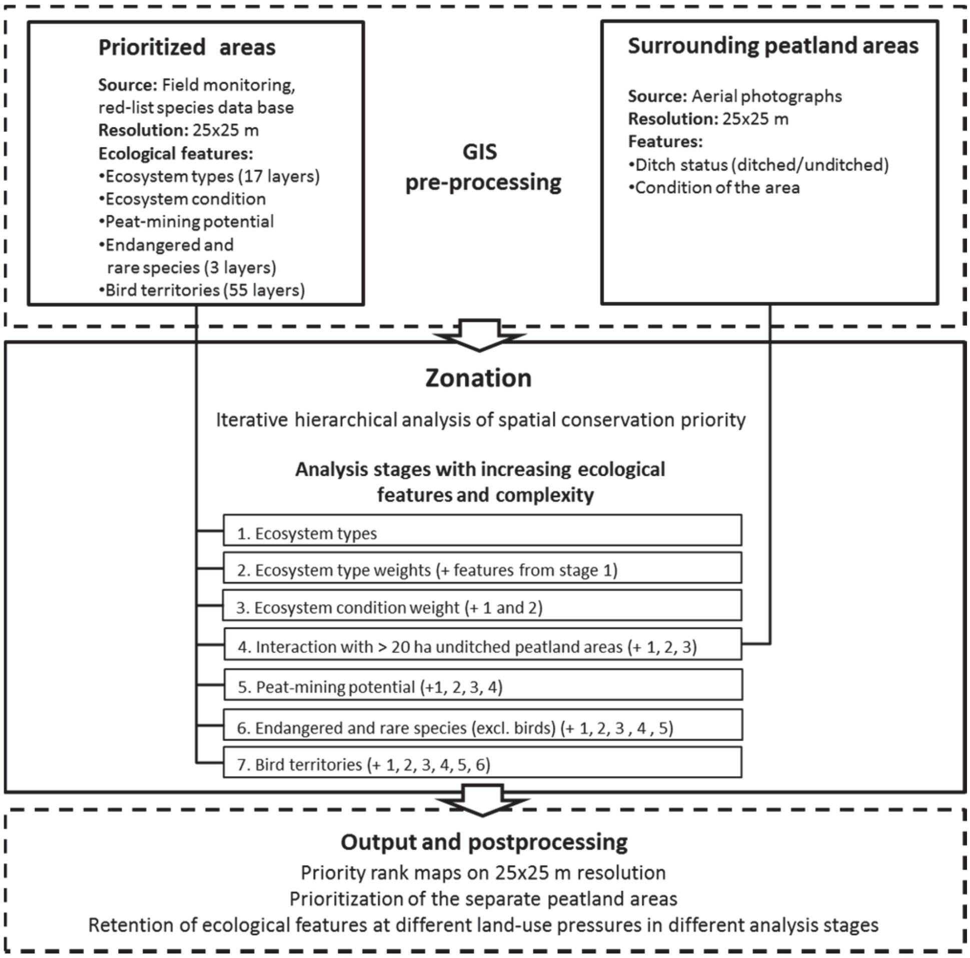

FIGURE 2 Flowchart of the analysis performed in chapter I, giving an overview of the workflow, used data and analysis stages.

2.2 Costs and reasons of a research-implementation gap

Here I follow up on the case study of ecological impact avoidance in spatial prioritization of peat-mining (I). I document the difference between the scientifically-based suggestion (hereafter scientists' suggestion), suggestion of the planning authorities (planners' suggestion), who also commissioned our analysis, and the final land-use zoning plan that was negotiated between planning authorities and stakeholders. I will also investigate the reasons between different suggestions for the zoning. Then, I will further analyse the apparent ecological and economic costs of the gap between research and implementation, and illustrate how quantitative and qualitative trade-offs in resource allocation projects can be exposed through replacement cost analysis (Cabeza and Moilanen 2006). In addition, I will investigate the phases of the whole planning-decision-making process in our case study to illustrate the effect of process complexity to the research-implementation gap.

I analyze three differences between scientists' suggestion, planners' suggestion, and the final land-use zoning, which was negotiated as a compromise between planning authorities and local stakeholder politics. First, by how many sites does the scientists' and the planners' suggestions differ from the final zoning plan? Second, how well do different zoning suggestion measure in terms of ecological and biodiversity performance? Third, what were the causes behind the differences between the scientists' and planners' suggestions and the final land-use zoning? Preferences underlying decisions that differed from the scientists' suggestion were not always traceable, but in many cases a clear reason could be identified.

Specifically, I compare five suggestions: 1) the scientists' suggestion based on the inverse prioritization analysis and extensive avoidance of impacts (I); 2) the scientists' suggestion using the same inverse prioritization, but a stakeholder-mandated higher requirement for peat extraction; 3) the scientists' suggestion modified by excluding certain sites from peat-mining because of high expected damage for surface waters and recreational value; 4) the planners' suggestion, 5) the final land-use zoning suggestion.

I describe the process and discus the results and the reasons behind the differences together with the planning authorities responsible for the zoning and commissioning the prioritization analysis. They are also my co-authors in the study (II) to enable proper discussion of the reasoning. The co-authors roles in the zoning project were: zoning project leader and expertise on peat-mining (OR) and leader of the ecological evaluation and expertise on peatland biology (RV).

2.3 Returning ecological value of degraded peatland ecosystems

2.3.1 Study design

To study the restoration of ecosystem structure and function I selected 38 previously unstudied *Sphagnum* peatlands from southern Finland. I included sites from four categories: drained peatlands (n = 9), previously drained and restored 3-7 years before the study (hereafter 5 years ago restored, n = 9), previously drained and restored 9-12 years before the study (hereafter 10 years ago restored, n = 10), and pristine peatlands (n = 10) (see III for example pictures of the categories). Based on topographic data and field observations, the study sites were hydrologically independent from each other. Distances between the study sites ranged from 200 m to 200 km. All of the disturbed sites were drained for forestry by the state ca. 40 years before the study. Original vegetation type and tree stands of the disturbed sites were similar to those of the chosen pristine control sites as determined by field observations and old aerial photographs. The now restored sites were designated to conservation in 1980’s with a subsequent decision to restore the drained sites inside the conservation areas. Time of restoration was available at the habitat database of the state owned land. Restoration measures included filling in the ditches with peat, construction of dams, and partial removal of tree stand in cases where drainage had significantly increased tree growth. The amount of trees removed was adjusted to mimic the pre-disturbance tree cover determined from aerial photographs. The used restoration measure is straight forward in the sense that it relies on natural re-establishment of target species’ populations from nearby relict sources and often costly transplantations of species or fine scale habitat engineering (e.g. for individual target species) was not applied. Restoration of the sites was conducted by Natural Heritage Services of Metsähallitus (governmental institution responsible for management of conservation areas).

The sampling design consisted of a systematic grid of 15 1-m² vegetation plots in a 10 × 20 meter area at each site. The sampling plots were placed in three transects running parallel to the ditch line 5, 10, and 15 meters from the ditch. The plots were located at 4 meter intervals along each transect forming a grid (Fig. 3). The location of the grid was randomized within the area of the focal habitat type at each site.

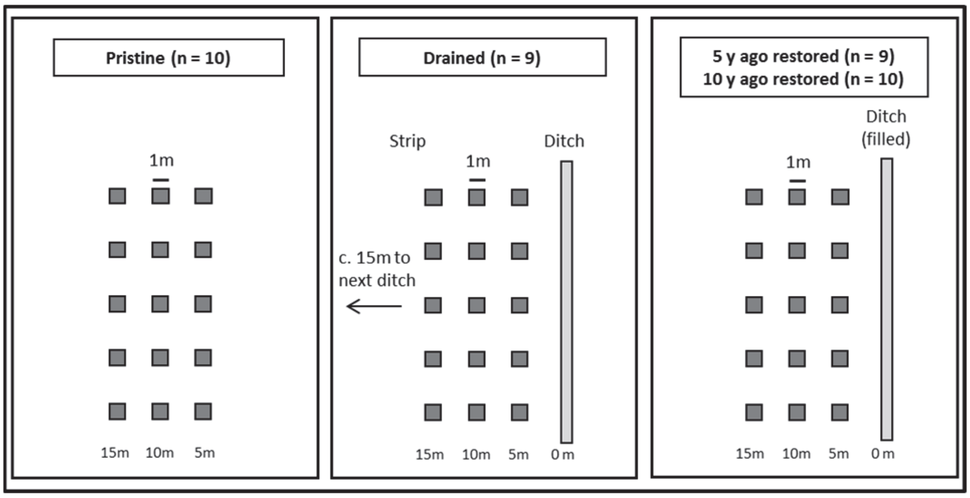

FIGURE 3 Set-up used for studying plant community composition, peat accumulation, and carbon sequestration of the 38 study sites in chapter III. The grid design consisting of 15 1-m² sampling plots at fixed distances from each other and the ditch line was similar in all treatments except for the ditch line missing from the pristine sites and being filled at the restored sites.

2.3.2 Measuring changes on peatland structure

I sampled the full vegetation at the 15 1-m^2 sampling plots at each site (Fig. 3). From each plot I recorded relative abundance as a % cover for all plant species based on visual estimation.

To study changes in vegetation I compared the similarity of the composition of the plant communities between drained, 5 and 10 years ago restored, and pristine sites. I compiled a plant community sample for each study site by calculating average relative abundances for each plant species over the 15 1-m^2 vegetation sampling plots. I used average values for each site, because my target was to assess general patterns of vegetation with respect to surface peat growth rate and carbon sequestration. To compare communities I used Bray-Curtis community (dis)similarity measure that considers both species' identities and their relative abundances (e.g. Magurran, 2004). I then studied the effect of treatment on plant community composition by comparing the Bray-Curtis community similarity of the study sites within and between the treatment groups. In addition, I performed NMS (Non-metric Multidimensional Scaling) ordination with Bray-Curtis similarity as the distance measure, for visual inspection of the community similarities. The estimation of the peatland functioning (see below) was, with the methods I used, possible at a site level and for this reason also the vegetation analysis was kept simple. More detailed analysis on the within site differences in the vegetation's response to restoration can be found in Haapalehto *et al.* (2014).

2.3.3 Measuring changes on peatland functions

I measured the changes in the peatland ecosystem functioning by sampling the most characteristic feature of peatland ecosystems, peat. I focused the measurements on the uppermost 20-25 cm surface peat layer (hereafter simply surface peat). This surface peat covers the main range of water table level fluctuation in natural sites and most of the layer that becomes exposed to increased aeration in drained sites.

First, I estimated the apparent annual growth rate of the surface peat. I collected 25-35 small (< 1.5 m) Scotch pines at each study site at the 10 x 20 meters sampling area. I determined the lengths from the pines' root collar to the peatland surface (i.e. from root to shoot transition to the top of the moss layer, marked to the shoot when collected) and the age of the pines (annual rings close to the root collar). By doing this I was able to estimate site specific peat growth rates (see Ohlson and Dahlberg, 1991 and Borggreve, 1889 for the "pine method" and III for more detailed description). I calculated the apparent annual vertical growth of surface peat as a linear regression coefficient between the rooting depth and age of the pines for each site. I then compared the coefficients between the treatments.

To study more detailed changes in the surface peat I collected six cores of surface peat (0-20 cm) with a side-cutting box sampler (sampler area: 8.3 × 8.4 cm) at each site. I collected the samples near the vegetation plots, at the 5, 10 and 15 m distances from the nearest ditch in drained and restored sites (three samples at each distance).

I first studied the effect of drainage on the accumulation of carbon (C) to the surface peat by estimating the C loss from the 0-20 cm peat samples during the approximately 40-year drainage period relative to the pristine sites. Decomposition leads to the loss of C, increase of bulk density and enrichment of ash content of peat. Drainage causes compaction of the peat and the amount of C per volume is not linearly comparable between treatments. Thus, I calculated an expected mass of C for the surface peat samples of drained peatlands by multiplying their observed ash and aluminium (Al) contents with the expected average C to ash and C to Al ratios obtained from the pristine peatlands (see III for detailed description of the chemical analyses). I was then able to estimate the relative loss of C (ΔC_{ASH} and ΔC_{AL}) for each sample as a difference between the expected and the observed mass of C of the samples from the drained sites (Grønlund *et al.* 2008, Leifeld *et al.* 2011). I used two measures (ΔC_{ASH} and ΔC_{AL}), because drainage may result in increased leaching of mineral cations (e.g. Prevost *et al.* 1999, Pitkänen *et al.* 2013), resulting in a potential bias in ΔC_{ASH} estimates towards underestimation of C loss. Aluminium is retained relatively strongly at cation exchange sites (Wieder *et al.* 1988) and this modification revealed considerably higher estimates of C loss than Δ_{CASH} (see results). For comparison, I also calculated ΔC estimates using other elements (III).

In addition to C loss I determined the recent apparent rate of C accumulation (RERCA) in surface peat after restoration and for comparable period for the pristine sites. By definition, RERCA only includes C bound in peat above a dated horizon and it is the net result of biomass production and decomposition (g C m-2 yr-1). Since the post-restoration time period varied among the sites in the 5 and 10 years ago restored categories, I used here a linear regression to model the cumulative C mass with time (years since restoration) and interpreted the slope as RERCA, i.e. increase of C store with one year increase of age (g m-2 yr-1). The post-restoration peat was separated in the laboratory from the visually distinctive older layer that represented the drainage period. The mass of C in each sample was calculated by multiplying dry mass with the measured C concentration. The linear regression was forced through origin (i.e. zero peat depth corresponded to zero age) and the slope was tested against the expected slope, in other words, the average 10-year RERCA of pristine sites, which was calculated according to the depth of 10-year old strata based on the pine method (see above).

2.4 How to implement global restoration targets

I investigate the dimension of restoration target setting by illustrating differences of purely total-affected-area based target setting versus target setting that considers both total-area-affected and the actual effects of the applied restoration actions. I will start by investigating what the "*restoration of at least 15 per cent of degraded ecosystems*" (Nagoya Treaty: Aichi Biodiversity Targets, CBD 2010) actually implies. I will then introduce a ten step procedure for how to apply target setting that considers the total area affected, the ecological effects of the applied methods and the costs of the actions per unit of area. In other words I will produce 1) a rationale for the large scale restoration target setting, 2) an operational model for the implementation of the rationale, and finally 3) a step by step platform for the operational model. The applied method is heuristic and the example at this moment only includes expert opinion on the restoration effects and costs. However, the expert opinion is obtained from the work of a multi-stakeholder working group for enhancement of the state of the habitats in Finland. This degraded habitats' enhancement plan in commissioned by the Finnish Ministry of Environment to investigate different possibilities to implement the global 15% restoration target agreed on the Nagoya treaty.

3 RESULTS

3.1 Avoiding biological diversity loss outside protected areas

3.1.1 Biodiversity to peat-mining trade-off in Central-Finland

My results show that the analysis applying the inverse spatial conservation prioritization approach was effective in identifying profitable trade-offs between peat-mining and biodiversity (I): on average 82% of the distributions of the observed main biodiversity features were remaining in the prioritized landscape up to the loss of circa 42% of peatland area and achieving 47% (c. 7000 ha) of the total peat-mining potential (Fig. 4, I).

In the random allocation scenario the biodiversity features' remaining distributions and the peat-mining area would on average change in direct proportion to total area allocated for peat-mining (I): 47% of the peat-mining potential would be achieved in 47% of the total area, retaining on average 53% of the biodiversity features' distributions. Consequently, at the prioritized peatlands, applying the inverse spatial conservation prioritization would result in reducing ecological losses on average by 62% (retaining 82% versus 53% of the biodiversity features) relative to the random allocation scenario where the biodiversity values of the areas would be ignored (I).

In the scenario where peatlands were allocated to peat-mining according to their peat-mining potential only (greedy-selection scenario), 47% of the peat-mining potential was achieved with only 34% of the total peatland area allocated to peat-mining (I). The high overall reduction in total area resulted in an ecologically better outcome than the random-allocation scenario, as the ecological values declined on average in direct proportion to area in both scenarios (I). Still, there was high variation in retention of different biodiversity features in the greedy-selection scenario (I) resulting in relatively high and unnecessary losses of some of the observed biodiversity features (I).

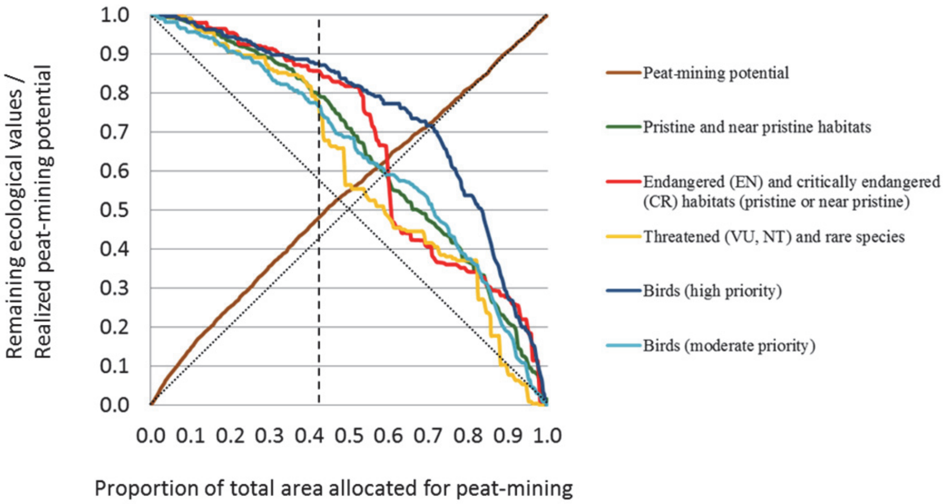

FIGURE 4 Performance curves of the final prioritization for the main ecological values and peat-mining potential. I have depicted the proportion of remaining ecological values and the proportion of realized peat-mining potential as functions of the proportion of the total area allocated to peat-mining. The thin black diagonal lines represent the decrease in ecological values and increase in realized peat-mining potential if they were in direct proportion to the area allocated to peat-mining (i.e. non prioritized scenario). The thin dashed vertical line represents the proportion of total area finally directed for the peat-mining.

3.1.2 Costs of and reasons behind research-implementation gap

Different zoning suggestions differed in the sites they suggested for peat-mining (I). The scientists' suggestion differed from the planners' suggestion in 102 cases, and from the final zoning in 107 cases. Furthermore, the planners' suggestion and final zoning suggestion differed in 71 cases. Most of the differences consider peatland sites with moderate ranks in the prioritization analysis (i.e. not the most or least valuable), however also some high and low rank peatland areas differed between the suggestions and the final zoning plan.

The set of peatlands suggested for peat-mining by the planning authorities contained a level of ecological values quite close to the set suggested by the scientists (I, II). The small differences in the remaining value of most of the ecological features incorporated into the analyses were consistent in that the scientists' solution maintained higher proportion of the biodiversity feature distributions in all but one feature, threatened species. Furthermore, as was expected the difference was larger when comparison was between the scientists' solution and the actual suggested zoning plan where more trade-offs were considered. When measured as a relative loss of biodiversity, the scientists' suggestion would have achieved 38% (11 percentage points) smaller losses of the observed ecosystem and species distributions (see II for details). In the scientists' suggestion the smaller ecological losses resulted in a trade-off of

4% (2 percentage points) less area suitable for peat-mining (II). However, the scientists' suggestion also included 17% (8 percentage points) less total area allocated into peat-mining use. The 4% less peat-mining area with 17% less total area used results from considering the site specific peat-mining potential (I, II), i.e. suggesting fewer but resource richer areas into peat-mining. In addition to mitigating ecological losses, this means higher economic cost-efficiency in a form of fewer production sites needed for the targeted amount of peat, resulting in the needed economic investments to be lower.

I found four plausible reasons explaining the differences between the suggestions. First, there is a clear difference in prioritizing observed locations of threatened species versus other biodiversity features. Second, there is a difference in considering peat-mining potential: the areas that the planners suggested for mining against the scientists' suggestion have an average peat-mining potential of 33%, while the areas the scientists suggested for peat-mining, but were not allocated for it, have an average peat-mining potential of 48%. Third, the usage of additional environmental data created further trade-offs in the final zoning plan, which shows as a decreased average retention of the ecological features. Fourth reason is the perspective difference recognized and agreed by all the authors (II): complementary based analyses aim to maximize performance of the whole set of chosen areas, whereas an expert opinion often emphasis importance of single sites. This shows as complementary based analyses achieving higher average retention of ecological features compared to emphasizing sites that have high ecological value when viewed individually.

The two different replacement-cost analyses, one for the increased allocation of area suitable for peat-mining (as mandated by stake-holders) and the other for forced retention of sites with potential surface water effects, shows that such forcing has an expected price in terms of reduced ecological values. While the reduction is only a moderate three percentage points on average, the representation of endangered and rare species declines nine percentage points, thereby revealing a significant trade-off.

Detailed description of the actual zoning project (II, sketched in Fig. 5) revealed that it, at least in this case, was a complex process with the zoning suggestion evolving and confirmed in five stages by different administrative institutions. It was commented in between twice by the stakeholder group and three times by landowners and anyone it considers. Still after the fourth stage, confirmed by the ministry of environment, it is open for complaints that will be dealt in the Supreme Administrative Court, whose decision will ultimately finalize the process.

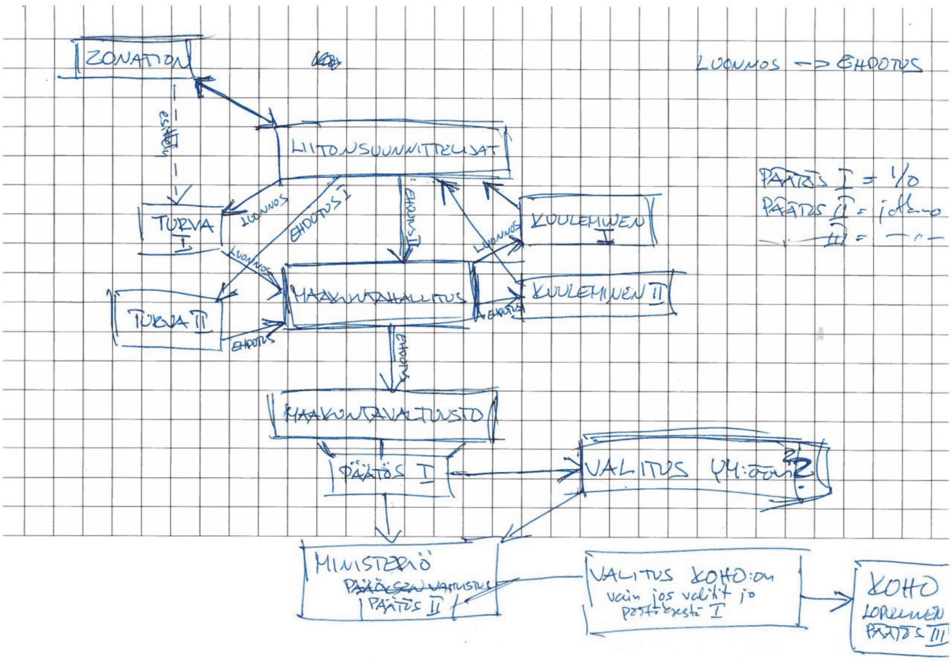

FIGURE 5 Scientist's perspective to the "ten+ step" zoning process. This is illustrated here as I first discussed and sketched it with the planning authorities. The authentic not reduced form demonstrates the planning and decision-making vortex that also scientists may have to face and survive to be able to effectively implement their results. See chapter II for polished version of the figure.

3.2 Returning ecological value of degraded peatland ecosystems

3.2.1 Measuring changes on oligotrophic sphagnum peatlands

The ecosystem structure measured as plant community composition was significantly different between the treatments. Most substantial differences were between pristine sites and all the other sites, i.e. drained, 5 years ago restored, and 10 years ago restored sites. More specifically: there was no significant difference in the community composition between drained and 5 years ago restored sites but the sites restored 10 years ago already showed significant dissimilarity to the drained sites. The effect of the treatment was perhaps most apparent in the visual inspection (III).

According to the peat age-depth models peatlands in all treatments accumulated some peat, but the recent net growth rate of the surface peat was significantly lower at the drained sites compared to pristine and restored sites (III), and the restored sites accumulated peat at a rate similar to that of the pristine sites. In addition, I found that the drainage had resulted in a substantial loss of C from the surface peat. The average per annum C loss estimates for the surface peat (for the 40 years' drainage period) were 129.3 (SE 58.5) (ΔC_{ASH}) and 167.8 (SE 72.7) (ΔC_{AL}) grams of C per square meter per year (III). It should be noted that this is not absolute emission of C to the atmosphere, but relative loss when compared to pristine controls, i.e. what would have been accumulated if not drained.

The rate of C accumulation into surface peat of restored sites was roughly linear after restoration (III). The annual post-restoration surface peat C accumulation (116.3 g m^{-2} yr^{-1}, SE 12.7) was still on average smaller than C accumulation in the pristine sites' surface peat (178.2 g m^{-2} yr^{-1}, SE 13.3). However the variation among the restored sites was large, some of them having even higher accumulation rates than the pristine average (III).

3.2.2 How to implement global restoration targets

The rationale for restoration target setting clearly shows the caveats of alternative ways, in which the set targets can be viewed (IV). Considering hectares alone, a 15% effort can result in significantly poor solution compared to a more holistic view of reversing ecosystem degradation by 15% (Fig. 6). When investigating the more holistic view it seems apparent that the 15% target is unrealistically challenging and probably impossible to ever be met. First, we need to realize that from an ecological perspective, ecosystem degradation has a minimum of two dimensions: the extent of area that has become degraded and the magnitude of the degradation at any given location. Thus, knowledge of the extent of the degraded area alone is not sufficient for providing a scientifically valid estimate of the magnitude of ecosystem degradation: it makes a great deal of difference whether an ecosystem has been only slightly degraded or almost completely lost (options B and C in Fig. 6). As third alternative, one could aim to restore for example 1/3 of the ecosystem condition in as much as 45% of the degraded landscape, resulting in a net effect of reversing 15% of ecosystem degradation (Fig. 6, option D). Option A (Fig. 6) is of course a worst case scenario, where cheapest restoration actions are implemented on 15% area, with perhaps some initial effect, but insufficient to achieve any significant recovery.

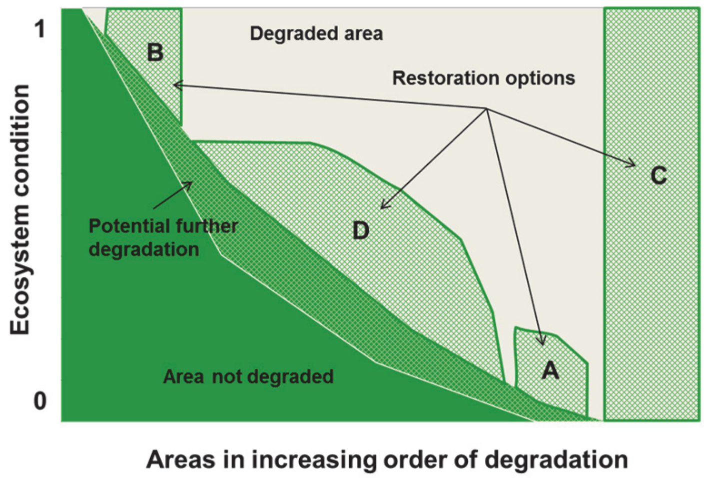

FIGURE 6 Options of the 15% restoration. The x axis describes the total area occupied by different restoration options in the landscape (A-D) and the area covered by each of the options represents their ecological impact. Options A, B and C occupy same total area of the landscape (15%) while options C and D have the same ecological impact. Here, the aggregate effect of (C) or (D) are close to 15% net effect on improvement of degraded ecosystem's condition whereas (A) or (B) achieve much less than 15% net effect on ecosystem condition, even when they imply restoration measures on 15% of the area.

When the effects of degradation and restoration are identified ecosystem specifically, a plan for large (national) scale ecological intervention planning can be devised. Following the logic in figure 6, I introduce an operational model to implement the suggested restoration rationale. The operational model, or a ten-step restoration prioritization procedure (IV) can be divided into four phases: steps 1 and 2 first define the focal ecosystems and identify components that have degraded, then, steps 3-5 determine the current state of the ecosystems, steps 6-8 determine the effects and cost-effectiveness of restoration measures, and the final two steps launch into political decision-making and actual implementation of the suggested actions. The included template (IV) presents instructions and an authentic (low-dimensional) worked-out example from Finland. It demonstrates how the procedure is currently successfully applied with boreal peatland ecosystems by a multi-stakeholder working group prioritizing restoration in an attempt to meet the 15% restoration target across the whole spectrum of ecosystems. Use of this procedure also allows for the monitoring of ecosystems' state and quantification of restoration success.

4 DISCUSSION

4.1 Avoiding biological diversity loss outside protected areas

4.1.1 Biodiversity to peat-mining trade-off in Central-Finland

The developed inverse application of spatial conservation prioritization proved to be successful in balancing economic requirements while minimizing ecological loss aggregated over multiple biodiversity features. My results show how over 80% of the most valuable biodiversity features of the prioritized peatlands could be saved without significantly compromising peat-mining interests. In fact, due to considering the spatial information on peat-mining potential, the analysis found a solution with also higher peat-mining efficiency compared to randomly allocating a set of the pre-selected areas to peat-mining. Although the greedy-selection scenario (i.e. prioritized according to peat-mining potential alone), achieved the same area suitable for peat-mining with minimal total area requirements, the inverse spatial conservation prioritization resulted in significantly higher ecological savings. The high variation in remaining biodiversity feature distributions in the greedy-selection scenario demonstrates the importance of complementarity based prioritization analyses and using as comprehensive data as possible (Wilson *et al.* 2009). If many biodiversity features are represented in the analysis by a coarse surrogate (like area) it is impossible to control the retention of most vulnerable features.

Intuitively, avoidance of unnecessary ecological losses should be the primary guideline for all actions aiming to counter the negative environmental effects of different development projects (Cuperus *et al.* 2001, ten Kate *et al.* 2004). However, finding best practices for avoiding negative effects and properly applying spatial conservation methods is still relatively rare in the unprotected areas without apparently high ecological value (e.g., Maiorano *et al.* 2008, Mathur and Sitha 2008, Chazdon *et al.* 2009). However, even partially degraded parts of the landscape can still substantially supplement conservation-area networks (e.g. Fischer and Lindenmayer 2002, Bengtsson *et al.* 2003, Laita *et al.* 2010). Global adoption of the inverse spatial conservation prioritization principle could result in avoiding a multitude of individually small but often unnecessary ecological losses accumulating into huge conservation gains at regional and global scales. In addition, holistic approaches are easier to justify: considering also the economic values in the case study of chapter I resulted in very positive feedback from the various stakeholders, also from the peat-mining industry.

As discussed in chapter I, this approach differs from several previous works dealing with resource allocation outside protected areas. Perhaps most importantly the presented method does not require *a priori* setting of targets. The practical and conceptual benefits of operating without targets have been demonstrated elsewhere (Wilhere 2007, Moilanen and Arponen 2011, Di Minin and Moilanen 2012, Laitila and Moilanen 2012). In the context of this thesis, the main benefit is that the target-free analysis structure allows the quantitative evaluation of the trade-offs between allocating the parts of the landscape to production and nonproduction areas. This is in the reverse order of the most common practices of systematic conservation planning (Margules and Pressey 2000, Cawardine *et al.* 2009), however, it is very useful for the fine adjustment of land-use plans and it is, in my opinion, in key role in presenting the results to the audience and finally in the process of intelligently deciding where to draw the line (Fig. 4, I, Wilhere 2007, Ferrier and Wintle 2009, Nelson *et al.* 2009).

Systematic conservation planning, including all aspects of ecological interventions should be flexible enough to maintain its effect when implemented and facing the challenges of the real life (Margules and Pressey 2000, Cabeza and Moilanen 2006, Sarkar *et al.* 2006, Knight *et al.* 2008, Arponen *et al.* 2010). However, to really make the critical difference, flexibility requirement should apply not only to conservation but to all forms of land-use planning. Fortunately, prioritization outside protected areas does not have to be an either/or battle between ecological values and economic returns (I, Polasky *et al.* 2005, Polasky *et al.* 2008, Perhans *et al.* 2011). Indeed, my results suggest that more holistic land-use planning reduces the need for flexibility while increasing possibilities for it. Inverse spatial conservation prioritization can be successful in avoiding a zero sum game in resource allocation by identifying areas that are profitable for development and least valuable for conservation. Hopefully the approaches demonstrated here and elsewhere (e.g. Polasky *et al.* 2008, Wilson *et al.* 2010) will lead the way, and development and adoption of holistic frameworks in resource allocation will prevail in the future.

4.1.2 Costs of and reasons behind research-implementation gap

Flow chart of the decision process (Fig. 5, II) of peat-mining allocation in chapter I demonstrates that gaps in delivering knowledge are likely to occur in several stages and forms. Perhaps the clearest indication of a research-implementation gap in our case study was still the difference between our scientifically-based suggestion and the preliminary zoning plan by the planning authorities, where the peat-mining potential seemed to be partially overlooked and focus somewhat shifted to the perceived ecological values of individual peatlands. This illustrates one of the points made by Game et al. (2013), namely, that it is often difficult to see conservation prioritization as a part of the general resource allocation problem. Furthermore, when local considerations and multiple stakeholders' interests were included into decisions, the purely ecological representativeness of the whole solution expectedly suffered from what it could have been (see also Ahlroth and Kotiaho 2009). In other words, complementarity and cost-efficiency ended up with reduced emphasis and the overall solution degraded in the perspective of biodiversity conservation.

In this study I had a chance to discuss the suggestions with the planning authorities and I was able to identify some reasons for the observed differences between the suggestions. There was a clear difference between the solutions in prioritizing observed locations of threatened species versus other biodiversity features (II). One reason for the relatively high priority of threatened species in the planners' suggestion and in the final zoning is that, at least in Finland, many of the threatened species have higher legitimating power than other biodiversity features. This is of course a valid consideration in decision-making. However, endangered species occurrence data may suffer from biased detection and it is deficient in preserving overall biodiversity and functional ecosystems. The suggestions also differed in how they consider the varying peat-mining potential of individual areas. Considering economic returns, like in the scientists' suggestion, would have made it possible to meet economic objectives with smaller total area sacrificed, thereby leading to higher proportion of ecological values retained (I, Margules and Pressey 2000, Naidoo *et al.* 2006). In addition, considering economic efficiency is a benefit on its own, not least because it provides a concrete demonstration of how decision support tools can find profitable trade-offs for the society. Finally, in addition to local level politics among various stakeholders impossible to trace here, the final zoning suggestion included additional environmental data, like surface water effects of peat-mining (II). Additional features create further trade-offs and thus result in a decreased average retention of originally prioritized ecological features (I) in the final zoning, showing in our results (II). I do not wish to say that these additional considerations are less important, but their inclusion to the original prioritization model (I) would have allowed a more holistic trade-off evaluation and intuitively a more efficient and more complementary outcome.

It is evident, that in many cases the experts (e.g. in this case the co-authoring planners) focus on the qualities of single areas instead of considering the whole prioritized set, i.e. the qualities of the whole group of areas suggested here for peat-mining or not to be mined. This is a very closely related issue to the lack of seeing conservation prioritization as allocation of resources (Game *et al.* 2013). It has, however, at least one very apparent reason: the planners, and in this case middle way authorities, often need to justify their suggestion at a site level. Although the final authorities may see the systematic planning solution (I) as a valid argument for the individual sites, it is not this obvious for landowners or in the eyes of law, if the landowners choose to challenge the legitimacy of a single site decision. At least in land-use planning in Finland, this seems to result in giving more weight to legitimating ecological features within single sites in the prioritization process.

The fact that some relevant political and environmental factors (II) were introduced only after the prioritization analysis had been completed underlines the common failure to consider all relevant factors as part of the same resource allocation (Game et al. 2013) or at least already in the beginning of the process. On the other hand, this failure can also be attributed to scientists, and in this case to me, due to my shortcoming in identifying all relevant decision criteria with the stakeholders and decision-makers. All in all, this reflects the complexity of real life resource allocation processes, with all possible things to be considered in a limited time budget. Indeed, the "ten+ step"planning-decision-making process (Fig. 5, II) related to my case study, was significantly more complex than what the process is often reduced to in the scientific literature (e.g. Game *et al.* 2013), perhaps indicating why the metaphor is bridging the gap instead of closing it. This may be one reason why finding best practices is difficult. The decision-making protocols are different from case to case and quite small things can alter the outcome. The tight schedules or starting the prioritization analysis process when the actual zoning project is already on the way, may force the scientists to jump into a moving train (like in chapter I). In the analysis level there are many routes to suboptimal solution (Moilanen 2008) and this applies also to implementation. We need to identify how to best implement, if the optimal implementation pathway (Knight et al. 2011) is already too late to take or too complex to handle within the time limits, which is probably often the case.

In an ideal situation, better communication in different stages of the process would have improved both the analysis and its uptake, reducing ecological impacts from what they will now be. Such communication and proper identification of objectives are part of the standard process of systematic conservation planning (Margules and Pressey 2000, Pressey and Bottrill 2008) and they are key concepts in operational models developed more precisely to implementation of results of conservation science (Knight *et al.* 2006, 2008, 2011). As a relevant consideration, I highlight the following: spatial land-use plans, such as the one in chapter I, have to be developed under constraints on time, money, and effort. For this reason, good availability of spatial data on ecological and economic features is of primary importance for the timely delivery of high-quality policy-relevant spatial prioritizations. If data is available, it is possible to concentrate on developing the utility of the analysis itself. If data is not available in the beginning, much time may be used in the collation of data and even lost if some options fail. Moreover, it is difficult to discuss and develop analysis structure with stakeholders when there is uncertainty about availability of information. In other words, early availability of data also facilitates easier communication through the process. This allows for example multiple iterations of the analysis and customized comparisons revealing the ecological and economic costs of compromises for the stakeholders and decision-makers thereby facilitating improved relevance of the final outcome. Now that methods exist for broad-scale, high-resolution, multi-feature spatial prioritization, long term investment into the development and maintenance of such data becomes more relevant than ever.

4.2 Returning ecological value of degraded peatland ecosystems

4.2.1 Measuring changes on oligotrophic sphagnum peatlands

My results show significant, however, incomplete recovery of both structure and function of boreal *Sphagnum* peatland ecosystem as a result of restoration actions. Stopping ecosystem level degradation and even reversing the trajectory to start recovery of the system through ecological restoration are the main goals in ecological interventions such as here (III, IV). The goal has two phases for a reason: many ecosystems, degraded or pristine, that appear static may in reality be slowly degrading further (Willis and Bhagwat 2009, Laurance *et al.* 2012) and ignoring this easily creates false impression on the interventions' success (Maron *et al.* 2012) on global scale. Indeed, the results on the change in peatland ecosystem structure (plant community composition) demonstrate that the implemented restoration actions were successful in stopping possible further degradation. However, the results also show that the recovery so far was only partial and it is likely to take significantly longer than the 5 to 10 years post-restoration period studied here. Our findings of partial recovery of plant community composition after peatland restoration are in line with earlier case-studies (Haapalehto *et al.* 2011, Hedberg *et al.* 2012). Although the actions used here were successful in reversing the trajectory, it may well need a lot more to fully regain the original community composition and ecosystem structure (Haapalehto 2014).

In the perspective of target setting in ecological intervention it is also important to be able to estimate the level of degradation created by a disturbance. The effect of drainage to boreal peatlands C storage function has been studied before. However, the results on effects of drainage to peat C storage are contradictory: both negative and positive C balances are reported in drained boreal peatlands (Minkkinen and Laine 1998, Lohila *et al.* 2011, Simola *et al.* 2012, Ojanen *et al.* 2010, 2012, 2013, Pitkänen *et al.* 2013). Contradictory or even missing results on degradation make it difficult to specify what is expected from the intervention actions, in other words, what can be considered as cost-efficient or a success. My analyses (III) indicate that, when compared to pristine peatlands' average C accumulation rate as a reference, drainage has resulted in substantial net shortfall of carbon accumulation in the surface peat over the long-term drainage period. However, it seems that the straightforward approach of filling the ditches is enough to jump-start the surface peat growth function already within few years after restoration (III). This is an essential first step towards regaining the long-term C storage function of forestry drained peatland ecosystems.

Potential ecosystem level consequences of biodiversity loss are gaining increasing attention (Hector and Bachi 2007, CBD 2010, Cardinale *et al.* 2012, Hooper *et al.* 2012, Reich *et al.* 2012). In ecological restoration the question is most tangible: How much of the original structure or community composition needs to be recovered in order to regain the original ecosystem functions (Bradshaw 1984, Cortina *et al.* 2006)? There is most likely some plasticity in for example the biodiversity-ecosystem functioning relationship, however, the magnitude or possible thresholds in the plasticity are not well known (Cardinale *et al.* 2012, Hooper *et al.* 2012, Naeem *et al.* 2012, Reich *et al.* 2012). In the perspective of restoration planning the magnitude of the plasticity plays an important role, because it directly relates to target setting and the net costs of restoration actions: strong relationship of ecosystem functions and very specific community composition or biodiversity *per se*, means that complicated and costly actions are very likely needed to achieve ecosystem level targets. According to my results the relationship of surface peat accumulation function and the composition of the plant community was considerably plastic as the peat growth was recovered already with minor recovery of the original vegetation (III). However, although the peat accumulation appeared to be seemingly plastic ecosystem function, the partial recovery of vegetation did not result in recovery of C accumulation function in the surface peat layer. As for many other ecosystem functions in wetlands (Moreno-Mateos *et al.* 2012) more comprehensive recovery of ecosystem structure or conditions seems to be needed also for the full recovery of C accumulation. As indicated by the results, the relationship between one ecosystem structural component and two ecosystem functions may differ even between closely linked functions. Further research is needed to fully understand the prerequisites for the recovery of ecosystem multifunctionality (Hector and Bachi 2007, Lucchese *et al.* 2010, Montoya *et al.* 2012, McCarter and Price 2013), not least to provide the society with more realistic expectations (Hilderbrand *et al.* 2005).

In ecological restoration the success may appear quite different with respect to time since restoration, especially when the successful outcome is naturally dynamic. Thresholds in ecosystem state are likely to result in nonlinear recovery of the structure in time (Suding and Hobbs 2009), which can cause quite large differences in the outcome in relatively short post-restoration time span, leaving the interpretation sensitive to recognizing these thresholds (Banks-Leite *et al.* 2014). On the other hand, the recovery rate of the ecosystem structure is intuitively slower towards the end of the recovery process, as some structural features or species are harder to recover than others, being for example more sensitive to the conditions or less efficient dispersers. Furthermore, it is unlikely that methods relying on unaided recolonization of species that are rare (or extinct) in the landscape's species pool will ever recover all the species that have gone locally extinct since the degradation (Haapalehto *et al.* 2011, Hedberg *et al.* 2012). Still, aided species recolonization on e.g. 1000 000 hectares of degraded peatlands that Finland has agreed to restore (Nagoya protocol, CBD 2010) means a way higher price tag than simply blocking the ditches and waiting. In the end, weighing the success brings us back to the "how-much-is-enough myth" (Wilhere 2008) and is ultimately beyond scientific evaluation: many outcomes are successful in their budgetary constraints and with the accepted uncertainty. When judging the successfulness of any restoration project, it is worth remembering that although it may take several decades for the ecosystem structure to recover, it is still usually relatively fast change compared to the original succession leading to the pre-disturbance state (Jones and Schmitz 2009). After all, restoration is a means to accelerate initial recovery dynamics (SER 2004) not make things appear out of nothing.

I also estimated the ecosystem service value for the carbon accumulated to the newly formed post-restoration peat layer. For example, the 15% restoration target (IV, CBD 2010) means approximately 1 million hectares peatland restoration in Finland. Assuming similar mean post-restoration carbon accumulation rates as found here, the total market value for the accumulated C in the post-restoration peat layer for the 15% restoration of degraded peatlands would be as high as 26 to 580 million USD annually, over the first decade after restoration. Considering the possible ecosystem service value (Bullock *et al.* 2011, Menz *et al.* 2013) as a part of a vast restoration target demonstrates well the magnitude of the situation. In this case study the value was estimated only for the surface peat and the whole ecosystem level carbon balance including the effect of the removed tree stand can of course be something quite different (III). Still, presenting these figures is likely to promote their adoption to common ecosystem manipulation evaluation, which then hopefully increases our understanding on the balance of possible benefits and often more apparent immediate costs of e.g. restoration actions (Menz *et al.* 2013).

4.2.2 How to implement global restoration targets

Rational investigation of the 15% restoration target exposes the overambitious nature of the Strategic Plan for Biodiversity 2011-2020 and Aichi Biodiversity Targets (CBD 2010): to obtain an ecologically justified 15% reduction in the degradation of ecosystems we need to apply heavy restoration measures across very large areas in extremely short time, simultaneously compensating for ongoing degradation elsewhere. In addition, it seems logical that the 15% restoration target (CBD 2010) should not be about area alone (unlike in e.g. Egoh *et al.* 2014). Alternatively, the 15% can be defined as an aggregate 15% return of all the degraded ecosystems' condition, i.e. restoring 15% of the degraded ecological features on the earth. In addition, we need to remember that the ongoing further ecosystem degradation should be included to the evaluation of achieving the target, with a negative effect: even when restoration measures are successfully implemented on a globally significant area, the net effect can be negative if ecosystems continue to degrade faster elsewhere (Laurance *et al.* 2012, Maron *et al.* 2012). It should be noted that despite a significant effort, also the European Commission commissioned protocol, the four-level model (Lammerant *et al.* 2013, four levels for ecosystem state), is only moderately better than a totally opportunistic decision. According to the four-level model the 15% restoration target is achieved by elevating ecosystems from one level to a higher one on at least 15% of degraded ecosystems' area. However, I note that only by elevating 15% of the completely degraded ecosystems to the highest level (if the restoration actually was total, which is highly unlikely) would achieve something like a 15% net effect (Fig. 6, option C) and any shortcoming from the absolute target state gives a smaller than 15% net effect on increasing ecosystems' condition.

Cost-effective and ecologically sound implementation of the 15% target demands open-minded consideration of different approaches and their effects and costs as well as unprejudiced evaluation of the outcome with respect to novelty of the resulting ecosystems (Hobbs *et al.* 2009). Calling for novel solutions is not a novel idea. In fact already over a decade now it has been acknowledged that novelty, in a form of unconventional ecosystems, may well prove to be a very rational way to approach such demanding goals (Seadstedt *et al.* 2008, Hobbs *et al.* 2009), when used with caution and reason and not a means to simply save in costs by lowering the bar (Arponen *et al.* 2010, Perring *et al.* 2014). This means that knowledge on ecosystem thresholds and resilience (Suding and Hobbs 2009, Lake 2012) becomes more necessary than ever to find cost-effective restoration solutions and to avoid the role of Don Quixote, with respect to both ecosystems and society. This also applies in off-setting policies that can easily be a slippery slope of setting the mind at ease with a zero-sum solution (Maron *et al.* 2012). Identifying the most relevant resistance thresholds in ecosystems' structure will be in key role in stopping further degradation (Suding and Hobbs 2009, Lake 2012, Banks-Leite *et al.* 2014) just as finding meaningful spatial trade-offs between safeguarding ecological values while producing local livelihood (I, Polasky 2008, Wilson *et al.* 2010).

I stress the importance of science based argumentation for restoration target setting and the need for reference ecosystem states if the degradation to be reversed and success of restoration are hoped to be quantified (IV). It is essential to notice that, although a pristine state of ecosystems is advocated for the reference state to quantitatively measure the amount of degradation, this does not mean that the reference state should also be the targeted state. First, allowing some degree of novelty in the targeted set of ecosystems may well prove to offer a much better overall ecological return for the investment. Second, also the historical and present role of human as part of the ecosystem is a relevant consideration (Higgs 2005, Clewell and Aronson 2006, Mace 2014), making either the definition of pristine complicated or ethically challenges the aim to achieve ecosystems without man in many parts of the Earth. Systematic and carefully argued reference points and ecosystem specific overall goals are needed however, to avoid the notorious harmful opportunism (Pressey and Bottrill 2008). This is because estimating the degree of degradation and then concentrating the 15 % areal restoration effort on the least degraded areas may be economically tempting but most likely inadequate in its total effect. The temptation of politically opportunistic choice of areas is both ecologically and economically deceptive: it can result in very small net benefits (Noss *et al.* 2012) as effective ecosystem or species recovery thresholds are very hard to identify but still having enormous cumulative costs operating with vast total area. This is hardly in line with the vast need for truly cost-effective and complementary solution for maintenance of biodiversity and ecosystem services.

Intuitively I find it likely that key solution considering total area improved, cost-effectiveness, and fully restored area is something between option C and D in figure 5. This is also likely to promote complementarity with respect to biodiversity and ecosystem functioning: to find best trade-off between net effects and minimal uncertainty intuitively requires a combination of “cheap novelty” and “costly naturalness”.

5 CONCLUSIONS – GLASS HALF FULL?

Although many problems have been scientifically described and methods for high-end analyses are available, it still seems that, at a global scale, the allocation of resources is most often a zero-sum game: more means more and no major short-cuts are offered. Thus, the principle aims of my PhD work were to evaluate the efficiency of modern conservation and restoration approaches in alleviating the challenges of resource allocation in a finite world, and to bring an ecological reality into the target setting.

From the perspectives of ecosystems' the results are promising. Inverse spatial conservation prioritization has the potential to successfully reduce ecological losses in land-use (I). Also, identifying the general reasons that can create a gap between systematic conservation plans and their implementation may help us closer to taking full advantage of these modern conservation methods (II). Furthermore, some important functions of ecosystems can be re-started and even fully restored already in the early stages of the post-restoration succession trajectory (III), demonstrating the huge, though still partly uncertain, potential of many intervention approaches (Hobbs and Cramer 2008). Chapter IV is a natural finale for this thesis: it combines a systematic conservation planning approach and the detailed ecology of the dynamics of disturbed ecosystems, demonstrating the necessity of both realms. It also demonstrates the pitfalls of combining politics with conservation biology, and the danger of building loose, unstable bridges over the apparent policy-implementation gap (IV, Tear *et al.* 2005). The inaccuracy arising from emphasising appearance over reality (Svancara *et al.* 2005, Maron *et al.* 2012) can easily result in a significant and unnecessary loss of cost-efficiency. Notably, common to all these studies is that they present methods that contribute to traditional conservation actions of expanding the total protected area (e.g. Kremen *et al.* 2008, Lehtomäki *et al.* 2009), and all of them take into consideration the cost-effectiveness of conserving biodiversity. As such, they all aim to tackle the challenge of allocating finite resources, most importantly area.

Still, looking at any of the results provided here one can wonder if the glass in the end is half full or half empty. Although the results look promising from my perspective, many could interpret them as insufficient or too slow. Or, one can think that they give a false impression of success when looking at restored or protected ecological values as a set instead of focusing on single site evaluations. This is of course a very tangible question in conservation in general, and perhaps especially relevant when interpreting the outcomes of ecological restoration (Bullock *et al.* 2011, Moreno-Mateos *et al.* 2012) and the success of the actions (e.g. Ruiz-Jaen and Aide 2005, Wortley *et al.* 2013). For example, the interpretation of a recent meta-analysis implied that a delay in the recovery of the structure of wetland ecosystems caused the restoration effort to fail in reaching the targets set for the functions of the ecosystems (Moreno-Mateos *et al.* 2012). Especially with restoration, we should be extra careful when determining a project as a success or a failure. Change always takes time (Moreno-Mateos *et al.* 2012), multiple thresholds can exist in the succession trajectory (Suding and Hobbs 2009) and the succession trajectory itself may not be self-evident (Suding *et al.* 2004). As a results it is difficult (if not impossible) to evaluate, when is the correct time to conclude what the restoration actions have achieved.

In a spatial conservation prioritization of multiple sites, the evaluation of the results may be even harder. Measuring success in achieved targets means, in practise, that setting loose targets increases the likelihood of success. Evaluating the analysis based on, for example, the performance of ecological features' (e.g. an increase in the proportions of the distributions protected) in relation to the total allocated area (I) is one way. However the performance measures are always limited in their comprehensiveness, just like the data and the ecological model of the analysis (Wilson *et al.* 2009). Although the relative representativeness of key ecological features is quantitative, and as such, a seemingly objective way of comparing the sets of areas of different suggestions, it can never fully concern the solutions ecological value *per se*. In addition, although the connectivity considerations applied, for example in focal prioritization (I), aim to respond to the sustainability of the solution at the landscape level, it is still unclear how well the ecological features of the suggested set of peatlands will stand the test of time. Finally, as we see, for example from the differences in suggestions for land-use (II), there are many differing arguments against and for the protection of an area, and it is thus not an easy task to claim success from one perspective over another.

The results of the inverse spatial conservation prioritization also provided a useful look into the protected - not protected dichotomy. An anonymous referee commented on my manuscript (I) stating that the case in question did not differ from any other form of protection, but was just a case of the gradual expansion of protected areas. Indeed, further actions can alter the studied landscape in two ways, by increasing either protection or economic land-use. However, if the current protected area network is expanded in the future, the inverse spatial conservation prioritization saves, and thus offers for future protection, the ecologically valuable areas that otherwise might be sacrificed during economic development, before the expected next expansion of conservation. This is first of all a clear difference in the stepwise increase of the protected area, but most importantly it alleviates the "how much is enough" problem by finding a more optimal (or less random) trade-off between conservation and economy. In case of further or different economic land-use the already applied ecological loss avoidance at least buys time: Before land-use is expanded the previously saved areas of higher ecological value can act as sources for species and thus increase ecological connectivity in the landscape, and remotely promote, for example, the success of restoring protected but partly degraded areas.

Explaining the results to the general public is also of increasing importance. The general success of restoration ecology, for example, is often challenged in the scientific literature (Hilderbrand *et al.* 2005, Cabin 2007, Miller and Hobbs 2007, Hobbs *et al.* 2011). However, it still seems that as policy makers, humans have at least a slightly distorted image of reality and of good decisions (e.g. Gilbert 2011). This can lead to an overly optimistic impression of our ability to, for example, re-create lost ecosystems or maintain all of the ecological features of the landscape, despite extensive land-use (Hilderbrand *et al.* 2005, Tear *et al.* 2005, Hobbs *et al.* 2011, Maron *et al.* 2012, Game *et al.* 2013).

Defining the research-implementation gap is not an easy task. First, already measuring the effects of conservation planning and a conservation effort is complicated. Second, the planning process has multiple links that can be confusing to define making the interpretation of the place of the gap difficult. In other words, the way the effects are measured and the way the whole process is defined must be clear, before actual gaps and inevitable (but initially missed) trade-offs can be distinguished from each other. Being aware of the gaps can help in identifying them already during the many phases of the planning process (Knight 2006, Knight *et al.* 2008). One aim of this thesis is to identify reasons and costs of the gaps, but most of all, to demonstrate how the differences between solutions can be quantified and how this can be used to demonstrate trade-offs and hopefully to avoid unnecessary gaps. This, however, requires proper adoption of the latest methods. Although high-end scientific considerations of spatial planning are rarely fully implemented, seeing spatial prioritization as "one tool" has been criticized by, for example, Game *et al.* (2013). This perspective can easily guide the decision-making away from a comprehensive resource allocation approach (Game *et al.* 2013). Nearly all ecological and societal considerations should be transferable into a prioritization model (Ferrier and Wintle 2009, Wilson *et al.* 2009). However, the "one of many tools" ideology can prevent quite achievable holistic goals if the plans are overruled by more conservative approaches from times when comprehensive spatial planning was technically impossible. Combining economic and ecological considerations in land-use has its risks (Arponen *et al.* 2010). However, it can in many cases be a very natural combination, which enables a more realistic evaluation of trade-offs, than does looking at ecological data alone (I, Wilson *et al.* 2010, Polasky *et al.* 2008, Polasky *et al.* 2014).

So, is the glass half full or half empty? Promising results from restoration and conservation, and especially from their combination, show that we are, in fact, standing on the edge of fate. However, we have already for some time now witnessed nature's decline despite the at least apparent increase in efforts to conserve and restore, as well as in a development in methodology. Although the question of "how much is enough?" is ultimately a philosophical one with many dimensions, it is quite certain that the glass should not appear to be full - not from any perspective. Now that scientific methods that can point out near optimal solutions have been developed, it is the dialogue between different parties in the scientists-planners-stakeholders-decision-makers vortex that determines what can be achieved at the end of the day. Chapters I and II demonstrate the relatively slight, yet clear gap between good solutions and good outcomes. Certainly, missing the opportunities of the modern high-end approaches because of an imperfect dialogue deserves to be called an epic failure instead of a problem, now more than ever. Finally, the aims of this thesis described in the introduction seem to be very science-oriented (which I have realized now, while writing this), which may well be seen as a consequence of and a reason for the research-implementation gap (Ehrenfeld 2000). The real aim of this thesis is, of course, to produce knowhow that is relevant for the holistic and cost-efficient protection of ecological features and that can be globally implemented in actual conservation work.

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YHTEENVETO (RÉSUMÉ IN FINNISH)

Ekosysteemien turvaaminen – kun suojelualueet eivät enää riitä

Ihmispopulaation valtavasta kasvusta seuraavat maankäyttöpaineet aiheuttavat merkittävän uhan ekosysteemien toiminnalle ja luonnon monimuotoisuuden säilymiselle Suomessa ja maailmanlaajuisesti. Huolimatta ponnisteluista luonnonsuojelullisten tavoitteiden täyttymiseksi näyttää siltä, että lajien sukupuutot tulevat edelleen yleistymään ja ekosysteemien heikentyminen on arkipäivää, jopa suojelualueilla. Olemassa olevien suojelualueiden riittämättömyys, muun muassa sukupuuttojen torjumisessa, on nostettu esille useissa yhteyksissä sekä tieteellisessä kirjallisuudessa että poliittisessa päätöksenteossa. Perinteisten suojelumenetelmien toimivuus yksinään onkin usein kyseenalaistettu ja tarve suojelun määrän ja laadun kustannustehokkaalle lisäämiselle haastaa tutkijoiden, suunnittelijoiden ja päätöksentekijöiden taidot kokonaisvaltaisemman, suojeluarvot huomioivan, maankäytön arkkitehteina. Toisaalta huolta herättää kykymme palauttaa tai korjata heikentyneiden ekosysteemien toiminta ja luontoarvot erityisesti nyt, kun heikentyneiden ekosysteemien ennallistamiselle on asetettu valtavat tavoitteet. Vuonna 2010, Convention on Biological Diversity –tapaamisessa, noin 200 valtiota allekirjoittivat sopimuksen sitoutuen ennallistamaan 15 % heikentyneistä ekosysteemeistä vuoteen 2020 mennessä. On kuitenkin epäselvää onko 15 % ennallistamistavoitteen saavuttaminen ylipäätään mahdollista ja riittääkö toisaalta sekään yhdessä suojelualueiden kanssa täyttämään samassa tapaamisessa asetetut tavoitteet ekosysteemien heikentymisen ja biodiversiteetin häviämisen lopettamiselle.

Väitöskirjassani tarkastelen ekologisten päätöksentekoanalyysien tehokkuutta luonnonsuojelun työkaluina. Tarkastelen erityisesti miten luonnonsuojelun priorisointimenetelmiä voidaan hyödyntää päätöksenteon apuvälineinä maankäytössä suojelualueiden ulkopuolella luoden täydentäviä vaihtoja rajallisten, suorien, suojelukeinojen tueksi. Tutkin toisaalta, kuinka paljon maankäytön negatiivisia ekologisia vaikutuksia voidaan lieventää ja toisaalta, kuinka kehitettyjen menetelmien käyttöönottoa voidaan tehostaa. Tämän lisäksi tutkin kuinka ekologisella ennallistamisella voidaan palauttaa heikentyneiden ekosysteemien rakennetta ja toimintaa. Tarkastelen ensin esimerkkinä ojituksen ja ennallistamisen vaikutuksia sararämeiden kasvillisuuteen suoekosysteemin rakenteena sekä turpeen kertymiseen ja hiilensidontaan ekosysteemin toimintoina. Lisäksi tarkastelen ennallistamiselle asetettujen maailmanlaajuisten tavoitteiden tulkintaa ja ennallistamisen laajamittaisen toteuttamisen haasteita. Väitöskirjassani maankäytön priorisointimenetelmiä ja ennallistamismahdollisuuksien tarkastelua yhdistää laadullisen tarkastelun ja kustannustehokkuuden näkökulma: ne eivät suojelukeinoina varsinaisesti vaadi lisäpinta-alan ohjaamista suojeluun, ja siksi niiden soveltaminen perinteisten suojelukeinojen lisänä tarjoaa mahdollisuuden lieventää ekosysteemien rakenteen ja toiminnan säilyttämiseen liittyviä kustannuksia ja niistä seuraavia haasteita.

Tulokseni osoittavat kuinka ekologisten arvojen huomioimisella maankäytön suunnittelussa voidaan tehokkaasti vähentää lajien ja elinympäristötyyppien harvinaistumista maisematasolla. Pieniä, mutta usein vältettävissä olevia, negatiivisia ekologisia vaikutuksia vähentämällä voidaan aikaansaada suuria kokonaissäästöjä, joiden merkitystä maisematasolla ei pidä väheksyä. Toisaalta, huomioimalla alueiden taloudellinen arvo, tässä tapauksessa turvetuotantopotentiaali, laatimani analyysin lopputulos oli myös taloudellisesti toimiva: suojeluarvon lisäksi myös ratkaisun turvetuotantopotentiaali oli korkeampi kuin jos kustannustehokkuus olisi jätetty huomioitta. Uusien priorisointimenetelmien avulla maankäytön ratkaisujen ekologista kokonaisvaltaisuutta pystytään kontrolloimaan entistä paremmin. Omassa analyysissäni pystyin suunnittelemaan maankäyttöä siten, että sen ulkopuolelle jäävä maisema olisi luontoarvoiltaan mahdollisimman monimuotoinen monella eri monimuotoisuuden tasolla. Käyttämäni menetelmän avulla tällaisen komplementaarisen ratkaisun kustannuksia ja hyötyjä pystytään lisäksi joustavasti tarkastelemaan suhteessa taloudellisen maankäytön laajuuteen. Voimme tarkastella kuinka paljon uhanalaisimpien lajien tai koskemattomimpien luontotyyppien edustavuus maisematasolla heikkenee jos maankäyttöastetta nostetaan, tai toisaalta kuinka paljon maankäytöstä pitäisi tinkiä, jotta uhanalaisen lajiston ja luontotyyppien kannalta arvokkaimmat alueet tai toivottu kokonaismäärä voitaisiin turvata.

Toisessa osatyössäni havaitsin, että näkökulmaerot alueiden ekologisen arvon tulkinnassa ja esimerkiksi päätöksenteon kannalta puutteelliset aineistot osaltaan hankaloittavat tulosten hyödyntämistä käytännössä. Toisaalta päätöksentekoprosessin monimutkaisuus itsessään vaikeuttaa tiedon siirtymistä prosessin eri vaiheissa ja näin vaikeuttaa priorisointianalyysien ja yleisemmin tieteellisen tiedon käyttöä päätöksenteon tukena. Lopputuloksen vertaileva tarkastelu on keskeinen lähestymistapa käytettäessä priorisointianalyysejä päätöksenteon apuvälineenä.

Havaitsin myös, että ekologinen ennallistaminen on tehokas tapa nopeuttaa ekosysteemin rakenteen ja toiminnan palautumista. Tutkimukseni osoitti, kuinka alun perin metsätalouskäyttöön ojitettujen soiden pintaturpeen kasvu saatiin palautumaan luonnontilaiselle tasolle jo ennallistamisprosessin alkuvaiheissa. Toisaalta havaitsin ennallistamisen positiivisten vaikutusten suoekosysteemin kasvillisuuden koostumukseen olevan hitaampia. Samoin ennallistettujen soiden pintaturpeen hiilensidonta ei vielä ollut luonnontilaisella tasolla 10 vuotta ennallistamisen jälkeen osoittaen, että toiset ekosysteemin toiminnot tarvitsevat palautuakseen täydellisemmän olosuhteiden ja rakenteen palautumisen, kun taas joidenkin ekosysteemin toimintojen, kuten tässä turpeen kasvun, suhde ekosysteemin rakenteeseen on löyhempi. Kasvillisuuden palautumista seurattaessa oli olettavissa, että prosessi kestää kauemmin kuin tässä tarkasteltu ennallistamista seuraava kymmenen vuoden aikajakso. Pintaturpeen kasvun nopea palautuminen osoitti kuitenkin, että joidenkin ekosysteemin toimintojen osalta voidaan ennallistamalla saavuttaa erittäin positiivisia tuloksia jo tällä, ekosysteemien luontaiseen kehitykseen verrattuna, hyvin lyhyellä aikavälillä.

Tarkastellessani ennallistamisen vaikutuksia maailmanlaajuisten tavoitteiden näkökulmasta havaitsin, että tavoitteet ovat usein paitsi epärealistisia myös epämääräisiä. Käytännön toiminnan ja vaikutusten kannalta on suuri merkitys esimerkiksi sillä, tarkastellaanko tavoitteita ennallistettavana pinta-alana vai ennallistamisen kokonaisvaikutuksen kautta. Pelkkänä pinta-alana ilmaistun tavoitteen vaikutuksia on hyvin hankala arvioida etukäteen, saman ennallistetun kokonaispinta-alan voidessa johtaa hyvinkin erilaisiin ennallistamisen kokonaisvaikutuksiin ennallistettavien ekosysteemien ennallistamista edeltäneestä tilasta ja toisaalta lopputuloksen onnistumisesta riippuen. Väitöskirjani viimeisessä osatyössä esittelen menetelmän, jonka avulla ennallistamistoimia voidaan tarkastella ja suunnitella ekosysteemien kokonaismuutoksen, heikentymisen ja palautumisen kautta. Menetelmässä huomioidaan ekosysteemikohtaisesti ihmistoiminnasta aiheutuneet rakenteen ja toiminnan muutokset ja eri ennallistamistoimien palauttava vaikutus sekä menetelmien kustannukset.

Yhdessä tulokseni osoittavat, että modernien luonnonsuojelun ja ekologisen ennallistamisen menetelmien avulla luonnonsuojelun kustannustehokkuutta pystytään huomattavasti lisäämään. Pystymme tehokkaasti vähentämään maankäytön negatiivisia ekologisia vaikutuksia suojelualueiden ulkopuolella ja samalla lisäämään suojelualueiden arvoa ekologisen ennallistamisen keinoilla. Jo olemassa olevien suojelualueiden laadun parantaminen ja turhien negatiivisten vaikutusten välttäminen ovat hyvä osoitus toimivista ja kustannustehokkaista menetelmistä, mutta ne helpottavat myös toimenpiteiden riittävyysongelmaa, koska ne eivät suoraan edellytä lisäpinta-alan allokoimista suojeluun. Näin ne toimivat tärkeinä täydentävinä menetelminä perinteiselle suojelulle. Edelleen kuitenkin monet tekijät menetelmien vaikeudesta päätöksentekoprosessin monimutkaisuuteen vaikeuttavat näidenkin menetelmien optimaalista laaja-alaista käyttöä. Vaikuttaakin siltä, että aikamme suurin haaste toimivien ekosysteemien säilyttämisen kannalta on toimivan vuoropuhelun löytyminen tutkijoiden, suunnittelijoiden ja päätöksentekijöiden välille.

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ORIGINAL PAPERS

I

USE OF INVERSE SPATIAL CONSERVATION PRIORITIZATION TO AVOID BIOLOGICAL DIVERSITY LOSS OUTSIDE PROTECTED AREAS

by

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Use of Inverse Spatial Conservation Prioritization to Avoid Biological Diversity Loss Outside Protected Areas

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Abstract: *Globally expanding human land use sets constantly increasing pressure for maintenance of biological diversity and functioning ecosystems. To fight the decline of biological diversity, conservation science has broken ground with methods such as the operational model of systematic conservation planning (SCP), which focuses on design and on-the-ground implementation of conservation areas. The most commonly used method in SCP is reserve selection that focuses on the spatial design of reserve networks and their expansion. We expanded these methods by introducing another form of spatial allocation of conservation effort relevant for land-use zoning at the landscape scale that avoids negative ecological effects of human land use outside protected areas. We call our method inverse spatial conservation prioritization. It can be used to identify areas suitable for economic development while simultaneously limiting total ecological and environmental effects of that development at the landscape level by identifying areas with highest economic but lowest ecological value. Our method is not based on a priori targets, and as such it is applicable to cases where the effects of land use on, for example, individual species or ecosystem types are relatively small and would not lead to violation of regional or national conservation targets. We applied our method to land-use allocation to peat mining. Our method identified a combination of profitable production areas that provides the needed area for peat production while retaining most of the landscape-level ecological value of the ecosystem. The results of this inverse spatial conservation prioritization are being used in land-use zoning in the province of Central Finland.*

Keywords: land-use zoning, land-use planning, site selection, spatial optimization, systematic conservation planning, Zonation software

Resumen: *La expansión global del uso de suelo por humanos establece un incremento constante en la presión para el mantenimiento de la biodiversidad y el funcionamiento de los ecosistemas. Para combatir la declinación de la biodiversidad, la ciencia de la conservación ha innovado métodos como el modelo operativo de planificación sistemática de la conservación (PSC), que se enfoca el diseño e implementación de áreas de conservación. El método usado más comúnmente en PSC es la selección de reservas que se concentra en el diseño espacial de redes de reservas y su expansión. Expandimos estos métodos mediante la introducción de otra forma de asignación espacial del esfuerzo de conservación relevante para la zonificación del uso de suelo a escala de paisaje que evita los efectos ecológicos negativos del uso de suelo por humanos afuera de áreas protegidas. Nuestro método se denomina priorización de conservación espacial inversa. Puede ser utilizado para identificar áreas adecuadas para el desarrollo económico al mismo tiempo que limitan los efectos ecológicos y ambientales de ese desarrollo a nivel de paisaje mediante la identificación de áreas con el mayor valor económico pero el menor valor ecológico. Nuestro método no se basa en objetivos definidos a priori, y como tal es aplicable a casos donde los efectos del uso de suelo sobre, por ejemplo, especies individuales o tipos de ecosistemas son relativamente pequeños y no violar objetivos de conservación regionales o nacionales. Aplicamos nuestro método a la asignación de uso de suelo para la explotación de turba. Nuestro método*

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identificó una combinación de áreas de producción rentables que proporcionan la superficie requerida para la producción de turba y al mismo tiempo retienen la mayor parte del valor ecológico del ecosistema a nivel de paisaje. Los resultados de esta priorización de conservación espacial inversa son utilizados en la zonificación de uso de suelo en la provincia de Finlandia Central.

Palabras Clave: optimización espacial, planificación de uso de suelo, planificación sistemática de la conservación, selección de sitios, software de zonificación, zonificación de uso de suelo

Introduction

The increasing human population sets universally high demands for land use (e.g., Foley et al. 2005; Millennium Ecosystem Assessment 2005; Polasky et al. 2008), and there is a constantly diminishing area remaining for the maintenance of biological diversity and ecosystem services (Chan et al. 2006; Fischer et al. 2007). Unfortunately, science has not, and may never, have an accurate answer to the question of how much area is enough to meet the varying goals for conservation of biological diversity and ecosystem functions (e.g., Svancara et al. 2005; Tear et al. 2005; Rondinini & Chiozza 2010). Despite apparent political will (Convention on Biological Diversity 2010), recent work shows that ecosystems are being degraded at alarming rates both outside and inside protected areas (Laurance et al. 2012).

To combat an overall decline in biological diversity, systematic conservation planning (SCP) was developed. This planning method focuses on both design and on-the-ground implementation of conservation (e.g., Sarkar et al. 2006; Margules & Sarkar 2007; Kukkala & Moilanen 2013). SCP offers tools to optimally design and expand reserves by identifying areas with the highest irreplaceability and complementarity (Sarkar et al. 2006; Pressey et al. 2007) and minimizing losses by considering spatial information on the probability of loss of ecological values in the landscape (e.g., Pressey et al. 2004; Nicholson et al. 2006; Visconti et al. 2011). Two target-based problem formulations, minimum-set and maximum-coverage planning, are integral to the SCP framework (Margules & Pressey 2000; ReVelle et al. 2002; Margules & Sarkar 2007). Minimum-set planning satisfies given ecological targets at minimum cost, and maximum-coverage planning satisfies as many targets as possible within the available budget when resources do not suffice to meet all targets. SCP can alleviate trade-offs between economic gains and ecological losses and improve conservation success because it maximizes the cost-effectiveness of conservation by minimizing area (and thus cost) needed for achieving a priori fixed ecological targets (Possingham et al. 2006).

Although SCP methods provide a systematic protocol for defining goals and targets for conservation (Pressey & Bottrill 2008; Carwardine et al. 2009), there is still uncertainty about whether the protected-area networks will ensure long-term persistence of populations and ecosystem functioning (e.g., Cabeza & Moilanen 2003; Kuussaari et al. 2009; Laurance et al. 2012). In response to this uncertainty, a shift in focus from the design of protected-area networks to more holistic approaches that cover the entire landscape have been suggested (e.g., Maiorano et al. 2008; Mathur & Sitha 2008; Chazdon et al. 2009). Tools have also been developed that integrate conservation planning with the needs of other land uses (e.g., Gordon et al. 2009; Watts et al. 2009; Willis et al. 2012).

One important aspect of conservation outside protected areas is the avoidance of negative environmental and ecological effects of economic development projects (Cuperus et al. 2001; Ten Kate et al. 2004). We investigated the application of spatial prioritization methods to avoid landscape-level negative effects (e.g., decreasing ecosystem area and declining species populations) of development. This application differs from the typical use of spatial prioritization in reserve selection. We devised inverse spatial conservation prioritization, a process that identifies areas for economic development that are the least ecologically valuable.

Because target-based planning is the most typical planning mode in SCP (Sarkar et al. 2006; Margules & Sarkar 2007; Pressey & Bottrill 2008), we first summarize target-based inverse analogues of the minimum-set and maximum-coverage approaches. In the inverse minimum-set problem, each ecological feature is given a maximum loss limit, and the method identifies the set of areas that generate maximal joint income from alternative land uses without violating any feature-level ecological loss limit. In the inverse maximum-coverage problem, the objective is to provide stated economic benefits while violating as few ecological loss limits as possible. However, many planning projects are less than ideally suited for a completely target-based design. Negative effects on the environment caused by any single economic development project can be small and do not necessarily cause any landscape-wide feature-level targets to be violated. A target can indeed be set for economic return or area needed for development, but minimization of ecological losses over multiple features can sometimes be better implemented via an aggregate benefit-based approach rather than via targets (Moilanen & Arponen 2011; Laitila & Moilanen 2012).

We propose that trade-offs between conservation and economic development may be investigated in a more informative and flexible way when they are not restricted to target-based approaches. We illustrate nontarget-based inverse spatial conservation prioritization by applying it to land-use zoning in the province of Central Finland. This zoning includes allocation of areas for peat mining. To satisfy the interests of stakeholders, a flexible preliminary target of 6000–12,000 ha for peat mining was considered. These areas were to be allocated so that ecological losses would be minimized. We used the Zonation approach to implement inverse spatial conservation prioritization. Zonation applies generic methods for how conservation value is aggregated across features, space, and time (Moilanen et al. 2009, 2011). Because our method operates without a priori targets and produces a continuous prioritization across the entire landscape, decision makers can examine the spatial and quantitative trade-offs between biological diversity features and peat mining before deciding how many and which areas will be allocated for peat extraction.

Methods

The focal landscape consisted of 306 partially ditched peatlands (36,503 ha). We excluded conservation areas and other unditched peatland areas from the analysis because they would not be allocated to peat mining in any case. Various stakeholders (peat mining companies, local energy company, nature conservation organizations, local and national environmental administration, and the Finnish Forest and Park Services) predetermined the areas included in the prioritization as possible candidates for peat extraction. However, not all stakeholders would choose the same set of areas for actual mining because the peatlands have variable peat-mining potential and contain different sets of biological diversity features.

Principles of Inverse Spatial Conservation Prioritization

We used the publicly available Zonation method and software (Moilanen et al. 2005, 2011, 2012) to perform land-use prioritization. Zonation identifies areas that are important for retaining environmental quality and landscape connectivity simultaneously for multiple features (species' populations, ecosystem types, etc.) in the landscape and thereby indirectly aims to retain persistence of all these features (e.g., Moilanen et al. 2005; Kremen et al. 2008; Moilanen et al. 2012).

The operational principle behind Zonation is to maximize the retention of range-size normalized occurrence levels of multiple features (species, ecosystems, etc.) according to their feature-specific weights, connectivity, and other considerations while retaining a complementarity-based balance across all features (Moilanen et al. 2011). Zonation starts from the assumption that in terms of ecological persistence it is best to conserve the entire landscape. Grid cells are iteratively removed (proposed to be allocated for economic land use) and aggregate loss of biological diversity features is minimized according to their weights and abundance remaining after each removal. The order of removal of grid cells is recorded to produce a ranking for the prioritization of the grids.

Heuristically, Zonation balances the abundance, quality, and connectivity of ecological features. Zonation can also account for multiple direct or indirect costs of conservation. We defined cost as lost potential for peat mining, which we measured in units of area suitable for peat mining (Geological Survey of Finland 2011). This cost is an opportunity cost incurred when an area is conserved and thus not available for peat extraction. Grid cells that remained to the end of the ranking were mutually complementary and had generally high-weighted feature richness and rarity and low peat-mining potential. Focusing on the low-priority end of a Zonation priority ranking (first grid cells removed) allowed us to implement the inverse spatial conservation prioritization principle: identification of areas that have comparatively low ecological values but high utility for other land uses.

Analysis Structure

We used the additive benefit function method to aggregate conservation value (ABF) (parameter $z = 0.25$) (Moilanen 2007). The ABF assumes conservation value is additive across features, and representation of each feature is converted to a value with a benefit function such as the one used in the canonical species-area curve (Arponen et al. 2005; Moilanen et al. 2011):

$$V(S) = \sum_j w_j R_j(S)^{z_j}, \tag{1}$$

where $V(S)$ is the value of the remaining set of grid cells S, w_j is the weight of ecological feature j, and z_j is a feature-specific exponent. The quantity $R_j(S)$ is the normalized distribution of the feature remaining in set S. Initially, when the full landscape remains, $R_j(S) = 1$ for all features. As the ranking proceeds, $R_j(S)$ declines for all features as areas are removed from conservation. The ABF removes grid cells from the remaining landscape (S) so that loss of $V(S)$ are minimized. Range-size normalization causes narrow-range features to have a relatively high effect on prioritization. The concave shape of the function implies that a balance is retained across features—when the representation of a feature declines, marginal losses for the feature increase continuously.

We conducted the analysis in 7 stages with 7 combinations of ecological features, feature weights, and analysis settings for ranking the conservation priority of areas (Fig. 1 & Table 1). The groups of features and analysis settings increased in complexity: (1) ecosystem types, (2) ecosystem types + ecosystem-type weights, (3) ecosystem types + ecosystem-type weights + ecosystem condition weight, (4) ecosystem types + ecosystem-type

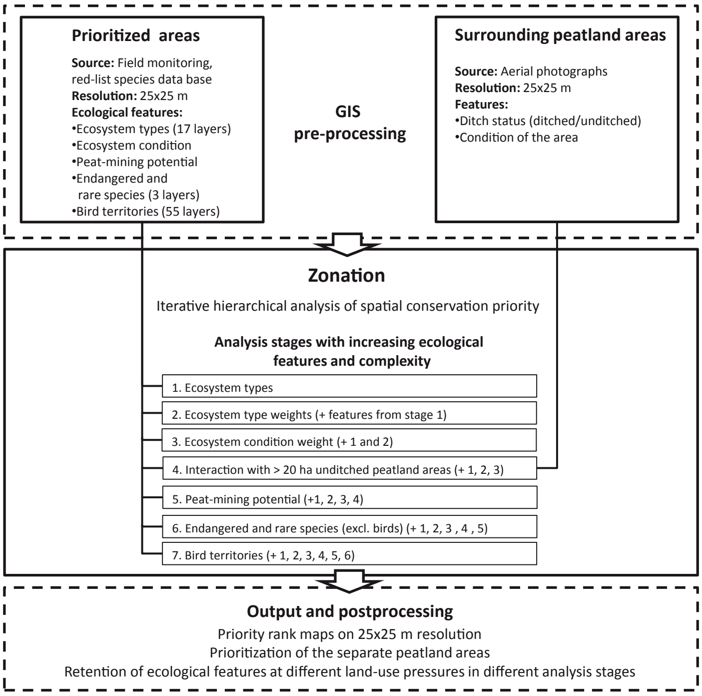

Figure 1. Process of the Zonation analysis that we used to perform the inverse spatial conservation prioritization for minimization of negative ecological effects of peat mining. Analysis stages include all the ecological-feature data used for conservation-priority ranking in the preceding stages together with new data or new analysis settings needed for the increasingly complex analyses (Table 1).

weights + ecosystem condition weight + connectivity interaction with unditched peatland areas >20 ha, (5) ecosystem types + ecosystem-type weights + ecosystem condition weight + connectivity interaction + peat-mining potential, (6) ecosystem types + ecosystem-type weights + ecosystem condition weight + connectivity interaction + peat-mining potential + red-listed and rare species, and (7) ecosystem types + ecosystem-type weights + ecosystem condition weight + connectivity interaction + peat-mining potential + red-listed and rare species + bird territories (see Supporting Information for details). We conducted analysis in stages so that we could verify the correctness of input data and analysis setup at each step.

We weighted ecological features according to an expert opinion on their relative importance (e.g., ecosystem types and condition were thought to best define the ecological value of peatland) (Table 1). Feature weights varied within and between feature groups (e.g., ecosystem types, endangered species occurrences, bird territories) (Table 1). We considered some core feature groups (e.g., ecosystem types) more important than other feature groups (e.g., bird territories), and some features within each feature group were more important than others (e.g., nutrient-rich fens versus nutrient poor pine mires). We based the final weight of each ecological feature on its relative importance within the feature group and by the relative importance of the feature group to other feature groups. Although assigning weights may resemble target setting, the weights were only used to affect the balance among features, and they do not delineate how much of each feature should be conserved per se. For example, a low-weight feature can have a high representation level if it is highly correlated in space with other features. The built-in range-size normalization introduces a tendency for Zonation analyses to give high priority to areas with occurrences of narrow-range features (Moilanen et al. 2011).

We initially performed the prioritization at a 25 × 25 m grid resolution. For the purpose of zoning, we constructed the final prioritization with entire peatland entities (306 planning units) as planning units. We did this because when a peatland is mined, the drainage of the area usually affects the entire peatland, not just the individual grid cells where peat has been extracted. Nevertheless, in other types of land use, effects can

Table 1. Ecological-feature data layers, feature weights, and analysis settings used in different stages of the analysis to prioritize peatlands.

Stage	*Ecological feature and analysis setting on which conservation-priority ranking is based*[a]	*Description (conservation status)*[b]	*Weight*
1	Ecosystem types (separate layer for each type)	eutrophic fens (CR)	1
		eutrophic spruce fen mires (EN)	1
		springs (EN)	1
		mesotrophic fens (VU)	1
		mesotrophic spruce fens (VU)	1
		mesotrophic pine fens (VU)	1
		oligotrophic fens (VU)	
		oligotrophic pine mires (VU)	1
		oligotrophic spruce fens (VU)	1
		oligotrophic pine fens (VU)	1
		swamps (NT)	1
		ombrotrophic fens (NT)	1
		ombrotrophic pine mires (NT)	1
		ombrotrophic pine fens	1
		moderately changed peatlands	1
		severely changed peatlands	1
		peat mining sites	1
2	Ecosystem-type weights	CR ecosystemt types	5
		EN ecosystems types	4
		VU ecosystems types	3
		NT ecosystems types	2
		ombrotrophic pine fens	1
		moderately changed peatlands	0.5
		severely changed peatlands	0.1
		peat mining sites	0.001
3	Condition or state of peatland area: past loss of habitat condition was modeled as a local decrease in the occurrence level of the peatland ecosystem at the affected location	pristine	1.0
		near pristine	0.8
		moderately changed peatland	0.5
		severely changed peatland	0.2
		peat miming site	0.0
4	Unditched, >20 ha peatland areas: this information was used in connectivity calculation of prioritized sites	unditched peatland areas of Central Finland and their condition	NA
5	Peat-mining potential	proportion of each peatland area suitable for peat mining	−5
6	Endangered and rare species (excluding birds): one species group layer for each level of conservation priority; value of the occurrence given only to the grid cell it was observed in	vulnerable species (4 species/29 observations)	3
		near threatened species (19/52)	2
		regionally threatened and rare species (18 / 121)	1
7	Bird territories: individual layers for all observed bird species; value of each territory partitioned to the grid cells of the peatland area it was observed in	birds, high priority (21 species layers/264 territories)	0.5
		birds, moderate priority (16 species layers/320 territories)	0.25
		birds, low priority (18 species layers/1357 territories)	0.05

[a]*Each successive stage of the analysis includes all features of previous stages (e.g., condition or state of peatland area also includes ecosystem type and ecosystem-type weight).*
[b]*Abbreviations: CR, critically endangered; EN, endangered; VU, vulnerable; NT, near threatened.*

remain localized, in which case the grid-based solution can help identify small areas that are most important to conserve.

We used the feature-specific performance curves automatically created in Zonation to investigate the quantitative trade-off between ecological features and peat-mining potential. We compared the outcome of the full prioritization with a random allocation scenario and with a greedy selection scenario in which peat extraction was maximized without regard to ecological values.

Results

We created a prioritization over the entire landscape (Fig. 2) and successfully found a solution in which the need for peat mining areas can be satisfied without major loss of different ecosystem types, endangered species, or bird territories (Fig. 3a & Table 2). The spatial priority map created in the Zonation analysis showed the priority of different peatlands (Fig. 2). In the present case, the more pertinent information was in the

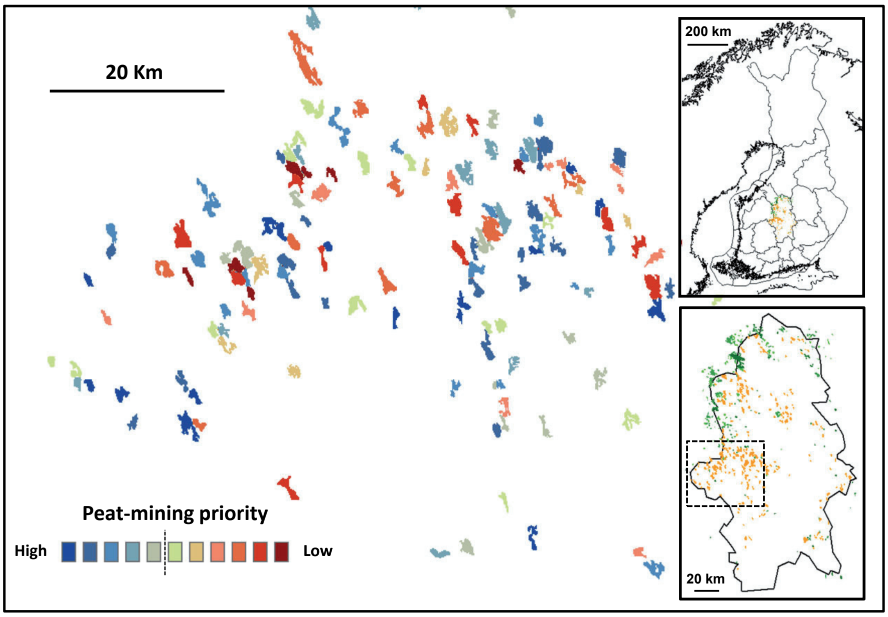

Figure 2. Close up of the inverse spatial conservation prioritization of peatlands under consideration for peat mining (high, high priority for peat mining; low, low priority for peat mining; line dividing priority color scheme, 7000 ha realized peat-mining potential). The upper inset shows the location of the study area in Finland, and the lower inset shows the spatial distributions of >20 ha of unditched peatland (green) and the candidate areas for peat mining we included (brown). Map outlines are from the National Land Survey of Finland (2010).

performance curves, which showed the status of each ecological feature (or feature group) throughout the prioritization (Fig. 3 & Table 2). The distributions of most of the ecological features remained at high levels up to the loss of approximately 42% of peatlands (Fig. 3a & Table 2). Thereafter, the retention of ecological features declined substantially (Fig. 3a). With 42% (approximately 15,200 ha) loss of the total peatland area, 47% (approximately 7000 ha) of the total peat-mining potential was achieved, and on average 82% of the distributions of biological diversity features were retained in the rest of the landscape (Fig. 3a & Table 2).

When no prioritization was used (i.e., spatial allocation of mining areas was random), the decrease in distributions of ecological features and the increase in realized peat-mining potential were on average in direct proportion to total area allocated for peat mining (see the decrease of nonprioritized features' distributions in greedy selection in Fig. 3b). In the scenario of random allocation of peatlands to peat mining, 47% of the peat-mining potential was achieved in 47% of the total area, leaving, on average, 53% of the ecological value, with potentially large random differences between individual features. Consequently, the inverse spatial conservation prioritization resulted in, on average, retention of 54% more of the ecological value (82% versus 53%) (Table 2) than random allocation of peatlands. The aggregate benefit of spatial prioritization was that ecological losses were reduced by, on average, 62% relative to random allocation.

In the economically realistic greedy-selection scenario, peatlands were allocated to mining in decreasing order of peat content (amount per area) without consideration of biological diversity features (Fig. 3b). In this scenario, 47% of peat-mining potential (approximately 7000 ha of peat-mining area) was achieved in 34% of the total peatland area, a significantly smaller area than that required by full prioritization or random allocation. Due to the high overall reduction in total area, the greedy scenario produced an ecologically better outcome than the random-allocation scenario. Because there was no strong correlation between the ecological features and peat-mining potential, ecological value declined in direct proportion to area, but there were relatively large

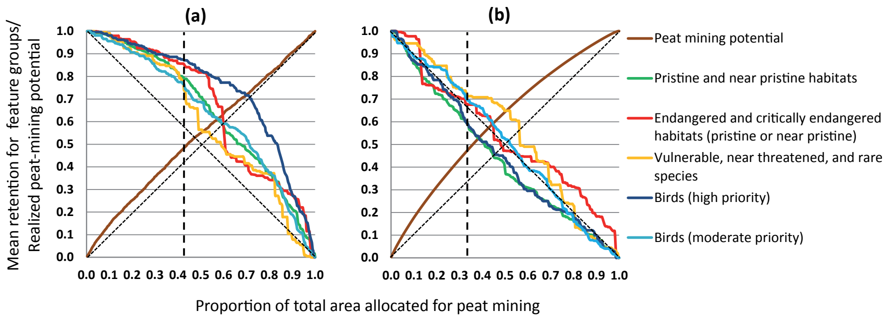

Figure 3. Performance curves for prioritizations of peatlands: (a) baseline analysis including all major ecological feature groups and peat-mining potential and (b) simple greedy selection (peatlands allocated to mining in decreasing order of proportion of area suitable for peat mining). Each declining line shows the average proportion remaining across features within one major biological diversity feature group, plotted as a function of declining area available for conservation. The thick increasing line shows the proportion of maximal peat-mining potential realized as a function of the proportion of area outside conservation. For example, 0.8 proportion remaining for birds of high priority means that on average 80% of the territories of these birds are in peatlands saved from peat mining. The thin black diagonal lines represent expected decrease in ecological values (Table 1) and expected increase in realized peat-mining potential if areas are randomly allocated into peat mining (i.e., the random selection scenario). The dashed vertical lines mark the 7000-ha realized peat-mining potential (Table 1).

difference among features. This resulted in relatively high and unnecessary losses of ecological value. For example, 44% of the pristine and near-pristine peatland area was lost as 42% of the distributions of high-priority bird species. On average, approximately 65% of the distributions of biological diversity features were retained in this scenario (Fig. 3b), which compares unfavorably with the approximately 82% retained by full prioritization of multiple ecological features (Fig. 3a). These figures are naturally case specific, but they illustrate the quantitative benefits generated by the propose analyses.

Table 2. Biological diversity gains with inverse spatial conservation prioritization with approximately 7000 ha of suitable area for peat mining realized.*

Biological diversity feature	*Distribution size in full study area*	*Remaining in full analysis*	*Remaining in random selection*	*Remaining in greedy selection*	*Gain compared with random selection*	*Gain compared with greedy selection*
Total area (ha)	36503	0.58 (21,040)	0.53 (19,347)	0.65 (23,864)	0.10 (1975)	−0.11 (−2,542)
Pristine and near pristine peatland area (ha)	4531	0.80 (3629)	0.53 (2401)	0.56 (2537)	0.51 (1228)	0.43 (1092)
Critically endangered and endangered areas (pristine and near pristine) (ha)	300	0.86 (257)	0.53 (159)	0.68 (204)	0.62 (98)	0.26 (53)
Endangered and rare species (occurrences)	202	0.78 (158)	0.53 (107)	0.71 (143)	0.48 (51)	0.10 (15)
Birds, high priority (territories)	264	0.88 (231)	0.53 (140)	0.58 (153)	0.65 (91)	0.51 (78)
Birds, moderate priority (territories)	320	0.77 (246)	0.53 (170)	0.68 (217)	0.45 (76)	0.13 (29)
Average remaining proportion (excluding total area)		0.82	0.53	0.64		
Average relative gain of the full analysis (excluding total area)					0.54	0.29

Retention and gain (full analysis compared with random and greedy selections) are expressed as fractions of total. Where relevant, absolute amounts corresponding to fractions are in parentheses.

Discussion

Our method of inverse spatial conservation prioritization balances economic requirements of one or several stakeholders while minimizing ecological loss aggregated over multiple biological diversity features. The spatial solutions derived from this method delineates areas ideal for avoidance of negative ecological effects. Our results demonstrated that the negative effects of human land use can be successfully avoided because we found that over 80% of the known ecological values of the planning area could be saved with a reasonable trade-off for peat-mining interests. In fact, when the peat-mining potential of areas was included in the analysis, the expected efficiency of peat mining increased relative to random allocation of areas to peat mining. Inverse spatial conservation prioritization also resulted in significant ecological savings compared with a greedy selection scenario, which only minimized the total area needed. On the basis of our results and other considerations, including negotiations with various stakeholders, the Regional Council of Central Finland decided that approximately 7000 ha of area suitable for peat mining will be allocated for peat extraction in the final land-use zoning plan. According to our analysis 7000 ha of suitable peat-mining area can be derived with 42% of the total peatland area allocated to peat mining and over 80% of the known ecological values of the planning area retained.

We call the principle applied here inverse spatial conservation prioritization because, in contrast to typical spatial prioritization, the idea is not to identify (for protection) areas with the highest ecological values. Rather, the objective is to identify the inverse end of the landscape, that is, areas with the lowest ecological values that are simultaneously the most appropriate for economic land uses (here peat mining). Of course one still needs to decide where to draw the line of acceptable ecological loss (Fig. 3). Estimation the consequences of this decision is plagued by uncertainty arising from incomplete data and understanding of ecological dynamics and, perhaps more importantly, changing political will. However, we believe, and others agree, that decision making is better justified and more acceptable when the trade-offs between economic gain and ecological loss are first examined quantitatively (e.g., Wilhere 2008).

It has been suggested that avoidance of negative ecological effects should be the primary goal of conservation efforts that counter development effects that may threaten the environment (Cuperus et al. 2001; Ten Kate et al. 2004). Even so, avoidance of negative effects is rarely discussed or properly applied in the context of land-use planning in unprotected areas that are not perceived to hold any great ecological value (e.g., Maiorano et al. 2008; Mathur & Sitha 2008; Chazdon et al. 2009). Nevertheless, it is known that even partially degraded unprotected areas can provide ecosystem services or hold ecological value that substantially supplement conservation-area networks (e.g., Fischer & Lindenmayer 2002; Bengtson et al. 2003; Laita et al. 2010). Integrating the inverse spatial conservation prioritization principle to global land-use planning could avoid a multitude of individually small ecological losses and could indirectly generate huge conservation gains at regional and global scales.

The proposed approach differs from previous related work that focused on targeting of protection measures to valuable sites that would experience substantial loss of biological diversity value in the absence of protection because we focused on ecologically low-value sites. One well-known approach to loss avoidance is to use a combination of vulnerability and irreplaceability to determine protection priorities for ecological features of interest (Gaston et al. 2002). Pressey et al. (2004) introduced the principle of maximal retention of biological diversity by loss minimization, which combined loss rates and presence of biological diversity in one measure. Ban and Vincent (2009) turned traditional target-setting around by setting targets for fishery yields and then minimizing the area (ecological cost) needed to satisfy these economic targets, thus working inversely from classic SCP problem formulations. The approach Ban and Vincent (2009) used allows spatial prioritization in which multiple target features are balanced with a single cost (i.e., a many-to-one structure). In their case, this was multiple targets for economic components (yields of separate fisheries to be satisfied with minimum area) and a single aggregate layer for area (area as a surrogate for biological diversity value), the allocation of which to fishery use was minimized as a cost. Instead of balancing distributions of multiple features with a single cost, our method has a computational structure in which multiple biological diversity features may be balanced against multiple costs (Moilanen et al. 2011), which is a major advantage because biological diversity value need not be represented by a single layer.

Recently, Klein et al. (2010), Weeks et al. (2010), Wilson et al. (2010), and Grantham et al. (2013) applied an approach that allows sites to be placed into one of several different planning zones, and the zone designation has different effects on economic gains and biological diversity features. All these authors focused on zoning with balanced benefits across multiple stakeholders. For example, Grantham et al. (2013) set separate protection targets for species, ecosystems, and coverage of communal fishing grounds, thereby identifying a solution that produces prespecified economic benefits across stakeholders while retaining target levels of biological diversity.

As a distinguishing feature, our work does not require a priori setting of targets, which is a practical and conceptual advantage (Laitila & Moilanen 2012). Here, an acceptable trade-off between production and nonproduction areas is quantified after prioritization with performance curves. This is in the reverse order of the common practice of SCP (Margules & Pressey 2000; Cawardine et al. 2009). The target-free inverse approach bypasses overall inefficiency that can arise from the target-setting model and processes because a poorly set target can consume a disproportionate fraction of resources and lead to inferior aggregate conservation performance (Moilanen & Arponen 2011; Di Minin & Moilanen 2012; Laitila & Moilanen 2012). This is a real possibility when many targets need to be set for different types of biological diversity features. Our work offers a different and flexible approach to balancing economic benefits and ecological effects of development through the use of spatial-prioritization tools.

Conservation prioritization should be flexible in the sense that it does not lose its effect when facing real life considerations (Margules & Pressey 2000; Cabeza & Moilanen 2006; Sarkar et al. 2006; Knight et al. 2008; Arponen et al. 2010). We contend that this flexibility requirement should apply to other forms of land use planning as well. Fortunately, prioritization outside protected areas does not have to be an either/or battle between ecological values and economic returns (Polasky et al. 2005; Polasky et al. 2008; Perhans et al. 2011). Inverse spatial conservation prioritization, as used here, can be successful in identifying profitable production areas while simultaneously safeguarding ecological values in an all-inclusive approach to land use decisions.

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Supporting Information

Information on the data and analysis stages we used are available online (Appendix S1). The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Supporting information

(Use of Inverse spatial conservation prioritization to avoid biodiversity loss outside protected areas, Kareksela et al. 2013)

Description of analysis stages

Stage 1. *Ecosystem types*. The first variant used only ecosystem type data without differential weights (all ecosystem types had weight 1.0, table 1). Ecosystem types were defined in the field according to the Finnish peatland classification, which consists of nearly one hundred specific peatland ecosystem types (Eurola et al. 1984). This data was gathered by biologists of the Regional Council of Central Finland. For the present analysis, ecosystem type data were grouped into 18 more robust classes according to the main peatland types and their nutrient levels including moderately and severely changed peatlands and peat mining areas as separate ecosystem types (table 1).

Stage 2. *Ecosystem-type weights*. Adding weighting for the ecosystem types. This analysis variant places higher emphasis on peatland ecosystem types according to their status in the habitats red list of Finland (Raunio et al. 2008; Kontula & Raunio 2009) (table 1).

Stage 3. *Condition weight*. Adding condition (naturalness) of each defined ecosystem within the focal sites to the prioritization criteria. This analysis variant adds further emphasis on focal sites that have experienced low human impacts. Biologists of the Regional Council of Central Finland defined the condition of each individual ecosystem fragment within the 306 focal sites in the field in terms of how much, if at all, the ecosystems were influenced by ditching. Condition was divided into five classes with coefficients from 0 to 1.0 (table 1).

Stage 4. *Interaction with unditched peatland areas over 20 hectares*. Adding a connectivity interaction to unditched peatlands of over 20 ha in size to the prioritization criteria. This data feature emphasizes the connectivity of the focal sites to larger unditched pristine or near-pristine peatlands. This connectivity consideration increases the value of the prioritized areas as the function of the distance to and condition of the unditched peatland areas over 20 ha in the landscape. The unditched peatlands over 20 ha were assigned local ecosystem quality (condition) ranging from 0.2 to 1.0 based on similar criteria as defined above (variant III) for the condition of the focal sites (table 1). Data for the quality of the unditched over 20 ha peatland areas was compiled by the Finnish Environment Institute based on remote sensing analysis and interpretation of aerial photographs as a part of a larger peatland evaluation project. The connectivity interaction was implemented using the method of Rayfield et al. (2009), using a mean spatial scale of 500 meters for the negative exponential spatial interaction kernel.

Stage 5. *Peat mining potential*. Adding peat mining potential to the prioritization criteria. This data feature balances ecological values with economic potential. Peat mining potential of the focal sites was based on the proportion of area of each peatland entity suitable for peat mining (over 1.5. meter peat depth). Peat mining potential was given a negative weight in the analysis (table 1), so instead of aiming to retain the feature, the analysis aims to remove grid cells with higher peat mining potential as early as possible (Moilanen et al. 2011). The opportunity cost used in the analysis, peat mining potential, is in units of area suitable for peat mining. This data was provided by the peatland surveys of Geological Survey of Finland (GTK; http://en.gtk.fi/expert_services/energy/peat/).

Stage 6. *Red listed and rare species*. Adding red listed and rare species observation data to the prioritization criteria. This feature emphasizes areas that contain red listed or rare peatland species. The red listed and rare species data was acquired from the field observations supplemented with data from the national red listed species data base HERTTA. Due to a low number of observations, individual species were not included as separate layers, but as species classes. Species were divided to three differently weighted classes according to their conservation status (Rassi et al. 2010) (table 1).

Stage 7. *Bird territories*. Adding bird observation data (1941 observed territories of 55 species) to the prioritization criteria. The Regional Council of Central Finland collated bird species data in 2007-2010. The biologists of the Regional Council of Central Finland divided bird species into three priority classes, and all of the bird observations were given weights corresponding to the priority class they belong to (table 1).

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II

EXPOSING ECOLOGICAL AND ECONOMIC COSTS OF RESEARCH-IMPLEMENTATION GAP AND COMPROMISES IN DECISION-MAKING.

by

Santtu Kareksela, Atte Moilanen, Olli Ristaniemi, Reima Välivaara & Janne S. Kotiaho 2014

Manuscript

Exposing ecological and economic costs of research-implementation gap and compromises in decision-making

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Key words: conservation implementation, decision-making, impact avoidance, inverse resource allocation problem, peat-mining, replacement cost, Zonation software

Abstract

The gap between conservation science and practice is frequently discussed. This gap feels most tangible when it is between spatial conservation prioritization case studies and their implementation in actual zoning. Here we explore reasons behind differences between actual zoning and one such prioritization analysis focusing on ecological impact avoidance of peat-mining. We obtained the final realized zoning plan to compare, analyze and determine the ecological costs of the possible gaps in the research–planning–decision-making chain in this project. We were able to quantify two gaps in two different phases of the project: between researchers and planners and between planners and decision-makers. We then discussed the results and the reasons behind these gaps together with the planning authorities responsible for the zoning and commissioning the spatial prioritization analysis. In our discussions we identified three major reasons for the gap between the suggestions of scientists and planning authorities: 1) an ease of valuing single sites independently versus valuing the landscape level solution for all sites concurrently, 2) planning for biodiversity conservation versus resource allocation 3) preference of single ecological features important for decision making versus considering all possible ecological features. There were also additional environmental considerations in the later stages of the zoning creating prioritization differences compared to the suggestions from the scientists and the planners. In addition to our findings we demonstrate how the high end tools can be useful in quantifying research-implementation gaps and identifying replacement costs of the many trade-offs in resource allocation projects. In depth analysis of the decision-making process combined with the discussions with the authorities also uncovered that in the real world the land-use decision-making is a very complex iterative process and the rather dualistic nature of the research-implementation gap concept is inadequate in describing the many phases. We conclude that complexity of a real-life planning-decision-making chain alone may set restrictions to how optimal the science based prioritization tools are in action.

Introduction

Recently Game *et al.* (2013) wrote about "six common mistakes in conservation priority setting", summarizing reasons for the frequent failure to implement scientific conservation recommendations in actual decision-making. The research-implementation gap is a serious matter influencing the cost-efficiency of individual cases *per se*, but in general it plagues all applied scientific effort (Ehrenfeld 2000, Cabin 2007, Hulme 2014) and may thus challenge the whole concept of applied ecology. Research-implementation gap has been discussed also earlier, both in the literature of conservation biology (e.g. Ehrenfeld 2000, Knight *et al.* 2008) and restoration ecology (e.g. Cabin 2007, Giardina *et al.* 2007). Progress towards better alignment between conservation scientists, planners and decision-makers has been frustratingly slow, despite conservation scientists proposing several ways to improve the usefulness of scientific analysis. Redford and Tabert (2000) and Knight (2006) stress the importance of "writing the wrongs", i.e. publishing also the shortcomings of implementation to prevent repeating avoidable mistakes. Cabin (2007) suggested that simplifying matters and using common sense could be more fruitful than possibly over-sophisticated analyses and experiments. Knight and Cowling (2007) demanded more support for taking advantage of opportunities while Pressey and Bottrill (2008) clarified the role of systematic conservation planning in distancing decision-making from harmful opportunism. Gilbert (2011) proposed that better psychological understanding of human decision-making could help us to close the gap. Game et al. (2013) emphasize the importance of understanding that conservation decision-making is fundamentally a matter of resource allocation. Finally, there are several studies and approaches offering solutions expressly to the matter (e.g. Margules and Pressey 2000, Knight *et al.* 2006, 2010), i.e. bridging the gap by offering operational models for more effective implementation.

Inspired by these discussions we follow up on our recent case study of ecological impact avoidance in spatial prioritization of peat-mining, in a regional zoning process (Kareksela *et al.* 2013). We document the quantitative difference between prioritization analysis based suggestion, suggestion by planning authorities, and the final land-use zoning and investigate why the suggestions and the final zoning differ. We then further analyze the apparent ecological and economic costs of the gap between research, planning and implementation, and illustrate how quantitative and qualitative trade-offs in resource allocation projects can be exposed through replacement cost analysis (Cabeza and Moilanen 2006).

In the original spatial prioritization (Kareksela *et al.* 2013), we prioritized 306 peatland areas in Central Finland. Instead of identifying areas with the highest conservation value, we developed an inverse approach to find an economically viable set of peatland areas, where peat-mining would least reduce landscape-level ecological values of the prioritized peatlands. Thus, this was an inverse analogue for traditional conservation resource allocation problem, in other words, environmental impact avoidance via allocation of resource extraction. This work was commissioned by the Regional Council of Central Finland (RCCF), who also collated substantial biodiversity data across the region and economic value for the prioritized areas (peat-mining potential: proportion of a peatland's area suitable for peat-mining). Our subsequent suggestion, based on the prioritization including all the produced spatial data, was delivered to the planning authorities of the RCCF following with several sessions were the results were explained and discussed. The analysis served as an important input into the preliminary zoning plan that was then put together and suggested by the planning authorities. The preliminary plan was then modified according to comments by different stakeholders before implemented as a final zoning decision. The results of the prioritization analysis were presented and explained to the stakeholder group. However, the more thorough use of the results was restricted to the planning authorities as originally RCCF commissioned the analysis independent from the stakeholder group to function as a decision support tool for their planning work. The final land-use zoning suggestion was agreed upon and made public by RCCF in May 2013 and it is currently in the final stage of the zoning process (Fig. 1) to make it legally binding.

Methods

We analyze three differences between the prioritization analysis based suggestion, alternative suggestion developed by the planning authorities, and the final land-use zoning, which was negotiated as a compromise between expert opinion and local stakeholder politics. First, by how many sites does the spatial prioritization analysis based suggestion (hereafter scientists' suggestion) differ from the suggestion of the planning authorities (planners' suggestion) and the final zoning? Second, how well do different zoning suggestion measure in terms of ecological and biodiversity performance? Third, what were the causes behind the differences between the prioritization analysis based suggestion, the suggestion by planning authorities, and the final land-use zoning? Preferences underlying decisions that differed from the prioritization analysis based suggestion were not always traceable, but some clear reason could be identified.

Specifically, we compared five suggestions: 1) suggestion based on the inverse prioritization analysis and extensive avoidance of impacts (Kareksela et al. 2013) (scientists' suggestion); 2) a prioritization analysis based suggestion using the same inverse prioritization, but a stakeholder-mandated higher requirement for peat extraction; 3) the prioritization analysis based suggestion modified by excluding certain sites from peat-mining because of high expected damage for surface waters and recreational value; 4) the suggestion made by planning authorities (planners' suggestion); 5) the actual land-use zoning suggestion. We also performed a simple replacement-cost analysis, i.e. determined ecological or alternatively economical costs of choosing a different type of trade-off than suggested by the original prioritization analysis (Kareksela *et al*. 2013).

Numeric comparison of the performance of different suggestions alone may be one-sided. Hence, here we report the analyses and discussions about the differences that were undertaken together with the planning authorities (OR, RV) responsible for the zoning and commissioning the prioritization analysis and the scientists performing the prioritization analysis (SK, AM, JSK). Together we describe the decision-making process and discus the results and the reasons behind the differences between the suggestions and the actual zoning. The planning authority authors' specific roles in the zoning project were the following: OR was the zoning project leader and has the expertise on peat-mining and RV was the leader of the ecological evaluation and has the expertise on peatland biology.

Results

Detailed analysis of the actual zoning project (Fig. 1) revealed that it was a very complex iterative process with the zoning suggestion evolving over several stages at five different administrative institutions. It was commented in between twice by the stakeholder group and four times by landowners and anyone it considers. The most time consuming phase of the zoning suggestion evolution is the middle way double circle concerning the planners, stakeholders, middle level decision-makers (Managing board) and landowners and other citizens and this is where most of the changes in the zoning suggestion are taking place. After the confirmation by the ministry of environment, the zoning suggestion is open for complaints that will be dealt in the Supreme Administrative Court, whose decision will ultimately finalize the process.

Scientists' suggestion and the planners' suggestion differed for 102 out of 301 peat lands, 35 of which we originally suggested for peat-mining and 67 for saving (Kareksela et al. 2013). Similarly, scientists' suggestion differed from the final zoning in 107 cases (50/57). In addition the planners' suggestion and final zoning differed in 71 peatlands of which 41 sites were suggested for peat-mining by the planners while suggested not for peat-mining in the zoning plan and 30 cases suggested for peat-mining in the zoning plan while suggested not for peat-mining by the planners. While differences were mostly for sites that had intermediate ranking in the scientists' prioritization, there were also notable differences for some sites that had very strong recommendations (high or low priority rank) in the scientists' prioritization.

The greatest differences in the remaining proportions of ecological features were between scientists' suggestion and the final land-use zoning (Table 1). Compared to the final zoning, scientists' suggestion delivered on average 11 percentage points higher representation across all biodiversity features in the area left outside peat-mining (Table 1). In terms of fractional loss of biodiversity, scientists' suggestion would have achieved 38% smaller losses of ecosystem and species distributions. Peatlands differ in their peat-mining potential i.e. in some peatlands the proportion of area actually suitable for mining is smaller than in other peatlands (Kareksela *et al.* 2013). Scientists' suggestion would have provided 4% (2 percentage points) less area actually suitable for peat-mining. However, in terms of total area consumed, the scientists' suggestion would have needed 17% (8 percentage points) less total area allocated into peat-mining use. Losing only 4% peat-mining area with 17% less total area used is a result of the scientists' suggestion allocating fewer but resource richer areas into peat-mining. In terms of economic efficiency this indicates savings as fewer production sites would have been needed to be established for the given amount of extracted resource thus lowering the total economic investments needed.

Three main reasons explained the differences between the scientists' and the planners' suggestions (Table 1). First, there is a difference in prioritizing observed locations of threatened species versus other biodiversity features (see Table 1). The most plausible explanation for the relatively high priority of threatened species in the planning authorities' suggestion and in the final zoning is that many of the threatened species have higher legitimating power than other biodiversity features. While this is a valid consideration in decision-making, endangered species occurrence data may suffer from biased detection and it is deficient in preserving overall biodiversity and functional ecosystems. Second, the suggestions differed in how they consider the varying peat-mining potential of individual areas. The average peat-mining potential across all prioritized areas is 42%. However, the areas that planning authorities suggested for mining against the scientists' suggestion have an average peat-mining potential of 33%, while the areas we suggested for peat-mining, but were not allocated for it, have an average peat-mining potential of 48%. Considering economic returns, like in the scientists' suggestion, would have made it possible to meet economic objectives with smaller total area sacrificed, thereby leading to higher proportion of ecological values retained (Table 1, e.g. Margules and Pressey 2000, Naidoo *et al.* 2006, Kareksela *et al.* 2013). Third, and closely related to the first two reasons, when discussed among the authors it was apparent that the expert opinion (here the planning authorities) often emphasis importance of single sites over the whole solution which is significantly different perspective compared to complementary based analyses like Zonation here (e.g. Kareksela *et al.* 2013). In addition, we identified a fourth reason creating a gap between scientists' suggestion and the actual zoning suggestion. This was the usage of additional environmental data (Table 2) in the final zoning, creating further trade-offs and thus resulting in a decreased average retention of the ecological features considered in the original prioritization analysis (Kareksela *et al.* 2013).

The two different replacement-cost analyses, one for the increased allocation of area suitable for peat-mining (as mandated by stake-holders) and the other for forced retention of sites with potential surface water effects, showed that such forcing has a price in terms of reduced ecological values. While the reduction is only a moderate three percentage points on average, the representation of endangered and rare species declines nine percentage points, thereby revealing a significant trade-off. In summary, the relatively low mean retention of ecosystem and species distributions in the final zoning follows from reduced emphasis on peat-mining potential, elevated emphasis on endangered species observations, and inclusion of additional considerations.

Discussion and lessons learned

Perhaps the clearest indication of a research-implementation gap was the difference between the prioritization analysis based suggestion and the preliminary zoning plan by the planning authorities. We found support to the points made by Game *et al.* (2013) as using the peat-mining potential data was significantly overruled by the ecological perspective. This and the fact that some relevant political and environmental factors (Table 2) were introduced only after the prioritization had been completed demonstrate a shortcoming of seeing these factors as part of the same resource allocation process (Game *et al.* 2013). On the other hand, this failure should also be attributed to scientists, due to our failure to identify all relevant decision criteria from stakeholders and decision-makers.

Second lesson to consider is the observation that in practice the focus is often on single areas instead of the whole solution, i.e. the quality of the whole group suggested for peat-mining or protection. This is a very closely related issue to the lack of seeing the zoning as a prioritization and allocation of resources (Game *et al.* 2013). When looking at individual sites there lies a danger that local considerations and stakeholders interests start driving decisions too strongly, and the balance of the whole solution suffers from what it could have been (Pressey and Bottrill 2008, Ahlroth and Kotiaho 2009). In other words, complementarity and cost-efficiency end up with reduced emphasis and the overall solution degrades. It has, however, at least one very apparent reason: the planners, or in this case middle way authorities, often need to justify their suggestion at a site level and it is easier to discuss the qualities that are present on the site than the complementing or connecting role of the site in the landscape. Although the final authorities may see the whole prioritization analysis result as a valid argument for the individual sites it is not this obvious for land owners, and perhaps also in the eyes of the law, if the land owners choose to challenge the legitimacy of the decision. Therefore, it may be that it is clearer to put more weight on the legitimating ecological features in the planning process, seen here as the high weight given to endangered species (mainly vulnerable, VU) in the final zoning suggestion. As many considerations exist it is not in the end easy to claim superiority of one solution over another. However, for example in the case of considering single sites over the whole solution, the resulting trade-offs can be evaluated by the replacement-cost methods represented in here (Table 1) and e.g. in Cabeza and Moilanen (2006) and Kareksela *et al.* (2013) and we feel that this would greatly benefit the decision-making process when properly applied.

Third and perhaps the most striking lesson from the scientists' perspective was the complexity of the decision-making process (Fig. 1). The fact that even the planning authorities' suggestion and the final zoning suggestion differed in 71 cases out of 301 shows that there are in fact multiple gaps in the whole process. As figure 1 further illustrates, here the challenge for applied conservation science is to transmit the scientific information in multiple occasions to multiple institutions within a single zoning process. It should also be noted that the suggested operational models to help implementation usually also add complexity to already complex situations. Following e.g. the implementation guidelines suggested in Knight *et al.* (2006, 2011) would leave us trying to tackle the complex decision-making vortex (Fig. 1) with something that the authors (Knight *et al.* 2006, 2011) describe as an operational model of "complex, heuristic, web-like structure", with nearly twenty different steps defined. Implementing prioritization analysis as a part of holistic land-use zoning project dealing with multiple trade-offs, all possible stakeholders, and administrative institutions is a lot more complicated than for example a "simple expansion" of a local protected area network. It is no longer about a gap but about multiple gaps as shown by our results, and even identifying the most obvious ones is hard work not to mention quantifying them all. Here we really need to think whether complex operational models for implementation strategies are suitable for bridging the gaps that partly result from complexity in the first place. Here, as pointed out by the planners and co-authors (OR and RV), the implementation of the spatial prioritization analysis would have benefitted from being applied earlier in the process i.e. already in the pre-face of the zoning project. This is of course essential in all the implementation models (Margules and Pressey 2000, Knight *et al.* 2006, 2011). So why did we not? In this case the authors of this article found each other when the project was already ongoing, i.e. the scientists jumped into a moving train, showing the flexibility needed also from the implementation models. Intuitively the beginning of the implementation process is in fact often not an ideal one and we still need to find best practices to apply when the possibly optimal implementation pathway is no longer available.

In an ideal world, increased communication through the process would have improved both the analysis and its uptake, reducing ecological impacts from what they will now be. Such communication and proper identification of objectives is part of the standard process of systematic conservation planning (Margules and Pressey 2000, Pressey and Bottrill 2008). As a relevant consideration, we highlight the following: real life spatial plans such as the one in Kareksela *et al.* (2013) have to be developed under constraints on time, money and work force. For this reason, good availability of spatial biodiversity data is of primary importance for the timely delivery of high-quality policy-relevant spatial prioritizations. If data is available, it is possible to concentrate on developing the utility of the analysis itself. If data is not available, much time may be used in the collation of data. Moreover, it is difficult to discuss and develop analysis structure with stakeholders when there is uncertainty about availability of ecological information on the prioritized areas. Early availability of data also facilitates easier communication through the process, allowing multiple iterations of the analysis and customized comparisons revealing the ecological and economic costs of compromises for the stakeholders and decision-makers thereby facilitating improved relevance of the final outcome. In addition to time and more concrete resources, gaps in training and experience of administrative personnel seems to be a relevant factor with respect to handling high-end prioritization analysis results, which may easily lead to neglecting results that are perhaps not always as self-evident as e.g. number of threatened species per site (suggested by the results in Walsh *et al.* 2014). Now that methods exist for broad-scale, high-resolution, multi-feature spatial prioritization, long term investment into the development and maintenance of data and pro-actively offering information on different analysis possibilities (e.g. Dicks *et al.* 2014) becomes more relevant than ever.

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Table 1. Comparison of retention of ecological values (proportion) and achieved proportion of area suitable for peat-mining (peat-mining potential) in different suggestions.

Proportion of biodiversity features remaining / peat-mining potential realized	*Scientists suggestion based on prioritization*	*Scientists suggestion with fixed peat-mining goal*[a]	*Scientists suggestion with forced exclusion of areas*[b]	*Suggestion of the planning authorities*	*Final land-use zoning*
Fraction of area remaining	0.60	0.58	0.60	0.52	0.52
Pristine and near pristine peatland area	0.81	0.79	0.77	0.77	0.69
EN and CR ecosystems (pristine or near pristine)	0.86	0.84	0.87	0.75	0.70
Endangered and rare species (occurrences)	0.79	0.69	0.69	0.83	0.83
Birds, high priority (territories)	0.88	0.87	0.87	0.80	0.72
Birds, medium priority	0.78	0.74	0.73	0.59	0.60
Mean ecosystem and species distributions remaining	0.82	0.79	0.79	0.75	0.71
Achieved proportion of the total peat-mining potential	0.45	0.47	0.45	0.47	0.47

[a]The threshold between peat-mining and no peat-mining is set so that area allocated to peat-mining equals that in the final zoning (=0.47).

[b]18 sites where peat-mining is expected to have negative effects on surface water systems are forcibly excluded from peat-mining, irrespective of ecological values or peat-mining potential in the sites.

Table 2. Documented* traceable reasons for cases where final zoning excluded areas from peat-mining differing from the scientists' (Kareksela et al. 2013) allocation suggestion.

Reason in the report	*Number of cases*
Expected negative effects of peat-mining to local surface water systems	18
Possible effects of peat-mining to nearby NATURA 2000 network areas	18
Unspecified nature values at site	9
Connectivity consideration (unspecified)	1

** Zoning report: http://www.keskisuomi.fi/4.vmk (in Finnish only, accessed March 2014).*

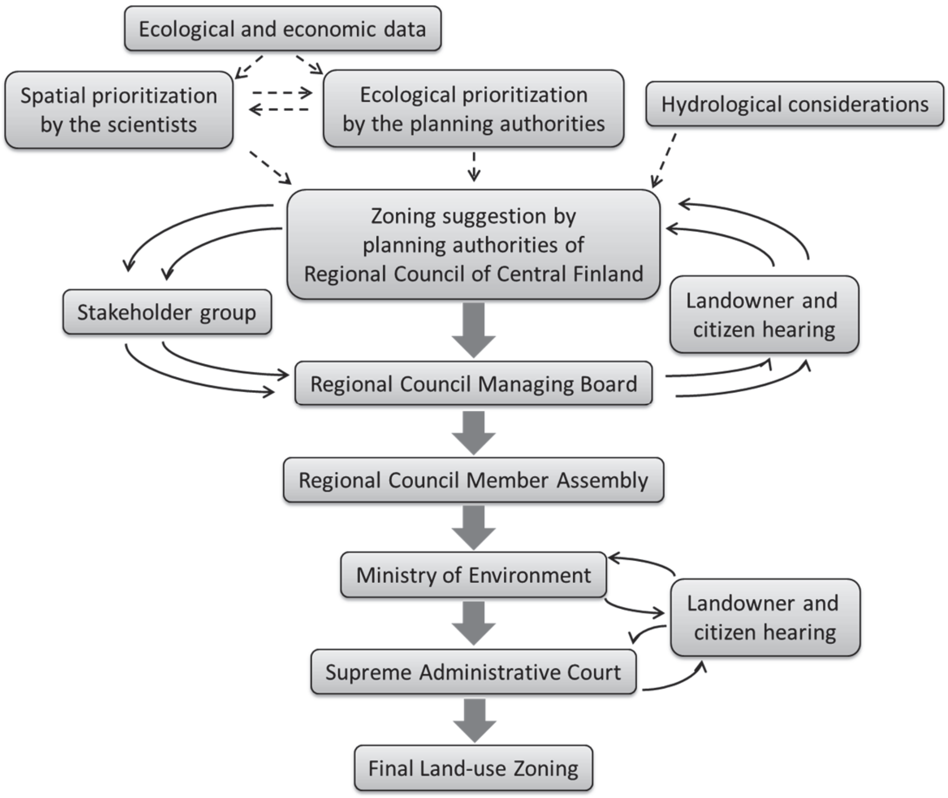

FIGURE 1 Schematic illustration of the land-use decision-making vortex in this case study. Dashed arrows represent flow of data and considerations, narrow arrows represent circulation of a version of the zoning suggestions and wide arrows represent the steps from a lower administrative institution to higher ones. The planning authorities (mainly OR, also an author of this paper) have presented the suggestion through the steps. The zoning plan was circulated twice between the planners, stakeholders, managing board and the landowners and citizens, before an official zoning suggestions proceeded from the planners to the managing board to be approved and sent forward in the chain. In other words, in this phase of the process the planning authorities produced three versions of the zoning suggestion, with the first two being called zoning plans and the third one a zoning suggestion.

III

FIGHTING CARBON LOSS OF DEGRADED PEATLANDS BY JUMP-STARTING ECOSYSTEM FUNCTIONING WITH ECOLOGICAL RESTORATION

by

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Fighting carbon loss of degraded peatlands by jump-starting ecosystem functioning with ecological restoration

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Keywords: acrotelm; carbon accumulation; carbon sequestration; plant community composition; ecosystem degradation; ecosystem recovery; ecosystem structure; peat; structure-function relationship; vegetation

Abstract

Degradation of ecosystems is one of the greatest concerns on the maintenance of biodiversity and ecosystem services. Ecological restoration fights degradation aiming at the recovery of ecosystem functions such as carbon (C) sequestration and ecosystem structures like plant communities responsible for the C sequestration function. We selected 38 pristine, drained and restored boreal peatland sites in Finland and asked i) what is the long term effect of drainage on the surface peat C storage, ii) can restoration recover ecosystem functioning (surface peat growth) and structure (plant community composition) and iii) is the recovery of the original structure needed for the recovery of ecosystem functions? We found that drainage had resulted in a substantial net loss of C from surface peat of drained sites. Restoration was successful in regaining natural growth rate in the surface peat already within 5 years after restoration. However, the regenerated surface peat sequestered C at a rate of 116.3 g m^{-2} yr^{-1}, when a comparable short-term rate was 178.2 g m^{-2} yr^{-1} at the pristine sites. The plant community compositions of the restored sites were considerably dissimilar to those of pristine sites still 10 years after restoration. We conclude that ecological restoration can be used to jump-start some key peatland ecosystem functions even without the recovery of original ecosystem structure (plant community composition). However, the re-establishment of other functions like C sequestration may require more profound recovery of conditions and ecosystem structure. We discuss the potential economic value of restored peatland ecosystems from the perspective of their C sequestration function.

Introduction

We are living an era of ecosystem manipulation. The development of human population has influenced ecosystems almost everywhere around the Earth (Foley *et al.*, 2005, Millennium Ecosystem Assessment 2005). Ecological restoration aims at the rehabilitation of ecosystem functions and structures, such as C sequestration and species communities, respectively, in order to respond to the threats of habitat and ecosystem level degradation (Vanha-Majamaa *et al.* 2007, Dobson 1997, Bradshaw 1996). Results of restoration, like increase of biodiversity or revival of populations of individual species, are promising (e.g. Benayas *et al.* 2009) and new targets for restoration of ecosystem functions and services (profitable and/or needed ecosystem functions) are being developed (Hobbs and Cramer 2008, Benayas *et al.* 2009, Aerts and Honnay 2011, Bullock *et al.* 2011, Suding 2011). Still, it seems unlikely that restored ecosystems will ever be exactly similar to their pristine targets and there are serious doubts concerning the recovery of ecosystem functions after restoration (Benayas *et al.* 2009, Zedler and Kercher 2005). The recovery of ecosystem functions is generally thought to depend on the recovery of original structures, but the relationship of ecosystems' functions and structure is still far from well understood (Bradshaw 1996, Cortina *et al.* 2006, Cardinale 2012).

Current ecological restoration frameworks face two major questions: i) can we return vital ecosystem functions and structures that are partially or totally lost and ii) if we can, is it economically feasible? The answer to the first question is dependent on the degree of ecosystem complexity and on our understanding of ecologically effective restoration practices (Cortina *et al.* 2006, Pocock *et al.* 2012). The second question is related to the complexity and thus cost of restoration actions needed, the opportunity costs (i.e. the value of current land use if not restored) and possible economic gains of restoration outcome. Increased ecosystem services, like societal benefits of C sequestration (Nelson *et al.* 2009, Alexander and McInnes 2012, Kettunen *et al.* 2012, Russi *et al.* 2013), is one example of possible economic gain. Being able to answer these questions is increasingly important because recently a global target was set to fight climate change and biodiversity loss by restoration of 15 % of degraded ecosystems by 2020 (Convention on Biological Diversity 2010). Restoration is also increasingly being called upon to compensate for the biodiversity values lost in development projects by restoring degraded sites elsewhere (Maron *et al.* 2012). Cost effective and rather straightforward restoration methods are most likely needed to reach such demanding targets (Perrings *et al.* 2010).

Increasing attention is paid to the ability of ecosystems to absorb CO_2 and store C (Feng 2005, Davidson and Janssens 2006). Compared to other ecosystems, peatlands are the largest reserves of organic C in the soil (e.g. Gorham 1991, Page *et al.* 2002, Zedler and Kercher 2005, Page *et al.* 2011): boreal and subarctic peatlands alone comprise *circa* 30 % of the global soil C pool (547 Pg C) (Yu 2011). In natural peatlands, C sequestration results from the deposition of plant biomass to form raw humus in the acrotelm (oxic surface peat layer) and its subsequent transition to long-term storage in the catotelm (anoxic stratum below water level fluctuation range) (e.g. Clymo 1984). However, wide-scale degradation caused by land-use threatens the C pool of peatlands on a global scale (Minkkinen *et al.* 2008, Oleszczuk *et al.* 2008, Hooijer *et al.* 2010, Simola *et al.* 2012, Moore *et al.* 2013, Pitkänen *et al.* 2013). For example, *ca.* 15 000 000 hectares of peatlands have been drained for forestry in the boreal region alone (Minkkinen *et al.* 2008), possibly causing emissions of C to the atmosphere especially due to increased aerobic decomposition (Silvola *et al.* 1996, Hooijer *et al.* 2010). There are contradictory results on effects of drainage to peat C storage, however. Indeed, both negative and positive C balances are reported in drained boreal peatlands (Minkkinen and Laine 1998, Minkkinen *et al.* 2008, Ojanen *et al.* 2010, 2012, 2013, Lohila *et al.* 2011, Simola *et al.* 2012, Pitkänen *et al.* 2013). A part of the controversy may be due to different time scales; many studies have measured current C balance (e.g. Lohila *et al.* 2011) while others have estimated total changes in C storage (e.g. Simola *et al.* 2012) during longer drainage periods. In addition, variation between different peatlands can be large (e.g. Ojanen *et al.* 2012), while the number of independent study sites is often small.

As a response to the overall degradation of peatlands, and potential effects on global C balance, a globally increasing trend towards peatland restoration has arisen (Parish *et al.* 2008, Erwin, 2009, Ramchunder *et al.* 2009, Thiele *et al.* 2009, Worrall 2009, European Commission 2011). In general, restoration actions have hierarchical aims. Re-establishment of natural water table level is expected to restore abiotic conditions needed to restart succession towards original species communities (Hobbs and Harris 2001, Gorham and Rochefort 2003). The regained high water table level and development of typical peatland vegetation is expected to result in the re-establishment of the natural acrotelm-catotelm stratification of peat and restart the original ecosystem function of C sequestration. Despite the recognized and increasing importance of peatland restoration the efficiency of the current methods in re-establishing the C sequestration function has not been studied much and research is concentrated mainly on peat mining areas (e.g. Cagampan and Waddington, 2008, Soini *et al.* 2010, Waddington *et al.* 2010, Moreno-Mateos *et al.* 2012, Strack and Zuback 2013) that globally cover only a minor fraction of degraded peatlands (Strack *et al.* 2008). In general, the recovery of biological structure has been found to precede the recovery of ecosystem functions in restored wetland ecosystems (Moreno-Mateos *et al.* 2012). Still, the relationship between the recovery of vegetation and C sequestration in boreal peatland ecosystems remains unexplored.

In this study, we address three questions related to degradation and restoration of C sequestration function of boreal peatlands. First, what is the long term effect of drainage on surface peat C storage? Second, are restored peatlands recovering the targeted pristine ecosystem function and structure? And third, is the recovery of the structure needed for the recovery of ecosystem function? To answer our questions, we examined the recovery of the surface peat, i.e. the peat forming acrotelm layer including the living plant biomass, where the C fixation, most of the decomposition of organic C, and transition of biomass to anaerobic storage layer catotelm take place (Clymo 1984, Francez and Vasander 1995, Gunnarsson *et al.* 2008). We first quantify the change of surface peat C storage due to drainage by using the C to ash and C to Al ratios of surface peat in pristine peatlands to determine the expected C mass for the surface peat of drained peatlands. We use the C to ash and C to Al ratios for the comparison, because drainage results in compaction of the surface peat, making the comparison of absolute C per volume values flawed for measuring the change in the C storage. Then, to determine whether ecosystems regain their original peat growth function after restoration, we compare the peat growth rate between pristine, drained and restored peatlands using data of age and rooting depth of pine seedlings. We also determine the recent apparent rate of C accumulation (RERCA) in surface peat of restored sites and compare it to the RERCA of pristine sites. To determine the recovery of the ecosystem structure, we compare the similarity of plant community composition between pristine, drained and restored peatlands. Finally, we discuss the question if recovery of the original ecosystem structures is needed for the ecosystem functions to recover.

Materials and methods

Study sites

For the study we selected 38 sites on previously unstudied *Sphagnum* peatlands in southern Finland, Europe. The average annual precipitation of the region is 675 mm and the annual mean temperature is +3.4^{o}C. We selected the sites so that they fell into one of four categories (treatments): i) drained peatlands (n = 9), ii) previously drained and restored 3-7 years before the study (hereafter restored 5 years ago, n = 9), iii) previously drained and restored 9-12 years before the study (hereafter restored 10 years ago, n = 10) and iv) pristine peatlands (n = 10) (see example pictures of the categories in Supporting information: Appendix 1). Distances between the study sites ranged from 200 m to 150 km. The study sites were independent from one another in their surface water flow (based on topographic data and field observations). Based on close examination of old and new aerial photographs accompanied with field observations, the original vegetation types and tree stands of the disturbed sites were roughly similar to those of the chosen pristine control sites. All of the disturbed sites were drained for forestry by the state *ca.* 40 years before the study (1960-1970) with ditch interval of 30-50 meters. Drainage had changed the peatlands' hydrology mainly by lowering the water table and altering the water chemistry (More detailed description of the hydrology of the studied peatlands can be found in Haapalehto *et al.* 2014). Tree growth (mainly *Pinus sylvestris* and *Betula pubescens*) had variably increased after the drainage. In 1980s some of the sites were designated to conservation with a subsequent decision to restore the drained sites within the conservation areas. Information on restoration year was available from the habitat database of the state owned land. Restoration measures included filling in the ditches with peat excavated near the ditches, construction of peat dams and removal of trees in cases where drainage had significantly increased tree growth. The amount of trees removed varied slightly so that all the restored sites had more or less the same tree cover in the end, mimicking the pre-disturbance tree cover determined from aerial photographs. The restoration measures may be considered rough and straightforward in the sense that they relied on natural re-establishment of populations of the target species from nearby relict sources. This means that the often laborious and costly transplantations of species or fine scale habitat engineering (e.g. for individual target species) were not applied. Restoration of the sites was conducted by Natural Heritage Services of Metsähallitus (governmental institution responsible for management of conservation areas).

At the pristine sites, vegetation was dominated by common peatland plants typical of oligotrophic lawn-level peatland (peatlands dominated by the intermediate surface between drier hummocks and the wettest level) vegetation, such as *Eriophorum vaginatum*, tall sedges (e.g. *Carex rostrata*) and *Sphagnum* mosses (*Sphagnum angustifolium*, *Sphagnum fallax* and *Sphagnum fuscum*). At the drained sites, common forest plants, such as the dwarf shrubs *Vaccinium myrtillus*, *Vaccinium uliginosum*, *Vaccinium vitis-idaea*, *Rhododendron tomentosum* and *Betula nana* dominated the field layer, *Pleurozium schreberi* along with *Sphagnum* mosses (*Sphagnum angustifolium*, *Sphagnum magellanicum* and *Sphagnum russowii*) being the most common species in the ground layer.

Sampling of surface peat and vegetation

The sampling was conducted using a systematic design of 15 1-m² vegetation plots in a 10 × 20 meter area at each site. The sampling plots were placed in three transects running parallel to the ditch line 5, 10 and 15 meters from the ditch. The plots were located at 4 meter intervals along each transect forming a grid (Fig. 1). The location of the grid was randomized within the area of the focal habitat type at each site.

We focused our peat sampling on the uppermost 20-25 cm surface peat layer (hereafter simply surface peat) that cover the main range of water table level fluctuation in natural sites and most of the surface layer exposed to increased aeration in drained sites. Six cores of surface peat were collected with a side-cutting box sampler (sampler area: 8.3 × 8.4 cm) at each site. Samples were collected close to the vegetation plots, at the 5, 10 and 15 m distances from the nearest ditch in drained and restored sites (three samples at each distance). Samples were divided into two segments (0-10 cm and 10-20 cm layers from the surface) in the field, sealed into plastic bags and stored frozen prior to analyses. The force needed to employ the box sampler typically causes compaction of surface peat samples and avoiding this requires careful operation in the field (Pitkänen *et al.* 2011). In addition, we carefully examined the peat samples in the laboratory and adjusted for the compaction by measuring the sample dimensions after reconstructing the erect posture of the *Sphagnum* mosses, whenever an evident compaction was observed. The corrected average depths of the pristine and drained 0-10 cm samples were 14.2 cm (+-2.1 SD) and 10.7 cm (+-1.0 SD), respectively.

Vegetation was sampled at the 15 1-m^2 vegetation plots at each site (Fig. 1). From each plot we recorded relative abundance as a % cover for all plant species based on visual estimation.

Carbon loss

To answer our first question about the effect of drainage on the amount of C in the surface peat, we estimated the C loss from the 0-20 cm peat samples during the approximately 40-year drainage period relative to the pristine sites. Decomposition leads to the loss of C, increase of bulk density and enrichment of ash content of peat. We calculated the expected mass of C for the surface peat samples of drained peatlands by multiplying their observed ash content with the average C to ash ratio obtained from the pristine peatlands (see Appendix 2 for detailed description of the chemical analyses). The loss of C (ΔC_{ASH}) for each sample was then estimated as the difference between the expected and the observed mass of C of the samples from the drained sites (Grønlund *et al.* 2008, Leifeld *et al.* 2011). The estimated C loss per m^2 for each drained site was then calculated by multiplying the average C loss of each sample with 1/sampler area (m^2) (see Appendix 3). However, drainage may result in increased leaching of mineral cations (e.g. Prevost *et al.* 1999, Pitkänen *et al.* 2013), thus reducing the mass of ash. This causes a potential bias in ΔC_{ASH} estimates towards underestimation of C loss as the decrease of mineral concentration results in higher C to ash ratio. Among the main cations, Al^{3+} is retained relatively strongly at cation exchange sites (Wieder *et al.* 1988) due to its trivalent charge and high charge to size ratio, i.e. ionic potential. Therefore, we modified the method by using aluminum concentration in place of total ash to yield ΔC_{AL}. Our modification revealed considerably higher estimates of C loss than ΔC_{ASH} (see results). For comparison, we also calculated ΔC estimates using other elements (Appendix 3). These calculations indicated the lowest ΔC in relation to readily leaching cations (Mn < Mg < Fe < Ca) and intermediate ΔC in relation to main nutrients (K < N < P) that are effectively retained by living organisms in the surface peat. The residual ash concentration (total ash – known mineral elements) most likely represented mainly silica (Si) and it indicated the highest ΔC. However, we did not use residual-ash in our estimate of C loss because of the uncertainty of its exact mineral constituents.

Surface peat growth rate

To study the surface peat growth rate we constructed empirical age-depth models of the surface peat layers of the study sites by the pine method (e.g. Borggreve 1889, Ohlson and Dahlberg 1991) and estimated the annual growth rate of the surface peat for each site. For this we collected 25-35 small (< 1.5 m) Scotch pines at each study site at the 10 x 20 meters sampling area. In cases where there were not enough pines in the actual sampling area, we extended the collection to similar area in the immediate vicinity. At each site, half of the pines were collected from hummocks and another half from lower-lying surfaces. At one 5 years ago restored site no pines were found and the site was excluded from this analysis. We determined the vertical distance from the root collar (root to shoot transition) of the trees to the peat surface to estimate rooting depth of each pine. The ages of pines were determined by counting the annual rings close to the root collar under a stereomicroscope. We then calculated the apparent annual vertical growth of surface peat as a linear regression coefficient between the rooting depth and age of the pines for each site and used analysis of variance (ANOVA, IBM SPSS Statistics 20) and appropriate post-hoc test to compare the coefficients between the treatments. We limited the age-depth data to the first 10 years, where the age-depth curve was close to linear. For further linearization, the data were first log transformed and regression curves were forced to pass through the origin, i.e. zero peat depth corresponded to zero age.

Recent apparent rate of carbon accumulation in surface peat

We determined the recent apparent rate of carbon accumulation (RERCA) in surface peat after restoration and for comparable period for the pristine sites. By definition, RERCA only includes C bound in peat above a dated horizon and it is the net result of biomass production and decomposition (g C m^{-2} yr^{-1}). Since decomposition continues with material of all ages, the accumulation rate will be the lower the older the material is and thus the accumulation pattern of C is nonlinear with time. However, within the 3-12 year time-scale of our study, we observed that the age-depth pattern and C accumulation was still nearly linear. Since the post-restoration time period varied among the sites in the 5 and 10 years ago restored categories, we used here linear regression to model the cumulative C mass with time (years since restoration) and interpreted the slope as RERCA, i.e. increase of C store with one year increase of age (g m^{-2} yr^{-1}). The post-restoration peat was separated from the older layer that represented the drainage period in the laboratory. The separation was based on visual inspection: there was typically a clear difference in the degree of humification and typical presence of bark and needles of trees and remains of species typical to drained peatland forests such as *Pleurozium schreberi* and *Vaccinium myrtillus*. Additionally, the post restoration peat layer was verified in all sites and with most of the individual cores by dating with annual increments of *Polytrichum strictum* and *Eriophorum vaginatum*, and in few cases *Trichophorum cespitosum* (data not shown). The post-restoration surface peat samples were dried to constant weight at 70 °C for the determination of dry weight. The mass of C in each sample was calculated by multiplying dry mass with the measured C concentration. The linear regression was forced through origin (i.e. zero peat depth corresponded to zero age) and the slope was tested against the expected slope (Extra sum-of-squares F test, GraphPad Prism 5 for Windows), i.e. the average 10-year RERCA of pristine sites, which was calculated according to the depth of 10-year old strata based on the pine method (see above).

Plant community composition

To study changes in vegetation, which is an essential element of ecosystem structure and vital for the recovery of the C sequestration function, we compared the similarity of the composition of the plant communities between drained, 5 and 10 years ago restored and pristine sites. For each of the 38 study sites we compiled plant community samples by calculating average relative abundances for each plant species over the 15 1-m^2 vegetation sampling plots. We used average values for each site, because we wanted to assess general patterns of vegetation with respect to surface peat growth rate and C sequestration. Both the identities and the abundances of the species were apparently affected by drainage as well as restoration. Therefore, we used Bray-Curtis community similarity measure considering both species identities and relative abundances (e.g. Magurran 2004). Effect of treatment on plant community composition was studied by comparing the Bray-Curtis community similarity of the study sites within and between the treatment groups with Non-Parametric MANOVA in PAST 2.17b (PERMANOVA, Anderson 2001, McArdle and Anderson 2001). For visual inspection we performed NMS (Non-metric Multidimensional Scaling) ordination in PCORD 5, using again the Bray-Curtis similarity as the distance measure, random starting points, 250 runs with the real data and 500 iterations for the final result. For the main purpose and comparisons of the current paper the vegetation analysis was kept simple, while a more detailed analysis of the plant community changes of these sites is under preparation.

Results

Carbon loss

The comparison of expected and observed C masses in surface peat of drained sites showed substantial loss of C due to drainage. The estimates for average C loss from the 0-20 cm surface peat samples were ΔC_{ASH} 5172 g m^{-2} (SE 2339 g m^{-2}) and ΔC_{AL} 6714 g m^{-2} (SE 2908 g m^{-2}). The focal peatlands were drained *ca.* 40 years prior to the study and thus the average per annum C loss (for the 40 years' drainage period) were 129.3 g m^{-2} (SE 58.5) and 167.8 g m^{-2} (SE 72.7) for ΔC_{ASH} and ΔC_{AL} (Fig. 1) respectively. In CO_2 equivalents these estimates equal to 474.5 and 615.8 g CO_2e m^{-2} yr^{-1}. See Supporting information for derived ΔC estimates for the other elements (Appendix 3) and total mineral concentrations (Appendix 2).

Surface peat growth rate

According to the peat age-depth models peatlands in all treatments accumulated some peat (Appendix 4), but there was a significant difference in the net growth rate (mm yr^{-1}) of surface peat among the treatments (ANOVA $F_{3, 33} = 6.06$, $p = 0.002$). Growth rate at the drained sites was significantly retarded when compared to the pristine, 5 years ago and 10 years ago restored sites (LSD pairwise comparison, for all $p < 0.003$), while there were no differences between the pristine and the 5 years or 10 years ago restored sites (LSD pairwise comparison, $p > 0.760$ for both). Full untransformed age-depth data for all treatments is depicted in appendix 4.

Recent apparent rate of C accumulation in surface peat

We observed a roughly linear rate of C accumulation into surface peat of restored sites after restoration. According to the slope of the regression model the regenerated surface peat layer of restored sites accumulated C with an average rate of 116.3 g m^{-2} yr^{-1} for the 3-12 years period (SE 12.7). In comparison, the RERCA for 10 years period for pristine sites was 178.2 g m^{-2} yr^{-1} (SE 13.3). The difference of the slope was statistically significant against the null-hypothesis of no deviance from the pristine RERCA ($F = 23.73$, $P < 0.001$), but the variation of the accumulation rate at the restored sites was large, and some of the restored sites had even higher post-restoration surface peat C accumulation than predicted by the pristine reference estimate (Fig. 2).

Plant community composition

The ecosystem structure measured as plant community composition was significantly different between the treatments (PERMANOVA $F = 4.719$, $p <$

0.001). Most substantial differences were between pristine sites and all the other sites i.e. drained, 5 years ago restored, and 10 years ago restored sites (pairwise comparisons of pristine to all others, for all $p < 0.006$). There was no difference in the community composition between drained and 5 years ago restored sites (pairwise comparison $p = 0.231$) but the sites restored 10 years ago already showed dissimilarity to the drained sites (pairwise comparisons $p = 0.024$). By visual inspection, the effect of the treatment can be seen on Axis 2 of the ordination (Fig. 3).

Discussion

Our analyses indicate that drainage-induced degradation of the peatland ecosystems results in significant net loss of carbon from surface peat when compared to the undisturbed state of the ecosystem. However, we also learned that the straightforward restoration by filling ditches can jump-start the ecosystem function of surface peat growth, which is an essential step towards re-establishing the long term carbon storage function of peatland ecosystems: already within few years after restoration the surface peat growth rates had recovered on average close to the level of pristine peatlands. Furthermore, the rate was essentially maintained over the post-restoration time span covered by our data. Elsewhere, the recovery of ecosystem structure has been suggested as a prerequisite to the recovery of ecosystem functions in wetlands (Moreno-Mateos *et al.* 2012). However, our analysis suggests that the employed restoration methods were successful in returning the surface peat growth function although the original ecosystem structure (plant community composition) was not yet recovered. This suggests a relatively loose relationship between these structural and functional ecosystem components. On the other hand, C sequestration rate to the newly formed surface peat was lower than at the pristine sites suggesting that some functions of these peatlands may need more profound recovery of the original structure and conditions to reach the targeted level.

There are several earlier estimates on C balance of similar drained ecosystems as studied here, but there is still no consensus on whether drained boreal peatlands function as sinks or sources of C (e.g. Silvola *et al.* 1996, Minkkinen and Laine, 1998, Minkkinen *et al.* 2008, Hooijer *et al.* 2010, Ojanen *et al.* 2010, Lohila *et al.* 2011, Simola *et al.* 2012, Pitkänen *et al.* 2013, Ojanen *et al.* 2012, 2013). Although environmental conditions undoubtedly add variance on drained peatlands' C balance (e.g. Ojanen *et al.* 2010, 2012), it appears likely that the lack of consensus stems in part from differences in approach and methodology. Peat core analyses and gas exchange measurements focus on different temporal scales: the gas exchange measurements focus on the real-time gas exchange of the ecosystem, and thus measure only the contemporary fluxes of the disturbed ecosystem. Peat core analyses, on the other hand, cover the cumulative effects on

C storage since the beginning of the disturbance (e.g. Simola *et al.* 2012). We observed considerable loss of C from surface peat due to drainage, while some gas exchange studies have indicated only moderate C loss or in some cases even slightly positive net C balance at similar sites (e.g. Minkkinen *et al.* 2008, Lohila *et al.* 2011, Ojanen *et al.* 2010, 2012, 2013). We suggest that when the aim is to understand the cumulative long term effects of disturbance on C storage of peatland ecosystems, the peat core analyses with undisturbed controls for calculating the net effects should be the preferred methodology. However, when the aim is to capture the current situation then gas exchange measurements can be preferable. In both cases, it is imperative to include also undisturbed reference sites into the study design to understand the net effects of degradation or restoration. Much of the extant literature has not done so, and thus it is difficult if not impossible to draw conclusions of the real net effects of the disturbances on global C balance. It should also be noted that our estimates derived from the surface peat layer only are not directly comparable to studies examining complete peat profiles. However, they are in line with recent studies that have observed reduced accumulation of biomass in the surface peat (Pitkänen *et al.* 2012) and a large net loss of C due to drainage from entire peat column (Simola *et al.* 2012, Pitkänen *et al.* 2013).

The reduced surface peat increment rate observed at the drained sites when compared to pristine sites is in line with a recent study, where a significant reduction in the biomass accumulation induced by increased decomposition was found in surface peat of forestry drained peatlands (Pitkänen *et al.* 2012). The growth of the surface peat layer is a prerequisite for subsequent deposition of biomass to lower anaerobic peat layers and long-term C accumulation in peatlands. Therefore, the re-establishment of the surface peat accumulation rate to the targeted level only a few years after restoration suggests a surprisingly rapid recovery of an important peatland ecosystem function. On the other hand, a small difference was still observed between the C accumulation rates of restored and pristine sites (see also Tolonen and Turunen 1996 for pristine RERCA over 35 years). C sequestration is a combination of growth and decomposition of vegetation, both affected by the hydrological conditions. Thus, the relatively small difference in the annual C sequestration rate between pristine and restored sites is probably partly due to a time-lag in the response of the plant community to new selection pressures set by the restoration actions. Indeed, relatively large annual variation in C sequestration is likely during the first years of post-restoration vegetation succession. This time period is characterized by initial reduction of forest vegetation and increasing domination of opportunistic rapidly growing early colonists like *Eriophorum vaginatum* followed by a state of *Sphagnum* mosses domination (e.g. Haapalehto *et al.* 2011). On the other hand, if the conditions for slow decomposition are effectively restored, any vegetation that is able to endure the physical conditions, should contribute to the C sequestration and peat accumulation with some variation caused by differences in the specific traits of plant species (e.g. De Deyn *et al.* 2008).

Our analysis of plant community composition suggests only limited recovery 10 years after restoration. Despite the communities still being distinct from the pristine communities, some post-restoration recovery was already taking place: the communities of the sites restored 10 years ago had evolved towards the pristine communities, while the communities of the sites restored 5 years ago were still indistinguishable from the drained communities. Our findings of partial recovery of plant community composition after peatland restoration are in line with earlier case-studies (Haapalehto *et al.* 2011, Hedberg *et al.* 2012). The dissimilarity in the plant community composition of pristine and restored sites is mainly due to i) some forest species still remaining in greater abundance than in pristine sites, ii) some pristine peatland species occurring in greater abundance than at pristine sites due to their ability to survive through the drainage period and to exploit the post restoration enhanced conditions, and iii) some pristine peatland species being absent from the restored sites due to local extinctions during drainage period and dispersal and/or re-establishment limitations (see e.g. Haapalehto *et al.* 2011, Hedberg *et al.* 2012). It should be noted, however, that the plant community dissimilarity occurring 10 years after restoration does not mean failure of restoration but only that it quite expectedly takes longer than 10 years for the structure of this ecosystem to fully recover (Jones and Schmitz 2009, Hedberg *et al.* 2012, Moreno-Mateos *et al.* 2012).

Potential ecosystem level consequences of biodiversity loss are gaining increasing attention (Hector and Bachi 2007, Convention on Biological Diversity 2010, Cardinale *et al.* 2012, Hooper *et al.* 2012, Reich *et al* 2012). In ecological restoration the question is most tangible: how much of the original structure or community composition needs to be recovered in order to regain the original ecosystem functions (Bradshaw 1984, Cortina *et al.* 2006)? Plasticity in the relationship of biodiversity and ecosystem functioning is likely although not yet well understood (Cardinale *et al.* 2012, Hooper *et al.* 2012, Naeem *et al.* 2012, Reich *et al.* 2012). From the practical restoration perspective the magnitude of plasticity in this relationship is important to understand because it directly relates to the net costs of restoration actions. Indeed, it is likely that the stronger and more causative the relationship of ecosystem functions and certain community composition or biodiversity *per se*, the more complicated and costly are the actions needed to reach the ecosystem level restoration targets. Our results suggest considerable plasticity in the studied structure-function relationship of peatland ecosystems as a valuable ecosystem function of surface peat accumulation was recovered already with minor recovery of the original composition of the plant community (for restored cut-away peatlands and C accumulation see Soini *et al.* 2010, Waddington *et al.* 2010). It appears also that while this kind of a plastic ecosystem function could, indeed, be re-established with minor recovery of ecosystem structure, this did not result in similar recovery of C accumulation in surface peat layer. Not surprisingly, this suggests that evolution of the relationship between one ecosystem structural component (plant community composition) and two ecosystem functions may differ even between closely linked functions (peat growth rate and C accumulation rate). Nevertheless, there is still work to be done to fully understand the magnitude of the recovery of the original structure needed for the full recovery of ecosystem multifunctionality (Lucchese et al. 2010, Hector and Bachi 2007, Montoya *et al.* 2012, McCarter and Price 2013).

Being aware of recent large scale international targets for restoration (Aichi targets of Convention on Biological Diversity 2010, Maron *et al.* 2012), it is interesting and necessary to consider also the economic value of restored ecosystems (Bullock *et al.* 2011, Menz *et al.* 2013). Estimating the economic value of C sequestration at restored peatlands is a relatively new idea (Nelson *et al.* 2009, Alexander and McInness 2012, Kettunen *et al.* 2012, Russi *et al.* 2013) and still far from straightforward (see e.g. Tanneberger and Wichtmann 2012). Nevertheless, the results like ours including the calculation of C fixed by the studied peat layer can be used to evaluate the economic value related to the accumulated C in the recovering ecosystem. The average and the highest prices in the voluntary C market for comparable terrestrial C projects in 2010 were 6 and 136 USD per credit (t CO_2e (carbon dioxide equivalents)), respectively (Peters-Stanley 2011, for future price assessments see also European Commission 2008, ten Brink *et al.* 2011). We estimated that on average 116.3 g C $m^{-2}yr^{-1}$ accumulated into the surface peat during the 3-12 years' post-restoration time period corresponding to 426.4 g $m^{-2}yr^{-1}$ of CO_2e. Although our estimates cover only the surface layer and a relatively short time span, we can estimate the market value for the C sequestered into the accumulated layer of surface peat. Thus, with the 2010 prices, the surface peat of the restored sites sequesters C at a rate corresponding to 26 – 580 USD ha^{-1} $year^{-1}$ for the 3-12 years' post-restoration time period (note that the accumulation of peat and input of C into long-term storage is not linear in time due to decomposition (e.g. Clymo 1984)). These are not trivial numbers. For example, in Finland alone there are *ca*. 1 million hectares of peatlands drained for forestry where the drainage has not been economically profitable in terms of increasing the timber growth as intended. According to our results and the 2010 C market prizes, the market value of surface-peat C of 1 million hectares of restored boreal *Sphagnum* peatlands would amount between 26 to 580 million USD annually over the first decade after restoration.

While considering the potential market values offers impressive figures, it should be noted that these figures show only the potential of the economic value of these restored ecosystems as they are and the true market value or the net C accumulation effect of restoration action could only be estimated by comparing net ecosystem C balance before and after restoration (see e.g. Kimmel and

Mander 2010). For example, with these data we cannot tell how restoration influences the lower peat layers. We find that long-term experiments e.g. on the changes in greenhouse gases (for restoration of cut-away peatlands see e.g. Waddington and Day 2007, Strack and Zuback 2013) and fluvial DOC fluxes (e.g. Moore *et al.* 2013) as well as the development of tree stands (e.g. Ojanen *et al.* 2013) are also needed to reliably estimate if peatland restoration may really produce tradable C-related ecosystem services. Our rough and unrealistic calculation above, hopefully, attracts scientific and societal interest to establish such studies.

Here we studied peatlands up to 12 years after restoration. This can be regarded as a relatively long time scale in ecosystem ecology studies (e.g. Reich *et al.* 2012). However, from the perspective of peatland ecology it is only a moment considering that it has taken several millennia for the boreal peatlands to accumulate their remarkable C storages. With this in mind, achieving recovery of peat growth with such rough and straightforward ecological restoration methods already within a decade post restoration certainly serves as a jump-start for the ecosystem functioning. Although it may well take several decades before original species communities are achieved (if they ever will be) it is very promising that it is not an insurmountable task to restore the needed amount of the original community composition to restart at least some of the essential ecosystem functions.

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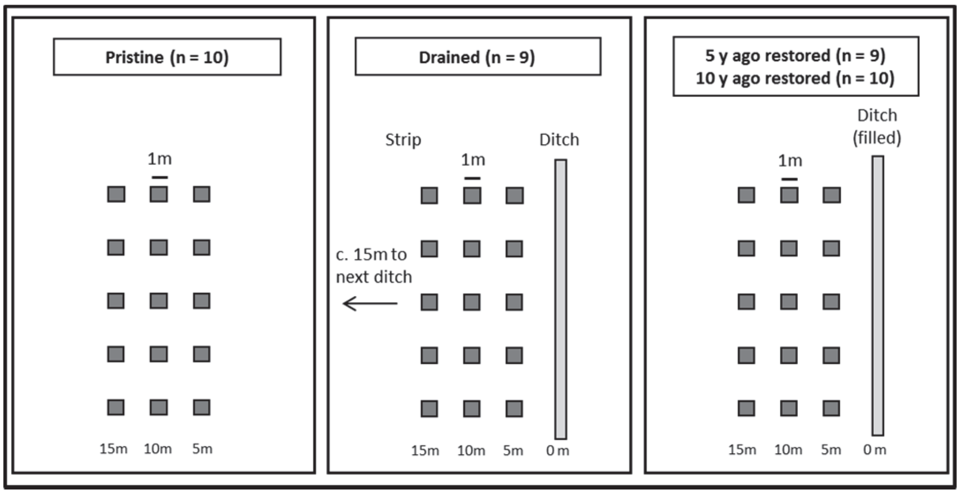

FIGURE 1 Experimental set up at pristine, drained and restored sites. Each grey square represents a 1-m^2 vegetation plot. At each site 2 peat core samples were taken at each distance to ditch (5, 10, and 15 m) close to the vegetation plots at both ends of each distance transect (i.e. altogether 6 peat core samples per site).

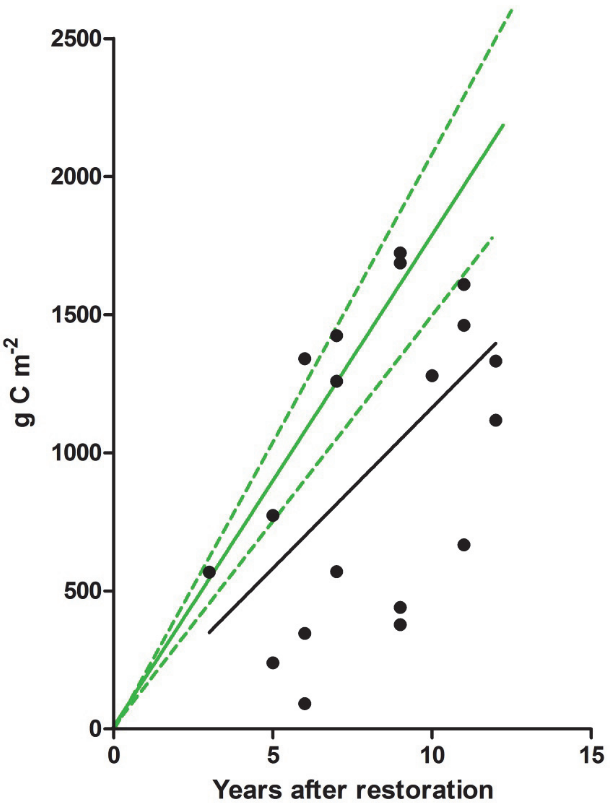

FIGURE 2 The mass of C (g m^{-2}) in the surface peat accumulated after restoration with the site-specific post-restoration years (black circles). Black line depicts the linear regression (cumulative C mass over time since restoration) fitted to the restored site's data (y = 116.3x). The green linear line from origin goes through the 10-year recent apparent rate of C accumulation (RERCA) of pristine sites (y = 178.2x, hatched lines 95% CI).

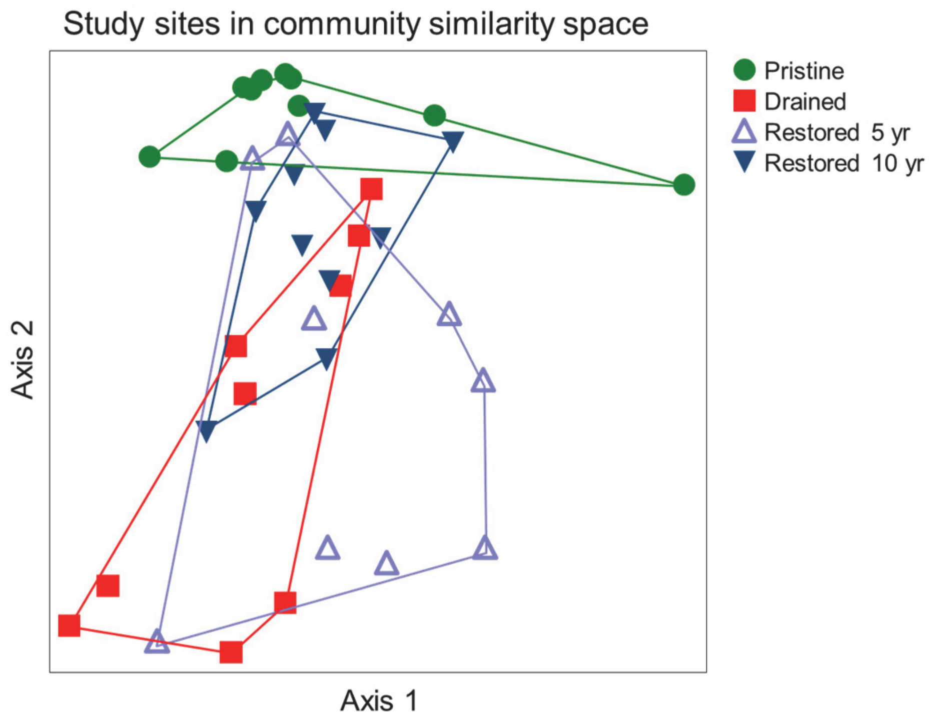

FIGURE 3 Vegetation community similarity between the pristine (filled circles), drained (filled square), 5 years ago restored (hollow triangle) and 10 years ago restored (filled triangle) study sites presented in an ordination space (NMS ordination with Bray-Curtis distance measure, 2-dimensional solution, stress = 11.22). Most distinctive differences between treatments are shown on the Axis 2 where the distribution of drained and 5 years ago restored sites are nearly identical and 10 years ago restored sites show a trend of clustering closer to the pristine sites, which are strongly clustered on the upper part of the Axis 2.

Supplementary Material

Appendix 1. Example photographs of the study sites of different treatments.

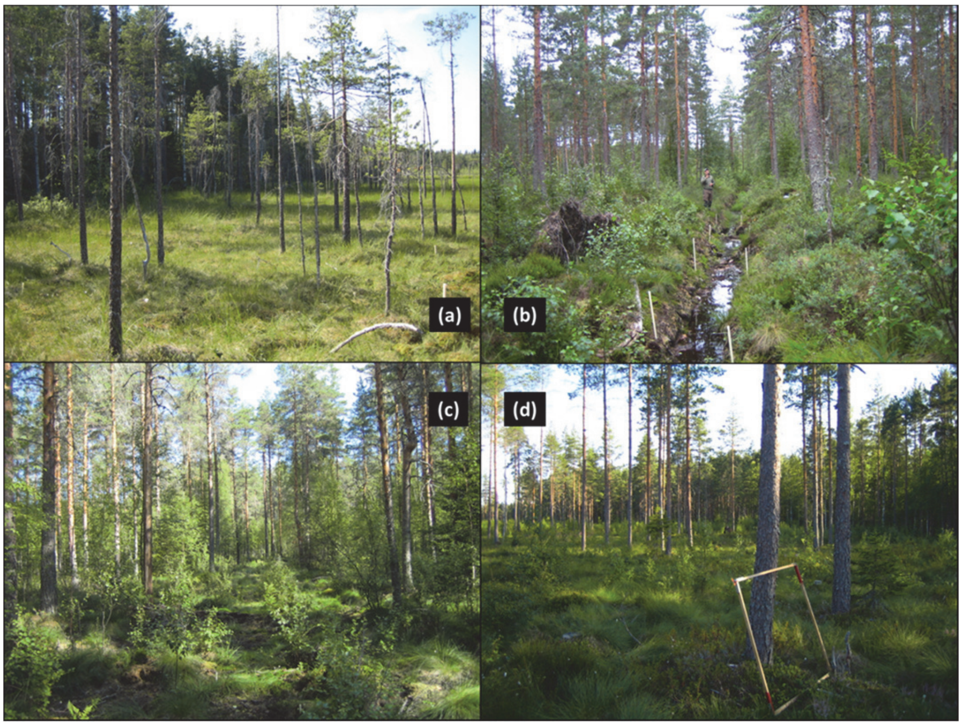

FIGURE S1 Picture of the study sites with different treatments. Panels: a) pristine, b) drained, c) restored 5 years ago, and d) restored 10 years ago.

Appendix 2. Chemical analyses of the peat samples and mean element concentrations and proportions for surface peat samples for pristine and drained sites.

Experimental

Chemicals

Ultra pure water was obtained by passing distilled water through an ELGA Elgastat Maxima (Elga Ltd, UK) and it was shown to have a resistivity greater than 18,2 MΩ/cm. The 1000 mg L^{-1} ICP multi-element standard solution IV (CertiPur, Merck, Darmstad, Germany) was used for determination of Al, Ca, Fe, K, Mg and Mn. For the determination of phosphorous the 1000 mg L^{-1} of phosphorous stock solution was made by appropriate weighing of KH_2PO_4 (pro analysi) also supplied by Merck. Acetanilid (pro analysi, Merck) was used for performance check of CHN-analyzer. 65 % nitric acid (Sigma Aldrich, Steinheim, Germany) and 30 % hydrogen peroxide (Riedel-de Haën, Seelze, Germany) was used in digestion of samples.

Analysis of peat samples

About 60 g of peat samples were collected regularly from each layer of the peat clods and all foreign matter, such as sticks, roots and stones were removed and also soft lumps were crushed with spatula. The samples were dried at 105 °C for 18 h in a heating oven (Heraeus Thermo Scientific, Langenselbold, Germany). After the drying the samples were milled using Franz Morat A 70 (Eisenbach, Germany) milling apparatus. About 7 g of the milled sample was ashed in a programmable furnace (Naber Industrieofenbau, Lilienthal, Germany) at 400 °C for 1h and then at 550 °C for 4h. Both milled and ashed samples were sealed in plastic bags in exigator. Moisture content (%), ash content (%) and organic matter content (%) (100 – moisture-% - ash-%) of the peat samples were calculated.
About 200 mg of the ashed peat samples were digested in sealed Teflon vessels in a Milestone microwave oven (Milestone Ethos Touch, Sorisole, Italy) after addition of 4 ml HNO_3 (65%) and 1 ml H_2O_2 (30%). The digestion program consisted of two steps: room temperature to 205 °C in 10 min and then heating at 205 °C for 10 min. After the digestion samples were filtered through Whatman No. 42 filter paper (Maidstone, UK) and then made up to 50 ml with 2% (v/v) HNO_3. The sample solutions were transferred into plastic screw-top bottle for storage in a refrigerator.

The concentration measurements of the selected elements in peat samples were performed with a PerkinElmer (Norwalk, CT, USA) model Optima 4300 DV inductively coupled plasma optical emission spectrometer (ICP-OES). A Scot-type double-pass spray chamber and cross-flow nebulizer was used throughout. The measurements were done using a nebulizer flow of 0.8 L min^{-1}, auxiliary gas flow of 0.2 L min^{-1}, plasma gas flow of 15 L min^{-1}, plasma power of 1300 W and sample flow of 1.5 L min^{-1}. The plasma was radially viewed. The concentrations of calibration standards for all the measured elements were 5, 15, 25, 35, 45 and 100 mg L^{-1} and they were made up with 2 % (v/v) HNO_3. The wavelengths used, linearity, limit of detection (LOD) and limit of quantification (LOQ) are shown in Table 1.

Table S1. The calibration parameters in ICP-OES measurements for each element analyzed.

Element	Wavelength (nm)	R^2	LOD (mg L^{-1})	LOQ (mg L^{-1})
Al	396.153	> 0.996	0.009	0.03
Ca	317.933	> 0.997	0.02	0.06
Fe	238.204	> 0.997	0.04	0.13
K	766.490	> 0.997	0.02	0.08
Mg	285.213	> 0.998	0.003	0.01
Mn	257.610	> 0.997	0.003	0.009
P	213.617	> 0.998	0.4	1.3

Carbon, hydrogen and nitrogen content of about 5 mg of dried and grinded peat samples were measured with a CHN analyzer (VARIO EL III Elementar Analysensystem, GmbH, Hanan, Germany).

Table S2. Mean element concentration (mg/g dry weight) and proportion (%) of N, C, H, and Ash for 0-10 cm and 10-20 cm surface peat samples for pristine and drained sites.

Pri		**Al**	**Ca**	**Fe**	**K**	**Mg**	**Mn**	**P**	**N**	**N (%)**	**C (%)**	**H (%)**	**Ash(%)**
0-10	Mean	0.378	2.934	1.871	1.590	0.843	0.159	0.470	6.671	0.667	46.987	5.905	2.764
	SD	0.197	1.237	1.200	0.430	0.255	0.059	0.118	1.305	0.130	0.375	0.044	0.806
10-20	Mean	0.716	3.026	2.256	0.294	0.694	0.031	0.423	9.442	0.944	47.591	5.892	3.292
	SD	0.319	1.429	1.113	0.086	0.272	0.024	0.051	2.565	0.256	0.417	0.097	0.993
Dra													
0-10	Mean	0.790	3.500	1.341	1.146	0.797	0.143	0.755	11.459	1.146	50.103	5.682	4.866
	SD	0.362	0.918	0.841	0.400	0.159	0.085	0.149	2.307	0.231	0.693	0.095	2.340
10-20	Mean	1.406	2.521	2.231	0.307	0.448	0.010	0.623	14.091	1.409	49.890	5.876	5.130
	SD	0.594	0.966	0.662	0.083	0.172	0.009	0.163	3.663	0.366	2.277	0.196	1.577

Appendix 3. Calculation of ΔC estimates and calculated ΔC estimates for surface peat with different minerals (or ash) as a comparison.

ΔC estimates (g m^{-2}) for surface peat (0-20 cm) of drained peatlands calculated as a difference between observed and expected carbon mass. The expected carbon mass is based on the observed mass of a mineral element in the drained peatland sample (Mi_d) multiplied by average ratio of carbon to mineral concentration observed in pristine peatlands (C_p/Mi_p). The observed carbon mass is the product of dry weight (DW_d, mass) and carbon concentration (C_d) of the sample. Value per m^2 for 0-20 cm peat depth is derived by multiplying the result per sample with 1/sampler area (A_s). Negative values in the table indicate relative loss of mineral element and positive values indicate relative loss of carbon.

$$\Delta C_{0-20}(\mathrm{g\ m^{-2}}) = \left(\frac{C_p}{Mi_p} \times Mi_d - DW_d \times C_d\right) \times \frac{1}{A_s}$$

Table S3. ΔC estimates (g m^{-2}) for surface peat (0-20 cm) with different minerals (or ash) as a comparison.

Mn		**Mg**		**Fe**		**Ca**	
Mean	-2467.7	Mean	-2301.4	Mean	-1442.3	Mean	-1043.9
SD	4322.0	SD	2554.3	SD	2404.9	SD	3124.3
SE	1440.7	SE	851.4	SE	801.6	SE	1041.4
Median	-3879.8	Median	-1945.4	Median	-1500.2	Median	-1150.4
N		**P**		**K**			
Mean	3915.7	Mean	4616.7	Mean	227.8		
SD	3953.1	SD	4041.2	SD	2463.3		
SE	1317.7	SE	1347.1	SE	821.1		
Median	3186.3	Median	3511.9	Median	1011.3		
Total ash		**Al**		**Residual ash**			
Mean	5172.4	Mean	6713.6	Mean	7293.2		
SD	7017.1	SD	8723.6	SD	9521.2		
SE	2339.0	SE	2907.9	SE	3173.7		
Median	2302.4	Median	4026.0	Median	3658.9		

Appendix 4. Age-depth data for peat accumulation.

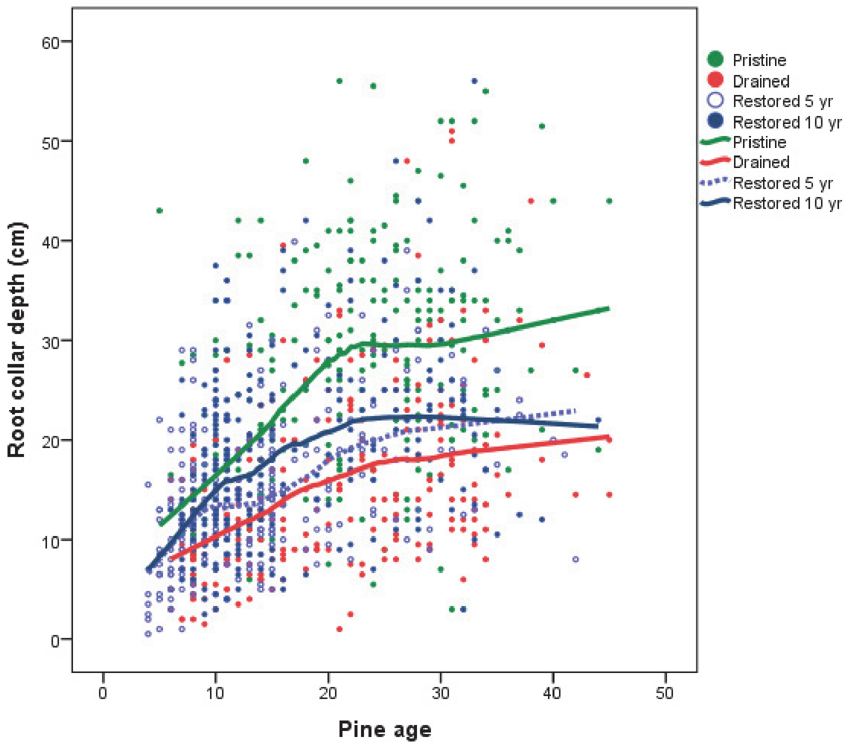

FIGURE S2 Untransformed age-depth data used in peat accumulation estimation. Note that in the analyses we used log transformed data, regression through the origin and only the first 10 years of the data to compare the peat growth between the treatments (see Methods, Peat growth).

IV

CONVENTION ON BIOLOGICAL DIVERSITY AND (NOT) RESTORING THE WORLD

by

Janne S. Kotiaho, Tuomas Haapalehto, Panu Halme, Santtu Kareksela, Anna Oldén, Jussi Päivinen & Atte Moilanen 2014

Manuscript

Convention on Biological Diversity and (not) restoring the world

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In October 2010, the tenth meeting of the Conference of the Parties of the Convention on Biological Diversity (CBD), agreed to translate the Strategic Plan for Biodiversity 2011-2020 and Aichi Biodiversity Targets (SPBABT) into national biodiversity strategies and action plans within two years (CBD 2010a). While only 20 Parties (out of 194) have since produced an updated action plan accommodating the SPBABT (CBD 2010b), the first Parties are now developing frameworks for the national implementation of it.

One major Aichi target is to restore at least 15% of degraded ecosystems globally by 2020: "*Target 15: By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks has been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification*" (CBD 2010c). To understand the challenges embedded in this target, we need to consider what the "*restoration of at least 15 per cent of degraded ecosystems*" actually implies. Doing so, we find the target will inevitably prove unrealistically challenging and impossible to ever be met.

First, we need to realize that from an ecological perspective, ecosystem degradation has a minimum of two dimensions: the extent of area that has become degraded and the magnitude of the degradation at any given location. We emphasize that knowledge of the extent of the degraded area alone is not sufficient for providing a scientifically valid estimate of the magnitude of ecosystem degradation, because it makes a great deal of difference whether an ecosystem has been only slightly degraded or almost completely lost. Therefore, implementing restoration measures on 15% of the degraded area is both ambiguous and most likely inadequate. We describe by a schematic presentation what scientifically valid 15% restoration by necessity means (Fig. 1) and illustrate the general implications of the 15% restoration target in the dominant ecosystem, forests, at our native locality Finland.

Forests of Finland cover nearly 150 000 km^2 or 49% of the land area, but less than 5% of them are in natural or semi-natural condition (Biodiversity.fi 2014) and the remaining 95% are variably degraded due to current or past intensive forestry operations. Timber is dominantly harvested by clear-felling and stumps are increasingly extracted for energy production. Harvest is usually followed by soil preparation and artificial regeneration by planting seedlings. These operations create single-species dominated forests that lack multi-storied forest structure, have greatly diminished volumes of dead wood and altered hydrological regime (Biodiversity.fi 2014). So, what might the 15% restoration in Finnish forests mean?

If we only considered the area as the basis of the 15%, one could, for example, leave $10m^3$ per hectare more dead wood on the 15% of the degraded forests (over 21 $000km^2$) after forestry operations. However, when the magnitude of the degradation is considered in addition to the area, it turns out that this action would translate into a net effect of reversing only 1.2% of ecosystem degradation (in only one out of several degraded components) of the forests of Finland (Fig. 1, item A). This improvement is based on the fact that currently there is on average c. $7m^3$/ha of dead wood in the forests of Finland, while in forests that are in natural or semi-natural condition is not uncommon that there is >100m^3/ha of dead wood (Biodiversity.fi 2014). Note also that even a complete and total restoration of ecosystem condition of 15% of only slightly degraded areas would not translate into a net effect of reversing 15% of ecosystem degradation (Fig. 1, item B). Therefore, an honest 15% restoration effort requires much more. One option could be complete and total restoration of ecosystem condition of 15% of completely degraded areas (Fig. 1, item C). This, however, is not likely to be feasible as restoration success - although often considerable (Benayas *et al.* 2009) - is nevertheless nearly always incomplete (Maron *et al.* 2012). Therefore, a more realistic option is to implement (heavy) restoration measures on an area much larger than 15%. For example, one could aim to restore 1/3 of the ecosystem condition in as much as 45% of the degraded landscape, resulting in a net effect of reversing 15% of ecosystem degradation (Fig. 1, item D).

What we learn from this example is that 15% restoration cannot logically be about area alone (like it is e.g. in Egoh *et al.* 2014). The only honest, ecologically justified, logically consistent and relatively unambiguous definition for the 15% is an aggregate 15% return of the ecological condition from the degraded state back towards less degraded state in every single degraded ecosystem of the world. An additional aspect that must not be forgotten is that ongoing habitat degradation in other areas should be counted in the balance of what has been achieved: even if significant restoration measures are implemented in several percent of the world, the net effect can still turn out negative if ecological condition continues to be degrading faster elsewhere (Laurance *et al*. 2012). Adding to lack of realism, the wording of the CBD suggests that restoration should happen before 2020. What would it require to meet this target in the Finnish Forest? If we immediately start implementing restoration measures that are effective in restoring 1/3 of the ecological condition on the focal area, we should need to restore over 52 000 hectares of forests every month until 2020. A good point of comparison is that during the past 20 years Finland has been able to restore less than 30 000 ha of forests to some (considerable) degree (Finnish Statistical Yearbook of Forestry 2013). This is three orders of magnitude less than what is implied by the CBD. Moreover, our example is likely to be conservative because compared to forest ecosystems in many parts of the world, forests in Finland may nevertheless be considered to be in relatively good condition.

We followed the heuristic logic of Fig. 1 and the example above, and developed a ten-step restoration prioritization procedure that retains the ecologically most relevant components while having otherwise minimal data demands and complexity (S1 and S2). High-end methods for optimal allocation of habitat restoration do exist (e.g. Noss *et al.* 2009, Wilson *et al.* 2011, Pouzols and Moilanen 2012), but these methods are data-hungry and their finesse is lost in a process that will require serious compromise to reach anything like the 15% target. Given the schedule and the magnitude of the task, lower-dimensional approaches such as the one described in (S1) are operationally more feasible. The procedure can be divided into four phases (Fig. 2): the first two steps of the procedure define the focal ecosystems and identify components that have degraded, steps 3-5 determine the current state of the ecosystems, steps 6-8 determine the effects and cost-effectiveness of restoration measures, and the final two steps launch into political decision making and implementation. To help the adoption of the procedure, we include a template (S2) with instructions and an authentic example from Finland, where the procedure is currently applied by a multi-stakeholder working group prioritizing restoration in an attempt to meet the 15% restoration target across the whole spectrum of ecosystems. Use of this procedure also allows for the monitoring of ecosystems' state and quantification of restoration success and perhaps even more importantly fills an important gap that was recently identified (Tittensor *et al.* 2014) in measuring progress towards SPBABT. In (S2) we provide a (low-dimensional) worked-out example for one other dominant Finnish ecosystem, peatlands, where altered hydrology (drainage) and increased tree cover are the most important degraded components.

In conclusion, above consideration of the 15% target exposes the overambitious nature of the SPBABT: to obtain a scientifically justified 15% reduction in the degradation of ecosystems we need to apply heavy restoration measures across very large areas in extremely short time, simultaneously compensating for ongoing degradation elsewhere. Unfortunately, the CBD agreements, although made with good intentions, appear to be repeatedly ridiculously unrealistic (e.g. significant reduction of the rate of biodiversity loss by 2010 (CBD 2010d)). Such recurrent lack of realism is likely to eat away the credibility of the CBD and cause the Parties to disregard the content. An example of possible disregard comes from the convention itself: the 194 Parties to the CBD in 2010 agreed that "*…longer-term actions to reduce the underlying causes of biodiversity are taking effect…*" (X/2, Annex, paragraph 10 (c) in (CBD 2010c)). The omission of the word "loss" after biodiversity is probably a simple mistake, but a mistake that reverses the intended meaning raises an issue about the sincerity of the CBD: has it been reduced to politically correct theatre that is verbally supportive of strategic goals but lacks genuine underlying intention to act upon problems?

Politicians, decision-makers and academics need to be honest and open about the magnitude of the global ecosystem restoration problem. While insufficient scientific information for policy and decision making is an obstacle for the implementation of the SPBABT, scientific uncertainty is not a valid excuse for inaction (CBD 2010c). Mankind has a tendency for self-deception and a narrow space-time perspective (Meadows *et al.* 1972) making seeking of excuses a likely problem for the implementation of the CBD's global attempt to impose limits on ourselves. The ecologically justified ten-step procedure (S1) and the ready-made template (S2) are our attempt at a pragmatic minimalist decision support tool to assist in the global ecological restoration task. The development of this tool stems from the hope that the lack of realism of the SPBABT does not discourage nations around the world from implementing significant restoration measures. It is implicit in the target of reversing 15% of ecosystem degradation by restoration that further degradation is taken into account in the balance. Therefore, Mankind can be proud if we can achieve even a modest fraction of the current 15% restoration target, because then the global trend of continuing habitat loss would already be stopped and reversed.

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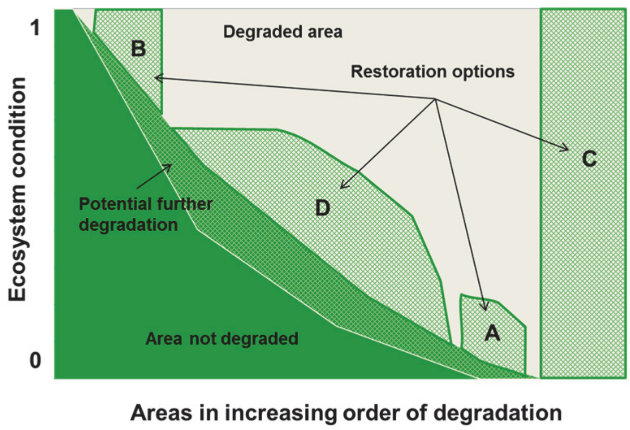

FIGURE 1 Options for restoration. This schematic illustrates how variably impacted areas of an ecosystem can either further degrade or be restored. The x axis describes the total area occupied by different restoration options in the landscape (A-D) and the area covered by each of the options represents their ecological impact. Options A, B and C occupy same total area of the landscape while options C and D have the same ecological impact. Scientifically valid 15 % restoration target cannot be achieved with small improvements made on 15% of the landscape area (A) or even with complete restoration of 15% of the slightly degraded areas (B). Instead, what is needed is either complete restoration of ecosystem condition on 15% of the completely degraded area (C) or significant partial restoration of much larger area (D). Here, the aggregate effect of (C) or (D) are close to ecologically valid 15% restoration target whereas (A) or (B) achieve much less than 15% even when they imply restoration measures on 15% of the area.

FIGURE 2 The four main phases of the ten steps for reversing the ecosystem degradation by cost-effective restoration measures.

Supplementary Materials:

Supplementary material S1: Description of a ten-step procedure for scientifically justified habitat restoration. The procedure described here is implemented numerically in template (S2).

From an ecological perspective, ecosystem degradation has a minimum of two dimensions: the extent of area that has become degraded and the magnitude of the degradation at any given location. Knowledge of the extent of the degraded area alone is not sufficient for providing a scientifically valid estimate of the magnitude of ecosystem degradation, because from an ecological perspective it makes a great deal of difference whether an ecosystem has been only slightly degraded or almost completely lost.

The existence of the two dimensions have not been appreciated in the literature discussing the Strategic Plan for Biodiversity 2011-2020 and Aichi Biodiversity Targets (SPBABT) (Egoh *et al.* 2014) or in one of the first attempts to operationalize it: the European Commission commissioned guidelines from Arcadis (Lammerant *et al.* 2013), the aim of which was to support the Member States of EU on the development of the strategic framework for setting priorities for ecosystem restoration. In this document a "*four-level model for ecosystem restoration*" has been described (Lammerant *et al.* 2013). Unfortunately, in its current form the four-level model is scientifically flawed because it does not properly take into account the magnitude of ecosystem degradation or improvement due to restoration and thus it does not allow assessment of the success of achieving the 15% restoration target of SPBABT (see also Tittensor *et al.* 2014). Moreover, even if a scientifically valid approach within the four-level model framework could be developed, in reality its operational implementation would be exceedingly difficult. This is primarily because for each focal ecosystem one would need to develop descriptors of the ecosystem condition at each of the four levels and for each degraded component in each ecosystem threshold values for moving between each of the levels should be determined.

Here we offer a simple ten-step procedure that is based on empirical continuous degraded components and there is no need for categorization or *a priori* target setting. This procedure fills an important gap in measuring progress towards SPBABT targets that was recently identified (Tittensor *et al.* 2014): there are no previous indicators to measure progress towards target 15. Ten-step procedure can be divided into four phases: I) the first two steps of the procedure define the focal ecosystems and identify components that have degraded, II) steps 3-5 determine the current state of the ecosystems, III) steps 6-8 determine the cost-effectiveness of restoration measures, and IV) the final two steps launch into political decision making and implementation. Supplement (S2) provides an annotated Excel-template for implementing this procedure, which is currently being applied to the Aichi 15% restoration problem across the whole spectrum of ecosystems in Finland.

1) *Decide focal ecosystem categories*. Categorization of ecosystems may be somewhat artificial (Lamarck 1809) but in practice necessary, as it facilitates expert-driven identification of ecosystem-specific degraded components and potential restoration measures. Each ecosystem needs an estimate of the area it covers. Ecosystem categories used in international biodiversity strategies, such as the habitat types in the Habitats Directive classification in Europe (Egoh *et al.* 2014, European Commission 2014), may provide a useful shortcut to categorization.

2) *Determine degraded components*. For each of the ecosystems identified in step 1, determine the main functional or structural components that may have degraded from the perspective of biodiversity and ecosystem services. Then, determine the area of the ecosystem that has been degraded by each of the component or their relevant combinations.

Here it is essential to keep in mind that ecosystem services are not a biological phenomenon, but that by definition they are the ecosystem functions that humans value (Millennium Ecosystem Assessment 2005), and there may be trade-offs between them and biodiversity (Bennett *et al.* 2009, Schröter *et al.* 2014). Where trade-offs occur, we must not let biodiversity be compromised for ecosystem services. This is because if we do, we are not really imposing limits to ourselves (Meadows *et al.* 1972), but rather, we let economic and other benefits drive further unsustainable exploitation of our environment.

3) *Determine current state and reference before degradation.* For each of the degraded components (step 2) we need to determine an approximate reference state before human-caused degradation. While the concept of reference state in ecology has often been problematized (Hunter 1996, Haila *et al.* 1997, Jackson and Hobbs 2009), conceptual or actual uncertainty about the natural state of an ecosystem should not be used as an excuse for lack of action. It is a simple fact that a reference state is needed for each of the degraded components; otherwise the degree of degradation and thus the extent of restoration success cannot be determined.

Note that the reference state needs to be based on ecology, not on societal value. It is also very important to keep in mind that the reference state is not a target; it is utilized to evaluate the current state and to estimate the amount of recovery of ecological value via alternative restoration measures. Targets for the desired amount of improvement are set separately (step 9). In addition to the reference state before degradation, we also need the approximate current state of the ecosystem for each of the degraded components.

4) *Determine loss of ecosystem condition related to each degraded component*. Determine the fraction of the ecosystem's overall condition lost when the component has been completely degraded. This is needed for two reasons: i) the current state is often not completely degraded and ii) restoration does not usually lead to complete recovery (6).

In most cases, the ecosystem condition is not zero even if there is a complete degradation of one of its components and the current state is determined by a combination of a number of variably degraded components.

5) *Determine overall ecosystem condition remaining at the current state*. Assuming multiplicative effects of the components on ecosystem condition, the current condition remaining in ecosystem, R^E, can be calculated as

$$R^E = \prod_{n=1}^{N^E} \left(1 - L_n^E(1 - n_{curr}/n_{ref})\right)$$

in which N^E is the number of relevant components in the focal ecosystem E, and L_n^E is the loss of ecosystem condition if component n is completely degraded, and n_{curr} and n_{ref} are the state of component n in the current state and in the reference state, respectively. When losses of ecosystem condition due to individual components are close to zero, the ecosystem condition remaining is close to one. Importantly, many degraded components can be measured by continuous variables and the current situation usually is only a partial degradation of the component. In the case there is a nonlinear relationship in how ecosystem condition changes with the change in state of any component, the equation above can easily be expanded by a function $f(n_{curr}/n_{ref})$, which can model e.g. a threshold effect via a sigmoid function.

6) *Determine potential restoration measures and their costs*. List all potential restoration measures which could plausibly be implemented to reduce the degradation. Estimate the cost of each. Consider both active and passive restoration measures (Benayas *et al.* 2009). In some cases the socio-economic benefits of restoration can be very clear (De Groot *et al.* 2013).

7) *Determine overall ecosystem condition gain related to each restoration measure.* Determine approximately how much each restoration action would recover the fractional loss of each degraded component. Note that very seldom does any restoration measure result in a complete recovery of the ecosystem (Maron *et al.* 2012).

8) *Determine cost efficiency of restoration measures*. Divide the benefit of each restoration measure (step 7) with its estimated per-unit cost (step 6).

9) *Determine the target*. While a target can generally be freely decided, the target given in the SPBABT is to restore 15% of the World's degraded ecosystems. When we estimate the extent of restoration needed for the 15 % target to be reached for example as we did for forests in the main or a fraction of peatlands in Finland (S2), we can see that this

target is clearly overambitious, resulting in restoration action needed across areas so vast that implementation goes beyond any available resources.

10) *Decision making and implementation*. Based on the knowledge gathered in the previous steps, the last step is to prioritize the restoration measures for ecosystems and initiate the implementation together with the society at large.

Supplementary material S2: A short guide and step by step instructions on how to use the Excel template for "Ten steps for reversing ecosystem degradation". This is a separate Excel template not provided here. It can be requested from the authors, prior to the formal publication of the manuscript.