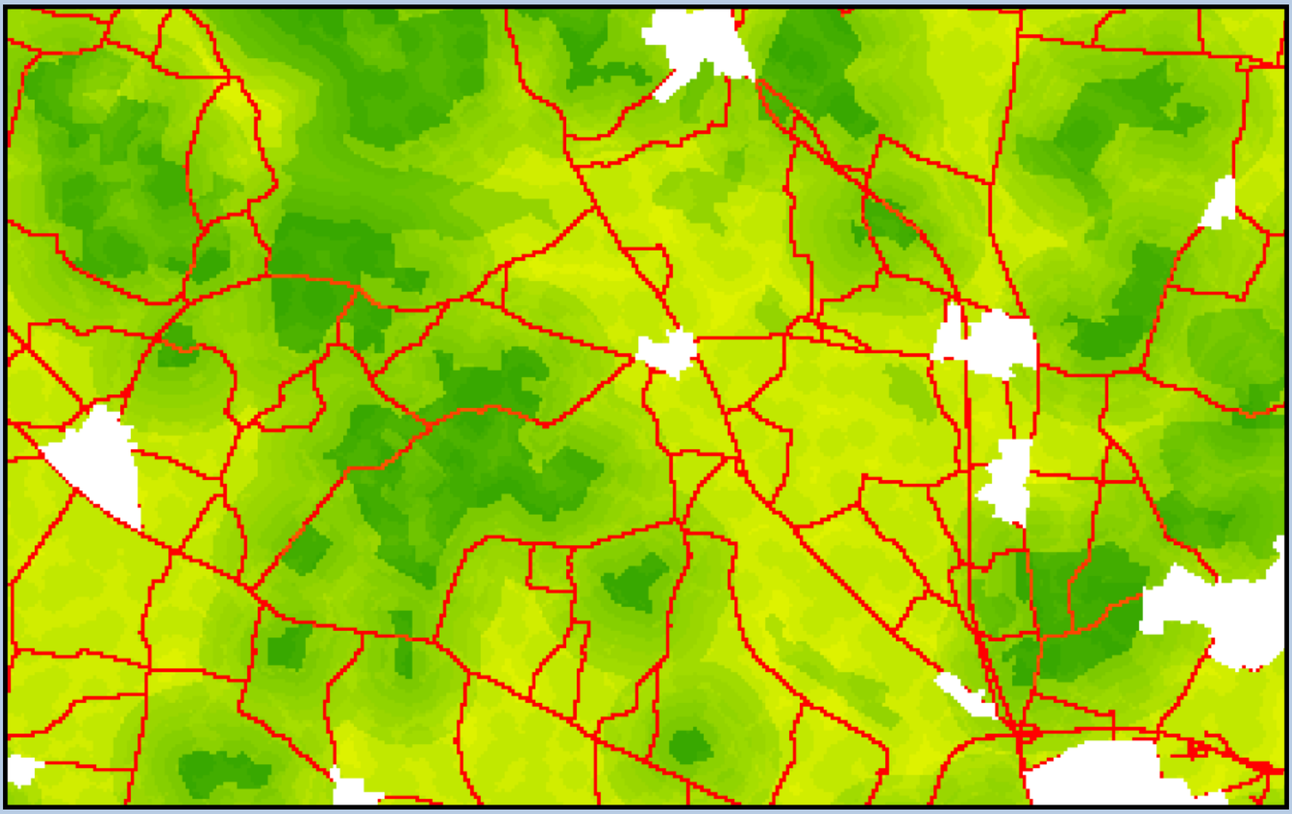
LEAST-COST MODELLING: A POTENTIAL TOOL FOR MAPPING ECOLOGICAL CORRIDORS IN DANISH MUNICIPALITIES?

LASSE FRISTRUP LEMMING
Master's thesis
Geoinformatics



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Abstract:
This project has sought to investigate the possibility of applying least-cost modelling as a tool for mapping ecological corridors in Danish municipalities. The main objective was to determine if the method was suitable for the municipal context and to investigate the best way of implementing the analysis method. An analysis was carried out using Næstved municipality as a study area and a small network of ecological corridors were delineated by using a habitat suitability map as a cost surface. The network of corridors was evaluated and an uncertainty analysis was executed to investigate the impact of uncertainty in the parameters used to generate the cost surface. The report concludes that the method may be too advanced and resource demanding to serve as a general tool in municipalities, although it may be useful for some more resourceful municipalities. While the analysis sought to model functional connectivity, the possibility is left open that a model based on structural connectivity may be simpler to use and thus more fitting for the planning context in the municipalities.

Preface

This Master's thesis was written by me, Lasse Fristrup Lemming, as part of my Geoinformatics Master's program at Aalborg University. The project started at the beginning of February 2016 and ended with the hand-in of the report on June 8th. The thesis concerns the use of least-cost modelling for mapping ecological corridors in Danish municipalities.

I would like to give my thanks to Lise Schrøder who provided me with plenty of useful feedback during the project. I would also like to thank Sofia Mulla Kølmel and Birte Hvarregaard from Næstved municipality's Department of Environment and Nature for answering the questions I had.

Table of contents

1 Introduction

The landscape of Denmark is one of the most intensively cultivated in Europe. About two thirds of it is devoted to agriculture and a tenth of it consists of urban areas, roads and other constructions, leaving little space for areas of higher nature quality, such as forests, heaths or bogs (Danish Nature Agency, 2014a). This has not always been the case. Prior to massive human intervention in the landscape, Denmark was largely covered by forests. The drive towards agricultural production eventually resulted in forests taking up merely 3% of the landscape at the beginning of the 1800's, a figure that has since been much improved due to reforestation (Natur- og Landbrugskommissionen, 2012). Nevertheless, the intensive cultivation of agricultural soils, the draining of wetlands, straightening of streams and other transformations of the landscape have had a deeply negative impact on biodiversity in Denmark (Wilhjelmudvalget, 2001a).

One significant driver of biodiversity loss is habitat fragmentation. As natural areas with high biodiversity are transformed to make room for agriculture, roads or urban areas, a process of fragmentation occurs, in which habitats become smaller and more isolated. The reduced size of the habitats has adverse effects for the ecological processes taking place within, and the areas become increasingly affected by edge effects, such as traffic noise from roads or pesticide runoff from agriculture (Wilhjelmudvalget, 2001b). A major problem associated with habitat fragmentation is the reduced capacity for inter-habitat movement of organisms. The roads and the intensively cultivated agricultural fields essentially act as barriers to movement. Dispersal of organisms between habitats is vital for biodiversity, as it makes populations resilient towards disturbances, enables recolonization and improves genetic exchange (Wilhjelmudvalget, 2001a).

The loss of biodiversity has been the subject of major political initiatives on a national, EU and global level. The EU's Habitat Directive of 1992 along with the Birds Directive of 1979 mandated the establishment of national networks of high value natural areas within the EU, called Natura 2000 areas, which were to receive special protection (Danish Nature Agency, n.d. b) (European Commission, 2016a). Within Denmark, a considerable number of strategies, laws and initiatives have been implemented with the aim of preserving endangered species and improving the general ecological quality of the Danish nature (Danish Nature Agency, 2014a). Many of the goals set forth in the various plans and strategies are, in part, intended to be achieved through the spatial planning work within Danish municipalities (Danish Nature Agency, 2014a). One of the tasks assigned to Danish municipalities is to designate areas that can serve as ecological corridors or potential ecological corridors. Ecological corridors are more or less visible features in the landscape that are mostly linear in shape and serve as connections between separated habitats. The purpose of these corridors is to improve the capacity of animals and plants to disperse between the natural areas in the landscape, thus mitigating the negative effects of habitat fragmentation (Danish Nature Agency, 2014a).

A 2013 report prepared for the Danish Nature Agency studied the work done in relation to ecological corridors in Danish municipalities (Hellesen, et al., 2013). The report found that municipalities lacked a common GIS-based method for mapping and appointing ecological corridors and suggested that work should be done towards creating such a method. Through studying the municipal plans and interviewing the municipalities, the authors of the report found only one systematic GIS-based approach for mapping ecological corridors in use (Hellesen, et al., 2013). However, the simplifying assumptions underlying the method raise questions about the appointed corridors' ecological validity. Finding a superior alternative would be ideal.

The protection and restoration of ecological corridors is a widely used conservation and nature management tool, not just in Denmark, but throughout the world (Wade, et al., 2015). A widely used method that has been applied for modelling these corridors is referred to as 'least-cost' modelling. It is based on the idea of finding a route between two locations that provides the least amount of resistance to movement. In these models, the landscape is divided into a grid of cells that each provide a certain amount of resistance (Adriaensen, et al., 2003). Resistance can be defined and understood in numerous ways, but a simple way of defining it could be that it represents the probability that an organism would be able or willing to cross a certain landscape element (Adriaensen, et al., 2003). The route drawn by the model can be expanded to a wide corridor, and this corridor can then be used as the spatial extent of an ecological corridor in a conservation project. The mapped corridor, which may be one among many corridors in the landscape, is considered to provide the best connectivity between two locations, assuming the model is correct (Wade, et al., 2015).

The question of whether least-cost modelling can be used in Danish municipal planning arises. The method could possibly provide more realistic representations of the best corridors available for the dispersal of species. However, the method's viability is dependent on whether the necessary data is available, if the assumptions made in the modelling process are reasonable, and whether the method and its increased complexity compared to other methods is suitable for Danish municipalities. If it did turn out that the method could be applied in within Danish municipalities, it would also be necessary to explore how this kind of modelling could be implemented in a manner that fits within the municipal context.

2 Problem statement

The discussion and issues mentioned in the introduction have led to the following problem statement (which consists of two research questions):

Problem statement:

> **How suitable would least-cost corridor analysis be as an approach for mapping ecological corridors within Danish municipalities? How should such an analysis be structured and implemented in order to fit the municipal context?**

Here, least-cost corridor (LCC) analysis refers to the use of least-cost modelling to identify corridors in the landscape. To avoid any confusion, similar terms will be clarified: Cost surface modelling is a group of models in which an area of analysis is divided into a grid of cells called a cost surface, where each cell has a numeric cost value. Different methods exist for mapping routes within this cost surface, one of which is called least-cost modelling. This method finds the route between two points that has the lowest accumulative cost. This route can be a one cell wide path with the lowest cost, which would be referred to as least-cost path (LCP) modelling. It can also be a wide swath of cells containing a set of routes with the lowest costs in the surface, which would be referred to as least-cost corridor modelling. Least-cost modelling will be described further in chapter 6. Least-cost corridor analysis should in this report simply be understood as an analysis that makes use of least-cost modelling to locate corridors.

The term 'ecological corridor' refers to the term found in the Danish Planning Act, and excludes the similar term 'potential ecological corridor' also found in the act.

3 Project design

This project seeks to answer the problem statement through two means: First, by digging deeply into the theory underlying least-cost modelling of ecological corridors and the municipal planning context in which this modelling would take place. Second, by conducting a concrete LCC analysis to map a network of ecological corridors within a small section of a Danish municipality. The knowledge gained from the first part combined with the hands-on experience gained from the second part will serve as a basis for a discussion that will seek an answer to the problem statement. The project design can be seen in Figure 1.

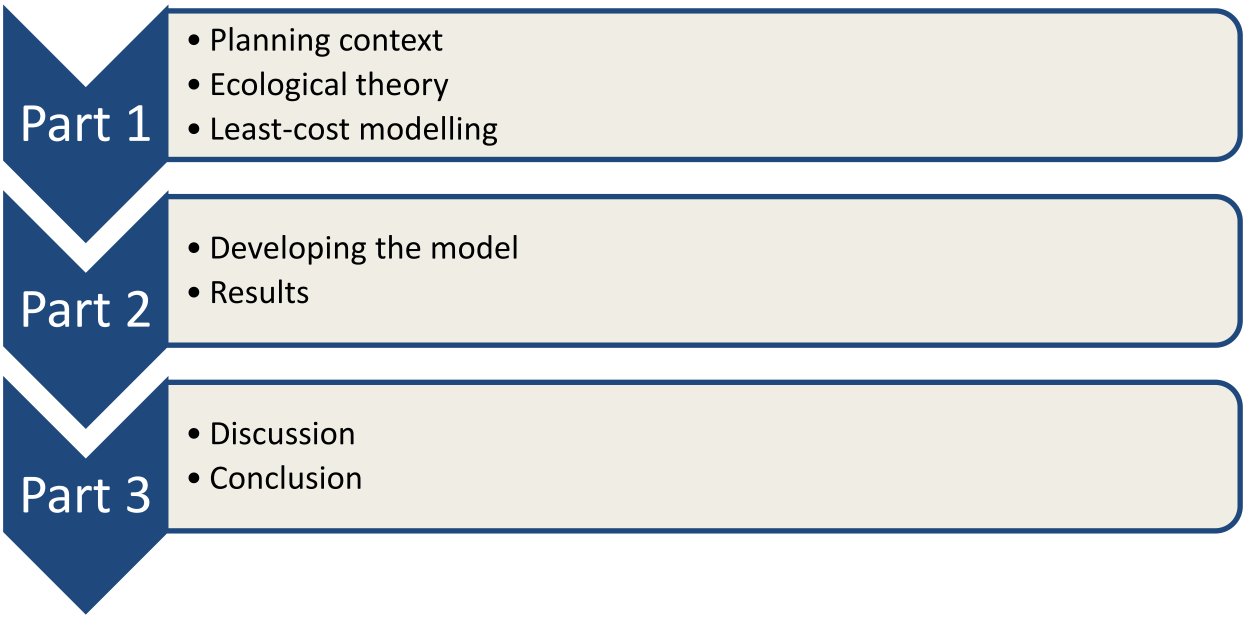

Figure 1. The project structure. The bullet points are chapter names, and the arrows show the different parts build upon each other.

This chapter will describe the structure of the report and how each chapter contributes to answering the problem statement. The selection of the case municipality for the LCC analysis will be described at the end.

Part 1

Answering the problem statement requires a thorough description of the political and administrative context in which the planning takes place. This will be found in chapter 4. This involves describing the underlying issue with diminishing biodiversity which is the reason ecological corridors have become a planning tool. It also involves describing the international, EU-level and national political framework that municipalities are working within. The requirements and recommendations that municipalities must consider when planning ecological corridors are described, followed by a description of the actual work being within municipalities with regards to planning ecological corridors.

In chapter 5, the reader is introduced to a number of theories in ecology that serve as the foundation for understanding ecological corridors and how they can be used as conservation tools. This chapter will discuss habitat fragmentation and landscape connectivity, introduce the patch-matrix-corridor theory of the landscape, and cover meta-population theory which describes how separated populations of organisms respond to disturbances and inter-patch migration. The issue of scale in landscape ecological research is covered, and the concept of ecological corridors is expanded upon. All this serves to provide the scientific backdrop for this type of landscape analysis.

In chapter 6, the reader is introduced to a thorough description of the scientific research underlying the field of least-cost modelling. The assumptions, the analytical structure and the graph-theoretical algorithm underlying cost-surface modelling will be described. The reader will be introduced to the many questions and challenges that inevitably arise in this sort of modelling.

Part 2

However, the issues related to least-cost modelling described in chapter 6 will inevitably only be general ones. In a concrete analysis many more issues that are uniquely tied to the specific problem being studied will arise. In addition, none of the research covered deals with the Danish municipal context. Therefore, a central goal of this project has been to perform a concrete analysis of a network of ecological corridors in a Danish municipality. It is argued that a purely theoretical walk-through of the available research and the municipal administrative context would not be sufficient to become familiar with the actual challenges and decisions municipalities would be faced with, were they to incorporate LCC analysis into their planning. An actual LCC analysis can serve as a test of whether the approach holds up when applied for municipal planning. Some of the challenges and issues that are uniquely tied to the Danish municipal context will be drawn out in the open and become available for analysis and discussion. As any individual LCC analysis is associated with its own unique issues, a case study of a single Danish municipality cannot provide an all-encompassing overview of the issues facing municipalities in general. However, the analysis can hopefully provide insight into general issues that municipalities could face, as well as some of their solutions.

The theory gained from **Part 1** concerning least-cost modelling, ecological theory and the planning context serves as a foundation for the LCC analysis. Chapter 7 will describe the considerations underlying the design and execution of the analysis. This includes how to map potential habitats, how the corridors between the habitats were delineated, how the value of the identified corridors was evaluated, and how the impact of uncertainty in the model was analyzed. In chapter 8, the results of the analysis are presented.

Part 3

The discussion in chapter 9 will then broaden the topic to debate the accuracy of the results and the potential improvements that could be made. It will also attempt to provide some sort of conclusion to the question of how municipalities should design their analyses if they were to employ LCC analysis as a tool in their planning, taking the conducted LCC analysis as a point of departure. LCC analysis will be compared to other known methods of mapping ecological corridors, and the important question of whether the approach is actually suitable for use in municipalities is discussed.

In chapter 10, the final conclusion is presented, which seeks to summarize the answers to the problem statement

3.1 Case municipality

A Danish municipality had to be selected for the analysis. Within a limited section of this municipality, LCC analysis would be applied to generate a network of ecological corridors between potential habitats of a specific species. Furthermore, an interview would be sought to gain insight into the municipality’s work with ecological corridors.

Næstved Municipality was chosen for this project. The reasons for this choice lie in the fact that it is a relatively large (in terms of space) municipality with low urbanization and plenty of nature. This means it can serve as an illustration of how the approach can be applied to other large municipalities with substantial amounts of nature. These municipalities are the key to ensuring a Danish landscape rich in biodiversity, whereas smaller and more urbanized municipalities are less important. Interviewing a municipality with these characteristics was also deemed likely to provide the most pertinent information concerning how municipalities work with ecological corridors. In addition, it has been argued that using models as a basis for designing ecological corridors is most useful when urbanization or other human-made barriers do not constrain the spatial extent of corridors (Beier, et al., 2008b), thus making Næstved more suitable than a highly urbanized municipality.

An interview was conducted with an employee at Næstved municipality's Department of Environment and Nature. The interview provided some useful information that will be referred to a few times throughout the report. It did not, as hoped, provide much information about how their own appointments of ecological corridors had been made, as these had not been updated since they were taken over from the previous county in the 2007 Local Government Reform. Following the interview, a few questions were asked over email. The communications can be seen in Appendix.

The location of Næstved municipality within Denmark can be seen in Figure 2. The figure also shows the current ecological corridors appointed in the municipal plan.

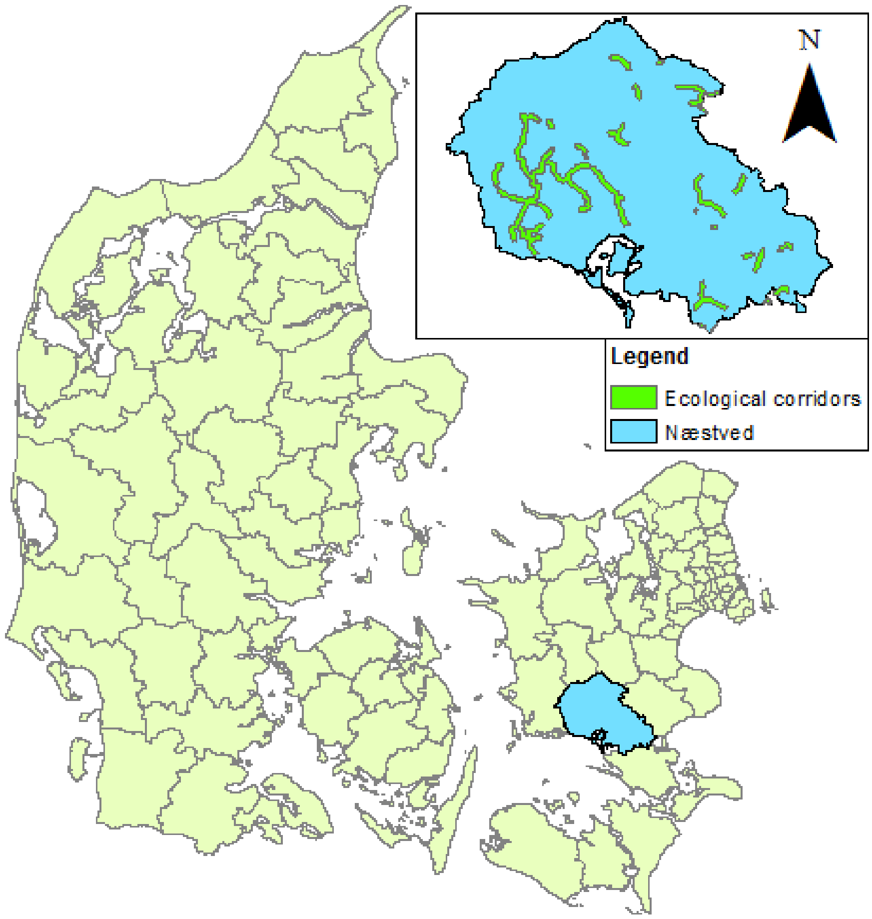

Figure 2. Shows the location of Næstved municipality within Denmark. Næstved is enlarged in the top right window and is overlaid with the ecological corridors appointed in the municipal plan. Data from (Erhvervsstyrelsen, 2015b) and (GeoDanmark, 2016).

4 Planning context

4.1 Biodiversity in Denmark

The landscape of Denmark is characterized by a high proportion of land allocated to agricultural uses and a limited amount of natural areas with long-term continuity and high ecological quality (Danish Nature Agency, 2014a). About 66% of Denmark's area is devoted to agricultural uses, 10% to urban areas, roads and other infrastructure, 14% to forests, and 9% to heaths, meadows, lakes and bogs. The current division of land use and the quality of the remaining natural areas can to a large degree be attributed to the past centuries of human exploitation of natural resources through activities such as agriculture, forestry, and urban development (Danish Nature Agency, 2014a). Intensive cultivation of forests and agricultural soils has reduced the capacity of these areas to support biodiversity, and the draining of wetlands and straightening of streams has likewise contributed negatively to biodiversity (Wilhjelmudvalget, 2001a).

Furthermore, the Danish landscape is characterized by a high degree of fragmentation. The natural areas are separated from each other by the dominant agricultural land use, the roads, railroads, and urban areas (Wilhjelmudvalget, 2001a). This fragmentation splits the natural areas into smaller units. This adversely affects the ecological processes taking place within, and it increases the amount of area affected by edge effects, such as runoff of fertilizer or pesticides from adjacent agriculture (Wilhjelmudvalget, 2001b). The size of animal and plant populations within the patches becomes smaller, and the isolation of the patches reduces migration between the populations. The flow of organisms between populations is vital for conserving biodiversity. Small populations are vulnerable to fluctuations in population size caused by disturbances, resource limitations or other causes, which may result in such populations dying out completely. On the other hand, large populations are more capable of recovering once conditions become favorable again. Migrating organisms from other patches can serve to boost the size of a failing population, can recolonize areas where the species has died out, as well as enable a genetic exchange between populations (Wilhjelmudvalget, 2001a). The presence of roads, railroads, agriculture, and so on, acts as a partial or complete barrier for many organisms that would otherwise migrate between populations (Danish Nature Agency, 2014a). For this reason, connectivity between natural areas is vital, and it is in this context that the political interest in ecological corridors must be viewed.

4.2 Planning framework for Danish municipalities

Danish nature and environmental planning and policies are situated within a global and regional framework of international conventions and EU Directives. A number of international conventions play a role in the conservation work of the Danish government and municipalities, such as the UN Convention on Biological Diversity and the Bern, Bonn and Ramsar conventions, which all aim to conserve specific species or habitat types, as well as biodiversity in general (Danish Nature Agency, 2014a). Within the EU, the Birds Directive from 1979 and the Habitats Directive from 1992 constitute the backbone of EU biodiversity policy (European Commission, 2016b) (Danish Nature Agency, n.d. b). These directives require the Danish government to put into place efforts to conserve certain species and habitats. The Birds Directive focused on providing designated protection zones for endangered bird species. The later Habitats Directive established protected areas for a wider range of species, as well as nature types (Danish Nature Agency, n.d. b). Combined, the protected zones of the Birds Directive and the Habitats Directive make up a network of protected areas called Natura 2000 (European Commission, 2016a). The Habitats Directive contains a number of annexes that list the nature types and species that are to be protected. Annex II lists a number of species whose core areas of habitat are to be included in the Natura 2000 network, Annex IV lists a number of species that must be strictly protected both within and outside Natura 2000 habitats, while Annex V lists species that can be exploited, but must be maintained in a favorable conservation status (European Commission, 2016c).

In addition to Natura 2000 areas, Denmark has implemented a general protection status for certain nature types over a certain size, such as heaths, meadows, and bogs (Danish Nature Agency, 2014a). These areas are sometimes referred to as Section 3 areas, in reference to the law establishing them. Furthermore, Denmark has four national parks. All in all, these protected nature areas, along with a number of areas protected by preservation orders, make up the key areas of the Danish nature network (Danish Nature Agency, 2014a).

With regards to national policies, shifting Danish governments have instituted a number of long term strategies and plans concerning the protection of the Danish environment (Danish Nature Agency, 2014a). To reach the objectives set forth in the national strategies, nature considerations are integrated into the spatial planning. In Denmark, spatial planning is a responsibility assigned purely to the municipalities in a highly decentralized system. Municipalities must adhere to the requirements listed in the Planning Act and must seek to implement the regional development plan set forth by the Region in which they are situated. The municipalities must also ensure that their plans can satisfy the requirements set forth in the 'List of State Interests' which is published by the government every four years. Municipalities must revise their municipal plans on an ongoing basis, and present an updated planning strategy every four years. A municipal plan includes an overall strategy for development and land use in the municipality; it includes guidelines describing how certain types of land use should be administered; and it includes a framework for local plans in the different areas of the municipality (Danish Nature Agency, 2014a).

The limited space in Denmark and the many different needs that must be accommodated can present a challenge, particularly with regards to the need for agricultural land versus the need for high-biodiversity nature. For this reason, planners often seek to employ a holistic approach to planning in order to accommodate multiple needs within the same area, whenever possible (Danish Nature Agency, 2014a).

In their municipal plan, the municipalities must account for a wide range of subjects related to conserving and improving nature. The most important task with regards to this project is the municipal task of designating areas as ecological corridors and potential ecological corridors and providing guidelines for these areas (Danish Nature Agency, 2014a). These guidelines are not directly, legally binding for the land owner. Instead they specify how the municipality intends to achieve its planning goals for these areas and how it must administrate the areas. For example, when a building permit application for an area in an ecological corridor is processed, the guidelines may result in a rejection of the application or a requirement that the building project will have to meet certain demands (see Appendix).

The purpose of these ecological corridors is to bind together the many different protected areas mentioned earlier, as well as raising the connectivity between habitats in general, thus improving the dispersal ability of plants and animals (Danish Nature Agency, 2014a). The corridors themselves can be in the form of either corridors in the strict sense, i.e. contiguous, dispersal-conducive areas that connect habitats, or stepping stones, i.e. a number of separate patches of a certain ecological quality strewn between the habitats, which can also support dispersal (Vejre, 2007). In most municipal plans, the delineation of ecological corridors has tended towards linear, contiguous corridors, however (see the maps at *kort.plansystem.dk*).

4.3 Recommendations for Danish municipalities and their work with ecological corridors

There are a number of requirements and recommendations that municipalities must consider when planning ecological corridors. The primary one is a requirement mentioned in the list of state interests, which states that municipalities must ensure that there is no construction of roads and buildings within the designated corridor (Erhvervsstyrelsen, 2015a). This requirement is usually implemented as a guideline for ecological corridors in the municipal plans. Other common guidelines have been described in a report by Hellesen, et al. (2013), which was prepared for the Danish Nature Agency and released in 2013. The report concerned the planning of nature interests and ecological corridors in the 2009 Municipal Plans. Among other things, the authors investigated how municipalities were handling ecological corridors in the planning process, including what methods they were using to delineate the corridors, and what guidelines for management of the corridors they were including in their plans. In addition to investigating the 2009 municipal plans themselves, they also interviewed the municipalities concerning the 2009 plans as well as their intentions for the 2013 plans. They found that the guidelines included in the plans could be divided into four types. Three of these guideline types were found in the large majority of municipal plans. Those three included a guideline requiring that the corridors were to be kept free from construction (as required in the list of state interests), a guideline stating that corridors should be improved in size and quality, and a guideline requiring that replacement biotopes or fauna passages are to be implemented if it is necessary to construct roads or buildings within the corridors. The fourth guideline was found in only 12 of the 98 municipalities, and stated that efforts should be made to remove or counteract barriers within the corridors (Hellesen, et al., 2013).

Another finding of the report was that only 39 municipalities had updated their area allocations for corridors and nature interests since taking over the task from the counties that were disbanded in the 2007 Local Government Reform (Hellesen, et al., 2013). Although things may have progressed since 2009, the case municipality of this project, Næstved, has also not yet updated the ecological corridors they took over from the counties (see Appendix). The report also recommended that a GIS method be developed that could be used by all municipalities for the delineation of ecological corridors (Hellesen, et al., 2013). No indications that such a method had been developed following the report could be found despite an intensive search.

Municipalities are advised to coordinate their appointments of ecological corridors with neighboring municipalities to ensure that areas of nature are properly connected across municipal borders (By- og Landskabsstyrelsen, 2008). Municipalities are also encouraged to design corridors in a way that accommodates specific species or supports specific nature types. Recommended candidates are Annex IV species or species that are listed as endangered on the Danish Red List (Hellesen, et al., 2013) (Vejre, 2007). The focus can be on specific species, groups of species, or even ecosystems whose conservation the municipality chooses to prioritize in their planning (Vejre, 2007).

Furthermore, municipalities are encouraged to make use of a planning approach called 'nature quality planning' (Vejre, 2007) (Hellesen, et al., 2013) (Wilhjelmudvalget, 2001b). Nature quality planning is an approach used to prioritize and administer nature areas according to their quality, to ensure that conservation efforts are directed towards the areas that are most vital for the overall geographical area under consideration. The approach is used for a variety of purposes, such as nature restoration, general conservation, and administration of various environmental grants (By- og Landskabsstyrelsen, 2008).

4.4 Approaches for the planning of ecological corridors used by Danish municipalities

With regards to concrete approaches used by Danish municipalities for the planning of ecological corridors, not much is known aside from a couple of approaches used by a few municipalities. The previously mentioned report concerning the planning of ecological corridors in the 2009 municipal plans found that no municipal plans mentioned using any specific methods for delineating ecological corridors. In the authors' subsequent interviews with the municipalities, they found only a single GIS-based method for appointing ecological corridors, used in the so-called "Trekantområde" (English: Triangle Area), a cooperating region of six municipalities (now eight) located around the Lillebælt area (Hellesen, et al., 2013). An additional GIS based method developed by the consultancy business Biomedia, similar to the one from Trekantområdet, has been used in Slagelse Municipality and others to appoint corridors (Danish Nature Agency, 2014b) (Hellesen, et al., 2013). The elusiveness of concrete methods for delineation of ecological corridors, GIS-based or not, perhaps indicates that municipalities tend to take a more ad-hoc approach to appointing their corridors. It is possible that municipalities have increased their use of formalized methods since the publishing of the aforementioned report, but no further methods were found during the search conducted in this project at hand. Despite not finding other methods in their study, Hellesen, et al. (2013) noted that some municipalities had specified what kind of areas they decided to include in their ecological corridors. These included Section 3 areas, stream systems, riparian zones, fences; ditches and similar, forests, and afforestation areas. Only 10 municipalities explicitly designed their corridors in a way intended to improve conditions for specific species or habitat types. The most common goal in that regard was to connect Natura 2000 areas. It should also be noted that a number of municipalities have not appointed any areas as ecological corridors due to having only minimal amounts of nature within their borders (Hellesen, et al., 2013). A cursory investigation conducted during this thesis also indicated that a large proportion of ecological corridors found in the latest plan data from the Danish municipalities corresponded very closely to Geodanmark data for streams.

The method developed for use in Trekantområdet is intended to connect nature areas of the same type, such as heaths, pastures or forests, or areas that contain endangered species. Creating corridors between Natura 2000 areas is also described as one of the method's main purposes. The method consists of a number of steps. First the analysts must define which types of areas they wish to connect. Based on this, relevant GIS layers are selected and are merged into one layer. Buffers are then created around the selected areas, and a different buffer is then subtracted from the result. The buffer distances used for this must represent the distance that species are able to disperse from their habitat, so that the end result ends up joining the areas of interest together in a smooth band, if the areas are within dispersal distance. The analyst must then identify the core areas that need to be connected. The result from the previous buffering operation is then trimmed so only areas connecting the core areas remain. Corridors crossing over unfeasible areas such as cities must also be removed (Vejle Kommune, 2013).

The approach used in Slagelse municipality and others was originally developed as a tool for pinpointing large-scale nature parks on Zealand as well as corridors connecting them. This is then applied on a smaller scale by Slagelse municipality. In the approach the analyst must define different types of corridors, with suggested types being wooden, coastal, wet, and dry corridors. The analyst identifies GIS data representing landscape features whose presence would indicate an area was suitable for a specific type of corridor, and then calculates the concentration of the objects within a grid net placed over the study area. The analyst then identifies linear areas with a high concentration of the landscape features that are able to connect the designated core areas. The method thus appears similar to a multicriteria-approach, although it does not include any weighing between factors.

5 Ecological theory

This chapter will describe the ecological theory that serves as the foundation of least-cost modelling of ecological corridors. It will touch upon the subjects of island biogeography theory and metapopulation theory that came to lay the ground for the field of landscape ecology. It will discuss the troublesome subject of scale which is also very important within landscape ecology. It will discuss the subjects of habitat fragmentation, connectivity, as well as the theory of source and sink habitats. It will describe the division of the landscape into the patch-matrix-corridor framework, followed by a discussion of the function and value of corridors.

5.1 Habitat fragmentation and connectivity

Projects involving the restoration and conservation of ecological corridors seek to mitigate the effects of habitat fragmentation and to preserve landscape connectivity (Wade, et al., 2015). As mentioned in chapter 4, habitat fragmentation caused by changing land use poses a major threat to biodiversity. Habitat fragmentation is the splitting apart of habitat into smaller parts. However, the process of habitat fragmentation tends to involve the conversion of habitat areas into other land use, such as roads or agriculture. Even though they are often lumped together, distinguishing between the effects of habitat loss and habitat fragmentation can be useful (McGarigal, et al., 2005). This distinction will be made in this report, although with the recognition that the effects are usually connected. Habitat fragmentation increases the total surface area of habitats in the landscape, causing more areas to be affected by edge effects, including noise from roads and fertilizer or pesticide runoff from adjacent agriculture (Wilhjelmudvalget, 2001b). Habitat fragmentation also reduces landscape connectivity; a term that refers to how much the landscape facilitates or impedes the movement of organisms (Taylor, et al., 1993) (Wade, et al., 2015) (McGarigal, et al., 2005). As the landscape loses connectivity, the ability of organisms to migrate, disperse and forage deteriorates (Wade, et al., 2015). A distinction can be made between structural and functional connectivity. The degree of functional connectivity in a landscape is always species-specific and it is dependent on the species' "scale of movement, perception of the landscape, resource needs, and behavioral responses to landscape elements and patterns" (Wade, et al., 2015). Structural connectivity, on the other hand, is not related to the behavior of an organism, but is rather a measure of the physical continuity of specific landscape characteristics (Wade, et al., 2015). Structural connectivity is occasionally used in connectivity studies as an umbrella for measuring connectivity of – frequently unspecified – groups of species, especially when knowledge of these species' behavior is not well understood (Wade, et al., 2015). However, protecting or restoring structural connectivity is not guaranteed to improve functional connectivity of species (Wade, et al., 2015).

5.2 Landscape ecology – origin and theories

Landscape ecology is a broad field of study which Turner, et al. (2001a) describe as the study of "the interaction between spatial pattern and ecological process, that is, the causes and consequence of spatial heterogeneity across a range of scales". The field came to prominence in Europe in the 1950's and 60's, and underwent major development when it reached an American audience in the 1980's. The main distinction to be found between landscape ecology and standard ecology is the former's focus on the importance of space and spatial patterns. Ecological understanding previously tended to treat landscape characteristics as something that was homogenous across space, whereas studies in landscape ecology focused on how ecological processes were affected by spatially heterogeneous landscapes composed of mosaics of ecosystems (Turner, et al., 2001a).

The theory of island biogeography had a large influence on the development of landscape ecology (Turner, et al., 2001a) (Schneekloth & Vejre, 2007). The theory arose in the 60's and consisted of two basic parts. The first part was the proposition that distance to an island from a source is inversely proportional, and the size of the island directly proportional, to the probability of a species reaching the island. The second part is that the probability of a species going extinct on an island becomes smaller as the size of the island increases (Turner, et al., 2001a). Although the theory concerned actual islands, it soon became applied to landscapes in general. Theorists visualized landscapes as being composed of an inhospitable matrix (water in the original theory) with variably sized patches of suitable habitat (islands) being strewn across this matrix (Wade, et al., 2015). Although the theory proved useful, biologists later came to find that the dichotomy between matrix and habitat patches was rarely as extreme as the theory suggested (Wade, et al., 2015).

Due to the aforementioned issue, among other criticism of island biogeography theory, landscape ecologists later came to draw more upon metapopulation theory (Turner, et al., 2001a) (Schneekloth & Vejre, 2007). Within this theory, separate subpopulations of a species, occupying separate habitat patches, are said to make up an interconnected set of subpopulations termed a metapopulation. Each subpopulation has a certain probability of undergoing local extinction, which would imply the metapopulation is eventually bound to become extinct as well. However, if subpopulations from one patch can recolonize other patches, it is possible for the metapopulation to persist over time even if individual subpopulations go extinct. That is, only as long as the rate of recolonization is higher than the rate of extinction (Turner, et al., 2001b). This was a simple observation with significant implications. It implies that destruction of habitat can shift the balance between extinction and recolonization, with consequences for the whole metapopulation. It also implies that reduced dispersal ability due to an increasingly hostile matrix will reduce recolonization rates, with potentially critical results as well (Turner, et al., 2001b).

Metapopulation theory has been extended to account for situations in which habitat patches have varying reproduction and mortality rates. When a habitat patch has a higher reproduction rate than its mortality rate, it is termed a source patch. It will produce more organisms than the patch can support, and the excess individuals will move towards other patches. Habitats where mortality rates are higher than reproduction rates are termed sink patches. A sink patch population will eventually go extinct unless it receives a sufficient quantity of dispersing individuals from source patches. The theory implies that source patches can keep the populations in a demographic equilibrium. It also implies that the loss of just a few source patches could lead to extinctions in sink patches, leading to large declines in the overall population. In relation to nature or conservation management, the theory implies that loss of high quality habitat cannot be counteracted by preservation of sink patches (Turner, et al., 2001b).

Metapopulation theory and island biogeography theory served as foundations for the predominant view of the landscape within landscape ecology, termed the patch-matrix-corridor theory (Wade, et al., 2015). This theory builds on the view of the landscape as being composed of a dominant and highly contiguous land cover type – the *matrix* – dotted with discrete *patches* that differ from the matrix in land cover or other characteristics (Wade, et al., 2015). A landscape may not always have an identifiable matrix (Turner, et al., 2001a). The patches may be connected by *corridors*, which are relatively linear landscape elements that are dissimilar to the matrix (Wade, et al., 2015). Recently landscape ecologists have begun to view the distinction between patches and matrix as being more fluid, recognizing that both matrix and patches may have spatially heterogeneous habitat quality. This new view has been described as a new paradigm, but it does still tend to preserve the view of organisms living in higher quality patches and merely travelling through the lower quality matrix in order to reach other higher quality patches. The organisms in this view will seek to travel along areas of low resistance in the spatially heterogeneous matrix through functional corridors (Wade, et al., 2015). This view of corridors existing as functional linkages in a heterogeneous landscape is essentially the basis for least-cost corridor modelling (Wade, et al., 2015).

5.3 Corridors

A number of different corridor types appear in the literature, and a wide range of different names are used to describe them, causing potential for confusion (Hess & Fischer, 2001). The term corridor can refer not just to landscape elements that provide functional connectivity in the landscape, but also to linear landscape elements that acts as filters or barriers. An example of a filter corridor would be a riparian buffer strip that prevents runoff of nutrients and pollutants from entering rivers, but still allows animal movement. A barrier corridor on the other hand blocks nearly all animal movement across it; the best example being roads (Hess & Fischer, 2001). This report is focused on corridors that provide functional connectivity, and barrier and filter corridors will not be covered. A number of different terms are used for corridors that provide functional connectivity. Some common ones include conservation corridor, greenway, dispersal corridor and wildlife corridor (Hess & Fischer, 2001). A term that has been used and which will also occur later in this report, is 'linkage'. Beier, at al. (2008b) use this term to refer to a swath of land that represents a spatial combination of individual corridors modelled for individual focal species. Wade, et al. (2015) use the term 'linkage' to refer to a network of more or less contiguous optional paths of any shape that provide connectivity, in order to create a distinction from the term 'corridor', which they define as a narrow and usually linear strip of land providing connectivity. Definitions of corridors vary widely, however, and some definitions put very little emphasis on the corridors being linear (Schneekloth & Vejre, 2007). Whenever describing corridors that provide functional connectivity, this report will use the term used in the Danish Planning Act; that is, 'ecological corridor'. The shorter 'corridor' will sometimes be used as shorthand for that.

The functional view of corridors, unlike the structural view, implies that corridors in the landscape do not have to be visually distinguishable, whether through land cover or other characteristics, in the landscape. The only thing that matters is that they improve the movement ability of organisms. However, the manner in which corridors provide connectivity can vary. Corridors can serve as conduits for movement, in which 'passage species' travel through in a short, discrete time period but they can also serve as habitat for 'corridor dwellers' that survive and reproduce within it, requiring several generations to spread across it (Hess & Fischer, 2001). To serve as a conduit it may only be necessary that the corridor is clear of obstacles, while serving as habitat requires the presence of the various resources required by a species (Wade, et al., 2015). Generally a habitat corridor will be wider than a conduit corridor, since edge effects would otherwise reduce the habitat quality (Hess & Fischer, 2001). If length were ignored, that would generally mean habitat corridors could also serve as conduits. In reality habitat corridors are often significantly longer, limiting their use as conduits for species with small dispersal distances. However, often the two functions blend together, as a corridor may serve as a conduit for one species but as a habitat for a different species (Beier & Noss, 1998). Being explicit about the intended function of a corridor is vital when discussing, studying, or designing ecological corridors (Hess & Fischer, 2001) (Wade, et al., 2015).

The protection and restoration of ecological corridors has often been proposed as a conservation tool for improving landscape connectivity and biodiversity (Hess & Fischer, 2001). The value of ecological corridors and their utility as a conservation tool had been a controversial issue for a period of time, but most researchers have come to recognize the value of corridors (Beier & Noss, 1998) (Wade, et al., 2015) (Schneekloth & Vejre, 2007). However, the value of a corridor is always dependent on its specific characteristics, the specific landscape and the species in question. In addition, there are other conservation measures available that may be more ecologically or cost effective, such as the preservation or restoration of large habitats (Wade, et al., 2015) (Ejrnæs & Nygaard, 2013) (Beier & Noss, 1998).

5.4 The issue of scale

The final topic in this chapter is scale; an important topic within landscape ecology (Wiens, 1989). Scale refers to the grain and extent of a study, or the grain and extent at which a process occurs or an organism relates to its environment (Turner, et al., 2001c). For a raster dataset, grain would be the cell size, whereas for a vector dataset grain would be the minimum mapping unit (the minimum size needed to be included as a separate object in the data). Extent is the overall study area. Scale can also refer to the temporal dimension of a phenomenon (Turner, et al., 2001c). It is well recognized within landscape ecology that the choice of scale for a study often has a large influence on the observed patterns or processes. Patterns or relations found at one scale may be completely different or non-existent at a different scale (Turner, et al., 2001c) (Wiens, 1989). The correct scale for an analysis depends entirely on the question being asked, and must be carefully considered when conducting an ecological analysis, including the modelling of ecological corridors (Turner, et al., 2001c) (Wade, et al., 2015).

6 Least-cost modelling

This chapter will describe how least-cost modelling can be used for the modelling of ecological corridors in landscapes. It will begin with an explanation of the principles underlying cost-surface modelling in general, which will then be followed up with a discussion of how it can applied to model ecological corridors.

6.1 What is least-cost modelling?

Least-cost modelling is a modelling methodology that does not solely find use within landscape ecology or modelling of ecological corridors. It is a general-purpose methodology for designing or mapping a least-cost path, corridor or cost-weighted distance across a surface (Mitchell, 2012). The surface is divided into a square grid where each cell is assigned a cost value. The nature of the path or the cost varies from project to project. For example, the purpose of a project could be to model the ideal path of a highway across a landscape, and the cost could be the literal amount of money it takes to construct a section of a highway on a certain type of land cover. The algorithm would then draw a route with the lowest possible construction cost between a source and a destination. However, when building a highway one might also want to take other factors into account, such as the economic impact on nearby towns or impact on sensitive wildlife habitat – thus expanding the cost definition to something that is more than just monetary (Mitchell, 2012). What this illustrates is that cost can represent anything of interest, whether it is money, time, energy expenditure or mortality rates for wildlife, or simply a unitless system representing a relative weighting of factors. Aside from modelling paths, which are single-cell wide lines in the surface, it is possible to model wider corridors. It can also be used to model the spread of some phenomenon from an origin, such as the spread of a wildfire or the dispersal distance of an animal from a habitat (Mitchell, 2012). It is thus a highly versatile tool.

The following description of the least-cost modelling process is based on the tools available in ArcGIS, but the process is essentially universal and comparable tools can be found in other GIS packages (GRASS Development Team, 2016). Any least-cost modelling process begins with the creation of a cost-raster, alternatively referred to as a cost surface (Esri, 2014). This is a raster in which cell is given a value signifying the cost of traversing it. One or more datasets are used as a basis for the cost-raster. The different datasets are reclassified to some comparable rating scale and are then added together into the final cost raster, with each input raster sometimes being weighted according to its desired influence in the final cost raster. This cost raster is then used in a cost distance calculation. A cost distance calculation results in an output raster in which each cell is assigned a value showing the accumulative cost incurred when moving through the least-cost route from the nearest source cell to the cell. The location of source cells is defined by the user in a source raster used as input for the calculation (Esri, 2014).

The algorithm for calculation of the cost distance raster is based on graph theory, in which the center of each cell is considered a node, and each node has links connected to each adjacent node. In the case of ArcGIS, adjacent nodes are any nodes located directly in the perpendicular and diagonal directions. The calculation of the cost incurred when travelling from one node to an adjacent node in a perpendicular direction is as seen in Equation 1 (Esri, 2014).

$$a1 = \frac{(cost1 + cost2)}{2}$$

Equation 1. Calculation of cost of moving from one cell to another in a perpendicular direction. From (Esri, 2014).

Where a1 = the cost of moving from cell 1 to cell 2, cost1 = cost of cell 1 and cost2 = cost of cell 2.

When moving in a diagonal direction, a different equation is used, as seen in Equation 2.

$$a1 = 1{,}414214 \frac{(cost1 + cost2)}{2}$$

Equation 2. Calculation of cost of moving from one cell to another in an diagonal direction. From (Esri, 2014).

In this calculation, the cost incurred is multiplied with the square root of 2 (i.e. 1,414214), to reflect the fact that the center of the of the node in the diagonal direction is further away, which thus makes the movement incur a larger cost.

As the algorithm moves from node to node, an accumulative cost is calculated. The accumulative cost of moving to a cell is simply calculated as the cost incurred from moving through all previous cells plus the cost of moving to the new cell (Esri, 2014).

An algorithm must be used to determine the least accumulative cost from a source to each cell for the cost distance output raster. The algorithm used in ArcGIS works by iteratively picking the cell with the lowest accumulative cost, assigning its accumulative cost to the corresponding cell in the output raster, and then examining the cost of connecting to the neighbors of that cell, proceeding like this until all cells have been evaluated (Esri, 2014). Figure 3 shows an example of a source raster and a cost raster.

1	1		

1	3	2	3.5
10	2	1	4
3	2	1	2.5
2	0	1	0

Figure 3. The left image is an example of a source raster, where the numbered cells are sources. The right image is an example of a cost raster. Created with inspiration from (Esri, 2014)

To provide a more precise description, the algorithm begins by assigning the accumulative cost 0 to each source cell, as there by definition is no movement here. Next, all neighbors of source cells are identified, and the cost of connecting them to a source is calculated. Figure 4 shows how the result of this step would look if the source and rasters in Figure 3 had been used. The neighboring cells are now active, and their individual accumulative costs are arranged in a list from lowest to highest. To be considered active, a cell must be a neighbor to a source cell or neighbor to a cell for which the lowest accumulative cost has been determined (Esri, 2014).

The cell on the active list with the lowest accumulative cost is selected, and its value is assigned to the output raster, since there are no routes with a lower cost to be found for this cell. The cell is removed from the active list. The neighbors of that cell now become active cells since a route to a source has opened for them. The accumulative costs of moving from the cell into the new cells are calculated. The accumulative cost of moving from the cell into neighboring cells that were already active is also calculated, but the calculated cost only replaces the old one if it is lower, which is never the case for cells directly touching a source (Esri, 2014).

0	0	2	
10	2	1	

Figure 4. The second iteration of the algorithm, using the above source and cost rasters.

However, the process then starts over, by selecting the next cell with the lowest accumulative cost on the active list, adding it to the output raster and examining its neighbors. As this process is repeated over and over, more effective paths may become uncovered, and active cells may experience a lowering of their accumulative cost. The process stops when every cell has been given a value in the output cost distance raster (Esri, 2014).

The cost distance output can be used as the end goal itself, as it can be used to show the spread of some phenomenon, or it can be a stepping point towards modelling the least-cost path or corridor. Modelling the least-cost path requires three inputs: The cost distance raster, a raster containing the destination location, and a backlink raster in which the value of each cell describes the direction to the next cell in the route to the nearest source. A single-cell wide path is then generated from the source (as defined in the cost distance raster) to the destination location using the directions from the backlink raster (Esri, 2014).

To generate a thicker corridor to connect two points, instead of using the single-cell wide least-cost path, one could apply a buffer to the path. However, an alternative approach exists for creating thick corridors whose spatial extent is restricted to areas with the lowest accumulative cost values. Creating such a corridor between two points requires the creation of two cost distance rasters. One cost distance raster using the first point as a source, and one cost distance raster using the other point as a source. By adding the two cost distance rasters together, the value of each cell will represent the least accumulative cost of the least-cost path going from the two points while crossing through the cell. A threshold value can be set in the resulting raster to exclude any cells with values above the threshold. The resulting corridor can be said to contain a number of (partially overlapping) paths connecting the two points, all of which have a lower accumulative cost than the paths outside the corridor (Esri, 2014). Figure 5 shows two examples of cost distance layers and Figure 6 shows the result when they are joined together.

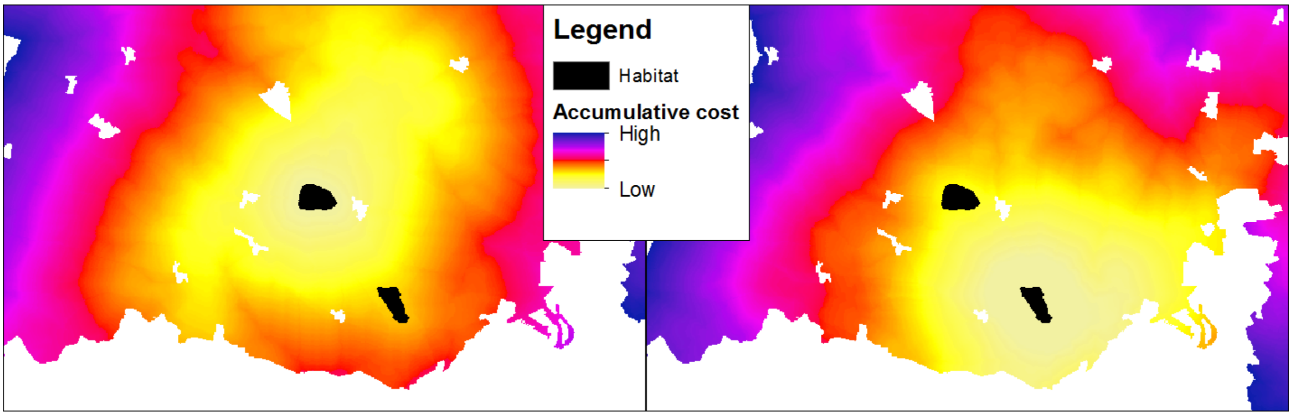

Figure 5. Two cost distance layers: One for each of two habitats.

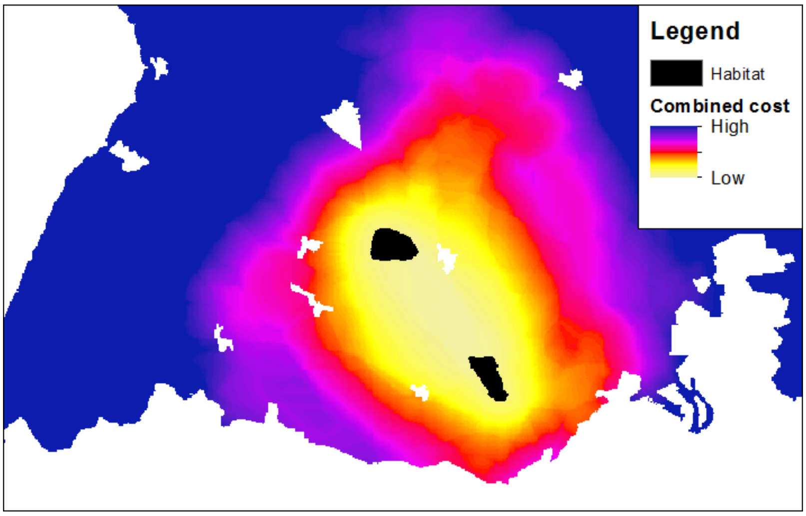

Figure 6. The two cost distance layers are added together to generate this corridor layer. No threshold value has been applied.

6.2 Least-cost modelling and ecological corridors

The use of least-cost modelling as a tool for mapping ecological corridors has become a substantial area of research and there are numerous examples of the approach being applied in concrete corridor design projects (Beier, et al., 2008b) (Wade, et al., 2015). The method has been transferred to this domain by applying the cost-surface as a representation of the resistance an organism is faced with when crossing certain landscape features, and the least-cost algorithm is used to model an ideal path or corridor through the landscape for the organism. This enables researchers and practitioners to incorporate both landscape features and the behavior of specific organisms when modelling corridors between habitats (Adriaensen, et al., 2003). Part of the appeal of the approach likely lies in its accessibility – the tools being available in common GIS packages – and because it does not require an extraordinary amount of data or computing power (Wade, et al., 2015) (Adriaensen, et al., 2003). Least-cost paths or corridors appear, on the surface, to be based on the organism having perfect knowledge of the landscape to determine the best route - an assumption that for most animals is likely far removed from biological reality (Wade, et al., 2015). However, for research purposes it is reasoned that a low cost corridor will statistically experience more wildlife crossings than less suitable corridors, and for conservation projects it is reasoned that, by their very nature, least-cost corridors provide better connectivity for organisms than corridors with higher resistance (Beier, et al., 2008b) (Wade, et al., 2015). Resistance is frequently used as a synonym for cost in the corridor modelling literature, and the resistance values are typically used to represent some form of ecological cost incurred when crossing a certain landscape features (Adriaensen, et al., 2003) (Wade, et al., 2015) (Beier, et al., 2008b). The ecological cost is typically understood to be factors such as travel time, energy expenditure or likelihood of mortality, however, the nature of the cost is in fact often vaguely defined (Wade, et al., 2015) (Zeller, et al., 2012). Adriaensen, et al. (2003) also offer a definition of resistance as a measure of the probability that an individual organism would be willing or able to cross a particular landscape element. Despite its benefits and popularity, the cost-surface approach for modelling corridors is often troubled by untested or unstated assumptions, which lessens the scientific credibility of the approach and makes it less appealing to stakeholders in conservation projects (Beier, et al., 2008b). In particular, the often highly subjective rating of resistance values for landscape features has been highlighted as a problem (Beier, et al., 2008b) (Wade, et al., 2015) (Zeller, et al., 2012). The following subchapter will seek to walk the reader through the numerous steps involved in cost-surface corridor modelling while discussing some of the issues and considerations mentioned in the substantial body of literature written about the topic.

6.3 Modelling least-cost ecological corridors: The process

This subchapter will describe an analytical structure for LCC analysis inspired mainly by two research papers, and it will describe the many considerations associated with each step in this analytical structure.

Wade, et al. (2015) published a comprehensive guide to cost-surface based connectivity modelling in cooperation with the U.S. Department of Agriculture. Their report focused not only on least-cost approaches, but also other path-finding algorithms that use resistance surfaces. In the guide they reviewed 47 connectivity modelling studies and 31 real projects in the U.S. to improve connectivity that used resistance surfaces as a basis for modelling. The report discussed a wide range of considerations researchers and practitioners must be mindful of, and among a wide range of other recommendations, they arrived at eight critical steps that must be handled when modelling linkage designs for wildlife. Although they explicitly choose the term ‘linkage’ rather than corridor, their definition of a linkage corresponds to common definitions of corridors. The eight steps they propose are explicitly directed towards the modelling process and do not touch upon the planning or implementation phases of corridor design projects. In a 2008 study, Beier, et al. (2008b) also provide an overview of 16 considerations that must be dealt with when designing linkages. While the two studies mentioned use the term ’linkage’, the steps and considerations they describe are equally suitable for corridor design.

An analytical structure, which will also be applied as a structure for the remainder of this chapter, has been created with inspiration from these two articles among others. The nine steps of the structure are listed in Figure 7. While the steps are presented in a linear fashion, during the modelling process it may sometimes be necessary to return to a previous step as a result of decision made at a later step.

(Wade, et al., 2015) 8 step approach	**Chapter and analytical structure of thesis**	(Beier, et al., 2008b) 16 questions for analysts
1. Define the type of connectivity to be modelled 2. Create resistance layer(s) 3. Define what is being connected 4. Calculate ecological distance 5. Map potential linkages 6. Validate potential linkages 7. Assess climate change effects (optional) 8. Quantify connectedness (optional)	1. Define connectivity type 2. Select focal species 3. Select scale of analysis 4. Identify variables for the resistance surface 5. Set resistance values and weights 6. Define locations to connect 7. Delineate the corridors 8. Evaluate the corridors 9. Validation and uncertainty analysis	1. How should the analysis area be defined? 2. How should focal species be identified? 3. What landscape factors should the model include? 4. What metric should be used for each factor 5. How should resistance of each class of pixels be estimated? 6. How should factor resistances be combined? 7. How should a corridor terminus be delineated? 8. How should habitat patches be delineated? 9. How should corridor dwellers be modeled? 10. How should continuous swaths of low-resistance pixels be identified? 11. How wide should a single-species corridor be? 12. How should corridors of multiple focal species be combined? 13. How wide should the linkage design be? 14. Is the best corridor any good? 15. How can the linkage design accommodate climate change? 16. How should the linkage design address barriers and management practices?

Figure 7. Eight steps in linkage design proposed by (Wade, et al., 2015), and 16 considerations in linkage design described by (Beier, et al., 2008b). Both are directly cited. A nine step analytical structure used for this thesis is in the center.

6.3.1 Step 1: Define connectivity type

As mentioned in chapter 5, a distinction is normally made between structural and functional connectivity. However, functional connectivity can be further subdivided into five different types that are related to movements of an organism at different spatial and temporal scales (Wade, et al., 2015). Wade, et al. suggest that explicitly stating what kind of connectivity a corridor design is intended to provide ought to be a requirement, as the type of connectivity not only affects the way the corridor should be designed, but failing to specify it makes it difficult to evaluate the quality of the corridor after its implementation. In total, Wade, et al. suggest there are six different types of connectivity that can be modelled (Wade, et al., 2015). *Daily Habitat* connectivity represents the daily movement of an organism to fulfill its needs for food water and shelter. It takes place on a small spatial and temporal scale. *Seasonal Migration* connectivity is related to the annual or seasonal movement of certain animals which takes place on a large spatial scale relative to the organism's daily movements. As part of the definition, this kind of movement requires movement back and forth between locations. *Demographic* connectivity is related to a large spatial scale movement of organisms between sub-populations, which is required to prevent decline of some sub-populations (recall the discussion concerning source and sink patches in chapter 5). *Genetic* connectivity is related to the transfer of genes between sub-populations and is associated with the same spatial scale as demographic connectivity. *Range Shift* connectivity is related to a large scale movement of populations due to changing climate or introduction of exotic species. It occurs on a much larger temporal scale than demographic or genetic movement. Future climate change means that some species will require connectivity enabling their range to shift in response to the changing conditions. Finally, there is *Structural* connectivity, which is based on the assumption that a lack of physical barriers and connections between landscape features with similar characteristics will be beneficial for biodiversity (Wade, et al., 2015). A common structural approach is to base the resistance grid values on the 'naturalness' or landscape integrity within each cell (Wade, et al., 2015) (Beier, et al., 2008b). A landscape possessing structural connectivity is not a guarantee of functional connectivity for organisms. However, Wade, et al. suggest the approach can be useful for "assessing general scenarios of land cover or ecosystem change" (Wade, et al., 2015).

All these types of connectivity are important for conservation of species. The choice of connectivity to focus on, which does not have to be restricted to a single type, has some implications as to how a corridor should be designed, mainly with regards to the spatial scale (Wade, et al., 2015). For example, a corridor design intended to improve an organism's ability to acquire its daily resources would likely be on a much smaller scale than a corridor design intended to improve demographic connectivity. However, one can argue that, in general, elements that improve one type of connectivity also tend to be beneficial for other types of connectivity. For example, improving a corridor's ability to function as habitat would provide benefits for any organism that must traverse the corridor over a short or long term period, which could be beneficial for possibly all six types of connectivity.

The issue of connectivity type raised by Wade, et al. can be tied to the discussion of whether corridors should serve as conduits for movement or as habitat itself. Each species has a certain distance it can cross before access to food, water and shelter becomes a priority. Thus if the corridor is longer than the species can cross in the short term, the corridor must be able to serve as habitat to some degree, allowing movement across the corridor to take place over days, years, or even generations. If the length of a corridor between two sub-populations is short, the corridor would be able to provide demographic connectivity even if it performs poorly as habitat (Wade, et al., 2015). However, a large scale corridor design intended to improve range shift connectivity would almost certainly have to function as habitat in order to support this long term movement.

The 'ecological corridors' that Danish municipalities must delineate for their municipal plans can in principle encompass almost any of these types of connectivity and spatial and temporal scales (Schneekloth & Vejre, 2007). Municipalities are recommended to design corridors that connect specific nature types or accommodate the needs of specific species. These two approaches can be seen as focusing on structural connectivity and functional connectivity respectively. Schneekloth & Vejre (2007), in a report written for the Danish Nature Agency, suggest that the municipalities' ecological corridors can be of a wide variety of sizes and only explicitly rule out trans- and intercontinental corridors. As long as the corridor serves to improve the movement ability of organisms and contributes to overall biodiversity, it is up to the municipalities and planners to determine what the purpose of a corridor should be (Schneekloth & Vejre, 2007). However, it is likely beyond the capacity of a municipality by itself to provide the conditions necessary for range shifts of species.

6.3.2 Step 2: Select focal species

Occasionally a corridor design project may begin with a specific organism in mind. However, this step has been placed after the selection of the connectivity type, as it is only relevant if functional connectivity is being modelled. When modelling functional connectivity, the corridor must be designed to accommodate one or more specific species. These are referred to as focal species. The purpose of a corridor design project may be to improve conditions for a specific, possibly endangered, species or the purpose may be to conserve a wider range of species and ecological processes. When attempting to create a corridor that can accommodate a wide range of species and ecological processes, Beier, et al. (2008b) recommend the use of a 'focal species approach', which involves focusing on a smaller array of species that can act as an umbrella for other species and processes. A number of authors have written about approaches to select the species to include in a focal species approach (Beier, et al., 2006) (Coppolillo, et al., 2004). Generally, it is recommended to choose a range of species that each fulfill specific roles in the ecosystem, and in doing so it is believed that the designed corridor or linkage design will be wide enough to accommodate other species that have similar roles (Beier, et al., 2008b). Usually, corridors are generated for each individual species and then combined into what is called a linkage design (Beier, et al., 2008b).

6.3.3 Step 3: Select scale of analysis

The choice of scale for the analysis is important, as was mentioned in chapter 5. The scale consists of the extent and resolution of the analysis.

When setting the extent of the analysis, it is recommended to include not only the area of interest, but also a buffer surrounding it. Failing to do so can result in potentially superior corridors being ignored (Wade, et al., 2015). In the case of Danish municipalities, who are advised to cooperate in the mapping of ecological corridors, it is particularly worth taking heed of this advice.

The grain of the analysis, expressed as the resolution of the resistance surface, should be based on the perceptual scale of the organism in relation to the type of connectivity being modelled (Wade, et al., 2015). Perceptual scale is defined by Wade, et al. as "the grain and extent of an organism's response to heterogeneity in the landscape" (Wade, et al., 2015). Perceptual scale is dependent on various qualities of the species, such as its vagility. Allometric rules, that assume a correlation between body size and spatial scale, can be used to approximate the scale of an organism, but are not perfect. Identifying the proper scale for an interaction between a species and the landscape is difficult (Turner, et al., 2001b). Wade, et al. (2015) found in their review of published corridor modelling studies that most researchers used a cell size between 30-100 m, seemingly as a direct result of data resolution constraints. There is also sometimes a tradeoff between high resolution and computational speed (Wade, et al., 2015). If the resolution chosen is arbitrary, uncertainty testing by running the model using different cell size resolutions will reveal the potential error associated with the grain choice for the analysis (Wade, et al., 2015). Uncertainty testing using a cell size smaller than the data supports will not provide any insights, however.

6.3.4 Step 4: Identify variables for the resistance surface

In the case where functional connectivity is being modelled, the ecological variables for the analysis are selected on the basis of the available scientific knowledge about the organism being studied. Usually, especially when setting resistance values on the basis of literature review or expert opinion, resistance values represent habitat suitability. There is very little literature written on travel costs through different landscape features (Beier, et al., 2006), so habitat suitability is almost always used as a proxy (Wade, et al.,

2015) (Adriaensen, et al., 2003) (Zeller, et al., 2012). The assumption here is that animals prefer to travel along areas that are similar to their habitat (Beier, et al., 2008b). This is an untested assumption, but is likely reasonable if the corridor is intended to accommodate long term movement and the corridor has to serve as habitat (Wade, et al., 2015). There are also empirical methods for determining the resistance values of the ecological variables, but before this the variables themselves must be chosen, and Wade, et al. recommend using habitat suitability as basis for this choice (Wade, et al., 2015). Habitat suitability tends to depend on things such as access to food, shelter, relationships with other species, and so on. However, because GIS data for factors like these are generally not available, proxy data is used instead (Beier, et al., 2008b). The most common variables used are land cover, road data, and human population density or location (Wade, et al., 2015) (Beier, et al., 2008b). Variables such as these are highly related to habitat requirements, but usually do not fully reflect all life requirements of the organism, making this a potential source of error (Beier, et al., 2008b). Data accuracy should also be considered when choosing variables, and as far as possible, only data layers with high accuracy should be included (Zeller, et al., 2012). There is also the risk that certain landscape elements are unstable or dynamic, making it unsafe to rely on the presence/absence of these elements when designing corridors that are meant to persist in the long term.

6.3.5 Step 5: Set resistance values and weights

Wade, et al. (2015) describes the assignment of resistance values as often being the most important step in least-cost modelling of ecological corridors, as it has a major impact on the end result. Using empirical data to set the resistance of landscape features is recommended, but most studies tend to rely on expert opinion informed by a literature review (Zeller, et al., 2012) (Beier, et al., 2008b). Resistance values set on the basis of expert opinion has been shown to perform worse than empirical methods of rating resistance (Zeller, et al., 2012). However, oftentimes empirical data is not available, and the use of expert opinion may be justified if an urgent conservation crisis is present (Zeller, et al., 2012) (Wade, et al., 2015). However, Wade, et al. (2015) state that experts should be restricted to people with more than just general knowledge about the subject in question and also suggest the use of Analytical Hierarchy Process (AHP) as a method of compiling expert opinion into resistance values (Wade, et al., 2015). AHP is an approach towards decision making which involves decision makers comparing different factors with each other in order to arrive at a mathematical weighting of the factors (Coyle, 2004). A number of empirical methods of estimating resistance values are available. The methods are based on examining the correlation between landscape features and either the presence of organisms, movements of organisms, or genetic distance between organisms or populations at different locations (Zeller, et al., 2012). While empirical methods of estimation are preferable to expert opinion, they are also faced with their own problems, which are described in detail by Zeller, et al. (2012). An issue with the use of presence/absence data is that it is only a proxy for the relationship between movement and landscape features, whereas an issue with using movement data is that movement within a home range is often used as a proxy for movement between sub-populations, even though the environment might affect these two types of movement differently (Zeller, et al., 2012).

Since models usually include more than one variable, which often spatially overlap in cells, it is necessary to combine the resistance values for these variables in some way. For example, a certain type of land cover and a road may both be present in a cell. The issue is complicated by the fact that variables often vary in their units of measurements and their range of variation (Beier, et al., 2008b). Converting variables to a common scale of resistance, such as from 1 to 100, is common practice. However, this may not be enough to reflect the relative importance of each variable, and so the question of how to combine variables remains (Beier, et al., 2008b). There are number of ways to combine variables in a way that seeks to account for relative importance of variables. A weighted sum is the most common, but a weighted product and a weighted geometric mean are also possibilities (Beier, et al., 2008b). Weighting parameters can be derived empirically, but when using expert opinion only, there is no way to be certain which method is the most accurate (Wade, et al., 2015). For this reason, it is recommended to perform uncertainty analysis to determine the effect of different weighting parameters (Wade, et al., 2015) (Beier, et al., 2008b).

6.3.6 Step 6: Define locations to connect

The areas that are to be connected by corridors can be referred to as termini. The choice of where to place the termini obviously has a great effect on the location of the corridors, and so the choice of termini must be carefully considered. Studies have used a variety of terminal types; common ones include protected areas and areas with empirically detected occurrence of the focal species (Wade, et al., 2015). However, the most common approach is possibly to derive habitat patches from the resistance surface, since many resistance surfaces are based on habitat suitability models (Wade, et al., 2015). To derive habitat patches from the resistance surface, three things have to be considered (Beier, et al., 2008b). First, what resistance score should serve as a threshold? Second, how large must an area be to be able to function as habitat? Third, how should isolated cells with low habitat suitability within otherwise suitable habitat be dealt with? A possible approach is described by Beier, et al. (2008a). The authors sought to map potential breeding patches and population cores, and to do so, they created a ‘neighborhood score’ for each cell on the basis of the average resistance score in a radius around the cell. The radius depended on the mobility of the species. They then joined areas above a certain neighborhood score and a certain size together to form the breeding patches and population cores. This approach makes it possible to generate contiguous areas of good habitat that can serve as termini by scrubbing away isolated low quality cells.

6.3.7 Step 7: Delineate corridors

Having created a resistance surface and selected the locations to connect, the modeller must then perform cost distance calculations. As described in subchapter 6.2, to model the least-cost path between two termini, only one cost distance calculation must be performed. However, to create a least-cost corridor, a cost distance calculation must be performed for each terminal, which must subsequently be combined. This creates a layer in which the value of each cell represents the cost of the least-cost path that connects the two termini while crossing the cell. By excluding cells with values above a certain threshold, a corridor-like area will remain, which connects the two termini. It may contain multiple departing paths and is not necessarily linear in shape. The corridor contains a number of partially overlapping paths connecting the two termini, all of which have a lower accumulative cost than the excluded paths.

Locating the least-cost corridor rather than the least-cost path is generally recommended as the preferred approach when attempting to model ecological corridors (Adriaensen, et al., 2003) (Wade, et al., 2015). The single-pixel wide least-cost path is associated with a number of problems. The path can sometimes be found within habitat that is otherwise poor, it is sensitive to errors in data, and its size is not appropriate for real world corridor conservation plans (Beier, et al., 2008b) (Wade, et al., 2015). In this thesis the main objective has been to study the mapping of least-cost corridors, although least-cost path has also been used to evaluate the corridor surrounding it, as will be described in chapter 7.

When mapping least-cost corridors, one is faced with the issue of setting a threshold value for excluding cells. An approach that has been used is to set a threshold equal to the LCP cost + n%. One study used the value 10%, although this was chosen arbitrarily (Pinto & Keitt, 2009). The percentage used should preferably be one that is biologically justified. It is easy to say that the corridor should be as wide as possible to avoid edge effects and provide maximum connectivity, but an excessively wide corridor can be expensive to implement and conflicts with other land use needs (Beier, et al., 2008b). According to Beier, et al. (2008b)., decision makers will usually be interested in the corridor design that is no wider than necessary to ensure connectivity, although Beier, et al. also recommend caution and propose a principle of finding the "narrowest width that is not likely to be regretted after the adjacent area is converted to human uses" (Beier, et al., 2008b). An analyst can also provide multiple size alternatives for the decision maker by creating a map of the corridor divided into multiple threshold intervals with different colors. The width of the corridor also depends on whether it should serve as a conduit for movement or as habitat. If habitat is the goal, a possible rule of thumb is to ensure a width roughly equal to the width of the organism's home range (Beier, et al., 2008b). However, this risks creating obstructions to movement when very territorial species have to live alongside each other in the corridor. For this reason, Beier et al. (2008b) recommends setting a corridor width that is "substantially larger than a home range width" (Beier, et al., 2008b). On the other hand, species that are merely passing through and do not have to live within the corridor do not need a corridor as wide as corridor dwellers (Hess & Fischer, 2001).

As said, the corridor generated may have multiple split paths. Some of these paths will be wider than others and some or all may end up with bottlenecks that are not wide enough to ensure connectivity. Loro, et al. (2015) suggest iteratively increasing the threshold value until there is at least one path with no bottlenecks. They propose determining a minimum width for a corridor, and applying a negative buffer to each iteration of the corridor half the size of the minimum width, followed by applying a positive buffer half the size of the minimum width. This will restore the corridor to its original width, but any areas with bottlenecks will disappear. The analyst can then determine if the two termini are still connected, and if not, continue iteratively increasing the threshold value (Loro, et al., 2015). However, setting a cost threshold high enough to exclude bottlenecks can cause other parts of the corridor to become impractically wide (Beier, et al., 2008b). The analyst must find a way to avoid bottlenecks and ensure a reasonable width throughout the corridor, possibly by manually excluding areas that become unnecessarily wide.

6.3.8 Step 8: Evaluate the corridors

Least-cost corridor modelling will always create a corridor between the two termini, but this does not mean that the resulting corridor is of any use, merely that it is theoretically better than any alternative corridor between the termini. It is sensible to provide decision makers with some sort of measure for the value of the corridor in terms of its usefulness for the organism of interest, the connectivity it adds to the overall network, or how good it is compared to other corridors. The accumulative cost of the least-cost path is not a particularly helpful measure if it cannot be translated into a less abstract measure, since cost merely represents the relative impedance of different factors (Beier, et al., 2008b). Wade, et al. (2015) recommend setting a maximum allowable cost for corridors, but determining how much ecological cost an organism can incur is not straightforward, and a cutoff value may not be necessary if the corridor represents actual habitat intended for cross-generational migration. However, accumulative cost can be used to compare potential corridors with each other, making it possible to rank potential corridors in order of their relative effective distance. The habitat patches making up the nodes of the network can also be evaluated in terms of their habitat quality, isolation or population size of the focal species. This would help determine which habitats are most important to improve connectivity to. Wade, et al. (2015) list a number of different measures that can be used to find the contribution of each corridor or patch to the overall connectivity of the patch-corridor network. Providing decision makers with some kind of measure to assist in prioritizing linkages or determine the value of the project is always recommended.

6.3.9 Step 9: Validation and uncertainty analysis

Least-cost corridor modelling relies on many untested assumptions and has many uncertainties. Yet there are numerous examples of this form of modelling serving as the foundation for large scale land management projects (Wade, et al., 2015) (Beier, et al., 2008b). Considering the money, effort and ramifications for wild life involved, decision makers deserve to know how accurate and trustworthy the underlying model is. Wade, et al. (2015) emphasizes the importance of attempting to validate the model's underlying assumptions and modelled corridors with independent data. Both Wade, et al. (2015) and Beier, et al. (2008b) advocate use of uncertainty testing to see how different model parameters and assumptions affect the results of the model, particularly when empirical data for validation is not available. A resistance surface model consists of several different elements that can undergo validation and uncertainty analysis, but only the most important - the resistance surface and the model results - will be covered here. Wade, et al. (2015) also suggests assessing the impact future climate change will have on the modelled corridors, since climate change can potentially alter the entire foundation of the model in the long term. However, inclusion of such climate change analyses in corridor modelling is often challenging and is relatively rare (Wade, et al., 2015).

As mentioned, the generation of the resistance surface is considered to be the most critical step in resistance-surface modelling. For this reason, it is particularly important to attempt to validate the resistance surface with independent data (Wade, et al., 2015). Wade, et al. (2015) suggest using empirical movement data to determine how well the resistance values for landscape features correspond to actual animals behavior. If movement data is not available, species occurrence data could be used to validate the habitat suitability model the resistance scores are based on. With regards to testing the model results as a whole, they propose three different approaches: 'Event/Predictive Validity', 'Comparison to Other Models' and 'Face Validity'. Event/Predictive Validity involves comparing actual animal behavior with the predictions of the model. The model can be compared to other models to see if the results are similar, while Face Validity involves independent experts judging the on-the-surface realism of the model. Each of these provide different degrees of validation, with Face Validity providing only weak validation (Wade, et al., 2015).

It is rare that a model can be considered fully validated, so uncertainty testing is recommended as a supplement or as an alternative. Most decisions made during the construction of the model, such as how to delineate termini, can undergo uncertainty testing to determine how robust the model is to choices that are affected by uncertainty. In particular, using uncertainty testing to determine the effect of alternative ratings of resistance is recommended by a number of authors (Wade, et al., 2015) (Beier, et al., 2008b) (Sawyer, et al., 2011).

A method for analyzing the effect of uncertainty in resistance ratings is described by Beier, et al. (2009). They describe their approach as a worst-case scenario approach. In short, the method involves creating alternative corridors using alternative resistance values, and examining how problematic the proposed corridor would be if one of the alternative resistance surfaces were correct. The method is designed for resistance surfaces based on expert opinion, but can in principle be used for empirically derived resistance surfaces if the estimates are judged to be uncertain. In the method, variables such as land cover and elevation are considered to be factors, and specific types of land cover or ranges of elevation are considered to be classes. Each factor has a weight from 0-100% used when combining factors. Each class was rated as habitat on a scale from 1-10, where low values represented ideal breeding habitat, medium-high values represented nonbreeding habitat, and high values represented areas the organism would try to avoid. Point estimates for factor weights and class resistance were assigned by experts who also had to provide minimum and maximum values for each factor and class, which represented the plausible range of biological uncertainty. The purpose of the method is to test resistance values spanning the range of biological uncertainty rather than every value possible. They propose creating worst-case scenarios where factor and class values are either as compressed or as dispersed as possible within their range of uncertainty, while preserving their rank order. The corridor generated with the point estimates is then compared to the corridors generated with the dispersed and compressed scenarios. For each alternative corridor, the percentage of the alternative corridor overlapped by the proposed corridor is calculated. In addition, the mean resistance of cells in the proposed corridor is compared to the mean resistance of cells in the alternative corridor, using the alternative resistance surface for both corridors. The two measures obtained this way reveal the potential error in corridor location and corridor resistance if the resistance estimates underlying the proposed corridor turn out to be false. This allows decision makers to understand the potential risks before embarking on the project (Beier, et al., 2009).

A number of studies have used uncertainty analysis to investigate how the setting of different resistance values can affect the location of modelled corridors. The results of these studies suggested that the locations of modelled corridors do not change significantly in response to different resistance ratings, as long as the resistance values of the variables are in the correct rank order. It has been suggested that this insensitivity is caused by urbanized areas limiting the possible locations of corridors, although a similar insensitivity was found in a study within an area with relatively little urbanization (Beier, et al., 2008b). These studies do not invalidate the need for uncertainty analysis, as the true rank order of variables may not be known, and there are numerous examples of studies that did find sensitivity to alternative resistance scores (Sawyer, et al., 2011).

7 Developing the model

As the previous chapter revealed, there are many issues to consider when modelling ecological corridors using least-cost corridor analysis. The considerations that went into designing the model of this project will be covered in this chapter, which follows the analytical structure that was proposed in chapter 6 and which can be seen in Figure 8. The nine steps will be described one by one in separate subchapters. In a few cases, the steps were not followed completely chronologically during the modelling process. The situations where this was the case will be mentioned in the respective sections.

Step 1: Define connectivity type
Step 2: Select focal species
Step 3: Select scale of analysis
Step 4: Identify variables for the resistance surface
Step 5: Set resistance values and weights
Step 6: Define locations to connect
Step 7: Delineate corridors
Step 8: Evaluate the corridors
Step 9: Validation and uncertainty analysis

Figure 8. Analytical structure used in the analysis.

Following the description of the nine steps, a subchapter will describe the nine steps in terms of how they were implemented using ArcGIS tools. ArcGIS was chosen as the GIS software for implementing the model for the simple reason that it was the most familiar GIS package to the author. However, the principles of LCC analysis described in this chapter and chapter 6 are not tied to any specific software package, and the analysis could also have been accomplished using other software. The implications, or lack thereof, of using ArcGIS for this analysis will be briefly discussed in chapter 9.

7.1 Step 1: Define connectivity type

The type of connectivity to model must be defined. A number of different connectivity types were discussed in chapter 6. The main distinction is between structural and functional connectivity, but functional connectivity can also be divided into five different subtypes.

A structural approach, based on organisms travelling across areas in the landscape that are less affected by anthropogenic activity, could be considered. However, a functional approach was preferred for a number of reasons. First of all, a structural corridor is no guarantee of functional connectivity. A functional approach can be designed to accommodate the specific needs of a species, and a focal species approach with a range of species can be used to accommodate a wider range of ecological processes and related species. Second, Danish municipalities are encouraged to design corridors that suit the needs of specific species or connect similar nature types. These two options allow for both structural and functional corridors, but focusing on functional connectivity makes it possible to test a method that more explicitly considers the needs of specific species, in comparison to the corridor design methods mentioned in chapter 4.

Functional connectivity was selected as the approach early on, but the specific subtype(s) of functional connectivity was not chosen until the focal species had been selected. The guidelines for municipalities do not explicitly set a scale for the corridors. However, it has been a long term goal to establish a nature network in Denmark, both on a national and municipal scale, and the ecological corridors are intended to play an integral role in this endeavor (Wilhjelmudvalget, 2001b) (Vejre, 2007) (By- og Landskabsstyrelsen, 2008) (Miljøministeriet, 2013). Creating a nation- or municipality wide nature network implies certain things about the scale of the corridors. A municipality wide network can include short corridors, but must include longer corridors in order to connect widely disconnected forests, as an example. For this reason, the focus in this project has been on relatively long corridors that can connect the major areas of habitat in the case municipality. This way the modelling method's value for the creation of a nature network can be evaluated. This implies certain things about the type of connectivity being modelled. Larger animals, like deer, may be able to travel widely enough to use disconnected forests as resource patches, and thus the chosen corridor scale may provide 'daily habitat' connectivity for this species. However, as described in the following step, the agile frog was chosen as the focal species. In comparison to deer, this species moves relatively slowly. The large scale corridors are unlikely to provide daily habitat connectivity for this species. Instead, the corridors can provide demographic and genetic connectivity, which will help the species proliferate widely across the landscape and lessen the chance of isolated populations becoming extirpated.

7.2 Step 2: Select focal species

Due to time constraints only one species was chosen as a focal species; that being the agile frog. This is sufficient to demonstrate the modelling of corridors. However, a focal species approach is usually understood as the selection of a number of species whose targeted conservation can act as umbrellas for other species and ensure protection of a range of vital ecological processes. A focus on a single species does not qualify as a focal species approach. Thus the use of the term 'focal species' when referring to the species of interest in this report should not be interpreted as though this analysis is making use of a focal species approach. A focal species approach to modelling corridors would likely be of benefit for municipalities, however. While some individual corridors designed exclusively for a single species can be useful, the nature networks of municipalities are intended to provide general biodiversity benefits, not just support a single species. The methods for selecting an array of focal species are outside the scope of this thesis. However, once the species are selected, a focal species approach in the context of corridor modelling mainly involves overlaying the individual species-specific corridors with each other in order to create a larger corridor (sometimes called a linkage or linkage design) (Beier, et al., 2008b) (Beier, et al., 2006). Thus, demonstrating the creation of a single species corridor goes a long way towards demonstrating how to implement a focal species approach, besides demonstrating a potential tool for conservation of individual threatened species.

As said, municipalities are encouraged to design corridors that fit the requirements of specific species. They are recommended to consider species whose flourishing best achieves the biodiversity goals described in various strategies and laws, with Annex IV and Red List species being specifically emphasized. Næstved municipality is particularly interested in Annex IV species, so a decision was made to narrow down the potential species of focus to this group. The Annex IV species list contains a range of different species from different taxonomic groups so it was necessary to narrow the list by removing species less suited for the project's purposes. Some animals are more suited for cost-surface analysis than others. It has been suggested that (some) trees take too long to spread their genes, during which time the landscape may become altered and the underlying GIS data inaccurate, and that (some) insects move in ways that are not closely tied to available GIS data (Beier, et al., 2009). Nevertheless, there are numerous examples of both insects, plants and even fish being incorporated into corridor modelling analyses (Beier, et al., 2008b). Nonetheless, it was useful to remove candidates that are less likely to be suitable for least-cost modelling. Flying and aquatic animals, insects and plants were thus ruled out. This left three mammals, a number of amphibians and the sandlizard.

Corridors between habitats that do not contain the target species have limited value, but are not always worthless. In principle, a network of such corridors can, over the course of time, if the habitat conditions are sufficient, enable dispersal from occupied habitats that are connected to the network. However, it is more interesting to model corridors for a species that has at least a minimal presence in the municipality, rather than a species with no existence. Within the mammal group, the common dormouse was the only terrestrial species with a possible presence in Næstved. Its presence is uncertain and highly limited however. Instead, the focus turned towards frogs and toads, which make up a majority of the remaining terrestrial Annex IV species suitable for cost-surface modelling. These species are very similar, so if modelling one of them proved successful, it would bode well for the modelling of all frogs and toads. Frogs and toads in fact appear to make up a majority of the Annex IV species that are 'plausible' for least-cost modelling (Danmarks Miljøundersøgelser, 2007). A number of frogs/toads were present in Næstved. The agile frog was chosen as the final candidate, as it had a significant presence and had habitat requirements that seemed sufficiently straightforward for a non-expert to model. In addition, the species has a wide dispersal ability compared to the other amphibians. This makes it more suitable for modelling the corridors as conduits that allow short term movement, rather than habitats that allow long term movement. There was no intention that the modelled corridors should function specifically as conduits or habitat. However, the agile frog's potential to function as a passage species provided an opportunity to evaluate the corridors' value as either type.

7.3 Step 3: Select scale of analysis

The scale of the analysis consists of the grain and extent. As mentioned in chapter 6, when selecting an extent it is advised to include a significant area outside the area of interest, and the grain should be determined on the basis of the focal species' perceptual scale.

The extent of the model was Næstved and all neighboring municipalities (see Figure 9). This extent was more than sufficient since all habitats were inside Næstved or directly adjacent. It was decided from the beginning that at least one of the habitats to connect with corridors should be across the municipal border outside Næstved, in order to illustrate how cross-border planning of corridors can be achieved.

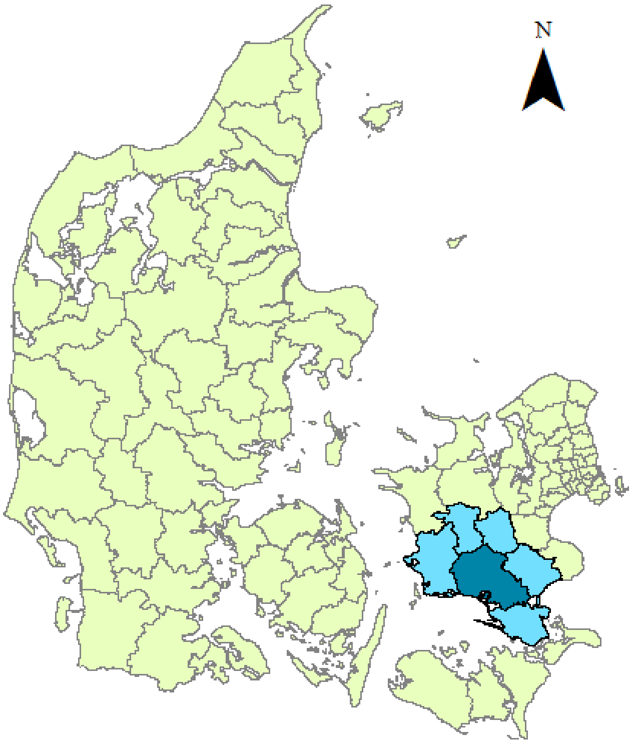

Figure 9. Spatial extent of the analysis area as it is situated within Denmark (Bornholm not pictured). Dark blue is Næstved, the case municipality. Light blue is the additional area included in the model. Data from (GeoDanmark, 2016)

No empirical studies of the spatial scale at which the agile frog interacts with the landscape was found. The agile frog is a relatively small organism and one would therefore expect its perceptual scale to be relatively small as well. In the end, the grain of the model was constrained by the resolution of the available data, and the cell size was set to 50 m. Step 3 thus came chronologically after step 4-5 in the modelling process, as the variables had to be selected and transformed to a suitable raster format. A 30-100 m cell size is typical of least-cost analyses, so this value is about right in the middle. As will be described in the following step, the analysis used four variables: Land cover, presence of roads, distance to deciduous/mixed forests and distance to lakes. A cell size of 50 m was able to accurately represent the shape of land cover and was four times smaller than the distance intervals used for the forest distance and lake distance variables (as will be described in step 5). The cell size also has the effect of creating a roughly 50 m buffer around roads, which could represent aversion to road noise. This is discussed further in the following two steps.

It is possible that the agile frog perceives the landscape at a finer grain than 50 m. However, it must also be considered that the goal is to model corridors on a scale much larger than this. The purpose is not to locate small paths traversed by the frog. It is possible that running the analysis with finer data and a correspondingly finer cell size in the model could result in minor changes to the location of the corridors. However, of the four variables used in the model, only the resolution of the land cover data is a constraint, and land cover class designations of areas are based on the typical land cover within the area. Thus the resistance of each cell is a close approximation to the average resistance within the hypothetical smaller cells constituting the 50 m cell. Given this, it seems unlikely that finer data and cell size would have much of an influence, but it cannot be ruled out.

7.4 Step 4: Identify variables for the resistance surface

As mentioned in chapter 6, there is very little data available concerning the travel cost of animals in various landscape elements. No literature describing travel costs for the agile frog was found for this thesis either, and so habitat suitability was used as a proxy instead, as is the common approach when estimating resistance scores from literature or expert opinion. Estimating functional connectivity using habitat suitability is a major assumption which will therefore be discussed in chapter 9.

A search was conducted for literature concerning factors affecting the habitat quality of agile frogs. Four important factors for habitat quality, that could be included using available GIS data, were found. These were: Land cover, roads, distance to deciduous/mixed forests and distance to lakes. A number of other relevant, but less important, factors for habitat were also found, but could not be modelled using available GIS data. The factors will be described further below.

A handbook written by the National Environmental Research Institute of Denmark to guide the management of Annex IV species describes the habitat requirements of the agile frog (Briggs & Damm, 2007). The agile frog needs to have access to lakes for use as breeding sites. These lakes can be of almost any quality, as long as they are not heavily polluted, sheltered from sun light, or populated by fish. In some parts of Denmark the agile frog exists without having access to any deciduous forests. However, almost everywhere else its presence is highly correlated with deciduous forests. It prefers to use lakes located in deciduous forests or up to 1 km away from it, and the population size is generally larger the closer it is located to such a forest. Deciduous forests provide food and shelter, whereas in the open country these resources are less available. Open areas of the forest that provide plenty of light are preferred. Forests of mixed deciduous and coniferous trees are also useful habitat, although pure deciduous forests are preferred. To a lesser degree, the frog will use meadows and pastures as resting places, especially where there is little forest. Swamps and wetlands in general can also be useful as resting or breeding sites. The frog is also very sensitive to traffic, as it moves relatively slowly across the roads. (Briggs & Damm, 2007)

It is clear that the four identified factors are highly relevant. However, land cover and distance to deciduous/mixed forests seem to overlap to some degree. The decision to include both of those factors was made to capture both the fact that different types of land cover provide different degrees of habitat quality, and the fact that distance to forests is highly important for the frog's access to certain resources, causing population sizes to become smaller the more distant they are from forests. Other research supports the idea of distance to forests being particularly important (Lippuner, 2014) (Laan & Verboom, 1990). Figure 10 shows the distribution of land cover in Næstved.

Some factors that were also important for habitat quality could not be captured in the available GIS data. Quality of lakes is one factor, open areas in forests is another. However, it can be argued that these two factors are more likely to change over time than the others, which could lower their value as factors for long term planning. Improving the quality of lakes and creating open areas in forests can also be steps that are taken following the appointment of a corridor. A number of smaller scale landscape elements also have value as resting places, including rocks, gardens and living fences, but the importance of these for habitat quality is considerably lower than the four identified factors. Fauna passages that improve the ability of amphibians to cross over roads are not particularly suitable as habitat themselves, but do significantly reduce travel costs in areas that are currently classified as roads. Including these in the model could improve the modelled corridors, but no data concerning their placement could be found.

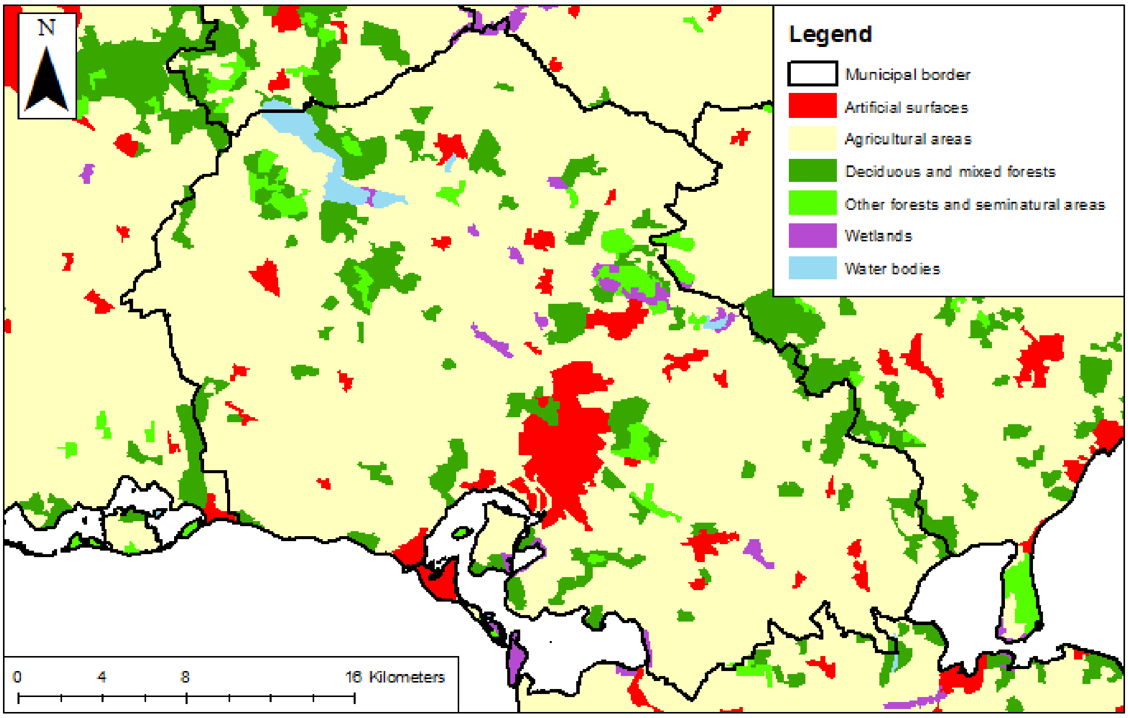

Figure 10. In this image the land cover is grouped into the 5 top hierarchical classes in the dataset. However, deciduous and mixed forests have been split from 'forest and seminatural areas' to highlight them. Data from (Styrelsen for Dataforsyning og Effektivisering, 2014) and (GeoDanmark, 2016)

7.4.1 Data and data quality

The data used for the four factors came from two sources: 'CORINE' land cover data and GeoDanmark (formerly known as FOT) data.

The GeoDanmark data is a collection of vector data maintained by the Danish municipalities and state (FOTdanmark, 2014). It contains a range of data relating to traffic, buildings, nature, hydrology, and others. Two object types in the dataset are relevant for this project; those being lakes and roads. Lakes are polygons and roads are polylines. The thematic accuracy of the lake and road data varies from 1-5%, in terms of errors. These errors include not just object type, but also attributes and attribute values. The points used to register the lake polygon can be off by 1-2 meters. The accuracy of the roads is such that the line is never off by more than 2 meters compared to the real road. All in all it is a highly accurate dataset. It should be noted that lakes are generally only registered if they are above 100 m^2, but in a few cases smaller lakes are included when they are judged to be of administrative or natural significance (FOTdanmark, 2014).

CORINE is an acronym for 'Coordination of Information on the Environment'. The CORINE project was started by the EU in 1985 and its purpose was to collect information about a range of environmental phenomena. One of its products was a geographic dataset describing the land cover of most of Europe. The project was later taken over by the European Environment Agency (The European Commission, n.d.) (European Environment Agency, 1995). The Corine Land Cover (CLC) dataset contains 44 different land cover types, with a geometric accuracy of 100 m or better, and a thematic accuracy of 85% or higher. The land cover assignments are produced using visual interpretation of high resolution satellite imagery in most countries, although some use semi-automatic classification approaches. The dataset was last updated for the year 2012. It has a minimum mapping unit of 25 hectares and it uses a minimum width of 100 m for linear areas (Copernicus Land Monitoring Services, 2016).

7.5 Step 5: Set resistance values and weights

Cost values for the factor classes, weights for the factors, and uncertainty ranges for the cost values had to be assigned. No empirical data was available for determining which values to assign to the different factor classes. When empirical data is not available, relying on one or more experts to assign values, supported by literature, is recommended (Beier, et al., 2008b). Basing cost values purely on literature is available as last resort. This project will depend on cost values assigned purely on the basis of literature, which is sufficient to demonstrate the modelling approach, but does lead to additional uncertainty in the results.

Before the costs, weights and uncertainty ranges can be assigned, it is necessary to find the best ways to represent the four factors in the cost surface layer. The data used for land cover and forest distance came from CORINE and is represented as vector polygons covering all terrestrial and some aquatic areas. Lakes and roads come from GeoDanmark data and are represented as small polygons and polylines respectively. Representing land cover as a cost surface was a matter of converting the data to raster format and assigning values to cells according to the land cover type within. The resulting cost raster can be seen in Figure 11.

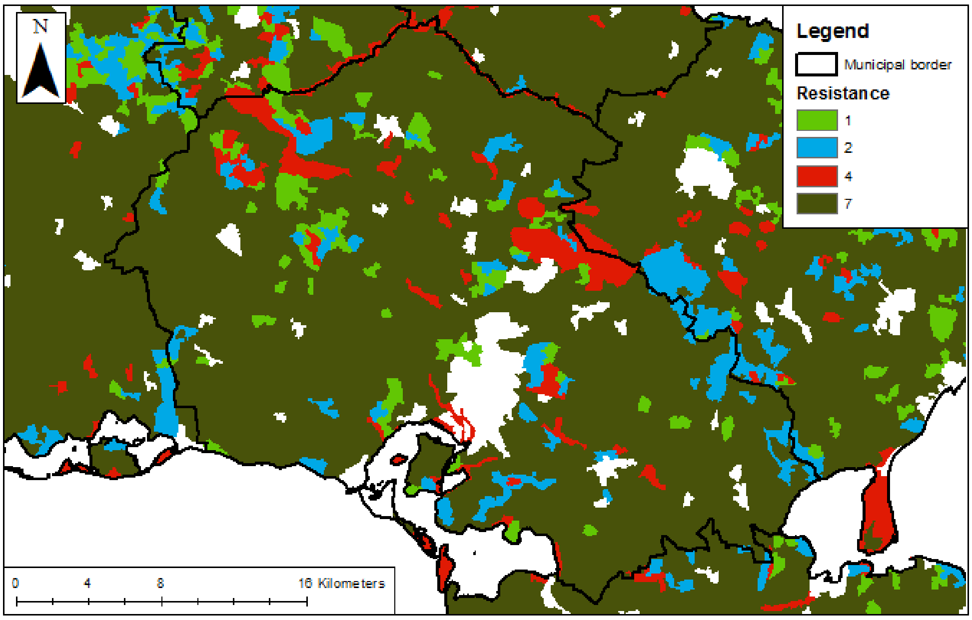

Figure 11. Land cover converted to a cost raster. Impenetrable surfaces such as cities are the color of the background, i.e. white. Data from (Styrelsen for Dataforsyning og Effektivisering, 2014) and (GeoDanmark, 2016).

Representing distance to lakes and forests required some more work. Although distance could be represented as both a continuous and interval-based value, intervals were chosen to simplify the assignment of values. A key question was which intervals to use. It seemed sensible to include an upper limit for distance at which point the distance to forests or lakes would no longer increase resistance, because these feature are already too far away from the organism to be of use. Basing this upper limit on the home range of the species seemed fitting. The aforementioned Annex IV species guide lists this home range as being approximately 1km in diameter, or 500 m in radius (Fog & Hesselsøe, 2007). This outer limit could be represented in the model as 500 m, if each cell in the model is understood as the center of a home range. The species would then at most be able to travel 500 m in each direction. Alternatively each cell can be considered to represent an arbitrary location within the home range, in which case a lake or forest may be located up to 1km away from the cell, if the cell is at the edge of the home range. Both approaches are problematic. A 500 m upper limit would mean that a cell located directly next to a lake, but 600 meters away from a forest, would be above the upper distance limit to the forest, despite the fact that the species is described as preferably living within 1 km of forests. A 1 km upper limit would mean that a cell could be placed just below 1 km away from a lake in one direction and a forest in the other direction; meaning none are above the upper limit, even though at least one is necessarily outside of the home range. In the end, the 1km upper limit was chosen, because an organism within a 500-999 m distance to both a lake and a forest on opposite sides would be able to choose between each resource, and would thus have an advantage over an organism located at a >1km distance from each resource. A 1km upper limit betters reflects this nuance than 500m upper limit. The distance values were grouped into five 200m intervals from 0 to 1000m, and one class above 1000. Both had to be converted to raster format, and layers were created that showed the Euclidean distance of each cell to the nearest relevant forest/lake. The cells in these layers could then be reclassified to reflect the resistance values assigned to each distance interval, resulting in two cost surfaces for these factors that can be seen in Figure 12 and Figure 13.

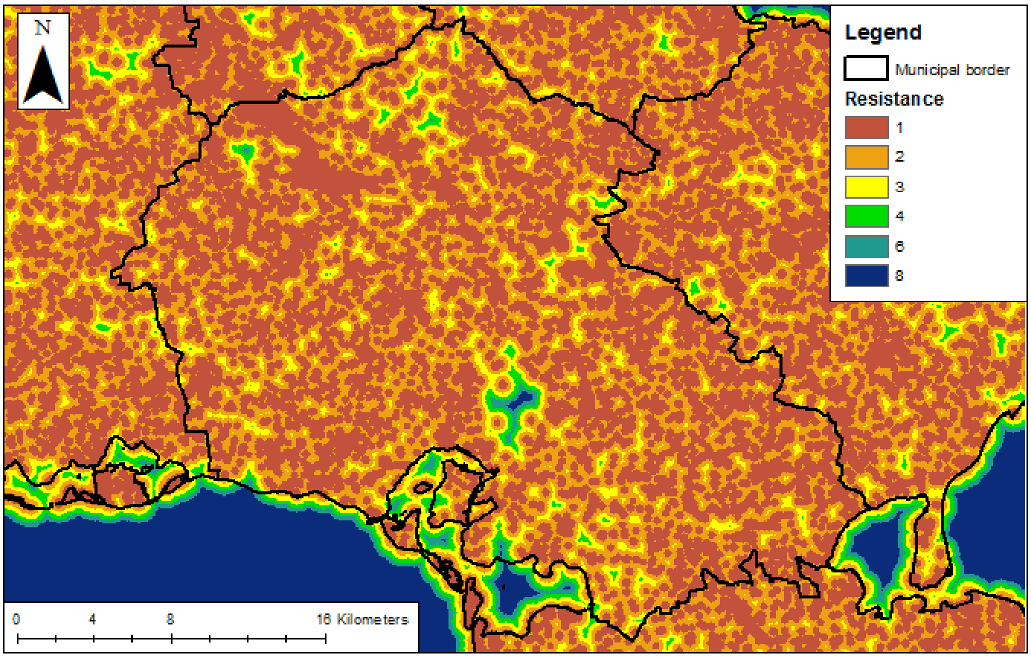

Figure 12. Cost raster for lake distance. Data from (GeoDanmark, 2016).

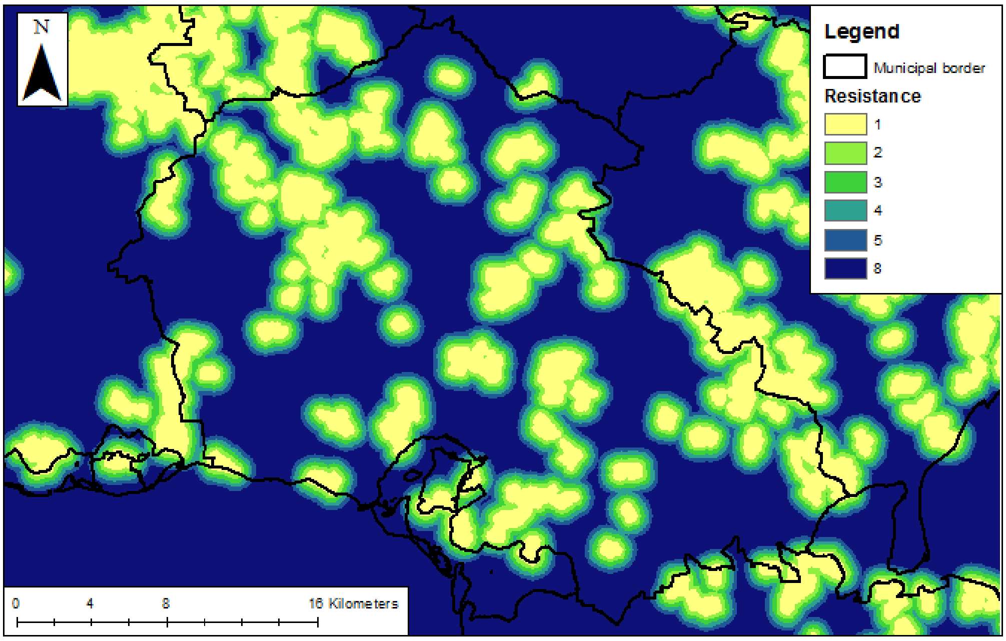

Figure 13. Cost raster for distance to deciduous or mixed forest. Data from (GeoDanmark, 2016) and (Copernicus Land Monitoring Services, 2016).

The road factor included a range of different road types as well as railroads. However, some roads are a larger obstacle than others and it was necessary to reflect this in the resistance layer. Highways were set to be impenetrable. Ray, et al. (2002) in a cost-surface modelling study using amphibians, set highways to be impenetrable on the basis that traffic intensity is highly correlated with mortality in amphibian species. In addition, municipalities would most likely not want to include a corridor that crosses a highway in their mapping of ecological corridors, given how unsuitable highways are for crossing of animals. Situations where the municipality intended to place a fauna passage at the crossing point could be an exception, but would be better suited for a separate analysis. Ray, et al. also write that railroad tracks have been shown to strongly affect some amphibian species, and hence assigned railroads as impenetrable in their study. However, a study of the effects of railroads on amphibians found that mortality depended on the agility of the species (Budzik & Budzik, 2014). They specifically found that the agile frog, as its name might suggest, was among the species agile enough to jump over railroad tracks, and found no occurrences of agile frogs killed in attempted railroad crossovers within the duration of their study. This should not be taken to imply that railroads are completely safe for passage, as the conditions in their study cannot necessarily be transferred completely to the local Danish context. It does imply that railroads are not impenetrable, however. Finally, many of the roads in the GeoDanmark road data are very lightly trafficked, and some are not even used by motorized vehicles. In order to simplify classification, roads and railroads were either classified as impassable, high friction, or as sufficiently low friction to be excluded from consideration in the model. This evidently does not reflect the complexity of the real world where each individual road has an individual traffic intensity and thus its own resistance to movement. However, this simplification was chosen because there was not enough information available to reliably judge the relative impact from the traffic and intensity of each road class and translate it to a resistance score. 'Tertiary' local roads and any road types not used for motorized traffic were excluded as they experience minimal traffic. A cost surface for the roads factor was then generated by assigning a resistance score for each cell depending on whether a road was present within the cell as well as the type of road (See Figure 14).

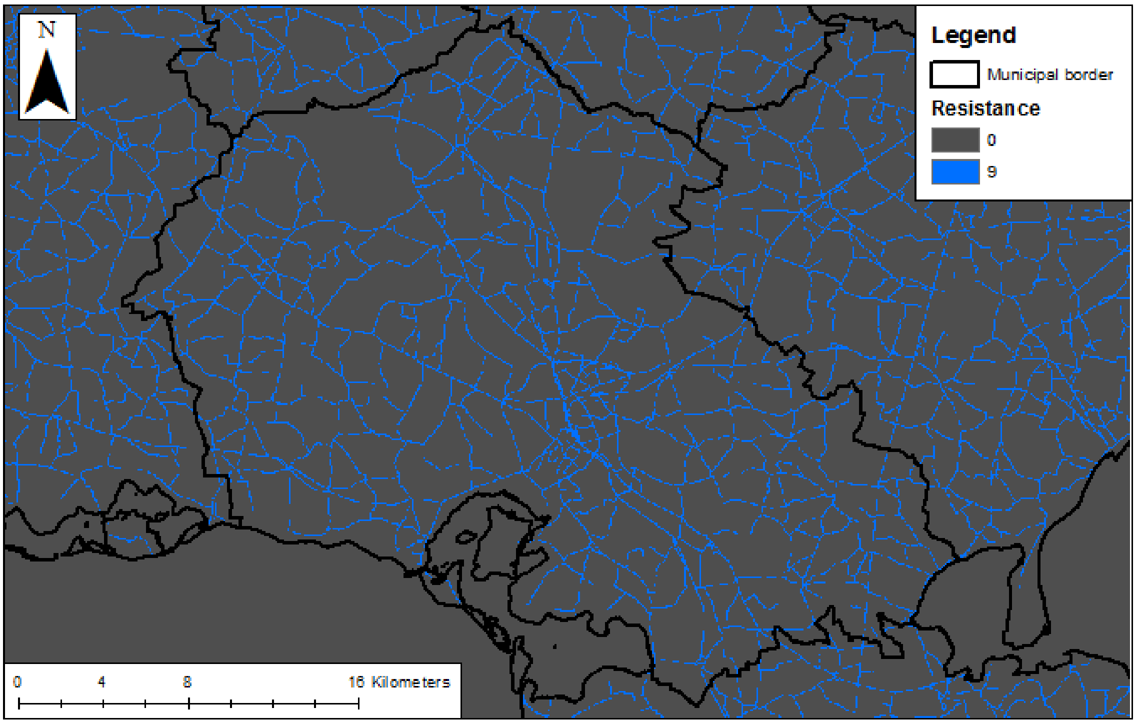

Figure 14. Cost raster for roads. Note: The impassable highway is not included in the image. Data from (GeoDanmark, 2016).

For assigning resistance values to the factor classes, a rating scale from 1 to 10 was used. Inspired by a rating scale used by Beier, et al. (2009) the values 1-3 represented the most preferred habitat, 6-7 indicated areas of limited worth as habitat and 8-10 indicated avoidance. The AHP rating method is a useful way to compile the opinions of multiple experts, and could also have been used in this project where only a single person rated the classes. As mentioned in chapter 6, as long as classes are in the correct rank order, the modelled corridors are robust to uncertainty in the point estimates of resistance. The rank orders for the four factors used in this project could be determined relatively precisely however, and thus the AHP approach would not have contributed much.

The resistance scores assigned to the land cover layer can be seen in Table 1. Five different values were used. All artificial surfaces and land cover types not found within Næstved were set to be impassable (NA). The most useful habitat, pure deciduous forest, was given the best score; 1. Mixed forest is close and was given the score 2. The score 7 was given to agricultural areas, with the exception of pasture. These agricultural areas provide few of the resources needed by the agile frog. The remaining land cover types which includes wetlands, pastures, coniferous forest and other semi natural areas, provide some value as habitat, and are given the score 4.

Class	Resistance	Uncertainty range
Artificial surfaces	NA	
Agricultural areas		
Non-irrigated arable land	7	5-8
Fruit trees and berry plantations	7	5-8
Pastures	4	3-6
Complex cultivation patterns	7	5-8
Land principally occupied by agriculture, with significant areas of natural vegetation	7	5-8
Forest and seminatural areas		
Broad-leaved forest	1	1
Coniferous forest	4	3-6
Mixed forest	2	1-3
Natural grasslands	4	3-6
Transitional woodland-shrub	4	3-6
Wetlands		
Inland marshes	4	3-6
Peat bogs	4	3-6
Salt marshes	4	3-6
Inland waters		
Water bodies	4	2-6
Coastal lagoons	NA	
Sea and ocean	NA	

Table 1. Resistance scores and uncertainty range for land cover.

For the road factor, the value 1 was assigned to areas with no roads. Highways were impassable, and railroads and other roads were given the value 9, to represent them being highly unsuitable for crossing. See Table 2 for a list of the resistance scores for the road factor and the two distance factors.

The resistance values for distance to forest increase linearly with distance at first, but increase at a higher rate from 800 meters and up. This choice was made to represent the fact that the quantity of egg-clutches in ponds have been shown to decrease exponentially with distance to forest (Wederkinch, 1988). All distances above 800 m were given the value 8 to represent avoidance of these areas. A similar relationship between distance and value was hypothesized for distance to lakes, and the classes for this factor were given the same values as distance to forest.

Class	Resistance	Uncertainty range
Presence of roads		
Highway	NA	
Other roads	9	8-10
Railroad	9	8-10
Non-road	1	
Distance to lakes		
0-200 meters	1	1
200-400 meters	2	1-3
400-600 meters	3	2-6
600-800 meters	4	3-8
800-1000 meters	6	4-8
Above 1000 meters	8	7-10
Distance to deciduous/mixed forest		
0-200 meters	1	1
200-400 meters	2	1-3
400-600 meters	3	2-6
600-800 meters	4	3-8
800-1000 meters	6	4-8
Above 1000 meters	8	7-10

Table 2. Resistance scores and uncertainty range for roads, distance to lakes and distance to deciduous/mixed forest.

The final cost raster had to be generated by adding the four factors together as a weighted sum. The weights must sum up to 100%. Because the value 9 for roads, while fitting the rating scale, did not seem to fully reflect the high avoidance and mortality associated with roads, the road factor was given a very high weight of 50%. The remaining three factors all seemed somewhat equal in importance. However, because the agile frog has been shown to exist in areas without forest, the two other factors were judged to be 1.5 times more important than distance to forests. This provided the final weights shown in Table 3.

Factor	Weight	Uncertainty range
Roads	50%	25-75
Alternative:	25%	
Distance to lakes	18,75%	12,5-50
Alternative:	12,5%	
Distance to deciduous/mixed forest	12,50%	12,5-25
Alternative:	25%	
Land cover	18,75%	12,5-37,5
Alternative:	37,5%	

Table 3. Weights and uncertainty ranges for the four factors. The 'Alternative' values show the weights used in the alternative scenario used in the uncertainty testing.

The point estimates provided for resistance scores and factor weights are highly uncertain. However, the rank order of classes within the distance factors are certain to be correct, and the rank order within the road class is highly unlikely to be incorrect (although road classes were simplified). The rank order of classes with the land cover factor are based on the available knowledge of habitat use, and are most likely dependable. As mentioned in chapter 6, a correct rank order of classes and factor weights goes a long way to ensure a model that is robust to uncertainty. The factor weights used here are more uncertain, however.

It is clear that uncertainty testing of this model would provide valuable additional information about how dependable the model is. The purpose of the uncertainty testing is to evaluate the worst case scenario in which the model is found to be as inaccurate as it could plausibly be. This should involve the testing of multiple scenarios of different resistance scores and factor weights, as described in chapter 6. In this project, only one alternative scenario will be evaluated, which is sufficient to demonstrate the approach.

Uncertainty ranges for both class resistance scores and factor weights were estimated and are shown in Table 1, Table 2 and Table 3. These are intended to represent the biologically plausible range of values for these factors and classes. These uncertainty ranges are estimates themselves and are not guaranteed to contain the 'true' values. Because the point estimates and uncertainty ranges were not set by an expert in the species, the uncertainty ranges were set to be relatively wide, particularly the factor weights. In the alternative scenario, the rank order of factor weights were changed from [Roads > Lake distance = Land cover > Forest distance], to the alternative [Land cover > Roads = Forest distance > Lake distance], staying within the uncertainty ranges for the weights. No alternative resistance scores were tested.

Following the creation of factor-specific cost surfaces for the two scenarios, the final cost surfaces could be generated by overlaying the four cost surface layers and performing a weighted sum of their resistance scores. The cost surface for the proposed scenario can be seen in Figure 15.

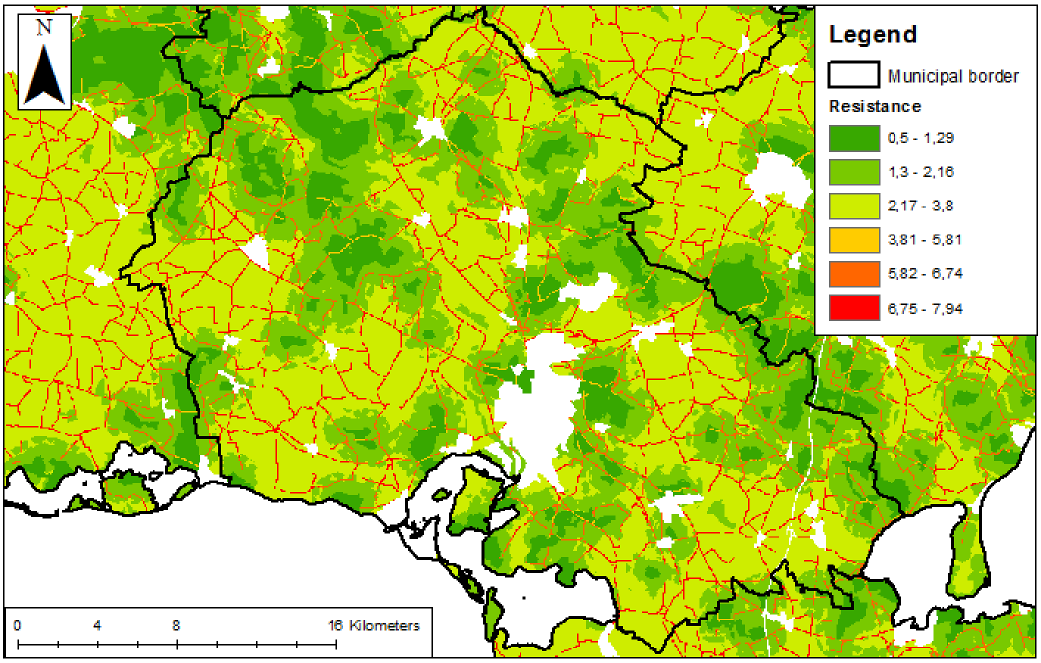

Figure 15. The final cost raster created using the proposed factor weighting. Values are divided into six classes using natural breaks. Data from (GeoDanmark, 2016) and (Styrelsen for Dataforsyning og Effektivisering, 2014).

7.6 Step 6: Define locations to connect

In this subchapter the method of selecting the termini to connect will be described. Following this, the approach used to decide which of these termini to connect with each other will be described.

7.6.1 Pin pointing habitats (or terminal areas)

The termini to connect had to be defined. A couple of options were available. Areas presently populated by the agile frog could be used as termini, but doing so would be missing an opportunity to provide better conditions for dispersal to uninhabited areas in the municipality. For this project, potential agile frog habitats would serve as the termini to be connected by corridors. The adjective 'potential' is used here to indicate that habitats will be determined on the basis of their habitat suitability, not whether these areas are presently populated by the species. For ease of writing, the word 'habitat' will be used henceforth to refer to these areas of high habitat suitability. By establishing or protecting corridors between all areas suitable for being populated, connectivity for the species can be improved throughout the municipality, and not just between the currently populated areas.

Since the cost raster generated in the previous step represents habitat quality, it makes sense to determine habitats on the basis of that dataset. Since high values on the surface raster indicates areas with low habitat quality, the habitats should be derived from areas with low resistance values. Setting a threshold value, below which all cells are treated as habitat is one approach, but has issues that will need to be solved. Expert involvement in choosing the threshold of sufficient habitat quality would be ideal. Alternatively, the threshold value could be determined on the basis of the available knowledge on habitat from literature. While a resistance scale was used in which the values represent something meaningful about the habitat quality, the subsequent weighing of the factors can result in a cost surface that does not accurately represent the meaning of the 1-10 scale. This is the case for the cost surface generated for this project. Here, no parts of the landscape have a value above 8, even though at least the roads should have a value of that severity. The values in the cost surface should instead be interpreted as relative cost for moving through different landscape elements (as mentioned in chapter 6). This should be taken into consideration before using the meaning of the 1-10 scale to guide the setting of the threshold. Instead, it might make more sense to compare cost surface values with the areas in the landscape which are known to be important for habitat. This was the approach taken in this project.

The literature suggests that the best habitats are in the vicinity of deciduous forests, which is reflected in the cost surface. This provided a useful guidance for the setting of the threshold value. Considering the habitats as roughly extending about a kilometer away from deciduous forests seemed sensible, as it would correspond to the distance the frogs are generally willing to depart from forests in order to spawn.

Patterns became easily visible in the cost-raster when the values were divided into six classes using natural breaks. Table 4 shows the six classes, listed from lowest to highest cost, although the characterizations of them were not entirely clear cut and had some overlap. The classes are the same as used in Figure 15.

6 classes identified with natural breaks
1. Deciduous/mixed forest
2. Area within close distance of deciduous/mixed forest
3. Areas of medium quality that could be characterized as the matrix in the landscape
4. Road in good habitat (Rare class)
5. Road in good habitat
6. Road in the matrix

Table 4. Cost surface divided into 6 classes using Natural Breaks.

Class number 2 could be seen as a potential threshold value, but a problem that is evident here is that many of the high quality habitat areas contain intersecting roads that are not suited as habitat themselves. This is an issue that could potentially arise in cost surface models for other species as well. Although the presence of roads does reduce the value of a habitat, their presence is not likely to render the habitats valueless. However, setting a threshold in the cost surface that leaves out areas with roads will cause habitats that are connected in the real world to be split up when the habitat polygons are generated later. On the other hand, a threshold high enough to include roads within good habitat would create some habitats that are clearly unsuitable (see Table 4). A method was needed that could include the value-diminishing effect of roads in the selection of habitats, without splitting up the habitats.

A method has already been described in chapter 6. The solution was to assign to each cell the average cost value of the cells found within a 500 m radius. The 500 m value was chosen because it represents the radius of the agile frog's home range (Fog & Hesselsøe, 2007). Thus the value of each cell becomes, in a sense, a measure of the value of a potential home range centered on this cell. The result of this operation was that clearly distinct areas of good habitat became perceptible, with roads no longer breaking up the habitats. On the basis of the new 'mean-cost raster', areas with a value below 1,8 were selected as habitats. A threshold value only slightly higher would have caused some habitats to melt together into agglomerations with highly irregular shapes, whereas the value 1,8 created habitats that closely corresponded to the desired 1 km buffer zones around forest patches and forest patch clusters.

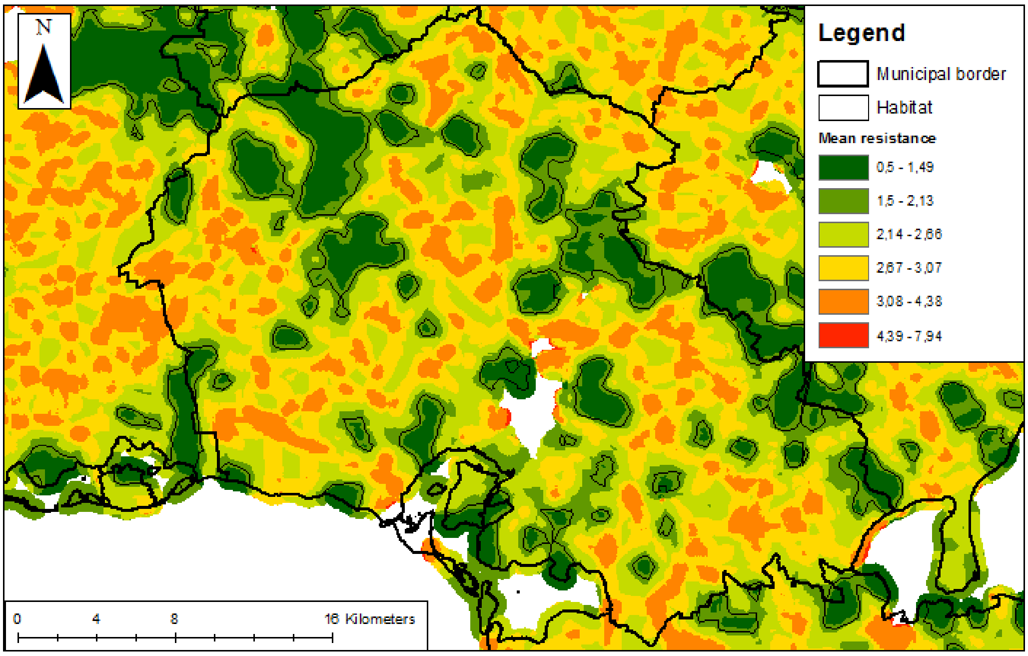

Figure 16. Mean-cost raster generated by, for each cell, taking the average cost within a 500 m radius in the original cost raster. The generated habitats are visible in the areas with lowest cost.

A number of habitats created with this approach were too small to realistically be suitable as habitat. The home range of the agile frog is listed as "approximately 1 km" (in diameter) in the handbook for Annex IV species (Fog & Hesselsøe, 2007). This corresponds to 0,79 km^2 home range, although it is fair to assume that not any patch shape of that size would be suitable, since a highly elongated patch would suffer from edge effects to a high degree. Nevertheless there is no easy way to account for shape, and elongated patches are rare, so the shape factor is ignored. Since the number in the handbook is listed as an approximation, a municipality might choose to include habitats with an area slightly below this, to ensure every possible habitat in the study area is accounted for. Alternatively, a municipality might choose to use the listed number, in order to focus on the habitat patches that are most important in the landscape. For this project, areas with a size below 0,1km^2 were removed. This was an arbitrary decision that could have been improved. However, the smallest habitat used in this study is 0,49 km^2, which is not unreasonable to include if one wants to consider all possible habitats.

A decision was made to focus on six habitats in the south-west corner of Næstved (see Figure 17). One of the patches is located mainly outside the municipal border. The six habitats were sufficient to demonstrate the principles of this analysis.

7.6.2 Determining which habitats to connect

Following this, it was necessary to determine which habitats should be joined together with corridors. Different approaches could be taken here. The most extensive and work intensive approach would be to create corridors between each and every habitat. However, municipalities would be unlikely to be interested in a collection of corridors that take up more space than necessary, which would limit spatial planning flexibility. A network of corridors that provides sufficient connectivity without taking up unnecessary space would be preferable. If taking the extensive approach, it would therefore make sense to evaluate which corridors are most important, and narrow the corridors down to these. One way to do this would be to gradually remove the corridors whose corresponding least cost paths have the lowest accumulative cost, while skipping any corridors whose removal would result in a patch becoming unconnected. The end result of this approach would be a near-optimal network of habitats connected by a minimal amount of corridors. To actually achieve a truly optimal network, it would be necessary to use some sort of algorithm that can properly calculate the optimal network. With that said, a minimal amount of corridors may not always be the best choice for achieving the intended conservation or biodiversity goals, which is very important to emphasize.

Having a number of corridors, perhaps in the range of 3-4, extending from each habitat was deemed desirable as this would benefit the overall connectivity of the network. One option considered was to connect each habitat with its 3-4 nearest habitats. Alternatively, to connect to all habitats within a fixed Euclidean distance. The problem with these two approaches is that some habitats found might be located behind another close habitat. Instead of trying to design a corridor that crosses through the intervening habitat, it makes more sense to use the intervening habitat as a separate terminal that connects to the outer habitat.

The final approach, which seems to be an approach that would be useful for municipalities, is to determine which habitats to join together using Thiessen polygons. In a space containing a number of points or objects, Thiessen polygons are subdivisions of the space, where each polygon is based around a point and includes all areas of the space that are closer to that point than the other points (Esri, 2014). In ArcGIS, Thiessen polygons can be generated for points representing the centers of the habitats, but tools can be downloaded that allow Thiessen polygons to be generated for the habitat polygons (see for example (UNTGeography, 2013)). The approach here is intended to be nothing more than a handy rule of thumb for deciding which habitats to connect, which can be seen as a policy decision rather than a matter of science. The resulting Thiessen polygons for each habitat do not border polygons for habitats that lie behind other habitats, solving the problem mentioned earlier. Furthermore, a large shared border with another Thiessen polygon tends to indicate that the neighboring habitat is close or isolated in that direction, making it a good candidate for connection. Occasionally a Thiessen polygon may share borders with a large number of polygons, and in that case it may be sensible to restrict each habitat to a specific number of corridors, such as 3-4. The habitats to connect to can be selected by taking the ones with the largest shared Thiessen polygon border first.

The habitats, the Thiessen polygons, and the chosen connections can be seen in Figure 17.

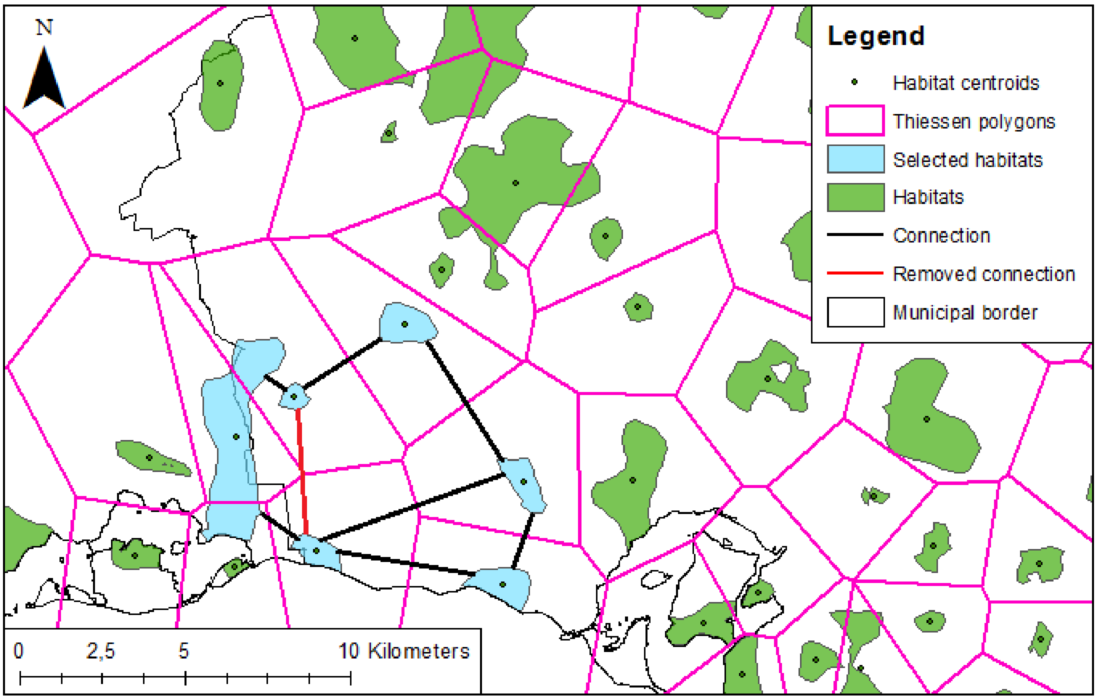

Figure 17. Blue polygons are the selected habitats. Black lines show which polygons were to be connected. The red connection was removed due to its corridor being redundant. Data from (GeoDanmark, 2016).

7.7 Step 7: Delineate corridors

The next step was to map the ecological corridors. The corridors themselves are mapped as least-cost corridors, but least cost paths are also generated to assist in evaluating the corridors, as described in step 8.

Cost distance layers were first generated using the cost raster (not the mean-cost raster) for each of the six selected habitats. Using the cost distance layers, least cost paths were created between the selected habitat pairs. This is the 1 cell wide path through the landscape between two terminals with the least accumulative cost. Then, using the cost distance layers of each habitat pair as input to the Corridor tool, a layer is created where the value of each cell is the accumulative cost of the least cost path connecting the two habitats while passing through the cell. By excluding cells with a value above a chosen threshold, corridors between the two habitats can be created, in which all cells within are part of a collection of cheap routes that are more permeable than the routes that were excluded.

It is recommended to set threshold values that ensure that the corridor is of sufficient width and free from inhibiting bottlenecks. The matter of how wide these corridors should be is not easy to answer. As mentioned in chapter 6, a common approach is to include the cells with a max value equal to the least-cost-path accumulative cost + n%, with n=1 for example. This is an easy approach that can be used to set thresholds for multiple corridors according to the same standard, although determining a threshold value with a scientific basis is not straightforward. Ideally, however, each corridor should be judged individually, and the cost threshold for being included in the corridor should be set in a way that ensures a sufficient width for the organism to travel through. If the corridor is to be used as habitat rather than as a conduit, the width of the corridor should be substantially more than the organism's home range width to accommodate corridor dwellers (as discussed in chapter 5).

For this analysis, a threshold of LCP-cost + 5% is chosen, as it created corridors of a subjectively reasonable width. One particular corridor became far too wide, and a threshold of LCP-cost + 1% was chosen here instead. The resulting corridors from this analysis are not intended to be final suggestions, as some are too thin or wide and others split into two paths, which may not be ideal. However, these corridors provide a foundation for demonstrating how to subsequently evaluate the corridors and test the effect of uncertainty in the parameters used to generate the cost-raster. The approach that should be used in a real world situation would consist of three steps. The first is to set a threshold value for the individual corridor that ensures a width equal or higher to what is biologically necessary. The second is to consider removing one split end if the corridor splits into two paths. The third is to include isolated cells within the corridor that had values above the threshold in the final corridor design, to make the design more aesthetically pleasing and easier to manage. This may not be a good idea if isolated the cells consist of urban areas or similar, however.

One of the corridors that was generated in the process described above crossed through a nearby habitat due to the lower resistance here. This corridor was superfluous and could be removed, because the two habitats it connected would already be connected through the intervening habitat. The removed corridor can be seen in Figure 17.

7.8 Step 8: Evaluate the corridors

The corridors that were generated need to be evaluated to determine which ones are most valuable and which ones are not valuable at all. As mentioned in chapter 6, assessing the value of the corridors generated through least-cost modelling can be a challenge. The main issue is that although the best corridor between two habitats may have been found, there is no guarantee that this corridor is good enough to provide the desired connectivity.

In this analysis, the corridors will be evaluated using only a single measure. The measure is a ranking of the least cost paths in terms of their 'Equivalent Euclidean Distance' (EED), a measure created for this project that will be described further ahead.

EED is an attempt to translate the accumulative cost values of the least cost paths into a Euclidean distance measure such as meters. The purpose is to evaluate how well the corridors serve as conduits, i.e. areas where an individual organism is able to disperse through within a relatively short time period, without actually inhabiting it. The long term dispersal distance of the agile frog is listed as 'above 1 km' in the Annex IV handbook, so it is prudent to determine if any corridors are significantly above 1 km long (Fog & Hesselsøe, 2007). However, if the length of the corridors were measured as Euclidean distance in meters, one would be ignoring the very purpose of least-cost modelling, which is to perceive the landscape as a heterogeneous space where each individual area provides a distinct resistance to movement. Therefore, the accumulative cost value of the corridors should be incorporated into the distance measure. The accumulative cost of the least-cost path is used as a proxy. EED is based on the assumption that the long term dispersal distance of the organism described in the scientific literature has occurred or been estimated as though it occurred in a corridor well suited for dispersal, i.e. a low cost corridor.

The equivalent Euclidean distance is measured the following way: The accumulative cost and the Euclidean distance in meters of all least-cost-paths are recorded. Cost per meter is calculated for each path. The path with the lowest cost ratio is used for calculating EED. The EED for each path is then calculated as the accumulative cost of the path divided by the cost ratio. For the path with the lowest cost-ratio, the EED is the same as its Euclidean distance, but EED is always higher than Euclidean distance for the other paths.

The rationale for using EED is as follows: First, it provides a distance in meters rather than cost-units, which is necessary to evaluate the corridor's value as a conduit, while at the same time more accurately representing the relative resistance of the corridors than Euclidean meters would. Second, the long term dispersal distance of the organism found in scientific literature is likely based on a dispersal that took place through a low cost corridor, thus making the cost-ratio found using the described method the most accurate estimate of the cost incurred per meter in the aforementioned low cost corridor. Third, if the assumption made in the second point is false, EED instead provides a conservative measure that underestimates the Euclidean distance in meters the organism can travel. A conservative measure reduces the chance of a corridor unsuitable as a conduit being regarded as suitable, thus making it the safer measure. If a corridor is found unsuitable to act as a conduit corridor, it can instead be treated as habitat corridor, which would normally entail ensuring that its width and other habitat qualities are sufficient.

7.9 Step 9: Validation and uncertainty analysis

As described in chapter 6, it is recommended to attempt to validate the resulting corridors and the generated cost surface using empirical data. Unfortunately, no empirical data was available for this purpose. Instead, the focus will be purely on uncertainty analysis, which will help reveal the effect of uncertainty on the corridors resulting from the analysis. As described in Step 5, one alternative scenario using different factor weights will be evaluated.

The corridors generated using the alternative cost-raster will be compared to the corridors generated with the original proposed cost-raster using two measures. The first measure is the difference in mean cell cost for each corridor-set (i.e. proposed corridor and alternative corridor), using the alternative cost-raster for both. The second measure is the percentage of the alternative corridor overlapped by the proposed corridor for each corridor-set. These two measures will show severe it is if the proposed corridors are wrong and the alternative corridors are more accurate. The two measures reveal how much the location of the proposed corridor would diverge from the 'real' alternative corridor, and how much worse the mean resistance would be in the proposed corridor compared to the 'real' alternative corridor.

The habitats generated using the proposed cost raster will also be used for the uncertainty analysis. This is slightly counter-intuitive, because if the alternative cost raster is correct, then the habitats might be in different places. However, it is necessary to use a common habitat dataset in order to compare the two scenarios in terms of corridor placement and mean cost, because otherwise the end points could be in different places and even different quantities. Since the proposed cost-raster is judged to be the most likely to be accurate, it makes sense to use that as a basis for pin pointing corridor in the uncertainty analysis.

7.10 Conceptual GIS model

This subchapter will present a conceptual model of the steps that were taken in ArcGIS in order to generate the final GIS layers and the measures used in the evaluation and uncertainty analysis of the corridors. Figure 18 shows how the input data is transformed to new GIS layers and how these layers are combined and altered to create other layers. The processes taking place in the model will be described below.

The first part of the model involves delineating the corridors. It starts with the four GIS datasets Corine Land Cover, and the Geodanmark datasets Vejmidte (English: *'road center'*), Jernbane (*'railroad'*) and Soe (*'Lake'*). These datasets have to be converted into four cost surfaces and one raster layer showing the locations of impassable highways. All the datasets are in vector format and thus need to be converted to raster format. The land cover cost surface is created by reclassifying the land cover classes into resistance values. Deciduous and mixed forests are separated from the CLC dataset, and along with the Lake dataset, become transformed into two datasets where each cell shows the Euclidean distance to the nearest forest or lake, respectively. The forest distance and lake distance cost surfaces are then generated by reclassifying the distances into resistance values. The road cost surface is created by assigning a resistance score to each cell that contains a road or railroad that fits the criteria set forth, while the highway raster is generated by assigning a NoData value (which is impassable) to cells with a highway and 0 elsewhere. The final cost raster had to be generated by adding the four factors together as a weighted sum, after which the highway raster was added to give highways a NoData value.

The final cost surface is transformed into a mean-cost surface by assigning to each cell the average cost value of the cells found within a 500 m radius. Cells with a mean-cost value about 1,8 were removed, and the remaining contiguous zones of cells were turned into polygon habitats. Habitats below 0,1km^2 in size were removed.

In order to generate least-cost paths and corridors, cost distance layers were generated for each of the six selected habitats using the final cost surface. The cost distance layers and habitats are used together to create the least-cost paths, while the two cost distance layers for each habitat-pair are added together to create the least-cost corridors. The lengths and accumulative costs of the least-cost paths are used to calculate EED values in the evaluation step.

The uncertainty analysis begins with the creation of an alternative cost surface and alternative least-cost corridors by applying different resistance values to the cost surfaces in the above process. The corridor overlap rasters are created by assigning the values 1 (corridor present) and NoData (corridor absent) to the proposed and the alternative corridor rasters, and then adding them together. The cell counts of the corridor overlap rasters and the alternative corridors are then used to calculate the percentage of the alternative corridor that is overlapped. To calculate the difference in mean cost (using the alternative cost surface) between the alternative and the proposed corridors, the cells in the alternative cost surface that are overlapped by the proposed corridor are extracted. The difference in the mean cost of cells is then calculated.

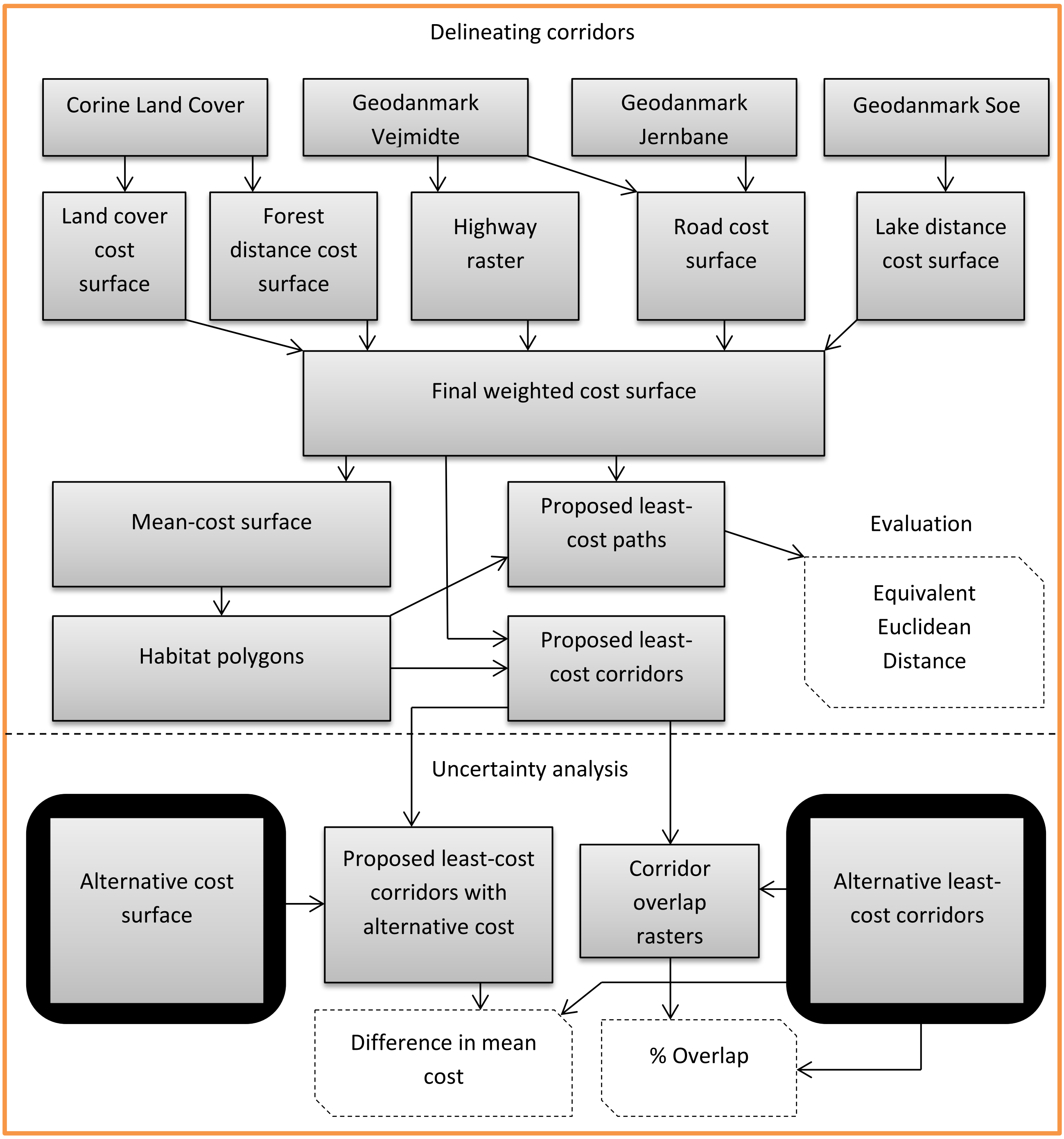

Figure 18. Grey boxes are GIS layers. Grey boxes within black boxes are GIS layers created using the alternative factor weights. Boxes with dotted lines are not GIS layers, but numeric measures used in the report.

8 Results

In this chapter the results of the analysis will be presented. The results include the proposed corridors that have been designed, the evaluation of these and the uncertainty analysis.

8.1 Proposed corridors

Figure 19 shows the proposed corridors and the least-cost paths within them. Table 5 shows the thresholds used to limit the extent of the corridors as well as how the thresholds were calculated. As can be seen, there are significant size differences between the corridors. If Corridor 3-5 (which connects habitat 3 and 5) had used LCP Cost + 5% instead of + 1%, the corridor would have been even wider. This suggests that it is not recommendable to limit a range of corridors using a threshold based on a static percentage of the least-cost path cost. Corridor 1-3 also suffers from a considerable bottleneck near habitat 1. To avoid situations like that it is recommended to incrementally increase the threshold value until the bottleneck disappears (as discussed in chapter 6). Corridor 2-4 noticeably splits into two highly separated corridors. A straight corridor is not possible because an urban area borders directly up to habitat 2. In a planning situation it may be worth considering removing one split end, or perhaps connecting habitat 1 to habitat 4, since one of the split ends uses habitat 1 as a stepping stone. Corridor 3-5 also noticeably splits into two; a situation also caused by the presence of an urban area. In general the different corridors seem to follow quite straight lines between the habitats. However, only a limited set of corridors have been modelled in this project and such a result may not always occur.

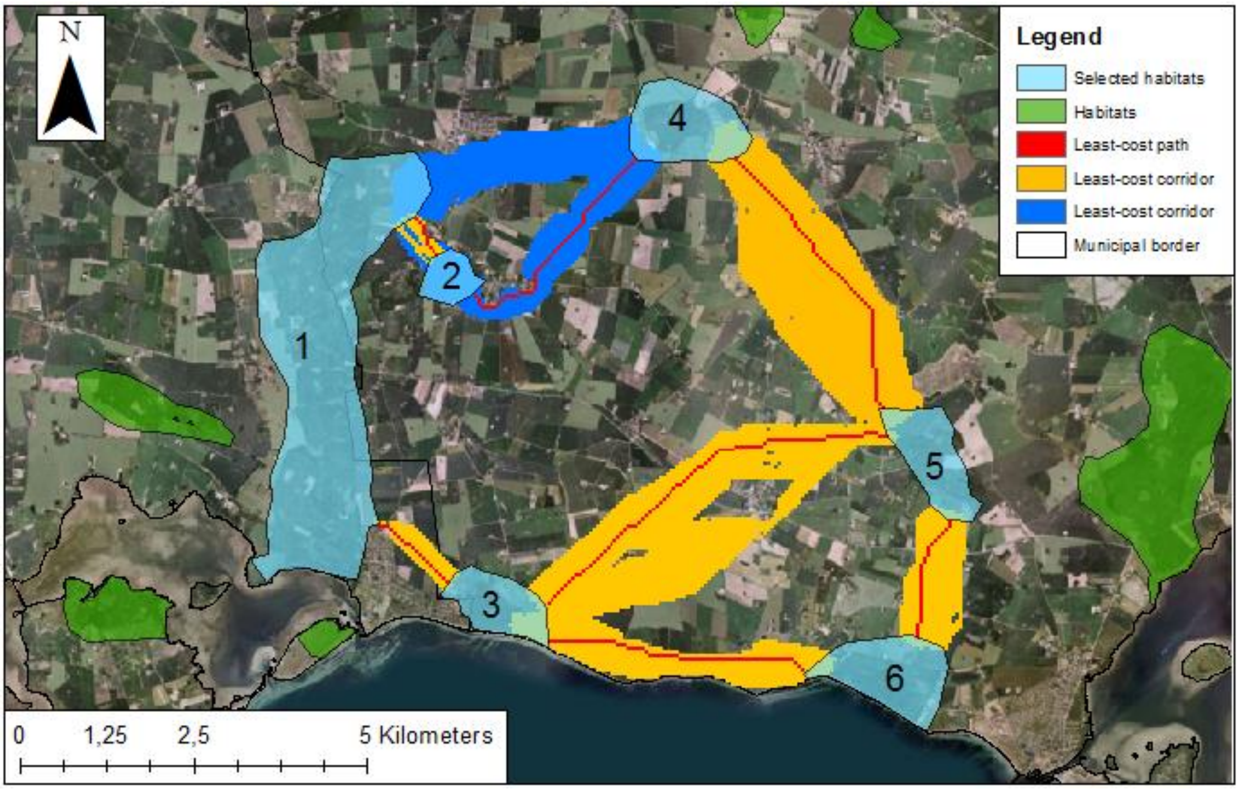

Figure 19. The image shows the proposed corridors and the six habitats they connect. The least-cost paths are also visible. Due to Corridor 2-4 overlapping Corridor 1-2, the former has been given a blue color so the corridors can be distinguished from each other. Data from (GeoDanmark, 2016) and (COWI, 2014).

Corridor (Proposed)	LCP Cost	Corridor threshold cost	% increase of LCP Cost
1-2	1518,519	1594,4	5%
1-3	2969,539	3118	5%
2-4	9052,865	9505,5	5%
3-5	14791,51	14939,4251	1%
3-6	9211,136	9671,6928	5%
4-5	11033,96	11585,658	5%
5-6	4444,667	4666,90035	5%

Table 5. For the proposed scenario shows the accumulative cost of the least-cost paths connecting the numbered habitats, the cost thresholds used to limit the corridor size, and the percentage of LCP Cost used to calculate the threshold.

Figure 20 shows the modelled corridors overlaid by the ecological corridors mapped in Næstved municipality's municipal plan (but unedited since being taken over from the previous county). It also shows the streams in the landscape in order to show how closely related these features appear to be. As can be seen, the municipal ecological corridors deviate widely from the modelled corridors. This should come as no surprise since the appointments in Næstved's plan seem to have been selected using a structural approach, largely based on the paths of streams, whereas the corridors modelled in this project have been exclusively designed to accommodate the agile frog. This illustrates how the functional landscape corridors used by a specific species can potentially end up being neglected if its needs and behavior are not explicitly taken into account when designing corridors.

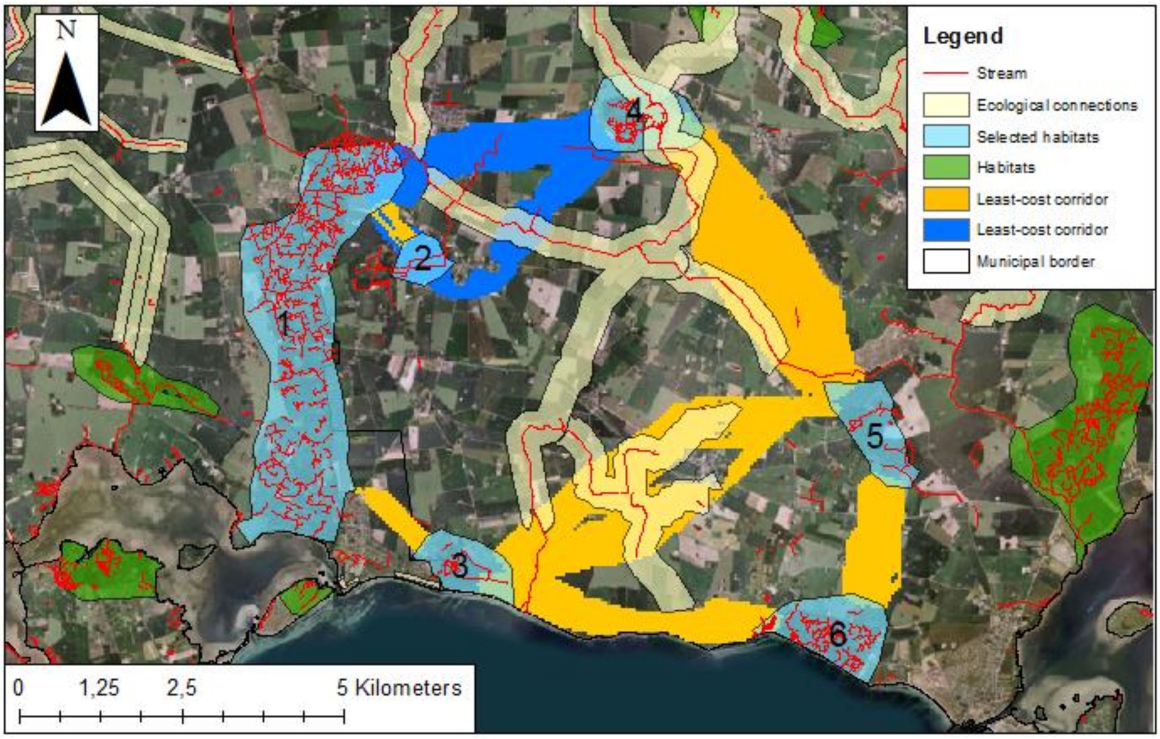

Figure 20. The image shows the habitats and proposed corridors. In addition, it shows the ecological corridors in the municipal plan. It also shows a stream layer to visualize how closely connected they are. Data from (Erhvervsstyrelsen, 2015b), (GeoDanmark, 2016) and (COWI, 2014).

Figure 21 shows the least-cost paths of the proposed corridors, overlaid on the cost-raster. The ideal routes for the agile frog do not entirely follow the shortest straight line between each habitat, but bend to avoid roads and cities and take advantage of low cost areas.

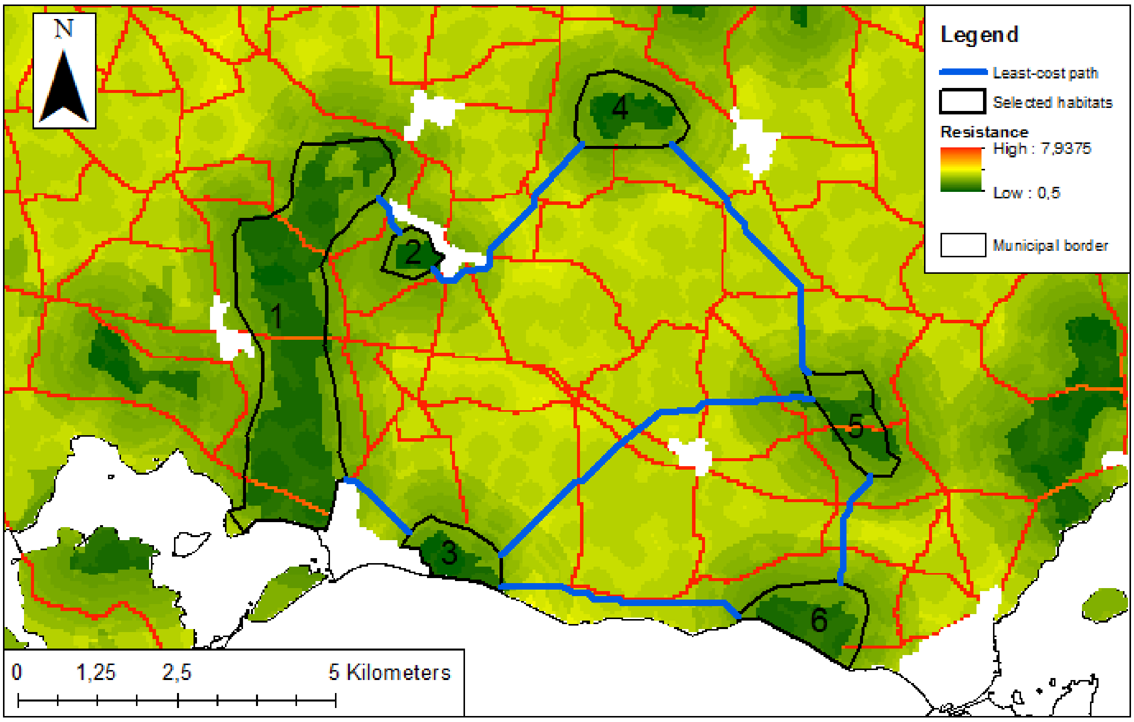

Figure 21. Shows the least-cost paths between the habitats on top of the cost-raster. Data from (GeoDanmark, 2016).

8.2 Evaluation of corridors

To evaluate the proposed corridors, their relative distances will be measured and ranked, and their value as conduit vs habitat corridors will be discussed.

Table 6 shows the accumulative cost, Euclidean distance, and equivalent Euclidean distance of the least-cost paths in the proposed scenario. As can be seen, the paths are ranked in the same order regardless of whether their length is calculated in Euclidean distance or EED. However, the relative distances of the different paths do change when their EED is considered. The main purpose of calculating the EED for the least-cost paths has been to evaluate the corridors' potential as non-habitat conduits using a measure of relative distance. It is the nature of the EED measure that it provides longer distances than a normal Euclidean distance. Thus the dispersal distance of the agile frog, as described in the Annex IV handbook, will be taken to represent an unusual occurrence that can be achieved only when the landscape provides minimal resistance. However, the dispersal distance of the agile frog is estimated to be 'Above 1 km', which is a highly imprecise estimate. Path 1-2 clearly fits within this estimate and should in principle be able to function as a conduit. The following two paths in the ranked list could possibly also serve as conduits. After that, the EED increases drastically, and it starts becoming questionable whether the frog can traverse these corridors without needing considerable resting periods and habitat resources. If those corridors are to serve as intermediate habitat connecting two main habitat areas, care should be taken to ensure that the corridors have the sufficient width and ecological quality to enable this. Part of this would be to ensure that the corridors have a width that is 'substantially larger than a home range width' (as mentioned in chapter

6). The home range of the agile frog is estimated to be approximately 1 km in diameter (Fog & Hesselsøe, 2007); a width that should ideally be enforced throughout the corridor, along with an additional buffer.

Path (proposed)	Accumulative cost	Euclidean length in meters	Ratio (cost per meter)	Equivalent Euclidean Distance (EED) in meters
1-2	1518,52	643,86	2,36	662,63
1-3	2969,54	1295,81	2,29	1295,81
5-6	4444,67	1850,25	2,40	1939,51
2-4	9052,87	3537,99	2,56	3950,38
3-6	9211,14	3893,58	2,37	4019,44
4-5	11033,96	4422,86	2,49	4814,86
3-5	14791,51	5845,34	2,53	6454,53

Table 6. The table shows the accumulative cost and Euclidean length of the least-cost paths in the proposed scenario. It shows the ratio of cost per meter and the lowest ratio is highlighted with orange. For each path the accumulative cost was divided by the highlighted ratio in order to calculate the EED. The paths are ranked from shortest/cheapest in ascending order.

8.3 Uncertainty testing of corridors

An alternative scenario where the cost raster has been generated using different weights will be analyzed. The corridors of the proposed scenario will be compared to the corridor of the alternative scenario, and the impact of uncertainty in the cost surface generation can be evaluated. As proposed by Beier, et al. (2009), two measure will be used for this purpose: The percent of an alternative corridor overlapped by the corresponding proposed corridor, and the difference between the mean resistance of an alternative corridor and the mean resistance of the corresponding proposed corridor (using the alternative cost surface). The measures will show how severe the consequences are for the quality of the proposed corridors if the alternative model turns out to be correct.

Figure 22 shows the alternative and proposed corridors and the areas where they overlap. Table 7 shows the calculation of the threshold values for the alternative corridors. Only a few changes are perceptible in the corridors. Corridor 2-4 noticeably loses one of its split ends. Other than that, there are just a few slight bulges towards different directions, but no major changes in shape. With regards to the least-cost paths, some are almost identical, some vary slightly, and then there is LCP 3-5, which takes a completely different route around a city.

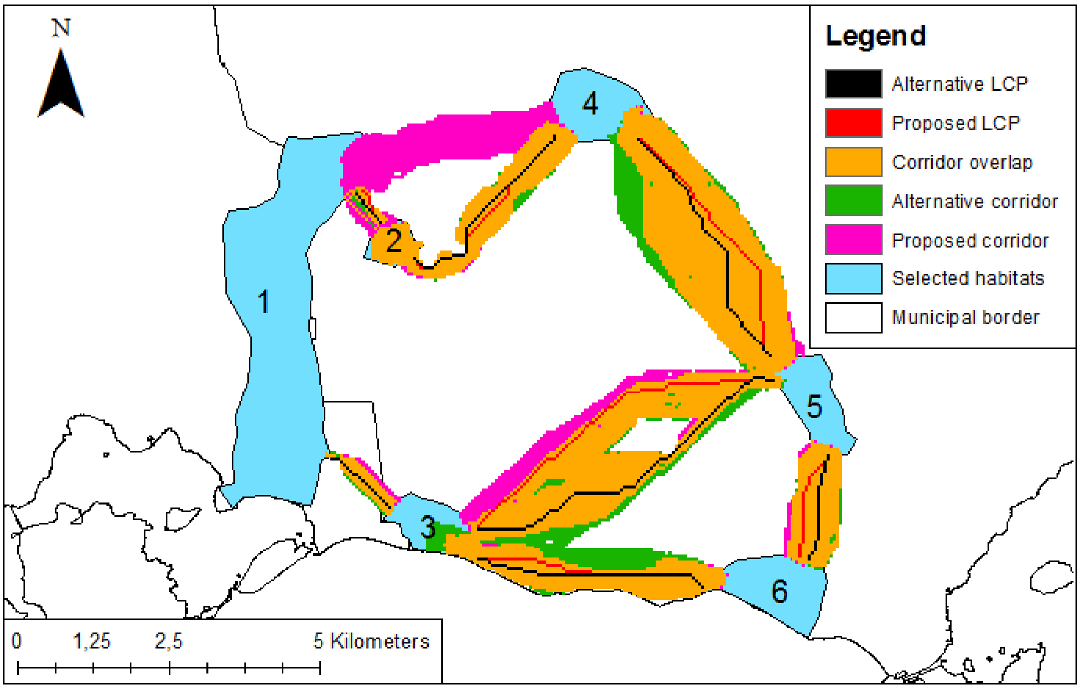

Figure 22. The figure shows the proposed and alternative corridors and their overlap. It also shows the two sets of least-cost paths.

Corridor (Alternative)	LCP Cost	1% increase of LCP Cost	% increase of LCP Cost
1-2	2446,581	2568,91005	5%
1-3	5007,232	5257,5936	5%
2-4	14833,53	15575,2065	5%
3-5	25538,69	25794,0769	1%
3-6	16281,58	17095,659	5%
4-5	19913,35	20909,0175	5%
5-6	7564,565	7942,79325	5%

Table 7. For the alternative scenario the table shows the accumulative cost of the least-cost paths connecting the numbered habitats, the cost thresholds used to limit the corridor size, and the percentage of LCP Cost used to calculate the threshold.

Table 8 shows how much each alternative corridor is overlapped by the corresponding proposed corridor. It varies from a minimum of 72,48 % to a maximum of 96,81 %, with a mean of 86,9 %. Table 9 shows the mean resistance of the alternative and proposed corridors as well as their difference. The mean difference was 0,01, which means that the proposed corridors have lower and hence superior mean resistances on average. Only two of the alternative corridors actually have a lower mean resistance. The differences are small, however, and the highest absolute difference was 0,049. It is important to remember that the alternative corridors consists of a selection of the paths with lowest costs between the two points, so one should be careful not to read too much into the higher mean resistance of the alternative corridors. The quality of a corridor also depends on other factors, such as whether the corridor has the right width, isn't affected by bottlenecks, and whether the cells are connected in a way that provides good routes. Only if the differences were much larger would it merit further analysis: High negative differences would suggest that the proposed corridors are of low quality given that the alternative scenario is true, and the first reaction to high positive differences should be to investigate if the calculation went wrong, since the least-cost algorithm is supposed to find the routes with the lowest cost.

Corridor	Alternative cell count	Overlapping cell count	% of alternative corridor overlapped by proposed corridor
1-2	79	67	84,81
1-3	157	140	89,17
2-4	941	911	96,81
3-5	2304	1910	82,9
3-6	1199	869	72,48
4-5	2690	2320	86,25
5-6	588	564	95,92
Mean:			86,9

Table 8. The table shows the % each alternative corridor that is overlapped by the corresponding proposed corridor. This is calculated using the cell count of each alternative corridor and the amount of cells where the proposed and alternative corridors overlap.

Proposed	Mean resistance	Alternative	Mean Resistance	Alternative mean minus proposed mean
1-2	3,519	1-2	3,516	-0,003
1-3	3,709	1-3	3,713	0,004
2-4	3,783	2-4	3,814	0,031
3-5	4,602	3-5	4,553	-0,049
3-6	3,983	3-6	4,016	0,033
4-5	4,484	4-5	4,51	0,026
5-6	4,062	5-6	4,09	0,028
Mean difference:				0,01

Table 9. The table shows the mean resistance of the cells in the proposed and alternative corridors as well as their difference. It also shows the mean difference. The resistance values are based on the cost surface of the alternative scenario.

9 Discussion

This chapter will contain a discussion of the accuracy of the results, the best way to implement an LCC analysis and a discussion of whether of the approach could be a useful tool for Danish municipalities.

Subchapter 9.1 will begin with a discussion of the accuracy of the conducted LCC analysis, covering the data quality, the underlying assumptions and the uncertainty analysis. On the basis of the lessons learned in the LCC analysis, the literature about LCC analysis, and the concrete issues facing Danish municipalities, subchapter 9.2 will seek an answer to the question of how best to implement LCC analysis in the planning of ecological corridors within the municipalities. Subchapter 9.3 will discuss the general advantages and disadvantages of LCC analysis, discuss the issues specific to the Danish municipal context, and finally compare the method to the two alternatives mentioned in chapter 4, with the purpose of answering whether LCC analysis is a useful approach for Danish municipalities.

9.1 How accurate are the results?

No attempts to validate the results with empirical data were conducted. Instead, this subchapter will deal with how the results could potentially have been affected by inaccuracies in data and uncertain assumptions. The uncertainty analysis will be discussed, which was conducted to obtain more information about how the uncertain weighting of factors could have affected the results. The face validity of the analysis will also be covered.

As mentioned in chapter 6, different types of validation exist. The simplest form of validity is Face Validity, which is achieved when the model and its behavior seems reasonable on the surface. If a model does not have Face Validity it is likely a sign of serious problems, but other than revealing highly flawed models it only provides weak validation. Assuming that the generated resistance surface is correct, the proposed corridors appear to be sensible in general. One potential problem can be seen near habitat 2, which is located directly besides an urban area and has corridors extending from it that almost hug the urban area (see Figure 19). It is possible that edge effects from the urban area could render these corridors less useful, although such an effect could not be deduced directly from the literature. Adding a buffer around urban areas and applying high resistance to them might be worth considering. Aside from this possible issue, the resulting corridors and least-cost paths seemed reasonable. Figure 21 shows how the paths avoid crossing cities and unnecessary roads while sticking to areas close to forests and lakes. The corridors themselves should not be considered to be the final designs, as some have bottlenecks and possibly unwanted split ends. The threshold values should be expanded iteratively until the corridors have a sufficient width and should then be trimmed where they are superfluous. However, the routes formed by the corridors have no apparent issues. They all follow quite straight paths between the habitats, except where presence of impenetrable urban areas forces them to diverge from their straight route. One exception was the redundant corridor between habitat 2 and 3 that was cut (see subchapter 7.7). This corridor bent from habitat 2 to habitat 1 in order to use the high quality habitat as a stepping stone towards habitat 3. Considering the four basic rules the route selection is based on; avoiding roads and cities, sticking close to lakes and deciduous forests, and keeping to suitable land cover; there is no reason to believe that straight corridors should be a problem in the given area. The model seems reasonable on the face of it, but having independent eyes to judge it would be better.

The uncertainty analysis revealed how an alternative resistance surface, with factor rank order changed from [Roads > Lake distance = Land cover > Forest distance] to [Land cover > Roads = Forest distance > Lake distance], affected the resulting corridors. The resulting alternative corridors were overlapped by the proposed corridors with a mean of 86,9 %. The corridors also had a mean difference in mean cell resistance of 0,01, which is miniscule. These results suggest the proposed corridor design, if it were to be implemented, would still provide roughly the same connectivity regardless of which model scenario was most accurate. However, only one alternative scenario was tested. The range of uncertainty in the resistance scores and factor weights permit many other possible scenarios to exist, and ideally a much larger array of scenarios should be tested.

There is a lack of detailed knowledge concerning the agile frog and its habitat preferences, dispersal distance and the energy costs of moving through different landscape features. More detailed knowledge about these topics could have improved the model.

It can be questioned if the model includes all relevant factors. Other factors were considered, and the reasons for leaving them out were discussed in chapter 7. A high resistance buffer around cities could be added, but is not recommended unless studies are conducted or uncovered that document that cities have edge effects on corridors. Some landscape features providing microhabitats, such as living fences, might advantageously be added to the model.

The resistance surface relies on four factors derived from data affected by different kinds of inaccuracies. With regards to the accuracy of the data used for the four selected factors, the GeoDanmark data is a rather accurate dataset. There is a thematic classification error rate of 1-5% affecting object type and attribute classification, and the points used to register the lake polygons and road lines are never off by more than a few meters. The accuracy of this dataset should not be an issue. The Corine Land Cover dataset is more inaccurate however. It has a geometric accuracy of 100 m or better and a thematic accuracy of 85% or higher. It also only registers different land cover if it is larger than 25 hectares or linear features wider than 100 m. The dataset is also from 2012 and is thus outdated by some years. Supplementing the land cover data with more accurate and current GeoDanmark data would be sensible. The representation of forests and cities is likely to be accurate as these are relatively easy to distinguish in the landscape.

A number of assumptions that underlie the model have been mentioned throughout the report. The validity of these assumptions can have a large influence on the correctness of the modelled results. Gauging the validity of these assumptions is difficult. However, the realism of these assumptions and others that occur in LCC modelling will be discussed in subchapter 9.3.

In conclusion, the model’s face validity and uncertainty test support the accuracy of the results. The GeoDanmark data is highly accurate, and while the land cover has some inaccuracies and only represents the landscape on a somewhat coarse-grained scale, the representation of the important classes of forests and cities is likely sufficiently accurate. The major issues that could potentially shake the model’s foundation are the untested assumptions that lie under it. These include the use of habitat selection rules as dispersal route selection rules, the subjective rating of resistance for landscape features, and more. If these assumptions are trustworthy, the model results should be trustworthy as well. These assumptions will be discussed in subchapter 9.3.1.

9.2 What is the best way for municipalities to implement LCC analysis?

This chapter will discuss the best way to implement an LCC analysis as part of a municipality's planning of ecological corridors.

By following the nine step approach described in this report, a municipality will be able to thoroughly consider the many decisions that must be made and potential problems that must be avoided. The eight step approach described by Wade, et al. (2015) and mentioned in chapter 6 covers almost the same issues but emphasizes different things. It also does not include the selection of focal species within its steps, although a focal species approach has been recommended by some authors as a way to ensure that the corridor and conservation projects can accommodate a wide range of species and ecological processes (Lambeck, 1997) (Beier, et al., 2008b).The 16 questions concerning corridor modelling described by (Beier, et al., 2008b) and also mentioned in chapter 6 have been largely integrated in the proposed nine step structure, but a corridor designer should consider all questions during the design process to ensure that no potential flaws in the model are missed. The nine step structure proposed in this report is not a thorough guide to LCC modelling, but rather an overall structure for analysis. The theory described in this report and the methods applied in the development of the model can serve as inspiration, but any project involving corridor design should involve critical reflection about the methods used and should include consideration of the latest research in the field.

There are a number of things that a municipality could do differently or attempt to include if it was to implement LCC analysis in the planning of ecological corridors. These are ideas that could potentially improve the results of an analysis. Some of these were not possible to include in the conducted analysis in this project, others did not fit, and others were too resource demanding for this project. The ideas include focusing on designing corridor between habitats that are known to be populated, using a focal species approach, using empirical data for determining cost values or for validation, using more alternative scenarios in the uncertainty analysis, accounting for other land use goals in the model, and finally, the idea of modelling structural connectivity rather than functional connectivity. The benefits and drawbacks of these ideas will be considered within the context of municipalities' needs and resources.

The model implemented in this project focused on connecting potential area with sufficient habitat quality to potentially be used as actual habitat by the species. An alternative approach would be to include data on the presence of the species in question when mapping the habitats to connect. The benefit of this idea would be that the resources involved with implementing corridor plans would not be wasted on areas that may not be suitable for habitation by the species. The habitat suitability model built without species presence data provides an indication of where the species might be able to live, but it is a simplification of reality and there may be circumstances on the ground that prevents the area from being suitable. However, there is a reasonable argument to be made that it is prudent to preserve functional corridors in the landscape that might facilitate the spread of the species in the future, if the habitat quality of the connecting areas were to be improved. Improving the corridors themselves might also lead to the species populating areas that previously could not sustain it, particularly with regards to sink patches that require a certain rate of migration to sustain their populations.

This project's model also focused on only a single species. Municipalities would be advised to use a focal species approach, and focus on a range of species whose conservation can lead to improved conditions for a wider range of species and ecological processes. Designing corridors solely for a single species would not be the best way to achieve connectivity and biodiversity goals for the municipality, except in unusual circumstances. There are essentially no downsides to creating corridors for a larger number of species, although there may resource constraints for municipalities that restrict how many species they can include in the approach. The resources available to municipalities will be discussed further ahead.

Using empirical data to determine resistance values or to validate the modelled corridors is always recommended, and goes a long way towards making the model results trustworthy. However, it is by no means certain that municipalities will have data available for this purpose. They would likely have to collect their own data, which would be a rather resource intensive task.

Municipalities should without a doubt include more than one alternative scenario in their uncertainty analysis. There should be multiple scenarios in which both resistance scores and factor weights are compressed and dispersed. Uncertainty analyses cannot provide the same validation as empirical data, but they are at least easier to implement than collecting and analyzing empirical data. The municipalities should use the uncertainty estimates of multiple experts to determine which ranges of class resistance and factor weights to test for uncertainty. There is no downside to including uncertainty analysis as part of a corridor design project, except for the additional man-hours which are quite moderate.

It is worth considering if the choice of modelling functional connectivity is not ideal for municipalities. For some municipalities, LCC analysis may require too many resources when considering their goals. For example, in the interview conducted with Næstved municipality, the interviewee expressed the view that this form of modelling was likely too complex for them to use (see Appendix). In order for the model output to generate corridor designs that provide general connectivity benefits for flora and fauna of interest, it is necessary to apply a focal species approach. A focal species approach requires even more resources than what was needed for this project's LCC analysis, and would likely involve a number of experts in different species working together over a period of time in order to select the species and determine which ecological variables to include and how to create resistance values for them. This places a significant burden on municipalities who are likely operating with a limited budget. If it was somehow possible to reduce the workload for the corridor planners in the municipalities by creating a ready-for-use model that required no additional configuration or parameter specification, this might be a preferable solution for some municipalities, even if it meant sacrificing some of the model's accuracy. This is where the possibility of modelling structural connectivity comes in. A number of studies have attempted to apply LCC modelling by treating 'naturalness' as the factor that determines landscape permeability for species (Beier, et al., 2008b). Naturalness should here be understood as the degree of human modification of the landscape. For example, such an approach could rank areas in terms of high to low resistance in the order of [urban area -> roads -> agricultural areas -> natural/semi-natural area]. Such an approach might be especially useful for modelling conduit corridors, in which habitat requirements might be less important, and structural barriers and dangers in the landscape may be more important. If such a model was created, it could be applied universally in all municipalities with only few adjustments. Exactly how to implement it would require further investigation and is outside the scope of this project. The issue of modelling structural versus functional connectivity will be discussed further in subchapter 9.3.1.

In conclusion, corridor planners are advised to use the nine step structure laid out in this report and to include the latest research in the field in order to make informed decisions about the many choices that arise during the modelling process. There are a number of definite or potential improvements, compared to the LCC analysis in this report, that municipalities could include when conducting their own analyses. Municipalities should consider if basing habitats on populated locations is better than basing it on habitat quality, and they should consider if modelling structural connectivity is preferable to modelling functional connectivity. Whether these two ideas are actual improvements is uncertain, and depends on the goals and resources of the municipality. Including more empirical data for model fitting and validation and conducting more extensive uncertainty analyses are clear improvements, but are also associated with additional expenditure of time and money. If modelling functional connectivity, it is generally advisable to use a focal species approach to improve conditions for a wider range of species and ecological processes.

9.3 Is LCC analysis a useful approach for municipalities?

This subchapter will start by discussing the general advantages and disadvantages that are associated with LCC modelling. This will used as a point of departure for a discussion concerning how useful the modelling approach is for municipalities' work with mapping ecological corridors. Finally, the approach is compared to the other corridor modelling approaches that have been mentioned in chapter 4.

9.3.1 General advantages and disadvantages

This subchapter will discuss the general advantages and disadvantages of LCC modelling to create corridor designs, without regard for the particular context of Danish municipal planning. Table 10 contains a summary of the main arguments for and against the use of the LCC approach and the table will be used as a point of departure for the discussion. The chapter will particularly deal with the disadvantages on the list, including how severe they are and how they can be handled.

The main arguments for the use of the approach is that it is easy to use in its basic form, and can be easily adjusted to fit in highly complex representations of reality. The tools are available in common GIS packages and the data used is widely available. It does not require excessive computational power and it can be easily tailored to model species-specific corridors. See Table 10. These are the main arguments in favor of using this as a modelling tool. Up ahead the potential issues that can negatively affect its usefulness will be discussed one by one. The final question that remains will be: Is the method better than the other alternatives? This will be discussed in subchapter 9.3.3.

One objection that has been voiced is that the model assumes organisms have perfect knowledge of the landscape and deliberately seek to travel to specific destinations. It is highly likely that most animals have some knowledge of the local landscape (Wade, et al., 2015); but the main counterargument here is the purpose of the model. The model is used to find the areas in the landscape that provide the best connectivity between two areas, so that good connectivity can be preserved even if the remaining parts of the landscape were to deteriorate in ecological quality.

A potentially large flaw in LCC model is that they usually assume that animals choose their routes using the same rules they use for selecting habitat. When the corridors are designed as habitat for corridor dwellers that over a lifetime or over generations must move or spread across the corridor, it can easily be argued that selecting corridors on the basis of the criteria that provide the best habitat is a sensible approach (Wade, et al., 2015). With regards to passage species that require landscape features that support quick movement, the benefit of using habitat as resistance criteria becomes less obvious. Actual physical barriers such as roads, buildings or in some cases rivers, may be more important in that context (Wade, et al., 2015). It is possible that there is a relation between animals' preferred habitats and the areas they move in, but without solid evidence supporting this assumption for a particular species, it should be considered merely a hypothesis.

Besides assuming that movement is related to habitat quality, most models also use expert opinion to set the resistance values associated with landscape features. A considerable amount of uncertainty is associated with this approach. The problem can be lessened, but rarely completely avoided, by deriving resistance values from empirical data. The issue can also be dealt with by conducting uncertainty testing using different resistance schemes.

The corridors generated using the method are not guaranteed to actually be good enough to provide the desired connectivity. For this reason, it is important that decision makers are provided with some sort of measures of the quality of the corridor or the benefits provided by it. This is a difficult task, but one option is to assess the value of a corridor as a conduit, which can be achieved by comparing the corridor's EED with the dispersal distance of the species in question. Various other suggestions for evaluating corridors can be found in the literature, particularly with regards to estimating the importance of a corridor or the patches it connects to in terms of the connectivity provided in the network. When evaluating the corridors, it can also be an idea to include consideration of how climate change will affect them, although it can be a challenge for municipalities to estimate changes that will occur so far ahead.

Advantages and disadvantages of LLC analysis	
List of advantages	List of disadvantages
Easy to use (Wade, et al., 2015)	Assumes organism has perfect knowledge of the landscape and seeks to travel to specific destinations (Wade, et al., 2015)
Available tools (Adriaensen, et al., 2003)	Usually assumes animals choose routes on basis of same rules they use for selecting habitat (Beier, et al., 2008b)
Available data (Wade, et al., 2015)	Sensitive to analysis grain and termini locations (Wade, et al., 2015)
No excessive computational requirements (Wade, et al., 2015)	The focal species approach may not fully capture all ecological processes (Beier, et al., 2008b)
Flexibility: Can be simple or complex (Wade, et al., 2015)	The available GIS data may not fully reflect the factors used by organisms to make movement decisions or select habitat (Beier, et al., 2008b)
Can be tailored to specific conservation need and to specific species (Wade, et al., 2015)	Resistance values and factor weights are often subjectively selected (Beier, et al., 2008b)
Better than the other options?	The least-cost corridor may not be good enough (Beier, et al., 2008b)
	Climate change is a threat to the models (Beier, et al., 2008b)
	Some organisms, such as plants, insects, birds, can be difficult to model in this framework (Beier, et al., 2008b)

Table 10. This table describes the main advantages and disadvantages of LCC analysis as described in the literature.

With regards to the remaining issues on the list, it must be said that issues related to the scale can be avoided with careful consideration of the focal species in question and the literature suggests that a carefully executed focal species approach is likely the most effective way of including consideration of a wide range of species and ecological processes. Many models have also included species that appear on the surface to be less fitting for LCC analysis, such as insects and plants. The available GIS data may not reflect every relevant habitat factor for every species, but it can come quite close, as illustrated by the agile frog in this project.

In conclusion, while some issues are quite unproblematic and the severity of others can be lessened by carefully considering the elements of the model, there are also some unavoidable and problematic assumptions that significantly reduce the trustworthiness of LCC models. For this reason, Wade, et al. (2015) suggest considering the results of LCC models as hypotheses rather than predictions.

9.3.2 LCC analysis in Danish municipal planning

This subchapter will discuss how well-suited the LCC approach for mapping ecological corridors is for Danish municipalities. The subchapter will describe the unique circumstances present in Danish municipalities, the resources available in municipalities and more.

The appointment of ecological corridors is a useful tool for municipalities with regards to the preservation or improvement of biodiversity. An area being designated as an ecological corridor has the effect of providing the area protection from damaging construction and usually means the municipality will take efforts into improving connectivity in that area. The LCC approach in this report selects areas as corridors on the basis of which areas provide the best short or long term dispersal of organisms across the landscape, with the reasoning that preserving or improving these areas will provide the best connectivity if the nearby areas were to deteriorate in ecological quality. Designing corridors is not the only option available in the municipal toolbox. An alternative is to work towards protecting or restoring habitats. Municipalities should always consider which tools in their toolbox provide the most benefits in a particular situation. The value of implementing corridors contra other solutions is a relevant factor to consider when deciding how much money and resources to spend on corridor modelling.

One issue that is relevant for the discussion of the use of LCC analysis within Danish municipalities is that some, typically highly urbanized, municipalities are very small and contain very few natural areas. Some of these municipalities do not occupy themselves with the appointment of ecological corridors in their municipal plan (Hellesen, et al., 2013). Another issue worth mentioning is that the Danish landscape is highly modified to suit human needs, particularly with regards to the widespread presence of agricultural production. Only small proportions of the country are completely free from human influence, the remaining areas can be characterized as small areas of isolated nature surrounded by a desert of agricultural fields separated by roads. This likely has the influence of restricting the possible routes and shapes of ecological corridors. This could mean that the shape of corridors modelled with the LCC approach are likely to be very insensitive to alternative resistance ratings. This could be seen as a benefit of LCC analysis, but could also point in the direction that simpler methods of discerning the placement of corridors in the landscape could be applied just as well.

The data required for the use of the approach in municipalities is available, considering strictly the type of data needed for a basic analysis. There is a wide range of environmental geodata available for use, covering the typical factors in LCC analysis, such as land cover, elevation, roads and urbanized areas. While this data represents what is typically used in LCC analyses, the results of an LCC analysis might be improved if GIS data related to more concrete resource requirements of organisms were available. In most cases, detailed empirical data concerning species presence or movement will be difficult to find and would have to be collected by the corridor designers themselves, a task which might exceed the available resources in municipalities. While plenty of literature can be found concerning the habitat requirements of species in Denmark, more knowledge about how landscape elements can be translated into resistance values would also benefit these types of analyses.

The analysis in this project was performed using ArcGIS, which is not used by every Danish municipality. However, least-cost tools can also be found in other commonly available GIS packages such as QGIS, and if the LCC method was found to be of use for municipalities, it should be possible to implement the approach on a platform that all municipalities can make use of. Another possibility is that the analyses could be outsourced to consulting businesses.

There is a possibility that LCC modelling will be too resource demanding for some municipalities in terms of work hours, money and available expertise. This conclusion is based on a number of observations. The interviewee at Næstved municipality's environmental department, while not being familiar with the exact intricacies of the method, stated that she believed it would be too advanced, although she left open the possibility that something like this could be used in the future. She said the appointment of corridors had been a low priority within the municipality, although they were planning to begin creating new plans in the summer (see Appendix). A seeming lack of urgency when it comes to mapping ecological corridors seems to be a general phenomenon. The slow progress with updating the ecological corridor plans after, taking them over from the previous counties, and the lack of systematic GIS-based methods that Hellesen, et al. found in their 2013 report about the planning of ecological corridors, indicates that ecological corridors are not high on the to-do lists of municipalities. The lack of GIS methods is surprising given that ecological corridors have been an element in the public land use planning framework for a very long time; being a part of the counties' regional plans since at least 1993, until the task was taken over by the municipalities in 2007 (Wilhjelmudvalget, 2001b). The apparent lack of urgency may be due to a low expected biodiversity-improving effect of corridors compared to other nature management options, which also lends support to the idea that municipalities are not interested in complex and resource intensive methods for mapping corridors. A final indication that municipalities like to keep it simple and inexpensive when it comes to planning corridors is an example collection of municipal corridor projects prepared by the Danish Nature Agency. Many of the examples in the list, which vary from small, local to larger, regional scale projects, often have no known budget and no subsequent evaluation (Danish Nature Agency, n.d. a). This is by no means the case for all projects in the list, however, and there is a variation in the scale, cost, intensity, and evaluation of the projects. All in all, the different observations mentioned point towards the suggestion that municipalities will rarely be interested in establishing large scale projects involving numerous experts selecting focal species, estimating resistance values, and collecting or validating using empirical data. Municipalities, for the most part, will not be interested purely in the most accurate modelling method, but will select a modelling method based on a tradeoff between accuracy and simplicity. The degree of accuracy that can be sacrificed for simplicity is also likely to vary between municipalities and between projects. The ideal target user of LCC modelling would perhaps be a municipality with a landscape consisting of a few isolated habitats, surrounded by a thick network of roads, working to conserve a highly threatened species. In such a case, the importance of creating connectivity in the landscape, compared to simply expanding habitat areas, might be sufficiently high to warrant extensive analysis using LCC modelling.

If LCC modelling of ecological corridors was to become an attractive method for use within Danish municipalities, a natural step to take would be for municipalities to share the models that they create. Since Denmark's climate and general environment is relatively homogenous, it should be possible to create models based around specific species that can be applied anywhere. By being able to use freely available models, municipalities would save significant time developing them, and would also be able to improve them if needed. It would be especially useful to have models available for the species on the Annex IV list and the Red List. This might go some way towards reducing the work load, but the idea of developing a model that outputs structural rather than functional corridors is also worth considering.

In conclusion, LCC modelling could be useful in some circumstances, but is likely too complex and work intensive to function as the standard approach for municipalities. It is likely something that would be used as a special case approach. However, efforts could be made to reduce the workload associated with the method.

9.3.3 Value of LCC analysis compared to other approaches

This subchapter will compare LCC analysis to the two other GIS methods mentioned in chapter 4, those being the approach used in Trekantområdet and the approach used in Slagelse Municipality. It has not been the goal of this project to perform an in-depth comparison between the described GIS-based methods. Knowledge of what the methods can do is, however, relevant when attempting to determine how valuable LCC modelling could be for municipalities. Municipalities will naturally seek to use the methods that can achieve their goals with the least demands on time and expenditure. A comparison has been included for this reason, but the comparison will rely on surface impressions garnered from reading the authors' summaries of their methods, rather than any hands-on experience with the methods. The methods will be compared on three parameters: Ease of use, accuracy and support for modelling single species.

When it comes to ease of use, the approach used in Trekantområdet is likely the easiest to use. The amount of work involved with the approach used in Slagelse is hard to discern, but since it focuses on modelling criteria for structural corridor, it would likely have a similar workload as modelling structural corridors with LCC analysis would. LCC analysis focused on modelling functional corridor is likely the most complex method to apply.

With regards to accuracy, the approach used in Trekantområdet is rather simplistic. It is based on the assumption that certain landscape elements can allow dispersal of species if they are close enough together. It does not account for the quality or size of these areas, only how close to each other they are. The guide for the method present different landscape feature combinations for modelling different types of corridors, e.g. wet corridors and dry corridors, but the landscape elements used in Trekantområdets own plans seem to be based purely on areas that are 'good nature', such as Section 3 areas and Natura 2000 areas (Vejle Kommune, 2013). The approach used by Slagelse can identify areas possessing the landscape components required for structural corridors, such as wooden, coastal, wet (wetland and meadow) and dry (heath and pasture) corridors. After the concentration of the landscape components is mapped, the analyst must identify linear areas containing high concentrations of these components, seemingly on a subjective basis. The relative importance of the landscape elements is not weighted when generating the concentration map, which seems to be a weakness. Overall the approach seems useful when it comes to mapping structural corridors. The LCC approach provides the most realistic output. It can consider habitat requirements of specific species and can join the corridors of different species together into a single corridor that can accommodate a wider range of species or ecological processes. It can find the most effective routes in the landscape and does not just map concentrations of good habitat area or landscape elements buffered together into bands. Unlike the others it can account for roads, and can find the routes that avoid roads as much as possible. It has a solid theoretical underpinning and the approach is trusted enough to have been used in numerous, major corridor design projects.

A capacity for modelling single species is found in the approach of Trekantområdet. The guide proposes that species specific corridors can be mapped by including landscape elements in which the species has been observed or where it has good habitat quality (Vejle Kommune, 2013). Slagelse's approach focuses purely on structural corridors, but could perhaps be adjusted to find habitat elements for an individual species. The LCC model can incorporate both habitat requirements and structural barriers to movement while finding the optimal route in the landscape, making LCC analysis the approach most suited for mapping functional connectivity.

In conclusion, the method of Trekantområdet seems very easy to use, but the validity of the appointed corridors seems weak. It can account for individual species to some degree. The method from Slagelse seems more realistic than the method of Trekantområdet, but also seems harder to use. It is designed for pin pointing structural corridors, but could possibly be adjusted to find species specific corridors. The LCC approach provides the most realistic output and is the only one that can account for roads. It might be too complex for general use, but there is a possibility that it can be simplified to a structural approach that is still more accurate than the other methods.

10 Conclusion

The findings of the project will be summarized. Recall the problem statement from the beginning of the report:

> ***Problem statement:***
>
> *How suitable would least-cost corridor analysis be as an approach for mapping ecological corridors within Danish municipalities? How should such an analysis be structured and implemented in order to fit the municipal context?*

The problem was investigated by studying the literature in the wide field of least-cost modelling and performing a concrete least-cost corridor analysis in a Danish municipality. The literature study served as the foundation for establishing an analytical structure that a municipality could apply, and helped uncover some of the many issues that municipalities must consider when executing such analyses. The analytical structure was demonstrated in practice through the LCC analysis, and it was shown how the context of Danish municipal planning could affect the choices made during model development. A few novel ideas were proposed, such as the use of Thiessen polygons to determine which habitats to connect and the use of Equivalent Euclidean Distance to measure corridors' value as conduit corridors. The corridors that were found appeared reasonable on the surface, and the uncertainty analysis found that an alternative weighting of factors, which altered the rank order of the factors, did not produce notably different results.

The suitability of least-cost corridor analysis as an approach for mapping ecological corridors in Danish Municipalities was discussed. It was concluded that it is likely the most accurate of the described methods. However, it is likely too advanced to serve as the basic tool of all municipalities, although it could possibly find use within some municipalities who possess the resources to use it. If a simpler model focusing on structural connectivity was created, it might be sufficiently easy to use to become generally applied. Through sharing of species-specific models with each other, the municipalities could reduce the workload associated with the method.

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Appendix - Communications with Næstved's Department of Environment and Nature

A telephone conversation and a few emails were exchanged with employees at Næstved's Department of Environment and Nature. The phone conversation will be summarized and an email that was used in the report will be shown in full.

Phone conversation summary

Conversation with Sofia Mulla Kølmel – Nature employee at Næstved's Department of Environment and Nature

In the phone conversation the employee was asked about the municipality's work with ecological corridors and nature management in general, how they cooperated with other municipalities, whether the least-cost modelling approach appeared as something they could find useful, and what GIS software they used in the municipality, among a few other things.

The main pieces of information that were gathered from the interview were: The municipality only uses the ecological corridors mapped by the previous county, partly because updating the ecological corridors was not considered a high priority. She also said lack of time was an issue when it came to making plans for ecological corridors. They are planning to begin working on planning new corridors this summer. She imagines that they might use the 'Landscape character assessment' method to help identify the corridors. The environmental department makes a draft of the corridor plan, and then other departments and politicians get involved afterwards. They use Spatial Suite and Mapinfo as their GIS software. Regarding least-cost modelling, she replied that it was likely too advanced, but might be something that could be used in the future. In their general nature management work they often focus on Annex IV species. Their work with ecological corridors is at such an early state they have not put thought into cooperating with other municipalities. They are more interested in bottom-up nature projects than top-down projects. She mentioned there is about five employees in the department.

Email exchange

Reply from Birte Hvarregaard – Nature Employee at Næstved's Department of Environment and Nature

Hej Lasse,

Måske er jeg ikke den helt rigtige til at svare dig, da vi i Center for Natur og Miljø ikke er dem, der bruger kommuneplanen i vores administration. Jeg sætter derfor en plankollega c.c. på denne mail. Bo, du er velkommen til at supplere mit svar, hvis du mener, der er brug for det.

Kommuneplanens retningslinjer er ikke direkte bindende for borgerne – men er bindende for kommunens administration efter Planloven. Det kan f.eks. være, når vi meddeler landzonetilladelse til et ansøgt byggeprojekt. Der skal retningslinjerne fra f.eks. de økologiske forbindelser lægges til grund for tilladelsen. Det kan være, kommunen af den grund må give afslag på en ansøgning, men det kan også være, at vi stiller nogle bestemte vilkår til projektets udførelse.

Hvis ikke kommunen træffer sine afgørelser under hensyntagen til kommuneplanens retningslinjer, er det noget af det, der kan klages over til Miljø- og Naturklagenævnet. Der er en række organisationer, myndigheder og enkeltpersoner, der er klageberettigede efter Planlovens bestemmelser.

Håber det var det, du ønskede svar på.

MVH Birte

Original email from Lasse Fristrup Lemming – Thesis author

Hej Birte.

Jeg har et spørgsmål jeg håber du kan besvare. De retningslinjer der er beskrevet i en kommuneplan (ikke kun i Næstved) som beskriver hvordan økologiske forbindelser skal forvaltes, f.eks. at der ikke må laves bebyggelse på arealerne... Hvordan håndhæves disse retningslinjer? Hvis en privat jordejer vælger at bebygge arealet alligevel, hvad sker der så? Jeg håber du kan svare, for jeg har ikke kunne finde et svar andre steder.

Med venlig hilsen,
Lasse