Department of Physical Geography

Stockholm University

The development of landscape structures affecting biodiversity in the Hanveden and Tyresta green wedges

Sigrid Andréen

Master's thesis
Physical Geography and Quaternary Geology, 30 Credits

NKA 128
2015

Department of Physical Geography

Stockholm University

Preface

This Master's thesis is Sigrid Andréen's degree project in Physical Geography and Quaternary Geology at the Department of Physical Geography, Stockholm University. The Master's thesis comprises 30 credits (one term of full-time studies).

Supervisor has been Regina Lindborg at the Department of Physical Geography, Stockholm University. Examiner has been Sara Cousins at the Department of Physical Geography, Stockholm University.

The author is responsible for the contents of this thesis.

Stockholm, 1 September 2015

Steffen Holzkämper
Director of studies

Abstract

The green wedges of Stockholm are meant to support a high level of biodiversity as well as cultural and recreational services but evaluating the spatial development of the wedges is difficult because their delineation has changed since they were first used in a regional development plan. This study examines a part of the Hanveden and Tyresta wedges in southern Stockholm, with the goal to use robust ecological theory to evaluate the development of the wedges from 1992 until today with focus on conserving a high level of biodiversity. Using an already existing GIS-based method of identifying connectivity weaknesses in the wedges, more weaknesses were found in 2010 than in 1992 although the total area of the wedges had only declined 3.3%. The shape of the wedges had also changed, with more narrow parts in 2010 than in 1992. To more effectively compare the development of factors in the landscape that are relevant for biodiversity, this study proposes a new method using the common shrew and hazel grouse as surrogate species. The total area loss for the common shrew was 2.96% from 1991 to 2013 and 2.23% for the hazel grouse. Fragmentation increased for both species. A large part of the greenspaces relevant to the surrogate species are covered by the green wedges, meaning that important cultural and recreational values identified by the county council are also present in areas relevant to the surrogate species. Using surrogate species to delineate and monitor the green wedges could enhance the cultural and recreational qualities of the wedges, emphasize the need for connectivity planning, identify ecologically important parts of the greenspaces as well as provide a tool for following up the development of the urban greenspaces of Stockholm. However, formulating goals relevant to biodiversity is important to fully evaluate development and municipal cooperation is needed.

Table of contents

1. Introduction

As urbanization intensifies around the globe, different environmental issues related to urbanization are becoming more and more important. Besides environmental problems such as pollution and overcrowding (Lemes de Oliviera, 2014), increased temperature and increased runoff from impervious surfaces, negative effects on biodiversity have been recognized as a growing environmental problem. Habitats are destroyed and fragmented and habitat isolation is increased as areas of native vegetation or farmland are turned in to built-up areas. A commonly found effect on species diversity is a decrease in native species and increase in alien species which have a higher tolerance of changes in soil and air composition. A general increase in species has also been noted, explained by increased landscape heterogeneity (Bryant, 2006).

There are many different concepts used when describing and planning urban green areas, definitions and abbreviations used in this paper can be seen in table 1. Large continuous areas can be described as green belts or green wedges. While the green belt is a circular structure surrounding the city, separating rural from urban and limiting urban growth, the green wedge is a radial formation that connects rural areas with urban areas along a gradient towards the center. The latter structure has been argued to be economically strategic, since land costs are typically smaller away from the center, making it easier for authorities to purchase land for parks (Lemes de Oliviera, 2014). Greenways is a concept that emphasizes the importance of connectedness between greenspaces. Greenways are linear elements in the landscape that make up an interconnected structure and encompass the need for both conservation, different types of human use and other ecosystem services (Ahern, 1995; Linehan, et al., 1995; Fábos, 2004; Pena, et al., 2010). Greenway plans are usually applied at the local and regional level (Ahern, 1995) and modern day examples where greenways have been applied in urban planning include Lisbon, Portugal (Ribeiro & Barão, 2006), Kohoku New Town, Japan (Yokohari, et al., 2006) and Indianapolis, United States (Lindsey, 2003).

While greenways are usually meant to serve multiple purposes, a fundamental part of their construction is to ensure environmental sustainability through connectivity. The terminology for this type of planning varies between different studies and study areas, but common names are ecological infrastructure, ecological network, habitat networks and wildlife corridors (Ahern, 1995). This paper will use the term "greenway" for all links between two or more patches, regardless of its purpose. The difficulties of using greenways in planning are many. Economic resources are usually limited which means conserving large patches of habitat may be preferred and there may be both political and economic troubles in securing land to create functioning greenways (Bryant, 2006). In a study of how network-analysis was used in practical land-use planning in the Stockholm county, difficulties in applying ecological theory were identified. One problem was the lack of tools for assessing connectivity and the effects of habitat fragmentation. There were few systematic methods used and there were difficulties in choosing study species as well as lack of input data regarding both species and the identification of habitats. Another important aspect was that connectivity was often not studied as an ecological issue, but rather as an issue of human recreation. Ecological benefits were thought to follow the strengthening of connectivity for human purposes (Bergsten & Zetterberg, 2013).

Table 1. Definitions and abbreviations.

Barrier	A structure in the landscape which hinders the flow of organisms and matter.
Biodiversity	A diversity of genes, species and ecosystems (Elander, et al., 2005).
Connectivity	The degree to which organisms and matter can move through the landscape (Mitchell, et al., 2015)
Edge	The outer part of a patch, which has a different species composition and abundance than the interior of the patch (Forman & Godron, 1981).
Fragmentation	The process of breaking apart areas of natural land cover (Tscharntke, et al., 2012).
Greenspace	Parks, gardens, forests and other natural areas that conserve ecosystem functions (Moseley, et al., 2013)
Greenway	A link between two or more patches created for ecological or other purposes (Ahern, 1995).
Green weak link	Parts of the green wedges of Stockholm that are less than 500 meters wide (Tillväxt, miljö och regionplanering, 2012).
Green wedge	A large area of undeveloped land that is identified as important for environmental as well as cultural and recreational purposes by planning authorities. In this paper the term is applied to the delineations used by the Stockholm county council.
Habitat-dispersal area	Area used for habitat and dispersal by surrogate species.
Landscape structure	The physical components of the landscape, often described as a matrix with patches of different quality, size, shape and connectivity (Forman & Godron, 1981)
Matrix	The area of a landscape which surrounds the patches (Forman & Godron, 1981)
Patch	A piece of land with a specific community or species assemblage, surrounded by areas of a different community structure (Forman & Godron, 1981).
SAR	Species-area relationship
SLOSS	Single Large Or Several Small
Surrogate species	A species chosen to represent other species or to serve as an indicator of different environmental qualities in conservation (Caro, 2010).

The structure of greenspaces in Stockholm is commonly described as a number of green wedges. The wedges are a historical remain of the physical development of the region, which has been centered around radial transportation routes. The number of wedges and their delineation has changed since 1991, when they were first used in a regional development plan, and the latest regional development plan in 2010 (figure 1) and they are being revised for the next regional plan (Larson, 2015). Using the green wedges in a planning context for Stockholm serves several purposes. Besides providing recreational areas they also provide room for historically interesting areas, ecological functions as well as scientific values. Other than that, they enable orientation in the landscape by dividing it in to comprehensible parts with different characteristics (Regionplane- och trafikkontoret, 1992).

A study of connectivity weaknesses in the green wedges was conducted for the regional development plan of 2010. Areas in wedges that were less than 500 meters wide were identified as “green weak links”. The distance of 500 meters was based on research regarding the distance needed from roads or development to achieve a feeling of being enclosed by nature (Tillväxt, miljö och regionplanering, 2012). The ecological grounds for a width of 500 meters are more unsure as there is little research on what distances are needed to ensure ecological connectivity. The wedges are not only used as a tool for planning ecological connectivity which means that trade-offs between the different goals have been made (Larson, 2015) and focus has shifted from conserving the ecology of the wedges, to more human-centered aspects such as recreation (Lundh Malmros, 2015). Because the extent as well as the purpose of the wedges has changed and continue to change, evaluation of the development of the wedges is difficult.

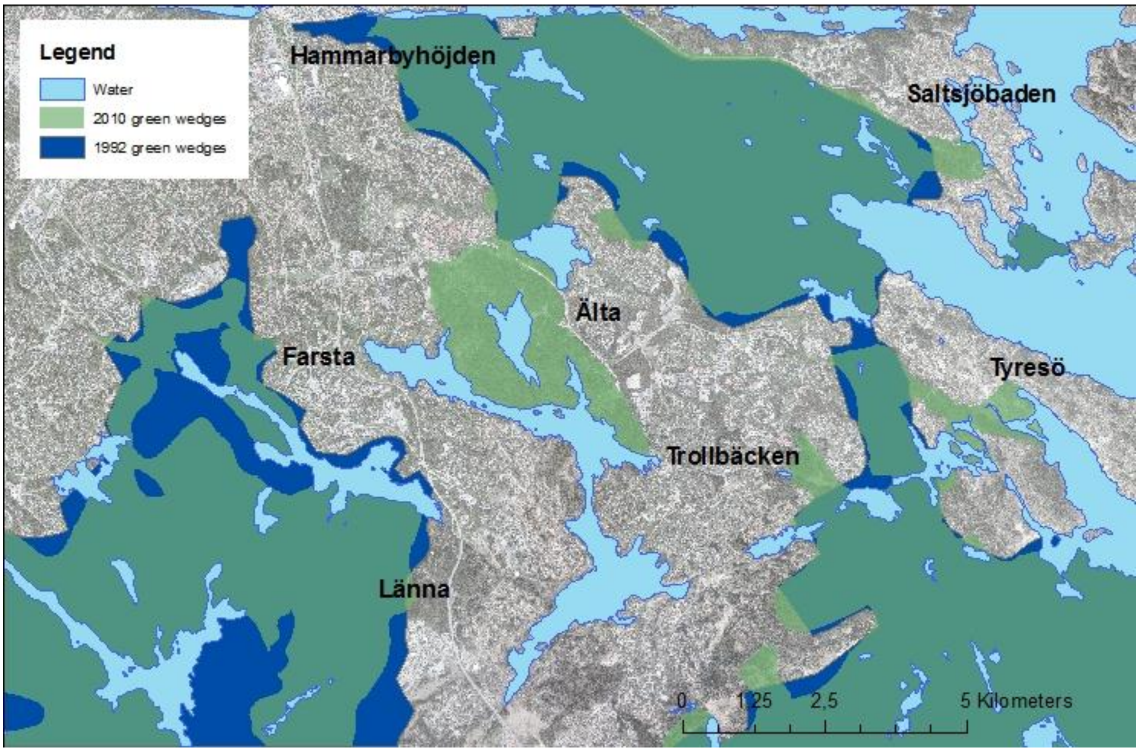

Figure 1. The extent of the wedges of the study area, in 1992 and 2010 (Regionplane- och trafikkontoret, 1992; Tillväxt- och regionplaneförvaltningen, 2010).

1.1 Purpose and research question

The purpose of this study is to use robust ecological theory to evaluate the development of the wedges during the period of 1992 to 2014 with focus on the goal to conserve a high level of biodiversity. One obstruction in this study is that the county council's criteria for delineation of the wedges has changed during the study period. Moreover, as the process of delineating the wedges is conducted in dialogue with the municipalities, it is also difficult to quantifiably evaluate the development of factors important to conserving biodiversity.

To evaluate the development of the wedges, the delineations set by the Stockholm county council in 1992 and 2010 are compared regarding their effects on biodiversity. Since the county council already has a method of identifying weak connections, this method is applied to the 1992 delineation and compared to the 2010 delineation. In addition, a new method using surrogate species is proposed, identifying greenspaces that could be of importance when delineating and evaluating the development of the wedges. Connectivity weaknesses and impact on biodiversity with this proposed method are compared to the previous methods.

The following questions are specifically addressed:

1. How has the county council's wedge delineation changed since 1992 and how does this correspond to the goal of conserving a high level of biodiversity?
2. How can the important parts of the landscape structure be identified to conserve biodiversity and provide quantifiable objectives?
3. How have connectivity weaknesses developed and how may these affect biodiversity?

2. Theoretical background

2.1 Biodiversity

Biodiversity is a key concept in ecology and is usually described as genetic diversity (many individuals within a species), species diversity (many different species) and ecological diversity (many communities within an ecosystem) (Elander, et al., 2005). Genetic, species and ecological biodiversity can be seen as different levels of organization, for which we need different assessment and conservation methods (Savard, et al., 2000). At the same time they overlap, so it is imperative that consideration is taken to all three levels when making decisions regarding planning and conservation. This also means that a lower level in the biodiversity hierarchy can sometimes be helped when assessing a higher level in the biodiversity hierarchy (Savard, et al., 2000; Elander, et al., 2005).

2.2 Creating a management regime

As with any landscape management regime, defining goals are central. With clear goals it is easier to prioritize different actions. What do we want from the landscape, is it mainly provisional, cultural, supporting or regulating ecosystem services? Do we want to conserve a single species or an entire guild? Do we want to conserve species or genetic diversity? Depending on our priorities, different actions can be considered and certain trade-offs between interests may be required. This is especially significant in multipurpose green structures (Ahern, 1995). Developing long-term quantifiable objectives is one of the most important aspects to take in to consideration when creating a landscape management regime (Lindenmayer, et al., 2008). When it comes to conservation of biodiversity, Savard et al. (2000) also stress that goals must be specific and achievable. The nature of the biodiversity concept, which is in itself very diverse, requires that the type of biodiversity wanted as well as the spatial scale are well defined.

Linehan et al. (1995) point out the importance of using ecological structure as a basis for landscape design. A functioning ecological structure will make room for other needs such as economic development and the need for open spaces as well. There may however exist conflicts between human wants and needs and ecological conservation. Lehvävirta et al. (2014) have shown that trampling can affect tree regeneration as well as species composition along paths. Some species such as pigeons are not are appreciated when they appear in large numbers (Savard, et al., 2000). Human settlement has been shown to increase land prices, making it costlier to set aside land for conservation. On the other hand, human population density has been shown to have a positive correlation to overall species richness and especially bird species richness, which could be explained by increased landscape heterogeneity and new nature types (Fairbanks, 2004).

2.3 Goals for the green wedges of Stockholm

As mentioned earlier, the green wedges are supposed to encompass many important aspects of the urban greenspaces: they provide easily accessible nature areas, quiet places and cultural experiences. They also act as air and water purifiers, temperature regulators and support biodiversity (Regionplanenämnden, 2010).

The importance of biodiversity in the wedges is mentioned in all regional development plans and some reports regarding greenspaces that have been published since the wedges were first used in planning. The wedges shall provide a rich flora and fauna for the inhabitants of Stockholm and be able to be used in research and education (Regionplane- och trafikkontoret, 1992). The 1991 regional development plan mentions legal requirements to conserve genetic diversity and the wide range of different nature types is seen as an indicator of good opportunities to support biodiversity (Regionplane- och trafikkontoret, 2002). The structure of the wedges is emphasized. Large core areas are meant to hold high nature values with a high level of biodiversity that can act as organism sources to less productive areas while greenspaces closer to development work as greenways connecting different parts of the greenspace structure to each other (Regionplane- och trafikkontoret, 1992). Concrete goals regarding the wedges are not mentioned in any regional development plan but the latest plan has an environmental assessment where different development alternatives are compared. With this plan, 1.2% of the current wedge area is planned to be turned in to development until the year 2030 (Regionplanenämnden, 2010).

2.4 The role of landscape structure

In landscape ecology, spatial aspects of ecosystems are an area of interest (Bastian, 2001) and as the urban population grows more and more thought is put into the spatial design of urban greenspaces (Bryant, 2006). Habitat fragmentation – the breaking apart of areas of natural land cover (Tscharntke, et al., 2012) is rapidly growing and has become an important area of research. While habitat fragmentation per se only involves turning large areas into smaller ones (Tscharntke, et al., 2012), the process of habitat fragmentation almost always includes a total loss of habitat, increased isolation of habitat patches and reduction in habitat patch size (Linehan, et al., 1995; Botequilha Leitão & Ahern, 2002). It is claimed to be a major threat to biodiversity worldwide (Botequilha Leitão & Ahern, 2002; Campos, et al., 2013), however this idea has been contested by researchers claiming that fragmentation actually increases habitat heterogeneity, in turn increasing biodiversity. An explanation to these two differing views could be that fragmentation is difficult to separate from other processes such as habitat loss (Tscharntke, et al., 2012). There are two ways of counteracting the effects of fragmentation: by increasing habitat area and increasing connectivity (Noss, 1987). Below, some important concepts related to landscape structure, fragmentation and biodiversity are reviewed.

2.4.1 Matrix, patches and edges

The landscape is often described as a matrix, containing patches of areas of varying suitability, depending on the focus of the study. Their size, shape and configuration within the matrix affect how they function as single patches as well as how they function together (Forman & Godron, 1981). The edge of a patch will have different qualities than the center of the patch (Forman & Godron, 1981; Harper, et al., 2005), a result of both biotic and abiotic factors such as changes in vegetation, soil, light and increased access for certain species. The edge influence on the focal patch is increased if the contrast between the patch and matrix is high and also depends on how old the edge is and if it is maintained or not (Harper, et al., 2005). The responses from different organisms will vary, for example birds are less affected by edge effects than herbs and mosses (Forman & Godron, 1981). Edge effects are not necessarily something negative. If species in need of conservation are typically found in edge areas, a fragmented landscape can be preferred (Higgs, 1981).

The influence of the edge effect is related to the shape of the patch. A round patch will have a larger interior unaffected area, than e.g. a rectangular patch of the same size. Consequently, a long and narrow patch may not have any interior at all (Forman & Godron, 1981). Roads are an important part of the urban landscape that create an edge effect on surrounding areas both by their physical contrast to the surrounding areas but also by noise contamination and pollution (Forman & Alexander, 1998; Kociolek, et al., 2011).

2.4.2 SAR and SLOSS

Related to fragmentation is one theory that has almost become a paradigm within the field: the species-area relationship (SAR) (Lomolino, 2001). The theory states that the number of species increases along with increased area, up to a certain point where the species-area relationship stagnates (Campos, et al., 2013; Lomolino, 2001)**.** This has been explained by the simple fact that a large area will most probably hold a larger habitat diversity than a small area, making it able to sustain a higher amount of species. Even when the number of habitats is removed from the equation, the total patch area influences the number of species (Forman & Godron, 1981), although with varying responses in different organism groups (Forman & Godron, 1981; Lomolino, 2001). SAR is used in many ways in research, among others to predict declines in biodiversity due to human land use change (Lomolino, 2001).

The SAR-relationship relates to a debate over whether a single large habitat should be preferred over several small habitats, also known as the SLOSS-debate. Single large patches are needed for conservation of rare species and can act as a source of organisms for near-by patches that are too small to sustain a whole life-cycle (Tjørve, 2010). Focusing on single large patches can also lead to a loss of species that need habitats outside of the focal area (Higgs, 1981). Several small patches can sustain a larger biodiversity (Burkey, 1988; Tjørve, 2010) and this type of system is less vulnerable to extinction, especially stochastic extinction (Higgs, 1981; Burkey, 1988; Tjørve, 2010). In cases where the goal is to conserve species groups that are mutually exclusive, several patches can be preferred (Higgs, 1981). Burkey (1988) noted that an important factor when researching the different consequences of single large or several small patches, is the underlying objective of the research. Proponents of single large patches try to minimize extinction while proponents of several small patches try to maximize biodiversity.

2.4.3 Connectivity

As reviewed earlier, connectivity between patches can enhance the chances of survival of an organism or even a whole population or species (Noss, 1987). Connectivity can be defined in several ways: as the degree of habitat connectivity for a specific species, as the connectivity of a specific land cover type, or as connectivity over multiple spatial scales (Lindenmayer, et al., 2008). Simply put, measures of connectivity describe how easily organisms and matter can move through the landscape (Mitchell, et al., 2015).

Different types of network theory is applied in the study of landscape connectivity. Although varying in their degree of complexity, they mainly utilize information about different species' habitat preferences and dispersal abilities to create visualizations of how a species or set of species can move though the landscape. These visualizations consist of elements such as habitat patches, corridors, hot spots and buffer areas (Mörtberg & Wallentinus, 2000).

In the urban environment structures such as green wedges, streams and hedgerows can be used to connect different greenspaces at varying scales (Elander, et al., 2005). Greenways are a way of improving connectivity, which have been advocated by several scientists. According to Noss (1987), they should be seen as a cost-effective complement to creating several large reserves, especially in a human-dominated and fragmented urban landscape. However, the importance of greenways for biodiversity conservation is widely contested, the main arguments for and against greenways can be seen in table 2. The design of the greenway is critical to its function and failing greenways may be the result of the design not suiting the needs of the studied species (Noss, 1987). Angold et al. (2006) did not find any evidence that greenways enhanced the diversity of plants and beetles in urban environments. They did on the other hand find that greenways enhanced dispersal of dormice and water voles and theorized that greenways could help small and medium sized mammals.

Table 2. Potential advantages and disadvantages of greenways for conservation of terrestrial species and habitats (after Noss, 1987).

Potential advantages	**Potential disadvantages**
1. Increased migration, which could A – Increase or maintain species richness and diversity B – Increase population sizes of particular species C – Decrease probability of extinction D – Permit species re-establishment E - Prevent inbreeding depression/maintain genetic diversity	1. Increase immigration, which could A – Facilitate the spread of diseases, pests, etc. B – Decrease the level of genetic variation between populations (outbreeding depression)
2. Increased foraging area for wide ranging species.	2. Facilitate spread of fire and other contagious catastrophes
3. Provide escape cover for movement between patches.	3. Increase exposure to hunters, poachers and predators
4. Increase accessibility to a mix of habitats	4. May not function for species not specifically studied

5. Provide alternative refuge from large disturbances	5. Cost and conflicts with conventional conservation direction of preserving endangered species
6. Provide greenbelts to: A – Limit urban growth B – Abate pollution C – Provide recreational opportunities D – Enhance and protect scenery E – Improve land values	

2.4.3 Barriers

Barriers are structures in the landscape that hinder the flow of organisms and matter. They can be natural structures such as water ways or ravines but they can also be man-made structures such as sound low walls (Ray, et al., 2002), built-up areas or infrastructure (Zetterberg, et al., 2010). In the urban environment roads are a common barrier with major effects on ecological processes. Mortality can be locally enhanced by collisions with cars and many species avoid roads due to traffic noise. Road width and traffic density are the most important factors affecting organism movement according to Forman and Alexander (1998) and there are several studies that show the negative impact of higher traffic density. The type of surface of the road may also affect species mobility (Mcgregor, et al., 2008). Paved roads have the largest negative effects on ecosystems and small, unpaved roads have been shown to facilitate seed dispersal, as they are an attractive place for mammals to defecate (Suárez-Esteban, et al., 2013). The mobility of birds is thought to be affected by road width (Kociolek, et al., 2011), a factor which influences the movement of other species as well. Studies have shown that carabid beetles and wolf spiders are hindered by roads that are only 2.5 meters wide and small mammals are far less likely to cross a lightly travelled 6-15 meter road than they are to move within adjacent habitats. The barrier effect is stated to be the most important ecological impact of roads (Forman & Alexander, 1998).

2.5 Surrogate species

Surrogate species are “species that are used to represent other species or aspects of the environment to attain a conservation objective” (Caro, 2010, p. 1). They are used as a shortcut to reach desired conservation goals such as conserving biodiversity, conserving certain rare or endangered species or monitoring environmental change. The idea is that by focusing conservation efforts on one, a few or a group of organisms, the results can be extrapolated to other areas of the ecosystem. Using surrogate species brings several advantages, for example the possibility to cover large areas, to map remotely instead of performing time consuming and costly field studies, including whole ecological processes and also being able to manage the environment for species which are yet unknown (Caro, 2010). At the same time, the choice of surrogate species is closely related to the reliability and effectiveness of the result, as noted by Linehan et al. (1995) in the case of creating greenways for wildlife connectivity.

Surrogate species can be divided into different categories depending on their function for the ecosystem and the organisms that they are to represent. The terms are used quite differently in different studies and are often confused and mixed together, which makes their evaluation difficult. Indicator species show changes in for example population sizes, environmental health, habitat quality and can be used to identify areas of high biodiversity or suitability for other species (Caro, 2010). A drawback to the use of indicator species is that there is often difficulty in deciding what to indicate as well as what species will be the best indicator (Simberloff, 1998). The use of umbrella species relies on the idea that a species has area and habitat requirements that encompass requirements of other species (Caro, 2010). The species chosen for this type of approach are typically large mammals, since large organisms tend to move over a larger area than small organisms (Caro & O'Doherty, 1999; Roberge & Anglestam, 2004) but smaller organisms can be used as well, as long as they are large relative to the organisms they are to encompass (Caro & O'Doherty, 1999). This is much in analogy with the species-area relationship (Lomolino, 2001; Campos, et al., 2013). Birds are another group that are frequently used as umbrella species (Roberge & Anglestam, 2004). In a review of 18 different studies of the usefulness of umbrella species, Roberge and Anglestam (2004) conclude that most studies showed limited effectiveness, only covering a few taxa. Using multiple species and systematically selecting different criteria for conservation could produce better results. This is however difficult due to the need of much data, sometimes on aspects that are difficult to study such as dispersal. A third type of surrogate species is the flagship species, not chosen as much for its ecological contribution as for its ability to raise public support for conservation efforts (Caro, 2010).

3. Methodology

3.1 Study area

In the latest regional development plan from 2010, the Stockholm region had ten green wedges, as shown in figure 2.

The study area was located in southern Stockholm and included parts of the Hanveden and Tyresta green wedges (figure 2). The Hanveden wedge is situated in the municipalities of Stockholm, Huddinge, Haninge and Botkyrka. It is identified by the county council as one of the most important wildlife areas of the region and is important for recreation. The wedge is also important for connectivity of species associated with old coniferous forest and broad-leaved deciduous forest. The Tyresta wedge starts quite near the center of Stockholm and continues out south-east, through the municipalities of Stockholm, Nacka, Tyresö and Haninge. Its most important feature is the large Tyresta national park with its old-growth forest. The wedge has a long shoreline and offers large nature areas close to the city (Regionplanenämnden, 2010).

The size of the study area was limited by the time available for the study. The location was chosen for two reasons, one being that its close proximity to the center of an expanding city makes it vulnerable to further development which may reduce the ecological qualities of the wedges. In the 2010 regional development plan there were six green weak links identified in the study area - parts of the wedges that did not provide sufficient connection (Regionplanenämnden, 2010). The study area was also chosen because it was currently the focus of a project by the county council on how to strengthen the green weak links, during the time this study was performed.

Figure 2. The ten green wedges and their green weak links as described in the 2010 regional development plan for Stockholm, with the study area pointed out in red (Tillväxt, miljö och regionplanering, 2010, cropped).

3.2 Study species

The common shrew and hazel grouse were chosen as study species. These are two generalist species with varying habitat requirements, dispersal abilities and impact on the environment. As generalists, they can inhabit many different nature types and will, if habitat size and connectivity allows for it, be present over a large area. As area size and biodiversity are positively correlated (Lomolino, 2001; Campos, et al., 2013), a large wedge area may be favorable. Working at a large scale could also ensure that the needs of smaller species are met within the requirements of larger species (Savard, et al., 2000). In this study the chosen surrogate species can be seen as umbrella species, covering the needs of other generalist species (Caro, 2010). Lastly, using this type of coarse grained approach fits well with the available data of the study area and the idea that the wedges are to be a large, regional structure.

3.2.1 The common shrew (Sorex araneus)

The common shrew (also called Eurasian shrew) is a small mammal with a size of about 48-80 mm and weighing 5-14 g. It inhabits a large part of Europe and Asia, from the British Isles to Lake Baikal (Stone, 1995) and from the Arctic coast to the Mediterranean Sea (IUCN, 2015), apart from arid areas and large parts of Italy, France, the Balkans and Iberia (Stone, 1995; IUCN, 2015).

The common shrew lives on the ground of many different nature types (Stone, 1995). It avoids very arid habitats and is most abundant in cool and damp areas with dense vegetation, such as riparian forests. It can also be found in woodland, scrub, road verges and other dryer areas (IUCN, 2015). In some studies the common shrew has been considered a grassland species, although it is also very common in forests (Hansson, 1987).

The area a shrew uses during its lifetime can be divided in to two categories: the first is the “foraging area”, which is the area needed to attain vital resources. The foraging area is enclosed by the “familiar area”, needed for exploratory activity. Exploration provides information about resources that are not necessarily needed at the moment, but which could increase later chances of survival. For the common shrew, the foraging area has a 30 meter radius while the radius of the familiar area is 180-240 meters (Shchipanov, et al., 2011). While their dispersal ability is often cited to be 4-5 kilometers over ice (Peltonen & Hanski, 1991) a more modest dispersal distance suggested is 119 meters (Wang & Grimm, 2007).

The diet of the common shrew consists mainly of invertebrates such as arthropods, worms and snails but plants are also part of the diet (Stone, 1995). It is an important part of the food chains in both soil and above ground (Klok & De Roos, 1998) and they also affect their environment by digging burrows to live in (Taylor, 2002) and create surface runways through vegetation (Stone, 1995).

Globally, the species is not considered threatened as it is one of the most abundant shrew species. There are however threats: they are affected by pollutants to such a degree that they are sometimes used as an indicator species for terrestrial pollution (Klok & De Roos, 1998). General habitat degradation is a threat to local populations (IUCN, 2015). In Western Europe increased cultivation and the following use of pesticides, the expanding road network and removal of important structures such as hedgerows affect insectivores, including the common shrew (Stone, 1995). Natural predators are different types of owl, foxes, stoats and kestrels (The Mammal Society, 2015).

3.2.2 Hazel grouse (Bonasa bonasia)

The hazel grouse can be found in many different forest habitats of both deciduous, mixed and coniferous type. It is a distinct forest species, and is rarely found outside forests. When choosing home range area, natural deciduous forests are highly ranked, followed by coniferous plantation, mixed forest and deciduous plantation (Rhim, 2006). As is does not disperse over large distances and is specific about site preferences, it is susceptible to habitat isolation (Rhim, 2006; Sahlsten, et al., 2010). Although the species as a whole is not threatened by extinction (Birdlife International, 2015), its numbers are declining locally due to intensive forestry (Rhim, 2006).

Several studies have been conducted on the dispersal ability of the hazel grouse, with varying results. Natal dispersal distance has been measured to a median of 800 meters, while adult hazel grouse have been found to move 340 meters (Sahlsten, et al., 2010). Dispersal ability of the hazel grouse has been found to depend on the matrix surrounding the habitat. In a study of habitat isolation in an agricultural and in a forest landscape, Åberg et al. (1995) found that no habitat fragments that were located more than 100 meters from other fragments in the agricultural landscape, were populated. In the forest landscape this distance was 2000 meters.

The size of the home range varies among individuals of different age and gender. In a study of habitat selection and home range, Rhim (2006) found that the median home range for adult males was 75.5 ha and for female, 46.2 ha. The home range size also varied between seasons: winter home ranges were almost twice as large as summer home ranges. Other studies have found the home range to be around 40 ha (Sahlsten, et al., 2010).

3.3 GIS analysis

3.3.1 Change detection with delineations and methods used by the county council

Changes in area size and patch shape were analyzed by comparing an existing GIS layer of the 2010 wedges (Tillväxt- och regionplaneförvaltningen, 2010) to a scanned and georeferenced map of the 1992 wedges, were areas of water had been removed (Regionplane- och trafikkontoret, 1992). The consulting company Ekologigruppen was hired by the county council to identify green weak links for the regional development plan of 2010. The method used by Ekologigruppen to identify green weak links was based on a GIS analysis combined with information from different reports regarding infrastructure barriers, habitat networks of old coniferous forests, oak and natural fields as well as current and planned nature reserves (Tillväxt, miljö och regionplanering, 2012). As there are no such reports describing the conditions in 1992, only the method for GIS analysis (Ekologigruppen, u.d.) was used in this study and applied to both years: a 50 meter point grid was placed over the wedges. Around every point, a 250 meter buffer was placed, creating 500 meter wide circles. All circles that were entirely within the wedge polygons were then selected and diffused. The difference between this selection and the original wedge polygons showed where the green structure was less than 500 meters wide, thus showing green weak links. The GIS analysis were performed in ArcGIS 10 (Esri, 2014).

3.3.2 Changed detection using surrogate species

Change detection using surrogate species was performed as a series of overlay analysis in ArcGIS (Esri, 2014). As a base for the mapping, a topographic map was used (Lantmäteriet, 2013). This method comes with certain limitations, as discussed later. The reason for using existing maps instead of orthophotos which could have provided more detail, was that there were no orthophotos from the early 1990's. As one of the aims of the study was to compile data for a systematic comparison of the two periods, maps that existed for both periods were chosen. The map from 2013 was acquired in digital form. As there was no map available for 1992, a map from 1991 was used

(Lantmäteriet, 1991). It was acquired in paper form and subsequently scanned and georeferenced against an orthophoto from 2011 (Lantmäteriet, 2011).

The ground layer of the 2013 map contained 12 different land classes, as shown in table 3.

Table 3. Land classes of the 2013 topographic map (Lantmäteriet, 2013).

Other open land	Other open land without forest outline
Leisure homes	High-rise buildings
Industrial area	Low-rise buildings
Deciduous forest	Coniferous and mixed forest
Precincts	Water
Water with diffuse shoreline	Arable land

The land class "other open land" contained many different types of areas. Using an orthophoto (Lantmäteriet, 2011) this land class was divided into three land classes: other open land, unmanaged open land and unpaved open land. Unmanaged open land was defined as areas far away from housing and other built-up areas, which had a high probability of containing a vegetation layer supporting small mammals. Unpaved open land was defined as areas such as golf courses, sports fields and parks, which were highly managed but could still work as dispersal areas for small mammals.

The 1991 map was created by adjusting the 2013 map to the topographic map from 1991. As there are no orthophotos from this year, no adjustments could be made to the two new land classes. The 1991 mapping relies on two assumptions:

- Areas classified as forest, arable land or other open land types in 2013, which are classified as development in 1991, are assumed to have been forest, arable land or other open land types in 1991.
- Areas classified as unmanaged open land or unpaved open land in 2013, which are classified as other open land in 1991, are assumed to have been unmanaged open land or unpaved open land in 1991.

Land classes used by the study species were of two categories: habitat land classes were suitable for the whole life cycle of the study species. Dispersal land classes could be used for movement but did not have qualities to support colonization. All other land classes were defined as barriers, constructing all movement by the study species. Parameters used to identify habitats of suitable size and dispersal areas are shown in table 4.

Table 4: Habitat and dispersal parameters for the common shrew and hazel grouse.

Landscape parameters	**Common shrew**	**Hazel grouse**
Habitat land classes	Deciduous forest (Stone, 1995; IUCN, 2015) Coniferous and mixed forest (Stone, 1995; IUCN, 2015) Other open land without forest outline (Hansson, 1987)	Deciduous forest (Rhim, 2006) Coniferous and mixed forest (Rhim, 2006)

	Unmanaged open land (Hansson, 1987)	
Dispersal land classes	All habitat land classes Arable land (IUCN, 2015) Unpaved open land	All habitat land classes Arable land (Åberg, et al., 1995)
Barriers	All other land classes	All other land classes
Habitat size	4.5 ha (Shchipanov, et al., 2011)	40 hectares (Sahlsten, et al., 2010)
Dispersal distance	119 meters (Wang & Grimm, 2007)	100 meters (Åberg, et al., 1995)

The habitat size of the common shrew was based on the 240 meter familiar area as described by Shchipanov et al. (2011). A circular area with a diameter of 240 meters gives a total area size of 45 239.93 square meters. Dispersal distance was identified for both study species by applying a buffer around the habitat and intersecting the buffered area with land classes suitable for dispersal. The result was an area which in this study is called habitat-dispersal area. Weaknesses in connectivity were defined as total breaks between patches.

3.4 Limitations

As with all studies, this study contains certain limitations and methodological problems. It is important to acknowledge the scope of the study. The green wedges of Stockholm are a regional planning tool serving multiple purposes. This study does not examine all of these purposes, the primary focus of this study is the ecological structure of the wedges. As the primary goal is to study the *difference* in green structure between two time-steps, the method used to delineate the green structure is not intended to encompass all of the factors relevant when delineating the green wedges in regional planning.

Using already existing maps means that the study relies on a classification scheme that was not intended for this particular study or even this type of study. A biotope map would have been preferable. This has affected the choice of study organism, which is why generalist species have been chosen instead of specialist species that demand very specific habitats. One way to diversify the classifications in this study has been to use an orthophoto as reference. As there was no orthophoto for 1991, certain assumptions have been made regarding land classes, as described earlier.

4. Results

4.1 Changes using delineations and methods used by the county council

With water removed, the total area of green wedge declined 3.3% between 1992 and 2010. It is reasonable to believe that this is from expanding development but it may also be explained by differences in mapping techniques and accuracy. The maps were designed for viewing in the scales of 1:150,000 and 1:100,000 so small inaccuracies could result in relatively large differences. The core parts of the wedges are the same for both years but in the 2010 form, an area in the center of the study area has been added and parts of the southwest area have been removed.

When using the GIS-based method to identify green weak links, eight links were found in 1992 and fourteen were found in 2010. While covering almost the same amount of area, the 2010 wedges (figure 4 b) have more narrow strips than the 1992 wedges (figure 4 a).

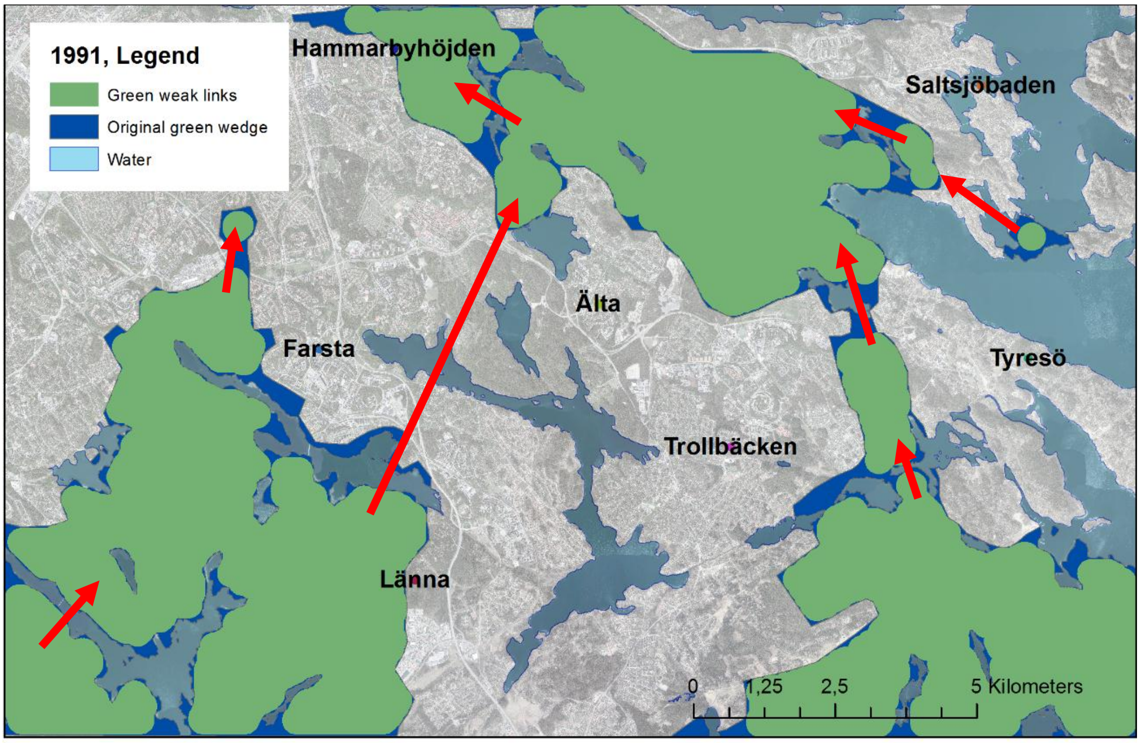

Figure 4 a. Green weak links in a part of the Hanveden and Tyresta wedges, 1992.

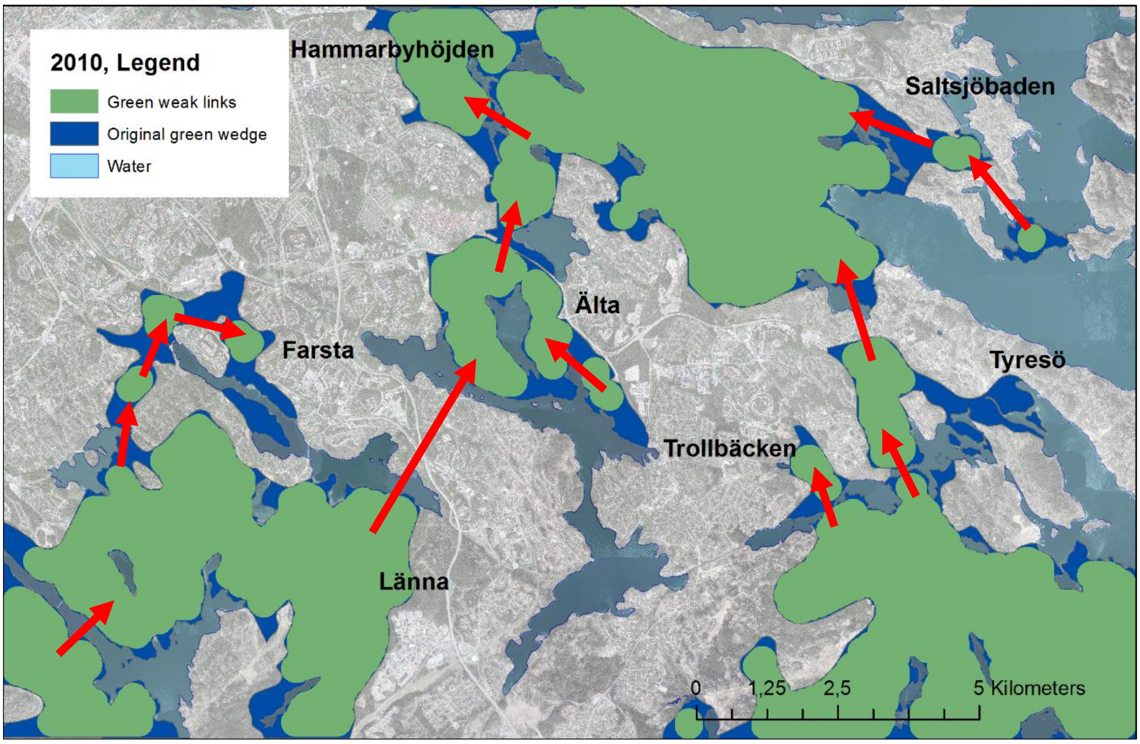

Figure 4 b. Green weak links in a part of the Hanveden and Tyresta wedges, 2010.

4.2 Changes using surrogate species

Linking the distribution patterns of study species to the existing delineations for 1992 and 2010 and the current method of identifying green weak links revealed different results.

4.2.1 Common shrew

The total area of land classes suitable as habitat for the common shrew decreased with 3 km^2 between 1991 and 2013, a change of -2.60% (table 5). When requirements of habitat size were added, the total area had decreased 2.44% during the study period. When suitable habitats were paired with dispersal areas at suitable distances (creating habitat-dispersal areas), 2.96% of land classes used by the common shrew had been changed in to other land classes unsuitable for the common shrew.

Table 5. Changes in areas potentially used by the common shrew in south Stockholm.

	1991 (km^2)	2013 (km^2)	% difference from 1991
Suitable habitat land classes	121.30	118.15	-2.60
Habitat: suitable land classes + patch size	113.84	111.07	-2.44
Habitat-dispersal area	131.86	127.96	-2.96

The biggest difference between the two time-steps can be seen in connectivity and patch size (figure 5). In 1991, two large patches of habitat-dispersal area, which do not connect to each other, could be identified. In 2013 the larger of the two patches had been divided into two separate patches. Between these large areas 37 patches of suitable land class and size could be found in 1991 and 39 were present in 2013. Three of the small 1991 patches had disappeared in 2013, while four patches had lost connectivity with the surrounding areas in 2013.

4.2.2 Hazel grouse

The total area of land classes suitable as habitat for the hazel grouse decreased 2.64% between 1991 and 2013 (table 6). When requirements of habitat size were added, the total area had decreased 2.16% during the study period. When suitable habitats were paired with dispersal areas at suitable distances, 2.23% of land cover used by the hazel grouse in 1991 had been changed in to other land classes unsuitable for the hazel grouse.

Table 6. Changes in areas potentially used by the hazel grouse in south Stockholm.

	1991 (km^2)	2013 (km^2)	% difference from 1991
Suitable habitat land classes	118.64	115.51	-2.64
Habitat: suitable land classes + patch size	70.61	69.09	-2.16
Habitat-dispersal area	72.23	70.62	-2.23

For both years, the habitat-dispersal area was divided into one north and one south patch, which were not connected (figure 6). Most of the land use change occurred along the edges, primarily in the center of the study area, between the two large patches. In the northern part, some open areas were regrown into land classes suitable for the hazel grouse.

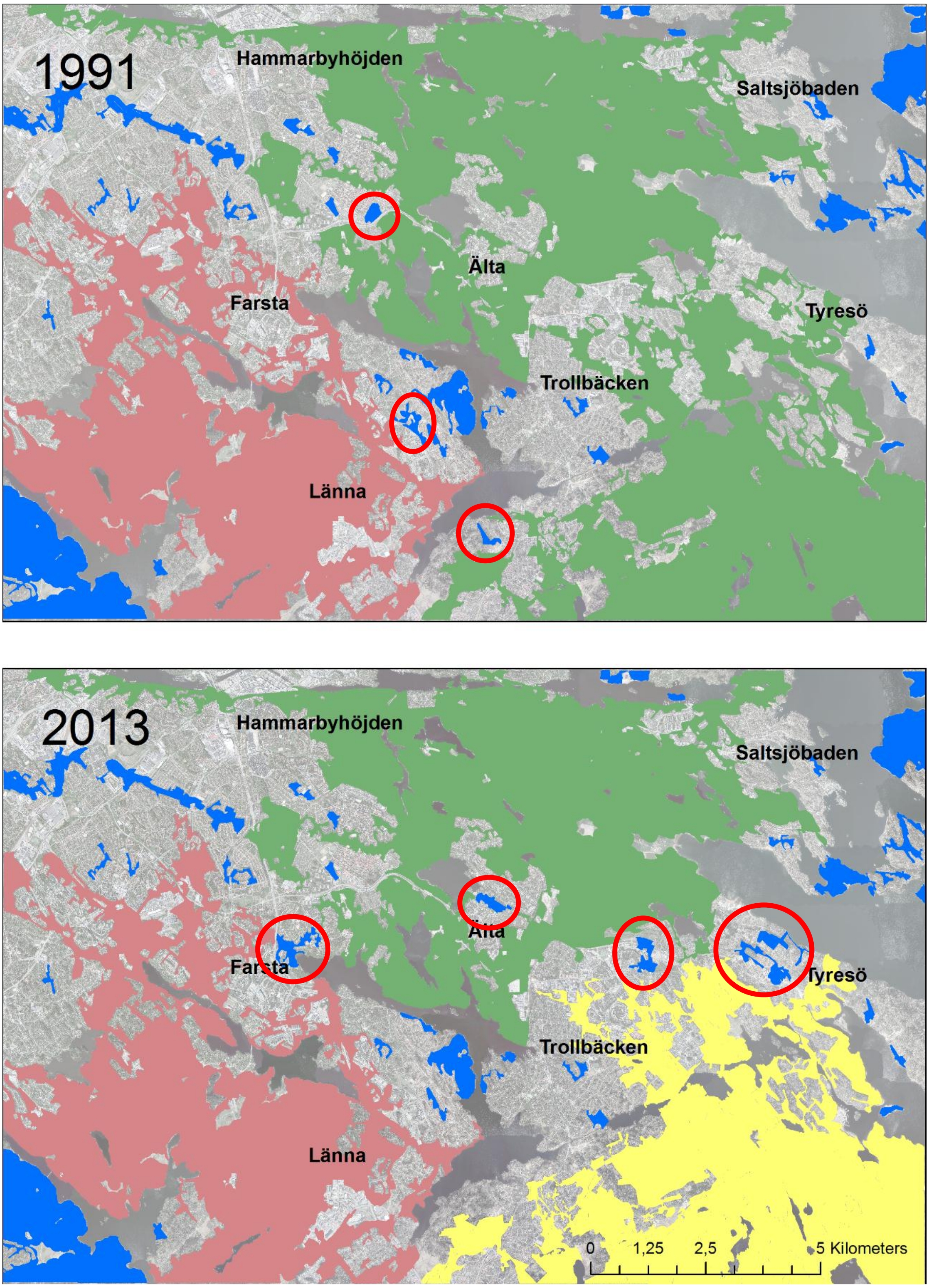

Figure 5. Habitat-dispersal area for the common shrew in 1991 and 2013. Two main areas can be found in 1991 (green and pink) and three main areas can be found in 2013 (green, pink and yellow). Blue patches are of suitable size and land class but are not connected to the larger areas. Three patches present in 1991 have disappeared, four isolated patches have been created by lost connectivity.

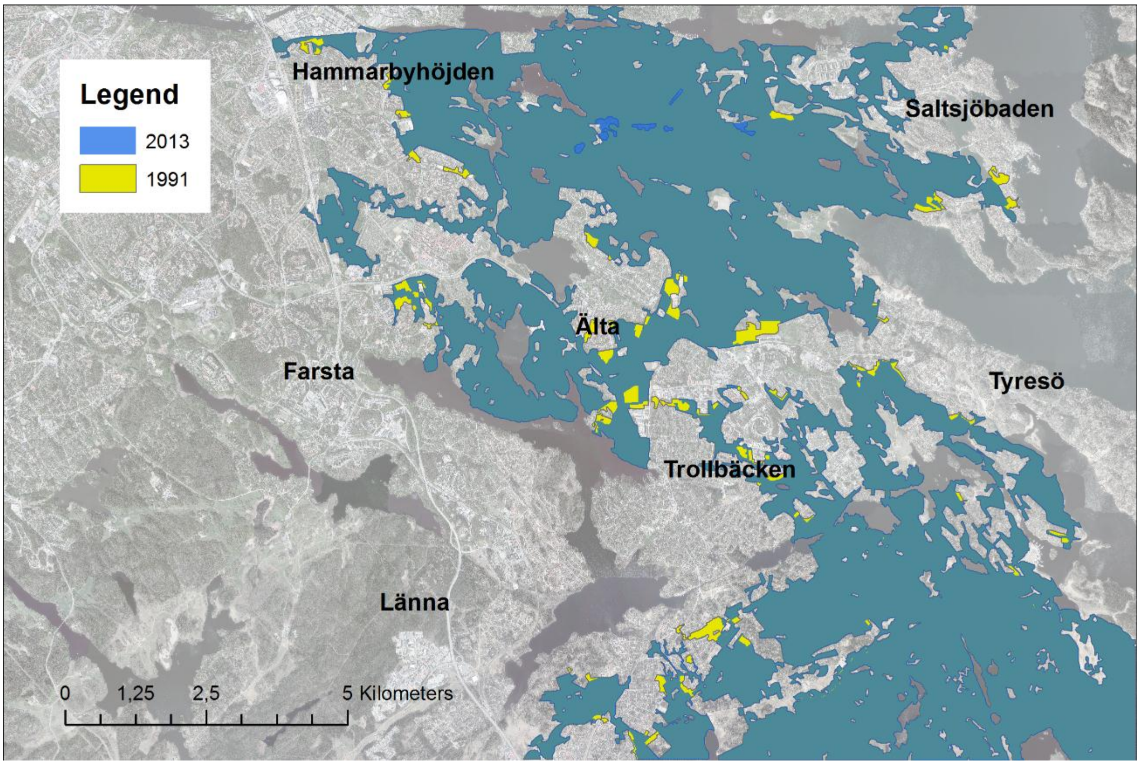

Figure 6. Habitat-dispersal area of the hazel grouse in 1991 and 2013. Two major areas can be identified: north and south. The connection between north and south has weakened during the study period.

4.3 Comparing present delineations and areas used by surrogate species

With the current wedge delineation used by the county council, most part of the hazel grouse habitat-dispersal area was covered (figure 7 a). In the south west side of the study area there was a large body of land classes that were suitable but of inadequate size. With improved connectivity between the southwest, north and east, forest living species requiring large areas could be benefited.

As the shrew can use many land cover types and does not require large areas, a large part of the green structure of the study area was covered, including almost all of what is currently defined as green wedge by the county council (figure 7 b). Most of what were potential habitat and dispersal land classes, are of the land type “unpaved open land”.

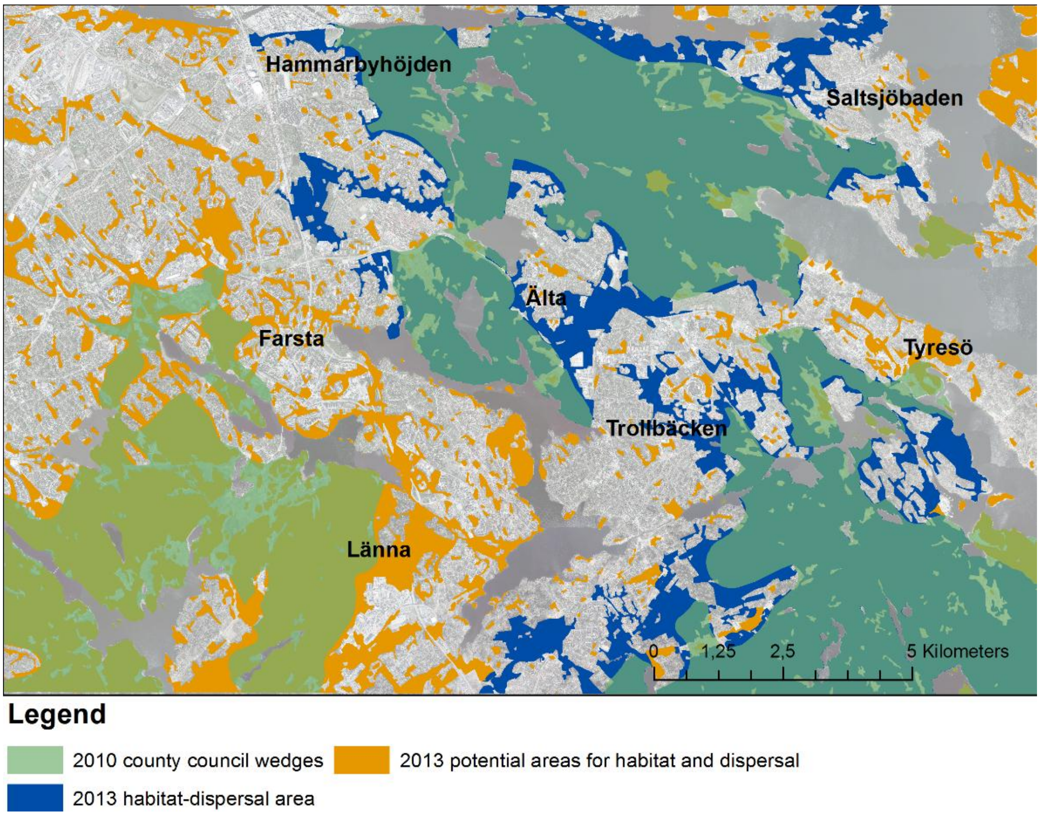

Figure 7 a. Hazel grouse habitat-dispersal area in 2013, land classes suitable for habitat and dispersal but of unsuitable size or location, and the present-day wedges.

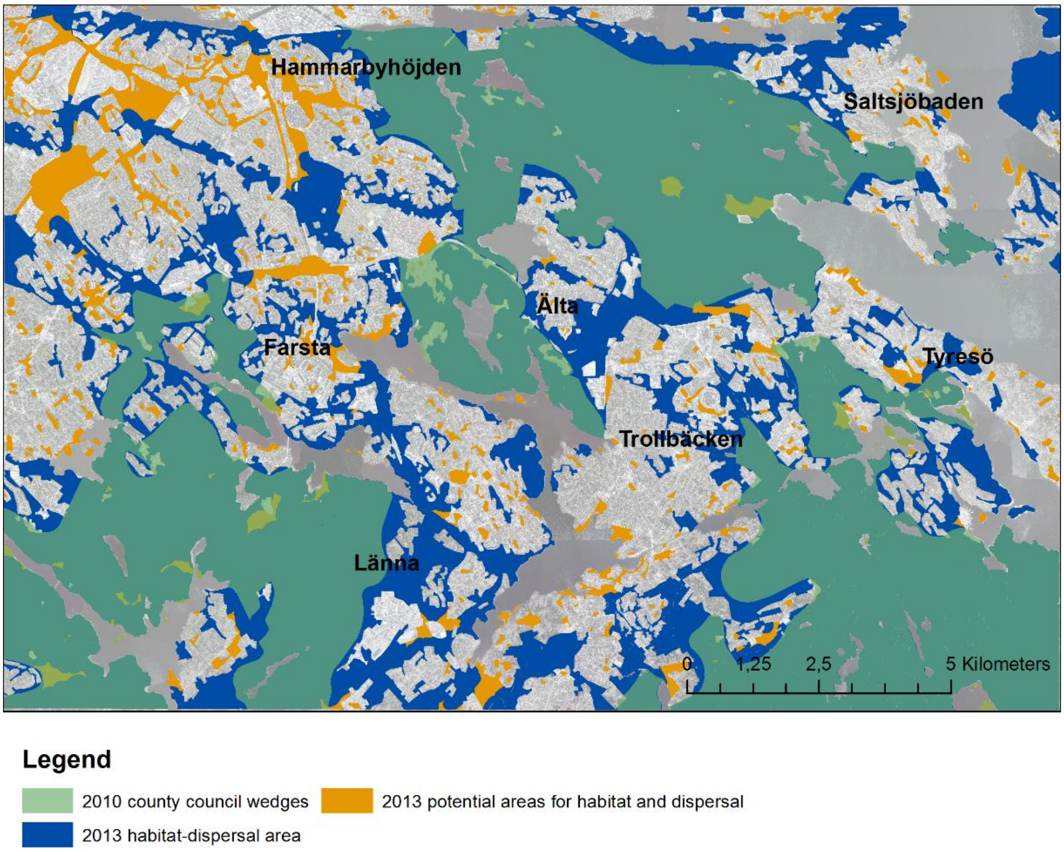

Figure 7 b. Common shrew habitat-dispersal area in 2013, land classes suitable for habitat and dispersal but of unsuitable size or location, and the present-day wedges.

5. Discussion

5.1 Changes in landscape structure can affect biodiversity

Using the county council's delineation to monitor land use changes relevant to biodiversity in the green wedges is problematic, since the delineation has changed during the study period. Even so, the total area of green wedge in the study has hardly changed at all, only 3.3%. This should be put in relation to the 34% population increase which the municipalities of the study area have received during the study period (Statistiska centralbyrån, 2014). Relying on SAR, this could mean that there has been little change affecting biodiversity. With SAR in mind the effect is little even when analyzing the landscape structure through surrogate species.

When analyzing connectivity of greenspaces this study has used two methods: one created on behalf of the county council where a weakness in connectivity, or green weak link, was defined as a part of the wedge that was less than 500 meters wide. The other method used surrogate species to identify important patches and a weakness in connectivity was defined as a total break in connectivity between two suitable patches. The first method showed a raised number of weak links during the study period. This can be because the 1992 wedges have a more compact shape than the 2010 wedges. The total loss of wedge area which has occurred during the study period could probably not alone explain such a rise in the number of weak links. The number of weaknesses has not increased when studying the hazel grouse. The large area requirements leave out many otherwise habitable areas and there was never any strip of land that fully connected the north and the south patch. This means that the number of weaknesses has not changed although the distance between the two patches has increased. Identifying weaknesses with the help of the common shrew gives a more complex landscape than for the hazel grouse and the county council's methods. Here, islands of suitable habitat are identified, which could be used to facilitate dispersal if only there was some connection between them (Noss, 1987). The development during the study period has been towards creating more of these islands, decreasing organism dispersal.

Although edge effects and barriers are not used in the GIS analysis of this study they can be believed to have an effect on the development of biodiversity (Forman & Godron, 1981; Forman & Alexander, 1998). The shape of the wedges has changed and they were more compact in 1992 than in 2010. Although this study has not identified any actual distance into the patch where edge effects are present, a larger number of thin segments increases the edge to interior ratio, reducing the amount of habitat available for species in need of environmental qualities present in the interior (Forman & Godron, 1981). Edges and barriers are likely to have an effect on what areas are suitable and not for the shrew and the hazel grouse. The hazel grouse is a forest species which means it could be affected by deteriorated forest edges. On the other hand it uses large land areas resulting in a small edge-to-interior ratio. The shrew uses smaller land areas which creates a higher degree of edge. However, the shrew has been shown to be able to live in and move over a wide variety of nature types, lessening the importance of edge effects. There is evidence that both birds and small mammals such as the shrew is affected by the presence of roads (Forman & Alexander, 1998; Kociolek, et al., 2011). This study has not pointed out any roads and their effects on habitat and dispersal but it is likely that they to some degree act as barriers for the surrogate species used in this study.

Are these changes in landscape structure positive of negative for biodiversity? A decrease in nature areas is possibly negative for species diversity (Lomolino, 2001). Fragmentation is however a more complex issue. If the goal is to conserve genetic or population diversity within a single species, some degree of heterogeneity through fragmentation may be wanted but not too much as recolonization would be hindered if the landscape becomes too fragmented. If the goal is to manage species diversity, fragmentation will have both negative and positive effects. Fragmentation may of course weaken connectivity making dispersal and recolonization more difficult (Lindenmayer, et al., 2008). On the other hand, there may be species that need to be separated which would benefit from some degree of fragmentation and too much focus on single large areas may lead to negative effects for species that are not present in these large areas (Higgs, 1981). Other than that, fragmentation may be beneficial if there are diseases that may gain increased transmission with increased connectivity (Noss, 1987).

The choice of surrogate species in this study relies mainly on SAR and the idea that a large nature area will be able to hold a high level of species diversity. So if a large area is positive, should we not focus on conserving every nature area and regard the found decline in nature area as something negative? It is important to remember that the species-area relationship dictates a rise in species only up to a certain point, the number of species cannot rise indefinitely (Lomolino, 2001). Identifying this point could be helpful, although it is a complicated task. In the meantime maybe studies like this one can help identify which areas are the most important, where surrogate species active at different spatial scales can complement each other.

5.2 Planning for biodiversity and other functions

When creating a greenspace structure and monitoring change several questions have to be addressed, such as what different functions the greenspace structure needs to fill and how these functions shall be prioritized. Relating to biodiversity, the SLOSS-problem has been shown to have different answers depending on whether the goal is to increase biodiversity or prevent extinction (Burkey, 1988). This question has not been directly addressed in the documents from the county council that have been reviewed in this study, making the priorities regarding the wedges unclear. The main idea to create very large nature areas could be a sign that a Single Large-strategy is preferred by the county council. Even so, looking at the development of habitat-dispersal areas for the shrew and hazel grouse, some degree of further fragmentation seems to be acceptable from a regional or municipal planner’s perspective. But also when using the delineations of the county council progressed fragmentation can be seen, possibly moving towards an (intended or unintended) focus on the maximization of biodiversity instead of minimization of extinction.

In the documents reviewed regarding goals for the green wedges biodiversity is mentioned several times but it is not specified which type of biodiversity is the main goal. Savard et al. (2000) emphasize the need to specify what type of biodiversity that is aimed for as well as the spatial scale. What type of biodiversity do we want, where? For example, Savard et al. (2000) suggest the goal "…to increase bird diversity in urban parks" instead of "…to enhance urban biodiversity". Monitoring and evaluating the development of biodiversity would be much easier with the former statement than with the latter. Goals also have to be set at the right organizational level. Greenway plans are usually a local and regional concern (Ahern, 1995) which makes the county council suitable for formulating goals regarding the greenspace structure. At the same time, the regional development plans produced by the county council have no legal authority. The only legally binding plans are the municipal local plans, concerning specific development projects (SFS 2010:900). This gives the municipalities an important role in development of the green structure for biodiversity, a fact that has also been recognized in the latest regional development plan where municipalities are urged to cooperate in matters regarding the green wedges (Regionplanenämnden, 2010).

Human recreation has been shown to have negative effects on the habitat of some organisms (Lehvävirta, et al., 2014) and there are potential conflicts between human use of nature areas and biodiversity conservation. In the case of the green wedges of Stockholm, one such conflict could result from the aim to provide easy access to nature areas. Just as the shape and size of a forest patch will affect its ecological qualities, the shape and size of a development patch will affect qualities relevant for the people living there. If as many people as possible are to live close to nature areas, development patches would need to have much edge and less interior. This would bring a risk of further fragmentation of nature areas, making ecological connectivity a crucial goal when planning for biodiversity.

5.3 Issues of methodology

Surrogate species are a widely used tool in conservation, although their usefulness is debatable. They provide a cost-effective tool for assessing and designing conservation efforts but demand information which is not always available (Caro, 2010), a fact that has also been pointed out by Bergsten and Zetterberg (2013) in their study of municipal planning. It would be unwise to think that simply focusing on the common shrew and hazel grouse could cover all ecological aspects needed to conserve biodiversity in the urban environment. As generalists they cover large areas of varying land classes which according to SAR should provide a high level of species diversity. The two species also function at different scales and land classes: where the common shrew acts as an umbrella species for small organisms in a variety of land classes the needs of the hazel grouse could secure the needs of other forest living organisms that require large areas. As the wedges are meant to hold values of regional interest, securing a large amount of nature areas may not be enough. Combining the needs of generalist species with more specialist and/or endangered species would help secure qualities important for biodiversity. The present wedge structure is founded on certain core areas, some of which have undergone extensive studies of existing species and biotopes (Regionplane- och trafikkontoret, 1992). Keeping track of the development of these core areas could be a good complement but when it comes to conserving large areas and to monitor urban development along the fringes of the greenspace structure, generalists like the ones used in this study would be more useful.

This study has relied on fairly simple GIS analysis. The use of more advanced GIS tools based on network theory and cost-distance analysis such as the work of Ray et al. (2002) could be helpful to determine the relative importance of different patches and corridors. However, these methods still rely on identifying relevant study species, a task which has been proven difficult (Caro, 2010). Another approach to create quantifiable indicators of change is to use landscape metrics such as different types of fragmentation indices. There is however difficulty in relating these indices to levels of biodiversity (Walz, 2011).

6. Conclusions

The green wedges have decreased in area, developed a less compact shape and the number of green weak links has risen during the study period. Based on theory regarding the influence of landscape structure on biodiversity, this development could have negative effects for biodiversity. A way to evaluate the development of factors relevant to biodiversity in the landscape structure, regardless of changes in the wedges delineations due to trade-offs between different interests, is to use surrogate species to identify important greenspaces. The results of this method show decreased greenspace area and reduced connectivity as well. This could have negative effects on biodiversity but it is important to pair these findings with relevant goals regarding biodiversity and other goals regarding for example recreation, to fully evaluate the development. Planning for biodiversity could provide protection for areas of interest for recreation, culture and other human-centered aspects of the greenspace structure. The regional level is an appropriate organizational level to formulate goals regarding biodiversity but the municipalities have an important role in the realization of these goals.

Looking forward, this study of generalist surrogate species and development of greenspace size and shape should be combined with studies of specialist species and development of habitat quality. Methods of GIS analysis that provide a more detailed picture of the movement of different organisms could also be used to prioritize between patches.

Thank you

Thank you to these two-legged and four-legged beings, in no particular order: Regina, Karin, Eva, Gurka, pappa, Spike, Oscar and Aldo. It takes a village to write a thesis.

References

Ahern, J., 1995. Greenways as a planning strategy. *Landscape and urban planning,* Volume 33, pp. 131-155.

Angold, P. G. et al., 2006. Biodiversity in urban habitat patches. *Science of the Total Environment,* Volume 360, pp. 196-204.

Bastian, O., 2001. Landscape ecology - Towards a unified discipline?. *Landscape Ecology,* Volume 19, pp. 757-766.

Bergsten, A. & Zetterberg, A., 2013. To model the landscape as a network - a practitioner's perspective. *Landscape and Urban Planning,* Volume 119, pp. 35-43.

Birdlife International, 2015. *Hazel Grouse Bonasa bonasia.* [Online]
Available at: http://www.birdlife.org/datazone/species/factsheet/22679494
[Accessed 05 05 2015].

Botequilha Leitão, A. & Ahern, J., 2002. Applying landscape ecological concepts and metrics in sustainable landscape planning. *Landscape and Urban Planning,* Volume 59, pp. 65-93.

Bryant, M. M., 2006. Urban landscape conservation and the role of ecological greenways at local and metropolitan scales. *Landscape and urban planning,* Volume 76, pp. 23-44.

Burkey, T. V., 1988. Extinction in nature reserves: the effect of fragmentation and the importance of migration between reserve fragments. *Oikos,* 55(1), pp. 75-81.

Campos, P. R. A., Rosas, A., de Oliviera, V. M. & Gomes, M. A. F., 2013. Effects of landscape structure on species diversity. *Plos One,* 8(6), pp. 1-10.

Caro, T., 2010. *Conservation by proxy. Indicator, umbrella, keystone, flagship, and other surrogate species.* Washington: Island Press.

Caro, T. M. & O'Doherty, G., 1999. On the use of surrogate species in conservation biology. *Conservation biology,* 13(4), pp. 805-814.

Ekologigruppen, n.d. *Metodförslag - grönstrukturanalys (unpublished) [Swedish].* Stockholm: Ekologigruppen.

Elander, I., Lundgren Alm, E., Malbert, B. & Sandström, U. G., 2005. Biodiversity in urban governance and planning: examples from Swedish cities. *Planning Theory & Practice,* September Vol 6, No. 3, 6(3), pp. 283-301.

Esri, 2014. *ArcGIS 10.2.2 for desktop.* s.l.:Esri.

Fábos, J. G., 2004. Greenway planning in the United States: its origin and recent case studies. *Landscape and urban planning,* Volume 68, pp. 321-342.

Fairbanks, D. H. K., 2004. Regional land-use impacts affecting avian richness patterns in Southern Africa - insights from historical avian atlas data. *Agriculture, Ecosystems and Environment,* Volume 101, pp. 269-288.

Forman, R. T. T. & Alexander, L. E., 1998. Roads and their major ecological effects. *Annual Review of Ecology and Systematics,* Volume 29, pp. 207-231.

Forman, R. T. T. & Godron, M., 1981. Patches and structural components for a landscape ecology. *BioScience,* 31(10), pp. 733-740.

Hansson, L., 1987. Dispersal routes of small mammals at an abandoned field in central Sweden. *Holarctic ecology,* Volume 10, pp. 154-159.

Harper, K. A. et al., 2005. Edge Influence on Forest Structure and Composition in Fragmented Landscapes. *Conservation Biology,* 19(3), pp. 768-782.

Higgs, A. J., 1981. Island biogeography theory and nature reserve design. *Journal of Biogeography,* Volume 8, pp. 117-124.

IUCN, 2015. *Sorex araneus.* [Online]
Available at: http://www.iucnredlist.org/details/29661/0
[Accessed 06 04 2015].

Klok, C. & De Roos, A. M., 1998. Effects of habitat size and quality on equilibrium density and extinction time of sorex araneus populations. *Journal of animal ecology,* Volume 67, pp. 195-209.

Kociolek, A. V., Clevenger, A. P., St. Clair, C. C. & Proppe, D. S., 2011. Effects of road networks on bird populations. *Conservation Biology,* 25(2), pp. 241-249.

Lantmäteriet, 1991. *Topographic map, Stockholm 10I NO and Stockholm 10I SO.* s.l.:s.n.

Lantmäteriet, 2011. *GSD-Orthophoto [Grant I2014/00691].* [Online]
Available at: https://maps.slu.se/get/
[Accessed 07 02 2015].

Lantmäteriet, 2013. *GSD-Topographic map [Grant I2014/00691].* [Online]
Available at: https://maps.slu.se/get/
[Accessed 17 03 2015].

Larson, A. M., 2015. *Landscape architect at Ekologigruppen* [Interview] (13 02 2015).

Lehvävirta, S. et al., 2014. Fragmentation and recreational use affects tree regeneration in urban forests. *Urban forestry & urban greening,* Volume 13, pp. 869-877.

Lemes de Oliviera, F., 2014. Green wedges: origins and development in Britain. *Planning perspectives,* 29(3), pp. 357-379.

Lindenmayer, D. et al., 2008. A checklist for ecological management of landscapes for conservation. *Ecology Letters,* Volume 11, pp. 78-91.

Lindsey, G., 2003. Sustainability and urban greenways. Indicators in Indianapolis. *APA Journal,* 69(2), pp. 165-180.

Linehan, J., Gross, M. & Finn, J., 1995. Greenway planning: developing a landscape ecological network approach. *Landscape and urban planning,* Volume 33, pp. 179-193.

Lomolino, M. V., 2001. The species-area relationship: new challenges for an old pattern. *Progress in physical geography,* 25(1), pp. 1-21.

Lundh Malmros, B., 2015. *Regional planner at the Stockholm county council* [Interview] (28 01 2015).

Mcgregor, R. L., Bender, D. J. & Fahrig, L., 2008. Do small mammals avoid roads because of the traffic?. *Journal of Applied Ecology,* Volume 45, pp. 117-123.

Mitchell, M. G. E. et al., 2015. Reframing landscape fragmentation's effects on ecosystem services. *Trends in ecology & evolution,* 30(4), pp. 190-198.

Moseley, D., Marzano, M., Checuti, J. & Watts, K., 2013. Green networks for people: Application of a functional approach to support the planning and management of greenspace. *Landscape and Urban Planning,* Volume 116, pp. 1-12.

Mörtberg, U. & Wallentinus, H.-G., 2000. Red-listed forest bird species in an urban environment - assessment of green space corridors. *Landscape and urban planning,* Volume 50, pp. 215-226.

Noss, R. F., 1987. Corridors in real landscapes: A reply to Simberloff and Cox. *Conservation biology,* 1(2), pp. 159-164.

Peltonen, A. & Hanski, I., 1991. Patterns of island occupancy explained by colonization and extinction rates in shrews. *Ecology,* 72(5), pp. 1698-1708.

Pena, S. B., Abreu, M. M., Teles, R. & Dalila, E.-S. M., 2010. A methodology for creating greenways though multidisciplinary sustainable landscape planning. *Journal of environmental management,* Volume 91, pp. 970-983.

Ray, N., Lehmann, A. & Joly, P., 2002. Modeling spatial distribution of amphibian populations: a GIS approach based on matrix permeability. *Biodiversity and conservation,* Volume 11, pp. 2143-2165.

Regionplane- och trafikkontoret, 1992. *Storstockholms gröna kilar [Swedish].* Stockholm: Stockholm county council.

Regionplane- och trafikkontoret, 2002. *RUFS 2001 [Swedish],* Stockholm: Stockholm county council.

Regionplanenämnden, 2010. *RUFS 2010 [Swedish],* Stockholm: Stockholm county council.

Rhim, S.-J., 2006. Home range and habitat selection of hazel grouse Bonasa bonasia in a temperate forest of South Korea. *Forest ecology and management,* Volume 226, pp. 22-25.

Ribeiro, L. & Barão, T., 2006. Greenways for recreation and maintenance of landscape equality: five case studies in Portugal. *Landscape and urban planning,* Volume 76, pp. 79-97.

Roberge, J.-M. & Anglestam, P., 2004. Usefulness of the umbrella species concept as a conservation tool. *Conservation biology,* 18(1), pp. 76-85.

Sahlsten, J., Wickström, F. & Höglund, J., 2010. Hazel grouse Bonasa Bonasia population dynamics in a fragmented landscape: a metapopulation approach. *Wildlife biology,* Volume 16, pp. 35-46.

Savard, J.-P. L., Clergeau, P. & Mennechez, G., 2000. Biodiversity concepts and urban ecosystems. *Landscape and urban planning,* Volume 48, pp. 131-142.

SFS 2010:900, 2010. *Plan- och bygglag [Swedish].* Stockholm: Socialdepartementet.

Shchipanov, N. A., Tumas'yan, F. A., Raspopova, A. A. & Kouptsov, A. V., 2011. Two types of using space in the resident common shrews Sorex araneus L.. *Biology bulletin,* 38(1), pp. 92-97.

Simberloff, D., 1998. Flagships, umbrellas and keystones: is single-species management passé in the landscape era?. *Biological conservation,* 83(3), pp. 247-257.

Statistiska centralbyrån, 2014. *Population by region, marital status, age and sex. Year 1968-2014.* Stockholm: Statistiska centralbyrån.

Stone, R. D., 1995. *Eurasion insectivores and tree shrews - status survey and conservation action plan,* Gland: IUCN/SSC.

Suárez-Esteban, A., Delibes, M. & Fedriani, J. M., 2013. Barriers or corridors? The overlooked role of unpaved roads in endozoochorous seed dispersal. *Journal of Applied Ecology,* Volume 50, pp. 767-774.

Taylor, M., 2002. *Sorex araneas Eurasian shrew.* [Online]
Available at: http://animaldiversity.org/accounts/Sorex_araneus/
[Accessed 06 04 2015].

The Mammal Society, 2015. *Common shrew - Sorex araneus.* [Online]
Available at: http://www.mammal.org.uk/species-factsheets/Common%20shrew
[Accessed 06 04 2015].

Tillväxt- och regionplaneförvaltningen, 2010. *Reg_grönstruktur_RUFS2010.* [Online]
Available at: http://www.trf.sll.se/rufskartor/Explore/Layers.aspx
[Accessed 05 02 2015].

Tillväxt, miljö och regionplanering, 2012. *Svaga samband i Stockholmsregionens gröna kilar rapport 5:2012 [Swedish],* Stockholm: Stockholm county council.

Tillväxt, m. o. r., 2010. *Regional green structure.* [Online]
Available at:
http://regionplanekontoret.lab.metamatrix.se/Global/Dokument/english/maps/Map-21-Regional-green-structure_h.jpg
[Accessed 22 04 2015].

Tjørve, E., 2010. Resolving the SLOSS-debate: Lessons from species-diversity models. *Journal of Theoretical Biology,* Volume 264, pp. 604-612.

Tscharntke, T. et al., 2012. Landscape moderation of biodiversity patterns and processes - eight hypothesis. *Biological Reviews,* Volume 87, pp. 661-685.

Walz, U., 2011. Landscape structure, landscape metrics and biodiversity. *Living reviews in landscape research,* 5(3), pp. 1-35.

Wang, M. & Grimm, V., 2007. Home range dynamics and population regulation: An individual-based model of the common shrew Sorex araneus. *Ecological Modelling,* 205(3-4), pp. 397-409.

Yokohari, M., Amemiya, M. & Amati, M., 2006. The history and future directions of greenways in Japanese New Towns. *Landscape and urban planning,* Volume 76, pp. 210-222.

Zetterberg, A., Mörtberg, U. M. & Balfors, B., 2010. Making graph theory for landscape ecological assessments, planning, and design. *Landscape and urban planning,* Volume 95, pp. 181-191.

Åberg, J., Jansson, G., Swenson, J. E. & Anglestam, P., 1995. The effects of matrix on the occurence of hazel grouse Bonasa bonasia in isolated habitat fragments. *Oecologia,* Volume 103, pp. 265-269.