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TOWARDS MEANINGFUL STRATEGIC ENVIRONMENTAL ASSESSMENT

LESSONS LEARNED FROM DANISH MINING

BY
MORTEN BIDSTRUP

DISSERTATION SUBMITTED 2016

AALBORG UNIVERSITY
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Dissertation submitted 2016

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PREFACE

This dissertation is the product of a three-year PhD fellowship, of which roughly two years were allocated for research. The fellowship was financed by Aalborg University in collaboration with The Danish Regions' Knowledge Centre for Environment and Resources. With an outset in the case of Danish mining, I have had the opportunity to explore how to apply strategic environmental assessment more meaningfully in practice. A playful colleague told me upon starting my fellowship: *"writing a PhD is like banging your head against a wall for three years!"* The process has resembled this description to some extent, but multiple individuals have been there to cushion the impact of *'the wall'* along the way.

I have had the pleasure to conduct my work in the Danish Centre for Environmental Assessment (DCEA), which is a tightly knit community of researchers working within the fields of impact assessment, life cycle assessment and public participation. I would like to thank all of my wonderful colleagues in DCEA for the supportive work environment and the many critical discussions. Among many, I would like to thank Lise Kirk Nordensgaard for proofreading my studies and Associate Professor Matthew Cashmore for his feedback on grey IA. I would also like to express my sincerest gratitude to my two supervisors Professor Anne Merrild Hansen and Associate Professor Massimo Pizzol, who have supported, motivated and challenged me throughout the process.

The research was highly influenced by my interaction with the international research community. I would like to thank my colleagues at the International Association for Impact Assessment for our many discussions at the yearly conferences, as well as I would like to thank Professor Maria Rosário Partidário from the University of Lisbon for our collaboration on cumulative effects and Associate Professor Sangwon Suh for our discussions on life cycle thinking and for providing me the opportunity to visit the University of California (Santa Barbara).

I will use this opportunity to thank the Danish mining planners whose practices I scrutinised. Thank you for your openness, helpfulness and positive attitude towards both me and the project. Thank you for answering my many questions, for introducing me to the IA issues of your field and for granting me the academic autonomy to develop the project in the direction of my interests.

Last but not least, I would like to thank my family and friends for their unconditional support. Most gratitude goes to my fiancée Yana for being there in all the ups and downs.

Morten Bidstrup

Aalborg, February 2016

ENGLISH SUMMARY

This dissertation explores how to apply Strategic Environmental Assessment (SEA) meaningfully. SEA applies to proposed plans, programmes and policies, and it aims to facilitate better and more transparent decision-making. In this dissertation, an SEA application is considered *'meaningful'* when its analysis and procedure fit the contextual setting and support the principal aim of facilitating better decisions.

The dissertation builds on the case of Danish mining. The Danish Regions prepare plans for how to secure the regional supply of raw materials for the construction sector. These plans liberate space for mining by zoning where private contractors may propose concrete mining projects. All mining plans must be evaluated by SEA, while all mining projects are subject to requirements for Environmental Impact Assessment (EIA). Still, there are several issues. Planners find that often it is not meaningful to assess cumulative effects and alternatives. Requirements for EIA are often omitted and the many assessments appear to have a local focus, by which indirect and distant impacts are not considered.

These issues are addressed in five studies with the purpose of exploring what lessons can be learned from Danish mining on meaningful SEA. Two studies explore why often the assessment of plan alternatives and cumulative effects is meaningless in Danish mining. Two others explore whether the observations on EIA omission and lacking consideration of indirect impacts are representative for wider Danish practice. The last study proposes and tests a procedure for how to apply life cycle assessment meaningfully to SEA with the purpose of considering indirect impacts.

First and foremost, the research contributes with *'lessons'* on why sometimes SEA is not meaningful. It was found that the purpose of the mining plans restricts the planners in considering both key alternatives and the diverse activities contributing to cumulative effects. This restriction relates to the planners' institutional power and influences the planners' perception of what can be considered their responsibility.

Second, the research contributes with *'lessons'* on how practitioners may try to make Impact Assessment (IA) meaningful. It was found that they may adopt a local assessment focus to fit the SEA to the contextual setting of the plan under evaluation. Also, practitioners may use IA screening to impose environmental improvements at a meaningful time or to omit requirements for (in their opinion) meaningless IAs.

At last, the research contributes with *'lessons'* on how SEA can be applied more meaningfully. It is recommended to fit SEA to the contextual setting by focusing it on the strategic capabilities of the planners. Having that in mind, it is recommended that planners try to rebel against contextual restrictions whenever possible. It was found that such rebellion can bring analytical improvements. A last recommendation is to be aware of and utilise the *'grey'* areas of the IA system.

DANSK RESUME

Denne Ph.d.-afhandling omhandler, hvordan Strategiske MiljøVurderinger (SMV) kan anvendes på en meningsfuld måde. SMV-værktøjet har til formål at fremme mere bæredygtige og gennemsigtige beslutninger på det strategiske niveau i planhierarkiet ved at sikre udarbejdelsen af miljøanalyser igennem en standardiseret procedure. En *'meningsfuld'* anvendelse af SMV er defineret, som når værktøjets analyser og procedurer passer til konteksten og bidrager til bedre beslutninger.

Afhandlingen tager udgangspunkt i råstofplanlægning. De Danske Regioner skal hvert fjerde år udarbejde råstofplaner, hvori der zoneres graveområder til fremtidens forsyning. Råstofplaner skal vurderes med SMV, og alle råstofprojekter er omfattet af kravene til Vurdering af Virkningen på Miljøet (VVM). Der er dog adskillige problemstillinger med den nuværende praksis. Råstofplanlæggerne oplever, at det sjældent er meningsfuldt at vurdere alternativer og kumulative påvirkninger. Krav til VVM bliver ofte undveget, og der tages sjældent højde for indirekte påvirkninger.

Disse problemstillinger adresseres gennem fem studier, hvis formål er at bidrage med ny viden om meningsfuld brug af SMV. To studier undersøger, hvorfor det kan forekomme meningsløst at vurdere visse alternativer og kumulative påvirkninger i en råstofsammenhæng. To andre studier undersøger, om sektorens undvigelse af VVM og lokale miljøfokus repræsenterer en bredere dansk miljøvurderingspraksis. Det sidste studie præsenterer og tester en procedure, hvormed livscyklusvurdering kan benyttes meningsfyldt i SMV til at adressere indirekte miljøpåvirkninger.

Afhandlingen bidrager med viden om, hvorfor brugen af SMV nogle gange ikke er meningsfuld. Råstofplanernes formål er at zonere nye graveområder. Dette snævre fokus forhindrer planlæggerne i at vurdere centrale forsyningsalternativer, ligesom det forhindrer dem i at vurdere forskelligartede bidrag til kumulative påvirkninger. Disse kontekstuelle forhindringer har udgangspunkt i planlæggernes institutionelle magt og influerer på deres opfattelse af, hvad der er deres miljømæssige ansvar.

Herudover bidrager afhandlingen med viden om, hvordan praktikere prøver at gøre miljøvurderinger meningsfulde. Et lokalt fokus kan være et udtryk for en tilpasning af en SMV til den relaterede plans kontekstuelle ramme. Praktikere kan også i nogle tilfælde benytte screening-processen til at fremme miljømæssige forbedringer på et meningsfuldt tidspunkt i beslutningsprocessen eller til at undvige en VVM, som de ikke betragter som meningsfuld.

Slutteligt bidrager afhandlingen med viden om, hvordan SMV kan anvendes på en mere meningsfuld måde. Det anbefales, at SMV tilpasses plan-konteksten ved at fokusere på planlæggernes strategiske råderum. Dette er dog ikke ensbetydende med, at praktikere ikke skal prøve at bryde deres kontekstuelle rammer, når det er muligt. Forskningen viser, at sådant kontekstuelt oprør kan føre til væsentlige analytiske forbedringer. Den sidste anbefaling er, at praktikere bør være opmærksomme på og forsøge at udnytte de uformelle, *'grå'* miljøvurderinger.

БЪЛГАРСКО РЕЗЮМЕ

Тази дисертация изследва темата за смисленото прилагане на стратегическата екологична оценка (СЕО). Инструментът СЕО се прилага към предложените планове, програми и обща политика, и има за цел да улесни устойчивото и прозрачно вземане на решения. В тази дисертация, прилагането на СЕО се смята за *'смислено'*, когато и анализът, и процедурата пасват в контекста и подкрепят основната цел – спомагане за вземане по-добри решения.

Дисертацията се базира на случай от датския добив на материали. Датските Региони подготвят планове за това как да се осигури регионалната доставката на суровини за строителния сектор. Тези планове освобождават пространство за добив на материали, чрез разделяне на зони, където частни изпълнители имат възможност да предложат конкретни минни проекти. Всички планове за добив на материали трябва да бъдат оценени чрез СЕО, докато всички проекти за добив трябва да отговарят на изискванията за оценка на въздействието върху околната среда (ОВОС). И все пак, изглежда, че съществуват няколко проблема. Лицата изготвящи планове откриват, че често не е смислено да се прави оценка на натрупващите се ефекти и алтернативи. Често изискванията за ОВОС се прескачат и много от оценките изглежда, че имат само локален фокус, който не взима предвид косвените въздействия.

Тези проблеми са разгледани в пет проучвания, с цел изследване на препоръките, които могат да се направят от датския добив на материали върху смислена СЕО. Две от проучванията изследват защо при датския добив на материали, оценката за план-алтернативи и натрупващи се ефекти, са често безсмислени. Две други проучвания изследват дали наблюдаване пропускането на ОВОС и не вземането под внимание на косвените въздействия, са представителни като по-широка датска практика. Последното проучване разглежда и тества процедура за смислено прилагане на оценката на жизнения цикъл към СЕО, с цел вземане предвид косвените въздействия.

На първо и основно място, тази научна разработка допринася с препоръка защо в някои случаи СЕО е безсмислено. Установено е, че целта на плановете за добив ограничава лицата, изготвящи тези планове, да вземат предвид ключови алтернативи и разнообразни дейности, допринасящи за натрупващи се ефекти. Тези ограничения относно институционната власт, която лицата изготвящи планове имат, влияе на разбирането им, за това кое може да бъде считано за тяхна отговорност.

На второ място, това проучване допринася с препоръка за това как практикуващите могат да опитат да направят смислена оценка на въздействието (ОВ). Установено е, че те могат да заимстват с местен фокус на оценката, за да може СЕО да съответства с контекста на плана, който се оценява. Освен това, практикуващите могат да използват скрийнинг на ОВ с цел налагане на подобрения на околната среда в разумно време или с цел пропускане на изискванията за безсмислени (по тяхно мнение) оценки.

В заключение, проучването дава своя принос с препоръки за това, как СЕО да се прилага по-смислено. Препоръчително е СЕО да се намести в контекста, чрез поставяне фокус на СЕО към стратегическите възможности на лицата, изготвящи плановете. С това предвид, препоръчително е при възможност, лицата, изготвящи планове, да се опитат да се противопоставят на контекстуалните ограничения. Установено е, че такова противопоставяне може да доведе до аналитични подобрения. Заключителната препоръка съдържа съвет да се имат предвид и оползотворяват т.н. '*сиви*' зони на ОВ.

1 INTRODUCTION

This chapter prepares the reader for the dissertation with a brief presentation of the context, aim and research questions of the PhD project and the five studies it is based upon. The chapter concludes with a reading guide for the dissertation.

1.1 A focus on meaningful assessment

Alongside the continuous development of modern societies, there seems to be an ever-increasing understanding of that the world is interconnected in systems. History has shown us that well-intended initiatives can lead to unwanted impacts on both bio-physical and socio-economic systems, and the Impacts Assessment (IA) of development proposals has thus grown to become an integrated part of decision-making across the world. The purpose of an IA is to facilitate environmentally sound decision-making by providing objective information on potential impacts and a platform for public participation. One type of IA is the Strategic Environmental Assessment (SEA), which applies to proposed programmes, plans and policies.

Though legal requirements for SEA have been in place for 15 years in Europe, the tool shows mixed performance. International scholars question the effectiveness of SEA and within the IA community there are continuing discussions on how to apply SEA more strategically to better address alternatives and cumulative effects. On a personal note, I have found that Danish SEAs are often so short and superficial they leave one wonder: *“Has there been a point to making this assessment?”*

The concept for the research of this dissertation sparked in late 2012 when the planners in charge of onshore mining in Denmark told members of the Danish Centre for Environmental Assessment (DCEA) that they struggle to see the meaning in some elements of SEA. The planners wish to apply SEA for finding green solutions and avoiding conflicts, but often they see little use in discussing alternatives and cumulative, distant impacts. Thus the PhD project initiated in April 2013 with the following aim:

To explore how to better address alternatives, cumulative effects and life cycle impacts in a meaningful way within the SEAs of Danish mining

The oxford dictionary defines *‘meaningful’* as something that has a worthwhile quality and purpose. Returning to the very purpose of IA, I argue that:

an application of SEA is meaningful when it provides valuable information in a way that is transparent and facilitates substantive improvements.

Section 2.2 describes how, principally, good SEA practice has a procedural, an analytical and a contextual side. I argue that an SEA is *‘meaningful’* for the planners and stakeholders when the procedural and analytical elements fit within the decision-making context. That is, when the SEA is applied at the right time and is focused on the right things to change developments for the better.

1.2 The research questions

Though having the primary aim of generating knowledge for the Danish mining planners on how to apply SEA more meaningfully, it was decided early-on to shape the project for an international, academic audience. The project was thus granted the following research question:

What lessons can be learned from Danish mining on
meaningful application of strategic environmental assessment?

The central research question was explored through three research sub-questions which concern why SEA is not meaningful, how the planners then act and what can be done. The sub-questions were developed during the first months of the PhD when I compared my observations from the Danish mining SEA practices to the scholarly literature (see chapter 5 and 6). The questions were tailored to provide *'lessons'* to the state-of-the-art. They are listed below:

a) *Why is the assessment of alternatives and cumulative effects poor in the strategic environmental assessments of Danish mining?*
b) *How representative are the observations on 'grey' screening practices and lacking life cycle thinking within Danish mining?*
c) *How can life cycle assessment be applied in the strategic environmental assessments of Danish mining?*

1.3 The format and studies

The dissertation held in the hand of the reader is a covering essay to a collection of five published studies (listed in table 1.1), which address the three research sub-questions. The purpose of the essay is to account for the overall research approach and to draw conclusions across the individual studies to ultimately answer the central research question. Sub-question (a) is answered in the studies 1 and 2, which explore the difficulties in addressing development alternatives and cumulative effects in the Danish mining plan SEAs. Sub-question (b) is answered in the studies 3 and 4, which explore the representativeness of the observations made on grey screening practices and lacking life cycle thinking, respectively. Sub-question (c) is answered in study 5, which proposes a procedure for SEA-LCA integration and tests the procedure on the case of Danish mining.

All five studies are published in internationally acknowledged, peer-reviewed journals. The studies 3 and 4 are solo contributions, while the remaining three are written in co-authorship with peers of various fields, institutions and nationalities. Signed co-author statements confirm that I can be attributed 80-90% of these studies' scientific contribution. At large, I have developed the idea, made the literature review, collected the data, written the manuscript and managed the review process for all studies.

1	The paradox of strategic environmental assessment	Bidstrup M Hansen AM	EIA Review 2014, 47, 29-35
2	Cumulative effects in strategic environmental assessment: the influence of plan boundaries	Bidstrup M Kørnøv L Partidario MR	EIA Review 2016, 57, 151-158
3	The 'grey' assessment practice of IA screening: prevalence, influence and applied rationale	Bidstrup M	EIA Review *in press*, E-pub ahead of print
4	Life cycle thinking in impact assessment: current practice and LCA gains	Bidstrup M	EIA Review 2015, 54, 72-79
5	Life cycle assessment in spatial planning: a procedure for addressing systemic impacts	Bidstrup M Pizzol M Schmidt JH	Journal of Cleaner Production 2015, 91, 136-144

Table 1.1: The five published studies of the dissertation.

Though the PhD project has just finished, the publications have already shown to have an academic impact. The studies 1 and 4 configured on the list of the 25 most downloaded articles from EIA Review during the spring of 2014 and the summer of 2015, respectively. Moreover, study 3 will be published in a 2016 special issue on quality in IA – by invitation from the editor – as a consequence of a much debated presentation of its results at the 35th annual conference of the International Association for Impact Assessment (Florence, 2015). It is still early days for conclusions on the long-term impact of the research since the first study was published less than 2 years ago (as of March 2016). Yet, the studies have already been downloaded (or viewed) more than 4000 times and cited by international peers.

1.4 Reading guide

The dissertation is divided into four broad *'parts'*.

Part I provides the *'contextual framework'* for the dissertation. The part opens with a description of the evolution, principles and general issues of SEA (chapter 2) before it moves on to describe the case of Danish mining (chapter 3). The part concludes with a presentation of systems thinking (chapter 4). Systems thinking is applied to elaborate on what meaningful SEA entails, to legitimise the focus on LCA in IA and to contextualise the issues of Danish mining.

Part II describes the *'research design'* of the PhD. It opens with a presentation of the state-of-the-art on some selected IA issues, from where it is believed there are lessons to be learned (chapter 5). Knowledge gaps in the scholarly literature on these issues are then taken as point of departure for developing the research sub-questions (chapter 6). Part II concludes with a description of the method – hereunder the research approach, the data collection and analysis, and the coverage of the five studies with respect to meaningful SEA (chapter 7).

Part III provides a *'synthesis of findings'*. Here, the studies' results are presented and related to the research sub-questions (chapter 8). The essay then moves on to discuss the representativeness of the case, my engagement with mining planners and the systems' perspective on the results (chapter 9). The part concludes by elaborating on what lessons were learned (chapter 10).

Part IV is named *'publications',* and it contains the five studies. The manuscripts of the studies have been copied into the word template of the dissertation and edited slightly[1]. Study 1 was found to need a last proofreading, and the numeration of all sections, figures and tables have been given the prefix *'p'* (for *'publication'*) in order to avoid confusion. The unedited, original publications can be found by following the posted doi links. The reader is referred to part IV for details on the method, assumptions, data, results and discussions of each study. Some overlaps with respect to IA and Danish mining may occur since the studies must be able to stand alone in their published forms.

[1] This was permitted by Laura Stingelin from Elsevier on January 12th 2016 with a reference to Elsevier's publishing rights – see: *https://www.elsevier.com/about/companyinformation/*

PART I

CONTEXTUAL FRAMEWORK

2 THE PRINCIPLES OF IMPACT ASSESSMENT

This chapter opens with a description of how IA evolved from being a North American tool for predicting the bio-physical impacts to today being a worldwide family of sub-tools, which all apply a broader concept of the environment. The procedural steps of an IA will then be presented before the chapter concludes with elaboration on the concept and rationale of SEA.

2.1 The evolution of impact assessment

The field of IA was born in the 20th Century from the increasing understanding of how man influences his surrounding environment (Morgan 2012). The first piece of IA legislation was approved in 1969 with the American National Environmental Protection Act (NEPA) of 1969, which required an environmental assessment made for proposed federal actions that may influence the environment significantly (Bina 2007; Morgan 2012; Therivel 2010). Legislation in other countries soon followed. A major step was the approval of the first European IA Directive, which in 1985 demanded all member states to implement requirements for IA of major projects in their national legislation (European Commision 2015). Though amended since then, both NEPA (US EPA 2000) and the European Directive (European Parliament 2014) remain active pieces of legislation. Today, 45 years after its conception, only two of the world's nations have not approved national legislation or signed international agreements on the application of IA (Morgan 2012). Multiple efforts have moreover been made on assigning IA to externally financed projects in developing countries, where IA legislation may not be strictly enforced. The OECD countries have agreed on procedures for IA with respect credit lending for projects (Morgan 2012), just as around 80 private institutions (covering over 70% of the project finance debt in emerging markets) have signed up for similar obligations (Equator Principles Association 2015).

The making of an impact assessment family

Yet, the evolution of IA is not solely related to its institutionalisation. The field of IA has today grown to encompass a *'family'* of sub-tools (IAIA 2009:1), which despite their differences in scope all share the common purpose of predicting the impacts of proposed actions (Pope et al. 2013). To some extent this family of tools developed from dissatisfaction with the early practices of IA (Morgan 2012).

IA was for many years predominantly assigned to the approval of large projects in the form of what is today known as an Environmental Impact Assessment (EIA). Though NEPA and other early understandings of IA did not confine application to the project level, the practical application of IA left practitioners and academics arguing that there was a need for an IA tool at the more strategic levels wherefrom future projects are framed (Bina 2007). Such advocacy resulted in the developments of the Strategic Environmental Assessment (SEA), which applies to proposals for

Policies, Plans, and Programmes (PPPs) affecting the environment significantly (Tetlow and Hanusch 2012). The strong focus on the biophysical environment in EIA and SEA sparked the development of the specialised Social Impact Assessment (SIA) – see Esteves et al. (2012) – and Health Impact Assessment (HIA) – see Harris-Roxas et al. (2012). Similarly, the intrinsic focus on the development under question (as opposed to the receiving environment) paved the way for the specialised Cumulative Affects Assessment (CEA) – Canter and Ross (2010) – and Sustainability Assessment (SA) – see Bond et al. (2012).

The most applied tools of the IA family are arguably the EIA and SEA, which both are legally required within all member states of the European Union (European Parliament 2001; 2014). While the division of IA into the project-oriented EIA and the PPP-oriented SEA is acknowledged, the independent role of SIA, HIA, CEA and SA remain more unclear. Requirements do exist for CEA in Canada (Government of Canada 2012: §19; Hegmann et al. 1999) and SA in the United Kingdom (UK Government 2004: §19), but many are of the opinion that social, health, cumulative and sustainability impacts can be addressed effectively in EIA and SEA. Textbooks on both EIA (Lawrence 2003; Morgan 1998; Weston 1997) and SEA (Dalal-Clayton and Sadler 2005; Fischer 2007; Partidário 2012; Therivel 2010) all apply a wider definitions of the 'environment' and stress the need for assessing cumulative impacts. Moreover, the International Association for Impact Assessment – IAIA – (2009:1) clarifies that the *'environment'* assessed in EIA and SEA has *"evolved from an initial focus on the biophysical components to a wider definition, including the physical-chemical, biological, visual, cultural and socio-economic components"*.

2.2 The impact assessment process

An IA can be defined as *"the process of identifying the future consequences of a current or proposed action"* with the aim of facilitating sustainable development (IAIA 2009:1). Such *'actions'* are onwards referred to as *'developments'* – a term covering projects, programmes, plans and policies. Though the many IA tools differ in scope and application, most are built around the following formal steps:

1. Screening
2. Scoping
3. Identification of alternatives
4. Impact analysis
5. Reporting
6. Follow-up

The first step of an IA is that of deciding whether it is needed. This step is referred to as *'screening'* and entails an evaluation of whether the proposed action poses a risk of significant impacts. Screening practices may vary (Pinho et al. 2010). In the European Union developments are screened by either comparison to some fixed criteria for when an action can be expected environmentally significant or by case-to-case evaluation (Parliament 2001; 2014). If it is found that an IA is needed, practitioners must then decide on which impacts to evaluate.

This second step is referred to as *'scoping'*. This procedure is put in place to allow IAs to be both rigorous and cost-effective since it legitimises a narrowing of scope to what essentially are the critical impacts (Lawrence 2007). The two initial steps of *'screening'* and 'scoping' are critical processes since they lead to a decision on both the need for and potential focus of an IA (Weston 2000).

The third step of an IA is the *'identification of alternatives'.* Alternatives represent the various ways of reaching the development objective, and they essentially provide a frame of reference to the impacts of the development. All alternatives are then analysed with respect to the issues identified in the scoping phase. This fourth step is referred to as the *'impact analysis'* of the IA. Impacts can be analysed by various tools of both qualitative and quantitative nature. IAs are overarching tools, which can include independent analytical tools, such as flow assessment, cost-benefit analysis or life cycle assessment (Finnveden and Moberg 2005).

The results of the IA and its recommendations for how to minimise and mitigate the impacts must then be documented and presented in the fifth step of *'reporting'.* The last step is *'follow-up'*. This entails the monitoring of whether the recommendations are followed in practice and the evaluation of whether they are effective.

When is an IA good?

There are many facets to what makes an IA application good (Joseph et al. 2015). Yet, the IAIA (1999) emphasises the importance of following the procedural IA steps – as presented above. With respect to screening, for instance, good practice entails that all significantly impacting developments are assessed. Second, good practice builds on some basic principles. IAs ought to be *"rigorous"*, *"relevant"*, *"participative"* and *"transparent"*, as well as they ought to support the wider objective of facilitating sustainable development (IAIA 1999). Thus I find that there are at least three sides to good practice IA:

1. **A procedural side** with a focus on insuring formal IA steps, here among the requirements for public participation.
2. **An analytical side** with a focus on indentifying consequences rigorously and presenting information in a transparent and objective manner.
3. **A contextual side** with a focus on providing relevant information in a way that supports decision-making and facilitates substantive changes.

This is is supported by Finnveden and Moberg (2005:1167), who define IAs as *"change-oriented procedural tools"* for environmental analysis.

2.3 The strategic environmental assessment

Sadler and Verheem (1996:26) define SEA as follows:

"SEA is a systematic process for evaluating the environmental consequences of proposed policy, plan or programme initiatives in order to ensure they are fully included and appropriately addressed at the earliest appropriate stage in decision making on par with economic and social considerations."

This definition differs from the general one of IA in two distinct ways: it confines the tool to PPPs and it emphasises the need for early application. As argued below, the first point was the rationale for developing SEA while the second point relates to an effective use of the tool.

A focus on strategic developments

The SEA developed because project-level IA (today known as EIA) tended to have a too delimited focus on project adjustments only (Bina 2007). The underlying rationale was that impact assessment on the more strategic levels of decision-making can lead to lower environmental impacts because developments are often proposed in a tiered planning system where policies guide the making of plans, programmes and eventually projects (Kalle and Arts 2013). Decision-making is rarely as rational (Kornov and Thissen 2000) or tiered (Noble 2009) as written in textbooks, but Therivel (2010:18) finds that modern day SEA *"offers the chance to influence the kinds of projects that are going to happen, not just the details once projects are being considered"*. An EIA of a wind turbine project, for instance, will most often only assess the impacts of that specific project and consider technical alternatives on how to realise it. Reversely, an SEA of the assigned energy PPP may consider the technologies and locations for future energy supply. The ability to look beyond single projects and consider more systemic alternatives has left authors advocating that SEA is a more appropriate tool for considering cumulative effects (Duinker and Greig 2006; Gunn and Noble 2011) and sustainability impacts (Stinchcombe and Gibson 2001; White and Noble 2013).

An increased focus on decision-making processes

SEA was named with the word *'strategy'* because it was intended for development proposals at the strategic levels of planning. Yet, this terminology left authors such as Noble (2000) and Cherp et al. (2007) questioning: When is an SEA truly strategic? The theory on SEA has changed since its conception. Coming from an initial idea of SEA as an EIA performed on PPP proposals, it was found gradually that for SEA to be effective, it must be integrated in the process of developing PPPs (Bina 2007; Lobos and Partidario 2014). SEA must be applied parallel to the development process to identify the *"decision windows"*, where environmental concern can be effectively addressed (Dalkmann et al. 2004). Table 2.1 presents one of the many models for integrating SEA in decision-making process.

PPP development process	SEA process
Formulate PPP objective	← Formulation of SEA objectives
Identify alternative ways to achieve the PPP objective	← Scoping ← Identification of alternatives
Choose preferred alternative	← Impact analysis
Make formal decision	← SEA reporting
Implement and monitor the PPP	← Follow-up

Table 2.1: Merge of SEA and the development process – modified from Therivel (2010:16)

The SEA has today developed unique characteristics, which distinguishes it from the project-oriented EIA. Noble (2000) lists the differences as presented in table 2.2. EIAs are typically applied to finished projects proposals with predefined objectives when no major changes can be made. SEAs, on the other hand, are applied during the PPP development when more fundamental questions can be asked.

EIA	SEA
… focuses on projects	… focuses on PPPs
… is made on projects with pre-determined goals and objectives	… is made on PPPs in context of broader visions and objectives
… asks "what are the impacts of our option?"	… asks: "What is the preferred option?"
… is reactive (after the project development process)	… is proactive (within the PPP development process)
… is narrow and highly detailed	… is broad and has a low level of detail

Table 2.2: The differences between an EIA and an SEA – modified from Noble (2000:204).

Still, it appears that SEA practice has not progressed parallel to the theoretical development of the tool. In a recent review of SEA practice, Lobos and Partidario (2014) conclude that SEA still predominantly is characterised as a technical and reactive assessment practice similar to that of EIA. SEA is often used to evaluate the impacts of a fully finished proposal for a plan or a programme. Thus Partidario (2012) argues that there exist two separate forms of IA application in practice: An EIA-based SEA, which focuses on impact evaluation, and a strategic approach to SEA, which focuses on influencing decision-making.

SEA practice is still not mature

This distinction aside, it has been found that SEA practice shows deficiencies worldwide (Pope et al. 2013). The assessment of alternatives and cumulative effects is often done poorly, and there are mixed findings on whether the tool indeed leads to substantive improvements of PPPs (Tetlow and Hanusch 2012). The issue of SEA effectiveness has received much attention in the scholarly literature (van Doren et al. 2013), but it is difficult to evaluate because the impact of an SEA can be both indirect and long-term (Acharibasam and Noble 2014). Van Buuren and Nooteboom (2009:146) argue: *"because the process is fluid and influenced by multiple factors, it is often impossible to pinpoint the exact impact of SEA on the final decision"*.

The contextual side to good practice SEA is receiving more and more focus in the scholarly literature. Yet, further research is still needed on how to apply SEA more meaningfully. This dissertation explores just that through a study on Danish mining. Chapter 5 presents a more elaborate literature review of the SEA issues, which were selected for further exploration.

3 IMPACT ASSESSMENT IN DANISH MINING

This chapter fulfils the function of presenting the case of Danish mining and the assigned impact assessment context. The chapter opens with a general description of Danish planning and impact assessment. It concludes with a presentation of the materials, impacts and regulation of Danish mining.

3.1 Denmark as planning context

Spatial competition in an affluent society

Denmark is an autonomous country situated in Scandinavia, Northern Europe. The country has a population of 5.7 million Danes (World Bank 2015d), and it has since 1973 been a member of the European Union (2015). Denmark is among the ten richest countries in the world measured in gross domestic product by capita (World Bank 2015a), and its economy is stable (Trading Economics 2015). Politically, the country is governed by the *'Nordic Model'*, which entails the existence of free market capitalism together with a strong social system. The level of taxation is high, but the responsibilities of the welfare state are extensive. This model has spread the wealth and made Denmark one of the most economically equal (CIA 2015), most *'happy'* (Helliwell et al. 2013) and least corrupt (Transparency International 2015) countries in the world. At large, Denmark is a peaceful place where the citizens trust the choices of decision-makers.

Yet, planning is not without its difficulties. Denmark is a small country of 43,000 km^2 – only one tenth the size of neighbouring Sweden (World Bank 2015b). Though 5.7 million inhabitants is not a large population internationally, Denmark has a population density twice the world's average and 20% above that of the EU (World Bank 2015c). Land use is currently divided as 10% urban areas, 66% agriculture and 23% nature (Statistics Denmark 2015a). There exists no vacant space since 100% of the onshore territory is used or zoned for a particular purpose. This point is demonstrated by Arler et al. (2015), who in a recent report conclude that the ambitions for future land use within food production, energy crops, forests, biodiversity, infrastructure and urbanisation add up to 140% of the territory! Said differently, the societal interests require more space than there is available.

A tiered planning system

These interests are managed through the national Planning Act (DMBG 2015:§1), which aims to secure a comprehensive planning that *"unites societal interests and facilitates the protection of the country's nature and environment, so that societal development can happen in a sustainable way"*. The act divides responsibility between the state, the five Danish Regions[2] and the 98 Danish municipalities. Most

[2] The names of the five Danish regions: North Denmark Region, Central Denmark Region, Southern Denmark Region, Zealand Region and Capital Region.

important is the formation of a planning hierarchy, which obligates plans and programmes to conform to plans of higher institutional tiers. This secures that all projects fit within the wider development goals. The institutions' responsibilities are described in multiple sector-specific laws[3]. Most often, the state and the regions produce wide, holistic plans for development, while the municipalities produce sector-specific plans and grant project permits.

3.2 Impact assessment in Denmark

The legal context

The European directives for EIA and SEA were introduced to Danish legislation in 1989 and 2004, respectively (Revsbech and Puggaard 2008). The requirements for EIA and SEA are currently implemented through various legal documents, but a proposal has been made to merge most legislation on IA in a joint act (DMEF 2015).

With respect to EIA, the requirements for projects are described in multiple legal documents. For some sectors such as water supply and farming, the requirements for EIA are to be found in the sector-specific acts (DMEF 2009; DMEF 2013d). Yet, most projects are subject to the EIA requirements of the National Planning Act (DMBG 2015) – as specified through the EIA declaration (DMEF 2014a). Generally, the permit-granting authority is responsible for both the screening procedure and for preparing the EIA. Yet, the Danish Ministry of Environment and Food – DMEF – may adopt the EIA competence for projects, which are of national importance or affect multiple institutions (DMEF 2014a).

With respect to SEA, the requirements are specified in the SEA Act (DMEF 2013c) and the ministerial SEA guidance (DMEF 2006). In short, SEA must be assigned to all plans and programmes within *"agriculture, forestry, fishery, energy, industry, transportation, waste management, water management, telecommunication, tourism, spatial planning and area use, which establish conditions for future projects"* of the kind mentioned in appendix 1 and 2 of the EIA declaration (DMEF 2013c:§3). The act specifies that responsible planning authority must prepare the SEA.

Peculiarities of Danish impact assessment

A distinctive feature of the Danish IA system is the role of the planners and administrative workers, who serve also as IA practitioners. Unlike in other countries, EIAs are made by the permit-granting institution rather than the project proponent. This might change with the coming amendment of the IA legislation, but for the time being, project proponents are only responsible of providing the information needed for conducting a meaningful assessment. With respect to SEA, the division of IA responsibilities means that planners often evaluate their own plans. There is room in the legislation for the institutions to hire external EIA or SEA consultants, but often public planners serve a *'double function'*.

[3] See an overview of the legislation and division of responsibility on www.mvfm.dk.

Another peculiarity of the Danish IA system derives from the National Planning Act (DMBG 2015), which states that some projects (subject to EIA) can only be permitted through *'plan appendices'* to municipal plans (subject to SEA). Hence, projects are often evaluated by both EIA and SEA – as pointed to in the most recent evaluation of European IA directives (European Commission 2009a; 2009b).

A last point about Danish IA is that it is cheap. Lyhne et al. (2015) argue that the Danish implementation of the European EIA directive has been characterised by a wish to fulfil only minimum requirements. With respect to SEA, the most recent evaluation of the SEA directive found that the costs of SEAs are lower in Denmark than in Slovenia, Estonia and Hungary (European Commission 2009b) – countries with a GDP per capita about three times lower than Denmark (World Bank 2015a).

3.3 Mining in Denmark

Products, sites and processes

Denmark is a country with minimal seismic activity and surface-near bedrock. Hence, the commercially available geology consist primarily of sedimentary materials, which settled during geological processes (Sørensen 2008). There are no ores of metals or deposits of precious stones. The marine territory of Denmark does hold oil and gas deposits as well as recent tests have pointed to the existence of deep shale gas deposits on land. Yet, this dissertation focuses exclusively on the onshore mining of the materials specified under the Mineral and Raw Materials Resource Act (2013b). These are sand, stone, chalk and various forms of clay – onwards referred to as *'raw materials'*.

Figure 3.1: A Danish gravel pit. The machinery for mining and sorting the sand and stone generate dust and noise in the countryside landscape. Photo: Rikke E. Biltoft.

Sand and stone make up an 85% mass fraction of all the mined raw materials (Statistics Denmark 2015b). Sand and stone are mined jointly in gravel pits similar to that of figure 3.1. The deposits were formed by glaciers, which transported vast amounts of material from the Scandinavian Peninsula during the last ice age, and which sorted the materials upon melting (Sørensen 2008). The sorting left behind stone and coarse fractions of sand, which today are widely used for foundations and as aggregates in concrete production. The sizes of gravel pits span from a few hectares to more than 100 hectares. During operation, heavy machinery is applied for excavating, sorting, cleaning and transportation.

Figure 3.2: A Danish lime pit. The white moon-like mining site is visible from far away. Aside from being noisy, lime pits generate fine white dust. Photo: Morten Bidstrup.

Figure 3.3: A clay pit for production of red bricks. The top-layer of the clay deposit is removed, after which the site is restored to its original purpose. Photo: Morten Bidstrup.

Chalk makes up a 10% mass fraction of the mined raw materials (Statistics Denmark 2015b). The resources were formed by the precipitation of dead shellfish in past times, and they are only near the surface on specific locations (Sørensen 2008). Chalk is used for cement production and other industrial purposes such as soil improvement and fodder. Most chalk in Denmark is used for cement production and dominated by a single player – Aalborg Portland (2015). Chalk is mined from lime pits – see figure 3.2 – which require machinery for excavation and transportation.

Clay makes up a 3% mass fraction of the mined materials (Statistics Denmark 2015b). Clay consists of fine particles, and the best deposits were formed by subtle settling processes in former glacial lakes (Sørensen 2008). Clay is used almost exclusively for brick production. The high level of chalk in the Danish geology makes most clay yellow when burned. However, the Danes have a strong preference for red brick houses, and most clay is thus mined where years of ground water flow has depleted chalk levels. This is generally the first 1-2 m of the clay layer. The shallow mining depth means that often clay pits are not as invasive as gravel pits or lime pits – see figure 3.3. At large, clay pits need only machinery for excavating.

The impacts of mining

Raw materials mining generates a range of impacts throughout the processes of establishing, operating and restoring a mining site – as accounted for by the Capital Region (2013). The establishment and closure of a mining site will always entail some sort of land use change, which may impact the biodiversity, food production and esthetical or historical value of the current land use. Moreover, the removal of the top soil and the subsequent excavation may decrease the natural groundwater protection. This is critical since the Danish water supply is based purely on groundwater. During operation, the heavy machinery may generate noise, dust and air pollutants in the local environment. Lastly, impacts are generated when the materials are distributed. The lorries leaving the mining sites generate traffic on countryside roads and cause issues of traffic safety in rural villages. The assigned CO_2 emissions are also significant due to the pure scale of the mining. Danish Statistics (2015c) estimates that the sector accounts for as much as 20% of the total transportation of goods (measured in ton·km) within the country.

Yet, the impacts of mining are not all negative. The sector generates revenue for rural communities, and the restoration of mining sites is often used as a possibility to create recreational value in a country, where the agricultural sector dominates the landscape. Mining sites have been planned as hunting grounds, mountain bike routes, put-and-take lakes, amphitheatres, energy storage facilities and even a rowing stadium. Moreover, recent research show that mining can improve biodiversity significantly both during and after the excavation process (KTC 2014).

A last point is that the impacts vary greatly. With respect to technology, it is clear that the mining of clay (figure 3.3) is less invasive than other types of mining. Some mining sites are small while others are big. Some mining pits are in operation for only a few years while others have permits spanning up to 40 years. With respect location, mines are more controversial when planned near conflicting interests, such as drinking water zones, biodiversity hotspots and valuable landscapes.

3.4 The regulation of Danish mining

The legal context

Mining activities are planned through the Mineral and Raw Material Resource Act – referred to as the *'Mining Act'* – which overall purpose is to ensure that exploitation of the raw material resources *"occurs as part of a sustainable development and according to a comprehensive weighting of interests"* (DMEF 2013b:§1). Central to the act is that commercial mining is not allowed without an extraction permit, which specifies the terms for site operation and restoration. The state grants permits for all mining on the marine territory, while the five Danish Regions plan and grant permits for onshore mining (DMEF 2014b).

The regional mining plans

The regulation of onshore mining is structured around the regional mining plans, which establish mining zones wherein contractors may apply for mining permits. Mining zones can span hundreds of hectares, while mining projects normally are not larger than 20 hectares. Mining zones regulate throughout the planning hierarchy by restricting the approval of infrastructure projects, urban development or any other action that could impact the access to the resources. Mining plans are made every fourth year and must secure raw materials for a minimum of 12 years' regional consumption. A schematic account of the planning process is presented in figure 3.4.

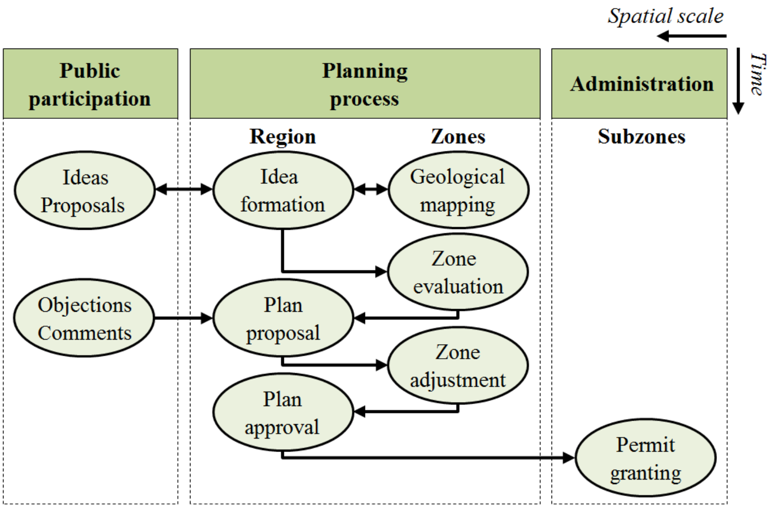

Figure 3.4: The planning process for onshore mining of raw materials in Denmark. The process alters between the three geographical levels: region, zone and sub-zone. The figure is modified from Bidstrup et. al (2016).

Planners start out with an idea formation phase, where proposals are made for both new zones and an overall strategy for how to supply the region in the years to come. The planners are during this phase supported by the public, private land owners and the mining industry, which all are invited to send in ideas and proposals. Zoning can only be legitimised if a sufficient amount of commercially accessible raw materials are present on the location, and thus the idea formation phase is supported by an on-going geological mapping. Each new zone is then evaluated and (potentially) used for making a full plan proposal. Next, a second public participation phase is initiated, where all interested parties are invited to send in comments or objections to the prioritisations made by the planners. Some zones are then adjusted before ultimately approving the final plan. Once the plan is approved, contractors can apply for mining permits for projects proposals. It is important to underline that the planners do not own the resources they zone. Hence, mining projects will only be proposed and initiated when there is a commercial interest from the land owners.

Impact assessment within Danish mining

The environmental effects of mining have made IA an integrated part of the planning process. Mining plans must be subject to SEA since they are within the topic *"industry"* and *"establish conditions for future projects"* (DMEF 2013c:§3). These SEAs are separated in their published form as an *'SEA report'* (accounting for the total impacts of the plan) and multiple *'zone reports'* (accounting for the impacts of each proposed zone). Proposed mining projects are subject to EIA legislation. An EIA is mandatory for all projects larger than 25 hectares or spanning more than 10 years, while EIA screening is required for the remaining projects (DMEF 2014a). The regional planners are authority for both the SEAs and EIAs.

Before initiating the PhD project in 2013, SEA had not been applied extensively within Danish mining planning. Legal requirements for SEA were established in 2004 (Revsbech and Puggaard 2008), but the Danish Regions were not formed as institutions before 2007 (Danish Government 2013). Though 9 years of SEA legislation and 6 years of regional planning could foster expectations of a well-founded practice, SEA had only been applied two times upon starting the project (2008 and 2012) due to the four-year planning cycle. The project developed when mining planners expressed that they struggled to apply SEA meaningfully to the 2012 plan, which for most regions marked the first extensive application of the tool.

4 IMPACT ASSESSMENT IN A WORLD OF SYSTEMS

The following chapter presents 'systems thinking' as a meta-theoretical framework. It opens with a presentation of what systems thinking is and how it is applied. The connection between systems thinking and IA is then accounted for and used to legitimise the onwards focus on life cycle thinking. At last, systems thinking is applied to conceptualise the regulatory context of the Danish mining planners.

4.1 Systems thinking as a theoretical framework

A systems view of the world

Many of the great discoveries share characteristics with the tale of Isaac Newton, who discovered the gravitational force under an apple tree. Though sitting in a complex environment, he reduced what he saw to a single element (apple) which moved from an elevated position (tree) to a lower one (ground). Yet, reductionism often fails to provide more elaborate analyses of events – especially with respect to complex societal problems (Meadows 2008). Take for instance the topic of how to reduce climate change. A reductionist may argue that climate change is caused by accumulated CO_2 in the atmosphere, and by such the solution is to reduce the rate of accumulation. Yet, most would reason that the topic is too complex for such a quick-fix solution. The current release of CO_2 is closely linked to our energy production and mobility, and quick solutions could thus limit the functioning and development of societies if not accompanied by alternatives to the carbon-based economy. The problem of climate change is systemic, and its solutions must consider both synergies and trade-offs. Though it is an extreme case, Capra (1996:3) emphasises:

> *"The more we study the major problems of our time, the more we come to realize that they cannot be understood in isolation. They are systemic problems, which means that they are interconnected and interdependent."*

System thinking breaks with the reductionistic sciences by advocating that the world consists of multiple functioning wholes, as opposed to single causal relationships (Boardman and Sauser 2013:27). The theory builds on the assumption that the world consists of interconnected biological, physical, economic, social and political systems, which all function within a grand system (Laszlo 1996). As Meadows (2008:97) writes: *"there are no separate systems. The world is a continuum"*.

The characteristics of a system

The broad spectrum of literature on systems theory and analysis contains diverse terminology and definitions. Yet, Meadow (2008:188) defines a system as:

> *"a set of elements or parts that is coherently organized and interconnected in a pattern or structure that produces a characteristic set of behaviours, often classified as its function or purpose."*

Key systems constituents are thus: function, elements, connections and boundary – see Meadows (2008:11-34) and Boardman & Sauser (2013:38-56). The *'elements'* are the building blocks of any system. They influence each other through *'connections'*, which allow the system to display a certain *'function'*. The system *'boundary'* is what separates the system's interior from its exterior.

The functioning of systems is complex, but important facets include hierarchy, variety, parsimony and feedback loops. The structuring of system elements in a *'hierarchy'* is important for system functioning because it simplifies the role of the individual system elements (Meadows 2008:82). Successful systems continue to function under stress, and a key to this success is their ability to adapt – referred to as '*variety'* – and constantly seek efficiency – referred to as *'parsimony'* (Boardman and Sauser 2013:86). Also, some connections may help stabilise systems by responding to the stress of a particular system element. This is referred to as a balancing *'feedback loop'* (Meadows 2008:28).

What is meant by theoretical framework?

The application of systems thinking as a meta-theoretical framework means that the theory provides a *'lens'* for contextualising problems and finding solutions. Through this lens, it will be argued that the mining planners act within an array of systems. Their plans are an element of both a *'resource system'* and the Danish *'planning system'*. The impacts of the plans are analysed by *'systems analysis'* in SEAs, which are elements within a greater *'IA system'* and ultimately aim to reduce the influences of the planning system on the receiving *'environmental systems'*. More concretely, systems thinking is applied to:

- Elaborate on what meaningful impact analysis entails (section 4.2)
- Legitimise the focus on LCA in IA (section 4.3)
- Contextualise the case of SEA in Danish mining (section 4.4)

The use of systems thinking is a bit atypical because I apply a theory home to the field of natural science to the societal context of meaningful SEA, which otherwise could be studied through a *'lens'* focusing on decision-making (March 1994), power (Morriss 2002) or strategy development (Cherp et al. 2007). Yet, I believe a systems approach can provide valuable lessons on the complex nature of the topic.

4.2 Impact assessment is systems thinking!

Few have linked systems thinking and IA in the scholarly literature. In fact, the brief viewpoint of Perdicoúlis (2016) appears to be the only work that describes the link explicitly. I am of the opinion that systems thinking is central to the purpose of IA – defined in section 2.2 as *"to identify the environmental consequences of proposed actions with the aim of facilitating sustainable development"*.

The *'identification of consequences'* is essentially an evaluation of how a proposed development (element) affects (connection) some entity of protection (element). From a systems perspective, the impact analysis of IA is an attempt to understand how system elements are connected with the aim of predicting system dynamics.

One hypothetical example of this could be a proposed establishment of a large gravel pit, which is subject to EIA due to its noise emission near a protected bird sanctuary – onwards referred to as the *'gravel pit example'*. In this case, the EIA must identify how the noise increase affects the sanctuary.

The *'environment'* is what IA ultimately aims to protect. It is a system element, but it is indeed also a functioning system *per se*. The sanctuary of the mining example is a functioning ecosystem, which functioning depends on many other elements and connections than those of the gravel pit. Additionally, the gravel pit may bring employment to the nearby community, which job market, economy and demography are systems as well. Thus the purpose of IA can be depicted as the study of how a proposed change in a subsystem (gravel production) may generate consequences throughout a larger system (life around the gravel pit).

The idiom *'sustainable development'* derives from the verb *'to sustain'*, which meaning from a systems perspective relates to the continued function of a system under stress. Hence, the proposed gravel pit facilitates local sustainability only if it does not affect the interrelated subsystems (for instance the local community or the sanctuary) to an extent where their functioning is jeopardised.

4.3 Life cycle assessment for wider systems analysis

A good impact analysis thus requires an understanding of how systems are interconnected. In this respect, authors have argued for years that the environmental stress of local decision can extend beyond the proximity of the development site through the connections of product systems. The argument is that developments may influence the demand for products, which in today's globalised economy is supplied from and produced in far regions, where impacts equally occur (Tukker 2000).

The raw materials production of the exemplified gravel pit, for instance, requires an intake of various machines (for mining and sorting), which all caused environmental stress elsewhere during their manufacturing and production. The impact analysis of some IAs could benefit from considering such impacts occurring across product supply chains – onwards referred to as Life Cycle Thinking (LCT). Multiple authors have advocated the use of the tool Life Cycle Assessment (LCA) for such system analysis (Björklund 2012; Finnveden et al. 2009; Loiseau et al. 2012; Tukker 2000).

Unlike IA, which focuses on proposed actions, LCA focuses on the impacts of products – a term covering both goods and services. The International Organization for Standardization (2006a:2) defines LCA as the *"compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle"*. An LCA application consists of the following four phases:

1) Definition of goal and scope
2) Inventory analysis
3) Impact assessment
4) Interpretation of results

Phase 1: Definition of goal and scope

The first phase is where all the model characteristics are determined. With a reference to the gravel pit example, one may want to analyse the impacts of producing gravel for the foundation of a particular highway. This study *'goal'* guides the choice on the product system under study (gravel pit) and the included life cycle processes (mining and sorting) – referred to as the *'scope'*. One must then define the *'function'* of the product system under analysis and the *'functional unit'* (FU) to which all impacts are quantified (European Commission 2010:60). In the example, the function of the system is to provide foundation materials, while the FU could be *'1 ton of foundation materials ready for use'*.

Phase 2: Inventory analysis

The second phase entails the compilation of all the inputs and outputs assigned to the FU – referred to as a *'life cycle inventory'* (LCI). The *'inputs'* comprise the intake of materials, energy, land and other resources related to the FU, while the *'outputs'* comprise the finished products in combinations with the related emissions of waste (European Commission 2010:196). In the gravel pit example, inputs could be *'machinery'* for mining and sorting materials and *'fuel'* for energy. These inputs lead to an output of the FU in combinations with various waste emissions (for instance particular pollutants and CO_2 from the combustion of fuel).

Inputs and outputs are categorised as those deriving from processes in either the system's foreground (specific to the system) or background (supporting the system) – see European Commission (2010:96). In the gravel pit example, the processes of mining and sorting the materials are in the foreground system, while the processes assigned to manufacturing and delivering the related machinery are in the background system. Extensive compilation of inputs and outputs from especially the background system can be a daunting task, and it is thus often aided by electronic databases (Finnveden et al. 2009).

Phase 3-4: Impact assessment and interpretation of results

The third phase of an LCA is to transform the output of emissions from the LCI into impacts – for instance, the transformation of various greenhouse gas emissions (CO_2 and CH_4) into global warming. For this purpose there exist many models, which all can be used in combination with LCI databases in modern LCA software (Finnveden et al. 2009). Phase three yields a long list of quantified impacts with respect to some pre-defined categories, which optionally can be weighted to provide some basis for impact comparison (European Commission 2010:282). At last, the results are interpreted and recommendations for decision-makers can be made.

4.4 A systems perspective on Danish mining

Systemic understanding can be facilitated by graphically displaying a system's elements and connections in a *'system diagram'*. There exists an array of methods for drawing system diagrams (Perdicoúlis 2016). One method is that of Boardman and Sauser (2013) which focuses on the qualitative connections between elements. A second method is that of Meadows (2008) which focuses on the quantitative flows and accumulation of a measurable entity. These two methods are referred to as the *'generic method'* and the *'resource-oriented method'*, respectively. They are applied onwards to contextualise the case of SEA in Danish mining.

Generic method: The function of the IA system

The method of Boardman and Sauser (2013) has a focus on mapping how system elements are interconnected to form a system with a particular function. The method describes elements with noun phrases and connections with verb phrases. It is used in figure 4.1 to depict the influence of the IA system on the planning system for Danish mining. EIAs and SEAs are applied to different elements of the planning system to lower the impacts on the receiving environment. The rationale of the SEAs is that improved mining plans yield less impacting mining projects.

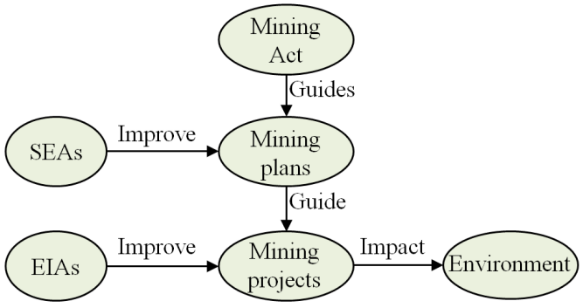

Figure 4.1: A generic system diagram of how the IA system influences the planning system for mining. The circles are element and the arrows are connections.

Drawing on the theory of section 4.1, it is clear that the IAs (elements) aim to improve (connection) the mining actions (elements) with the purpose (function) of reducing their impact (connection) on the receiving environment (element). The functioning of the system is ensured by various means. The IA system applies SEA and EIA (variety) to different elements of the planning system (hierarchy) as cost-effective as possible (parsimony).

Resource-oriented method: There is a need for mining!

The method of Meadows (2008) differs from the generic method by its focus on quantifying how resources accumulate or deplete in stocks (measurable elements) by in- and outflows (measurable connections). The method is applied in figure 4.2 to depict the Danish raw materials resource system. The rate of construction (flow) requires an intake of raw material products (stock). These raw materials can be supplied either by mining raw materials from virgin resource deposits (flow) or by recycling demolition waste (flow).

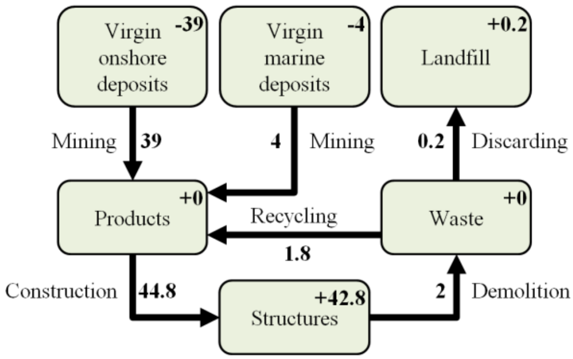

Figure 4.2: A system diagram of the raw material resource system. The yearly flow (arrows) and stock accumulation (boxes) of raw materials in Denmark are depicted in 'million tons'.

Over the past five years, the annual mining of materials has been 39 million tons onshore and four million tons offshore – assuming a material density of 1.5 ton/m^3 (Statistics Denmark 2015b). With respect to recycling, the latest waste statistics state that 90% of all demolition waste is recycled and that the total annual amount adds up to roughly two million tons – when subtracting wood, glass, metals and other products not deriving from raw materials (DMEF 2013e).

Drawing on the theory from section 4.1, it is evident that the functioning of the resource system depends on the rate of mining and demolition. The recycled materials add up to only 4% of the current demand for raw materials – an optimistic estimate since it is doubtful whether all of the recycled materials can substitute virgin materials. Roughly 42.8 million tons of raw materials accumulate in Danish structures annually, and it can thus be concluded that there is a continuous need for mining virgin deposits with the current rate of construction and demolition.

Combining the methods: the responsibilities of the mining planners

The two methods represent different schools of thought on how to depict systems. They are combined in figure 4.3 to form a conceptual model of the regional mining planners' responsibilities with respect to the raw material resource system.

Drawing on the theory from section 4.1, one can consider the responsibility of the Danish mining planners a grand feedback loop which stabilises the raw materials resource system by ensuring a sufficient and socially accepted outflow of raw materials from the onshore resource deposits. The mining plans respond to increases in demand for raw material products, while the mining permits is what ultimately controls the flow of virgin materials to the product stock. Geological mapping ensures that the stock of resources in the virgin deposits is not depleted.

Though the Danish Regions are legally obliged to secure a sufficient supply of raw materials, their responsibilities cover only a corner of the raw material resource system – see figure 4.3. They have no authority to influence the rate of construction (flow to structures), demolition (flow to waste) or recycling (flow back to products). They control only one means of system regulation: *onshore mining*.

There are currently ministerial wishes for the planners to reduce the impacts of onshore mining as much as possible (DMEF 2014c), but this is getting ever-harder as more deposits are depleted. The number and scale of conflicts are expected to increase since many of the remaining deposits are situated in zones with other spatial interests. The only viable way of maintaining the product stock, while reducing the impacts of onshore mining, is to increase the flow from the marine resource deposits, and the Danish Regions (2014) have thus requested a national mining strategy that includes maritime mining. Reports have been published on the possibilities and restrictions for further maritime mining (DMEF 2013a; RKCER 2013), and some even talk about a *"hidden treasure trove"* (Jensen and Nielsen 1998). Yet, the obligations of the Danish Regions remain unchanged for now.

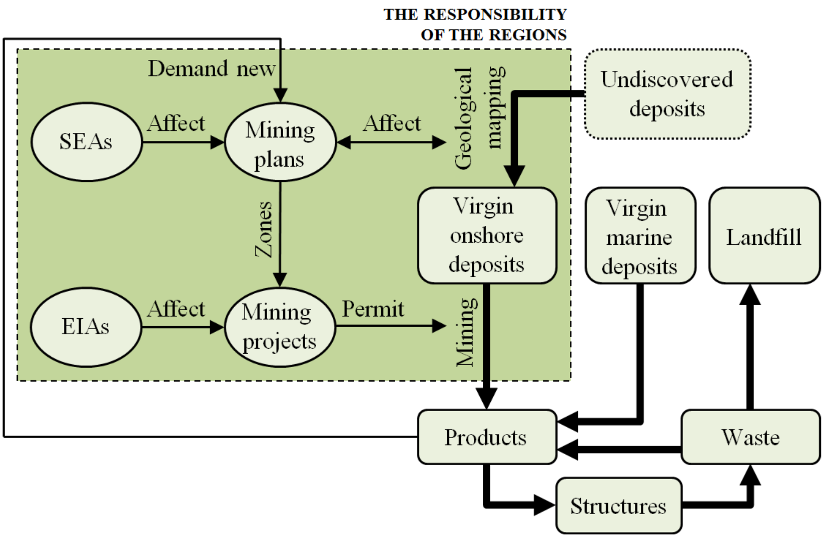

Figure 4.3: A conceptual system diagram of the Danish Regions' legally assigned responsibility with respect to the wider raw material resource system. Circles are elements, thin arrows are 'connections', thick arrows are 'flows' and boxes are 'stocks'.

A last point to make about figure 4.3 is that it illustrates how the Danish mining planners act within a world of systems. Representing a system *per se*, mining plans aim to secure the functioning of the raw material resource system. The planners apply different tools of the IA system with the purpose of minimising the stress on the interconnected social, economic and bio-physical systems.

PART II

RESEARCH DESIGN

5 STATE-OF-THE-ART ON SELECTED IMPACT ASSESSMENT ISSUES

This chapter opens with a section on how the following IA issues of Danish mining were selected as topics to explore further: alternatives, cumulative effects, grey screening practices and application of LCA. The scholarly state-of-the-art on each issue is then described. The chapter yields five knowledge gaps which point to where further research could contribute with insight. These gaps are used when formulating the research questions of the dissertation in chapter 6.

5.1 Identification of impact assessment issues

The research of the PhD took point of departure in some IA issues, which were carefully selected for further exploration. Two issues were raised as *'frustrations'* by the mining planners upon initiating the project while two others developed from my *'observations'* on the practices of the sector.

Frustrations: assessment of alternatives and cumulative effects

The catalyst for the project was the mining planners' frustrations on how to assess alternatives and cumulative effects (CE). With respect to alternatives, the prevalent opinion was that mining plans represent the optimal solution since they are the product of careful considerations. With respect to cumulative effects, planners expressed that they have limited power because they only establish zones for mining. The location and timing of the mining activities, which could act cumulatively with other activities, ultimately depend on the concrete project proposals of private proponents'. In short, the mining planners found that the assessment of alternatives and CE is often not meaningful.

Observation: few environmental impact assessments

My first observation was that few EIAs appear to follow the mining SEAs. One could interpret this absence as a sign of an environmentally sound and non-intrusive mining sector. Yet, I was met with an alternative and indeed very different rationale when questioning the phenomenon in the first months of the PhD.

The 2013 Raw Materials Mining Assembly[4] entailed a field trip to a gravel pit, which according to the NGO Danish Raw Materials (Danske Råstoffer) was to be considered a prime example of good practice. The gravel pit is displayed on the front page of the dissertation in the lower left corner of the picture. The site covers more than 100 hectares, where raw materials are mined both above and below the groundwater table. The mining occurs no more than 50m from a suburban street in a medium sized village, which is surrounded by other active mining projects. The spatial extent, groundwater interference, urban location and cumulative impact of

[4] The Raw Materials Mining Assembly (*Råstofårsmøde)* is an annual event organized for planners, industry representatives, consultants and academics working within Danish mining.

this gravel pit could each independently legitimise an EIA. Yet, the proponent had never been met with such a claim! Upon asking him why, he revealed that he had purposely applied for multiple smaller, consecutive mining permits, which then all had passed the EIA screening. He explained that he sees EIA as nothing but an expensive and time-consuming administrative burden since it is much more effective to formulate projects in close dialogue with the local authority and the neighbours. Though this argument is rational, the scale of his current mining impacts raised a pressing question: *Has he been a bit too close with the local authority?*

The prevalence of this IA practice within the mining sector was later confirmed by the individuals granting mining permits in four of the most mining municipalities[5] (the municipalities granted mining permits until the 2014 revision of the Mining Act) and the consultant Jakob Christensen (2015). All defend the practice as a common way to secure better projects in a resource-efficient manner. Yet, the deliberate IA circumvention of this particular project proponent provoked me to such an extent that pre-screening practices was selected as an issue to study.

Observation: poor assessment of indirect, global impacts

My second observation was that the SEAs focus mostly on the local impacts of new mining zones – as seen when comparing the SEA reports and zone reports. With respect to the written extent, the zone reports' add up to hundreds of pages while generally the SEA reports cover only around 15 pages. With respect to content, the zone reports are detailed while the SEA reports have a summarising and not particularly analytical format. In fact, the SEA reports focus mostly on describing the legal context and background for the SEA.

The transport of raw materials appears to be the only impact scaled up to the regional level. The contributions to indirect and global impacts are largely not addressed (Bidstrup and Hansen 2014), though there are some rationalisations on the link between transport and global warming. The only exception is the Capital Region's SEA (2013), which applies LCA to concretise and quantify the climate impacts of some supply scenarios. I saw an analytical potential in such use of LCA, and the introduction of LCA in IA was thus selected as an issue to study.

5.2 Alternatives in strategic environmental assessment

The assessment of reasonable alternatives is a central IA element (IAIA 1999; 2009), which is mandatory according to both the European SEA Directive (European Parliament 2001) and the Danish implementation hereof (DMEF 2013c). Alternatives may range from fundamental *'WHY?'* alternatives to more operational *'HOW?'* alternatives (Stoeglehner 2010; Therivel 2010). They may represent alternative strategies, value-choices, locations, technologies or timing (González et al. 2015). A myriad of alternatives may exist for any given development. Yet, these are only *'reasonable'* when they are politically realistic, implementable within the timeframe, and both economically and technically viable (González et al. 2015).

[5] I contacted the municipalities: Kalundborg, Roskilde, Aabenraa and Silkeborg.

Stoeglehner (2010) describes the alternatives as what ultimately makes SEA strategic. The identification of alternatives may help proponents to consider more environmentally sound options when shaping their development, while the assessment of alternatives may provide decision-makers a yardstick for comparing the impacts to what could have been. As Steinemann (2001:3) frames it: *"the quality of a decision depends on the quality of alternatives from which to choose"*.

Therivel (2010:130) describes the development and assessment of alternatives in SEA as a *"key stage"*, which is *"very hard to do well, and very easy do to badly"*. The consideration of alternatives in SEA has been found to be universally poor (González et al. 2015; Pope et al. 2013; Tetlow and Hanusch 2012), as supported by experiences from Canada (Noble 2004; 2009), Austria (Stoeglehner 2010), Finland (Soderman and Kallio 2009), Italy and England (Bragagnolo et al. 2012). When assessed, alternatives are often subjective, narrowly defined and retrofitted to support a preferred option (González et al. 2015; Steinemann 2001). Fundamental alternatives are rarely considered and no-built alternatives are not popular by the planning authorities, whose purpose is to facilitate development (Steinemann 2001). In practice, the assessment of alternatives is often reduced to a vague assessment of the 0-alternative (Bragagnolo et al. 2012; Soderman and Kallio 2009) following the mantra: *doing something is better than doing nothing*.

With respect to causality, the most recent evaluation of the European SEA Directive finds that practitioners struggle with the concept *'reasonable alternative'* because there is little legal guidance (European Parliament 2001: 129). Others find that the consideration of alternatives is poor because practitioners apply SEA too late in the planning process (Bragagnolo et al. 2012; Stoeglehner 2010). Lastly, some echo the words of the mining planners. Soderman and Kallio (2009) find Finish planners expressing that there are no resonable alternatives in their planning context.

Knowledge gap: (1) Why are alternatives poor?

It has been extensively documented that alternatives often are poorly considered in SEA practice, but few appear to question the causality. More research is needed on whether non-strategic SEA applications and difficulties in grasping the term *'reasonable'* are the only explanations for this worldwide SEA deficiency.

5.3 Cumulative effects in strategic environmental assessment

As it was the case with alternatives, the assessment of CE is an important IA element (IAIA 1999), which is mandatory according to both the European SEA Directive (Parliament 2001) and the Danish implementation hereof (DMEF 2013c). There exist different definitions of CE, but a commonly used one is: *"changes to the environment that are caused by an action in combination with other past, present and future human actions"* (Hegmann et al. 1999: 3). A key point in this definition is that CE assessment focuses on the impact on and capacity of the receiving environment – not only the development under study (Duinker and Greig 2006; Gunn and Noble 2011; Hegmann and Yarranton 2011; Therivel and Ross 2007). In CE assessments, the environment is analysed as Valued Components, or VCs

(Canter 2015; Canter and Ross 2010; Johnson et al. 2011; Olagunju and Gunn 2015). Whether representing a bio-physical or socio-economic indicator, the receiving environment is what an IA ultimately aims to protect, and thus Duinker and Greig (2006:157) argue that CE are *"the only real effects worth assessing"*.

Still, CE assessment remains one of the most enduring IA challenges (Gunn and Noble 2011). Also SEAs show poor performance (Tetlow and Hanusch 2012), as studies from Canada (Noble 2009), Finland (Soderman and Kallio 2009), Germany (Weiland 2010), Italy and England (Bragagnolo et al. 2012; Cooper 2011) document. This is critical because the potential for better CE assessment was among the *"underpinning justifications"* for developing and implementing SEA in legislation (Pope et al. 2013). SEA is widely considered the most appropriate IA tool for CE assessment because it applies to developments with broad boundaries, which may cover the many actions causing CE (Duinker and Greig 2006; Gunn and Noble 2011; Johnson et al. 2011; Therivel 2010).

Much research has explored why CE assessment is poor in SEA. Some argue that the poor CE assessment may derive from late, non-strategic uses of SEA (Cooper 2011; Hegmann and Yarranton 2011) or a lack of conceptual understanding (Gunn and Noble 2011) and legal guidance (Weiland 2010). Others argue that there may be an institutional side to the phenomenon since the means for regulating the many actions causing CE are often segmented across institutions (Chilima et al. 2013; Kristensen et al. 2013; Sheelanere et al. 2013). A last hypothesis is that the intrinsic SEA focus on evaluating (and approving) developments may clash with the VC-oriented focus of CE assessment (Duinker and Greig 2006; Gunn and Noble 2011).

Knowledge gap: (2) Do plan boundaries restrict CE assessment?

The causality for poor CE assessment in SEA has been extensively studied, but none appear to question the following underpinning assumption: *SEA is appropriate for CE assessment because it applies to developments with wide boundaries.* There is a need for elaboration on whether these boundaries indeed can be expected wide enough to encompass the activities causing CE and what happens when they are not.

5.4 Impact assessment screening and 'grey' practices

Within the scholarly literature, multiple studies are published on the factors influencing the effectiveness and quality of IA – as listed by van Doren et al. (2013). Yet, most studies appear to be confined to what goes on once an IA has been deemed necessary. The initial process of screening and the mechanisms surrounding it has not been granted much attention (Pinho et al. 2010; Weston 2000).

The great majority of screening processes do not lead to a need for IA (McGillivray 2011; Nielsen et al. 2005; Wood and Becker 2005). There can be many reasons for this. As it was the case with Danish mining, the literature points to the existence of an explanation different from that of non-intrusive development proposals. Often, there is a willingness to pass proposals because IA requires resources from both the proponent and the authority (João and McLauchlan 2014; Macintosh and Waugh

2014; Nielsen et al. 2005; Weston 2000). Weston (2011) argues that among some development authorities there exists a culture of IA resistance. The term *‘grey IA’* is often used in Denmark to describe the adjustment of development proposals prior to or during screening processes with the aim of avoiding IA (Christensen 2015).

Screening can be done by two approaches: by application of threshold criteria, beyond which significant impacts can be expected, and by case-to-case evaluation (Pinho et al. 2010). Grey IA takes place within both types of screening. With respect to the threshold approach, Pinho et al. (2010) find that often proponents propose developments that stay just below the criteria. Also, there is a continuous problem within the EU of proponents dodging EIA requirements by slicing up bigger projects to multiple smaller ones (European Commission 2009a:137). With respect to the case-by-case approach, grey IA is possible because such screening ultimately relies on a discretionary judgement (Weston 2000; Wood and Becker 2005), which can be subjective, normative and made within a local-political context (Lawrence 2007).

Grey IA is common under the American NEPA Act, but it does not belong to formal IA practice in most other countries (McGillivray 2011). Few have studied the phenomenon, but its rationale is disputed. One argument is that is undermines the very purpose of IA when applied as a means of omitting IA requirements (McGillivray 2011; Weston 2011). A counter argument is that it facilitates early integration of environmental concern (McGillivray 2011; Nielsen et al. 2005).

Knowledge gap: (3) How prevalent and influential is grey IA?

Little is still known about pre-screening practices outside an American NEPA context. The work of Nielsen et al. (2005) remains the only European study with an exclusive focus on the topic, but the data is 15 years old and has a clear bias on agricultural projects. Moreover, all studies are on EIA. None appear to have studied whether the same kind of adjustments takes place in SEA.

5.5 Life cycle assessment in impact assessment

LCA application in IA has been advocated for nearly two decades (Owens 1997; Tukker 2000). Multiple LCA studies evaluate spatial developments normally covered by IA – such as sanitary systems (Lemos et al. 2013; Niero et al. 2014) and waste treatment systems (Prapaspongsa et al. 2010). Yet, few appear to apply the tool in an IA context explicitly. Important facets of LCA application in IA are: why and how should such tool integration occur?

With respect to *‘why’*, the prevalent argument is that LCA can extend the analytical scope of IA to include long-term, global impacts occurring across supply chains – see section 4.3. At large, the material on LCA-IA integration has the character of theoretical advocacy in research papers (Finnveden and Moberg 2005; Finnveden et al. 2003; Loiseau et al. 2012; Tukker 2000) and textbooks (Fischer 2007:47; Therivel 2010:316) or the character of a single LCA demonstration on a specific plan (Björklund 2012) or project (Cornejo et al. 2005; Manuilova et al. 2009; Židonienė and Kruopienė 2015) proposal.

With respect to *'how'*, little guidance existed on LCA-IA integration upon starting the PhD project. Since then, two papers have been published. The first paper is Loiseau et al. (2013), which presents an LCA procedure for evaluating the multiple imbedded functions of an area in land use planning. The second paper is Zidonienė and Kruopienė (2015), which presents a procedure for evaluating proposed manufacturing projects (subject to EIA). Both authors argue that LCA may facilitate an evaluation of the impacts occurring beyond the proximity of the development site. Zidonienė and Kruopienė (2015) further argue that LCA may help identify EIA alternatives since minor modifications can generate diverse impacts elsewhere.

Knowledge gaps: (4) How prevalent is LCT currently in IA practice?
(5) How can LCA be applied in SEA?

There are generally few studies on LCA-IA integration. The *'why'* facet appears to be most explored in the academic literature, but most studies are not rooted in observations from actual IA practice and none compare the analytical perspective of LCA to that which would have existed otherwise. Thus little is still known about what benefits LCA application can bring to the current IA practice. With respect to the *'how'* facet, there exists no procedure for integrating LCA in SEA.

6 RESEARCH QUESTIONS

This chapter presents the central research question of the dissertation and its three underpinning sub-questions, which are rooted in the knowledge gaps of chapter 5.

6.1 The central research question

The project sparked from the Danish mining planners' frustrations on how to apply SEA in a way that suits their planning context and adds value to decision-making, but it also aimed to generate knowledge of wider international relevance. Hence, the central research question was formulated as follows:

What lessons can be learned from Danish mining on
meaningful application of strategic environmental assessment?

The research responds to the problem of SEAs showing deficiencies worldwide (see section 2.3) by analysing the case of Danish mining for wider *'lessons'* – defined as a contribution to the state-of-the-art. Thus the research focused on the *'knowledge gaps'* of chapter 5 through three research sub-questions.

6.2 The three research sub-questions

First, it was found that more knowledge is needed on why the assessment of alternatives and cumulative effects is poor in SEAs worldwide. Hence, the case of Danish mining can provide *'lessons'* through question (a):

a) *Why is the assessment of alternatives and cumulative effects poor in the strategic environmental assessments of Danish mining?*

Second, it was found that little is known about the prevalence of the '*grey*' screening practices and the analytical potential of applying LCA in IA. Indirectly, Danish mining can thus provide *'lessons'* through question (b):

b) *How representative are the observations on 'grey' screening practices and lacking life cycle thinking within Danish mining?*

A third and last point from chapter 5 is that currently there exists no procedure for applying LCA in SEA. Danish mining can therefore provide *'lessons'* through the answer to question (c):

c) *How can life cycle assessment be applied in the strategic environmental assessments of Danish mining?*

The research sub-questions respond to the central research question by exploring why SEA is not meaningful, how the planners then act and what can be done.

7 METHOD

This chapter presents the method for answering the research questions and the approach to collecting and analysing data. It accounts for how the studies relate to Danish mining and cover the many facets of meaningful SEA.

7.1 A foundation of five studies

The dissertation is based on five published studies. Each of these addresses a *'knowledge gap'* of chapter 5. Table 7.1 shows how the studies relate to the research sub-questions of chapter 6.

STUDIES	RESEARCH SUB-QUESTIONS
1. Bidstrup and Hansen (2014): *The paradox of SEA* **2.** Bidstrup et al. (2016): *CE in SEA – the influence of plan boundaries*	**a.** Why is the assessment of alternatives and CE poor in the SEAs of Danish mining?
3. Bidstrup (*in press*): *The 'grey' assessment practice of IA screening – prevalence, influence and applied rationale* **4.** Bidstrup (2015): *LCT in IA – current practice and LCA gains*	**b.** How representative are the observations on 'grey' screening practices and lacking LCT within Danish mining?
5. Bidstrup et al. (2015): *LCA in spatial planning – a procedure for addressing systemic impacts*	**c.** How can LCA be applied in the SEAs of Danish mining?

Table 7.1: The relation between the five studies and the research sub-questions. The first two studies feed into question (a), the two next studies feed into sub-question (b) and the last study feed into sub-question (c).

7.2 The role of Danish mining

All five studies developed from the narrative of the Danish mining planners, who struggle to apply SEA meaningfully. Flyvbjerg (2006:237) defends such a research approach by arguing that good narratives can facilitate a truthful description of the real world, which often is both complex and contradictory. Danish mining serves as a broad study field, from which it is believed wider SEA *'lessons'* can be learned. The dissertation is structured around five case studies, which vary in their typology and relation to Danish mining – as accounted for in this section.

With respect to typology, Yin (2002:5) categorises case studies as follows:

- **Explorative** studies define questions or hypotheses for further research.
- **Explanatory** studies answer questions and describe causal relationships.
- **Descriptive** case studies describe phenomena.

Danish mining was treated as a grand *'explorative'* case study during the process of identifying IA issues in the initial phases of the project – see section 5.1. The case studies 1 and 2 (on alternatives and CE) are *'explanatory'* because they aim to find the causality for common SEA deficiencies. The case studies 3 and 4 (on LCT and grey IA) are *'descriptive'* since they aim to describe whether Danish IA as a whole shares characteristics with the practices of the mining sector. Hence, the five studies comprise two explanatory case studies, two descriptive case studies and one methodological study which does not fall into the categories of Yin (2002).

The studies' relation to Danish mining is illustrated as a methodological *'distance'* in figure 7.1. The studies 1 and 2 (on alternatives and CE) are closest to the study field because they have an explicit focus on the practices of the mining planners. Study 5 (on LCA-SEA integration) is further from the study field. It tests a proposed procedure on Danish mining, but its focus is methodological. The studies 3 and 4 (on LCT and grey IA) are furthest away from the study field because they do not focus on Danish mining *per se*.

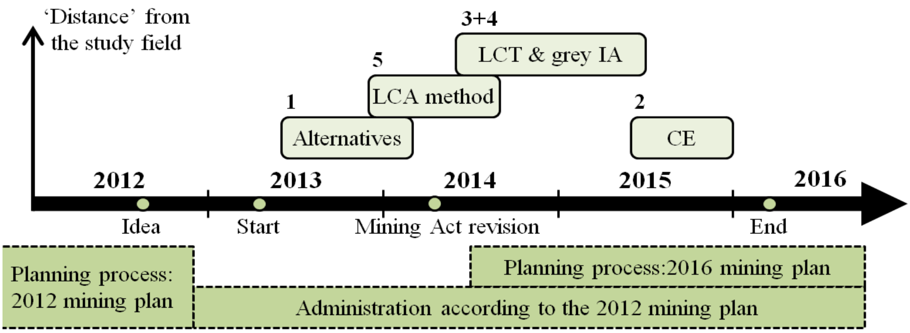

Figure 7.1: The PhD project ran parallel to the planning process (green boxes). Some of the studies (round boxes) are 'distanced' further from the study field than others.

7.3 The research approach

The research of the PhD project is *'applied'* since all studies relate to the societal issue of meaningful SEA. The research is also *'participative'* since I relied strongly on the inside knowledge and perspective of the Danish mining planners. A last point is that the research builds on *'cumulative knowledge creation'* through the linkage between the five studies and the scholarly literature – see chapter 5.

Douven (2011) states that within the theories of science there exist three approaches to inference – as briefly accounted for below:

- **Deduction** is when a universal premise is applied to project the causal relationship of a case.
- **Induction** is when an observed causal relationship of a case is applied to establish a universal premise.
- **Abduction** is when an observed effect is applied with a universal premise to predict the likely cause.

My PhD project applies an inductive approach since it collects empirical data on IA practices within a delimited study field with the aim of providing *'lessons'* of wider relevance. It is hypothesised that the five case studies yield results (on alternatives, CE, LCT and grey IA), which can induce universal SEA theory and deductively apply to the behaviour of other non-related SEA applications.

Altering between test and dialog

On a more operational level, the research builds on a participative and cyclical research model, which alters between dialog and test – as illustrated on figure 7.2. Work meetings with the mining planners, technical visits to mining sites and engagement in seminars and discussions with practitioners set the scene for a serious of questions, which were tested in various studies. The dissemination of the obtained results was projected in two directions: 1) to scientific journals for feedback and publication, and 2) back to the mining planners and IA practitioners for further dialogue – thus re-entering the research cycle. Some questions developed after several loops. For instance, the sector-wide scale of the grey screening practices was realised only after the results on alternatives (study 1) were presented. Likewise, the idea for exploring LCT in IA developed upon presenting the results from the application of the proposed generic procedure for LCA in SEA (study 5).

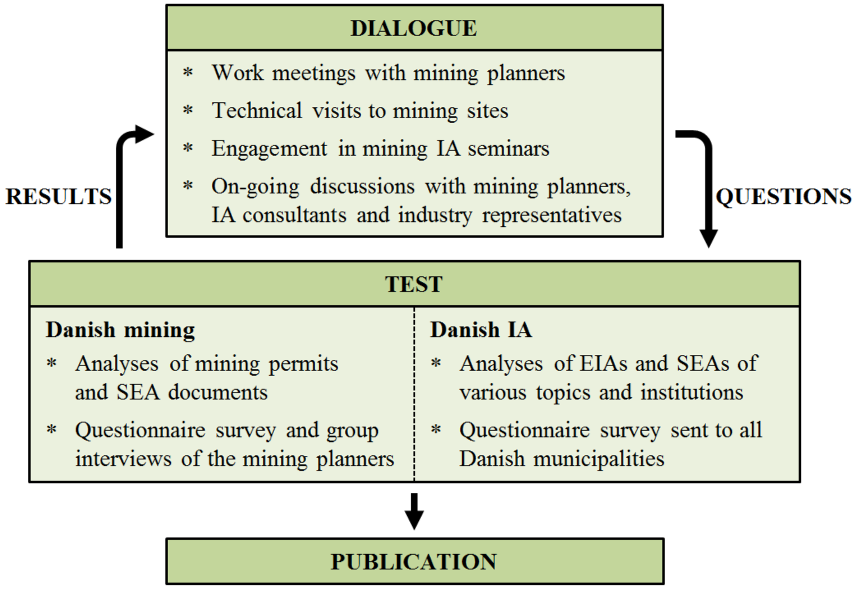

Figure 7.2: The research model of the PhD project. Dialogue with the Danish community of mining planners and IA practitioners yielded questions for testing. The results of these tests were published but also validated and challenged in further dialogue.

Application of mixed methods

With respect to possible test methods, Maxwell (2005:22) argues that quantitative methods are suitable for documenting the extent of a certain phenomenon while qualitative methods are suitable for exploring causal explanation as well as the meaning and context of certain events. Both of these method categories are used. The studies 1 and 2 apply qualitative methods (interviews, dialog and document analysis) because they focus exclusively on the context of Danish mining and the imbedded restrictions for meaningful assessment. The studies 3 and 4 apply quantitative methods (analysis of numerous IA documents and questionnaire responds) because they aim to explore the extent to which the observations from Danish mining represent the practices of Danish IA as a whole. Study 5 applies a combined approach since a quantitative LCA is taken as point of departure a qualitative discussion of the wider application of a procedure for integrating LCA in SEA. In reality, though, all studies had both quantitative and qualitative elements.

Role as researcher

The interaction between me (researcher) and the mining planners (study field) points to the importance of the project's social setting. Drawing from an broad literature review, Kørnøv et al. (2011) argue that there exist three modes of research:

1. Research that is autonomous and independent of societal partners.
2. Research that is driven by societal partners.
3. Research that has an outset in the problems of societal partners, but which goals and methods are negotiable.

One's research rarely belongs 100% to only one mode, and it is possible to alter between modes. Yet, Kørnøv et al. (2011) argue that mode three research within the field of SEA brings the opportunity to act as a '*change agent*' because one can provide critical feedback and change attitudes within the funding organisations.

My PhD builds on mode 3 research since it initiated with an outset in the SEA difficulties of the mining planners but then developed rather autonomously. This autonomy allowed me to make the studies 3 and 4 (on grey IA and LCT), which focused little on the study field *per se*. My research moved slightly towards mode 1 during these studies – a period where I was distanced also geographically from the study field due to a stay at the University of California, Santa Barbara. When applied, the mode 3 research proved to be influential. The planners would object if certain conclusions were mistaken (a validating role) or was not meaningful within the context they make SEA (a challenging role).

7.4 The data collection and analysis

The research data can be categorised as being either contextual (from the *'dialogue'*) or analytical (from the *'test'*). The contextual data were collected in an informal and participative process, which allowed me to gain an understanding of the study field and served to both validate and challenge results. The analytical data collection was more concrete because these data are all assigned to the five studies.

Contextual data

Two rounds of formal work meetings were conducted – during which I visited each of the regions to discuss SEA difficulties and early results. The first round took place in October 2013 and the second round took place in June 2015. The meetings were complemented with on-going, informal communication by e-mail and phone.

Further knowledge on Danish mining was gained during seven field trips to:

1. A gravel mining site and a brickworks in Southern Denmark. Focus was on evaluating the impacts of production facilities – Sept. 2013.
2. A laboratory for geological samples near the city of Vejle. Focus was on how to secure materials of adequate quality – Sept. 2013.
3. A chalk mining site near the city of Aalborg. Focus was on understanding the impacts of production and means of distribution – Oct. 2013.
4. Various production sites for clay near the city of Randers. Focus was on understanding how impacts differ from site to site – Oct. 2013.
5. A gravel mining site in the intensively mined areas near the city of Sorø. Focus was on understanding the process for granting permits – Oct. 2013.
6. An industrial dock in Copenhagen for landing maritime and imported materials. Focus was on understanding transport mechanisms – Oct. 2013.
7. A large nature restoration project of an old peat mining site near the city of Aalborg. Focus was on the opportunities of post-mining sites – Sept. 2014

The contextual data on the peculiarities and issues of Danish mining were further complemented by my attendance in seminars, where my results were presented, discussed and challenged. I attended the annual Raw Materials Mining Assembly (*'Råstofårsmøde'*) three times, the annual Environmental Assessment Day (*'Miljøvurderingsdag'*) four times, and the annual conference of the International Association for Impact Assessment (IAIA) three times.

Analytical data

Data for research sub-question (a) were collected and analysed as described in the studies 1 and 2. All five 2012 mining plan SEAs as well as 15 zone reports were analysed for the extent to which assessments of alternatives and CE are documented in writing. This analysis was complemented by a questionnaire survey and five semi-structured group interviews. The survey focused on mining plan alternatives and the strategic nature of the SEAs. It was responded to by nine key-planners, who all elaborated on their responses in follow-up discussions. The five interviews focused on the mining planners' understanding of CE, their current practices on CE assessment and the extent to which their sector-specific planning context limits their considerations. Each interview took around 50 minutes and was transcribed.

Data for research sub-question (b) were collected and analysed as described in the studies 3 and 4. The representativeness of the observations on grey IA in Danish mining was explored through a questionnaire survey. The survey was distributed to the environmental department of all 98 Danish municipalities and inquired about the commonness and influence of the practice as well as the extent to which the practice is driven by an economic rationale. A total of 121 IA practitioners responded – in total, 102 EIA practitioners and 84 SEA practitioners. The representativeness of the observations on lacking LCT in Danish mining was explored through a document analysis of 85 IAs (37 SEAs and 48 EIAs), which comprise the topics urban planning (18), infrastructure (14), urban structures (15), energy (19), raw materials (8) and livestock (11). The analysis explored the extent to which LCT could be considered analytically appropriate and compared this fraction to the number of assessments applying LCT with LCA or other means of analysis.

Data for addressing research sub-question (c) were collected as described in study 5. The question did not require data *per se* since its answer is a procedure for LCA-SEA integration. However, the test of the procedure required data on the extraction intensity and the land use both prior to and after extraction (retrieved from 44 mining permits) as well as data on the thickness of the resource layers (retrieved from the 313 mining zones of the five 2012 mining plans).

7.5 The facets of meaningful SEA under study

As explained in the introduction, this dissertation builds on the assumption that an SEA is meaningful when it fits the decision-making context of the development under study. The figures 7.3 and 7.4 illustrate how the five studies explore different facets of both the SEA procedure and analysis in this regard.

With respect to the SEA procedure, study 3 explores how IA practitioner may alter the *'screening'* to better fit their decision-making context. Furthermore, the studies 1 and 2 explore how the contextual setting may impact both the *'identification of alternatives'* and *'impact analysis'*. Study 2 builds on the experiences from study 1 and presents a procedure for how to shape LCA application to the contextual setting and strategic capabilities of planners.

The purpose of the SEA analysis is to predict how a plan may cause a chain of events similar to that of figure 7.4. Plans regulate projects, which stress the local environment directly and environments elsewhere indirectly through e.g. product demands. The studies 1 and 2 explore how the contextual setting may influence the analysis of both alternatives (for regulating projects) and cumulative effects (on the receiving environment). The studies 4 and 5 explore the analytical benefits of LCA usage with attention to both the purpose and contextual setting of plans.

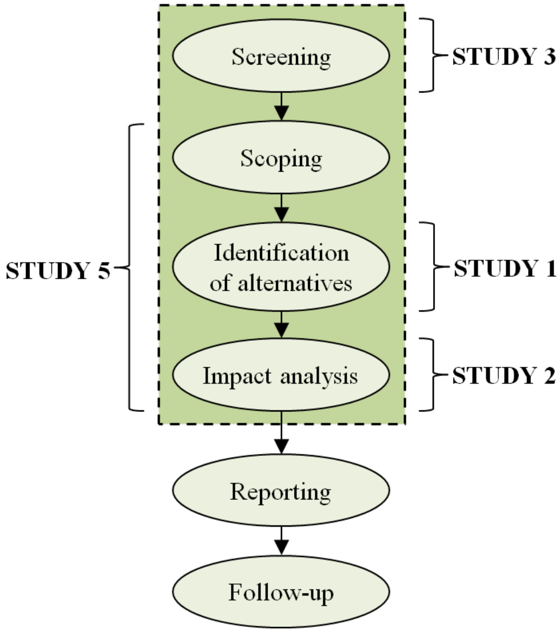

Figure 7.3: Four of the studies focus on the contextual influences on the SEA process.

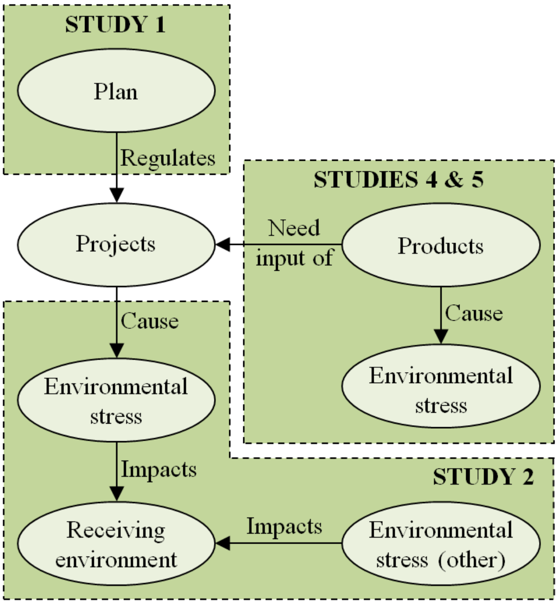

Figure 7.4: Four of the studies focus on the contextual influences on the SEA analysis.

PART III

SYTHESIS OF FINDINGS

8 FINDINGS

This chapter presents the findings from the five studies. The first section responds to research sub-question (a) and includes the findings on what restricts the assessment of alternatives and cumulative effects in the mining SEAs. The two following sections respond to research sub-question (b) and include the findings on the state of 'grey' practices and LCT in Danish IA. The last section responds to research sub-question (c) and presents a procedure for how to apply LCA in SEA.

8.1 The challenging alternatives and cumulative effects

The studies 1 (Bidstrup and Hansen 2014) and 2 (Bidstrup et al. 2016) explore the assessment of alternatives and cumulative effects (CE) in the mining SEAs.

Discrepancy between the process and documentation of the SEAs

In four of the SEA documents the identification and assessment of alternatives cover no more than a single paragraph, wherein it is argued that the plan is better than having no plan or using the old one (Bidstrup and Hansen 2014). Only one region assesses other alternatives. Likewise, it was found in Bidstrup et al. (2016) that the CE assessments lacks both detail and rigorosity in writing. Only one SEA describes how the joint activities of the mining plans may lower CE (with respect to *'transport'*, *'resource security'* and *'community benefits'*). The remaining four SEAs either do not mention CE or refer briefly to the consideration of CE (without any information on how such assessment has taken place). No examples were found of explicit CE assessment in the 15 zone reports – see table 8.1

	SEA REPORTS		ZONE REPORTS	
	Explicit assessment	Incl. implicit assessment	Explicit assessment	Incl. implicit assessment
Landscape	+	+		+++++
Traffic	+	+++		+
Groundwater				+
Biodiversity				
Community benefits	+	+		
Resource security	+	+++++		

Table 8.1: CE assessment in the SEA and zone reports with respect to six impact categories. Each '+' refers to CE practice in one region. Adopted from Bidstrup et al. (2016).

It is further explored in Bidstrup et al. (2016) whether the SEA reports assess CE implicitly – defined as assessments, which relate activities to the joint stress on a VC but are not labelled *'CE'*. As displayed in table 8.1, it was found that most plans provide a detailed account of how the plans contribute to the cumulative stress on the *'resource security'* in the region. All SEAs account for the current stress on the indicator, the projected stress in the years to come and the influence of the planned actions in this regard. Similarly, implicit practices were identified for regional traffic and local landscape impacts.

This discrepancy between what is explicitly written and what is actually done was supported further by the interviews. Surprisingly, every mining plan is the product of an iterative planning process, where alternative locations for mining zones have been considered carefully from the idea phase to the time of plan approval. Thus Bidstrup and Hansen (2014) conclude that the mining plans, in fact, are products of an on-going and integrated assessment of *'hidden alternatives'*. Similarly, it is reported in Bidstrup et al. (2016) that CE assessment has been an integrated part of the planning process. Four regions formed strategies for lowering CE in the early stage of the 2012 planning process. These strategies were used actively to map and select the mining zones for the subsequent plan proposal. Hence, it appears that the SEA reports do not represent the SEA processes truthfully. Both alternatives and CE have been assessed and managed throughout the planning process, though there is still room for improvement.

The contextual setting of mining plans

It was found that the poor assessment of alternatives and CE are not deliberate. The planners have a wish for their SEAs to focus on broad alternatives and they acknowledge that they still focus too little on CE (Bidstrup and Hansen 2014). Rather, it appears that the contextual setting is to blame.

Bidstrup and Hansen (2014) describe the contextual limitations with respect to plan alternatives in mining plans. From a purely geological perspective, onshore resource deposits are getting ever-more scarce and thus there are not always alternative locations at hand. New locations can be found by geological mapping, but this activity is expensive for the regions and is thus posing an economic limitation to the sheer number of feasible alternatives. The last and possibly most influential limitation to the assessment of alternative is institutional. A sufficient supply of raw materials can be secured by other means than merely more onshore mining, such as increased recycling or imports, a lowered consumption rate or offshore mining. Yet, these means are all beyond the responsibility of the planners (see figure 4.3). As stated in Bidstrup and Hansen (2014: 33): *"Though representing the highest managerial level within land-based aggregates planning, their task is to liberate sufficient space for extraction through zoning – not to rethink supply"*.

Bidstrup et al. (2016) describe how also CE assessment is influenced by the mining plans' boundaries with respect to geography (regional), time (12 years) and topic (only mining). The regional boundary has little influence since the most important CE occur on a smaller geographical scale. The 12-year time boundary is too narrow to encompass the relevant CE, but it does not appear to restrict the assessment since it is independent from the assessment and management of the subsequent mining projects causing the CE. The topical boundary restricts a meaningful assessment of CE because many of the activities causing CE together with the mining activities are beyond the influence of the planners. Often the planners have little knowledge of on the non-mining activities contributing to CE. Thus, data availability and cross-institutional collaboration appear to be important means for quality assurance when no responsibility for CE is legally appointed (Bidstrup et al. 2016).

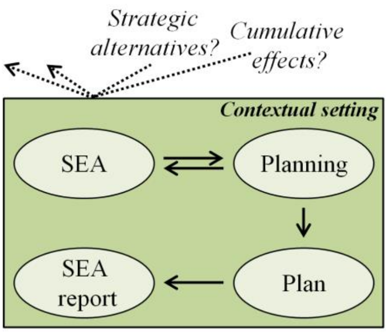

Figure 8.1: The assessment of some alternatives and cumulative effects are hindered by the contextual setting of plans. Additionally, the SEA report may be a product of the plan rather than the SEA process. Adopted and modified from Bidstrup & Hansen (2014).

The influence of the contextual setting on the assessment of alternatives and CE is depicted in figure 8.1. The SEA is applied actively in the planning process but the planners work within a setting, which limits the extent to which certain alternatives and CE can be addressed.

The capacity of the planners

A last point of the two studies is that further capacity building could improve the practices on alternatives and CE – however contextually limited they may be.

The mining planners struggle to identify alternatives because few alternatives exist when a plan proposal has been generated through the iterative planning process. This expectation of alternatives as *'alternatives to the full plan proposal'* generated a situation where the SEA reports reflect the plan as opposed to the integrated planning process – see figure 8.1. The alternatives of the planning practice are found in the way the plans are shaped. The true alternatives reflect the prioritisations and value-choices by which locations are proposed, assessed, selected and rejected for the plan. The true alternatives are procedural – not complete plan schemes. The planners are already assessing such alternatives, but further documentation of this practice could improve the transparency of the SEAs.

It was found that principally there is a good understanding of CE among the mining planners. Yet, the assessment of CE is conceptually questionably since, at large, the planners focus on the joint stress of their own activities as opposed to the total stress on and capacity of the environment (Bidstrup et al. 2016). They apply a plan-focus instead of the receptor-focus.

8.2 The 'grey' practices of Danish impact assessment

Study 3 (Bidstrup *In press*) explores the prevalence, influence and rationale of the assessment-like procedures occurring during or before IA screening – referred to as *'grey'* IA and explained in section 5.4. The purpose was to explore whether grey IA takes place also in wider Danish IA practice.

The prevalence of grey IA

Responds from the questionnaire reveal that 72% of the EIA practitioners and 80% of the SEA practitioners have knowledge of grey IA occurring in their own municipality. Figure 8.2 shows the commonness of grey IA, as expressed by those familiar with the practice. The median value for commonness is *'4'* for both EIA and SEA, and thus grey IA appears to be a widely prevalent practice, which is *'common'* in most municipalities.

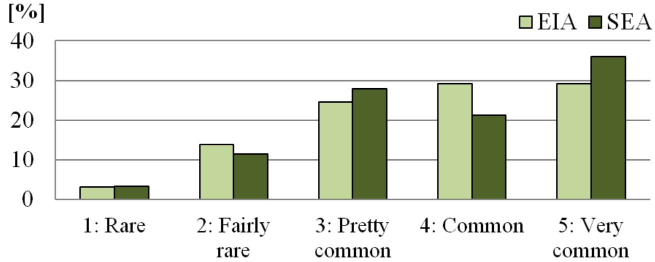

Figure 8.2: The commonness of grey IA among those familiar with the practice. Adopted from Bidstrup (in press).

The influence of grey IA

The practitioners familiar with grey IA were then asked to rank the extent to which the practice influences the outcomes of the subsequent, formal screening procedures. It was found in Bidstrup (*in press*) that grey IA has *'some'* to *'large'* influence on screening procedures. Digging further into the practitioners ranking *'low influence'* and *'high influence'*, it appears grey EIA is more influential than grey SEA.

The rationale(s) of grey IA

At last, the questionnaire inquired about the extent to which grey IA is motivated by the opportunity to save the time and money for a full IA – onwards referred to as the *'economic rationale'*. The practitioners were more in doubt about this question, as illustrated on figure 8.3. It is concluded in Bidstrup (*in press*) that the potential economic savings of grey IA motivate SEA practitioners to *'some'* extent and EIA practitioners to a *'large'* extent. In fact, 26% of the EIA practitioners express that the economic rationale motivates them to a *'very large extent'*.

The questionnaire's sole focus on the economic rationale for grey IA provoked many practitioners. A total of 26 written comments were received, and half of these had the purpose of categorically rejecting economic motives as the only explanation for the prevalence of grey IA. One practitioner assured: *"The adjustment and dialog taking place prior to and during screening is a lot about us wanting to ensure a good project and to avoid environmental impacts"*. Others found the practice to be merely *"environmental consultation"* or a sign of a *"healthy"* IA system. A key argument was that grey IA can be a means of securing the fulfilment of environmental objectives, and thus it is concluded in Bidstrup (*in press*) that there exists a *'green rationale'* to grey IA as well.

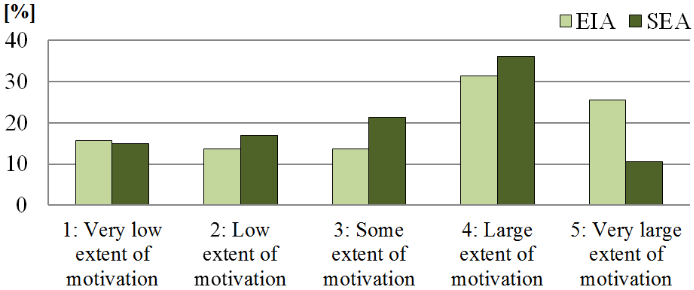

Figure 8.3: The extent to which grey IA practice is motivated by the opportunity to save the time and resources of a full-scale IA. Adopted from Bidstrup (in press).

The observations made on grey IA within the case of Danish mining (see section 5.1) therefore appear to be somewhat representative for wider Danish IA practice. Grey assessment does take place during or before screening. It is prevalent and influential, but it can be practiced with multiple rationales.

8.3 Life cycle thinking in Danish impact assessment

Study 4 (Bidstrup 2015) explores the extent to which the local assessment focus of the Danish mining sector is representative for Danish IA. Results were generated on the analytical appropriateness of LCT, the current application of LCT and the analytical gains assigned to further application of LCA.

An application of LCT in IA was in the study defined as having two facets. First and foremost, it entails the adoption of a product-oriented paradigm, by which impacts and alternatives are related to the product provision of the assessed development. Second, in entails the consideration of both up- and downstream impacts.

The analytical appropriateness of LCT in IA

It was assumed in Bidstrup (2015) that LCT is *'analytically appropriate'* in IAs of developments which supply a product. Such developments may generate life cycle impacts because they influence the means of product provision. Having this assumption in mind, it was found that LCT is appropriate for 87% of the studied IAs. Examples of *'products'* were electricity, transport, water and housing. The LCT perspective was found appropriate for all EIAs and 70% of the SEA. The SEAs found inappropriate for LCT were within the topic *'urban planning'* (see figure 8.4) since many of these had a broader focus than that of a sole product. Still, two such SEAs were found to rely on external LCA results as a means to legitimise strategic prioritisation, and thus it appears LCT can serve some purpose in most IAs.

The application of LCA in IA

Despite the widespread appropriateness of LCT, few IAs apply LCA for such analytical perspective. It was found that only 22% of the IAs are supported by external LCA results, while as little as 7% apply the tool actively – see figure 8.4.

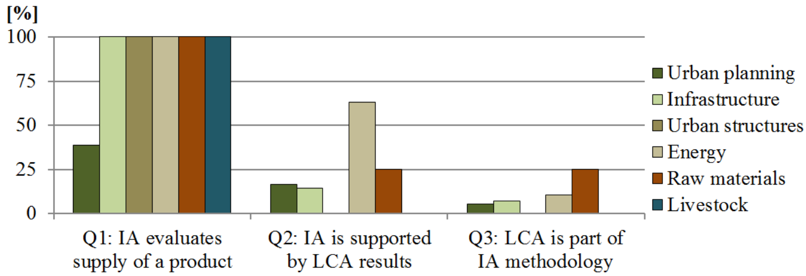

Figure 8.4: The analytical appropriateness of LCT (Q_1) and application of LCA (Q_2 & Q_3) across the topics of the 85 IAs. Adopted from Bidstrup (2015).

Single examples of LCA use were found within the topics *'urban structures'*, *'infrastructure'* and *'raw materials'* – here among the SEA of the Capital Region's 2012 mining plan – but only EIAs within the topic *'energy'* appear to draw on the tool consistently. LCA calculations are widely drawn on to estimate the CO_2 savings of wind turbine projects. Often, LCA is used to highlight the reduction of global impacts as a means to legitimise the negative local ones, such as noise or the industrial disturbance of particular landscapes (Bidstrup 2015).

The application of LCT without LCA

Figure 8.5 presents how the population of IAs apply LCT without LCA. It was found that without LCA IAs rarely relate impacts to product provision. IAs within *'energy'* appears to be an exception since these often manages to relate the impacts to the scale of the energy production. Still, most appear to consider alternatives with the same product provision. EIAs on road construction projects, for instance, often consider the different ways of delivering some fixed demand for mobility (the product), such as an alternative road trajectory or traffic congestion with the current infrastructure. Thus it is concluded in Bidstrup (2015) that most IAs do relate impacts to the product provision of developments somewhat implicitly.

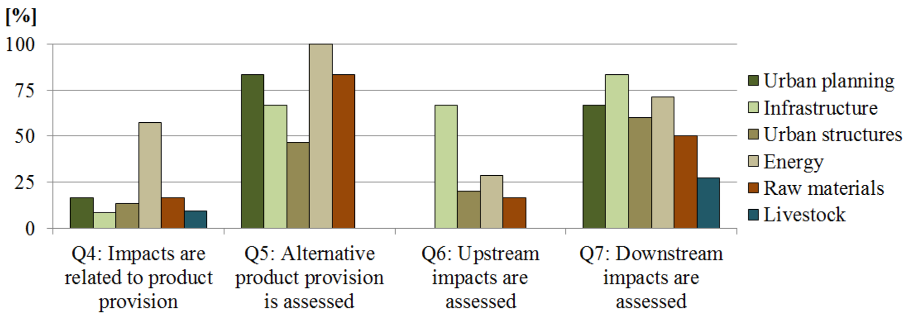

Figure 8.5: LCT across the topics of the 57 IAs, which do not apply LCA. LCT entails a product-oriented focus (Q_4 & Q_5) and the consideration of up- and downstream (Q_6 & Q_7) impacts. Adopted from Bidstrup (2015).

Few IAs assess upstream impacts while most assess downstream impacts – see figure 8.5. It was observed in Bidstrup (2015) that often upstream processes relate to the supply of construction materials while downstream processes relate to the use phase. In such cases, upstream impacts occur elsewhere while downstream impacts occur on the location of the development. Also, it was observed that the assessment of LCT (without LCA) is mostly qualitative.

The potential LCA gains

The findings from Bidstrup (2015) allow comparison between the level of LCT in IA with and without LCA. As it was the case with Danish mining (see section 5.1), it is concluded that further use of LCA can bring analytical gains. LCA can facilitate a more explicit focus on the product provision of development proposals and a more rigorous assessment practice, where distant, upstream impacts of the background system are considered as well and where impacts are communicated quantitatively. This can be exemplified through the case of a proposed apartment complex. Here, LCA could help quantify the impacts in relation to the supply of for instance 100 apartments (FU) and account for how the inputs for the construction, use and demolition phase may cause global warming and other impacts throughout the related production systems. Such analytical advancement would extent the assessment focus beyond the proximity of the development site and make it easier to compare the impacts of the project to that of an alternative apartment complex.

8.4 A procedure for applying life cycle assessment

Study 5 (Bidstrup et al. 2015) proposes and tests a procedure for applying LCA to SEA in a spatial planning context. The procedure was applied to the case of Danish mining for the purpose of testing its performance.

Description of the procedure

The merge of LCA and SEA posed two key methodological challenges. First of all, it was important to find a way to align the tools' divergent impact focuses on products (across production systems) and proposed developments (geographically delimited), respectively. Second, I had a wish to use the acquired knowledge on SEA and contextual setting (see section 8.1) to develop a procedure, which can be applied meaningfully. The proposed solution to these challenges is to focus the application of LCA application on how planners (working within their context) can influence the provision of their plan's product. Focus is on assessing how planners may drive product impacts, not on assessing the impacts of the plan *per se*.

The procedure of Bidstrup et al. (2015) is listed below:

Step 1: Identification and quantification of planning variables
Step 2: Development of an LCA model
Step 3: Formulation of planning scenarios
Step 4: Analysis of life cycle impacts
Step 5: Formulation of planning recommendations

The first step of the procedure is to identify and describe the relation between planning decisions and the subsequent product provision – referred to as *'planning variables'*. Second, one must build an LCA model of the production system with the processes of the variables in the foreground system. Planning scenarios, which are based on the variables and represent different planning prioritisations, are then formulated and analysed. The last step is to interpret the results and develop recommendation for planning practice.

An SEA procedure entails an *'impact analysis'*, where a variety of analytical tools and techniques can be taken in use – see section 2.2. Most studies on LCA-IA integration propose to apply LCA as one such tool. The procedure from Bidstrup et al. (2015) differs slightly from such proposals since it facilitates also the development of alternatives (formulated and analysed as *'planning scenarios'*). The procedure should be applied when an SEA *'scoping'* shows concern for significant impacts occurring across supply chains. This may be the case when a proposed development results in diverse ways of providing a product.

Test on the case of Danish mining

Step 1 led to the identification of four planning variables: *'transport'*, *'extraction intensity'*, *'resource thickness'* and *'site restoration'*. The planners influence the transport of raw materials and the thickness of the resource deposits through zoning. The intensity and restoration of mining projects are specified in mining permits.

Step 2 entailed the development of a cradle-to-gate LCA model, which accounts for how the product *'gravel'* is produced and delivered to a construction site. The FU was defined as *'1 m^3 of raw materials, from an average Danish gravel pit, delivered to user'*. The model made it possible to calculate how variations in the variable *'transport'* influence the demand for inputs such as lorries, roads and gasoline, while variations in the variables *'extraction intensity'*, *'resource thickness'* and *'site restoration'* influence the demand for land both during and after the mining process.

The scenarios of step 3 were formulated as planning extremes. The test was built as a sensitivity analysis, which aimed to contribute with knowledge on the variables most prone to generate impacts. The test had one baseline scenario (representing average planning) and ten extreme scenarios.

The analysis of step 4 had two facets: a scoping of the important impact categories and a subsequent analysis of the ten extreme scenarios. Figure 8.6 shows that the impact categories *'respiratory inorganics'* and *'global warming'* are particularly harmful when weighting the baseline scenario with the monetising Stepwise method. Figure 8.7 provides indications as to how the planners can manage such impacts. Plan induced changes in transport can generate some impact deviation (scenario 1-2), while the mining intensity has little importance (scenario 3-4). The restoration plans are very influential due to the loss in land productivity. The conversion from intensive agricultural land to non-productive land generates large impact increases (scenario 8-10). This is especially the case when the resource thickness is low since this increases the land conversion rate per cubic meter mined material (scenario 9).

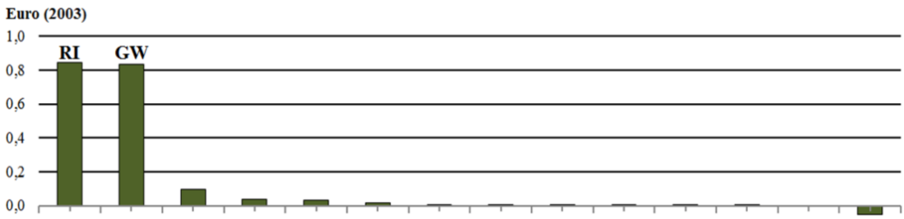

Figure 8.6: The weighted impact analysis of the baseline scenario. Among the multiple impact categories, two stand out as particularly important: respiratory inorganics (RI) and global warming (GW). Adopted and modified from Bidstrup et al. (2015).

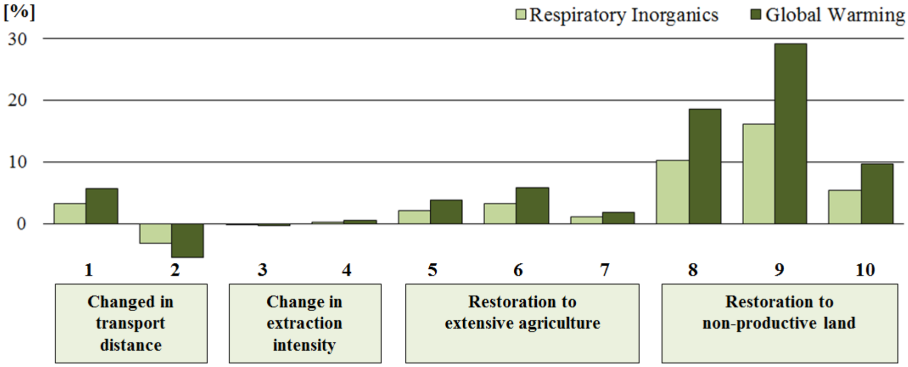

Figure 8.7: The impact analysis of the ten scenarios with respect to respiratory inorganics and global warming. All results are presented as % deviation from the baseline scenario. Adopted and modified from Bidstrup et al. (2015).

With an outset in this analysis, recommendations are formulated in Bidstrup et al. (2015) for how the mining planners can mitigate life cycle impacts (step 5). Planners can consider the intensity of mining projects a local concern only, but they should have a focus on reducing transport distances. Planners should restore mining sites to productive land on locations with a thin resource layer. They should only restore to nature where the productivity of the land is not high or where there is a thick resource layer (high resource yield per converted square meter).

Lessons learned from the test

The release of particulate pollutants and greenhouse gasses are already concerns of the mining planners, but air pollution is primarily considered a local phenomenon while global warming is solely related to transport – not land occupation. The restoration of mining sites to nature or recreational areas is in today's planning often used as a means to generate local accept. Yet, the study shows that such decisions can generate unwanted impacts elsewhere. Thus the procedure was found to provide new, valuable knowledge on how the planners can extent their impact analysis beyond their region – as further argued in Bidstrup et al. (2015).

The findings of the study were presented to an audience of IA practitioners at DCEA's annual Environmental Assessment Day seminar (miljøvurderingsdag) in 2014. Few members of the audience appeared to question the method or results, but the recommendations on how to manage such indirect effects sparked a vivid debate. A large fraction of the practitioners expressed that they are working in (and hired by) local institutions, which have an interest in lowering the impacts on their own (taxpaying) citizens. The life cycle impacts were perceived as too *'academic'* to have importance when generating local accept for projects or plans. The idea of re-establishing locally polluting agriculture instead of locally attractive recreational areas with the sole argument of reducing the impacts of a statistically occurring land conversion or intensification elsewhere was, for many, unfeasible!

9 DISCUSSIONS

This chapter provides a discussion of the research as a prelude to the conclusions in chapter 10. The first two sections address the representativeness of the Danish case and the validity of the obtained data. The chapter concludes by 'taking a step back' to elaborate on how systems thinking has contributed.

9.1 The representativeness of the Danish case

As explained previously, the research of this dissertation is inductive and applies the single case of mining and IA in Denmark to draw conclusions of wider relevance. Yet, Douven (2011) warns that what is observed may not always be adequate for establishing a universal premise. It is possible that Danish IA may not represent IA practice as a whole. This makes it critical to discuss the peculiarities of the Danish case – as presented in chapter 3. Four of these are: the institutional system, the spatial competition, the low corruption rate and the low price of the IAs.

The first peculiarity of the case is the segmented institutional system, which in the studies 1 and 2 was found to restrict the assessment of both plan alternatives and CE. Indeed, one might question whether the strategically limited planning context of Danish mining is representative for all plans subject to SEA. It is possible that elsewhere plans are made in institutions with a wider array of responsibilities. The contextual setting would arguably be less restrictive in such a case.

Second, Danish planning was described in section 3.1 as a *'spatial competition'* due to the high population density and the fact that all land currently is zoned (and thus also reserved) for a particular purpose. The lack of vacant space means that mining plans are generated through an iterative process, wherein the concerns and interests of the many local stakeholders are considered carefully. This process leads to the lack of reasonable alternatives once a plan proposal has been formulated. Moreover, the multiple local interests provide some explanation as to why planners struggle to see the importance of considering also the indirect impacts occurring across supply chains: *it is hard enough to take into account all the local interests as it is!* Thus, the lack of alternatives and interest in addressing life cycle impacts may not be representative for IAs made in countries with a lower population density.

Third, Denmark is the least corrupt country in the world. This is especially relevant for study 3. The contrast between the initial observations on grey IA and my idealistic world view (being a Danish citizen) is what sparked the study. Possibly, the grey practices had not caused the same frustrations to a PhD student working in a societal context, where the intensions of decision-makers are less trusted. Also, it was found that the effect of the practice depends on the rationale by which it is practiced. One could question whether grey IA is more prone to be practiced on the expense of the environment and public participation in a more corrupt IA context.

A last peculiarity is that Danish IA is cheap. Lyhne et al. (2015) depict Danish IA as a small, economic *'Volkswagen Beetle'* in a comparison to the more expensive Dutch IA system, which is depicted as a *'Rolls Roys'*. The studies 3 and 4 provide indications as to how the Danes keep the costs low: *grey screening practices reduce the need for formal IAs and life cycle impacts are often accounted for by qualitative means of analysis rather than the expensive LCA*. It can be questioned whether the prevalence of grey IA and the lack of LCA usage is representative to IA systems more expensive than the Danish one.

The existence of these peculiarities means that the results from the case of Danish mining cannot be scaled up uncritically to describe IA tendencies worldwide. This is not a problem with respect to the studies 1 and 2 since these merely conclude that the contextual setting of plans <u>can</u> influence the assessment of alternatives and CE. The conclusions from study 5 are not impacted either since the case of Danish mining merely serves as a platform for testing a proposed procedure. The peculiarities of the Danish case are most relevant for the studies 3 and 4 since these have an explicit focus on documenting the existence of some phenomena – respectively grey IA and LCT in IA. The results from these studies may not represent IA practices in more corrupt or extensive IA systems. Their conclusions apply to the Danish case and are less valid for other countries.

9.2 Engaging with the research field

The applied research approach meant that I engaged continuously with the mining planners. Though described as a *'dialogue'* in chapter 7, often this engagement had a character of *'confrontation'* between the theory and practice of SEA. From a theoretical perspective, I perceived grey IA as cheating the system, I perceived lacking LCT as a deficiency and I struggled to see the problem in just getting better at assessing alternatives and CE. From a practical perspective, my argumentation was too academic and not fitting the context. It was through this collision of world views I learned lessons on meaningful SEA.

The influence of the project

The PhD project ran parallel to the planning process of the 2016 mining plan, which it aimed to support. I sent an email to the planners on January the 12th 2016, wherein I inquired about the influence of the research on their practices. The purpose was to explore if I have acted as a change agent – as conceptualised in section 7.3.

It was found that the project has had limited direct influence on the process and SEAs of the 2016 plan. This is not all surprising since change agency was not at the centre of the research design. Research sub-question (b) does not relate to Danish mining explicitly, while sub-question (a) explores <u>why</u> the SEAs lack quality. Only sub-question (c) provides tangible recommendations for practice.

However, it was found that the research has contributed indirectly. The planners of the Central Denmark Region express that they *"have been forced to be more critical"*, while the planners of the Capital Region express that the confrontation has been both *"useful"* and *"educational"*. From the Zealand Region, one planner further stresses that the results may be used in future political processes: *"I strongly believe that some of what you found out about our room to manoeuvre can be used onwards to push for an improvement of the mining planners' strategic capabilities"*.

Thus I have been a *'change agent'* to some extent. Though substantive changes are not present at the moment, the project facilitated discussion and provided arguments for onwards improvement of the SEA practices. Among the multiple approaches to change agency, the planners unanimously express that I have fulfilled the role of a *"critical partner for discussion"*.

Concerns about data validity

The participative research approach raises questions about data validity since the influenced planners are also a primary source of data. Study 2 was initiated more than two years into the study when the topic of CE had been discussed on multiple occasions. Also, the planners of the Capital Region highlight that their SEA from 2012 (the most thorough of the five) was supported by discussions with members of DCEA prior to the PhD project. Thus the results from both the SEA reports and the interviews appear to be somehow under the influence of prior IA research.

The big question is whether this influence on the data represents a source of error. In particular, one can question the validity of some findings from the studies 1 and 2. Do the planners wish to apply SEA more strategically, or have they just been introduced to this theoretical ideal during prior interactions? Did the planners have a good conceptual understand of CE before the PhD started, or have this understanding evolved parallel to the project? The latter options may be the case, but more importantly: *Does it matter?* The purpose of the studies was to understand what currently limits the assessment of alternatives and CE. The studies were never intended to describe the challenges of a pristine practice – if one such ever existed.

9.3 A systems perspective on the findings

Chapter 4 described how the research of the dissertation builds on systems thinking as the overarching meta-theoretical framework. This choice has contributed with analytical perspective and helped to highlight how some of the systems surrounding the Danish mining planners have deficiencies.

The contributions of systems 'lens'

The application of systems thinking helped to widen the analytical perspective to focus on not only the SEAs of Danish mining. Indeed, the planners act within an array of interconnected systems with diverse purposes – see table 9.1. The mining planning system for onshore raw materials (see figure 3.4) is a part within a greater, institutionalised and hierarchical planning system. The mining planning system balances the raw material resource system (see figure 4.3). Also, it is influenced by the IA system (see figure 4.1) due to its impact on environmental systems.

SYSTEM	PURPOSE
Mining planning system	*to balance the raw material resource system*
Grand planning system	*to secure comprehensive planning, which unites societal interests and facilitates sustainable development*
Raw material resource system	*to preserve the construction industry*
Impact assessment system	*to ensure that impacting developments are assessed in a rigorous and transparent way, which helps to reduce the negative impacts on environmental system*
Environmental system(s)	*to preserve life*

Table 9.1: The systems related to the planning of raw materials mining. The systems' purposes are interpreted from their behaviour, as recommended by Meadows (2008:14).

The studies 1 and 2 highlight how the poor considerations of plan alternatives and CE are related to the segmentation of the institutionalised grand planning system. Study 3 describes how efforts to secure parsimony in the grand planning system (reduction of the costs and time for the approval of developments) may spark an informal dialogue, which alters the procedures of the IA system. The studies 4 and 5 explored how mining activities are related to environmental systems beyond the boundaries of the mining planning system. Hence, systems thinking helped to describe the IA issues of chapter 5 (alternatives, CE, Grey IA and LCT) as deriving from and being related to a world of surrounding systems. This contextualisation contributed to the process of scrutinising the Danish case for wider IA lessons.

Systemic deficiencies

All of the related systems serve a certain purpose – see table 9.1. The grand planning system serves to secure a comprehensive and sustainable planning (DMBG 2015), the mining planning system serves to balance the raw materials system and the IA system serves to reduce negative influences on environmental systems through the influence of rigorous and transparent assessment procedures.

The research suggests that the purposes of the IA system is somewhat compromised by the grand planning system. The topical focus of only onshore mining limits the mining planners' ability to consider CE and certain alternatives, while the regional and local-political nature of the planning makes LCT somewhat undesirable. Thus the case study demonstrates a systemic cascade effect, by which some element of the grand planning system restricts a sub-system (the IA system) in achieving its purpose (rigorous assessment). Ironically, such influence on the IA system may ultimately work against the grand planning system's very purpose of facilitating sustainable development. This is not to say that the Danish mining planners do not want sustainable development or that they are not facilitating such currently. The project just shows that their contextual setting restricts them in doing so.

A second systemic deficiency is that of grey IA. The various IA tools and their formal procedural steps represent a system, which is acknowledged across the scholarly literature. This is why I was provoked when I witnessed how some mining companies are omitting screening in collaboration with the local authorities: *they were breaking the rules of the system!* Study 3 initiated under the assumption that breaking the rules of the IA systems works against its purpose. Yet, the existence of the green rationale for grey IA provides an alternative explanation. It was found that arguably it is possible to rebel against the procedural purpose of IA and at the same time fulfil its contextual purpose of limiting the impacts on environmental systems.

10 CONCLUSION

This chapter concludes the dissertation. It opens with a description of how the five studies answer to the research sub-questions and contribute to the state-of-the-art. It then moves on to address the central research question of the dissertation. The last section elaborates on areas for further work.

10.1 What lessons were learned?

With an outset in the questionable performance of SEAs worldwide, the PhD project set out to explore whether there were lessons to be learned from the case of raw materials mining in Denmark. Focus was on exploring *'meaningful SEA'* – a term defined as when the SEA procedure and analysis fit the decision-making context. Through three research sub-questions (a-c), I explored why sometimes SEA is not meaningful, how planners then act and what can be done. The research contributed to the state-of-the-art by addressing the knowledge gaps of chapter 5.

a) Why is the assessment of alternatives and cumulative effects poor in the strategic environmental assessments of Danish mining?

It was found that the contextual setting restricts rigorous assessment. Key means for supply alternatives and many of the activities contributing to CE are beyond the institutional responsibility of the mining planners. Having this limitation in mind, it appears that to some extent both alternatives and CE are assessed and managed throughout the planning process implicitly. Further attention to procedural alternatives (rather than plan alternatives) and the receiving environment (rather than the stress of plan activities only) could improve these practices substantially.

These findings contribute to the state-of-the-art by providing insight as to why alternatives are assessed poorly in SEA. The demonstration of how restrictive contextual settings can be is novel. A second contribution is that of proving how CE assessment is not always meaningful in SEA. Also the developments of SEA have boundaries which can restrict assessment.

b) How representative are the observations on 'grey' screening practices and lacking life cycle thinking within Danish mining?

It was found that the observations on grey screening practices within the Danish mining sector are fully representative for wider IA practice in Denmark. Grey IA is common and influences the need for subsequent formal IA procedures. As it was the case for Danish mining, it was found that grey IA is more nuanced than *'proponents cheating the system'*. The practice can be used to circumvent IA requirements at the expense of both transparency and the environment, but it can also be a means of influencing projects and plans for the better at an early stage in the decision-making process. These findings contribute to the state-of-the-art with insight on the prevalence, influence and rationale of the practice outside a NEPA context and within the fields of both EIA and SEA.

Also the observations on lacking LCT were found to be representative for Danish IA practice. It is widely appropriate to assess the indirect, global impacts occurring across supply chains, but LCA is rarely applied for adding such perspective. Some LCT is present, but LCA can facilitate a more rigorous impact analysis with a more explicit focus on the impacts of the product provision. These findings contribute to the state-of-the-art by demonstrating the analytical benefits of further LCA application within current IA practice.

c) How can life cycle assessment be applied in the strategic environmental assessments of Danish mining?

A procedure for how to apply LCA to SEA was proposed. It relates all impacts to the product of the planning context and focuses on the strategic capabilities of planners. Application of the procedure to the case of Danish mining generated valuable results on how local prioritisations can spark unforeseen impacts throughout the interconnected product systems. The procedure *per se* represents a contribution to the state-of-the-art.

Central question: What lessons can be learned from Danish mining on meaningful application of strategic environmental assessment?

Drawing conclusions across the five studies, it is evident that the case of Danish mining has provided lessons on why sometimes elements of SEA are not meaningful, how the planners then act and what can be done to improve practice.

The case of Danish mining demonstrates that the contextual setting of SEAs can make certain assessments meaningless. This is by limiting the strategic means of the practitioners, but it is also by imposing some limit to their sense of responsibility. Another point is that conceptual difficulties may inhibit even the most well-willing of practitioners in applying SEA meaningfully.

The case further demonstrates how practitioners may try to make IA meaningful. They may respond to the contextually limited and local-political nature of their planning task by adopting a local assessment focus. They may also start to perceive IA as an administrative task with questionable output and, as a consequence hereof, use the screening process as some light edition of IA – with various rationales.

Three recommendations were formulated on how to apply SEA more meaningfully:

1) Fit SEA to the contextual setting.
2) Rebel against the contextual setting.
3) Be aware of the IA system's grey areas.

The first recommendation is to fit SEA to its contextual setting. The Danish mining planners, for instance, should focus on procedural alternatives rather than plan alternatives and focus the usage of LCA (and other tools) on planning variables.

The second recommendation is to try to rebel against the contextual setting. The case of Danish mining suggests that inter-institutional collaboration, further availability of data and appointment of leadership for cross-institutional issues are means to defy contextual limitations. More importantly, the study demonstrates that there may be significant analytical gains assigned to such rebellion.

The third and last recommendation is to be aware of the grey areas of the IA system. The case of Danish mining suggests that the projects following plans are regulated as much through informal dialogue as they are through formal EIA processes. SEAs should provide clear indications to how projects can fulfil the green objectives of plans. SEAs should strive to improve the grey IAs made with a green rationale and restrict the ones made with an economic rationale only.

10.2 Further research

Meaningful application of SEA is a broad, overarching topic, and thus there remain many follow-up questions which I did not have the chance to answer in the time allocated for the PhD project. A few of these are accounted for below.

One issue is that of exploring how to report integrated SEAs in way which does the assessment process justice. The case of Danish mining demonstrates that there can be large discrepancies between the considerations of the SEA report and the SEA process. This issue may seem unimportant to some, but it ultimately relates to the IA purpose of securing transparent decision-making.

Lots of follow-up questions remain with respect to grey IA. It was found that there exist both an economic and a green rationale for the practice, but which of these is most prevalent? The answer to this question may help to highlight whether the practice is driving or jeopardising environmental improvements. Furthermore, study 3 did not provide data on the kind and scale of the changes made by grey IA. Is the practice facilitating substantive changes? If this is the case, this common and widely prevalent practice may contribute to the effectiveness of the IA system as a whole. A last element of interest is that of further exploring the differences between grey EIA and grey SEA. It was found that grey EIA is more influential and more led by the economic rationale, but little is known about why. Is it due to differences in assessment costs? Or, are EIAs just easier to *'avoid'*?

The topic of LCA use in IA is receiving ever-more attention in the international literature. It was concluded in study 4 that LCA is widely appropriate from an analytical perspective, but little is still known about when this is the case from a procedural and contextual perspective. When can the analytical gains of LCA be expected substantial enough to legitimise the costs of the tool? The answer to this question may help highlight when the application of LCA in SEA is *'meaningful'*.

PART IV

PUBLICATIONS

P_1 The paradox of strategic environmental assessment

P_2 Cumulative effects in strategic environmental assessment:
The influence of plan boundaries

P_3 The 'grey' assessment practice of IA screening:
Prevalence, influence and applied rationale

P_4 Life cycle thinking in impact assessment:
Current practice and LCA gains

P_5 Life cycle assessment in spatial planning:
A procedure for addressing systemic impacts

STUDY #1

Environmental Impact Assessment Review 2014, 47, 29-35
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THE PARADOX OF STRATEGIC ENVIRONMENTAL ASSESSMENT

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Abstract

Strategic Environmental Assessment (SEA) is a tool that can facilitate sustainable development and improve decision-making by introducing environmental concern early in planning processes. However, various international studies conclude that current planning practice is not taking full advantage of the tool, and therefore we define the paradox of SEA as the methodological ambiguity of non-strategic SEA. This article explores causality through at three-step case study on aggregates extraction planning in Denmark, which consists of a document analysis; a questionnaire survey and follow-up communication with key planners. Though the environmental reports on one hand largely lack strategic considerations, practitioners express a strong wish for strategy and reveal that their SEAs in fact have been an integrated part of the planning process. Institutional context is found to be the most significant barrier for strategy and this suggests that non-strategic planning setups may influence SEA practice more than non-strategic planning. Planners may try to execute strategy within the confinements of SEA-restricted planning contexts; however, such efforts can be overlooked if evaluated by a narrow criterion for strategy formation. Consequently, the paradox may also spark from challenged documentation. These findings contribute to the common under-standing of SEA quality, but further research is needed on how to communicate and influence the strategic options which arguably remain inside non-strategic planning realities.

$P_1$1 Introduction

A paradox is a seemingly contradictory statement that may nonetheless be true, like: Strategic Environmental Assessment (SEA) is not strategic. This is a provocative statement since SEA is implemented into national legislation in countries worldwide based on the belief that it secures strategic considerations in decision-making on the policy, plan and programme (PPP) levels of activity. McGimpsey and Morgan (2013) describe mandatory inclusion of strategic alternatives and assessment of systemic effects as the primary benefit of introducing SEA in a non-mandatory planning context; yet, Tetlow and Hanusch (2012) conclude that especially these strategic elements appear to be lacking in practice. Such experiences from Canada, Austria, England, Finland, China, Greenland and Italy have been published (Bragagnolo et al. 2012; Hansen and Kørnøv 2010; Noble 2004; Stoeglehner 2010; SÖDerman and Kallio 2009; Zhou and Sheate 2011).

The suggested solutions for avoiding this paradox differ according to the different reasoning suggested. Stoeglehner (2010) argues that a change of planning paradigms towards more future-oriented approaches is required, while Bragagnolo et al. (2012) point at a need to increase focus on scoping and include relevant alternatives. Some studies find that practitioners do not assign significant value to the task of conducting SEA and perceive it as an administrative burden (Stoeglehner 2010; Zhou and Sheate 2011). A study on SEA in Belgium prior to the implementation of the European SEA Directive showed that the enthusiasm to make good strategic SEAs was greatest among Environmental Assessment (EA) experts and green NGOs while administrative workers were more sceptical (Devuyst et al. 2000). Reversely, other authors find SEA practitioners driven by the acknowledgement of a need to include environmental concerns at the PPP level of planning (see e.g. Noble (2004), Zhou and Sheate (2011), Kristensen et al. (2013) and Devuyst et al. (2000)).

The general overview provided by these studies opens up for a line of new questions, which seem important to answer in order to achieve an understanding of why SEAs apparently fail on strategy. These are questions like: Why do planners who appreciate SEA produce non-strategic assessments? Why are some SEAs considered of low value and perceived by planners as an administrative burden? And, why are planners sceptical towards the implementation and purpose of the tool? This article explores the causality behind the paradox of non-strategic SEAs through a case study, drawing on the experience with regional SEA of construction aggregate extraction plans in Denmark and focusing on the role of planners in relation to the inclusion of strategic elements in the SEAs. First, the article presents the concept of strategy in SEA. Secondly, a description of the planning context and the case study methodology will be provided. The article then presents findings and discusses whether environmental assessments of aggregate extraction plans in Denmark can be strategic, given the institutional structure of the sector. Finally, it compares case study findings with the international experiences that served as a point of departure in order to elaborate on the causality of the paradox of SEA.

$P_1$2 The concept of strategy in SEA

The term *'strategic environmental assessment'* has been around for a few decades now (see Therivel et. al. (1992)) and various opinions and interpretations of its societal purpose exist. Therivel (2010) defines SEA as *"a process that aims to integrate environmental and sustainability considerations into strategic decision-making"*, while Partidário (2012) argues that the purpose of an SEA is *"to help understand the development context of the strategy being assessed, to appropriately identify problems and potentials, ... and to assess environmental and sustainable viable options ... that will achieve strategic objectives"*.

SEA developed from the field of environmental impact assessment (EIA), but several methodological differences exist between the two tools. While EIA represents a reactive technical tool for mitigating (and preferably avoiding) the impacts of proposed projects, Noble (2000) argues that SEA is a tool for proactive and broad assessment of development alternatives for PPPs. However, the difference between the two tools is not always easy to spot since some SEAs in practice share many characteristics with EIA methodology – commonly referred to as EIA-based SEAs. Authors within the Impact Assessment (IA) community have in this regard argued that it is necessary to distinguish between *'strategic SEA'* and *'EIA-based SEA*' (Partidário 2012).

Though commonly referred to as a tool, SEA is a process which can improve decision-making and spark sustainable development. The strategic SEA is therefore related to the planning objectives, the timing of the planning process and the inclusion of what is referred to as strategic elements – e.g. the assessment of alternatives and cumulative impacts. Inspired by Therivel (2010) and Partidário (2012), figure $P_1$1 illustrates our interpretation of a strategic SEA planning setup. The SEA process (box 1) is here closely assigned to the decision making process (box 2), why alternatives, cumulative effects and other systemic sustainability impacts are continuously taken into account in an iterative fashion. The product of this process is an environmental report (box 3) that documents the SEA considerations, as required by e.g. The European Parliament (2001), and the approved plan (box 4), which ideally has been adjusted in accordance with the environmental concern of the strategic planning process.

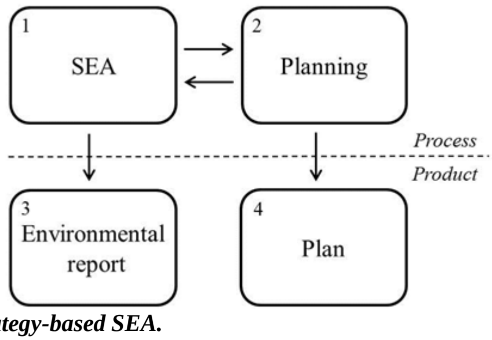

Figure $P_1$1: The strategy-based SEA.

The definition of *'strategy'* in SEA has received quite a bit of attention within the IA community. Noble (2000) summarises the term as "*the determination of objectives and means"* and *"the adoption of courses of action to achieve specified ends*". Cherp et al. (2007) investigate the concept of *'strategy formation'* in SEA. They point out that generally strategic elements are perceived as introduced in formal processes (based on a rational decision-making model), whereas in reality strategy formulation often happen in an informal process where the strategies are emergent rather than deliberate. Cherp et al. (2007) argue that mainstream SEA methodology applies a prescriptive notion of strategy formation in which the *'ideal'* strategy must be established prior to planning. Yet, a descriptive strategy formation, which fits the planning context and can be adjusted as challenges emerge, may prove more efficient since it represents the actual planning practice (Cherp et al. 2007).

We explore the paradox of non-strategic SEA by analysing the different elements of the SEA planning model – presented in figure $P_1$1. First, a document review and analysis investigates the strategic elements in the environmental reports (box 3) based on a prescriptive notion of strategy. Secondly, a questionnaire survey and follow-up communication apply a descriptive notion of strategy for exploring the interaction between the SEA process (box 1) and the planning process (box 2) in order to uncover how plans (box 4) are developed. The purpose of the analysis is to gain understanding about what the level of strategy is, where planners would like to see their tool application develop and why they are not doing it.

$P_1$3 Methodology

$P_1$3.1 Danish aggregates planning as case study

The case chosen as a subject of analysis is the SEAs related to the regional planning of mineral resource extraction in Denmark – commonly referred to as aggregates extraction planning due to the materials' societal purpose. The public sector in Denmark is divided between the state, the five regions and the 98 municipalities (DMIH 2005). The regions are responsible for the health system, transport, education, environmental development, handling of soil pollution and resource planning. In Denmark, the primary tool to secure the inclusion of environmental considerations at the strategic level in relation to the aggregates industry is SEA of regional resource planning; more specifically, the regional aggregate extraction plans, which identify and zone resource deposits (DMEF 2013b). The Danish aggregate extraction plans are forthcoming referred to as *'aggregate plans'*, while the assigned SEA documents will be referred to as *'environmental reports'*. No further centralised national management scheme exists, and the regional level thus remains the highest managerial level for aggregate extraction in Denmark. The regional planning must tier down to the municipal level where further project specific EIAs are undertaken in relation to technology applications for extraction licences (DMEF 2013b).

SEA in the aggregates sector of Denmark offers a good platform for an interesting and relevant case study on why SEAs lack strategy since the regional planners on several occasions have expressed difficulties in applying the tool. The aggregate planning context appears rather straight forward at first glance, and Denmark is a small country with a long history of environmental planning; hence, difficulties in applying SEA with a desired level of strategy and inclusion of strategic elements would be unexpected. A case study can provide a practical and exact illustration of specific challenges within the field subject (Rendtorff et al. 2009) and it can be exploratory, descriptive or explanatory (Yin 1993:5). The study presented in this article is based on a case study methodology of Yin (2003b), and it can be characterised as *'explanatory'*. Focus is on understanding the role of SEA in decision-making with an emphasis on exploring the three whys presented in the introduction. Common types of data in explanatory case studies are the data from documents, archival records, interviews, and participant observations (Yin 2003b:86). The case study in scope applies a mixture of these data collection forms, and the sources, types and uses of data are further described in the following paragraphs in relation to each step of the methodology.

$P_1$3.2 Case study methodology

The document review was conducted by comparing the five environmental reports (Capital Region 2013; Central Denmark Region 2012; North Denmark Region 2012; Southern Denmark Region 2012; Zealand Region 2012) to a list of principles for good SEA methodology and decision support. This list is presented in table $P_1$1 and it is based on both legal requirements and guidelines in the literature; namely, the European SEA Directive (European Parliament 2001), the OECD SEA guideline (OECD 2006) and a European SEA guideline (Partidário 2012). The demands of the European SEA Directive are applied since public planners in Denmark and the rest of Europe are legally obliged to comply with the content of this document (European Parliament 2001). Two guidelines are applied in order to compare the environmental reports to what is generally considered good practice within the field of SEA. The reasoning behind picking two different guidelines was that guidelines tend to vary depending on their interpretation of SEA objectives. The analysis was conducted by reviewing the environmental reports with a focus on registration and description of the 17 principles from table $P_1$1. Focus was solely on the content of the environmental reports, and thus process related elements (such as SEA timing) were not analysed. The focus was on the documentation available for the public.

The second part of the case study methodology consisted of a questionnaire survey among the key planners responsible for the 2012 SEA process. Danish aggregates planning is conducted by a small circle of specialists, and answers were received from nine planners, which represent and were appointed by the five regions. Rather than merely describing lacking strategic features of the environmental reports, this second analysis explored the underlying causality. Noble (2000) argues that a cornerstone in improving SEA quality is to focus more on the strategic component and thus move away from the widely used EIA-based SEA approach – but, do the planners agree with this point of view?

	Principles for SEA	SEA directive	SEA guidelines
1	Led by defined objectives	(x)	(x)
2	Incorporates the broad notion of sustainability	(x)	(x)
3	Applies a systemic perspective	(x)	(x)
4	Assesses development alternatives	(x)	(x)
5	Evaluates impacts on a baseline	(x)	(x)
6	Applies scenario-building		(x)
7	Evaluates impacts based on the context of the plan	(x)	(x)
8	Based on a transparent assessment methodology with defined principles and indicators		(x)
9	Considers both direct and indirect impacts	(x)	(x)
10	Considers cumulative effects	(x)	(x)
11	Considers both short-term and long-term effects	(x)	(x)
12	Considers probability, duration, frequency, reversibility, magnitude and spatial extend of impacts.	(x)	
13	Describes trade-offs		(x)
14	Describes conflicts of interest		(x)
15	Highlights opportunities and risks		(x)
16	Describes mitigation measures	(x)	
17	Provides reasoning for the best or chosen alternative	(x)	

Table $P_1$1: 17 Principles for good SEA methodology and information for decision support. (OECD, 2006; Partidário, 2012; European Parliament, 2001).

The questionnaire survey aimed at determining the extent to which planners agree with the strategic nature of their SEAs. The questionnaire consisted of 17 questions – see table $P_1$2. 14 questions are a direct modification of the seven differences between EIAs and SEAs described by Noble (2000). The remaining three questions refer to the strategic nature of the planning context. The planners are asked both how they perceive their current SEA and how they perceive an ideal aggregate SEA. The distinction between *'current'* and *'ideal'* enabled an analysis of the direction in which the planners would like to develop the tool.

The last element of the case study methodology was subsequent face-to-face and written communications with the planners, during which they were given a chance to elaborate on their questionnaire responses. On some occasions, we (the authors) requested these inputs, but more often than not they were sparked by objections or challenging comments from the planners when preliminary results were presented. This latter form of communication thus enabled us to verify or reject emerging interpretations deriving from result synthesis between the two prior parts of the case study methodology. In short, this last step ensured the bond to planning reality.

The characteristics of the aggregate SEAs			
	When is SEA used in planning?	*Towards the end*	*During plan development*
1	Currently:	( )	( )
2	Ideally:	( )	( )
	How is your SEA made?	*Strictly in the context of aggregate supply planning*	*In the context of broader visions, goals and objectives for regional development*
3	Currently:	( )	( )
4	Ideally:	( )	( )
	Which question resembles SEA scope?	*"How does the plan affect the environment?"*	*"What is the preferred option among our supply alternatives?"*
5	Currently:	( )	( )
6	Ideally:	( )	( )
	How is your SEA made?	*By assessing future impacts of the plan*	*By planning in accordance with visions established of the region*
7	Currently:	( )	( )
8	Ideally:	( )	( )
	How is your SEA made?	*Reactively*	*Proactively*
9	Currently:	( )	( )
10	Ideally:	( )	( )
	What characterises an SEA?	*It sums up individual screenings of extraction zones*	*It addresses the choice of the overall best supply option*
11	Currently:	( )	( )
12	Ideally:	( )	( )
	What characterises an SEA?	*It has a narrow focus and a high level of detail*	*It has a broad focus and a low level of detail*
13	Currently:	( )	( )
14	Ideally:	( )	( )

The strategic nature of an ideal aggregate SEAs			
		Yes	**No**
15	*It establishes a broad framework for future regional supply?*	( )	( )
16	*It evaluates which supply option that will be best for the region in the long run?*	( )	( )
17	*It focuses on sustainability rather than merely environmental impacts?*	( )	( )

Table $P_1$2: The questionnaire sent out to key planners (translated from Danish).

$P_1$4 Findings

$P_1$4.1 Characteristics of the SEAs

The environmental reports differ quite a bit between the five regions, whereas the SEA processes are almost identical. The comparability of the processes largely has its explanation in the legal framework since the national Act on Raw Materials provides mandatory requirements for public participation, the rights to complain, documentation and deadlines for each of these elements (DMEF 2013b). Without exception, all SEAs refer directly to this act and the SEA Directive when describing the purpose of SEA; hence, legal compliance rather than better development appears to be the main argument for conducting SEA.

The environmental reports

The environmental reports contain many of the elements of table $P_1$1. All five environmental reports position their aggregate plan in relation to defined sustainability objectives, they communicate the inherent trade-offs and conflicts of interests in the aggregate planning context, and they describe plan impacts in relation to a baseline. Three regions further present concrete planning objectives of respectively lowering transport impacts (2 regions) and land occupation (1 region). Despite these good elements, strategic considerations appear to be lacking in most of the environmental reports.

Aggregate plans clear and zone land for future extraction, and thus they generally consist of many small land use changes that all undergo an individual EIA-based screening. These screening documents are all rather comprehensive; however, most regions fail to establish a connection between these separate, minor changes and the impact of the overall plan. The Region of Central Denmark is a good example of this as their SEA methodology is defined as the sum of all the individual scoping reports for proposed quarries, i.e. the plan is accepted if all the individual changes are accepted. Moreover, the environmental reports primarily describe the impacts on local communities (e.g. traffic and noise), whereas global impacts are largely left out. Matters of cumulative effects and indirect impacts are not addressed.

Additionally, the SEAs are in severe lack of plan alternatives. According to authors such as Therivel (2010), inclusion of plan alternatives is the heart of SEA since its very purpose is to facilitate a choice of the preferred strategic action among the alternatives at hand. For this reason, it is quite astonishing that only the Capital Region succeeds in describing any other alternative than that of not approving the plan, commonly referred to as the 0-alternative. The assessments of 0-alternatives are in general given less weight than the full plan proposal, and they are often not compared with respect to all assessment indicators. These findings leave the Capital Region as the only region, which reasons why the plan at hand is a good option when compared to strategic alternatives – a demand from the SEA Directive. The remaining four environmental reports defend the plan by rationalising that it has *"minor impacts"*. However, one might ask: *compared to what?*

The SEA process

The findings from the environmental report analysis indicate poor SEA quality; however, analysis of the SEA process provided a different picture. Planners acknowledge their lacking focus on cumulative effects and broad systemic impacts, but the critique of their lacking strategic considerations in regard to plan alternatives (as opposed to only considering the 0-alternative) created frustration. Planners from the Capital Region argued that a *"limited number of alternatives"* existed when writing the environmental report since the plan and SEA were *"developed simultaneously in an iterative process that has assessed and environmentally optimized alternative solutions continuously"*. This statement is supported by the assigned documents for public participation (referred to as *'the white book'*) and the supporting scoping reports for gravel pits proposals. They describe how all plans in fact have been adjusted or changed during the planning process as a result of environmental and public concern. The Zealand Region refers to these changes as *"possible alternatives"*, but only the Capital Region briefly describes an alternative wherein the reader is presented the implications of the sum of these adjustments. As such information is generally excluded from the environmental reports, it is concluded that most of the plans are in fact products of an on-going assessment of hidden plan alternatives. The term *'hidden'* refers to the absence of these alternatives in the environmental reports available for the public. Prior criticism was solely based on lacking assessment of alternatives in the environmental reports; however, if the SEA tool has been applied to continuously improve the plan during the planning process, as recommended by authors such as Noble (2000), Therivel (2010) or Partidário (2012), one might argue that an assessment of alternatives (of the whole plan) becomes redundant in the final decision-making phase. Planners from the Region of Central Denmark have in this regard stated on several occasions that they see the assessment of alternatives as a *pro forma* task rather than a meaningful planning process. Strategy has taken place during the planning process; yet, the focus on plan alternatives seems to demotivate the planners because a full plan will only be available when all strategic decision windows have passed.

$P_1$4.2 Perception of planners

As touched upon in the introduction, one hypothesis on the causality of non-strategic SEAs is lacking will or understanding among planners to improve their planning practice. Our results show that this is not the case for aggregates planning in Denmark. Despite the lacking strategic considerations in the environmental reports, our questionnaire survey proves that the very same planners generally agree to that an ideal aggregate SEA ought to assess supply alternatives, be pro-active and establish a broad framework for future extraction. Key planners of all five regions agree on that their SEAs should do more than simply sum-up screening results of extraction proposals and they acknowledge that their present environmental reports tend to focus too much on the plan at hand and too little on extraction alternatives. There is a wish and will for the SEAs to move from reactively assessing a plan towards proactively assessing alternatives.

Lastly, the questionnaire provided one key observation. Though a comprehensive description on how to answer the questionnaire stated that the planners could only pick one of the listed options (the strategic option or the EIA-based option), four regions ended up answering *"both"* for several questions. Such *"both"* answers appeared in 13 out of 17 questions. The planners apparently see the need for their SEAs to fulfil both a strategic and a more EIA-based function!

$P_1$4.3 Institutional constraints

The document analysis proved that the environmental reports have a low level of strategy, which inevitably results in a poor foundation for the regional decision-makers when approving the plans. This is peculiar since the questionnaire survey reversely proves that there is a wish to produce high quality, strategic SEAs. Based on the follow-up communication with the key planners, our results suggest that true strategic considerations are constrained by the institutional context. The planners are, so to say, caught between a wish for strategy and their institutional reality.

Returning to the *"both"* answers

As touched upon already, the legislation dictates that the role of the regions is to zone areas for future extraction and thus tier directly to the municipal level where extraction permits are granted (DMEF 2013b). Kalle and Arts (2013) highlight that such tiering is a vital element in ensuring good decisions. On one hand, the aggregate plans must produce a tangible output for subsequent municipal management. Yet, they also represent the most suitable means to strategic management since no aggregates extraction planning occurs on any managerial level higher than the regional. Surely, the demand for both tangible zoning and strategic thinking explains why the planners answered *"both"* in the questionnaires.

What alternatives do planners have?

One must further understand that an assessment of alternatives is a complex matter when dealing with finite geological resources. Firstly, a deposit needs to be mapped and described before it can be characterised as a resource. Geological resource mapping is a costly activity for the regional budgets, and it is thus politically unacceptable to map areas merely for the sake of evaluating them as alternatives. Secondly, alternatives can only be evaluated when one has several possibilities at hand. Planners from the Capital Region express that they are having severe difficulties in finding sufficient resources due to *"geological conditions and the high population density"*. Several suitable extraction sites may not always exist in close proximity. Hence, the planners normally start the planning process by mapping suitable deposits, after which public participation and individual EIA-based scoping will determine whether extraction is feasible. This is arguably the root of the SEAs' local focus, and it gives an indication as to why planners generally experience an evaluation of full plan alternatives as a *pro forma* task.

Still, one might argue that truly strategic alternatives do not have to exclusively concern land-based extraction sites in close proximity. Partidário (2012) describes *'strategy'* as taking a step back and perceiving the overall objectives from a larger perspective. The main objective of an aggregate plan is to supply society with sufficient resources in a responsible way, and strategic alternatives could therefore be to increase recycling, to increase marine extraction, to lower consumption or to import from neighbouring countries. However, recycling is managed by the municipalities, marine extraction is managed by the state and consumption patterns are a result of the free market forces. Consequently, most systemic alternatives are beyond the institutional power granted the regional planners. Though representing the highest managerial level within land-based aggregates planning, their task is to liberate sufficient space for extraction through zoning – not to rethink supply. A truly strategic aggregate planning seems to conflict with the institutional reality of the sector. This argument is supported by the planners who find that their SEA *"encompasses the possibilities available within the given legislative framework"*.

The reality of SEA

All European PPPs that may significantly influence the environment, must by law, be subject to an SEA. Yet, in the case of aggregate planning in Denmark, SEA reality lacks conformity with the prescriptive principles for a good SEA due to institutional barriers. The findings are illustrated in figure $P_1 2$, which differs from the idealistic strategic SEA model of figure $P_1 1$.

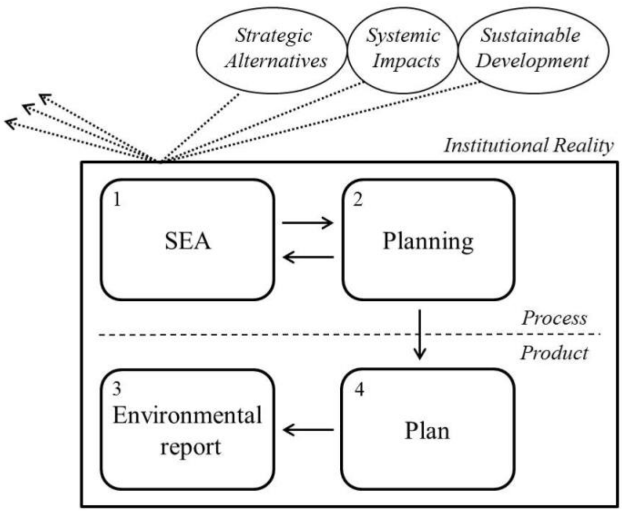

Figure $P_1$2: The SEA reality of SEA practitioners in the case study.

In short, SEA practitioners are found to work within an institutional reality that in some cases acts as a barrier for the consideration of strategic alternatives, systemic impacts and a broader notion of sustainability. Moreover, the case study concludes that environmental reports in some cases derive from the finished plans rather than the SEA processes, and that this phenomenon can be sparked by an interpretation of SEA alternatives as solely *'plan alternatives'*. It is tempting to assume that an environmental report which only focuses on the finished plan is a sign of poor SEA quality, but the case study results suggest that an integrated, strategic SEA process is not necessarily synonymous with a transparent environmental report.

$P_1$5 Unfolding the paradox of SEA

As touched upon in the introduction, many studies have been conducted on SEA performance prior to this article. Stoeglehner (2010), Bragagnolo, Geneletti, & Fischer (2012), Söderman & Kallio (2009) and Zhou & Sheate (2011) also found that the SEAs they analysed lacked proper assessment of systemic alternatives. Likewise, Noble (2004) found in an SEA review that planners face difficulties in including a broader notion of sustainability in their SEA context, and the studies Söderman & Kallio (2009) and Noble (2004) describe how SEA practitioners are having a hard time seeing the purpose of truly strategic SEAs. Thus it is evident that the case study on aggregate planning in Denmark shares many characteristics with these prior experiences. We believe that it contributes with knowledge relevant for planning contexts much different from the one in scope.

$P_1$5.1 Caught between a wish for strategy and institutional reality

Most prior studies focus primarily on describing the lacking features of SEA, but some elaborate on why strategy appears to be missing. Lacking insight, methodological misuses of SEA and a snapshot of the progress of an emerging tool are presented as the causality for poor conformity with SEA principles (Devuyst et al., 2000; Noble, 2004; Söderman and Kallio, 2009; Zhou and Sheate, 2011). However, our study suggests a different cause. In the case of aggregates planning in Denmark, the regions remain the only applicable institution to impose strategy-based planning, but their embedded task and institutional reality prohibit them in doing exactly that. The results suggest that truly strategic SEAs may neither be possible nor meaningful for all planning contexts requiring an SEA by law.

Though rather untouched in the literature, descriptions of institutional constraints can be found. Noble (2004) describes *“institutional limitations”* as a frequent cause for insufficient SEA practice, and Kristensen et al. (2013) highlight *“government structures arranged around... political boundaries”* as the main cause for lacking strategic leadership in regard to managing cumulative effects. Moreover, Finish planners have expressed that assessment of plan alternatives does not make sense in their context (Söderman and Kallio 2009). Thus we argue that the paradox of non-strategic SEA can be a product of the institutional reality surrounding practitioners.

Certain experiences emerge when looking for description of this phenomenon in SEA guidelines. Partidario (2000) argues that the strategic nature and characteristics of SEA vary due to the vast span of decision arenas which SEA has to cover, and guidelines stress that SEA must focus on improving decision-making rather than fitting a certain format (Partidário, 2012; Therivel, 2010). The case study in scope reveals a different perspective to the debate since it suggests that certain planning contexts and institutional setups may actually prohibit strategy – even at the highest managerial level. With that in mind one can question: *Do non-strategic SEAs have their roots in non-strategic planning or non-strategic institutional setups?* We argue that the latter option may be valid for certain SEA contexts.

$P_1$5.2 What are SEA alternatives?

The case study proved that a context-relevant assessment of alternatives in an integrated SEA planning process may not yield a transparent level of strategy in the environmental reports by default. The European SEA Directive demands that environmental reports should describe and evaluate *"the likely significant environmental effects of implementing the plan or programme, and reasonable alternatives"* (European Parliament, 2001), and the regional planners of Denmark clearly interpret this formulation as *'plan alternatives'*. The research suggests that such a rigid interpretation of SEA alternatives can leave practitioners with little room to conduct an environmental report with transparent strategy since the notion of plan alternatives may be unfeasible within certain planning processes and exclude the true alternatives of the institutional reality. On a methodological level, our case study suggests that the more integrated and strategy-oriented models for SEA interacting with planning can prove hard to transparently document.

$P_1$5.3 How does one evaluate strategy?

The problem is that the aggregate plans are programmes (considered non-strategic) on the highest managerial level (where strategy should be executed). Therivel (2010) prescribe that higher levels of decision-making ought to address the systemic nature of decisions with *'why'* questions, while more technical *'how'* questions are suitable for lower tiers. But how should questions be asked when whys do not fit the context of the highest managerial level? More importantly, how does one ask the right questions at the right time that enables influence? The case of Danish aggregate extraction is interesting because it represents such an attempt.

The study demonstrates that practitioners can find themselves in an institutional context where they are required to produce assessments that are both strategy-based and EIA-based. Such contexts generate barriers for strategic planning; but still they do leave room for influential, decision-oriented inclusion of environmental concern. Can such concern be characterised as strategy? We argue that an on-going, iterative adjustment of plan content as a result of the uncertain societal and geological externalities represents an emergent, informal and descriptive strategy formation. This is of interest since the principles of table $P_1$1 represent a rather prescriptive recipe for strategy, while an SEA evaluation based on written environmental reports assumes a formal and deliberate notion of strategy formation. In other words, the document review analysed *'strategy'* in the SEAs with a too narrow criteria for strategy formation. Cherp et al. (2007) conclude that future SEAs must adapt to include both emergent and informal strategy formations as a means of granting SEA influence. The case on Danish aggregates demonstrates that such uses of SEA do exist. Yet, a narrow notion of strategy formation may deem such SEAs non-strategic.

$P_1$5.4 Returning to the whys

The exploration of the paradox of *'non-strategic SEA'* provides a foundation for elaborating on the whys brought forward in the introduction. There might be many valid reasons for why SEAs worldwide fail on strategy, but the case study on aggregates planning in Denmark provides new insight. We find that planners who see the need for strategic SEA can be severely limited in their execution of strategy by non-strategic institutional contexts. Tthe case study shows that planners may try to operate strategically within such limited planning realities. A too rigid and one-sighted interpretation of SEA alternatives and objectives can nonetheless under such circumstances make SEA reporting seem like a *pro forma* task and in this way spark a written focus that does not match the actual SEA process. These findings suggest a new hypothesis for why some planners are sceptical towards SEA and why they perceive it as an administrative burden. Namely, that SEA can appear as an academic tool which is hard to implement in different planning realities.

$P_1$6 Conclusion

Therivel (2010) argues that SEA has the potential to *"make the world a greener and more liveable place"*. Yet, in practice the tool appears to fail on its inherent promise: strategy. We explored this paradox of non-strategic SEA through a case study on aggregate extraction planning in Denmark.

We found that the paradox can spark from planning context because certain institutional setups subject to SEA (even on the highest managerial level) simply do not leave room for broad strategic considerations. This leads to the conclusion that a much deeper paradox must be addressed. Namely, that of parliaments delegating strategic planning responsibility to institutions with limited strategic capabilities.

We further found that the paradox of non-strategic SEA can derive from challenged documentation rather than poor planning *per se*. Planners had great difficulties in addressing plan alternatives due to an on-going, iterative and in many ways strategic practice, and this indicates a risk assigned to the more integrated and strategic SEA models. Influential and descriptive strategy formation may simply be perceived as non-strategic when evaluated by prescriptive strategy ideals. In other words, the principles of table $P_1$1 are a good starting point when making an SEA, but they provide no framework for assessing whether an SEA has been strategic or not.

We acknowledge that these conclusions by no means fully explain the lack of SEA quality worldwide. Assessment of cumulative effects remains absent in the hidden alternatives of the Danish Regions, and capacity building might therefore still be a way forward in relation to improving SEA quality. In that sense, this article complements the findings of prior research. Rather than shooting down existing interpretations, it is merely aimed at adding perspective to the debate on SEA quality. This perspective could be met by further research on how to define strategy in accordance with the strategic capabilities of practitioners as wells as further research on how to communicate and improve the hidden (and somewhat truly strategic) alternatives of various SEA realities.

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STUDY #2

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CUMULATIVE EFFECTS IN STRATEGIC ENVIRONEMENTAL ASSESSMENT: THE INFLUENCE OF PLAN BOUNDARIES

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Abstract

Cumulative effects (CE) assessment is lacking quality in impact assessment (IA) worldwide. It has been argued that the strategic environmental assessment (SEA) provides a suitable IA framework for addressing CE because it is applied to developments with broad boundaries, but few have tested this claim. Through a case study on the Danish mining sector, this article explores how plan boundaries influence the analytical boundaries applied for assessing CE in SEA. The case was studied through document analysis in combination with semi-structured group interviews of the responsible planners, who also serve as SEA practitioners. It was found that CE are to some extent assessed and managed implicitly throughout the planning process. However, this is through a focus on lowering the cumulative stress of mining rather than the cumulative stress on and capacity of the receiving environment. Plan boundaries do influence CE assessment, though all boundaries are not equally influential. The geographical and time boundaries of the Danish mining plans are broad or flexible enough to accommodate a meaningful assessment of CE, but the topical boundary is restrictive. The study indicates that collaboration among planning authorities and legally appointed CE leadership may facilitate better practice on CE assessment in sector-specific SEA contexts. However, most pressing is the need for relating assessment to the receiving environment as opposed to solely the stress of a proposed plan.

$P_2$1 Introduction

The field of Impact Assessment (IA) covers a broad range of procedural tools, which all aim to facilitate transparent decision-making and sustainable development through the identification and evaluation of the impacts assigned to proposed developments (IAIA 1999; 2009). The International Association for Impact Assessment (1999) stresses that good IA practice includes an assessment of the contribution to cumulative effects (CE), commonly defined as *"changes to the environment that are caused by an action in combination with other past, present and future human actions"* (Hegmann et al. 1999:3). CE assessment focuses on the total stress on Valued Components (VCs), which for societal or scientific reasons are considered important (Canter 2015; Canter and Ross 2010; Hegmann et al. 1999; Johnson et al. 2011). This focus on the capacity of and stress on the receiving environmental (communicated as a VC) rather than solely the stress of solely the development under evaluation is a cornerstone in CE assessment (Duinker and Greig 2006; Gunn and Noble 2011; Hegmann and Yarranton 2011; Therivel and Ross 2007). Despite its importance, CE are assessed poorly in IAs worldwide (Morgan 2012; Pope et al. 2013; Tetlow and Hanusch 2012). Aside from explanations such as lacking conceptual understanding (Gunn and Noble, 2011) and legal guidance (Weiland 2010), recent research has found that the institutional segmentation of IA responsibility can pose barriers for effectively addressing CE (Chilima et al. 2013; Kristensen et al. 2013; Sheelanere et al. 2013).

It has been argued extensively that the strategic environmental assessment (SEA) provides the most appropriate IA platform for CE assessment (Cocklin et al. 1992; Duinker and Greig 2006; Gunn and Noble 2011; Johnson et al. 2011; Therivel 2010) – though some SEAs show poor CE performance also (Bragagnolo et al. 2012; Cooper 2011; Noble 2009). The prevalent argument is that SEA *"offers the chance to influence the kinds of projects that are going to happen"* (Therivel, 2010:18) because the developments under evaluation in SEA (programmes, plans and policies) cover multiple actions on a larger scale of space and time than for instance the project-oriented Environmental Impact Assessment – referred to as EIA (Therivel and Ross, 2007). Yet, the developments subject to SEA are ultimately still bounded. This article proceeds under the assumption that there exist two types of boundaries for any CE assessment made in an IA context: an analytical boundary and a development boundary.

The *'analytical boundary'* marks the scale of space and time applied for considering the multiple (and often diverse) actions causing CE on a particular VC – as described in CE guidelines, such as CEAA (2012) and IFC (2013). João (2007:489) finds that the choice of an appropriate analytical scale (and thus also boundary) is critical in IA because it *"affects the problem addressed, the options found and the impacts evaluated"*. CE often occur on different scales among and within impact categories, and a multi-scale approach is thus often needed (João 2002; Karstens et al. 2007; Therivel and Ross 2007). For instance, a certain action may generate CE in the near proximity during the time of construction (narrow scale), while it also plays a part in larger, regional CE over the timespan of multiple years (wide scale).

The *'development boundary'* is in this study defined as the coverage of the development under evaluation. By that we mean that all proposed developments by default influence a set of actions, which may span across geography, time and topics, and which may cause CE. A proposed project is often a single action *per se*, which will be established on a particular location during a short period of time. Reversely, a proposed plan may cover multiple types of actions, which will take place within a larger planning area during a planning period. All developments can thus be characterised as having a set of geographical, time and topical boundaries – some more narrow than others.

Though not stated explicitly, much of the advocacy for CE assessment in SEA revolves around the argument that development boundaries influence the analytical boundaries, i.e. wider development boundaries allow better consideration of the multiple actions causing CE. Karstens et al. (2007:389) find that the decision-makers proposing and evaluating developments *"are often limited in their powers by the scale of the political system"*, just as Bidstrup and Hansen (2014:32) find that planners can be limited by their *"institutional reality"*. However, the influence of development boundaries on the analytical boundaries applied for evaluating impacts in IA is poorly studied. The research of Bragagnolo et al. (2012) does show that the assessment of CE in SEA can be bounded by the plan under study, but critical questions remain. Can development boundaries in SEA be expected broad enough to encompass the analytical boundaries appropriate for considering the actions contributing to CE, spanning across various topics and applied on various locations at various times? If not, are they then restricting CE assessment?

This present study explores sector plans – a bounded development type commonly evaluated by SEA. Through a case study of Danish mining, the study tests the following hypothesis: *Plan boundaries influence the analytical boundaries applied for CE assessment in SEA*. Attention to one sector in one country was chosen as a means of deepening analysis to comprise also implicit CE assessment. The hypothesis was tested by exploring four topics: a) the understanding of CE among the SEA practitioners, b) the current practice on assessing CE, c) the extent to which plan actions are related to environmental stress beyond plan boundaries, and d) the opportunities for overcoming plan boundaries. The article opens with a short description of the case study context. The method is then described, after which results are presented with respect to each of the four topics. The article concludes with a discussion of the adequacy of CE assessment in SEA and the lessons learned.

$P_2$2 Case study context: mining plans in Denmark

Denmark is a country in Northern Europe and a member of the European Union. The European SEA Directive (European Parliament 2001) is implemented in Danish legislation through the national SEA Act (DMEF 2013c), which states that all plans and programmes posing a risk of significant impacts must be evaluated by SEA. The act specifies that CE assessment is a mandatory element.

This study focuses on the plans regulating the on-shore mining of mineral and raw material resources for the construction sector – such as sand, stone and chalk. In Denmark, planning is structured around the national planning hierarchy, which comprises a state level, 5 regions and 98 municipalities (DMIH, 2005). The national act on Mineral and Raw Material Resources (DMEF 2013b) specifies that each region must produce a plan every fourth year – onwards referred to as a *'mining plan'* – which accounts for how the supply of resources can be ensured for the coming 12 years. Supply is ensured through establishment of mining zones, within which contractors then can apply for mining permits for mining projects. The plan boundaries of the case are thus:

Geographical boundary: regional
Time boundary: 12 years
Topical boundary: mining

The relation between mining plans, zones and projects is presented in table $P_2 1$, while a schematic overview of the planning process is presented in figure $P_2 1$. The table and figure are based on the legal framework (DMEF 2013b) and interviews with the mining planners. The planning process consists of six phases. First, planners form ideas for a supply strategy and potential locations for future mining zones. The planners are during this phase supported by an 8 weeks public hearing, where stakeholders are invited to send in ideas and proposals for future supply. Proposed locations can only be taken into consideration if they hold substantial resources, and thus phase one is supported by geological mapping (phase two). Each proposed mining location is then evaluated in phase three, during which the onsite impacts are weighed in relation to both the size of the resource deposit (estimated in phase two) and the supply strategy (formed in phase one). The results of these multiple evaluations are then used to establish a full plan proposal in phase four. This proposal is subject to further 8 weeks of public hearing, where stakeholders are now invited to object and comment on the prioritisations and decisions of the planners. The hearing often results in an adjustment of mining zones (phase five) before ultimately approving the mining plan (phase six). The planning process alters between a local zone focus and a regional plan focus.

	Regulation	**Focus**	**IA**	**Documentation**
PLAN	Plan approval	∗ Plan ∗ Zones	SEA	∗ SEA report ∗ Zone reports
PROJECT	Mining permit	∗ Sub-zones	EIA	∗ Environmental impact statement

Table $P_2 1$: Mining plans consist of mining zones, within which contractors can apply for mining permits for concrete mining projects. Mining plans are evaluated by SEA, while mining projects may be evaluated by EIA. The study focuses on the SEAs. These are documented through an 'SEA report' and multiple 'zone reports'.

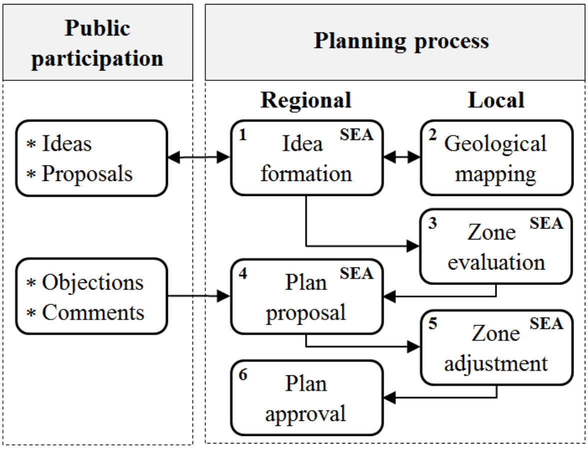

Figure $P_2$1: The planning process for Danish mining is divided in six phases, which alter between the regional and local level. Public participation takes place in phase one and four, while SEA is drawn upon in the phases one, three, four and five.

SEA is drawn upon throughout the planning process. Broad environmental considerations are made when brainstorming ideas for a supply strategy in phase one, while assessment on a local zone level is an integrated part of phase three. The knowledge on local impacts near proposed mining zones is used to concretize plan-wise impacts in phase four, before returning to the local zone level in phase five. The local and regional assessments are separated in published form, though they jointly make up the SEA. Plan-wide impacts are communicated in an *'SEA report'* while more detailed accounts of the impacts of each mining zone are attached as multiple independent *'zone reports'*. These latter reports are made in the planning process before contractors may propose specific projects, and thus they should not be confused with the environmental impact statements assigned to project EIA – see table $P_2$1. A last thing to clarify is that the SEAs of Danish mining are sector-specific SEAs. Though each covering the geographical area of a Danish Region (a public administrative authority headed by democratically elected politicians), they have little in common with the broad SEA type *'Regional SEA'* – as further clarified in section $P_2$5.3.

$P_2$3 Method and data

As described in the introduction, the study design explored four topics:

a) The understanding of CE among the SEA practitioners
b) The current practice on assessing CE
c) The extent to which plan actions are related to environmental stress beyond the plan boundaries
d) The opportunities for overcoming plan boundaries

The authors have experienced that CE assessment is not always done (or articulated) well in SEA practice, and the method was thus tailored to identify both implicit and explicit assessment of CE. This focus on also implicit assessment practices lead to a series of measures. First and foremost, it was considered important to explore both the conceptual understanding of CE (a) and the current practices on CE assessment (b) before drawing conclusions on the influence of plan boundaries. Second, it was chosen to study the influence of plan boundaries as the extent to which practitioners relate their actions to environmental stress extending beyond the plan boundaries (c), rather than whether explicit CE assessments apply such a perspective. Assuming that plan boundaries would have some kind of influence, the study questioned how the influences of plan boundaries can be overcome (d).

Data were collected through document analysis of the five 2012 mining plan SEAs and semi-structured group interviews with the responsible mining planners, who also serve as SEA practitioners. The document analysis provided insight on the written extent of CE assessment in relation to the plan boundaries (b and c). The interviews served to deepen the results with knowledge about the conceptual understanding of the mining planners (a), the SEA process (elaboration of b and c) and the opportunities for overcoming plan boundaries (d).

A key element in the analysis was to explore whether the Danish mining planners relate their proposed actions to the total stress on and capacity of the receiving environment (as argued in the introduction). Principally, there is nothing wrong in focusing the analysis on some measurable indicator for environmental stress rather than a VC *per se*, but CE assessment has only taken place if this stress is ultimately related to the functioning of the receiving environment.

$P_2$3.1 A focus on both explicit and implicit CE assessment

The term *'explicit CE assessment'* covers assessments in the SEA reports or the zone reports that refer to CE directly. The term *'implicit CE assessment'* covers assessments, which are not labelled *'CE'* explicitly, but which a) relate the added stress to the functioning of a VC or b) were highlighted during the interviews with respect to CE. For the sake of clarity, the analysis on implicit CE assessment focused on six generic VCs:

1. Landscape
2. Traffic
3. Groundwater
4. Biodiversity
5. Community benefits
6. Resource security

These generic VCs were selected with an outset in known impacts of the Danish mining sector. The VCs cover both bio-physical and socioeconomic impact categories, which may be affected both positively and negatively. *'Landscapes'* are altered whenever a mining site is taken in or out of use, just as the mined materials always generate *'traffic'* when distributed. The Danish drinking water supply is based purely on *'groundwater'*, and both the quality and quantity of this resource may be affected when mining materials under the groundwater table. *'Biodiversity'* can likewise be affected since mining both destroy and create nature through land conversion. The category *'community benefits'* covers the economic benefits of mining, the assigned employment, the health impacts and the opportunities generated when restoring mining sites. Lastly, *'resource security'* is an important parameter since planners must balance how the supply of finite mineral resources can be secured within both a short-term and long-term horizon.

$P_2$3.2 Document analysis

The objects of study were the SEAs of the five 2012 mining plans (Capital Region, 2013; Central Denmark Region, 2012; North Denmark Region, 2012; Southern Denmark Region, 2012; Zealand Region, 2012), which are the most recent. Each SEA consists of one SEA report and 21 to 52 zone reports, of which three random were studied. Thus the study sample covered a total of five SEA reports and 15 zone reports. The purpose was to gain a general understanding of the written extent of CE assessment, which then was to be further explored in the subsequent interviews.

Explicit CE assessment
Are cumulative effects mentioned?
To what extent are CE assessed explicitly?
Implicit CE assessment
Which of the six generic VCs are assessed?
To what extent is the cumulative stress related to VCs?
Plan boundaries
To what extent do the SEAs consider the interplay with actions occurring beyond ...
... the geographical boundary?
... the time boundary?
... the topical boundary?

Table $P_2$2: The framework applied for analysing the SEAs of the 2012 mining plans.

The documents were studied by the use of the framework from table $P_2$2. As previously explained, the focus was on mapping both the explicit and implicit assessment of CE as well as on exploring whether the SEAs relate plan actions to environmental stress beyond the plan boundaries. CE were studied on three different geographical scales: a local zone scale, a regional plan scale and a supra-regional scale. Implicit CE assessment practices were identified without a standardized framework. Such a simple approach was possible because the written extent of the documents was manageable (in total around 250 pages) while both the SEA reports and zone reports were topically divided into sections addressing impacts on the generic VCs explicitly.

$P_2$3.3 Clarifying interviews

A semi-structured group interview was conducted in each of the five regions between the 8th and 17th of June 2015. All interviews lasted between 40 min and a full hour. The regions were asked to invite who they found most appropriate to represent and explain their SEA practice. This resulted in the participation of three to five mining planners from each region – adding up to 18 individuals. Though small, the sample size was considered sufficient to explore the practices on CE assessment within the case study context. With an outset in the four topics of the research design (previously denoted a to d), the interviews explored the conceptual understanding of CE, the process of assessing CE, the perceived influence of plan boundaries, and the restrictions and opportunities for improving CE assessment. The interviews were structured around the framework presented in table $P_2$3.

Conceptual understanding of CE assessment
How would you define CE?
Is CE assessment an important element in SEA?
Is there a difference between CE assessment on plan level and zone level?

CE assessment and the planning process
Were CE discussed in the 2012 idea phase?
Were CE discussed when assessing zone proposals?
Were CE found significant for some zone proposals?
Did CE influence the plan proposal?
If any, who initiated CE assessment and why did they do it?

Plan boundaries
Did you consider ...
... the stress on inter-regional, national or global VCs?
... the cumulative interplay with actions beyond a 12-year timespan?
... the cumulative interplay with non-mining actions?

Opportunities and restriction for better CE assessment
What would enable better CE assessment?
What is restricting better CE assessment?

Table $P_2$3: The framework applied for the semi-structured interviews. The questions have been translated from Danish to English.

$P_2$4 Results

$P_2$4.1 Conceptual understanding of CE

Planners were asked to define CE freely at the beginning of each interview. The definitions varied from region to region, but all agreed on two basic principles. First, there was strong consensus on that CE assessment must focus on how multiple actions affect the same VC. One group of planners defined CE as *"all kinds of contributions, which reinforces the total effect"*, while another group defined CE as *"when multiple impacts act jointly"*. Second, all groups found that CE can extend beyond mining plan boundaries. It was argued that mining can contribute to CE beyond the regional boundaries. None of the planners defined CE as confined with respect to time, but all agreed on that CE can spark through the interplay with non-mining actions such as wind turbines, farms and various sources of traffic. With minimal intervention from the authors, the planners thus provided a fairly concise definition of CE. When subsequently reading out loud the definition of CE provided by the Danish SEA guidance (DMEF 2006) – which is close to identical the definition of the introduction – all planners expressed that this covers their interpretation. This was despite the fact that none appeared to be familiar with this guidance beforehand. Hence, the interviews proved that there exists a good conceptual understanding of CE among the Danish mining planners.

$P_2$4.2 The general level of CE assessment

The SEA documents

Despite conceptual understanding, CE assessment is poorly addressed explicitly – see table $P_2$4. With respect to the regional plan level, one SEA report does not even mention CE, while three others describe shortly that CE have been assessed (without any further information about how, when and where). Only one region accounts for how the actions of the plan lead to joint CE – in the specific case, a lowered need for transport, reduced landscape impacts, a higher resource security and increased community benefits. The SEA reports were further analysed for signs of implicit CE assessment in regard to the six generic VCs of section $P_2$3.1. Each of the VCs are accounted for throughout all five SEAs, but impacts on *'landscape'*, *'groundwater'*, *'biodiversity'* and *'community benefits'* are only mentioned briefly with respect to the plan level. Three SEAs describe in a short fashion how their plan may result in less regional *'traffic'* through a focus on minimising transport distances. Yet, only the VC of *'resource security'* is thoroughly assessed in writing. All SEAs account for how resource supply can be secured within the region through new mining zones in combination with recycling initiatives and maritime excavation, while taking the current stress on resource security into consideration. However, this result is not all surprising – as one planner later expressed: *"the primary goal [of the plan] is to account for the supply situation within the region"*.

	REGIONAL PLAN LEVEL		**LOCAL ZONE LEVEL**	
	Explicit assessment	**Implicit assessment included**	**Explicit assessment**	**Implicit assessment included**
Landscape	+	+		+++++
Traffic	+	+++		+
Groundwater				+
Biodiversity				
Community benefits	+	+		
Resource security	+	+++++		

Table $P_2$4: The extent of CE assessment documented in the 2012 mining plan SEAs. Each '+' refers to one of the five SEAs, which either explicitly or implicitly relate mining plan actions to the generic VCs. Results on regional plan level assessments were retrieved from the SEA reports, results on local zone level assessment were retrieved from the zone reports.

With respect to the local zone level, the picture is similar. Assessment of CE is not described explicitly in any of the analysed zone reports. Yet, it was found that all five regions assess the impacts of proposed mining zones as related to the current stress on countryside landscapes. This approach qualifies as CE assessment since the reports consider whether the current landscape will be stressed beyond a point where it could be considered significantly changed. In addition, single examples of zone reports that take into account the current traffic load within an area or the current stress on a groundwater reservoir were found – see table $P_2$4.

The SEA process

With respect to the regional plan level, the interviews revealed that CE on *'landscape'*, *'groundwater'*, *'biodiversity'* and *'community benefits'* have not been addressed. Planners acknowledge that they influence regional CE on these VCs, but they have not considered this in their SEAs. On a more positive remark, it was found that four of the five groups had formulated a supply strategy on reducing the region-wide cumulative transport of raw materials during the first phase of the planning process (see figure $P_2$1). With this strategy, they hoped to lower CO_2 emissions and keep the market price of the resources low. For at least one of the regions, this strategy had legitimised public investment to map resources (phase 2) in areas with scarce supply as a means of sparking commercial interest for mining on *"desired locations"*. Additionally, three regions had applied the strategy actively for selecting those of the incoming zone proposals that were to be accepted for the plan (phase 4). One region, for instance, had been strict on not pointing out new areas close to operational gravel pits, while it simultaneously had accepted almost all proposals in close proximity to a future tunnel project. An interviewee framed the practice as follows: *"we are wearing 'different glasses' for areas with sufficient raw materials, than for areas where there are not enough"*. When confronted with the lack of explicit CE assessment in the SEA reports, one respondent assured: *"we have discussed it a lot – especially in regard to traffic – and made an effort to scrutinise CE. But it is not written down"*. His colleague further elaborated: *"CE is not a word you can use in the public debate! In that case you will have to reformulate it as 'traffic load', for instance"*. Hence, plan-wide considerations on CE with respect to *'traffic'* did shape the 2012 plans in at least three of the regions.

With respect to the local zone level, the interviews confirmed that it is common to evaluate the impacts on *'landscape'*, while a receptor-oriented CE approach occasionally is applied to the assessment of *'traffic'* and *'groundwater'*. Yet, only one group of planners was able to provide an example of a case where a zone had been adjusted or omitted from the plan due to accumulated impacts. Multiple zone proposals were adjusted during the 2012 planning process, but these decisions were most often based on the implications of the mining zone *per se*, rather than conflicts regarding the interplay with other actions. In fact, two groups argued that the existence of current cumulative issues on a location may favour the appointment of that specific area for future mining. One interviewee provided an example of this rationale in regard to landscape evaluations: *"[if] we have an area, which is already affected. It is not in correspondence with the landscape we had 50 or 200 years ago, it is not well-preserved and the original structures are destroyed in one way or another. Then one can say: This is not valuable … It looks terrible as it is!"*

The interviews proved to verify the findings of the document analysis. Assessment of CE with respect to *'resource security'* and *'traffic'* is common at the regional plan level, while the assessment of cumulative landscape effects is common on a local zone level. Hence, CE assessment has clearly been a bigger part of the SEA process than communicated explicitly in the SEA documents.

$P_2$4.3 Relation to cumulative stress beyond plan boundaries

Geographical boundary

None of the SEA reports describe impacts extending beyond the regional geographical boundary as an explicit focus with regard to minimising the CE of the plan. Also, the three generic VCs of *'landscape'*, *'groundwater'* and *'community benefits'* are not related to stress occurring on any level higher than regional, as well as it rarely is the case for *'traffic'* and *'resource security'* – see table $P_2$5. One planner responded to this lack of perspective during the interviews: *"the environmental consequences of mining raw materials are first and foremost local"*. Said differently, the planners find the geographical boundary to be wide enough to encompass most relevant CE, which their plan actions may contribute to.

	GEOGRAPHICAL **Assessment of contribution to inter-regional, national or global stress on VCs**	**TIME** **Assessment of stress on VCs beyond a 12-year timespan**	**TOPIC** **Assessment of stress on VCs from non-mining actions**
Landscape		++++	+++++
Traffic	++	++	+
Groundwater		+	+
Biodiversity	(+++++)[a]	+++	
Community benefits		+++++	
Resource security	++	+++++	++

Table $P_2$5: The extent to which the five 2012 mining plan SEAs relate plan actions to environmental stress beyond the plan boundaries with respect to each of the generic VCs. Each '+' refers to one SEA. The table aggregates explicit, implicit, regional and local CE assessment. a: biodiversity impacts on higher geographical scales are somewhat managed through the consideration of biodiversity zones.

Still, it was found that *'traffic'* and *'resource security'* are accounted for on an inter-regional level between the neighbouring Region Zealand and Capital Region. This is due to a dependency, which has been much debated as a planning dilemma within the Danish mining community. The capital of Denmark, Copenhagen, houses multiple projects, which demand a steady intake of materials – for instance the construction of the new metro ring (Metro Corporation 2015). Yet, the Capital Region can from a geological perspective not produce a sufficient amount of raw materials of adequate quality to support its activities, and it is thus highly dependent on a resource import from the Zealand Region (Capital Region, 2013). This dependency was highlighted by both groups of planners during the interviews. As one planner stated: *"raw material supply in the Capital Region ... has a tremendous effect on what goes on in Region Zealand"*. Hence, the Zealandic planners have established mining zones near the border of Capital Region to lower the joint CE.

A surprising result was that all five SEAs appear to somewhat manage both national and international CE on *'biodiversity'* by rigorously accounting for and minimising the potential impacts on spatial biodiversity zones. Such zones are established across the Danish landscapes with the purpose of conserving biotopes and species of fauna and flora which are threatened by cumulative stress on either a national (§3 zones from the Danish Nature Conservation Act) or international (Natura2000 zones from the European Habitat Directive) scale. Though arguably representing effective CE management, the practice has little in common with a proper CE assessment. The SEAs do not relate the added stress to the functioning of ecological systems on these supra-regional scales, and the interviews revealed that none of the planners had considered this practice as related to CE! As one planner framed it: *"we take into consideration that there is a general ban on planning in Natura2000 zones. We do not pose cumulative arguments"*. When asked about why they then consider national and international biodiversity, the planners pointed to legal compliance.

Time boundary

It was found that most planners relate their plans to environmental stress extending beyond the time boundary of 12 years – see table $P_2$5. The time boundary is solely mentioned in relation to the *'resource security'* provided by the plans, which in all cases actually is more than 12 years! For the remaining five generic VCs, the SEAs focus on the impacts occurring during or after excavation, rather than the timespan of the plan *per se*. When inquired about impacts occurring after 12 years, one planner responded: *"we are aware of that the zones we select will remain in many years ... Our thoughts are that we must create something valuable"*. Hence, planners find that the time boundary is too narrow to encompass the environmental stress, which the plan actions may contribute to.

Topical boundary

The SEAs show poor performance in relating the plan actions to the environmental stress of actions occurring beyond the topical boundary – that is, actions not related to mining. One SEA contextualises its supply strategy within the overall traffic situation of the region, while two others aim at preserving virgin resources through a focus on increased recycling (managed by the municipalities). Yet, non-mining actions are generally not considered – see table $P_2$5. The only real exception is the local *'landscape'* assessments, which universally relate the proposed mining actions to the industrial character and visible vulnerability of the area under analysis.

This exception aside, the SEAs generally stick to assessing the cumulative stress of their own actions without relating this stress to the receiving environment. Questions on this topic generated frustration. When asking one group about the lack of data on the multiple types of actions affecting a concrete groundwater reservoir, the answer was: *"the municipality's comments guided us when we proposed the zone, because they are groundwater authority … We do not have the expertise in house …"*. When asking another group of planners about why they then actually had included information about the current stress on a different groundwater reservoir, the answer was: *"we use the information we can get our hands on. There is a contamination from the city, and that information was by chance available in our institution"*. Availability of data was a theme raised by all five groups of planners. Before proposing new zones, planners ask all stakeholders to provide information, which could be of importance when considering a specific location for mining. All publicly available GIS themes are moreover downloaded and taken into consideration upon assessing the zone. Yet, it is difficult for the planners to consider CE if information on existing or planned actions is neither sent to them nor publicly available.

$P_2$4.4 Overcoming plan boundaries in CE assessment

It was found in section $P_2$4.3 that plan boundaries do influence the analytical boundaries applied for assessing CE in SEA. Yet, the study provides indication as to how the analysis can extend beyond the plan boundaries when deemed appropriate.

Planners expressed that they neither have the financial resources nor the time for considering CE outside their planning boundaries. As one said: *"there is a practical, pragmatic reality, which bounds the level of detail and the resources we can use"*. Hence, ways must be found for assessing the contributions to CE in a resource-efficient manner. Planners do their best to gain knowledge on existing problems by, for instance, downloading GIS themes with geographical zones for biodiversity, wind turbine projects, urban development and much more. Thus it appears that increased availability of data is a key element in securing meaningful consideration of CE. When such data do not exist, it was highlighted that coordination and dialog between both institutions and stakeholders is critical. A second theme was the absence of formal requirements for CE assessment. Legislation and guidelines all highlight that CE should be addressed, but there is a lack of appointed leadership regarding the management of CE. The planners felt that some of the questions on assessment beyond plan boundaries were equally beyond their institutional responsibility. These results are supported by Sheelanere et al. (2013), who also find that there is a need for appointed leadership as well as coordination and collaboration among authorities when CE extend beyond managerial bounds.

$P_2$5 Discussions

$P_2$5.1 Limitations of the study

The findings of the article are based on a case study of a single sector in a single country. This sector accounts for five SEAs every fourth year and houses roughly 18 planners, who also function as SEA practitioners. The authors are confident about the validity and robustness of the results, but the study design raises critical questions on representativeness. Indeed, it can be argued that a) the mining sector is not comparable to other planning contexts subject to SEA and that

b) Danish mining, which primarily produces products for the construction sector, is not universally comparable to mining in other countries. Thus the results of this study cannot be scaled up uncritically to cover tendencies within SEA practice as a whole. Through a case study, the research aims at analysing CE assessment practices in SEA deeper than previous studies and in this process explore the challenges and opportunities assigned to moving CE assessment to SEA – nothing more.

$P_2$5.2 Plan boundaries are not equally influential

Multiple authors have advocated that CE assessment can be done well in SEA due to the wider boundaries of the developments under evaluation. Yet, one can argue that this claim is true only if the plan boundaries are either 1) wide enough to encompass the actions causing CE or 2) not restricting the assessment of CE – see figure $P_2$2a. This is elaborated in figure $P_2$2b with respect to the case study context.

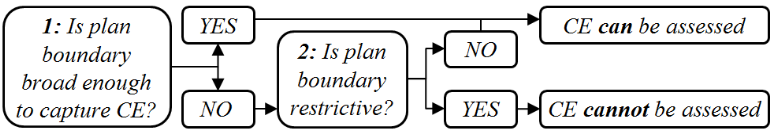

Figure $P_2$2a: A flow diagramme of the relation between plan boundaries and CE assessment. Assessment of CE is possible if the plan boundaries are wide or non-restrictive.

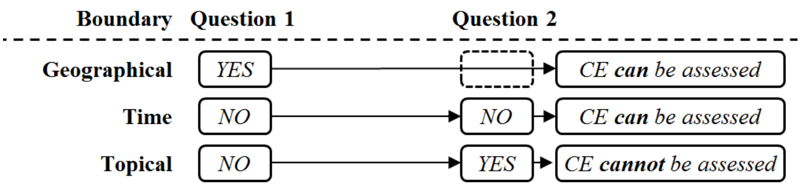

Figure $P_2$2b: Results on mining plan boundaries. The numerated 'questions' refer to figure $P_2$2a. The geographical and time boundaries allow CE assessment, while the topical boundary was found to be too narrow and restrictive for CE assessment.

In regard to the geographical boundary, the planners expressed that the regional scale is sufficiently broad to cover most significant CE. Impacts are rarely contextualised on spatial scales larger than the regional one (see table $P_2$5), and planners were reluctant to do so. Thus CE assessment is meaningful within the geographical plan boundary, though the regional focus is somewhat restricting.

The time boundary was found to be too narrow for capturing relevant CE since the mining and subsequent site restoration (which generate the impacts) extend beyond a 12-year horizon. However, the boundary is not particularly restrictive. The plan requirement on the 12 years relates solely to the amount of resources which need to be available in the mining zones. The start, duration and end of the impacts depend on the time at which an application for a mining permit is received, the quantity of resources in the zone and the fluctuations in the resource demands of the market. The impacting actions are somewhat independent of the plan's timeframe, and thus the narrow time boundary does not restrict a meaningful assessment of CE.

Questions on the topical boundary generated much frustration among the mining planners. Multiple mining sites within close proximity can generate CE, which the topical boundary is broad enough to capture. However, all five groups reached the conclusion that both local and regional CE can arise from the interplay with various actions not related to mining. These actions are often planned and regulated in different institutions, and this generates a lack of information, institutional power and ownership. By such, the topical plan boundary is restricting a meaningful assessment of CE.

The case study demonstrates that plan boundaries are not equally influential. Some boundaries may prove wide enough for a meaningful CE assessment, while others remain narrow and restrictive. Furthermore, the study indicates that there may be an important institutional side to the discussion of plan boundaries. Assessment and management of CE takes place within formal and informal institutional boundaries, which influence the planners' recognition and attention towards CE as well as their perception of CE relevance, institutional responsibility and legitimacy – a result supported by Kristensen et al. (2013). It has been argued that the institutional setup surrounding SEAs underpins the CE assessment in the sense that it can have a facilitating or constraining effect (Chilima et al. 2013; Sheelanere et al. 2013).

$P_2$5.3 Assessment of CE and the multiple SEA types

With an outset in the Danish mining sector, it was found that development boundaries can influence the analytical boundary applied for assessing CE. Yet, it is important to emphasise that this study focused on only one development type subject to SEA. SEA applies to a broad spectrum of proposed developments, ranging from sets of multiple projects to policy-driven structured actions (Harriman and Noble 2008; Partidário 2000). Programmes and narrow plans subject to SEA are often assessed by reactively evaluating some already proposed actions, while broader plans and policies ideally are assessed with a broader and more forward-looking, strategic approach (Lobos and Partidario 2014; Noble 2000; Partidário 2012). Hence, the quality of CE assessment in SEA may depend tremendously on whether one deals with an operational, single sector plan (like the Danish mining plans) or a more strategy-oriented development.

An example of one such an SEA context is Regional SEA (R-SEA), which appears to be especially well-established in Canada. R-SEA has an explicit focus on identifying and evaluating the CE of a region, and by such it applies wide topical boundaries and focuses on the receiving environment by default (CCME 2009). The effectiveness of R-SEA depends on whether it builds on strategic visions and is properly tiered to the operational permit-granting managerial level (Gunn and Noble 2009; Johnson et al. 2011), but R-SEA is arguably the most appropriate SEA context for CE assessment (Duinker and Greig, 2006; Harriman and Noble, 2008).

The results of this present study suggest that SEA practitioners are prone to consider CE as solely the joint stress of the actions occurring within the development boundaries. We suspect that the intrinsic focus of proposed, bounded developments (as opposed to the receiving environment) is what prohibits meaningful assessment of CE in IAs made outside a R-SEA context. One might hypothesise that for CE assessment to work in SEA there is a need for a conceptual change of focus rather than wider plan boundaries *per se*. However, more research is needed before this hypothesis can be verified or rejected.

$P_2$6 Conclusions

Assessment of CE is an important IA element, which is done badly across the world. Authors have argued that SEA provides a suitable platform for addressing CE due to the broader boundaries of programmes, plans and policies, but few have challenged this claim. Through a case study on the Danish mining sector, this article explored the following hypothesis: *Plan boundaries influence the analytical boundaries applied for CE assessment in SEA*.

It was found that there exists a good conceptual understanding of CE among the Danish mining planners. However, this understanding is only partially applied in practice. At large, planners fail to relate the cumulative stress arising from the actions of the mining plans to the total stress on and capacity of the receiving environment. Most practice on CE assessment is furthermore both implicit and undocumented. Having these deficiencies in mind, it was found that CE are somewhat assessed and managed throughout the planning process. With respect to *'traffic'*, for instance, strategies for lowering the cumulative transport are formulated early in the planning process and used for establishing new mining zones. Hence, the study suggests that CE assessment may be a bigger part of IA practice than previously described if one casts aside the term *'CE'* and instead focuses on the nature of the evaluations made.

Plan boundaries were found to influence the analytical boundary applied for CE assessment, but all boundaries are not equally influential. The geographical boundary was found to be broad enough to encompass the actions leading to the most relevant CE. The time boundary is too narrow *per se*, but it does not influence the CE assessment. The topical boundary restricts a meaningful assessment of CE. One might argue that to be effective CE assessment must move beyond narrowly defined SEA contexts (such as sector SEA) to broad IA platforms similar to R-SEA. Yet, all proposed developments are essentially bounded and CE assessment will thus arguably remain restricted in most IA applications.

More resources, data, collaboration, leadership and legislation can facilitate better CE assessment in bounded SEAs. However, the study suggests that a conceptual change may be needed to further relate assessments to the receiving environment. Thus the study confirms the concerns of Gunn and Noble (2011), who anticipated conceptual challenges in applying a receptor-oriented CE analysis in the development-oriented SEA.

STUDY #3

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THE 'GREY' ASSESSMENT PRACTICE OF IA SCREENING: PREVALENCE, INFLUENCE AND APPLIED RATIONALE

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Abstract

Research focusing on the practices surrounding screening in Impact Assessment (IA) is limited. Yet, it has been found that development proposals sometimes are adjusted through an informal dialog with IA practitioners prior to or during screening. Such practice is often referred to as *'grey IA'* in Denmark. This article explores the prevalence, influence and applied rationale of grey IA. Through a questionnaire, data were collected from 121 IA practitioners working within the fields of environmental impact assessment and strategic environmental assessment. It was found that grey IA is a common practice, which influences the outcomes of formal screening procedures through the consideration of impacts on neighbours and spatial zones of protection. Grey IA is to some extent motivated by the opportunity to save the resources required for full-scale IA, but an additional *'green'* rationale also exists. Grey IA may influence the effectiveness of IA systems, but further research is needed before any conclusions can be made.

$P_3$1 Introduction

Impact Assessment (IA) refers to the process of identifying and evaluating the future consequences of proposed developments with the purpose of facilitating environmentally sound development through transparent decision-support (IAIA 2009). The multiple scales at which developments can be proposed, in combination with the diversity of impacts they can afflict, have caused IA methodology to develop into a jigsaw puzzle of sub-tools (Pope et al., 2013). The most widely used are arguably the Environmental Impact Assessment (EIA) and the Strategic Environmental Assessment (SEA), which apply to the project level and the plan, programme or policy levels, respectively (Morgan, 2012; Tetlow and Hanusch, 2012).

Today IAs accompany decision-making in all but two of the world's nations (Morgan, 2012). However, the tools are showing mixed performance, and thus IA applications often fail to deliver the promise of facilitating better decisions (Bidstrup and Hansen 2014; Cashmore et al. 2004; Phylip-Jones and Fischer ; Pope et al. 2013; Tetlow and Hanusch 2012; Therivel et al. 2009). Extensive research has been undertaken on the various factors influencing IA quality (van Doren et al. 2013), but few have explored the dynamics of the initial screening procedure, which determines whether an IA is needed (Pinho et al. 2010; Weston 2011).

Screening practices are complex, though their purpose is simple. Research has shown that screening can serve as an environmental evaluation because it represents the first encounter with the IA system (McGillivray 2011; Nielsen et al. 2005; Weston 2011). Such practice is common under the American NEPA act, which allows IA authorities to openly pass proposed actions subject to conditional mitigation measures (McGillivray, 2011), but it is more informal outside a NEPA context. In Denmark, it is often referred to as *'grey IA'*. The most extensive study on this phenomenon is described by Nielsen et al. (2005), who in an analysis of 98 EIA screening decisions found that 45% had been adjusted either prior to or during the screening procedure through dialog between the developer, his private consultant and in some cases the IA authority. All studies on grey practice have dealt with EIA screening only. However, a recent discussion session among 60 IA practitioners have indicated that SEA screening may likewise serve multiple purposes different from that of merely evaluating whether an assessment is needed (Hansen 2014).

Other studies have found that among development authorities there exists a culture of IA resistance (Weston, 2011), which is rooted in a wish to save resources and expressed through screening processes by a willingness to pass proposed developments without IA (João and McLauchlan 2014; Macintosh and Waugh 2014; Weston 2000). Weston (2011) argues that such resistance may undermine the purpose and effectiveness of IA.

Through a questionnaire distributed among Danish IA practitioners, this article explores the prevalence of grey IA practices, the influence of grey IA on the outcome of screening procedures and the extent to which grey IA is rooted in economically motivated IA resistance. The study was structured around the following research hypothesis: *Grey IA is widely prevalent in Danish IA practice, it influences the outcomes of screening procedures, and it is motivated by the opportunity to save resources*. The study aimed to elaborate further on the informal mechanisms surrounding screening procedures and in this way contribute to the debate about the multiple factors influencing the performance of IA. The article opens with a brief presentation of screening procedures and grey IA. Subsequently, the methodology for data collection is explained, after which the results on the prevalence, influence and rationale of grey IA are presented and discussed. Lastly, the findings are contextualised within the debate on IA effectiveness.

$P_3$2 Screening and grey practice in IA

The International Association for Impact Assessment – IAIA - (1999) describes the purpose of screening as that of determining whether a development proposal should be subject to IA. Within the 28 member states of the European Union (EU), IA is legally required for all projects and plans which are likely to cause significant environmental effects (European Parliament 2001; 2014). Hence, a screening is ultimately an evaluation of environmental *'significance'* in regard to development activities. A screening can be made either by comparing characteristics of the proposal to fixed criteria for when projects and plans can be considered environmentally significant or as a case-by-case evaluation (European Parliament 2001; 2014). Screening practices vary substantially within the EU, but most nations apply a combination of the two approaches (Pinho et al., 2010).

Figure $P_3$1 illustrates the procedure for the screening of a proposed project or plan. Theoretically, a developer will independently produce a plan or project proposal, which then is submitted to the competent authority upon requesting a development permit. Having no prior knowledge of this proposal, the authority assigns an IA practitioner to conduct a screening procedure aimed at determining whether the development is likely to cause significant impacts. This procedure can be depicted as a pass/fail test. If the proposal is found to produce no significant impacts, the screening is *'passed'* and administrative procedures for granting a permit can be initiated. If the proposal is found to cause significant impacts, the screening is *'failed'* and an IA must be assigned. This IA will evaluate the potential impacts and communicate how these can be avoided, reduced or mitigated.

Reality does, however, sometimes differ from theory. In practice, developers are often not interested in having an IA assigned to their proposal since such an undertaking will delay the permit (Nielsen et al., 2005) – as illustrated in Figure $P_3$1. An IA-based permit can, moreover, contain conditions (as a means of avoiding, reducing or mitigating impacts) which could be difficult and expensive to incorporate at a late stage in the process (McGillivray, 2011). IA practitioners can likewise be reluctant to *'fail'* too many screenings since each IA will demand time, resources and perhaps a need for the capacity of external experts (João and McLauchlan, 2014; McGillivray, 2011; Weston, 2000). Hence, several factors are generating an overall interest in avoiding *'failed'* screenings.

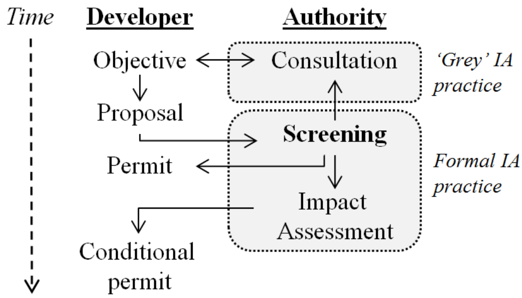

Figure $P_3$1: The screening procedure. IA is required for all proposals, which are likely to cause 'significant' impacts. However, a grey and informal practice may change proposals.

This mutual interest in avoiding formal IA practice can spark an informal, *'grey'* IA practice (see figure $P_3$1), which aims at adjusting proposals to a state where they will not display a concern for significant environmental impacts in subsequent, formal screening procedures. Such practice can either have the form of developers consulting IA practitioners while shaping their development proposals or the form of IA practitioners encouraging developers to withdraw the proposal and reconsider certain elements. Grey practice is possible because case-by-case evaluations of significance ultimately rely on a discretionary judgement (Wood and Becker 2005), which can be subjective, normative, value-based and even political (Lawrence 2007). The practice is referred to as *'informal'* because it occurs outside the formal framework and guidelines for full-scale IA. In Denmark, many refer to this practice as *'grey'* because it is happens rather undocumented in the *'shadows'* of the traditional IA system. It may also refer to the practice being somewhat less complete than a full-scale IA, for which reason it lacks certain *'colours'*.

$P_3$3 Method

$P_3$3.1 A focus on Danish municipal screening

IA authority is in Denmark divided across the tiers of the national planning hierarchy. Policies and strategies for development are made on a national scale, after which more detailed development plans are produced by the five regions and the 98 municipalities. The institution proposing a development policy, programme or plan is responsible for the assigned SEA (DMEF 2013c), while the permit-granting authority most often is responsible for the EIA assigned to a proposed project (DMBG 2015). In practice, this means that *'developer'* and *'IA authority'* typically are found within the same institution for SEA practice, while they often are separated for EIA practice (where many projects are proposed by private actors). This does, however, not change the formal requirements for IA screening.

The municipalities are by far the most influential and diverse IA authority in Denmark since they propose the bulk of all development plans (for instance in regard to water, sanitation, waste, heating, business and dwelling) and administer the assigned permits. Hence, the study focused on municipal screening practices. In regard to SEA, the research solely focused on plans since this term covers programmes as well in a Danish context. Grey practice was not explored in regard to policies since these generally are made on higher institutional tiers than the municipal one. The term *'IA practitioner'* does in this study cover municipal employees appointed to screen incoming plans and project proposals for IA and (if necessary) conduct the assigned assessments.

$P_3$3.2 Designing a questionnaire survey

The focus on *'prevalence'* generated a need for a large sample of IA experiences, and it was therefore chosen to collect data through an electronic questionnaire. Expecting that a lengthy survey could discourage some practitioners to voluntarily contribute with data on their working practices, the questionnaire was designed to consist of only nine questions (abbreviated Q_1 to Q_9). It can be found in an English version as appendix P_3A. The questionnaire inquired about the IA practice of the respondent (Q_1), the prevalence of grey IA (Q_2 to Q_4), the influence of grey IA (Q_5), the economic rationale for grey IA (Q_6 to Q_7), and the environmental considerations of grey IA (Q_8). It furthermore enabled each respondent to provide comments (Q_9).

When studying prevalence, it is as important to receive data from practitioners unfamiliar with grey IA as it is to collect data from those who apply grey IA. For this reason, the questionnaire opened with a question on the IA competences of the respondent (Q_1) before he/she was confronted with a description of and questions in regard to grey practice. Any practitioner logging out after such confrontation would in this way still leave data. Question one furthermore had the purpose of enabling division of the data sample into pools of respectively EIA and SEA practitioners.

The *'prevalence'* of grey IA was explored through questions on the existence (Q_2 and Q_3) and commonness (Q_4) of such practice in the municipality of each respondent. The *'influence'* of grey IA was studied through inquiry about the extent to which the practice affects subsequent screening outcomes (Q_5). The *'rationale'* for grey IA was explored by questioning practitioners about the extent to which the practice is motivated by the opportunity to save resources (Q_6), digging into whether it exists to a similar degree when full-scale IA cannot be avoided (Q_7).

An additional question eight (Q_8) was included for further perspective. It inquired practitioners about the extent to which multiple environmental parameters are considered when conducting grey IA. These parameters were categorised into three broad themes, covering concerns of interested parties, overlaps with spatial zones of protection and holistic implications – see appendix P_3A. By doing so, themes of impacts were studied rather than specific impact categories. These themes were based on the managerial reality of municipal IA practitioners, who need to include the public while working in a zone-oriented spatial context where many inter-connected plans and strategies apply.

Though used freely within certain circles of the Danish IA community, the term *'grey IA'* does imply an expectation of an opaque and somewhat less extensive IA practice (as argued in section $P_3$2). For the sake of objectivity in the questionnaire, the term was therefore not used explicitly. The questionnaire was designed as *'open'*, allowing all practitioners with access to a link to fill in their experiences.

$P_3$3.3 Data collection and analysis

A link to the questionnaire was distributed to the environmental departments of all 98 Danish municipalities on September the 23rd 2014. The survey closed for answers on October 20th 2014, leaving the municipalities four weeks to respond. In total 121 responses were collected from IA practitioners, of which 100 were fully completed and 21 were partially completed. The population consisted of 65 practitioners working with both EIA and SEA, 37 practitioners working exclusively with EIA and 19 practitioners working exclusively with SEA. Consequently, the study had a sample size of 102 (65+37) EIA practitioners and 84 (65+19) SEA practitioners. This is comparable to the related studies of Weston (2000), which had 115 respondents, and Wood and Becker (2005), which had 107 respondents. A sample of 26 written comments was obtained through question nine. These comments have been translated to English with respect to both message and structure.

The data obtained from the questionnaire can be categorised as either nominal or ordinal. There is no statistical order between IA types or yes/no answers and the first three questions (Q_1 to Q_3) thus yielded nominal data. Respondents were in the subsequent questions asked to rank the non-numerical concepts of commonness (Q_4), influence (Q_5), motivation (Q_6), similar adjustment (Q_7) and environmental consideration (Q_8) from *'1'* to *'5'* (expressed literally in the text to improve readability). Data from these five questions is ordinal since it, from a statistical perspective, provides some degree of order, though it lacks measurability. The collected data were largely analysed by the use of frequency diagrams. For the ordinal data, median values were applied as central tendency measures.

$P_3$4 Results

$P_3$4.1 The prevalence of grey IA

Responses to the questionnaire revealed that 72% of EIA practitioners and 80% of SEA practitioners know of grey practice occurring in their own municipality. Among these practitioners, most appear to be familiar with the phenomenon (question four). A median value of *'4'* was obtained for both EIA and SEA, and thus it appears grey IA is *'common'* – see figure $P_3$2. No great differences can be found between EIA and SEA practices. However, a slightly higher fraction of SEA practitioners appear to describe grey practice as *'very common'*.

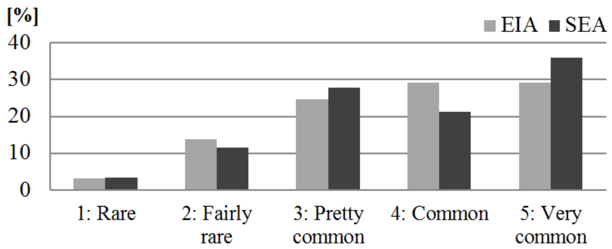

Figure $P_3$2: The commonness of grey IA among those familiar with the practice (Q_4).

$P_3$4.2 The influence of grey IA

In regard to question five, practitioners graded the influence of grey practice in EIA and SEA a median value of *'3.5'* and *'3'*, respectively. With a reference to the terminology of figure $P_3$3, it appears grey IA has *'some'* to *'large'* influence on whether a full-scale IA will be required in subsequent, formal screening procedures. In fact, 18% of grey EIA practitioners stated that grey practice influences to a *'very large extent'* while only 5% found grey practice to influence to a *'very low extent'*.

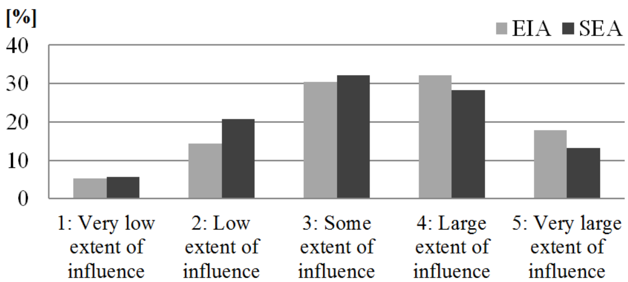

Figure $P_3$3: The influence of grey IA on the outcome of screening procedures (Q_5).

$P_3$4.3 The economic rationale for grey IA

In regard to question six, it was found that grey IA is motivated by the opportunity to save the resources required for full-scale IA. Yet, differences exist between EIA and SEA. The fraction of practitioners motivated to a *'very large extent'* was 26% for EIA practice but only 11% for SEA practice. Likewise, the median values of the samples were found to be *'4'* for EIA but *'3'* for SEA. With a reference to the terminology of figure $P_3$4, it appears that the economic opportunities assigned to grey IA to a *'large'* extent motivate EIA practitioners, while they only to *'some'* extent cover the rationale applied for justifying grey SEA.

Some practitioners verified the research hypothesis of the study by confirming that efforts are made within municipal screening practices to *"avoid impact assessment, since it can be both time and resource demanding"*. One practitioner elaborated further on this rationale by expressing that the opportunities for economic savings differ from case to case. This particular practitioner has experienced that for big projects it *"costs more resources trying to avoid EIA through pre-screening adaptation, than it would have cost to make an EIA right away"*.

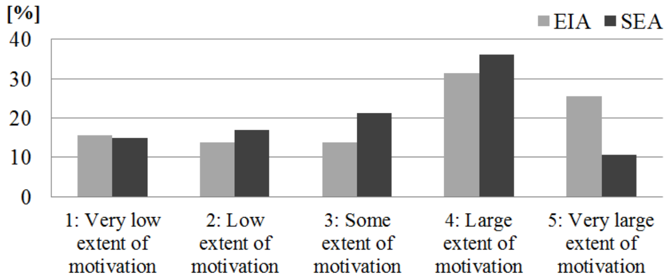

Figure $P_3$4: The extent to which grey IA practice is motivated by the opportunity to save the time and resources of a full-scale IA (Q_6).

The respondents showed a greater degree of doubt in question seven when asked about whether similar practice is performed to proposed developments, which undoubtedly would require IA. To this question, the percentage of *"no knowledge"* answers increased from around 10% (Q_5 andQ_6) to 33% for EIA and 23% for SEA. The results of the remaining sample are depicted in figure $P_3$5.

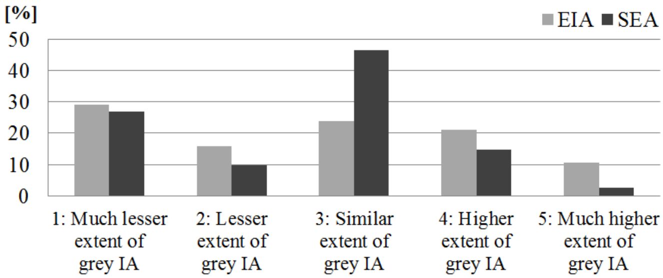

Figure $P_3$5: The extent of grey practice when full-scale IA cannot be avoided (Q_7).

EIA practitioners show little consistency on this question. SEA practitioners are divided into two larger fractions: one where grey IA is practiced to a similar extent and another where it is practiced to a much lesser extent. Median values of *'3'* for both EIA and SEA indicate that grey IA is practiced to a similar extent.

$P_3$4.4 The environmental considerations of grey IA

Figure $P_3$6 provides insight regarding the analytical scope of grey IA. The concerns of neighbours and the influences of activities on zoned areas of protection are considered to a *'large extent'* (median values of *'4'*), while the relation to other plans and strategies as well as cumulative impacts are considered to *'some'* extent (median values of *'3'*). Consideration of global impacts and the concerns of NGOs appear to receive a *'low'* extent of consideration (median values of *'2'*).

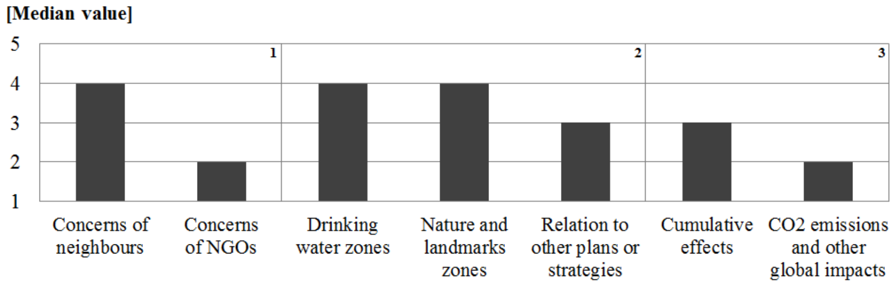

Figure P$_3$6: The environmental considerations of grey IA (Q$_8$). The categorisation of impacts: (1) concerns of interested parties, (2) impacts on spatial zones of protection, and (3) holistic implications. High median values indicate a high degree of consideration.

P$_3$5 Discussion

P$_3$5.1 The limitations of the research design

Research on the *'prevalence'* of a certain practice requires a large study sample. Hence, data were collected through an electronic questionnaire, which was designed in a short format to encourage a high respond rate. This research strategy arguably worked since responses were obtained from 102 EIA practitioners and 84 SEA practitioners, but it also proved to limit the study. The questionnaire restricted the author in asking follow-up questions, while the short format limited analysis of multiple other facets of grey IA.

The *'influence'* of grey IA was solely studied as the extent to which it affects screening outcomes. Knowing now that the practice is influential, it could have been of great value to explore further how screening decisions are altered. Does grey practice facilitate genuine, substantive changes in development proposals, or is it only a means of minor, cosmetic adjustments? This question remains unanswered. The study of the *'rationale'* was likewise limited since only the economic motivation for grey practice was explored (questions six and seven). A study on alternative rationales could have provided further input on the nature of grey practice. Hence, the focus on *'prevalence'* proved to somehow restrict the analysis of both *'influence'* and *'rationale'*.

In addition, more in-depth questioning could have explored the mechanisms creating and driving grey IA, the differences between grey EIA and grey SEA, and the practical differences (or lack thereof) between grey IA and the early adjustments of formal practice. The obtained results on the prevalence and influence of grey IA have made these related issues important to explore and thus the study has generated a need for further research.

P$_3$5.2 Grey IA is here to stay

The case study demonstrates that grey IA still exists in Denmark, though a decade has passed since the data of Nielsen et al. (2005) were collected. This is noticeable in a Danish context since a nationwide political reform fundamentally altered the institutional landscape in 2007. The reform set out to make the public sector more efficient by clustering its functions in larger institutions, and it entailed a complete restructuring of tasks where 13 counties and 270 municipalities were merged into five regions and 98 municipalities (DMIH, 2005). The new, bigger municipalities were granted authority of most development tasks and the assigned permits, while the new regional institutions were granted the tasks of managing the national health care system and formulating broad development strategies for their region as a whole (DMIH, 2005). In practice, this meant that most responsibilities for IA were moved from the county level to the municipal level, requiring the intake of new IA staff and the formulation of new IA practices in newly formed institutions. Many of today's municipal IA practitioners are the same individuals who used to practice IA in the former counties, but the reform meant that IA practitioners from various counties were mixed among each other and among municipal employees new to the field of IA. The fact that grey practice has survived such turbulent conditions indicates that it is a resilient phenomenon to which IA practitioners assign value. It proves that grey practice (at least in Denmark) is here to stay.

Screening practices vary substantially across nations (Pinho et al., 2010), and it is therefore not possible to universally apply the results of this study to all IA contexts. Grey IA has been documented in both the USA and the UK (McGillivray, 2011). Yet, it is quite possible that some form of grey practice, through which practitioners operationalize rigid IA requirements, exists in other countries – as recently discussed by Macintosh and Waugh (2014).

$P_3$5.3 Grey SEA and its characteristics

Prior to this study, grey practice had only been described in regard to EIA. Yet, it was found that grey SEA actually is more prevalent than grey EIA, though less motivated by the opportunity to save resources. These differences could be rooted in the structure of the Danish IA system, where plans (unlike projects) normally are proposed by the same institution performing the SEA screening (though occurring in different departments). In this regard, Hansen (2014) found that Danish planners often draw on their in-house IA capacity to better their planning by using screening templates as procedural tools for both internal communication and external documentation. Similar tendencies have been found by Weston (2011), who in an English context found planning authorities to conduct project appraisal, and Phylip-Jones and Fischer (2015), who found German planners to address environmental concerns before SEA was formally applied. Thus there are indications pointing in the direction of grey SEA being a way of integrating SEA elements early in the planning process. Further research is, however, needed in this regard.

$P_3$5.4 The alternative '*green*' rationale for grey IA

The author formulated question six based on the research hypothesis, which stated that grey practice is motivated by the opportunity to save resources. By doing so, it was somewhat expected that grey IA serves as a form of *'discount IA'* which through an economic rationale circumvents formal requirements for full-scale IA – in line with the concerns of McGillivray (2011) and Weston (2011).

It was found that grey IA is motivated by an economic rationale, and one respondent did in this regard admit that the practitioners of his institution have *"speculated in how to avoid EIA"*. However, 13 out of the 26 comments provided in the open question nine (50%) were found to have the purpose of directly opposing the economy-driven motivation for grey IA. Some practitioners clearly felt that the study focus on this sole explanation for the prevalence of grey IA did not do their practice justice. One respondent ensured: *"The adjustment and dialog taking place prior to and during screening is a lot about us wanting to ensure a good project and avoid environmental impacts"*. Considering that the practice is referred to as *'grey'* because it somewhat happens in the *'shadows of the IA system'*, it was found that practitioners remain remarkably open about its existence. One respondent found grey IA to be a sign of a *"healthy"* management practice, while another see *"little difference between that and their general consultation tasks for developers unfamiliar with legislation"*. As one respondent rhetorically questioned: *"... is it not merely a good thing to use the screening process as a working tool for optimizing one's plan or project proposal?"*

Though grey IA is motivated by the opportunity to save resources, it appears that additional rationales for its prevalence can be found. The study indicates that there exists a *'green'* rationale, by which practitioners may facilitate better developments early on in the IA process through dialog. This is of interest since early integration is widely recognised as a key-element in securing both effective SEA (Partidário 2012; Stoeglehner et al. 2009; Therivel 2010; van Buuren and Nooteboom 2009; van Doren et al. 2013) and EIA (Cashmore et al. 2004; IAIA 1999).

$P_3$5.5 Grey practice and IA effectiveness

Knowing now that grey IA is both prevalent and influential, the study can be contextualised within the debate on effectiveness. The early work of Sadler (1996) describes *'effectiveness'* as the measure of *"how well something works or whether it works as intended and meets the purposes for which it is designed"*. Within the context of grey IA, that *'something'* could be discussed as both *'screening'* and *'IA'*.

Screening effectiveness relates to the procedural purpose of a screening procedure, which according to the International Association for Impact Assessment (1999) is to determine whether an IA is needed. The conditions for determining such a need can be discussed. However, a reasonable interpretation could be that a screening procedure is effective if it succeeds in highlighting the necessity for assigning IA to a proposed development, which is likely to cause significant environmental impacts. It can be argued that screening effectiveness is decreased when grey practice is used for circumventing IA with a sole economic rationale. Instances of this have been seen in Denmark, where developers in the mining sector are encouraged to apply for smaller consecutive extraction permits with the purpose of avoiding that full-scale EIA, which their multiple smaller projects would require jointly[6]. Internationally, such misuse of IA screening (often referred to as '*salami-slicing*') was highlighted

[6] This practice is publicly known within the Danish mining sector. It has been documented by the author through multiple face-to-face interviews and e-mails.

as a continuous challenge in the official report on the application and effectiveness of the European EIA Directive (European Commision 2009a). Yet, the existence of the additional *'green'* rationale for grey IA fundamentally changes the assumptions of the study since it suggests that pre-screening procedures can have the aim of improving development proposals to an extent where they might not generate significant impacts. The influence of grey IA on screening effectiveness may in this way depend on the rationale applied by the practitioners.

In regard to IA, effectiveness is conceptually more disputable. The intended purpose of IA is pluralistic (Bond and Morrison-Saunders 2013), and no universal definition or criterion for IA effectiveness thus exists (van Doren et al., 2013). Sadler (1996) argues that IA effectiveness can be categorised as *'procedural'* (compliance with established principles and formal requirements), *'transactive'* (minimal use of time and resources) and *'substantive'* (facilitation of environmentally sound decisions). Yet, Baker and McLelland (2003) suggest that there is a normative side to effectiveness as well, in the sense that IAs should facilitate institutional learning, change views and accommodate societal discourses (Chanchitpricha and Bond 2013; Rozema and Bond 2015). One could define even more types of effectiveness by further subdividing and interpreting the pluralistic purpose of IA. The discussion of grey practice will only be contextualised through the three initial categorisations of Sadler (1996), though this is a simplification.

Cashmore et al. (2009) argue that IA effectiveness can be evaluated on both a macro level (assessment system) and a micro level (individual application). This present study does not support any conclusions on micro level effectiveness, but the findings do provide insight on how the Danish IA system performs as a whole.

Formally speaking, screening procedures should solely evaluate whether proposed developments pose a risk of significant impacts. Yet, the assessment-like procedures and informal conversation arranged around screening practices in Denmark assign this particular IA step an alternative function. Such practice conflicts with the procedural purpose (and conservative depiction) of screening as merely a pass/fail test, and thus it can be argued that grey IA influences the procedural effectiveness of the IA system negatively. Yet, it was found that grey practice can be undertaken with a *'green'* rationale, by which resources are saved through early environmental improvement of proposed developments. Under such circumstances, grey practice arguably increases transactive effectiveness.

Cashmore et al. (2004) argue that lacking conformance with formal requirements does not mean that IA practice *de facto* is ineffective. Hence, a critical issue within the debate on grey practice appears to be that of evaluating whether it facilitates IA in achieving its substantive purpose of supporting sustainable development. The study did not analyse the nature of the changes made to development proposals by grey practice, and thus no grand conclusions can be drawn in regard to substantive effectiveness. Yet, it was found that adjustment occurs in most municipalities, within which around 55% describe it as either *'common'* or *'very common'* – see figure $P_3$2. In comparison, Nielsen et al. (2005) and Wood and Becker (2005) found full-scale EIA to cover only 3% and 0.1% of the proposed projects, respectively. In the present study, one practitioner elaborated as follows: *"Regardless of our adjustments, we have very few projects resulting in a full-scale IA"*. Hence, the study suggests that grey IA influences more plan and project proposals than full-scale IA. By such, it can be hypothesised that grey practices influence the substantive effectiveness of the IA system. If legitimised through a sole economic rationale and tailored for deliberately circumventing IA requirements, one might expect that grey practice decreases substantive effectiveness. If legitimised through a green rationale and undertaken with pro-active environmental improvement in mind, one might reversely expect that grey practice contributes to substantive effectiveness. Testing this hypothesis in future research is of importance.

$P_3$6 Conclusion

This present study set out to explore the prevalence, influence and rationale of the grey assessment-like practices surrounding IA screening. The study inquired 121 Danish IA practitioners about their personal experiences with grey IA through a questionnaire survey built around the following research hypothesis: *Grey IA is widely prevalent in Danish IA practice, it influences the outcomes of screening procedures, and it is motivated by the opportunity to save resources*.

Grey IA was found to be a common practice within both EIA and SEA, which through considerations of the impacts on neighbours and spatial zones of protection influences the perceived need for full-scale assessment. The practice has proven resilient enough to survive widespread reforms in the institutional IA system, and thus it is fair to assume that in Denmark it is here to stay. Grey IA is motivated by the opportunity to save resources, but additional explanations for its prevalence also exist. Here among is a '*green*' rationale, by which grey IA is a means of environmentally improving development proposals early in the design process. Hence, the first two claims of the research hypothesis were verified while the last claim was only partially verified.

Having found that grey IA is both influential and widely prevalent in Danish IA practice, the study raises multiple questions for further research. What types of changes are made in grey IA? How and why do grey EIA and grey SEA differ? What rationales for grey IA are dominating? And, to what extent is grey IA really *'grey'* if practitioners openly defend it as environmental consultancy?

The study was conclusively contextualised within the debate on effectiveness. The influence of grey IA on screening effectiveness depends on the rationale applied by practitioners. If grey IA is conducted with the sole purpose of saving resources, it may undermine screening effectiveness. Yet, this might not be the case if grey IA is made with a proactive, environmental rationale as point of departure. From a holistic IA perspective, it was argued that grey practice sacrifices procedural effectiveness for the sake of increasing the transactive one. In addition, the study findings tentatively indicate that grey practice may influence substantive effectiveness by altering more proposed developments than full-scale IA.

The article demonstrates that IA practitioners act in an IA system, which consists of more than formal requirements. Grey areas exist wherein practitioners can act autonomously. As expressed by one respondent:

> *"It is good that someone is investigating and focusing on this area. But one must remember that planning reality may differ from formal procedures, though practiced within the boundaries of legislation. The world is neither black nor white. Rather, it is grey."*

Acknowledgements

The author wishes to thank the 121 municipal IA practitioners who participated in the study. Furthermore, this research would not have been possible without the insight of Associate Professor Anne Merrild Hansen, who held a workshop on quality in SEA screening at DCEAs 2014 Environmental Assessment Day seminar, and Associate Professor Matthew Cashmore, who provided critical feedback in the later iterations of the study.

Appendix P_3A: Questionnaire

The questionnaire distributed to the municipal IA practitioners is displayed in a translated form. It was designed in the software programme SurveyXact (www.surveyxact.dk). Question eight did originally show after question four, but it has been moved to after the questions on the influence and rationale for grey IA practice (Q_5 to Q_7) for pedagogical reasons upon writing the article.

1) What type of IA are you practicing?	EIA ( )	SEA ( )	Both EIA & SEA ( )	I do not practice IA ( )		
The term SCREENING refers to an initial evaluation of whether a proposed project or plan may result in significant environmental effects. The primary purpose of a screening is to determine whether an IA is needed. However, studies and seminars hosted by the Danish Centre for Environmental Assessment have indicated that screening often is given an assessment-like function in practice. It appears that dialog with developers prior to (or during) EIA screening can result in project adjustments. Similarly, it appears that some plan proposals are adjusted as a consequence of a pre-screening procedure before the finished plan proposal is made public.						
2) Do you have knowledge of such practice in your municipality?	Yes ( )	No ( )				
3) When does such practice occur?	Yes	No	N.K.*			
In EIA screening of project proposals:	( )	( )	( )			
In SEA screening of plan proposals:	( )	( )	( )			
4) How common is this practice in your opinion?	(1= rare, 5 = very common)					
	1	2	3	4	5	
In EIA screening of project proposals:	( )	( )	( )	( )	( )	
In SEA screening of plan proposals:	( )	( )	( )	( )	( )	
5) To what extent does such adjustment influence the outcome of subsequent screening (the need for an IA)?	(1 = no extent, 5 = a large extent)					
	1	2	3	4	5	N.K.
In EIA screening of project proposals:	( )	( )	( )	( )	( )	( )
In SEA screening of plan proposals:	( )	( )	( )	( )	( )	( )
6) To what extent is such adjustment motivated by an opportunity to save the time and resources assigned to an IA?	(1 = a very low extent, 5 = a very large extent)					
	1	2	3	4	5	N.K.
In EIA screening of project proposals:	( )	( )	( )	( )	( )	( )
In SEA screening of plan proposals:	( )	( )	( )	( )	( )	( )
7) To what extent is similar adjustment made to a proposed project or plan which regardless of adjustment would require an IA?	(1 = a much lesser extent, 5 = a much higher extent)					
	1	2	3	4	5	N.K.
In EIA screening of project proposals:	( )	( )	( )	( )	( )	( )
In SEA screening of plan proposals:	( )	( )	( )	( )	( )	( )
8) To what extent are the following environmental parameters considered during the adjustment of proposed projects and/or plans?	(1 = a very low extent, 5 = a very large extent)					
	1	2	3	4	5	
Concern of neighbours:	( )	( )	( )	( )	( )	
Concern of NGOs:	( )	( )	( )	( )	( )	
Impact on drinking water zones:	( )	( )	( )	( )	( )	
Impact on protected nature and landmarks:	( )	( )	( )	( )	( )	
Other conflicting geographical zoning:	( )	( )	( )	( )	( )	
The relation to other plans and strategies:	( )	( )	( )	( )	( )	
Cumulative effects generated with other:	( )	( )	( )	( )	( )	
CO_2 emissions and other global impacts:	( )	( )	( )	( )	( )	
9) Do you have comments about the screening practices in your municipality?	[______________________]					

* *'No Knowledge' is abbreviated 'N.K.'*

STUDY #4

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LIFE CYCLE THINKING IN IMPACT ASSESSMENT: CURRENT PRACTICE AND LCA GAINS

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Abstract

It has been advocated that life cycle thinking (LCT) should be applied in impact assessment (IA) to a greater extent since some development proposals pose a risk of significant impacts throughout the interconnected activities of product systems. Multiple authors have proposed the usage of life cycle assessment (LCA) for such analytical advancement, but little to no research on this tool application has been founded in IA practice so far. The aim of this article is to elaborate further on the gains assigned to application of LCA. The research builds on a review of 85 Danish IA reports, which were analysed for analytical appropriateness and application of LCT. Through a focus on the non-technical summary, the conclusion and the use of specific search words, passages containing LCT were searched for in each IA report. These passages were then analysed with a generic framework. The results reveal that LCT is appropriate for most of the IAs, but that LCA is rarely applied to provide such a perspective. Without LCA, the IAs show mixed performance in regard to LCT. Most IAs do consider the product provision of development proposals, but they rarely relate impacts to this function explicitly. Many IAs do consider downstream impacts, but assessments of upstream, distant impacts are generally absent. It is concluded that multiple analytical gains can be attributed to greater application of LCA in IA practice, though some level of LCT already exists.

$P_4$1 Introduction

Most proposed projects, plans and other development schemes have the inherent purpose of changing the societal status quo by for instance increasing economic activity, mobility or resource availability. Such proposals may generate unwanted societal effects, and impact assessment (IA) has thus grown to become a widely used and legally required tool for public decision support worldwide (Morgan, 2012). An IA can be divided into the four procedural steps of screening, scoping, impact analysis and reporting. *'Screening'* refers to the initial evaluation of whether a proposed development is likely to cause significant impacts. If this is the case, practitioners must identify the issues of concern in the *'scoping'* phase. The impacts of the development are evaluated and compared to reasonable alternatives in the *'impact analysis'*. Lastly, an IA report, which documents the process and the embedded considerations, must be prepared – see IAIA (1999) for further knowledge on IA principles. The field of IA has developed in recent years to comprise a mixture of tools. Yet, two of the most common are the Environmental Impacts Assessment (EIA), which is applied for projects, and the Strategic Environmental Assessment (SEA), which is applied at the strategic stages of decision-making in regard to a region or a sector as a whole (Morgan, 2012; Pope et al., 2013; Tetlow and Hanusch, 2012; Therivel, 2010).

IA can be characterised as a procedural type of tool for environmental analysis since it serves both an analytical and a procedural purpose (Finnveden and Moberg 2005; IAIA 2009). The analytical purpose is to ensure information on environmental impacts through the application of appropriate sub-tools for rigorous assessment; the procedural purpose is to provide a framework for a transparent and participative decision-making process (IAIA, 1999). Several authors have argued that life cycle assessment (LCA) is an appropriate analytical tool for application in both EIA and SEA (Björklund 2012; Finnveden and Moberg 2005; Finnveden et al. 2003; Fischer 2007; Jeswani et al. 2010; Loiseau et al. 2012; Manuilova et al. 2009; Tukker 2000), and recent research has proposed formal procedures for such integration (Bidstrup et al. 2015; Loiseau et al. 2013; Židonienė and Kruopienė 2015).

LCA is a tool for assessing the environmental impacts of a *'product'* (a term covering both goods and services) throughout its life cycle by compilation and evaluation of the inputs and outputs assigned to the interconnected product systems (ISO 2006a). The inputs of resources and the assigned outputs of emissions are typically compiled by the aid of large, electronic databases which contain data on the causal relationships of economic systems, see for instance Ecoinvent Centre (2013). The environmental evaluation is conducted by the use of LCA software, which projects and quantifies how outputs from the technosphere result in environmental impacts, such as global warming, release of respiratory inorganics and eutrophication (Finnveden et al. 2009). The most distinct feature of LCA is the exclusive focus on product provision and the application of a life cycle perspective (Finnveden et al. 2009; Finnveden and Moberg 2005). An LCA of a production line will in this way not focus on the production facility *per se*. Rather, impacts will be attributed each produced unit (the *'function'* of the system) and span from the impacts assigned to the acquisition and refinement of the resources required for its production (*'upstream impacts'*) to the impacts imposed during its use and disposal (*'downstream impacts'*) – see European Commission (2010).

The rationale for applying the product-oriented LCA tool in the procedural IA tool lies in the realisation that proposed developments are often product systems components, which can influence product flows (and the assigned emissions). This point is well-demonstrated in the case study of Björklund (2012) on energy planning, the case study of Bidstrup et al. (2015) on raw materials extraction and the case study of Židonienė and Kruopienė (2015) on a proposed industrial project. Since proposed developments may influence product systems, it is important for IAs to consider impacts across the life cycle of the embedded product provision. This requirement is onwards referred to as the application of life cycle thinking (LCT).

The introduction of a function-oriented assessment paradigm and the consideration of up- and downstream impacts are widely advocated as analytical benefits of applying LCT in IA (Bidstrup et al. 2015; Björklund 2012; Finnveden et al. 2003; Loiseau et al. 2012; Manuilova et al. 2009; Tukker 2000). A function-oriented assessment paradigm can add perspective to IA since proposed developments are then assessed in relation to their embedded product provision, as opposed to assessment merely in relation to the total scale of impacts. This may guide IA practitioners when assessing projects on industrial expansion since a potential lowering of the relative impact per produced unit (and thus also a lowering of impacts across the production system under analysis) may otherwise be concealed within an apparent increase in total, local impacts. Consideration of up- and downstream activities are important because large system-wide impact deviations may exist among development alternatives with seemingly similar on-site impacts.

The reasons for applying LCT in IA are extensively described in the literature. Yet, research on the topic has mostly had the form of theoretical advocacy (Finnveden and Moberg 2005; Loiseau et al. 2012; Tukker 2000) or demonstrations of LCA applications on deliberately chosen cases (Bidstrup et al. 2015; Björklund 2012; Cornejo et al. 2005; Manuilova et al. 2009; Židonienė and Kruopienė 2015). Little to no research appears to be rooted in observations from IA practice.

Through a review of 85 Danish IA reports, this article explores the extent to which LCT is applied in current IA practice. This is done by analysing the sample for a) appropriateness of LCT, b) application of LCT with LCA, and c) application of a function-oriented assessment paradigm, where up- and downstream impacts are considered, with alternative means of analysis. The aim of the research is to add perspective to the continuous discussion on the analytical gains assigned to LCA use in IA. The article opens with a description of the Danish IA context. Second, the method for collection and analysis of data is presented. The appropriateness and application of LCT in IA practice is then accounted for with respect to the various types and topics of IA. With an outset in the performance of the 85 IAs, the article concludes by discussing the opportunities and challenges assigned to further application of LCA.

$P_4$2 Method

$P_4$2.1 IA practice in Denmark

The application of LCT in IA was analysed through a case study on Danish practice. Denmark is a small country in Scandinavia and a member state of the European Union (EU). Hence, Danish IA practice is largely regulated through the national implementation of the EU directives on SEA and EIA (European Parliament 2001; 2014). In regard to SEA, the Danish Ministry of Environment and Food – DMEF – (2013c) requires evaluation of both strategic means for reaching development objectives (referred to as *'plan SEA'*) and compilations of multiple projects with the same development purpose (referred to as *'programme SEA'*). In regard to EIA, the ministry distinguishes between livestock expansion projects and other development projects due to the economic and political importance of the agricultural sector (DMEF 2009; 2014a). The requirements for EIA are the same, but different and more standardised procedures apply to these projects due to their similarity and frequency. These practices were therefore evaluated as separate branches of EIA, referred to as either *'classic EIA'* or *'agricultural EIA'*.

The Danish institutional system consists of 98 municipalities, five regions and the state. The municipalities prepare plans within most topics (for instance water, industry, waste) and manage the assigned permits for projects. The regions are county-like institutions which cover larger geographical areas. They prepare broad development strategies and manage the activities of the mining sector. The state prepares broad environmental strategies and manages permits for particular projects of national interest. In Denmark, EIA authority is assigned to the permit-granting institution, while SEA authority is assigned to the institution proposing the plan or programme under evaluation (DMEF 2013c; 2014a). Hence, most IAs are made at the municipal level, though some are prepared by the regions, the DMEF and the national road directorate (an institution that manages all state roads).

$P_4$2.2 Data collection

Data were sought through the information available in IA reports. These had to be downloaded manually from the website of single institutions since no central database for IAs exists in Denmark. A data set of 85 IA reports, adding up to approximately 9500 pages (excluding appendices), was retrieved in pdf format from the website of ten populous municipalities and the five regions. The data collection focused on obtaining a good mix of recently prepared IA reports. Hence, efforts were made to ensure that certain types and topics of IA or certain IA institutions were not overrepresented in the data set. Table $P_4$1 provides a rough overview of all the data, while a more detailed account is attached as appendix P_4A.

Of the 85 IA reports, 51 (29 EIAs and 22 SEAs) were prepared between 2012 and 2014 – the oldest is from 2007. The 2009-revision of the EIA directive estimates that approximately 130 Danish EIAs are produced annually, while the corresponding revision of the SEA directive found no available statistics (European Commission 2009a; 2009b). The 29 most recent EIAs thus represent roughly 8% of the national average. This is, however, only a qualified guess.

	Urban planning	Construction		Production		
		Infra-structure	Urban structures	Energy	Raw materials	Livestock
Plan SEA	13	0	0	0	1	0
Programme SEA	5	1	6	5	6	0
Classic EIA	0	13	9	14	1	0
Agricultural EIA	0	0	0	0	0	11

Table $P_4$1: An overview of the 85 analysed IA reports. See Appendix A1 for additional information on name, publishing year, page number and IA authority of each report.

In regard to IA types, the study included 14 plan SEAs, 23 programme SEAs, 37 classic EIAs and 11 agricultural EIAs – in total 37 SEAs and 48 EIAs. No universally acknowledged criterion exists for when to categorise developments as plans, programmes or projects, respectively. In practice, some plans could be characterised as programmes consisting of multiple projects, while certain programmes could be characterised as big projects. In addition, it was found that practitioners often name IA reports as both EIA and SEA due to peculiarities of Danish legislation, where projects must be approved through *'plan supplements'*. Hence, the categorisation of IA types often relied on professional judgement in regard to when a compilation of activities was broad and strategic enough to be considered a *'plan'* or detailed and local enough to be categorised as a *'project'*.

In regard to IA topics, the study included 18 IAs on urban planning, 29 IAs on construction and 38 IAs on production. The term '*urban planning*' was used for plans and programmes with holistic objectives, fulfilling either multiple functions (e.g. municipal development plans) or urban functions which fall into neither the category of *'construction'* nor *'production'* (e.g. waste water plans). Developments categorised as *'construction'* included infrastructure (13), such as roads, harbours and airports, and urban structures (15), such as housing, office buildings or hospitals. Developments categorized as *'production'* included energy (19), raw material mining (8) and livestock (11). Table $P_4$1 shows that plan SEAs primarily covered the topic urban planning, while the specialised agricultural EIAs covered all the livestock projects. The programme SEAs and classic EIAs had a more generic nature, spanning across multiple topics of proposed construction and production.

$P_4$2.3 A framework for analysis of LCT

The data analysis of this study explored a) the appropriateness of LCT in IA, b) the application of LCT through LCA and c) the extent to which IA practice applies LCT through different means of analysis than LCA. With a reference to the introduction, LCT was interpreted as the application of a function-oriented assessment paradigm by which up- and downstream impacts are considered. A generic framework consisting of seven questions (referred to as Q_1 to Q_7) was applied when analysing every IA report – see table $P_4$2. For clarity, all questions were structured in a short *'yes/no'*-format, but this does not mean that the research looked only for *'yes/no'*-answers. Indeed, the study also focused on how LCT is applied in IA.

Appropriateness of LCT	
Q_1	*Does the IA evaluate proposed supply of a single product?*
Application of LCT through LCA	
Q_2	*Is the IA supported by LCA results?*
Q_3	*If yes, is an LCA made within the assessment?*
Application of function-oriented paradigm	
Q_4	*Are impacts evaluated in relation to the proposed product provision?*
Q_5	*Are alternatives with similar product provision compared?*
Consideration of up- and downstream impacts	
Q_6	*Are impacts of inputs supporting the product provision assessed?*
Q_7	*Are impacts of inputs assigned to later life cycle stages assessed?*

Table $P_4$2: The framework applied for analysis of each IA report.

'Appropriateness' of LCT was in this study viewed from a strictly analytical perspective, evaluating whether such information would be of value for later decision support. LCT was assumed analytically appropriate for all IAs on proposed changes in supply of a single product (Q_1) since development alternatives within such IAs directly influence impacts of related product systems. A *'single product'* could (in an IA context) be: electricity, solid waste treatment, housing and mobility. Supply of a single product (such as 1 kWh of electricity) will often be established through an input of multiple sub-products (such as various fuels). The existence of multiple inputs was not viewed as changing the single product focus of such developments. Indeed, it can be argued that the need for sub-products to fulfil the function of a development to a large extent constitutes the rationale for applying a life cycle perspective. Question one thus served as a filter to determine whether any given IA was suitable for LCT.

In regard to the application of LCT through LCA, a distinction was made between IAs supported by external LCA results (Q_2) and IAs applying LCA as part of the assessment methodology (Q_3), since these approaches arguably represent two very different degrees of LCA application. Following the same line of thought, a distinction was made between IAs applying a function-oriented assessment paradigm explicitly (Q_4), by relating impacts of the proposed development to the scale of product provision, and implicitly (Q_5), by evaluating the impacts of development alternatives which fulfil a similar function. Consideration of up- and downstream impacts was framed as an evaluation of the inputs supporting the product provision of a development either prior to (Q_6) or subsequent to (Q_7) the life cycle stage under analysis. Most of the IAs did, in practice, concern the construction phase of new infrastructure or production facilities, and upstream impacts were therefore generally assigned to resource consumption while downstream impacts were assigned to the impacts imposed during the use or demolition phase.

$P_4$2.4 Data analysis strategy

Because the study aimed at exploring the prevalence of LCT in IA practice, it was deemed necessary to apply a large data set. This, however, meant that it was not possible to scrutinise every single IA report.

Thus the following data analysis strategy was applied:

1. Study the non-technical summary and conclusion for elements of LCT.
2. Scan of the remaining report using the 20 search words from table $P_4$3.
3. Use table $P_4$2 to analyse the passages of the IA report, which were found relevant through the two prior steps.

Phrases in the non-technical summary (or the usage of a search word) were thus used to locate text passages which could help answer the questions of table $P_4$2. Passages found on electricity use over the lifespan of windmills in a proposed energy project would, for instance, direct attention towards the evaluation of electricity production in that specific IA. Though a data set of 85 IA reports (covering thousands of pages) and the application of a search word strategy could be interpreted as a sole quantitative approach, the research design ultimately relied on qualitative analysis of specific text passages. The aim was to gain an understanding of the extent of LCT in practice – not to count the use of search words.

		Application of LCA	**Function-oriented paradigm**	**Up- and downstream impacts**
LCA terminology	life cycle	x	x	x
	LCA	x	x	x
	footprint	x	x	x
	function		x	
	market		x	
	demand		x	
	supply chain			x
	upstream			x
	downstream			x
	product system			x
IA terminology	emission	x		
	relative		x	
	impact per		x	
	stress per		x	
	alternative		x	
	resource			x
	material			x
	use phase			x
	indirect			x
	secondary			x

Table $P_4$3: The search words applied for pin-pointing passages with LCT in the IA reports. The search words have been translated from Danish to English.

Twenty search words were selected for the purpose of locating passages with LCT in the IA reports. The first ten were adopted from LCA terminology provided in ISO standards (ISO 2006a; 2006b) and the ILCD handbook (European Commission 2010). They helped pin-point *'life cycle'* assessment or thinking, *'footprint'* considerations, use of a *'function'*-oriented approach (covering also *'functional unit'*), thoughts on the *'demand'* and *'market'* for the provided product, impacts across *'supply chains'* and the relation to *'production systems'*. However, it became clear as the analysis progressed that these terms are rarely applied in IA practice, and ten additional IA-based search words were thus added to the list. These were largely based on the wording of the concrete IA reports and developed iteratively as more and more passages with LCT were identified. LCA application was found often under the assessment of *'emissions'*, while a function-oriented assessment paradigm expressed as impacts *'relative'* to production was searched for. Question five from table $P_4$2 was tried answered by reading all passages on *'alternatives'*, while considerations of up- and downstream inputs were searched for in regard to *'resources'*, *'materials'* and the subsequent *'use phase'*. Lastly, considerations on up- and downstream impacts were also searched for as *'indirect'* and *'secondary'* effects since these terms are referred to in the EIA and SEA directives. Ideally, even more search words could have been added.

$P_4$3 Results

$P_4$3.1 The appropriateness of LCT

The study found it widely appropriate to apply LCT in Danish IA practice, with as many as 87% of all IAs evaluating supply of a single product (Q1). It was found that LCT can be applied across most types (figure $P_4$1) and topics (figure $P_4$2) of IA. All IAs on construction and production were found by the author to be analytically appropriate for LCT. Yet, one might argue that this result is not all surprising since proposals for new infrastructure and urban structures are often made with a function in mind (for which there is a demand). Likewise, it is rather logical that IAs on proposed production facilities fundamentally evaluate the conditions under which increased product provision can be established.

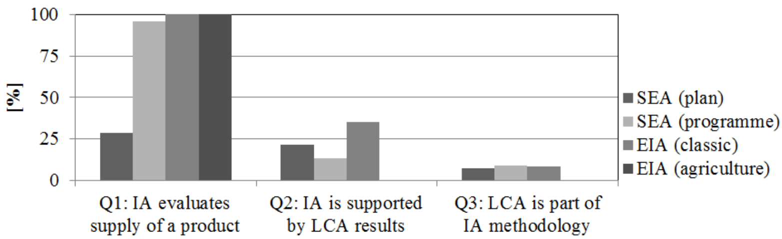

Figure $P_4$1: The applicability and application of LCT through LCA among IA types.

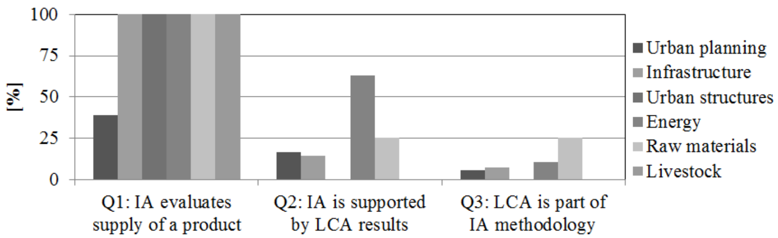

Figure $P_4$2: The applicability and application of LCT through LCA among IA topics.

Plan SEAs were found to be somewhat less appropriate for LCT. This is mainly because several of these were categorised as urban planning (see table $P_4$1), of which many were found to have a focus different from that of product provision (see figure $P_4$2). SEAs deemed inappropriate for LCT were typically municipal development plans, which for instance sum up local initiatives for increasing public health, business activities and overall quality of life. Two of these IAs were, however, found to refer to external LCA studies as a means of legitimising political prioritisations. Hence, LCT can be (and is being) drawn upon regardless of the analytical appropriateness of the IA topic.

$P_4$3.2 The application of LCT through LCA

Despite the widespread appropriateness of LCT in Danish IA practice, it was found that LCA is rarely used to provide such perspective. Among the 85 IA reports, only 22% was supported by LCA results (Q_2) while as little as 7% applied LCA as part of the IA methodology (Q_3). Instead of actively applying LCA as an *'appropriate tool'* in the impact analysis, it appears that IA practitioners primarily draw upon LCA results from external sources.

The classic EIAs had the highest prevalence of LCA application (35%) while none of the 11 agricultural EIAs took use of the tool (figure $P_4$1). There are great deviations across the various topics of IA (figure $P_4$2). As many as 63% of energy production IAs were found to be supported by LCA, while LCA support ranged between 14% and 25% for IAs on urban planning, construction of infrastructure and production of raw materials. No LCA support was found among the IAs on construction of urban structures or production of livestock.

It was found that LCA application rarely expresses LCT across the various impact categories of modern LCA tools. Certain projects on wind turbine erection did account for the cumulative energy demand throughout the life cycle of a wind turbine as a means of legitimising their purpose. However, LCA was used almost exclusively for calculating global warming impacts.

Lastly, it was found that the technical terminologies of LCA and IA differ completely – even when LCA results are drawn upon. Only 18% of the IAs included search words from LCA terminology, while as little as 6% included terms other than the search word *'life cycle'*. Within an IA context, *'footprints'* generally refer to traces of wildlife, while the terms *'upstream'* and *'downstream'* most often refer to hydrological or chemical impacts on aquatic systems.

$P_4$3.3 Application of a function-oriented paradigm without LCA

When studying the IAs that did not apply LCA, only 18% explicitly assess proposed developments in relation to the imbedded product provision (Q_4). This, however, does not mean that IA practitioners are unaware of the developments' function in product systems. In fact, all plan SEAs, 63% of the programme SEAs and 71% of the classic EIAs were found to implicitly evaluate the cleanest provision of the embedded product through comparison of multiple alternatives that serve the same function (Q_5), see figure $P_4$3. When discriminating between various topics of IA, it was found that IAs on energy production perform the best, see figure $P_4$4

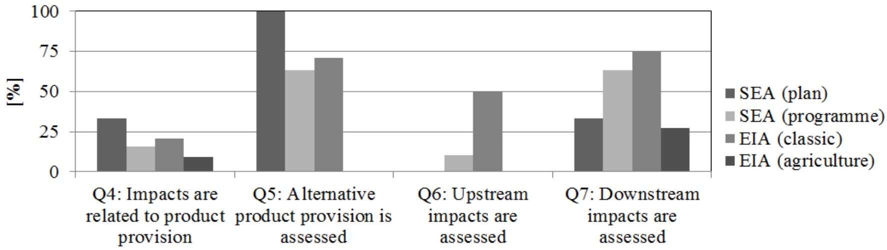

Figure P$_4$3: The performance across types of IA in applying a function-oriented assessment paradigm and considering up- and downstream impacts (without LCA).

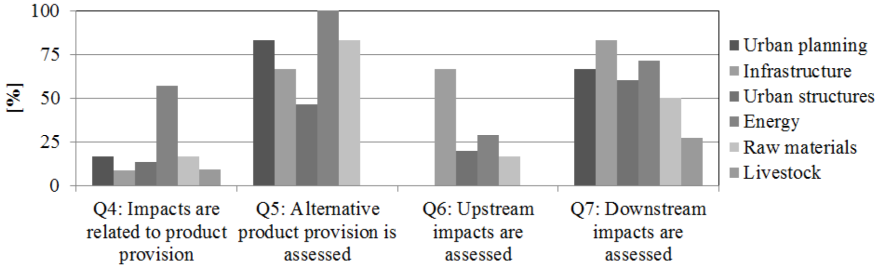

Figure P$_4$4: The performance across topics of IA in applying a function-oriented assessment paradigm and considering up- and downstream impacts (without LCA).

Of the IAs which did not apply LCA, 61% relate impacts to the imbedded function of the development either explicitly or implicitly. The remaining 39% do not consider the embedded product provision. These IAs stick to conclusions on whether the proposed project is acceptable on the proposed location, and by doing so they turn a blind eye to the potential impacts of providing the embedded function of the proposed development elsewhere or in a different way. This appears to be an issue within agricultural EIAs and to some extent within IAs on urban structures.

Of the 61% that relate impacts to the embedded function without LCA, 77% were found to do so in a qualitative way. This was most commonly done implicitly by comparing alternative ways of fulfilling the proposed function, but some practitioners were more explicit in their qualitative assessment. One example of this is an EIA regarding a proposed expansion of a cement factory: *"Jointly, the expansion will lead to both greater use of resources and production of waste products. However, the resource use will, if considered in regard to each produced ton of cement, be lowered due to application of best available technology"*.

P$_4$3.4 Consideration of up-and downstream impacts without LCA

Among the IAs which did not apply LCA, the study found that only 25% assess the impacts of inputs supporting the product provision upstream in production systems (Q$_6$). Such considerations were most prevalent in classic EIAs (see figure P$_4$3) of infrastructural projects (see figure P$_4$4). Upstream impacts are typically considered when the use of natural resources is found significant by the IA authorities. In such cases, efforts were directed towards minimising the use of virgin resources by accounting for how recycled materials could be used.

Regarding downstream activities, however, the IAs performed better. Of the IAs which did not apply LCA, 60% were found to evaluate downstream impacts by different means of analysis (Q_7). IAs on urban planning did for instance project how strategic efforts could lower traffic loads and thus also air pollution, while construction projects for urban structures quite commonly assess energy use in the subsequent use phase. Figure $P_4$3 shows how downstream impacts are well-accounted for across the types of IA. Yet, the agricultural IAs did, once again, prove to perform worse than the remaining types and topics of IA.

Of the IAs which accounted for up- or downstream impacts without LCA, 58% were found to do so in a qualitative way. Upstream impacts were often assessed qualitatively by discussing the impacts of the chosen type of construction materials rather than the scale of material use *per se*. Downstream impacts were addressed qualitatively by arguing that new office buildings, for instance, would lower future energy requirements when compared to the current use of old and less energy efficient ones. Up- and downstream impacts were in this study evaluated as assigned to the inputs supporting the life cycle stages of the provided product. In this regard, it is noticeable that such inputs generally are considered impact categories in their own right. The use of raw materials or electricity is for instance often used as metrics for comparing alternatives. Few IAs express such inputs by generic metrics for environmental impact.

$P_4$4 Discussion

$P_4$4.1 The representativeness of the case study

It can rightfully be questioned whether Danish practice is representative of IA practice as a whole. Denmark does, with its population of 5.6 million and its size of 43,000 km^2 (Statistics Denmark 2014), only account for a small fraction of the IAs produced worldwide. Moreover, Denmark is among the 10 richest countries in the world (World Bank 2015a) and has strict IA legislation, which is enforced through its obligations to the European Union (European Parliament 2001; 2014). Hence, Danish IA practice may be more representative for practices in affluent, well-regulated countries than developing countries, for instance.

One may further question how representative the data set then is for Danish practice as a whole. The IA reports were only retrieved from 10 of the 98 municipalities, and it is estimated that the 85 reports represent only a fraction of the IAs prepared annually. Hence, there may be practices in the remaining IA institutions, or the many excluded IA reports, which have not been highlighted by this present study.

$P_4$4.2 Research design and validity of results

The study applied a data analysis strategy where non-technical summaries in combination with word searches were applied to pin-point areas in IA reports, which were to be studied in depth – see section $P_4$2.4. This strategy made it possible to focus on a large sample of IA reports. However, it raises issues of validity which need to be addressed.

First, it can be criticised that a study on the application of analytical *'thinking'* focuses on written documentation rather than the thoughts and perceptions of IA practitioners. It is assumed that the IA reports truthfully represent the collective thinking process of both the screening and scoping process, and this is a source of error since the analysis of IA processes may differ significantly from what is documented later on – see Bidstrup and Hansen (2014). It must here be stressed that the term *'life cycle thinking'* refers to the analytical consideration of impacts occurring across supply chains when providing a product, rather than to the cognitive capabilities or activities of practitioners. LCT is nothing but an analytical perspective, which can be applied or omitted during assessment.

Second, certain LCA-like methodology, function-oriented assessment paradigms or considerations of up- and downstream impacts might have been missed if these elements have not been formulated clearly in the non-technical summary and conclusion or generated a *'hit'* via the search words of table $P_4$3. The amount of both IA reports and search words could have been substantially extended if the author had used digital software for *'text mining'*, by which the occurrence of words and sentences can be registered. Yet, application of such software and a larger data set would arguably have resulted in a fundamentally different research design where specific wording would be in focus rather than the context in which the wording is used. With the current research design, validity is only impacted if additional words would have referred to passages in the IA documents, which the current 20 words (in combination with the non-technical summary and the conclusion) have not directed attention towards. However, this is a possibility.

The study deliberately balances between qualitative and quantitative research since it aims at describing both the practice and prevalence of LCT in IA. From a qualitative perspective, validity could have been increased by focusing on fewer IAs and then considering each specific IA process (including the perceptions and actions of practitioners). From a quantitative perspective, it can be argued that a larger sample and application of more search words could have added robustness and increased validity to the study. Nevertheless, the 85 reports constitute a significantly larger sample than those provided in previous studies on the topic (primarily single case studies). In addition, one can question whether LCT truly has been applied in IAs where no references to such can be found in the non-technical summary, in the conclusion or by the 20 applied search words.

$P_4$4.3 Indications on the geographical extent of LCT in IA practice

With an outset in the provided argumentation for introducing LCT in IA practice, the analytical framework of table $P_4$2 was tailored to answer if current IA practice considers impacts of up- and downstream inputs. It was found that IAs are generally better at assessing downstream impacts than the upstream ones – see section $P_4$3.4. As the analysis progressed, however, an interesting relationship was observed between the impacts considered and the geographical extent of assessment. In an IA context, upstream impacts are typically assigned to distant and supporting activities (for instance the production of construction materials), while downstream impacts of the use and end-of-life phase typically occur onsite. Hence, the study indicates that

IA practitioners may not have a preference for downstream impacts *per se*. They may simply prioritise impacts occurring on-site, and within the spatial boundaries of their institution, to those assigned distant activities which they cannot regulate. This is an analytical deficiency of IA which has been touched upon by other authors. Bidstrup et al. (2015) found that indirect, distant impacts had the greatest variations in local-oriented IA practice, while Loiseau et al. (2013) argue that indirect impacts of a proposed plan can dwarf the direct, local ones.

$P_4$4.4 Analytical gains of greater LCA application

It has been widely argued that application of LCA in IA can widen the assessment perspective by facilitating the consideration of the up- and downstream impacts assigned to the function of proposed developments. Such *'widening'* implies that a difference exists between the perspective applied with and without LCA. Yet, the existence of this *'difference'* has until now primarily been based on theoretical reflections (Finnveden and Moberg, 2005; Loiseau et al., 2012; Tukker, 2000). The perspective added to IA by LCA has been demonstrated in various case studies (Björklund, 2012; Cornejo et al., 2005; Manuilova et al., 2009; Židonienė and Kruopienė, 2015), but only the work of Bidstrup et al. (2015) appear to briefly compare this perspective to one without LCA. This present article allows further elaboration on how the LCT of current IA practice compares to the LCA approach to LCT. It was found that further application of LCA can lead to four analytical gains:

1. A more explicit focus on product provision
2. A more rigorous evaluation of life cycle impacts
3. An impact-oriented methodology for evaluating resource use
4. A framework for quantitative comparison of alternatives

LCA application could enable the majority of IAs to be more explicit about how a proposed local development represents a component in a larger production system, with impacts that are equally important to mitigate. LCA application could facilitate an understanding of proposed developments as societal responds to market demands. Within agricultural IAs, this perspective is currently non-existent.

LCA application would moreover entail the use of computer software for LCT, which makes it possible to evaluate the impacts of activities across all life cycle stages of a product. This includes not only an evaluation of the primary inputs needed for a specific proposed development (*'foreground system'*) but also an evaluation of the sub-inputs needed to support these primary inputs (*'background system'*) – see the European Commission (2010). Current IA practice rarely assesses inputs extending beyond those occurring onsite in the foreground system.

Computerised LCA tools model the relationships between inputs and outputs of the technosphere and the assigned stress on the biosphere in regard to broad environmental metrics. This contrasts the practices of Danish IA, where resource inputs are often treated as environmental indicators *per se*. Hence, application of LCA could facilitate a more impact-oriented methodology where e.g. climate change impacts of material use and energy use in a construction project can be compared. Such perspective is largely lacking in current IA practice, where climate impacts are rarely attributed to other than combustion processes in the foreground system.

Lastly, application of LCA could provide a framework for quantifying issues which are primarily qualitative in today's practice. It could facilitate practitioners in expressing the scale of impact deviation among proposed alternatives and measures for impact mitigation.

$P_4$4.5 When can and should LCA be applied for LCT?

Frameworks for application of LCA have been proposed for both EIA (Židonienė and Kruopienė, 2015) and SEA (Bidstrup et al. 2015). It is in this present study argued that application of LCT is analytically appropriate in IAs concerning the supply of a single product. Yet, the study of Loiseau et al. (2013) demonstrates that LCA can be applied in a multipurpose development context too. No universal rules thus seem to apply as to when LCA. Data needs could, however, be a problem in plan SEAs where much uncertainty exists and where no detailed account of inputs is at hand – as argued by Björklund (2012).

Yet, the fact that LCA can be applied for improved LCT in IA does not mean that the tool should be applied. This choice ultimately relates to the *'procedural appropriateness'* of the tool application. IA practitioners are during the IA scoping process faced with the task of deciding which impacts that can be expected significant. This decision then guides them in selecting whether LCA is an appropriate analytical tool for environmental evaluation. Practitioners should always strive towards reaching the objective of an IA in the most cost-effective way. Yet, LCA application may increase both costs and time requirements for the assessment (Fischer 2007; Tukker 2000), and use of the tool can thus not be legitimised in all IA contexts (Björklund, 2012). The procedural IA principles of *'significance'* and *'cost-effectiveness'* are thus built-in mechanisms that in practice limit the usage of LCA.

IAIA (1999) stresses that good IAs should be *'focused'*. LCA should, of course, not be applied in IAs where the gain is minimal or where tool application represents an unreasonable increase in costs. Yet, the perceptions of '*gain*' and *'unreasonable costs'* are often based on a discretionary and value-based judgement, which is made in a local-political context and which relies on the experience of each practitioner (Lawrence 2007; Weston 2011; Wood and Becker 2005). Hence, LCA will only be applied in IA when authorities, operating according to their local agenda, deem life cycle impacts significant enough to legitimise an increased use of IA resources. This has been discussed by both Björklund (2012) and Tukker (2000).

Thus it can be argued that the analytical deficiencies of Danish IA practice in applying LCT may be signs of lacking perception of significance, represented through a deliberate prioritisation of resources, rather than a sign of bad IA quality *per se*. Though widely absent in the analysed IA reports, LCA support was found to be common within the topic of energy (see figure $P_4$2), with all wind turbine erection projects drawing on LCA results. Hence, it appears that life cycle impacts are primarily scoped significant within energy projects in current Danish practice. The absent considerations of upstream (and largely distant) impacts were most common within classic EIAs of infrastructural projects. This is interesting since exactly these IAs generally are larger and more thorough than the remaining ones. One indicator of this is the written extent of the IA reports, which within this type and topic of IA was found to be twice as big as the average IA and more than three times bigger than the poorly performing agricultural EIAs. Hence, the study indicates that assessment of upstream, distant impacts is primarily legitimised for larger and possibly more expensive IAs in current practice. Further research could explore the prevalence and rationale of such discretionary judgements among IA practitioners. Little has still been said about when LCA should be applied in IA.

$P_4$5 Conclusion

Various authors have advocated that LCT is needed in IA and that application of LCA is an appropriate means of achieving such analytical perspective. However, little research has based such advocacy on observations from practice, and the gains of LCA application in IA have thus not been legitimised. This article explored the appropriateness and application of LCT in Danish IA practice.

The study found that LCT is widely appropriate in IA – spanning across the various types and topics of assessment. Yet, LCA is rarely applied as a tool for LCT outside the context of renewable energy projects. When applied, LCA is often used for justifying local impacts by highlighting the climate and energy benefits of developments. In regard to wind turbine erection projects, for instance, LCA data is used as a counterargument for impacts such as shadows, noise and disturbance of the landscape. Across the various topics and types of IA, LCA appears to be a tool used primarily for evaluation of climate impacts. It is often drawn upon from external sources in a *'carbon footprint'*-like format.

Without LCA, the IAs were found to apply LCT to some extent. They rarely evaluate impacts explicitly in relation to the embedded function of the development, but they do seem to apply a function-oriented paradigm implicitly by considering alternatives with a similar product provision. Multiple IAs were found to consider on-site, downstream impacts, as well as EIAs of large infrastructural projects often consider upstream impacts assigned to resource use. However, IA practice as a whole was found to perform poor in considering the impacts of upstream activities extending beyond the proximity of the development site. It was moreover noticed that resource and energy inputs are often assessed as impacts *per se*. They are rarely assigned to emission outputs and expressed by generic impact indicators.

Classic EIAs proved to perform best. They have the highest prevalence of LCT through both LCA and other means of analysis. SEAs on plans and programmes performed second best. They often consider both function and downstream impacts. Agricultural EIAs were found to perform the worst, at large failing to apply LCT.

With an outset in the performance of current IA practice, it was possible to elaborate on the analytical gains of LCA application. LCA can, indeed, widen the perspective of IA practice. However, this is not through the introduction of LCT *per se* – such perspective already exists in thorough IAs. Instead, it is by helping IAs to be more explicit on their embedded function and by facilitating a more rigorous, impact-oriented and quantitative assessment practice in regard to resource use.

Appendix P_4A can be found online at http://dx.doi.org/10.1016/j.eiar.2015.05.003.

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LIFE CYCLE ASSESSMENT IN SPATIAL PLANNING: A PROCEDURE FOR ADDRESSING SYSTEMIC IMPACTS

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Abstract

Spatial planning establishes conditions for societal patterns of production and consumption. However, the assigned Strategic Environmental Assessments (SEA) tend to have a too narrow focus. In particular, there is a need for applying a system perspective in SEA, extending assessment beyond the spatial boundaries of a plan to further focus on global, indirect and cumulative impacts. These impacts are referred to as *'systemic impacts'*. This study proposes a Life Cycle Assessment (LCA) procedure which can be adopted in SEAs of various types of planning. The procedure represents a first step towards operationalizing LCA in SEA by adjusting LCA methodology to focus on the ways planners and planning processes can influence the environmental impacts of interconnected activities. The proposed procedure was tested on a case study of Danish extraction planning, and it was found to generate new knowledge for decision support. The procedure enabled identification of key systemic impacts, as well as it enabled formulation of recommendations for how to address these impacts in planning processes. On a more general level, this article demonstrates an application of LCA which until now has received little attention, and it highlights the role of spatial planners in facilitating cleaner production.

$P_5$1 Introduction

Patterns of traffic, industry, production and resource supply form the backbone of local and regional development. However, they may also generate unwanted impacts, and environmental assessment has thus for decades been an integrated part of preparing the spatial plans regulating these activities. In particular, the Strategic Environmental Assessment (SEA) is highlighted as a tool which can facilitate sustainable development (Fischer 2007; Partidário 2012; Therivel 2010) by introducing sustainability in planning processes and generating transparency about alternatives (Stinchcombe and Gibson 2001). Because SEA is performed at the plan level (or higher), it allows influence on the combination and the characteristics of project proposals from a wider perspective. In doing that, it enables a consideration of development alternatives where cumulative and synergistic impacts can be considered (Johnson et al. 2011; Therivel 2010). This is an important characteristic of SEA since a strategic viewpoint can provide an alternative perspective on the rationale of decisions. A projected wastewater discharge from a proposed project may e.g. seem negligible when viewed upon independently, while it could represent a contribution to a cumulative wastewater overload on a local recipient when viewed upon strategically. A new resource intensive industry may reversely seem polluting locally, while it may represent an opportunity for industrial ecology and cleaner production from a strategic viewpoint.

Yet, experiences from various international studies conclude that current SEA practice has major shortcomings in this regard. Studies from both Europe (Bragagnolo et al., 2012; Stoeglehner, 2010; Söderman and Kallio, 2009), North America (Noble, 2004) and Asia (Zhou and Sheate, 2011) all conclude that SEAs tend to have a too narrow scope and that they do not address cumulative impacts. Tetlow and Hanusch (2012) emphasise that such SEA improvement is generally needed. Yet, such improvement entails assessing impacts which extend beyond the geographical boarders of the region in scope and beyond the timeframe of the plan in scope, changing the assessment paradigm from a plan/project focus towards a system focus (Gunn and Noble, 2011).

Thus SEA must, in addition to assessment of direct and onsite impacts, also focus on how a plan's embedded activities influence and interact in systems. This, however, is difficult since impacts exist on diverse scales of space (local, regional, national and global) and time (short-term vs long-term), often appearing indirectly (an action sparked elsewhere). With an outset in the assessment scope currently lacking in SEA practice, this article proposes the term *'systemic impacts'* to cover the global and long-term impacts (induced both directly and indirectly) of proposed plan activities.

Introduction of Life Cycle Assessment (LCA) in SEA practice has since the late 1990's been advocated as a potential means for addressing such systemic impacts in spatial planning (Owens 1997; Tukker 2000). LCA is the study of impacts assigned to societal products or services, and it is defined as a "*compilation and evaluation of the inputs, outputs and the potential environmental impacts of a product system throughout its life cycle*" (ISO 2006a). The ability to predict global, long-term and indirect impacts across the life cycle of products and services makes LCA an analytical tool which can complement SEA (Björklund 2012; Finnveden and Moberg 2005; Fischer 2007; Jeswani et al. 2010; Loiseau et al. 2012; Manuilova et al. 2009). Yet, the communities of scientists and practitioners working with respectively SEA and LCA remain rather segregated despite their common focus on supporting environmentally sound decisions. A standard is currently in development on how to apply LCA to policy proposals (WRI 2014), but there exist little consensus on how to apply the tool in planning.

Many LCA studies have dealt with topics typically covered by SEA, such as water management (Lemos et al. 2013; Niero et al. 2014), forest management (Berg and Lindholm 2005; Valente et al. 2011) and waste management (Prapaspongsa et al. 2010; Quek and Balasubramanian 2014). Yet, such studies are typically limited to concluding on a preferred technical option, after which it is assumed that someone somewhere will be supported by these LCA results when making a decision. Some authors have described how LCA can support decisions in regard to a proposed plan (Björklund 2012; Lundie et al. 2004; Nilsson et al. 2005), while recent work propose *"territorial LCA"* for baseline analysis of the functions on a given territory (Loiseau et al., 2013). However, little LCA research has until now focussed on what happens in between baseline studies and finished plan proposals – the process of planning. This research builds on the idea that LCA knowledge must add value within planning processes in order to be influential in practice. SEA is the established tool through which such support can be provided.

Integrating LCA as an analytical tool in SEA is ultimately an act of operationalizing LCA in spatial planning processes. However, this requires research on how to adapt LCA methodology to fit SEA and spatial planning processes as well as research on how to use such LCA application to support better decision-making in practice. The research of this article primarily focuses on adapting LCA methodology by proposing and testing an LCA procedure which can be adopted in the analytical phase of SEA. The research is interdisciplinary and it represents a first step towards bridging the research communities working with environmental analysis through respectively LCA and SEA. The article opens with a brief description of how LCA could fit within the framework of SEA, after which the proposed LCA procedure is presented. The procedure is then tested on a case study of Danish extraction planning, which is taken as a starting point for discussion and reflection on the performance and limitations of the procedure. The article concludes by summing up the experiences gained from the test.

$P_5$2 The proposed procedure for LCA in SEA

$P_5$2.1 Ensuring better planning with SEA

SEA is required for plans likely to generate substantial environmental impacts in the EU (European Parliament 2001), USA (US EPA 2000), Australia, China, Korea (Fischer 2007), South Africa and several other countries (OECD 2006). Many definitions of SEA exist; however, Fischer (2007) describes it as a *"systematic decision support process, aiming to ensure that environmental and possibly other sustainability aspects are considered"* when for instance preparing spatial plans.

SEA processes can vary, but frameworks typically include:

a) screening for the necessity of SEA,
b) scoping of the issues that need to be addressed,
c) assessment of planning alternatives, and
d) environmental reporting.

The basic purpose of planning is to determine a suitable course of action (a plan) for reaching desirable development objectives. A planning process will typically yield a plan proposal (based on initial prioritisations), which then subsequently is adjusted and approved in cooperation with key stakeholders. Decision-making theory and decision processes are broad research topics which go beyond the scope of this present article. However, it is widely recognised that integration of SEA in the planning process (as opposed to using SEA solely for plan approval) is a key element in producing effective and influential decision support (Partidário, 2012; Therivel, 2010; van Doren et al., 2013). When integrated, SEA generates knowledge on how to avoid, minimise or compensate environmental burdens while planners are considering the alternatives way of reaching planning objectives (Fischer, 2007).

$P_5$2.2 Fitting LCA in SEA

As recommended by Fischer (2007), this study proposes to introduce LCA as a *"technique"* in the assessment of planning alternatives within the SEA framework (bullet c in section $P_5$2.1). This, however, can be challenging due to the very same differences which make the tools complementary.

SEAs typically focus on alternative ways of reaching the development objectives of the plan in scope, considering alternative configurations of activities and/or applications of technology within the spatial boundaries of a region in scope. Yet, this focus on the development of and the impacts on a specific region contrasts with the product-oriented paradigm of LCA, which focuses on the total impacts assigned to a Functional Unit (FU). Hence, the merge of these two tools for environmental analysis depends on the extent to which the spatially delimited development objectives of SEA influence a quantifiable flow of products or services which can be expressed as an FU and modelled with LCA. In essence, LCA can only add value in SEAs which change the demand for products and services or which influence the ways by which these are supplied.

The proposed procedure focuses therefore on how planning choices can influence production and service systems. Planning is, quite simply, established as a model variable which through regulation of activities within a region generates differences in the demand for, or the supply of, products and services. The proposed procedure should be perceived as a supplementary analysis, which can be applied in SEAs where the scoping phase (bullet b in section $P_5$2.1) reveals a concern for systemic impacts assigned to influence on product systems. The procedure should be applied in the early stages of assessing alternatives, allowing SEA practitioners to use the recommendations when developing the plan.

$P_5$2.3 Proposed procedure

Step 1: Identification and quantification of planning variables
Step 2: development of an LCA model that includes planning variables
Step 3: Formulation of planning scenarios
Step 4: Analysis of life cycle impacts
Step 5: Formulation of planning recommendations

Step 1 identifies and quantifies the pathways by which planners can influence production and service systems. These pathways are central to the LCA model and they are named *'planning variables'*. Planning variables can be identified through a variety of methods, e.g. interviews with decision-makers, analysis of planning documents or available statistics.

In step 2, an LCA model and a suitable FU are developed, depicting the relationship between planning and the production or service activities influenced hereby. The planning variables are then formulated as foreground parameters in the model. It is recommended to construct the LCA model by consequential modelling principles, characterised by a focus on the causal links by which systems interact. Consequential modelling accounts for how product systems are *"expected to change as a consequence of a change in demand for the functional unit"* (UNEP/SETAC 2011). The choice of analysing the effects of planning with consequential model is supported by Finnveden et al. (2009) who argue that consequential modelling must be applied in all *"decision LCAs"*, and Björklund (2012) who likewise uses consequential modelling for integration of LCA in SEA. Causal links in the foreground system (direct consequence of planning) can be established through dialog with decision-makers, planners or experts in market mechanisms, while causal links of the background system (results of changes in demand for products or services) are embedded in modern LCA inventory databases.

In step 3, planning scenarios are formulated. It is recommended that application of the procedure includes a baseline scenario which can act as a point of reference to the remaining scenarios. The remaining scenarios should then represent possible planning prioritisations within the quantitative span of the planning variables.

In step 4, the systemic impacts are calculated. Although optional in LCA methodology, a weighting step can be applied to the baseline scenario in order to identify the most relevant impact categories. These impact categories can then be analysed in the remaining scenarios.

Step 5 entails a formulation of recommendations. Such recommendations should be based on the extent to which certain planning variables hold the possibility to generate large systemic impacts, referred to as *'sensitive'* planning variables.

$P_5$3 Method

$P_5$3.1 The case study context

The proposed procedure was tested on a case study about aggregate extraction planning in Denmark. The term *'aggregate'* comprises sand, stone, gravel, chalk, clay and other mineral products used by the construction sector. These resources are widely used, and the sector accounts for approximately 56% of the total resource extraction mass flow within the European Union (FORWAST 2011).

Danish extraction planning is managed through the national Act on Raw Materials, which states that supply must be met through a weighting of societal sustainability interests (DMEF 2013b). Planners zone suitable sites in regional extraction plans, after which private landowners and entrepreneurs can apply for extraction permits within these zones. Mandatory SEAs of extraction plans introduce sustainability principles and management recommendations on the strategic level of resource planning, while environmental screening and project-specific assessments are taken in use prior to proposing new zones or granting extraction permits. Despite this integrated focus on sustainability, Bidstrup and Hansen (2014) report that current SEA practice shows limitations. Most SEAs do not assess systemic impacts, and strategic effort appears more directed towards the environmental screening of individual zone proposals (Bidstrup and Hansen 2014). The assigned documents for public participation and communication with planners further reveal that zone proposals in particular are adjusted to minimise the impacts on local communities. For these reasons, Danish extraction planning is considered a representative case for lacking assessment of systemic impacts beyond the spatial boundaries of a plan.

$P_5$3.2 Step 1: identification and quantification of planning variables

Application of the procedure had the purpose of analysing the extent to which Danish aggregate planners can influence the systemic impacts assigned to subsequent aggregate production. Both qualitative and quantitative data were collected. Qualitative data collection included: field visits to extraction sites; written and oral communication with representatives from the industry, NGOs, the Danish Ministry for Environment and Food, key-planners and key-municipal administrative staff; and document reviews of all five active aggregate plans, the assigned SEA reports and selected extraction permits. The quantitative data collection included analysis of all 313 extraction zones mapped and described in the 2012 aggregate plans (Capital Region, 2013; Central Denmark Region, 2012; North Denmark Region, 2012; Southern Denmark Region, 2012; Zealand Region, 2012), as well as it included an analysis of all 44 extraction permits granted after 2007 by four Danish municipalities (Kalundborg, Silkeborg, Sorø and Roskilde) which are among the most extracting in the country. Four planning variables were identified: 1) transport, 2) extraction intensity, 3) resource thickness, and 4) site restoration.

Aggregate transport patterns are largely driven by the price of the products since entrepreneurs optimize their costs by choosing the cheapest products which fulfil quality requirements. Yet, the price of aggregate products depends heavily on transport distance due to their low value-weight ratio. Planners can regulate the location, size and overall supply structure of aggregate zones, and strategic aggregate planning can in this way facilitate certain transport patterns by establishing e.g. centralised or decentralized supply schemes. The latest data shows transport distances varying between 30 km and 44 km across the five Danish regions (Hejlesen and Larsen 2007) with a geometrical average of 35.5 km.

Secondly, planning can influence the extraction intensity of resources through permit requirements such as the maximum yearly extracted quantity. Planning can thirdly influence resource thickness and other quality indicators of future extraction sites through zoning. What planners influence is obviously not the geology itself but the exploitation of the geology by e.g. avoiding to zone specific sites. The resource thickness across the 313 extraction zones and the permitted extraction intensity of the 44 extraction permits are reported in table $P_5$1. Standard deviations above 70% indicate that these planning variable differ substantially between extraction sites.

Resource thickness, [m]			**Permitted extraction intensity,** [$m^3/(m^2 \cdot year)$]		
Average	Thinnest 20%	Thickest 80%	Average	Most intensive municipality	Least intensive Municipality
3.6 ± 80%	< 2,3	> 6,9	2.0 ± 76%	3.4	1.2

Table $P_5$1: The resource thickness of the 313 zoned resource deposits in the 2012 aggregates plans and the permitted extraction intensity of the 44 extraction permits studied.

Lastly, planning changes land use during extraction as well as it can influence future land use through site restoration plans (establishing e.g. nature, recreational areas, or agriculture). Production of aggregates and the subsequent site restoration result in land use on a location which prior to this activity had a different purpose. 80% of the 44 extraction permits were found to establish aggregate production on former farmland. Of these, 75% will be restored as nature or recreational areas subsequently, while the remaining 25% will be restored as extensive agriculture with restrictions on application of fertilizer and pesticides. Changing the land use of productive farm land and occupying it for another purpose can be problematic from a consequential modelling perspective since this action will generate land use changes elsewhere in order to restore prior production capacity. Such impacts are commonly referred to as indirect land use changes (iLUC).

$P_5$3.3 Step 2: Development of an LCA model that includes planning variables

The LCA model was structured as a cradle-to-gate study in order to explore the extent to which planning can influence the impacts generated by industry from the initial stages of land and resource acquisition (cradle) until the commercial aggregate resources are delivered at consumer (gate). Transport to consumer was included because this life cycle stage was identified as a planning variable in step 1. The FU of the study was defined as *'1 m^3 of construction aggregates, from an average Danish gravel pit, delivered to user'*. Gravel pits yield a variety of commercial products consisting of sand and stone, and these resources represents an 80% mass fraction of the Danish aggregate extraction (Statistics Denmark 2015b). A scheme of the LCA model is depicted on figure $P_5$1.

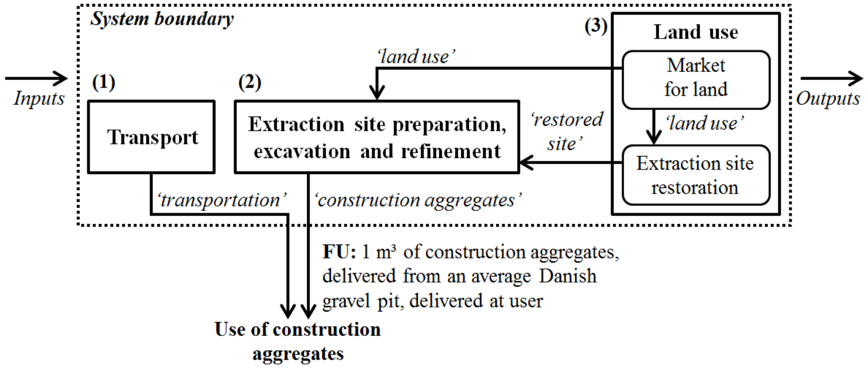


Figure P$_5$1: The applied LCA model. The numerated boxes are grouped activities, and the dotted line is the system boundary. Arrows indicate flows of products or services.

Aggregate production entails an initial acquisition of land (on which the top soil must be removed and stored) and infrastructure. The raw materials are excavated and refined in the extraction phase, after which commercial aggregate products are transported to a construction site. When the land has been fully mined, each extraction site is restored to a new societal purpose. The environmental exchanges from the life cycle stages were categorised in three groups of activities – as illustrated on figure P$_5$1. These groups are: 1) transport, 2) extraction site preparation, excavation and refinement, and 3) land use. It is within these grouped activities the planning variables of step 1 were tested.

The grouped activities of the LCA model

The 44 extraction permits indicate that an average truckload of aggregates leaving Danish gravel pits weighs around 20 tons. Hence, *'transport'* was modelled with *"Transport, freight, lorry 16-32 metric ton, EURO5 RER| transport, freight, lorry 16-32 metric ton, EURO5 | Conseq, U"* from Ecoinvent Centre (2013).

'Extraction site preparation, excavation and refinement' was grouped as one activity, which represents all the impacts planners cannot influence (for instance the machinery used in the gravel pit). This activity is independent from planning variables and it is thus kept constant throughout the analysis. It was included in the LCA model for the sake of completeness since the study aimed at determining the influences of planning on the impacts of a finished aggregate product. The activity was modelled with *"Gravel Round (RoW)| gravel and sand quarry operation| Conseq, U"* from Ecoinvent Centre (2013).

The primary consequence of *'land use'* was assumed to be that of occupying productive farm land for a different purpose than agricultural production. The iLUC impacts of such land occupation were calculated with the model of Schmidt et al. (2015), which likewise has been used in e.g. Dalgaard et al. (2014), Schmidt (2015) and Schmidt and Muñoz (2014). This consequential model mimics the global market response when increasing the demand for productive land by modelling a supply of arable land from a *'market for land'* (see figure P$_5$1). Arable land is supplied by establishing or intensifying farmland elsewhere, generating impacts through deforestation and supply of fertiliser, respectively. Inventory data for determining these emissions were obtained from Schmidt and Muñoz (2014, chp 3.5). Occupying one square meter of Danish farm land in one year will theoretically increase the demand for land with 1.15m^2·year of global arable land since the potential productivity of Danish farm land is 15% higher than the global average. Hence, iLUC impacts can be calculated by feeding the model input data on the land occupation (m^2·year) imposed by each FU.

Introduction of planning variables

The second part of step 2 is to formulate the identified planning variables of step 1 in the LCA model. The planning variable of *'transport'* is modelled within the grouped activity with the similar name. The remaining planning variables of '*extraction intensity*', '*resource thickness*' and '*site restoration*' relate to an induced occupation of land, and they were therefore modelled in the grouped activity '*land use*'. Transposing these planning variables into land occupation values (needed as input data for the iLUC model) required some adjustment. The land occupation induced by aggregate production was divided as deriving from either the extraction process (function of *'extraction intensity'*) or the subsequent restoration (function of *'resource thickness'* and *'site restoration'*). Equation 1 describes the relation between extraction intensity *(I)* and the land occupation it entails.

$$\textbf{Land Occupation}_{\textbf{Extraction}} = \frac{1}{\mathrm{I} \cdot \rho}\ ,\left[\frac{m^2 \cdot year}{ton}\right] \qquad \textbf{(1)}$$

I is the extraction intensity [$\frac{m3}{m^2 \cdot year}$] and ρ is the density of gravel [$\frac{ton}{m^3}$].

Equation 2 describes the land occupation assigned to restoration. The planning variable of *'resource thickness' (T)* expresses the rate of conversion since 10m^3 of resources converts 10 m^2 of land from a 1m thick resource while it only converts 1 m^2 of land from a 10 m thick resource. The planning variable '*site restoration*' was expressed by a production factor *(f)*, which represents the decrease in productivity from prior to future land use. Hence, *f* was given the value *'1'* when converting agricultural land into non-productive land, such as nature. Extensive agriculture with a ban on usage of fertilizer and pesticides was assumed equal to organic farming, resulting in a 20% reduction of yield according to Seufert et al. (2012). Hence, this land use change was assumed to generate an f-value of '*0.2*'.

The last element of equation 2 is the occupational timespan *(Δt)*. It could be argued that an extraction site is unoccupied after it is restored to nature. However, each restored area will theoretically occupy land suitable for agricultural production and thus still generate iLUC, as described by Schmidt et al. (2015). The timespan adequate for calculating these impacts of permanent land conversion must represent the time until which land occupation is projected to cause no more iLUC. From a consequential modelling perspective, this situation will occur only when the demand for productive land reaches its maximum and starts decreasing since a conversion of productive land under these conditions will not cause indirect measures to restore prior production capacity. To our knowledge, no research exists on when this market situation could be reached, and any estimates of such a timespan will be subject to much uncertainty. This study estimated a *Δt* of 50 years.

$$\textbf{Land Occupation}_{\textbf{Restoration}} = \mathbf{f} \cdot \frac{1}{\mathbf{T} \cdot \boldsymbol{\rho}} \cdot \boldsymbol{\Delta}\mathbf{t}, \left[\frac{m^2 \cdot year}{ton}\right] \quad \textbf{(2)}$$

f is the production factor [−], T is the resource thickness [$\frac{m3}{m^2}$], ρ is the density of gravel [$\frac{ton}{m^3}$] and Δt is the occupational timespan [years].

The resulting land occupation inputs for the grouped land use activity are reported in table $P_5$2. It is here evident that restoration generates substantially higher land occupation than variations in extraction intensity.

Land occupation imposed by extraction *[(m² · year)/ton]*			**Land occupation imposed by restoration** *[(m² · year)/ton]*			
Average extraction intensity	*Highest municipal extraction intensity*	*Lowest municipal extraction intensity*	*Average resource thickness*	*20% thinnest resources*	*20% thickest resources*	
0.28	0.17	0.48	1.59	2.48	0.83	*Extensive agriculture*
			7.94	12.42	4.14	*Nature*

Table $P_5$2: Input data for the grouped activity 'land use' under different configurations of the planning variables 'extraction intensity', 'resource thickness' and 'site restoration'.

$P_5$3.4 Step 3: formulation of planning scenarios

The baseline scenario, representing average planning, applied a default input value (production of 1 m^3) for the grouped model activity *'preparation, excavation and refinement',* while average input values were used for *'transport'* (35.5 km) and *'land occupation'* deriving from extraction (0.28 (m^2·year)/ton). All scenarios assumed an initial land use of intensive agriculture, but the baseline analysis further assumed that agricultural activity can be fully restored after extraction (f-factor of 0). Hence, no land occupation was assumed to derive from *'site restoration'*.

Scenarios 1 and 2 explored the systemic impacts of the planning variable *'transport'* by modelling respectively a 10% increase and decrease of the average transport distance. Consequently, the scenarios adjusted the model activity *'transport'* in relation to the baseline scenario.

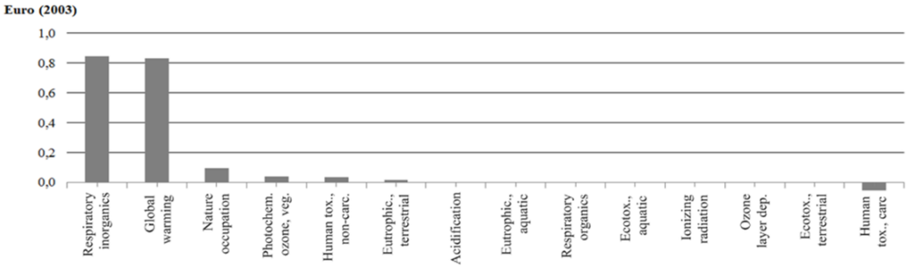
Scenarios 3 and 4 explored the systemic impacts of the planning variable *'extraction intensity'* by modelling the impacts assigned to respectively the most and the least intensive municipal extraction practice. The scenarios adjusted the model activity of *'land use'* in relation to the baseline scenario (using the values of table $P_5$2).

Scenarios 5 to 10 explored the systemic impacts of restoration as a function of the planning variables *'site restoration'* and *'resource thickness'*. Consequently, the scenarios adjusted the model activity of *'land use'* in relation to the baseline scenario. The scenarios analysed the impacts of changing intensive agricultural land to respectively extensive agriculture (scenarios 5–7) and nature (scenarios 8–10). The average thick, the 20% thinnest and the 20% thickest of the 313 resource zones were modelled for each restoration option (using the values of table $P_5$2).

$P_5$4 Results

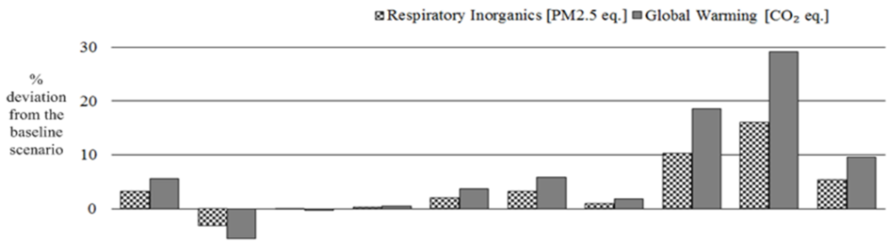
$P_5$4.1 Step 4: analysis of life cycle impacts

The Life Cycle Inventory (LCI) for all scenarios can be found as supplementary information. Life Cycle Impact Assessment (LCIA) results were calculated with the Stepwise method, described by Weidema (2009). Stepwise weights end-point impact categories by monetisation, thus making it possible to express diverse LCA impacts by the same unit. Stepwise is one among many LCA methods which could be used in the application of the procedure. Figure $P_5$2 depicts the weighted results of the baseline scenario. The largest weighted impacts are by far respiratory inorganics (RI) and global warming (GW). Nature occupation is the third largest impact, but it is around 90% lower than the two prior. Hence, RI and GW ought to be in focus when mitigating the systemic impacts of the sector. The scenario results for these two impact categories are depicted in figure $P_5$3.



Respiratory inorganics	Global warming	Nature occupation	Photochem. ozone, veg.	Human tox., non-carc.	Eutrophic., terrestrial	Acidification	Eutrophic., aquatic	Respiratory organics	Ecotox., aquatic	Ionizing radiation	Ozone layer dep.	Ecotox., terrestrial	Human tox., carc
kg PM2.5-eq	*kg CO2-eq*	*m2-years agr*	*m2*ppm* hours*	*kg C2H3Cl-eq*	*m2 UES*	*m2 UES*	*kg NO3-eq*	*Pers *ppm*h*	*kg TEG-eq w*	*Bq C-14-eq*	*kg CFC-11-eq*	*kg TEG-eq s*	*kg C2H3Cl-eq*
1.3·E-2	1.0·E+1	7,9·E-1	1.1·E+2	1,3·E-1	1.6·E+0	6.8·E-1	2.5·E-2	9.6·E-3	2.9·E+2	9.1·E+1	6.2·E-7	1.1·E-4	-1.9E-1

Figure $P_5$2: The weighted results of the baseline analysis (T = 35.5 km, I = 0.28 m^2·year/ton). FU is '1m^3 of construction aggregates from an average Danish gravel pit delivered to user', and the box presents the characterised life cycle impact results.



Scenario:	1	2	3	4	5	6	7	8	9	10
	Change in Transport Distance		Change in Extraction Intensity		Restoration to Extensive Agriculture			Restoration to non-productive land		
Transport: [*km*]	**39.1**	**32.0**	**35.5**	**35.5**	**35.5**	**35.5**	**35.5**	**35.5**	**35.5**	**35.5**
Land Use: [m^2·*year/ton*]	0.28 + 0.00 **0.28**	0.28 + 0.00 **0.28**	0.17 + 0.00 **0.17**	0.48 + 0.00 **0.48**	0.28 + 1.59 **1.87**	0.28 + 2.48 **2.76**	0.28 + 0.83 **1.11**	0.28 + 7.94 **8.22**	0.28 + 12.42 **12.70**	0.28 + 4.14 **4.42**

Figure $P_5$3: Scenario analysis results. The results of the 10 scenarios are presented as % deviation from the baseline scenario (RI = 1.3·10^{-2} kg PM2.5 eq., GWP = 10.0 kg CO_2 eq.). The boxes depict the activity inputs for each scenario.

The sensitivity of *'transport'*

33% of the 0.013 kg $PM_{2.5}$ equivalents and 56% of the 10 kg CO_2 equivalents emitted from the supply of 1 ton aggregates derive from traffic, and it is thus a life cycle process with a high potential for mitigation. In this regard, scenario 1 and 2 showed that a 10% increase or decrease in transport distance generates impact deviations of respectively ±3.3% and ±5.5%. It is thus an influential planning variable which can generate impact deviations.

The sensitivity of *'extraction intensity'*

By comparison, the land use of average extraction adds up to only 0.4% and 0.7% of the weighted impacts in the baseline analysis. The empirical data showed great variation in municipal planning practice. Yet, scenario 3 showed that the most intensively extracting municipality only generates a negative impact deviation of 0.1–0.3%, while scenario 4 showed that the least intensively extracting municipality only generates a positive impact deviation of 0.3–0.5%. Thus *'extraction intensity'* is not a planning variable which generates large impact deviations.

The sensitivity of *'resource thickness'*

The restoration scenarios (5–10) showed great impact variations deriving from the variability of the planning variable *'resource thickness'*. Restoration in the 20% thickest extraction zones only generated impact increases of respectively 1.0% and 1.9% for restoration to extensive agriculture (scenario 7) as well as increases of respectively 5.4% and 9.6% for restoration to nature (scenario 10). Reversely, restoration in the 20% thinnest extraction zones generated impact increases as high as respectively 3.3% and 5.9% for restoration to extensive agriculture (scenario 6) and increases of respectively 16.2% and 29.1% for restoration to nature (scenario 9) – a threefold increase! Consequently, *'resource thickness'* represents a sensitive planning variable which can generate large impact deviations.

The sensitivity of *'site restoration'*

The scenarios 5 to 10 revealed that restoration to extensive agriculture generated much lower impact increases than restoration to nature due to lower land occupation inputs (see table $P_5$2). Extraction in an average extraction zone only generates impact increases of respectively 2.1% and 3.8% for restoration to extensive agriculture (scenario 5) while it generates impact increases of as high as respectively 10.3% and 18.6% for restoration to nature (scenario 8). Hence, *'site restoration'* represents a sensitive planning variable which in combination with the planning variable of *'resource thickness'* can generate substantial impact deviations.

$P_5$4.2 Step 5: formulation of planning recommendations

By analysing impact sensitivity in regard to the different planning variables (step 4), the study allowed formulation of five recommendations for aggregate extraction planning in Denmark:

1. Planning should primarily be concerned with systemic impacts in regard to global warming and the release of respiratory inorganics.
2. A substantial part of these systemic impacts are caused by transport. Hence, measures to minimise transport should be pursued.
3. Even large changes in extraction intensity generate only minor differences in systemic impacts. Hence, resource intensity can freely be adjusted as a means of minimising local impacts.
4. Restoration plans can in combination with resource thickness generate large systemic impacts. Hence, restoration to productive land should be pursued in zones with thin resources.
5. While nature restoration may generate local beneficial impacts, it may also cause negative systemic impacts. To prevent this, nature restoration should primarily be made where a) it does not substitute highly productive land or b) the aggregate yield is high due to a high resource thickness.

Formulating such strategic recommendations early in the assessment phase (bullet c in section $P_5$2.1) of a Danish aggregate SEA could facilitate planners in the iterative process of finding new extraction zones, granting extraction permits and establishing restoration plans. These recommendations could help planners in choosing between alternative extraction sites, as well as they could help in mitigating systemic impacts by for instance prolonging extraction permits in order to retrieve as many resources as possible before nature restoration.

$P_5$5 Discussion

$P_5$5.1 The performance of the procedure

Respiratory inorganics and global warming are both well-established impact categories in current SEA practice. However, the release of respiratory inorganics is today primary treated as a local phenomenon bothering neighbours of extraction sites or aggregate transport routes, and the procedure can thus help to broaden the focus of current practice. The relationship between transport and global warming, on the other hand, is already known by planners, and the procedure did in this regard only manage to confirm the necessity of strengthening current efforts.

Resource extraction intensity had prior to this study not been discussed as an instrument for strategic planning. Rather, it is perceived as an instrument for reaching a local balance between the wishes of industry (increased income by intensive extraction) and extraction site neighbours (reduction of safety and health impacts by extensive extraction). Application of the procedure proved that this planning practice does not generate any substantial systemic impacts.

Restoration to nature is in today's practice largely perceived as a way to generate public support for extraction projects since prospects of a new recreational area nearby can seem attractive for neighbours. Application of the procedure revealed that this spatial prioritisation can generate large systemic impacts. This is new knowledge for planners, and it demonstrates that local impact mitigation in fact can cause substantial increases in systemic impacts.

$P_5$5.2 The LCA assumptions on land use

The applied model for assessing the impacts of land use is a source of uncertainty. Firstly, it is possible that other land use impacts than that of merely occupying the land could occur, such as e.g. reduced natural protection of the on-site ground water resource. Second, the iLUC calculations applied a roughly estimated (and probably underestimated) occupational timespan of 50 years for permanent land use changes. A more pessimistic estimate of a 100 year time horizon would generate iLUC impacts twice as big.

Lastly, impacts on biodiversity were not accounted for in depth, though somewhat accounted for as *'nature occupation'* in the stepwise method. The European biodiversity is currently under pressure (European Commision 2011). Yet, little space is left for biodiversity in Denmark with agriculture and urbanised areas adding up to 76% of total land use (Statistics Denmark 2014). Restoring old extraction sites as new nature could improve national biodiversity. However, occupying former Danish farmland as nature generates an increase in demand for productive land, which under current market conditions will be met through land conversion and/or intensification elsewhere (Schmidt et al., 2015). Restoring old extraction sites as nature is therefore likely to generate local (on site) biodiversity improvements together with a biodiversity loss elsewhere. This balance between biodiversity loos and gain is complex. Different methodologies for accounting for biodiversity impacts in LCA have been proposed, but none of these are currently fully operational (Koellner et al. 2013) and no consensus exists on an appropriate technique (Penman et al. 2010). To our knowledge, there exists no LCA methodology, by which it is possible to project and compare the biodiversity loss, gain and change caused both directly and indirectly by spatial planning. It is fair to assume that on-site land use impacts will be addressed in the local-oriented SEA procedures without the assistance of LCA. Yet, the abovementioned intrinsic limitations of LCA exemplify that the tool cannot calculate all types of systemic impacts with a high degree of accuracy.

$P_5$5.3 Addressing systemic impacts with LCA

As touched upon in the introduction, planning influences a variety of systems on a variety of scales. This makes it worth asking: which systemic impacts can LCA and the proposed procedure help to address?

Generally speaking, LCA can help to address the direct and indirect, global, long-term impacts of planning prioritisations within the boundaries of the available LCA databases. Established LCA methodologies can currently not sufficiently cover all impacts typically dealt with in an SEA (Björklund, 2012), but broadening LCA methodology to describe the impacts on bio-physical systems together with the impacts on social and economic systems (thus encompassing the pillars of sustainability) has been recognised as an important improvement potential (Jeswani et al., 2010; Weidema, 2009). It must be stressed that current databases of LCA makes it difficult to assess local and temporal impacts (though spatial and time weighting can be applied), and issues of impact accumulation on any other spatial scale than global thus is better addressed with other analytical tools.

$P_5$5.4 Implementation of the proposed procedure

The proposed procedure differs from prior studies on operationalizing LCA in SEA by its effort on adjusting LCA methodology to fit planning processes. In doing that, it builds on the belief that fitting LCA within an SEA (integrated in planning) can help to support better decisions. Implementation of the procedure and the assigned decision-making were, however, deemed beyond the scope of this study.

It could be of interest to further analyse the LCA capacity of SEA practitioners, their willingness to use LCA, the potential costs assigned to application of the procedure and/or the opinion of local decision-makers in balancing global against local impacts. Future work could also focus on fitting the procedure to assess alternatives in SEAs of policies or Environmental Impact Assessments (EIA) of projects – in line with the recent study of Židoniené and Kruopiené (2015).

At last, it is important to highlight that several studies have found SEAs to be ineffective in influencing planning outcomes (Pope et al., 2013; Tetlow and Hanusch, 2012). Given that current SEA practices lack effectiveness, future work should consider if including more analytical tools herein is what is needed. It may very well be that the greatest challenge to implementation lies not in the proposed LCA procedure *per se,* but in the SEA framework through which it is implemented.

$P_5$6 Conclusion

LCA has been advocated as a tool which can create needed improvement for SEA. However, little research has to date dealt with how to use the tool actively in planning. This article represents a first step towards operationalizing LCA in planning by proposing a procedure which focuses on the capabilities of planners.

When applied to the case study of Danish aggregate planning, the procedure proved to perform well. It generated new knowledge on how to identify and address key systemic impacts, as well as it helped to highlight the trade-offs in regard to mitigation of local and global impacts. It was, however, argued that LCA cannot cover all the systemic impacts SEAs must consider, though broadening and increased accuracy of LCA methodology could make the procedure more applicable.

The application of LCA within SEA may challenge spatial planners since LCA entails a change in assessment paradigm where the influence on product flows is in focus instead of the region of planning *per se*. Yet, the authors foresee that increased application and development of the proposed procedure can lead to better decision support, as well as it may help to unveil the potential role of spatial planners in making future production patterns more sustainable. This, however, requires engagement and a wish for collaboration among the researchers and practitioners working within respectively LCA and SEA.

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