EFFECT OF UNCERTAINTY IN SPATIAL DATASETS ON THE RELIABILITY OF CONNECTIVITY ANALYSIS FOR A PROTECTED AREA NETWORK IN WHISTLER, BRITISH COLUMBIA

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ABSTRACT

This study explored the application of graph theory and GIS to spatially-explicit environmental planning and the uncertainty associated with these techniques. Graph theory and GIS can be used to identify patches important to maintaining ecological connectivity. A case study, which applied graph theory to reserve design showed species with intermediate dispersal ranges had the greatest sensitivity to patch removal. A sensitivity analysis on spatial data demonstrated that the order of patch importance could change with variations in the data. Cost surface definition, and data resolution had the greatest impact on the order of patch removal and the total area retained, while the classification of the habitat map had little impact on the results. Sensitivity to uncertainties within the data was related to dispersal capability with intermediate ranges having the greatest sensitivity. This research supports presentation of alternative results, produced by sensitivity analyses, to decision-makers to mitigate against GIS uncertainties.

DEDICATION

To Steve who supported me throughout this degree despite the 4000km that separated us

AND

To all those who have come before me and all those yet to come who dedicate their lives to protecting the world around us.

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LIST OF ABBREVIATIONS AND ACRONYMS

ESRI – Environmental Systems Research Institute

GIS – Geographic Information Systems

PAN – Protected Area Network

TEM – Terrestrial Ecosystem Mapping

RIC - Resource Inventory Committee

RMOW – Resort Municipality of Whistler

SELES - Spatially Explicit Landscape Event Simulator

INTRODUCTION

Spatially-explicit planning has become an integral part of resource and environmental management, especially for conservation efforts. Geographic Information Systems (GIS) are powerful tools that can have immense influence in the decision making process (Norton & Williams 1992; Crosetto & Tarantola 2001; Kyriakidis & Dungan 2001). However, there is often little consideration at the management level given to the possible uncertainties and errors associated with the inputs and outputs of GIS-based models. Stokes and Morrison (2003), illustrated how the disregard of potential uncertainties and errors in GIS-based conservation planning may create misleading results. They were asked by the Save the Redwoods League for advice in refining an existing GIS analysis. When they examined the original analysis, they discovered the analysis had erroneously attributed high conservation values to degraded areas such as cities and logging operations while assigning low values to relatively intact areas (Stokes & Morrison 2003). These results, even though the model was based upon conservation principles, could potentially have led to poor decisions such as purchasing lands for conservation that failed to meet the organization's objectives. It is therefore important to critically analyze GIS-based models, and present uncertainties associated with the models to those making decisions. This research focuses on the question of how GIS uncertainties could potentially impact ecological decisions. In order, to address this question this research examined how uncertainty in spatial input data impacted the results of a graph-based model used to determine a patch's importance to maintaining connectivity.

The issue of uncertainty in GIS-based models is an important area of research, which has been well examined in Geographical Information Science, however very little research has thoroughly examined this issue in the ecological application of GIS (Roloff & Kernohan 1999; Elith et al. 2002; Walsh et al. 1987; Thappa & Bossler 1992). It is important to examine this issue from an ecological and management perspective due to the limited amount of resources available for conservation around the world and due to the possibility of extinction events, if poor decisions are made (Schwartz 1999). It is essential that resources are used effectively and efficiently, and the uncertainty associated with the results be communicated to the decision-makers to ensure the most effective allocation of resources. This ineffectiveness is concerning due to the often irreversibility of ecological decisions.

Recently, graph theory, as discussed below, has been applied to ecological problems, in particular to issues of connectivity. There has been no thorough examination of how this ecological application of graph theory is impacted by potential uncertainties associated with GIS. As graph theory becomes more popular in ecology and more uses are developed, the more prevalent this methodology may become in decision making, especially in regards to protected areas. It is therefore necessary to understand how spatial models based on graph theory are impacted by uncertainties in GIS. In order to determine this one must first identify potential sources for error and uncertainty in graph-based models. This topic is addressed in chapter two of this study.

Cantwell & Forman (1993) introduced methodology using graph theory to compare landscape mosaics and concluded that the method would be useful to ecology. Keitt et al. (1997) built upon this work, through combining percolation theory with non-uniform graphs in order to develop a method to assess the contribution of individual habitat patches to landscape connectivity and also to determine the sensitivity of landscape connectivity to scale. This research further developed earlier techniques to create mathematical measures to describe aspects such as patch importance, improving the applicability to management. Bunn et al. (2000) further explored the application of graph theory to examine connectivity from the perspective of species with similar habitat requirements but different dispersal capabilities. These researchers stress an important advantage to graph theory is the ability to perform an analysis without long-term population data, which is both timely and costly to collect. Urban and Keitt (2001) also examined the application of graph theory to conservation. Similar to past research, they examined the importance of patches to overall landscape connectivity, and discovered key patches important for the persistence of the Mexican Spotted Owl (*Strix occidentalis lucidus*). They also maintain that graph theory is an important tool for conservation biology and landscape ecology. Recently Rothley & Rae (unpublished) developed methodology using graph-based connectivity metrics to aid in reserve design. This research used graph theory to rank patches, in order of importance for maintaining connectivity in the landscape. These patches were then removed in order of least importance until the landscape became completely disconnected. This information was utilized to analyze the trade-offs between reserve size and connectivity. Past research on the application of graph theory to ecology and conservation have mainly focused upon the development of methodology and have not examined the impact of uncertainty on these models.

Currently the Resort Municipality of Whistler, British Columbia, is undertaking an environmental initiative that includes the creation of protected area networks (PAN) (Waldron 1999). These networks are of particular importance, as the development of Whistler's tourism industry could potentially alter the pattern of connectivity in the valley. The situation in Whistler is critical, as it fast approaches its bed unit cap and due to the potential for further development, in relation 2010 Winter Olympic Games. This pressures the municipality to allow for further development of infrastructures and facilities in the valley. Whistler utilized the ecosystem-based approach or coarse filter methodology to develop the PANs and is now attempting to determine which areas in the network are essential to maintain connectivity in the valley.

There are various research papers that deal with the issue of spatial uncertainty in ecology. Some of the earliest work in ecology regarding spatial uncertainty tested the sensitivity of habitat suitability models to grid cell size. This research found that appearance of highly suitable habitat could be masked when larger grid size cells were used in the analysis (Laymon & Reid 1986). Stoms et al. (1992) examined how uncertainties in both input data and model assumptions impacted wildlife habitat models. This work was followed by further research, which provided methodology to quantify possible uncertainties in habitat models through the use of confidence intervals (Bender et al. 1996). More recently, Elith et al. (2002) investigated methods to spatially represent uncertainty associated with logistic habitat models and noted the lack of acknowledgement of potential uncertainties in generalized linear models published in ecological literature. Research relating to the uncertainty associated with spatial reserve design has focused on factors such as survey methodology, and taxonomic diversity and these factors impact on selection algorithms (Flather et al. 1997; Freitag & Jaarsveld 1998). DeGenst et al. (2001) research explored the uncertainty associated with a buffer-based connectivity analysis performed for the red squirrel (*Sciurus vulgaris*) and found that their analysis was particularly sensitive to classification error of the landscape, and the parameters of dispersal capacity and landscape effect. These past research studies support this study's examination of the uncertainty associated with the use of graph-based connectivity metrics for reserve design.

The uncertainty of models is related to many factors such as errors in input data, natural variations, and assumptions (Elith et al. 2002). Conroy and Noon (1996) discuss the negative impact that such uncertainties in data or models may have on the outcome of the decision-making process and even state that some conservation models such as species-habitat models have "doubtful reliability". Davey and Stockwell (1991) note that uncertainty is inevitable when dealing with wildlife habitat and stems from vagueness, inaccuracies in classification, beliefs, random chance, and random variation. These researchers also note that the common ways that uncertainty has been dealt with is to either ignore it or use only certain knowledge. Unfortunately, if uncertainty is ignored no measure of confidence can be obtained and the later option only acts to limit research (Davey and Stockwell 1991).

In particular, the graph-based model presented in this study could be impacted by uncertainties within the definition of the two main data inputs, which are the habitat patch map and the cost surface. These are the two main components of the model and directly impact the landscape metrics that were calculated in this study. The other factor that could potentially impact this analysis is the spatial resolution at which these data sources are represented since the spatial resolution of these surfaces could potentially alter their appearances thereby having an indirect impact on the calculation of the landscape metrics.

The study presented in the following two chapters will explore both the ecological application of graph theory and the uncertainty that this GIS spatial data may bring to decision-making. Chapter one will present a case study, which applies the methodology presented in Rothley & Rae (unpublished) to the protected area network problem in Whistler. This application of graph theory will then be utilized to perform a sensitivity analysis, which will be presented in chapter 2. The uncertainty examined will focus on the resolution of the spatial input data, definition of the cost surface, and classification of the patch map. Chapter 2 will also review the possible sources of uncertainty and error that maybe associated with the spatial input data. It is also the intent of this project to identify how GIS uncertainty in models potentially impact the decision-making process and propose methodology to mitigate against this issue.

Objectives

1) To present an application of graph theory to reserve design in Whistler, British Columbia, in order to determine an option for maintenance of second growth patches for the protected area network
2) To identify and discuss possible sources of uncertainty in a SELES based connectivity analysis for the Whistler, British Columbia protected area network

3) To determine how spatial resolutions of 10m, 20m, 30m and 40m cell sizes of the initial spatial inputs impact the final connectivity measure and visual appearance of the output map
4) To determine the sensitivity of the model to variations in the cost surface, through a deconstruction of the cost surface utilized in the initial analysis, and to examine how this alters the connectivity measure and visual appearance of the output map
5) To determine the sensitivity of the SELES model to the definition and classification of the habitat patch map to two alternative queries based upon RIC 1998 definitions.

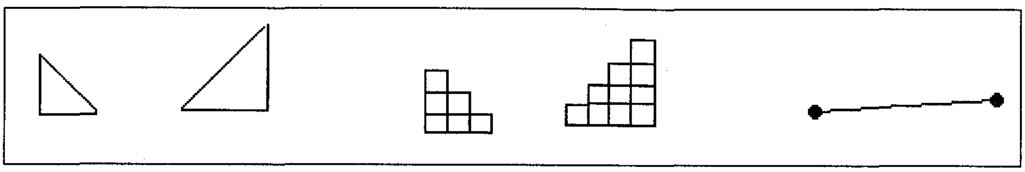
CHAPTER 1

Introduction

Graph Theory

Spatial landscape data has become an important component in conservation planning (Urban & Keitt 2001). There are three main forms of spatial data utilized in the analysis of landscapes including: spatial point patterns, geostatistical data, and lattices (Urban & Keitt 2001). Spatial data lattices, describe the landscape with measurements or values, and are used in geographical information systems (GIS) either commonly as vector or raster data structures (Urban & Keitt 2001). Lattice data can also be represented in the form of a graph. This technique has been most extensively applied to geography, information technology, and computer science (Bunn et al. 2000). Recently, the literature has begun to introduce the use of graph theory to ecology, in particular to measure the connectivity of a landscape (Bunn et al. 2000; Urban & Keitt 2001;Keitt et al. 1997). The graph represents the landscape as nodes and edges. In this study nodes correspond to patches, and edges denote dispersal between the nodes (Urban & Keitt 2001). Figure 1-1 illustrates the differences between the three forms of GIS data. The vector data form represents features as points, lines or polygons whereas raster data structures represent features as grid cells with values describing the different features on the landscape. The graph, a form of lattice data, represents the landscape with points depicting features and lines (edges), which describe how the features are connected.

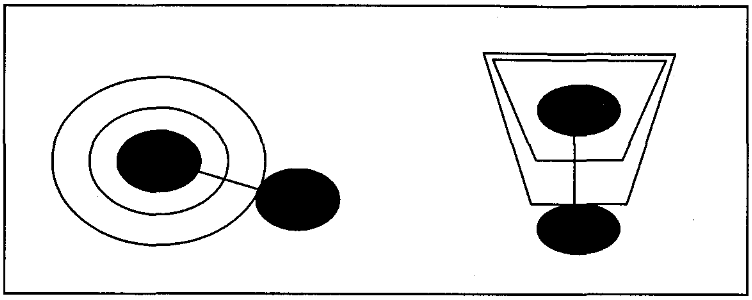
Figure 1-1 Visual differences between three GIS data structures; From left to right vector, raster, and graph data structures



In order to create a graph, there must be rules, which describe how nodes are connected by edges on the landscape (Cantwell & Forman 1993). Connections can be determined either through Euclidean distance or the use of cost surfaces (Urban & Keitt 2001; Fall unpublished). Euclidean distances simply represent the connection between nodes as a straight-line distance, ignoring the heterogeneous nature of the non-habitat matrix and the subsequent differences in the ability to disperse in various habitats within the matrix. Cost surfaces account for differences in the dispersal ability within the non-habitat matrix, since the edge represents the least cost pathway between patches, which is not necessarily a straight line. In a cost surface, cells represent the movement cost for an organism to traverse through a particular area. In general, high costs of travel would be assigned to features such as highways, steep cliffs, or lakes depending upon the species and their mode of dispersal, whereas low costs would be attributed to habitat favourable to movement such as forests.

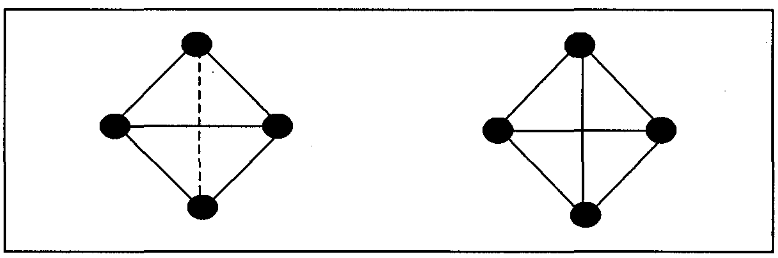
The software SELES, used in this research, determines connections between graphs, through a "growing" operation where patches increase in size at various rates depending on the cost surface, until the "growing" patch touches another patch on the landscape, a connection or edge is then formed between these two patches (Figure 1-2)(Fall & Fall 2001).

Figure 1-2 Differences between edge definition using Euclidean distances (left) and cost distances (right)



The connections or edges that occur on the landscape are dependent upon the threshold distance used in the analysis. From a conservation perspective, this distance could represent a value such as dispersal capability. Connections that are greater than the threshold distance are considered non-existent. The number of connections that are formed between patches depends upon the graph type, the minimal planar graph draws a single connection between patches, and does not allow these connection to cross whereas the complete graph, forms connections between all possible patches, allowing connections to cross (Figure 1-3) (Fall unpublished). The complete graph is utilized in this analysis, since this graph more accurately portrays dispersal behaviour of flora and fauna in the landscape due to the fact that the crossing of two edges does not make movement along that edge less efficient or impossible.

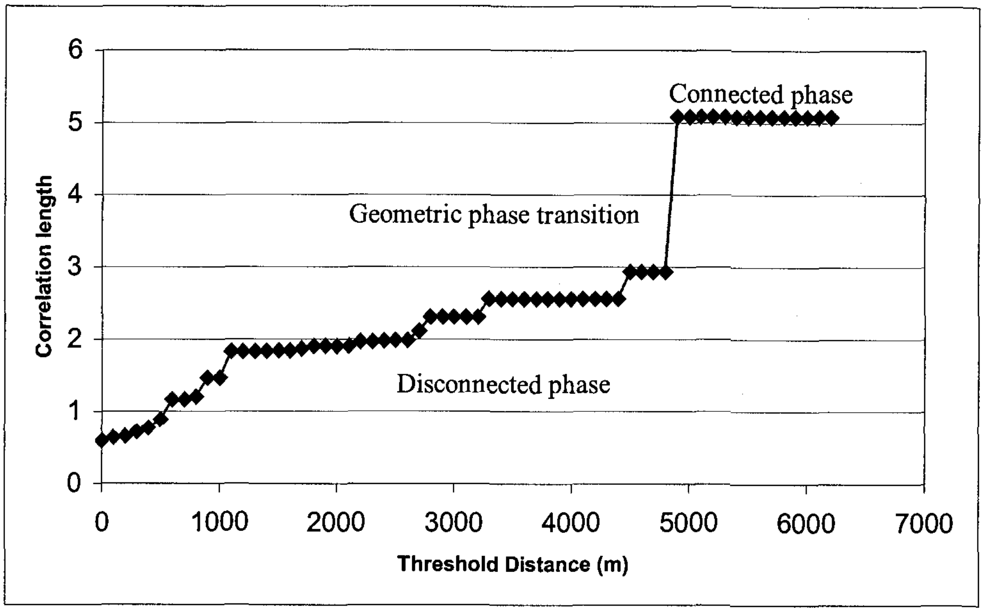
Figure 1-3 Differences between edge definitions of minimal planar graphs(left) and complete graphs(right) (adapted from Fall (unpublished))



Graph Analysis

Analysis of the graph, should occur on only those thresholds, which represent distances where the landscape appears fragmented (Keitt et al. 1997; Fall unpublished). These distances are identified on a graph of threshold distance vs correlation length, as illustrated by figure 1-4, where the landscape is in a disconnected phase or fragmented from a threshold distance of 0m to approximately 5000m, at which a geometric phase transition occurs. This transition represented by the straight line that connects the set of points in the disconnected phase with the set of points where the graph plateaus, indicating a connected landscape.

Figure 1-4 Graph of threshold vs. correlation length depicting transitions zones as identified by Keitt et al. (1997)



The critical threshold distance is the value at which the landscape changes from fragmented to completely connected, in the case of figure 1-4 the critical threshold is approximately 5000m. In order, to assign clusters a graph is created where all possible connection between nodes are identified. The critical threshold is then used to remove all edges, which are greater than the threshold distance and the cluster is then considered groups of nodes that have paths between them.

Statistical analysis on the graph can be performed at either the component (cluster) or graph (landscape) level (Fall unpublished). At the component level the common measures, which are examined include number of patches, number of patch cells, centroid location and radius of gyration. The probability of a connection between habitat patches is dependent upon the distance between them and the connectivity is related to the average size of a connected cluster (Keitt et al. 1997). In order to measure the average size of a cluster, the radius of this entity is used, however, due to its irregular shape the radius of gyration is calculated instead. This value represents the distance that a randomly moving particle will travel before encountering the edge of the cluster. Ecologically the radius of gyration corresponds to the average dispersal range (Keitt et al. 1997). The radius of gyration (R) is calculated as follows:

$$R = 1/n\sum_{i=1}^{n}\sqrt{(x_i - \overline{x})^2 + (y_i - \overline{y})^2}$$

Where n is the number of cells

x_i is the horizontal location of a cell i

x is the horizontal location of the centroid

y_i is the vertical location of a cell i

y is the vertical location of the centroid

At the landscape level, the measures used to describe the graph are the number of clusters and correlation length (Fall unpublished). The correlation length represents the "size weighted average connectivity of a set of clusters" and is the overall measure of connectivity for the landscape, the greater the correlation length the more connected the landscape (Keitt et al. 1997). Correlation length is calculated using the following equation:

$$C_d = \frac{\sum_{j=1}^{m} n_j R_j}{n}$$

Where d is the threshold distance

m is the number of connected components

n_j is the number of cells in component Rj

R_j is the radius of gyration for cluster j

n is the total number of patch cells

The final analysis, which is important in studying landscape connectivity relates to how the correlation length changes as patches, edges, and combinations of patches and edges are removed from the graph. There are two measures, patch importance index and per area patch importance index, which are utilized to describe changes in correlation length (Fall unpublished). The patch importance index (I) indicates the importance of individual patches to the landscape where the per area importance (A) describes this importance per unit area (Keitt et al. 1997). The formulas for these two statistics are as follows:

$$I_d(i) = \frac{C_d - C_d(i)}{C_d}$$

Where d is the threshold distance

C_d is graph level correlation length at d

C_i is correlation length for a patch i at d

I is the importance index for a patch i at d

$$A_d(i) = \frac{I_d(i)}{a_i}$$

Where d is the threshold distance

$I_d(i)$ – is the importance index for a patch i at d

A_i – is the area of patch I in ha

Graph theory is currently being applied to many aspects of ecology (D'Eon et al. 2002; Keitt et al. 1997). The following section will introduce the application of graph theory to protected area design.

Application of Graph Theory to Protected Area Design

There are many objectives to be considered when designing a protected area, and connectivity is one such environmental objective (Wilcox & Murphy 1985). Connectivity, as considered by this paper, is representative of the "degree to which the landscape facilitates or impedes the movement of individuals among resource patches and populations."(Pither & Taylor 1998). It is important to develop tools to aid in the protected area design process, which will allow the quantitative measurement of connectivity for alternative designs (Briers 2002). However, there are many challenges for the development of such a tool. Although, landscape connectivity is greatly dependent upon the spatial distribution of habitat and the scale at which the movement occurs, it is also greatly impacted by the characteristics of the organism itself (Keitt et al. 1997; With et al. 1997). These characteristics differ between species and can include: an organism's ability to cross barriers, perceptivity, and risk averseness (With et al. 1997). It is therefore difficult to assign a single value of connectivity to the landscape, since this value is as much dependent on the organism as the spatial arrangement and characteristics of the landscape. Another issue lies in prioritizing connectivity between various types of habitat and defining the characteristics of the landscape that constitute a connection (Rothley 1999). It is also difficult to determine where the objective of connectivity should fall within a design framework with other objectives such as size, representativeness, and cost (Rothley 1999;Bedward et al. 1992;Cabeza & Moilanen 2001;McDonnell et al. 2002).

There are different methodologies that are used to examine landscape connectivity including: empirical studies, computer simulations, and mathematical indices. Empirical studies examine how an organism utilizes and moves within the landscape, and this information is used to develop assumptions about the connectivity of an organism's environment (see Fahrig & Merriam 1985;Pither & Taylor 1998;Arnold et al. 1993). Computer simulations measure parameters such as dispersal success, search times, and distributions of virtual organisms within a landscape (Tischendorf & Fahrig 2000). The final methodology mathematical indices use non-species specific metrics to quantitatively characterize landscape patterns. These indices can range from simple values such as number of habitat patches to complex formulas that include various characteristics of the landscape (Bogarert et al. 2000;Hanski & Ovaskainen 2000).

As previously mentioned, reserve design connectivity is an important aspect to consider. Since the most conservative reserve from an environmental perspective protects everything, this becomes a strong starting point for reserve design, instead of selecting individual parcels of land to protect, one would start by protecting everything then take away pieces and look at trade-offs between the social and economic objectives. In order, to ensure the reserve serves its protection objective, one aspect that is important to consider is connectivity when deciding which parcels of land to remove from the most conservative reserve. To determine which parcels of land are important in ensuring connectivity in the landscape a neutral landscape model can be used based upon percolation theory. With and King (1997)support this use of neutral landscape models, as they note that results from these models are useful in exploring alternatives of reserve design.

Graph theory, as introduced above, is a type of percolation based model which is useful in attempting to identify critical patches, which are important to maintaining the overall connectivity of the landscape (Keitt et al. 1997). The patch importance index can be calculated for every habitat patch in the landscape determining an order of importance. The greater the value of the patch importance index the greater value the patch has to maintaining the connectivity of the landscape. This order can then be used to remove individual patches from the landscape and then the connectivity of the landscape can be re-examined to determine the impact of removing the patch, and subsequent removal order.

This chapter presents the application of a reserve selection methodology to satisfy a connectivity criterion, as developed by Rothley & Rae (unpublished), to the creation of the protected area network in Whistler, British Columbia.

Methodology

Study Area

The Resort Municipality of Whistler (RMOW) in British Columbia, Canada is an approximately 13,000ha mountainous region comprised of the coastal western hemlock (62%), mountain hemlock (19%), alpine tundra (15%), and Engelmann spruce sub-apline fir (4%) biogeoclimatic zones. The municipality is bounded on the east and west by the Coast Mountains and bisected in a southwest to northeast direction by a highway and by a valley containing a series of lakes connected by creeks (Figure 1-5). The population of RMOW has increased dramatically from 531 in 1976 to 8,896 in 2001 (Statistics Canada 2001) and the landscape has become increasingly fragmented as a result of logging, the introduction of trails for an internationally known skiing resort, and development to support a high volume of summer and winter tourism.

The RMOW PAN steering committee determined that second-growth forests, that comprise nearly 15% of RMOW, would be included in the PAN, if they contributed to connectivity. According to the British Columbia Ministry of Forests Biodiversity Guidebook (British Columbia Ministry of Forests 1995), the coastal western hemlock biogeoclimatic zone is categorized as "natural disturbance type 1" for which the frequency of cross-elevational connectivity is high for natural ecosystems. Therefore, the importance of each second growth patch in maintaining cross-elevational connectivity was examined.

Geospatial Data

The software SELES utilized for this analysis requires three inputs: a patch map, cost surface, and boundary map. The data to create these inputs was obtained from the terrestrial ecosystem mapping (TEM) project, which has been implemented by the RMOW to support its environmental strategy. The ecosystem maps were developed by B.A. Blackwell Associates Ltd. and mapping procedures followed the methods outlined by the Resource Inventory Committee (RIC) in their document Standard for Terrestrial Ecosystem Mapping in British Columbia (Green 2004).

The patch map of second growth forests, which had potential for inclusion in the PAN was created through querying the TEM data for structural stage 5. This structural stage is considered "young forest- self-thinning evident with canopy layers developed; more open than Pole/Sapling stage; usually 40-80 yrs"(Green 2004). Since this study was interested in cross-elevation movement 10 target patches were added to the patch map at high elevations to act as anchors for the destination and origin of movement (Malcolm & Revelle 2002). The patch map was then converted to a raster data set with a cell size of 50 meters and reclassified as a binary map with the value 1 representing patches. Patches of a size less than 1ha were assumed to be of poor quality and were filtered out of the subsequent analysis. The final map contained 66 patches, which ranged in size from 1ha to 373.25 ha.

It was decided that the cost surface for this analysis would be based upon slope due to Whistler's mountainous terrain, which makes this variable applicable to multiple organisms. The use of slope was utilized in this study, however, there are other methodologies which could be applied to generate a cost surface. In order to generate the initial slope surface a 20m elevation contour grid was interpolated, using a cell size of 50m and then a slope calculation was performed in Arc View 3.2a (Environmental Systems Research Institute 2000). The values for slope ranged from 0 to 65, however, SELES requires that there be no zeros in the cost surface therefore values of zeros in the slope surface were replaced by value 1. Even though slope is considered a gravity surface where it is easier to move down the slope than up the slope, it is assumed that the cost surface is symmetrical meaning that the cost to travel from A to B is equal to the travel from B to A (Bunn et al. 2000). This type of cost surface captures the assumption that organisms in the Whistler Valley prefer travel through areas with lower slopes. Areas considered hazardous or difficult for organisms to travel through, including the major highway, developed areas, and water features, had their slope values replaced with the value of the greatest slope i.e. 65. For the highway, areas where diagonal travel through the highway feature could occur at a low cost were considered holes in the cost surface and filled with values of 65, to prevent organisms from crossing the highway without crossing at least one high cost cell (Adriaensen et al. 2003).

Analysis

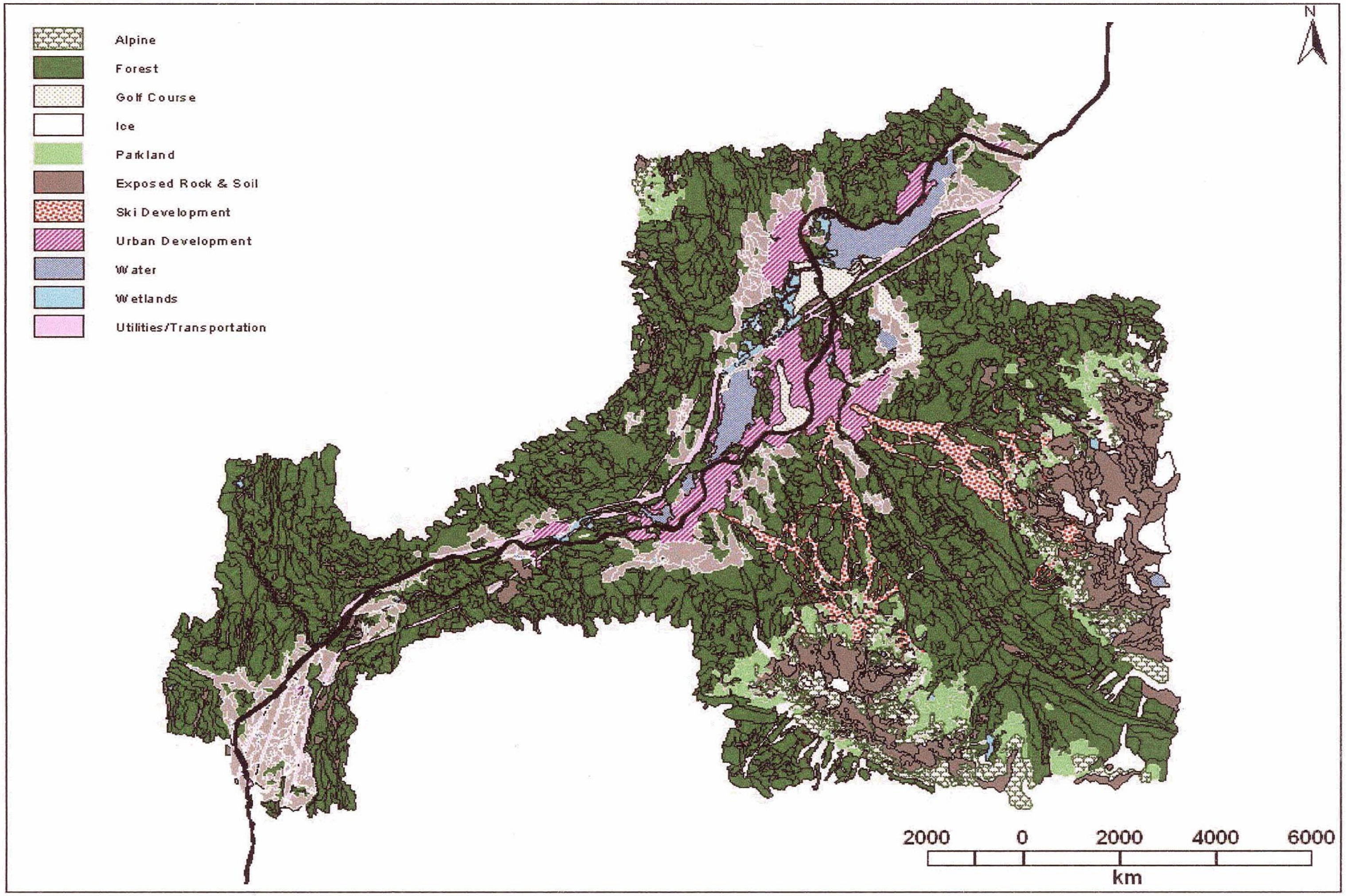
This research was performed using the program SELES, which supports graph-based connectivity analysis (Fall & Fall 2001). SELES converted the second growth patch map to nodes, located at the centroid of each patch. Since organisms may have multiple pathways between patches a complete graph format was chosen, allowing edges or paths to be formed between each pair of nodes, regardless of the existence of another pathway. Edges were delineated based upon the least cost pathway between a pair of nodes. From an organism's perspective patches are considered connected if the distance between them is less than or equal to the organism's maximum dispersal distance, *d*. For every *d* value, a cluster is defined which contains patches that are connected to a least one other patch in the landscape.

The SELES analysis examined maximum dispersal distances at 500 meter intervals, between the range of 0m to 38,500m. The upper range was chosen because this value fell within the connected phase of the correlation length vs threshold distance graph, discussed above, for the Whistler landscape. The value of the correlation length (C_d) was determined, which refers to the average distance an organism, if randomly placed in a cluster, can disperse until it reaches the boundary of a cluster (Keitt et al. 1997). The importance index I for each patch was also calculated based upon edge removal, where the paths to a node are removed from the landscape. In essence, I under this scenario represents the percentage change in the correlation length of the landscape when all the edges to the patch are removed. Edges were removed rather than entire patches, since non-inclusion of a habitat patch within the protected area will not mean that the removed patch will be converted to non-habitat immediately. Patches, which are attributed high importance index values are considered stepping stone, since removal of these patches results in a high loss of connectivity within the landscape.

To determine an order of patch importance, the patches were sorted by the maximum I value across all thresholds and then ordered according to the smallest value of this maximum I, representing the greatest importance value a patch has over all threshold distances. The least important patch was considered the patch with the lowest minmax, not including the target patches, and was removed from the analysis. This created a pruning strategy where patches, least important to connectivity, were removed based on the ordering above and then the landscape graph and values were recalculated. Subsequent patches were then removed until only the target patches remained on the landscape. A curve representing the trade-off between the area removed and landscape connectivity was created to analyze the impact of patch removal.

The recommended area to be retained for the reserve, for this analysis, was considered to be the minimum patch area necessary to ensure that a disperser does not experience a distinct change in the way in which it perceives the connectivity of the landscape. This area was determined from the area trade-off curve, by identifying the point at which a transition zone occurs (i.e. immediate steep drop in correlation length over a minimal amount of area). The recommended option for the reserve will be the option for the disperser, which requires the most area to maintain cross-elevational connectivity. If connectivity is maintained for this organism, which requires the most area, it will also be maintained for the rest of the dispersal abilities that require less area.

Figure 1-5 Land-use within the municipality of Whistler, British Columbia



RESULTS

This analysis identified a total of 56 second growth patches, which cover an area of 1236ha (Figure 1-6). The maximum dispersal capability(*d*) (greatest distance between two patches) of the intact landscape is 30,518.14m(*d*) with the cost to travel between patches ranging from 50 - 182981.2. For the purpose of this analysis 5 classes of hypothetical dispersers were examined. The "best disperser" was considered to have a maximum dispersal capability of 38500m(*d*), a value greater than the maximum dispersal capability of the landscape. The remaining classes examined are as follows: intermediate disperser "a" *d*=20000m, intermediate disperser "b" *d*=10000m, intermediate disperser "c" *d*=6000m, and "poor" disperser *d* = 1000m. With all 56 patches in tact the C*d* was 11 times greater for the best disperser (6.663736) than for the "poor" disperser (0.606625).

As illustrated by figure 1-7, when patches are removed the impact on correlation length depends on the dispersal capability of the organism. The correlation length of the poor disperser only changed 14% between the intact landscape and the landscape with only a single patch remaining, while the best disperser experienced an 85% reduction in correlation length over the same loss of area. For the intermediate dispersers, the change in correlation length that occurred between the intact landscape to a single patch landscape was as follows: "a" experienced an 89% loss, "b" an 82% loss, and "c" a 75% loss.

Another important aspect of the graph, is the location of steep changes in correlation length, which represent transitional zones where the landscape changes from a connected phase into a disconnected phase. The most noticeable drop in connectivity occurs for intermediate “a” disperser as the 48th(38% loss of area) is removed relating to a sharp drop in correlation length of 57%. Intermediate dispersers “b” and “c” have two distinct transitional zones. For disperser “b” this occurs when the 43rd (33% area removed) & 45th (36% area removed) are removed representing a respective drop in connectivity of 32% and 60%. Disperser “c” similarly had the 43rd and 45th patch removals resulting in transitional zones, which represented a change of correlation length of 17% and 61%. For the poorest disperser, there is no evident transition zone, instead the connectivity gradually decreases, as patches are removed until reaching the lowest correlation length of 0.52432 where the graph levels out at approximately 768ha. For the best disperser, there are also no distinct transitional zones, however, there are evident steps of connectivity loss representing a more gradual loss. All the intermediate dispersers reach the same lowest value of correlation length as the “poor” disperser at the following area removed: “a” at 58% area removed (517ha), “b”at 52% area removed(596ha), and for “c”at 39% area removed (751ha). For the best disperser the connectivity of the landscape only appears to level out as the last patch is removed reaching its lowest value of 0.685 with 100% of the area removed.

Figure 1-8 demonstrates that size of a patch was not the only factor, which determined at which iteration it was removed from the landscape. Although, many of the initial patches were those of smaller area, iteration 11 saw the removal of a 21.5ha patch, while a 1.5ha patch remained until the 49th iteration. Patches of small area that were removed in later iterations are considered to be “stepping stone” patches (Keitt et al. 1997). Figure 1-9 illustrates the % of area removed, the map is classified into four sections representing approximate steps of 25% area. The first 22% of area removed represents 39 or 70% of the total number of patches on the landscape. While the next class, 28% of area removed, represented only 11 patches, and the remaining 50% of the total area was represent by 5 patches.

Mapping the order of patch removal using 4 equal intervals, yields some interesting visual results, which demonstrates the importance of small patches, as stepping stones. Figure 1-10 illustrates the order of patch removal with the lighter patches representing those removed earliest and the darker patches are later removals. An interesting series of patches which demonstrates the stepping stones' importance is the 9,10,11 series of patches in the northeast section of the map. These three patches are small in comparison to some of the other patches on the map, however, 10 and 11 were removed late in the analysis and most likely represent stepping stones, while patch 9 of similar size was removed relatively earlier, and does not contribute as much to connectivity. Also patch 29, which is of relatively large size, was deemed by the analysis to have little value to maintaining the connectivity of the landscape, and was removed in the first 25% of patches.

The alternative patch retention options, which maintain the relative state of connectivity in the landscape with the minimum amount of area, are illustrated in figure 1-11, 1-12, 1-13 for the intermediate disperser levels. An alternative is not included for the poor disperser, since the landscape is already deemed to have low connectivity for this dispersal distance and only by adding patches will an alternative be created that is viewed as connected for this organism. There is also no option for the "best" disperser, since the landscape never truly becomes disconnected with patch remove, as illustrated by the lack of a distinct drop off in the area trade-off curve. The area of the alternative designs for intermediate disperser are as follows: "a" 766.75ha, "b" and "c" 795.5ha.

Figure 1-6 Second growth forest patch map, as defined by RIC (1998), in Whistler British Columbia and target patches used as input the habitat map in SELES analysis

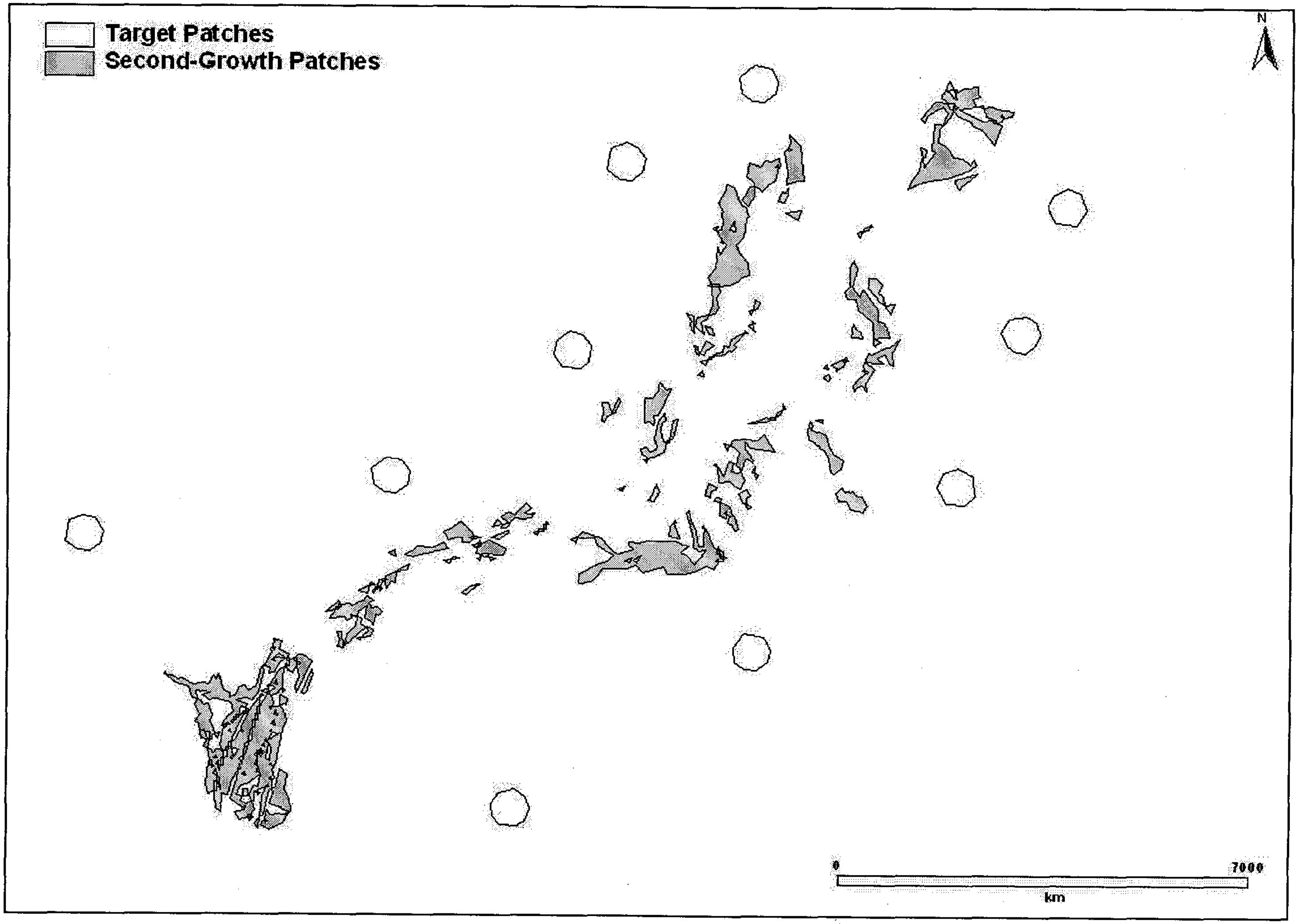


Figure 1-7 Area trade-off curve illustrating the relation between correlation length and area as patches are removed from the landscape

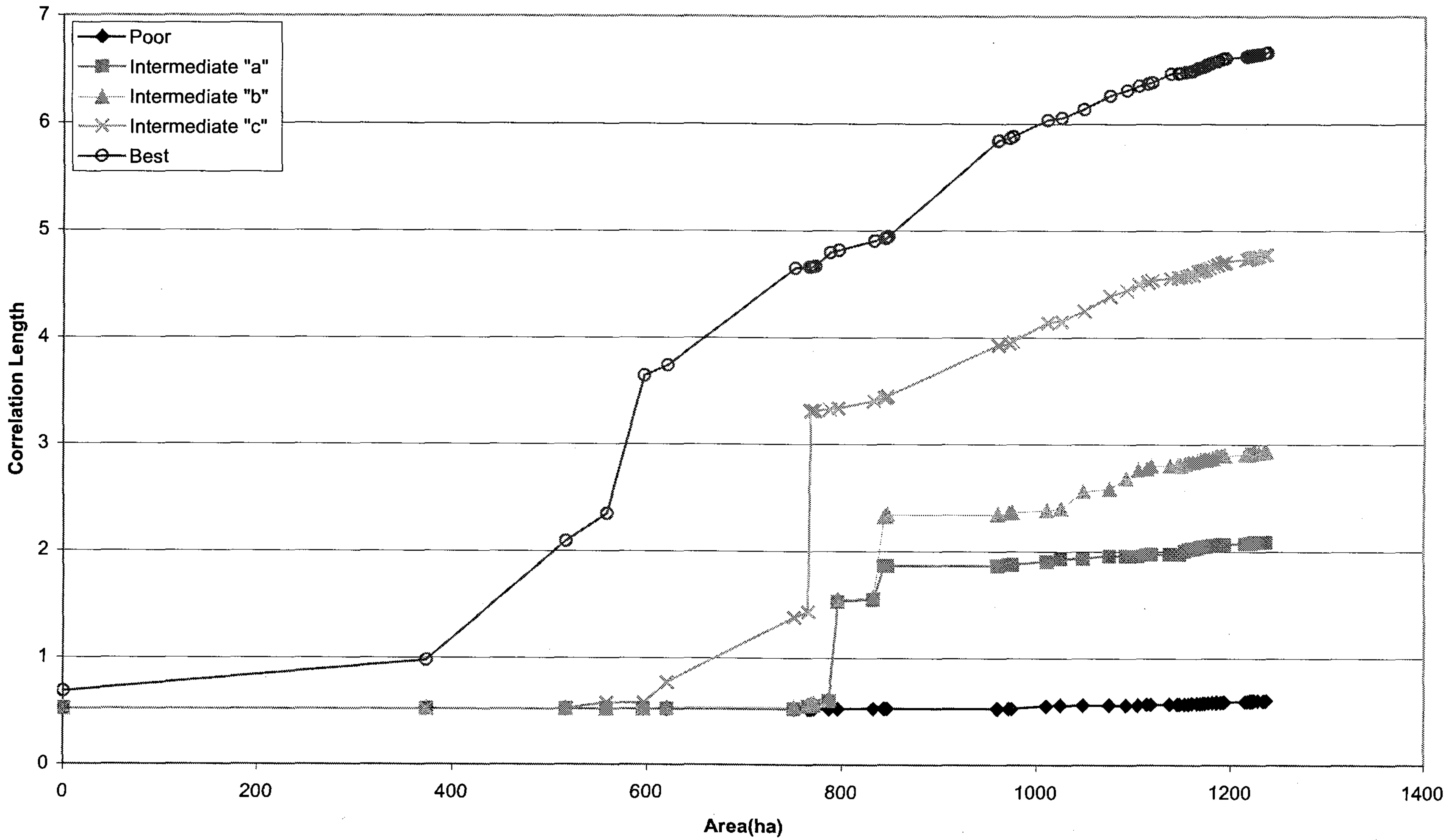


Figure 1-8 Graph depicting the area of the patch which was removed at each iteration of the patch removal simulation

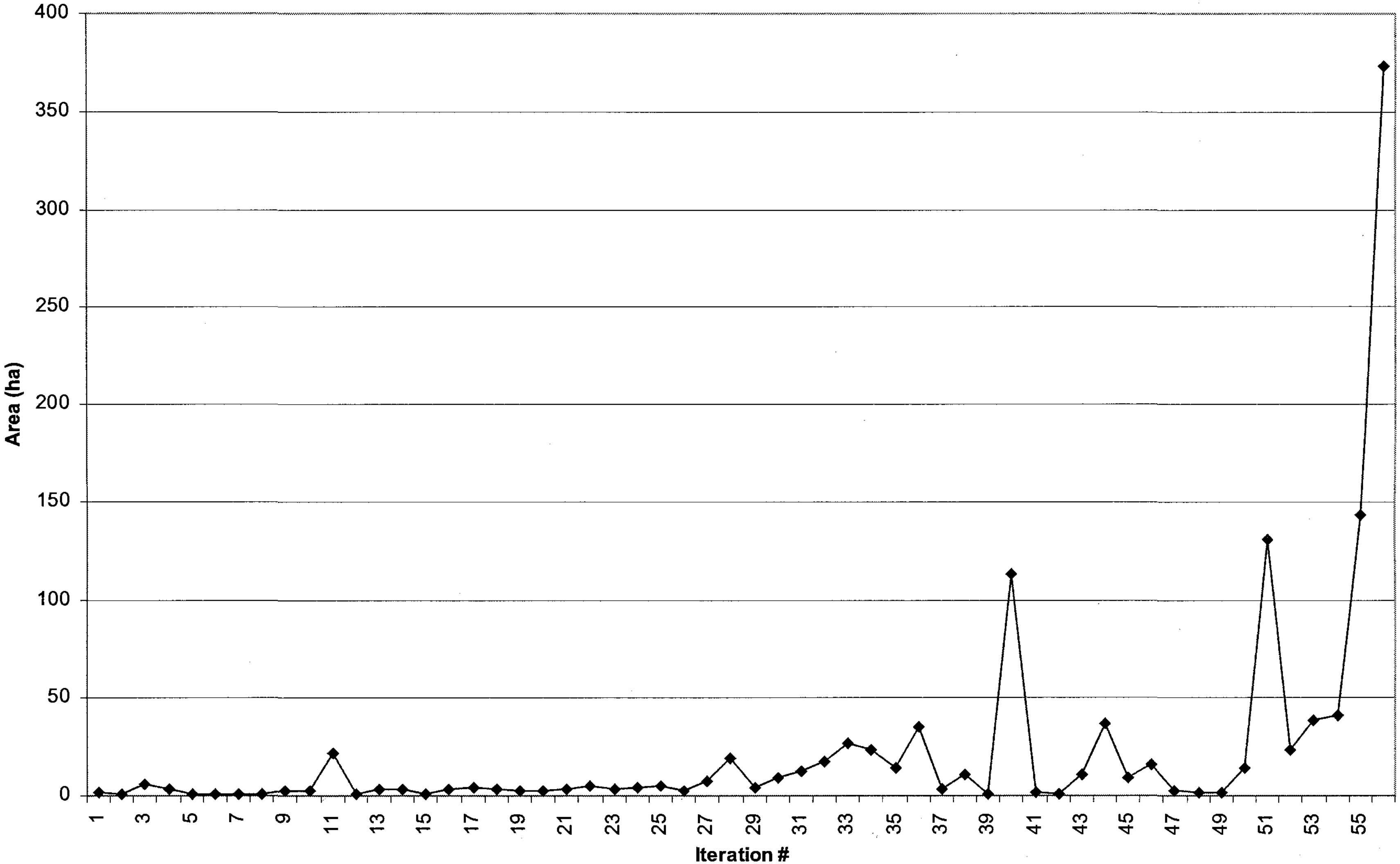


Figure 1-9 % area removed map classified by quartiles with each class representing 25% of the second-growth landscape

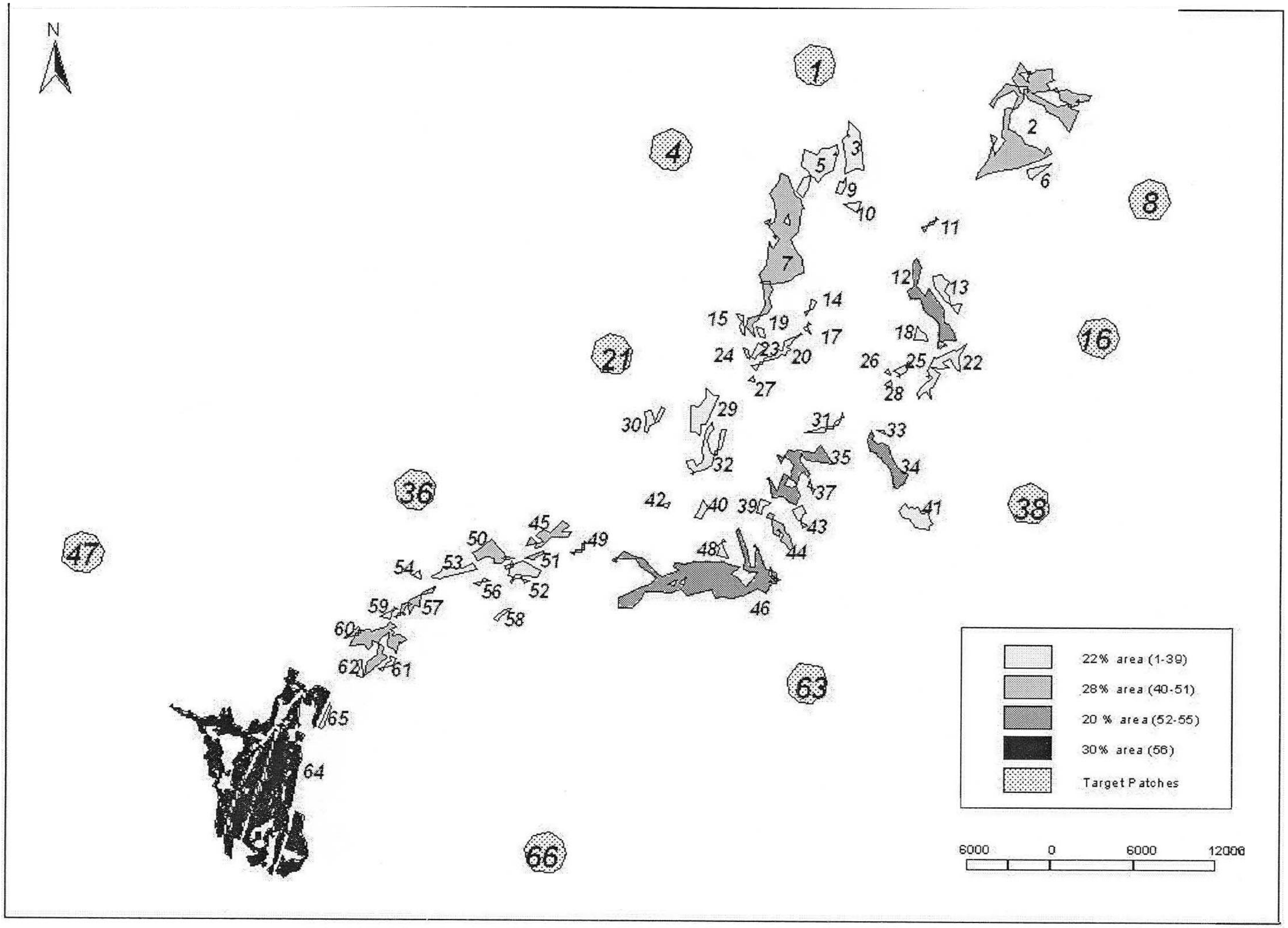


Figure 1-10 Map depicting iteration at which patch was removed classified by 4 equal intervals

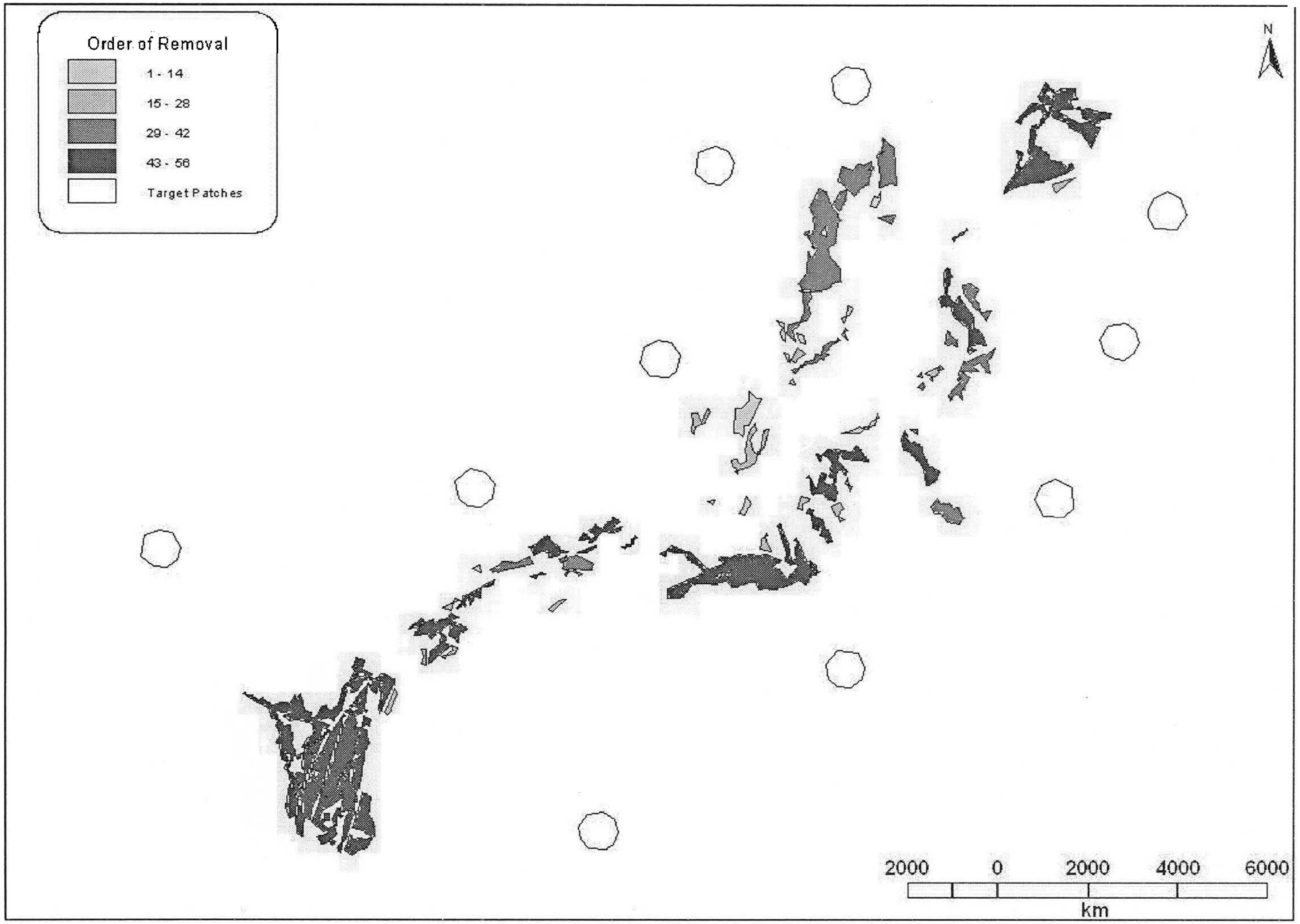


Figure 1-11 Patch retention option for intermediate disperser “a”

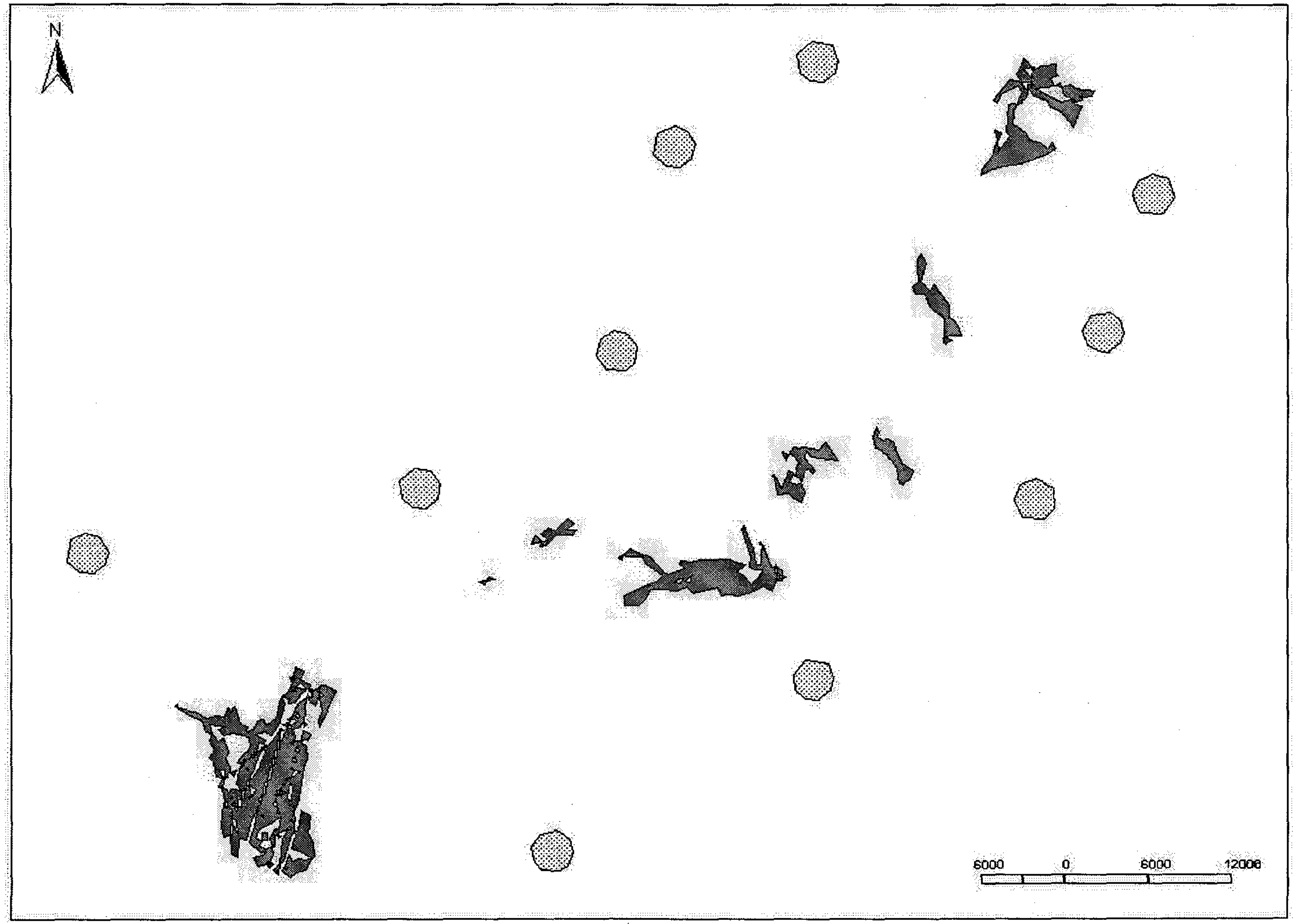


Figure 1-12 Patch retention option for intermediate disperser “b”.

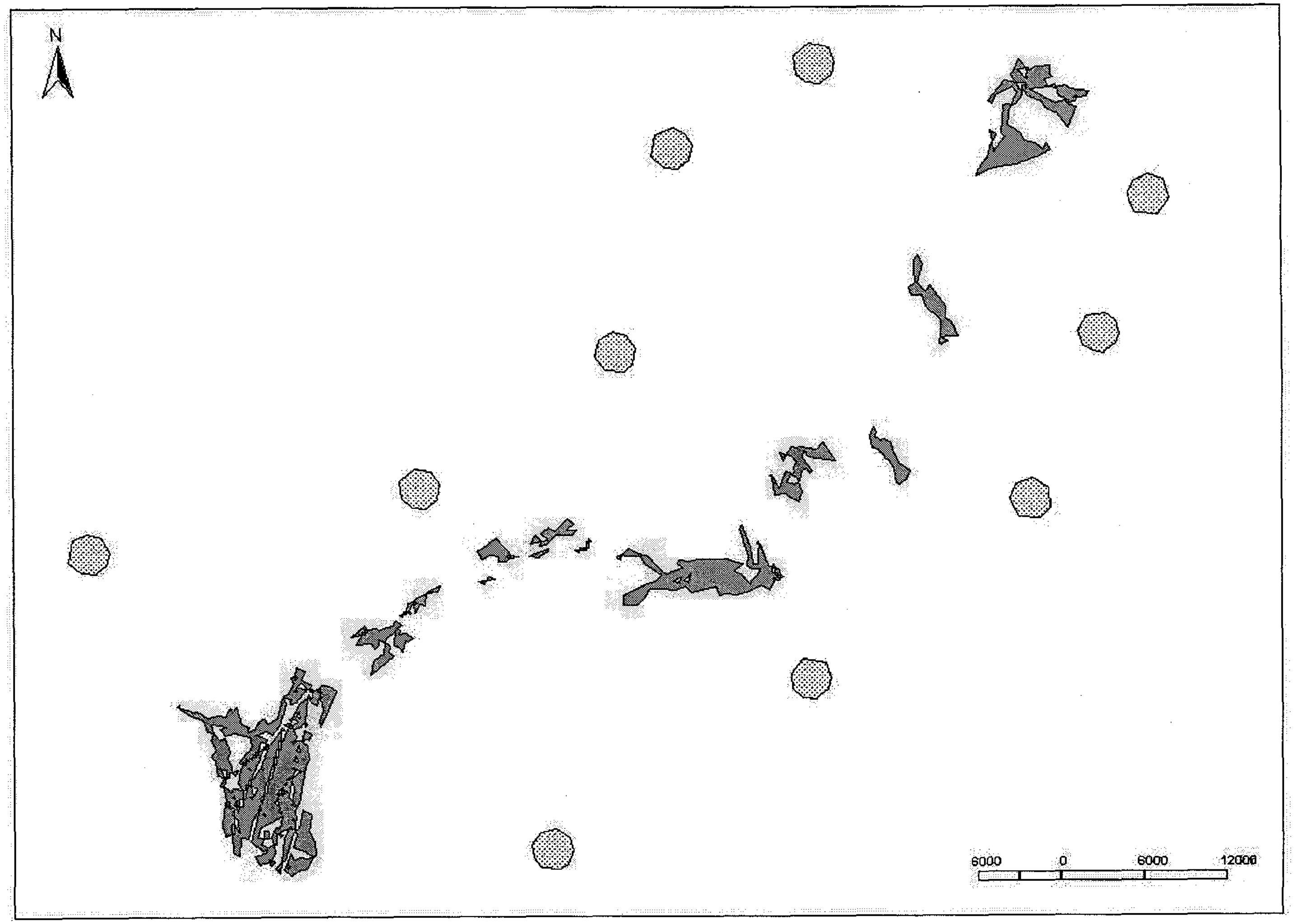
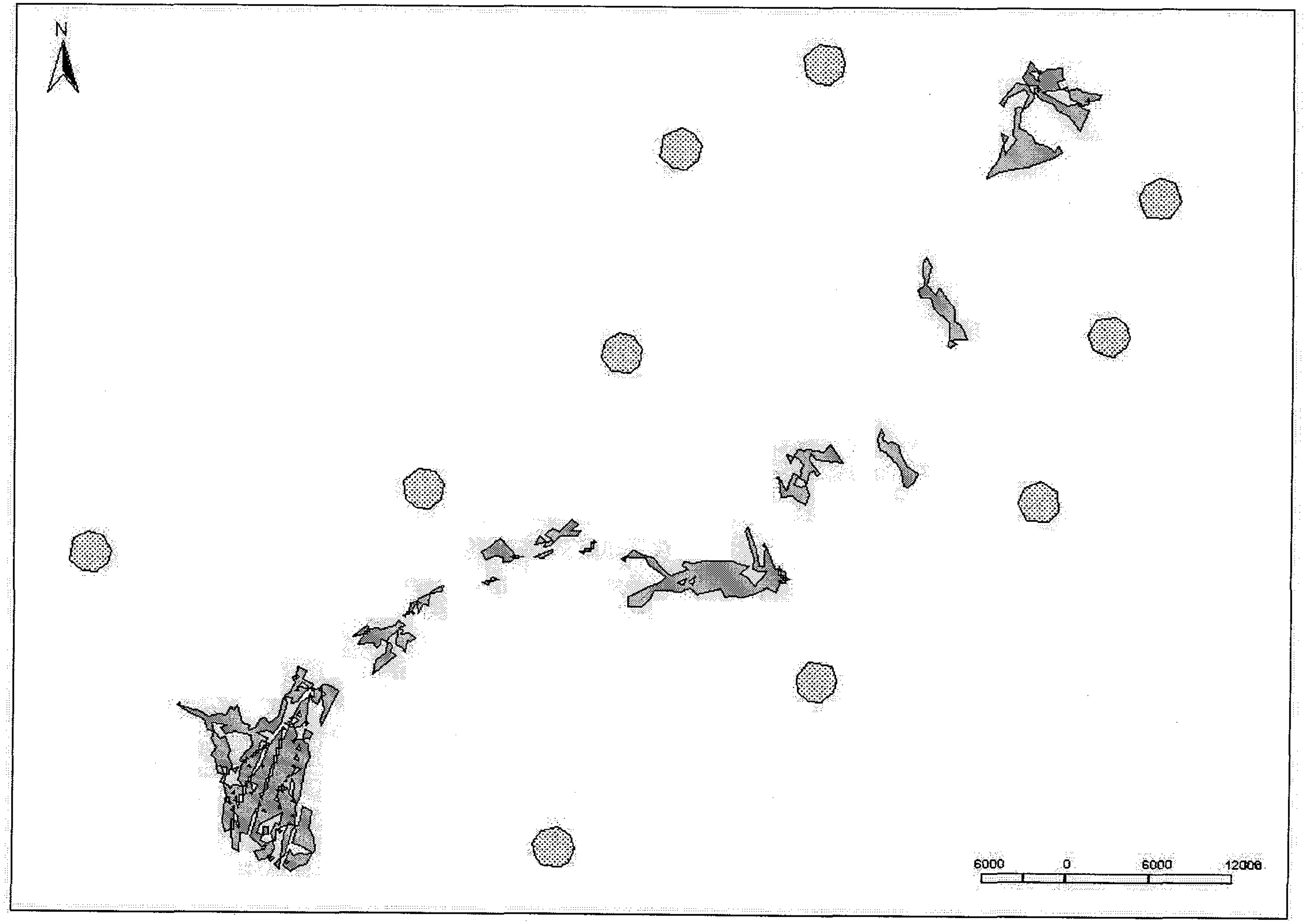


Figure 1-13 Patch retention option for intermediate disperser “c”.



Discussion

The results of this study support three main ideas. First, the differences that occur in the area trade-off curves and the final reserve alternatives support a multi-species approach, since examining only a single dispersal capability may have led to different results. Secondly, the persistence of small patches in the removal process suggests that patch size alone is not a reliable predictor of the importance of the patch to maintaining connectivity. If one was only to consider patch size in reserve design, small patches which act as stepping stones could be removed resulting in the fragmentation of the landscape for many organisms. Finally, this study illustrates that one must be cautious when determining an approach to protect connectivity in a reserve. A logical assumption when designing a reserve may be to take a cautious approach and protect connectivity for the poorest disperser in the landscape. However, as these results demonstrate the best disperser had the greatest change in connectivity when area was removed, while the poorest disperser already viewed the landscape as disconnected and removal of area had little impact on the degree of connectivity for this organism. These results therefore suggest that it maybe more cautious to protect connectivity for the "better" dispersers in the landscape.

In terms of connectivity, the patch retention alternative that would be recommended for Whistler would most likely be the alternative for intermediate disperser "a"&"b". All the retention alternatives had the same core patches within their network, and there were no particular patches, which aided movement for one disperser level that weren't present for a 'better' disperser. It is also important to remember that connectivity is only one important objective when considering a reserve design. The recommendation of this study will also be incorporated with other objectives in the decision making process, which will lead to the development of the final reserve network.

Further research, should focus on combining this methodology with a more comprehensive approach to reserve design, since connectivity alone is not sufficient to develop a comprehensive reserve that meets many conservation objectives. There is also much uncertainty that has not yet been examined with this methodology. This research did not consider that some patches may act to stop dispersal (Adrianensen et al. 2003). The potential impact of these "sponge" patches on this analysis is that if included in the network they may act to limit movement and actually lead to potential disconnection in the landscape. Other uncertainties may occur in the geospatial input data, which may impact the reliability of the landscape metrics. This issue will be the topic of the next chapter.

CHAPTER 2

Introduction

Information produced through the use of geographic information systems(GIS) is being used more frequently in the management of environmental resources (Crosetto et al. 2002; Crosetto & Tarantola 2001; Edwards & Lowell 1996; Norton & Williams 1992). Crosetto et al. (2002) note that an increasing amount of politically sensitive decisions are being made based upon the information derived from spatial models. Unfortunately, those making decisions often have little understanding about GIS and are unaware of the potential for uncertainties or error surrounding output information (Hunter 2001). Uncertainties in GIS outputs could lead to misinterpretation of results and ultimately flawed information being applied to important decisions. This may lead to poor decisions which in environmental management and conservation may have high costs due to the irreversible nature of these decisions (Norton & Williams 1992; Hunter 2001). GIS does not mitigate against this circumstance, as it derives new information without producing any measure of its reliability (Lanter & Veregin 1992). There has been an increasing interest in error and uncertainty analysis within the GIS literature, however, there has been little application of this research in ecology (Hunter et al. 1995).

GIS outputs can be impacted by both error and uncertainty. An error occurs when there is a measurable known difference between the produced information and reality, while with uncertainty this difference is unknown (Hunter et al. 1995; Stoms et al. 1992). In order to examine uncertainty, a sensitivity analysis can be used which produces alternative results, which are compared to the initial output; while error is examined by comparing the initial output to an independent measure of truth (Stoms et al. 1992). This chapter will examine the use of GIS in ecological research and in particular its recent applications in conjunction with graph theory. This chapter will also examine the awareness and mitigation of GIS uncertainty and error within ecological research. The final objective of this chapter is to present a sensitivity analysis, which will examine the uncertainty surrounding the reserve selection technique discussed in chapter one.

GIS and Uncertainty in Ecology

Ecology has many spatial problems that suit the application of GIS. GIS has been applied to many different problems in ecology including habitat modelling, home range analysis, and reserve design to mention a few (Pereira & Itami 1991;Schadt et al. 2002). Many of the products produced from GIS research are used to aid in policy development for the management of both habitat and species (Woolf et al. 2002; McComb et al. 2002).

One of the most predominant uses of GIS in ecological research is for the modeling of habitat suitability. In the late eighties, Donovan et al. (1987) introduced the application of GIS to the development of habitat suitability models. At that time, there was no mention about the potential uncertainties and errors that GIS may introduce into the model. However, since that time there have been many papers, which have addressed issues of uncertainty in these types of models (see Roloff & Kernohan 1999); most noteably, Stoms et al. (1992) who dealt directly with the issue of GIS data. Other papers, such as Bender et al. (1996) have attempted to illustrate uncertainties within the model through the use of confidence intervals, however, this examination was not directly related to GIS uncertainties. Habitat models are an important tool in decision making as illustrated by Woolf et al. (2002) whose model of bobcat habitat in Illinois contributed to the delisting of the species in this state. However, this model presented no indication of potential uncertainties associated with GIS spatial data. In the papers examined for this study the methodology to deal with uncertainty varied from simply mentioning potential issues, such as cell size selection in a model of red squirrel habitat, to providing complete sensitivity analyses (Schadt et al. 2002; Pereira & Itami 1991). However, many of these papers do not directly mention uncertainties related to spatial data and GIS processing (see McComb et al. 2002; Woolf et al. 2002; Roloff & Kernohan 1999). Uncertainties related to GIS are especially of importance when dealing with spatial outputs where locations of boundaries may have significant impacts on decisions.

Another common use of GIS in ecology, is to map information such as species distribution and species richness (Elith et al. 2002; Conroy & Noon 1996). Elith et al. (2002) discuss many issues of uncertainty in habitat maps and note that the verification of models against independent data that does not have a spatial component will not determine problems in the spatial predictions. Conroy and Noon (1996) also present issues of uncertainty surrounding biodiversity mapping such as the Gap analysis program. They suggest problems may lie in the coarse resolution at which species richness data is collected and mapped. They also note that mapped representations of species richness data are sensitive to errors from both the input data and the models themselves. The uncertainty surrounding these maps is important since many of the spatial patterns observed are considered as first approximations for land use decisions such as reserve locations (Conroy & Noon 1996).

A more recent use of GIS in ecology has been the application of graph theory to landscape scale studies. The initial use of graph theory in landscape ecology was by Cantwell & Forman (1993) who used graph theory to compare and identify landscape patterns. The intent of their methodology was for use in examining how policy changes can impact landscapes over time. In their study, they made no mention of the potential uncertainties associated with this modelling technique or the spatial input information. In the late nineties, graph theory was applied to the study of landscape fragmentation and connectivity. Keitt et al. (1997) first examined the application of graph theory to connectivity and developed a methodology to quantify habitat connectivity at multiple scales. Their methodology also allowed for the ranking of patches according to their value in maintaining the connectivity of the landscape. Later Bunn et al. (2000) expanded upon this methodology examining landscape connectivity from the perspective of the American mink (*Mustela vison*), and the prothonotary warbler (*Protonotaria citrea*), within a metapopulation context. This paper expanded upon Keitt et al.'s (1997) original paper by applying least cost modelling to estimate the functional distance between patches rather than the Euclidean approach used by Keitt et al. (1997). Bunn et al.'s (2000) paper also did not discuss or examine the potential uncertainties or errors associated with graph theory or its spatial inputs. Urban and Keitt (2001) formerly introduced the application of graph theory to both connectivity analysis and ecology. In their paper, they presented methodology using graph theory and landscape metrics to examine landscape connectivity and a patch's importance to maintaining this connectivity. However Urban and Keitt's (2001) paper also does not directly address issues of uncertainty or errors within the model and how this could potentially impact results and decision made based upon their approach. The importance of graph theory to ecology, as described by Urban & Keitt (2001), is that it allows connectivity to be measured both from population processes and landscape pattern creating a process-based measure, which is more applicable to conservation planning. Due to the increasing use and value of graph theory in ecological research and conservation planning, it is important to address the issue of uncertainty and error in GIS and its impact on the output of the model.

Sources of Uncertainty and Error in the Application of Graph Theory

There are many potential sources of uncertainties and error in the application of graph theory, both within the model and the spatial input data. The uncertainties of the model arise from ecological assumptions, its algorithms and also the landscape generalization, which occur as a basic premise of graph theory. The spatial input data that the model uses for its analysis are prone to all of the sources of uncertainties and errors related to the application of GIS. The graph theory model requires spatial representations of the travel cost, habitat patches, and study area, which for this study were all produced in Arc View 3.2 (ESRI 2000).

Sources of uncertainty related to GIS can be due to the initial collection of the data, input of the data into GIS, or the processing and transformation of the data within GIS (Davey & Stockwell 1991;Thappa & Bossler 1992;Burrough & McDonnell 1998). It is important to examine all sources of potential error or uncertainties within this model, in order to better understand its limitations. This section will review the sources of error and uncertainty related to the application of graph theory to reserve design presented in chapter 1.

GRAPH THEORY UNCERTAINTIES & ASSUMPTIONS

The conversion of reality into an artificial system requires many assumptions to create a version of reality that can be easily handled by these systems. Modelling habitat relations in a computer is especially difficult and uncertain due to the problems accounting for randomness and chance events within these relationships (Davey & Stockwell 1991). An example of how this uncertainty may occur in the application of graph theory is the lack of consideration of patch characteristics, which may cause it to act as a sink rather than facilitating movement. Also graph theory simplifies movement basing it only on travel cost and distance between patches, ignoring other potential driving forces such as availability of mates or resource quality. Another principle not accounted for in graph theory is the dependence of habitat utilization on population levels (Davey & Stockwell 1991). Any of these factors have the potential to alter results produced by the application of graph theory to reserve design.

There is an extensive body of research in Geographic Information Science examining the uncertainties and errors within GIS (Hunter et al. 1995). The application of graph theory to reserve design is susceptible to all these errors due to the spatial input data required for the analysis. The primary introduction of uncertainty and error into GIS occurs in the initial collection and input of spatial data into a GIS system. Uncertainties and error can be introduced due to imprecision or error in measurements, temporal inconsistencies in data sets, or digitizing errors when the data is brought into GIS

(Thappa & Bossler 1992; Burrough & Mc Donnell 1998). The most common methods of creating spatial data is to digitize information from maps or input data from remote sensing sources such as air photos. When digitizing information from other map sources, the user places a cursor over an object and in a sense traces points over the elements in the map. These points are then used to generate vector features such as points, lines and polygons. The error in digitizing is related to the precision of the user when they create the initial trace points and also any operation, which acts to join the created points or generalize the resultant features (Youcai & Wenbao 1997). There are also many potential sources of uncertainty and error when processing remote sensing data for use in analyses. Some potential sources for these uncertainties and errors are as follows: sensor systems, geometric rectification, data conversion, classification, and data generalization (Lunetta et al. 1991). The uncertainties and errors that result from the collection and creation of GIS spatial data could potentially impact the arrangement of nodes and the definition of edges within a graph theory model. This may alter the distances between the patches, hence altering dispersal distances and potentially changing the order of patch importance within the reserve selection technique. The appearance of the cost surface may also be impacted by these initial sources of uncertainty and error due to changes in the location of feature's boundaries. This change may alter the classification of the landscape when the raster surface is created for the definition of the cost surface. This will also impact the definition of the edges due to potential changes in the least cost pathways.

Uncertainties may also occur during processing of GIS spatial data for use in the graph theory model. These processing errors may include the conversion of data from vector to raster formats, querying information from data layers, and buffering features

(Veregin 1989;Congaiton 1997;Morris 2003). The conversion of data from vector to raster formats can cause uncertainties related to the classification of cells, and the location of boundaries (Congaiton 1997). For example, when a vector layer is gridded a cell may contain more than one value, known as a "cut cell", since this cell may only be attributed a single value, the accuracy of this classification is not 100% (Congaiton 1997). In the graph theory model, this error may alter both the definition of the cost surface and the boundaries of patches inputted into the analysis. Creating new surfaces by querying spatial data is another source of potential uncertainty in processing. In this analysis, second growth forest patches were created by querying a dataset created by the Terrestrial Ecosystem Mapping (TEM) initiative. Morris (2003) notes that the definition of such a query can be problematic, due to the fuzzy nature of such query events. Features in the landscape may be partial members to the query set and maybe excluded in the analysis due to the crisp nature of a query. The definition of second growth patches in the original data set was not crisp, second growth forests could have been represented as either the 1st 2nd or 3rd ecosystem component therefore, the definition of the second growth patch classification was uncertain. (An ecosystem component is considered a habitat type, such as second growth forests, which in combination with other habitat comprises an ecosystem. The numerical order of an ecosystem component refers to the prevalence of that habitat in the system, with the 1st ecosystem component being the most prevalent and the 3rd component being the least prevalent.) In this analysis, as discussed above, these sources of uncertainty could potential alter both the distance between patches due to node location, the presence and absence of patches, and also the definition of edges, due to potential differences in the least cost pathway.

Another common problem in raster-based analyses, as in the graph theory model, is the size of the cells used in the grids or the spatial resolution of the analysis. The cell size can alter the quality of the conversion and the accuracy of the boundaries (Congaiton 1997). If too large a cell size is chosen, features may be lost from the analysis, especially narrow features such as roads or rivers. However, if too small a cell size is selected, it can lead to extremely long processing times (Congaiton 1997). In regards to graph theory, spatial resolution could potentially alter both the cost and the patch input surfaces. The appearance of the cost surface could change because of increased classification accuracy through the use of smaller cell sizes. This increased accuracy may lead to the occurrence of barriers in areas while larger cell sizes may result in the feature not being classified. These barriers to dispersal would substantially alter the location of least cost pathways and the order of patch importance to connectivity. The location of the patch boundaries and the number of patches would also have the potential to change, since smaller cell sizes could yield more patches through either those that are classified due to smaller spatial resolution or the splitting of other patches in the analysis where gaps in between the patches were too small to be identified at coarser resolutions. These sources of uncertainty have a greater potential to alter the final order of the importance of patches to connectivity, due to the impact on the patch number and boundary location. There is also potential for significant change in the cost surface that could perhaps alter the least cost pathways identified in the original analysis.

The cost surface is another potential source of uncertainty as a result of the subjective nature of its definition. Due to the importance of the cost surface in defining edges and dispersal distance, this uncertainty would most likely have one of the greatest impact on any analyses using graph theory. Adrianensen et al. (2003) stress the importance, when defining a cost surface, of having both high quality input maps and relevant choices for both resistance values and landscape features used in the cost analysis. Defining cost is a difficult task especially within ecological systems as many potential problems can occur within the surface (Adrianensen et al. 2003). One of the most pertinent problems for this analysis is the ability of an organism to cross over linear features without occurring the high cost that is associated with the feature. Linear features when converted to raster format may appear as a line of diagonal cells joined at a single corner. If diagonal movement is allowed in the model then an organism can move through this "hole" in the surface without incurring any of the costs, and in the case of the highway in the Whistler Valley, this could potentially have significant implications for the final results (Adrianensen et al. 2003). The choice of the resistance values and potential "holes" in the surface will alter the definition of the edges and ultimately impact the order of importance for patches within this analysis. Due to the array of methodologies and features that can be included in a cost surface, it is important to examine the sensitivity of the analysis to alternative cost surfaces.

In this chapter, we will examine the sensitivity of the Rothley & Rae (unpublished) ecological application of graph theory, to uncertainties within the spatial input data. The specific uncertainties that will be examined include the spatial resolution of the analysis, definition of the cost surface, and classification of the patch map. Various alternatives of the graph theory analysis will be produced and compared to determine the range of potential outputs. A discussion of how uncertainties in the input spatial data could potential impact the decisions made in Whistler B.C regarding the protected area network will also be discussed.

Methodology

As previously mentioned, three aspects of uncertainty in regards to the spatial input data were examined for this research including: classification of the patch map, spatial resolution of input surfaces, and definition of the cost surface. Uncertainty was examined using a sensitivity analysis, where input surfaces were varied and the analysis discussed in chapter one repeated. Results produced were then compared with the original results of chapter one, and analyzed for changes in previous trends.

Geospatial Data Creation

Classification Data Creation

The uncertainty with regards to the classification of the patch maps was related to the definition of second growth forests. In chapter one, second growth forests were defined as structural stage 5 within three ecosystem components, as defined by RIC (See Waldron 1999). However, there are three separate ecosystem components within the TEM data which includes 1st ecosystem component or dominant component, 2nd ecosystem component or sub-dominant component, and 3rd ecosystem component or trace component. The inclusion of both 2nd and 3rd ecosystem components may create patches that are not truly representative of a homogeneous second growth forest. Defining the patches in this manner may have included some areas with other features such as old growth forest already deemed within the PAN objectives as protected. Also, including more area alters the boundaries of the patches present in the map, which could potentially impact the connectivity of the landscape.

In order, to compare how the definition of the second growth patch map impacted the analysis, two patch maps were created by querying Whistler's TEM database. The first patch map, was queried again for structural stage 5, however, only those areas within the 1st ecosystem component were considered, creating a map of all the patches in the area that were dominantly second growth forests (Classification analysis 1)(Figure 2-1). The second patch map was also queried for structural stage 5 within the 1st ecosystem component, and also included stage 5 within the 2nd ecosystem component, creating a map representative of the dominant and sub-dominant patches of second growth forests (Classification analysis 2)(Figure 2-2). These two patch maps were then merged with the target patches utilized in the original analysis to create the high to low elevation movement across the valley. The maps were then rasterized with a cell size of 50mX50m, and reclassified as binary with the value one representing patches and zero representing the matrix between the patches. The cost and area input maps were left as the maps used in the original analysis.

Figure 2-1 Classification of habitat patch map considering only second growth forests within the 1st ecosystem component. Numbers represent the patches identification assigned by SELES

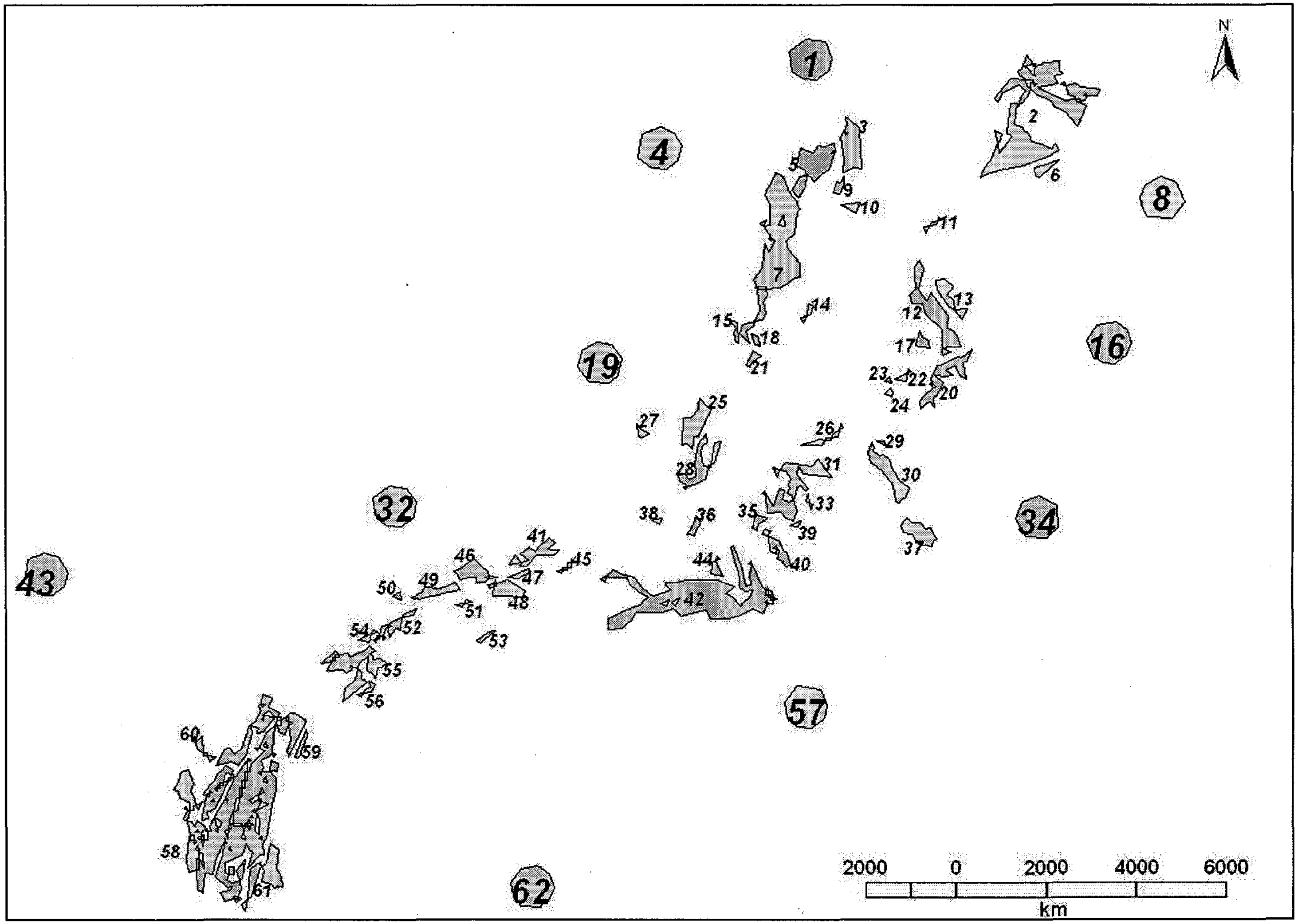
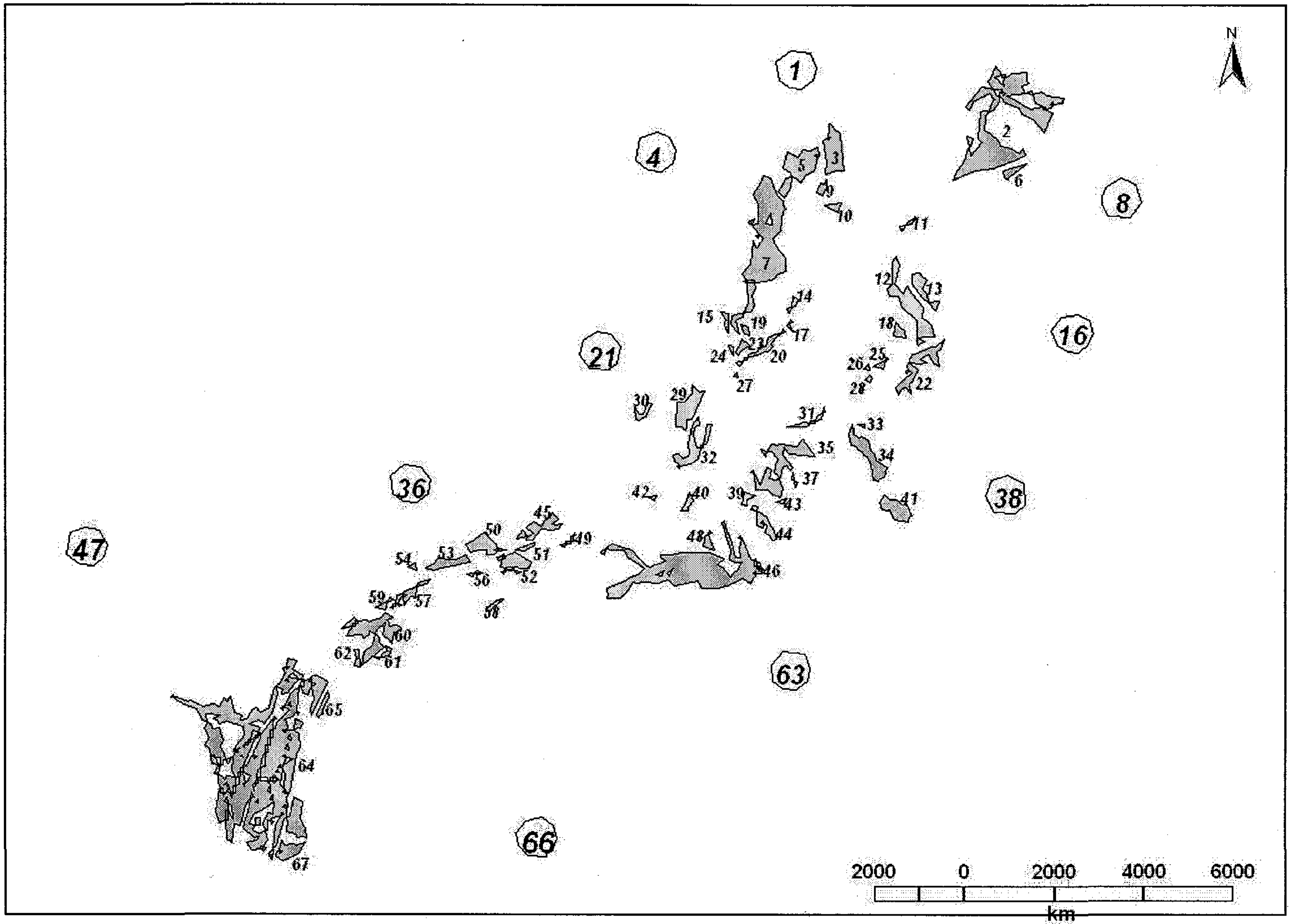


Figure 2-2 Classification of patch map considering only second-growth forests within the 1st & 2nd ecosystem component. Numbers represent patch identification assigned by SELES



Resolution Data Creation

The spatial input data in the original analysis was produced at a cell size of 50m X 50m. In order, to examine how the cell size or spatial resolution of the analysis impacts the final results of the connectivity analysis each of the inputs including the patch map, area map and cost surface were recreated at cell sizes of 10m, 20m, 30m, and 40m. The patch and area map were simply rasterized again from the vector format at each of the specified cell sizes (Figure 2-3-2-6). The cost surface however, could not just be rasterized again, due to the number of processing steps in the creation of the surface. To create the cost surface at the various cell sizes the elevation contours were first converted to grids at each of the specified cell sizes in Arc View 3.2a (ESRI 2000). There were four separate coverages which represented elevation in Whistler, and the resultant grids were merged into a single grid. This created a digital elevation model of the Whistler valley, which was converted to slope surfaces using the spatial analysis extension for Arc View 3.2a (ESRI 2000). The slope map was then converted to an integer surface. The final step in creating the cost surfaces was adding the high cost travel areas to the map, which included: the major highway, developed areas and water features. Each of these features was rasterized at the cell sizes specified above. The highway feature, areas where the simulated organism could potential move across the highway without incurring any of the cost, were identified and area was added to the feature, which would force movement across at least one high cost cell of the highway. This added area was then rasterized at the corresponding cell size. The high cost areas were then incorporated into the slope surface by replacing the cell value that corresponded with the feature with the highest slope value, which was 65. The final step in the creation of the cost surface was to replace all the zero values in the surface with the value one, since SELES does not allow for movement across cells without cost. Figure 2-7 displays the four spatial resolutions of the cost surface that were examined in this study.

Figure 2-3 Second growth patch map at resolution cell size 40m. Numbers represent patch identification assigned by SELES

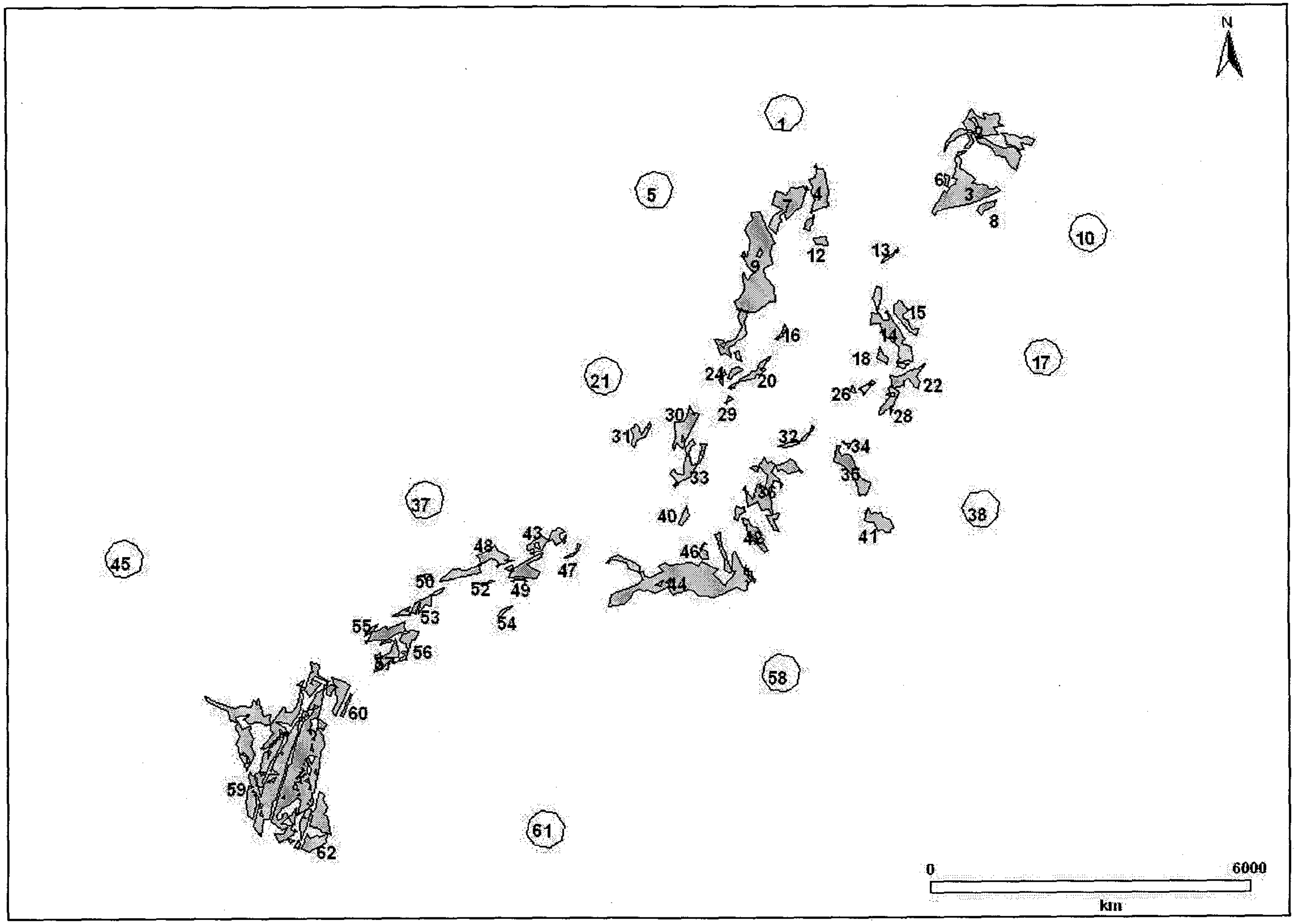


Figure 2-4 Patch map at resolution of cell size 30m. Numbers represent patch identification assigned by SELES

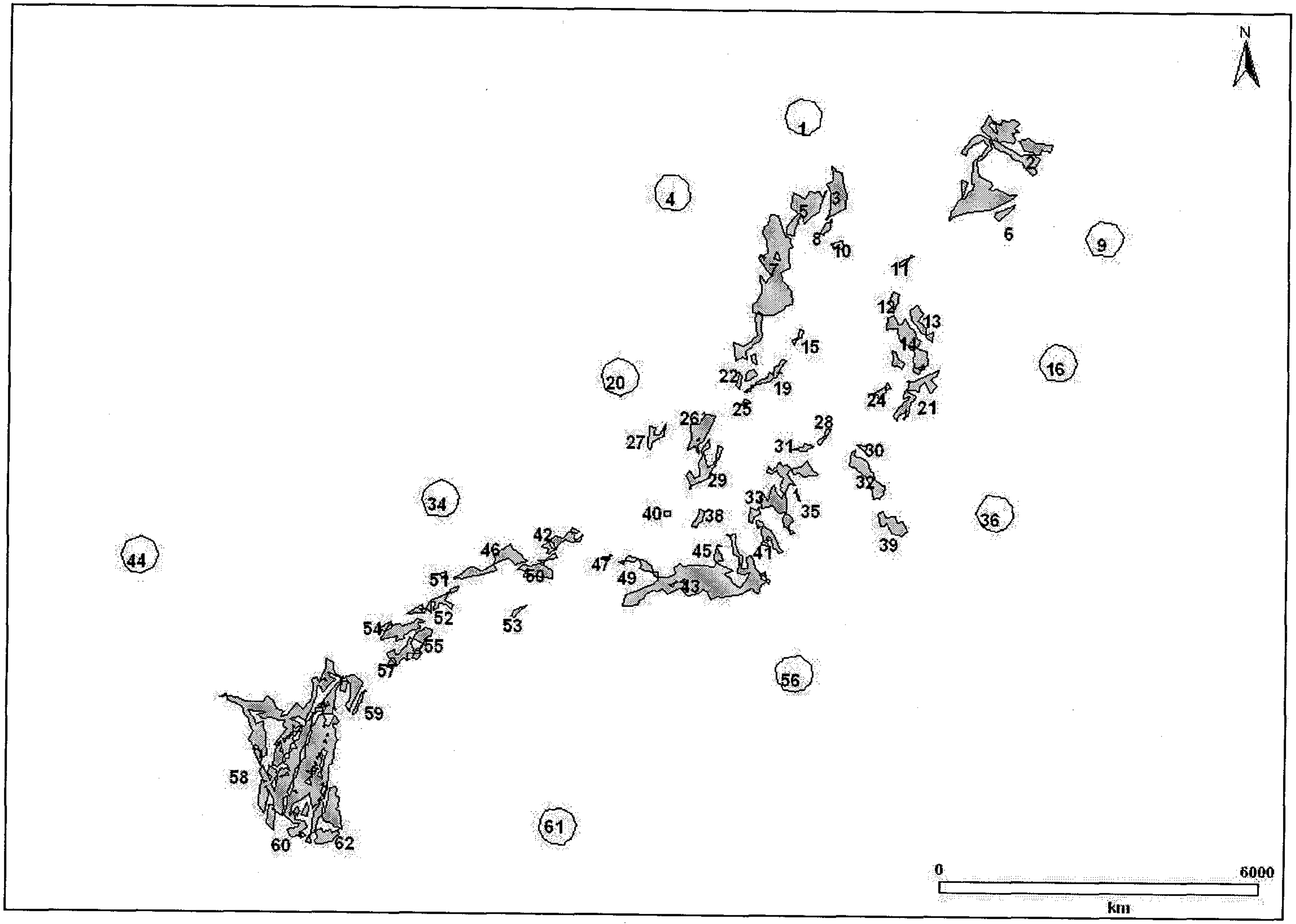


Figure 2-5 Patch map at resolution cell size 20m. Numbers represent patch identification assigned by SELES

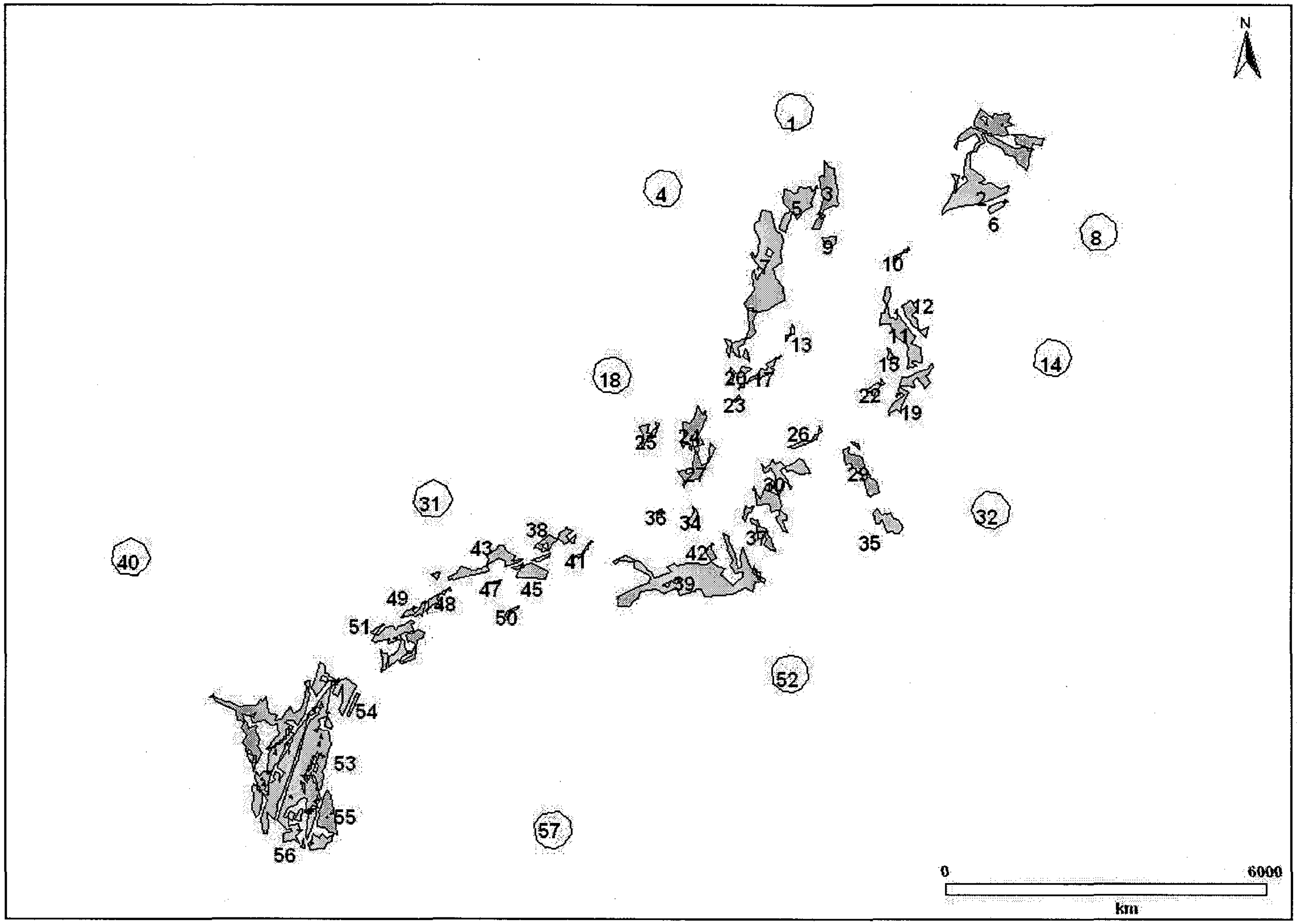


Figure 2-6 Patch map at resolution cell size 10m. Numbers represent patch identification assigned by SELES

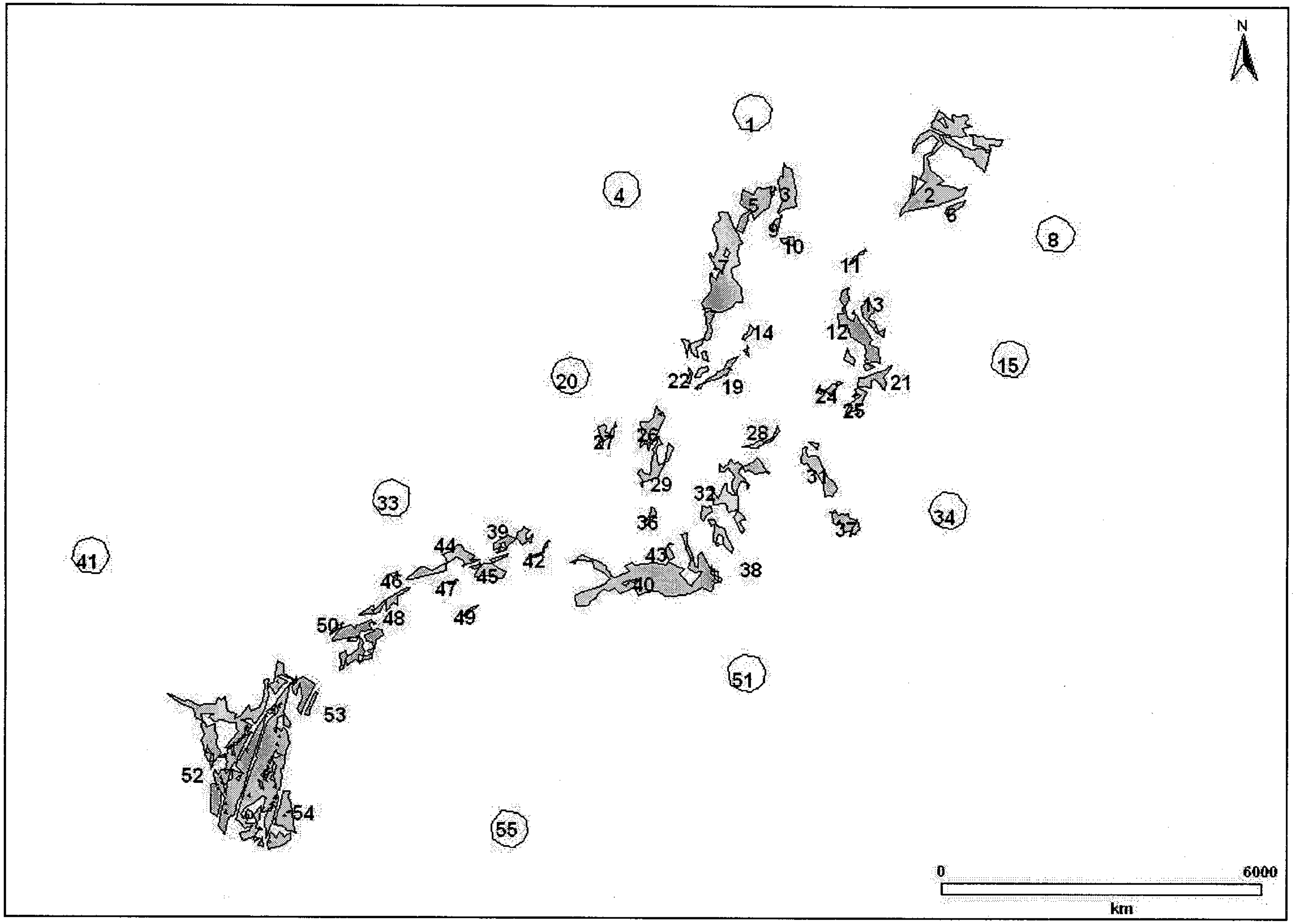
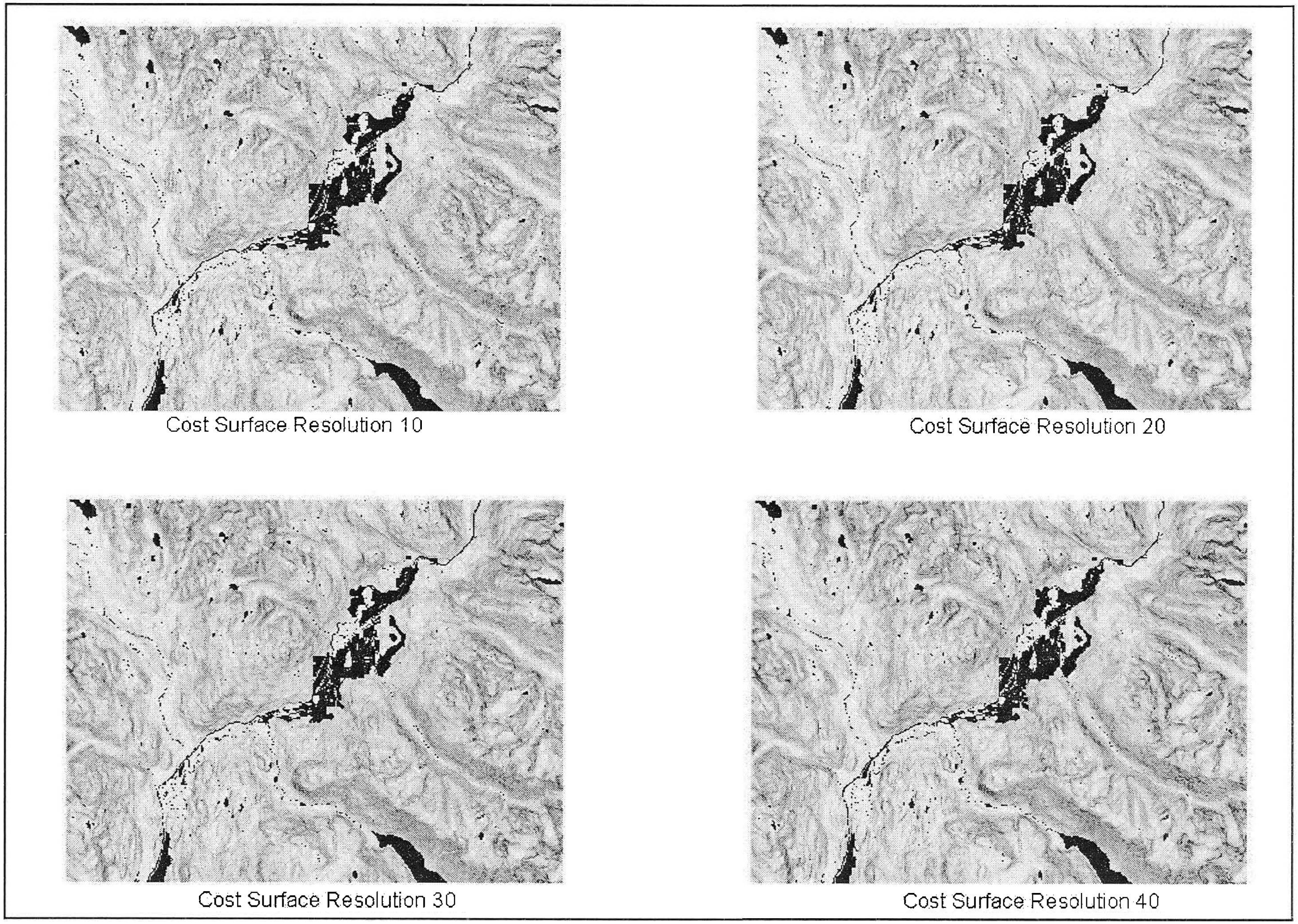


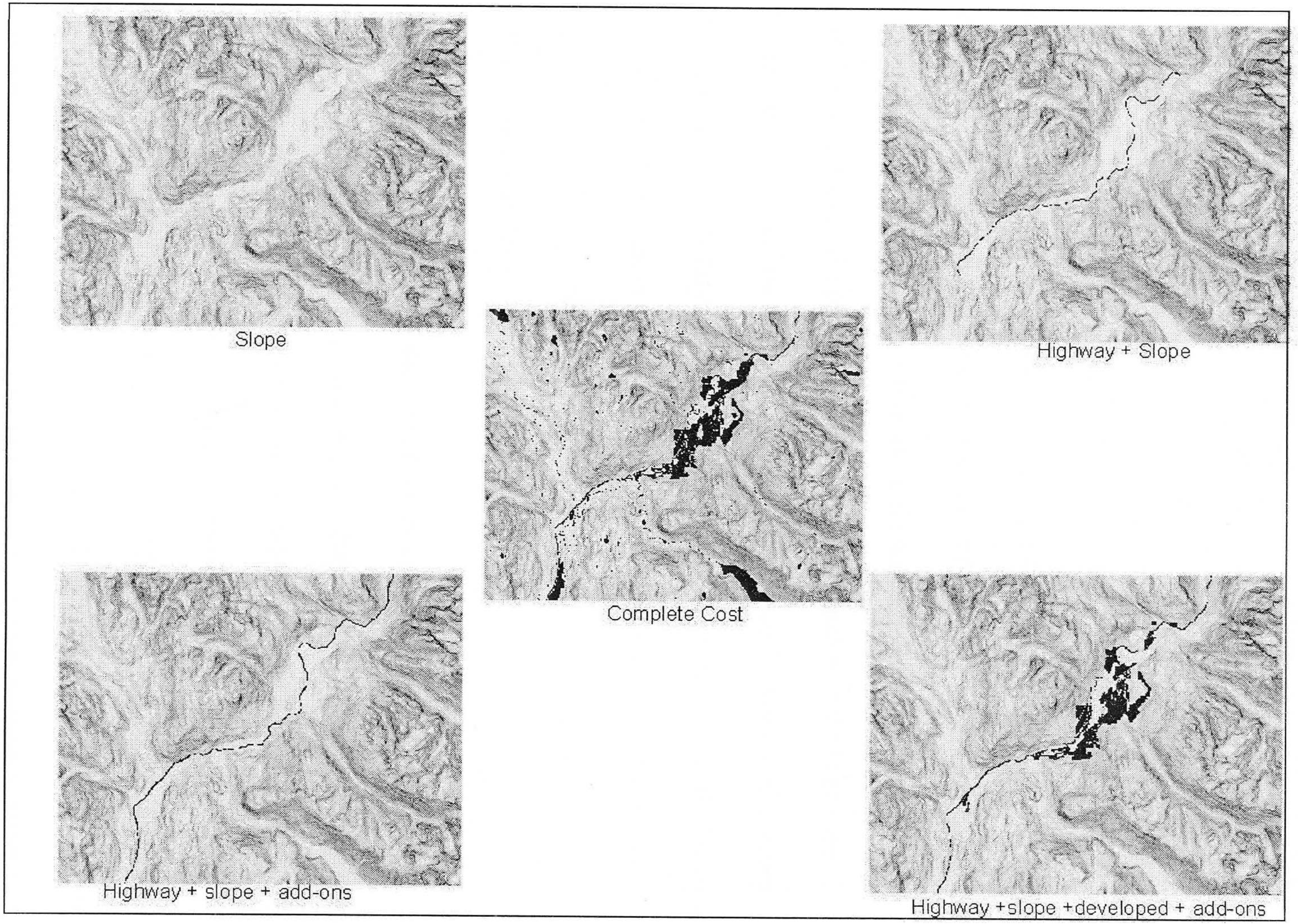
Figure 2-7 Comparison of the four input cost surfaces representing cell sizes of 10m, 20m, 30m, 40m utilized in the resolution analysis



Cost Data Creation

There are many ways which a cost surface could potential be developed for an area. In Whistler, the approach chosen was to use slope as a basis to define movement cost, with the assumption that slope would be a cost parameter, which was attributable to multiple species. Areas that were considered of high cost to animal movement were then added to the surface as barriers to dispersal. Cost could have also been defined by assigning movement cost to habitat types, however, this often represents only a certain group of organisms. Since the protected area network (PAN) design takes a multi-species approach, it was decided that the sensitivity analysis re-examine the slope cost surface with various modification to the high cost areas. Five different cost surfaces were examined in the sensitivity analysis. These surfaces were created by removing the highway, developed areas, and water from the original cost surface, assuming the original surface represented the highest level of movement cost in the valley. The surfaces were created with 50mX50m cell size and utilized the original slope surface as the base map. The first cost surface added the high cost features of the highway, highway add-ons, and developed area, replacing the cell values on the map where the features were located with the highest slope cost of 65. The next two cost surfaces represented the slope cost surface with the high cost area of highway and highway add-ons for the second surface, and just the highway for the third surface. The third surface represents the highway feature without added areas, and allows organisms to move across the highway in some areas without incurring any of the cost associated with this feature. The fourth surface represented just the slope cost values with no areas of high cost represented on the map. The final surface utilized in this analysis represents movement without varying cost, where each cell was given an identical value creating a uniform surface. Since all cells have the same value the least cost distance is the shortest path between two nodes, which geometrically is the straight-line distance, thereby basing connections on Euclidean distances. Figure 2-8 illustrates the visual differences between all the cost surfaces utilized in this study. The patch and area maps for this analysis were kept as the maps utilized in the original analysis.

Figure 2-8 Comparison of the input cost surfaces (excluding uniform cost surface) utilized in the cost analysis



Results

Classification Uncertainty

Table 2-1 Comparison between landscape statistics of original analysis and classification analyses

	Original	Classification analysis 1	Classification analysis 2
Maximum Cost	182981	182981	182981
Minimum Cost	50	50	50
Patch Area	1236	1153	1223.5
Number of Patches	56	52	57

Table 2-1 compares the landscape statistics for the original analysis performed in chapter one with the two analyses, which altered the classification of the second growth patches. Overall the landscapes appear to have only minor differences between the varying classification analyses. The maximum and minimum cost paths are identical for all the analyses. However, the patch area, and number of patches in the landscape all vary. For the first classification analysis (considering only the 1st ecosystem component), the area of the patches is reduced by 83ha and there are 4 fewer patches in the landscape. While the second classification analysis, which considered both 1st and 2nd ecosystem components had 12.5 ha less area and one more patch in the landscape than the patch map in chapter one's analysis.

The area trade-off curve for the classification analysis did not yield many substantial differences from the original analysis (Figure 2-9,2-10). Although, one minor difference occurs in the first classification analysis, where the best disperser experiences a more distinct change in correlation length at an approximate area of 600ha(Figure 2-9). This drop represents a type of phase transition, where there is now a distinct difference between how the organism perceives the landscape between the two zones (i.e sudden immediate drop of connectivity with little area removed). This is a noteable difference from the original analysis where the best disperser experienced no transitions during any of the patch removals, which suggested that there was not a specific group of patches, which ensured the connectivity for this organism, and connectivity just decreased as the area of the landscape was reduced (Figure 1-7).

A similar trend is notable to the original analysis, where predominantly small patches tend to be removed earlier in the analysis while the larger patches often were removed near the end of the analysis (Figure 2-11,2-12). However, stepping stone patches were still evident in the analyses, which are illustrated by the several small patches of less than 5ha that were removed after the 30th iteration in both classification analyses. The pattern of the patch removal, however, was slightly different, especially between the original analysis (Figure 1-8) and classification analysis 1 (Figure 2-11). In classification analysis 1, two larger patches of 21.5ha and 19.5ha where removed around iteration 10 while only one large patch was removed at this stage in the original analysis and in classification analysis 2. The remainder of the iterations were similar to the original analysis for both classification analyses.

The order of patch removal is also illustrated in figures 2-13 and 2-14. In comparison to the original analysis (Figure 1-9) the maps of patch removal order appear very similar to both classification analysis 1 (Figure 2-13) and classification analysis 2 (Figure 2-14). The majority of the last group of patches removed, distinguished by the darkest colour, remained the same. This suggests that the most important patches to maintaining connectivity in the valley were not impacted by the classification query that built the patch map used in the analysis.

The suggested patches to be included in the patch retention option are those patches present in the area before a “transition zone” occurs on the area trade-off curve, where one of the dispersers experience a steep drop in connectivity of the landscape. For the classification analysis, the reserve for classification analysis 1, which included only the second growth patches present in the first ecosystem component reserve, would be comprised of a total area of 756.25 ha (Figure 2-15). This area is attributed to the transition zone experienced by both intermediate dispersers “b” and “c”. In classification analysis 2, which considered second growth patches from both the first and second ecosystem component would suggest a reserve that conserved 827.75 ha, which again is related to the transition zone that occurs for intermediate dispersers “b” and “c” (Figure 2-16). In comparison to the original analysis’s suggestion of maintaining 795.5ha (Figures 1-11 & 1-12), the two alternative analyses would have produced both a lower estimate from classification analysis 1 and a higher estimate, classification analysis 2, of the area that should be maintained to ensure the connectivity of the valley..

Figure 2-9 Classification analysis considering only the 1st ecosystem component area trade-off curve illustrating the relation between correlation length and area as patches are removed from the landscape

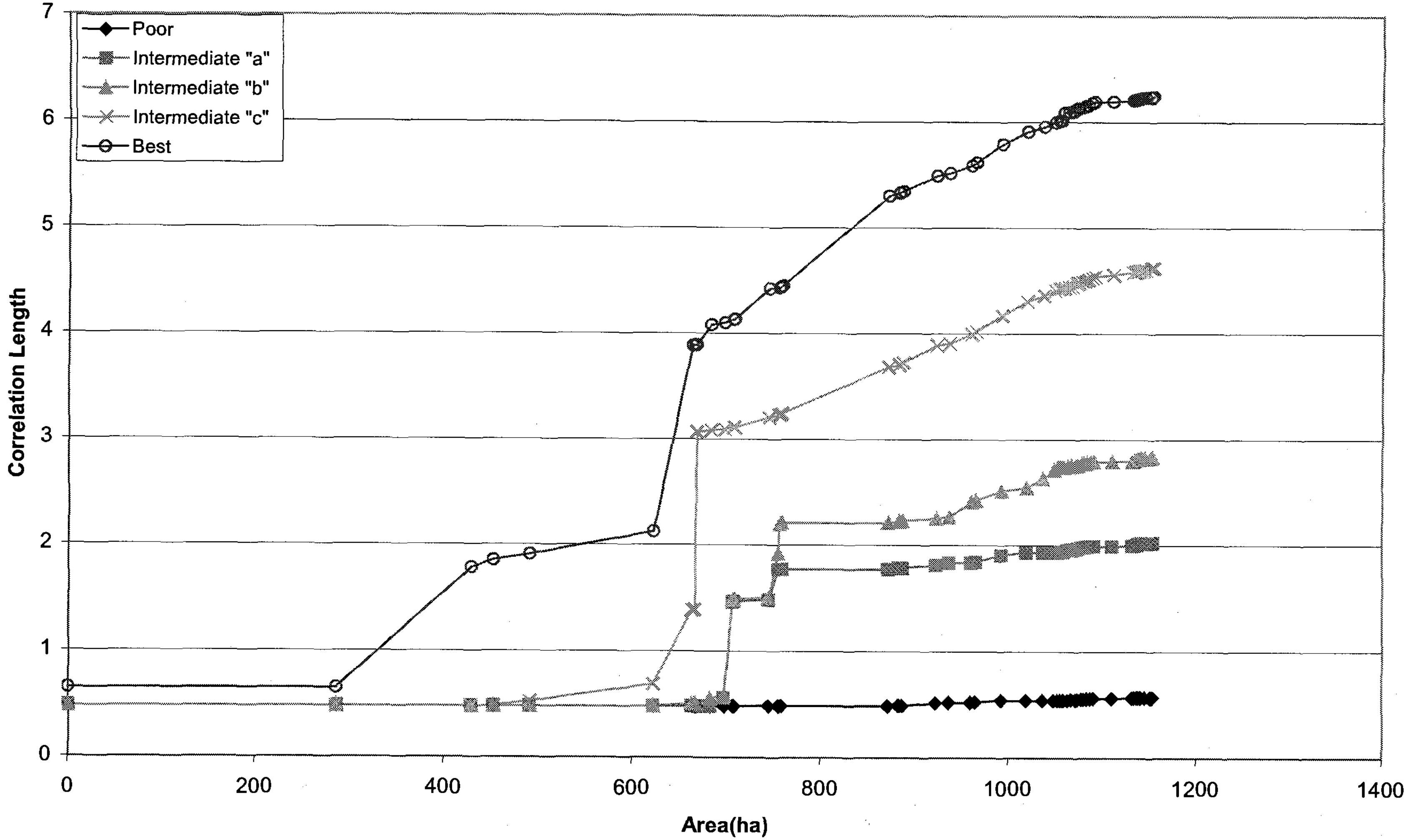


Figure 2-10 Classification analysis considering only the 1st & 2nd ecosystem component area trade-off curve illustrating relation between correlation length and area as patches are removed from the landscape

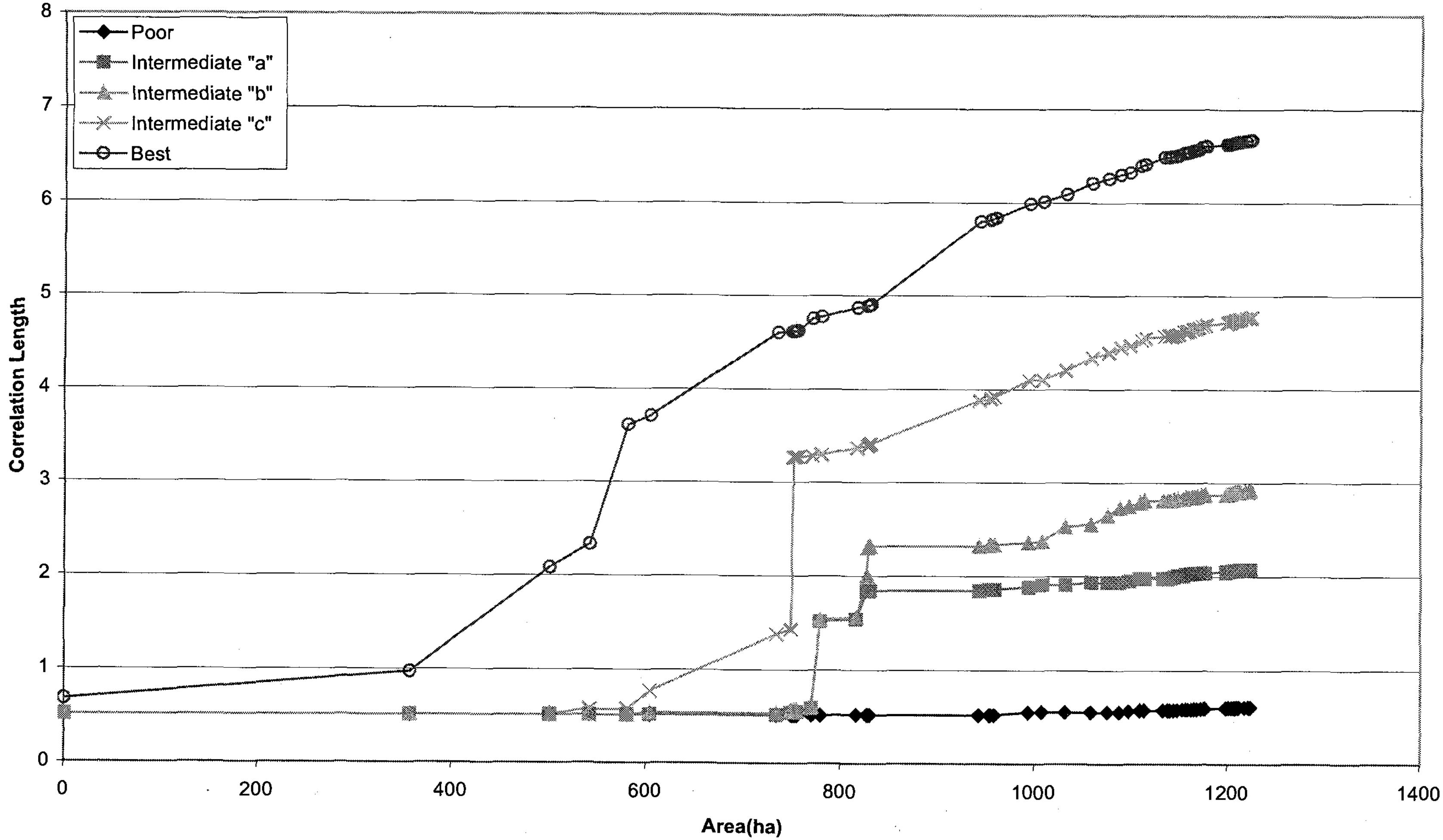


Figure 2-11 Classification analysis considering only the 1st ecosystem component graph of patch area in relation to the iteration of removal during the simulation

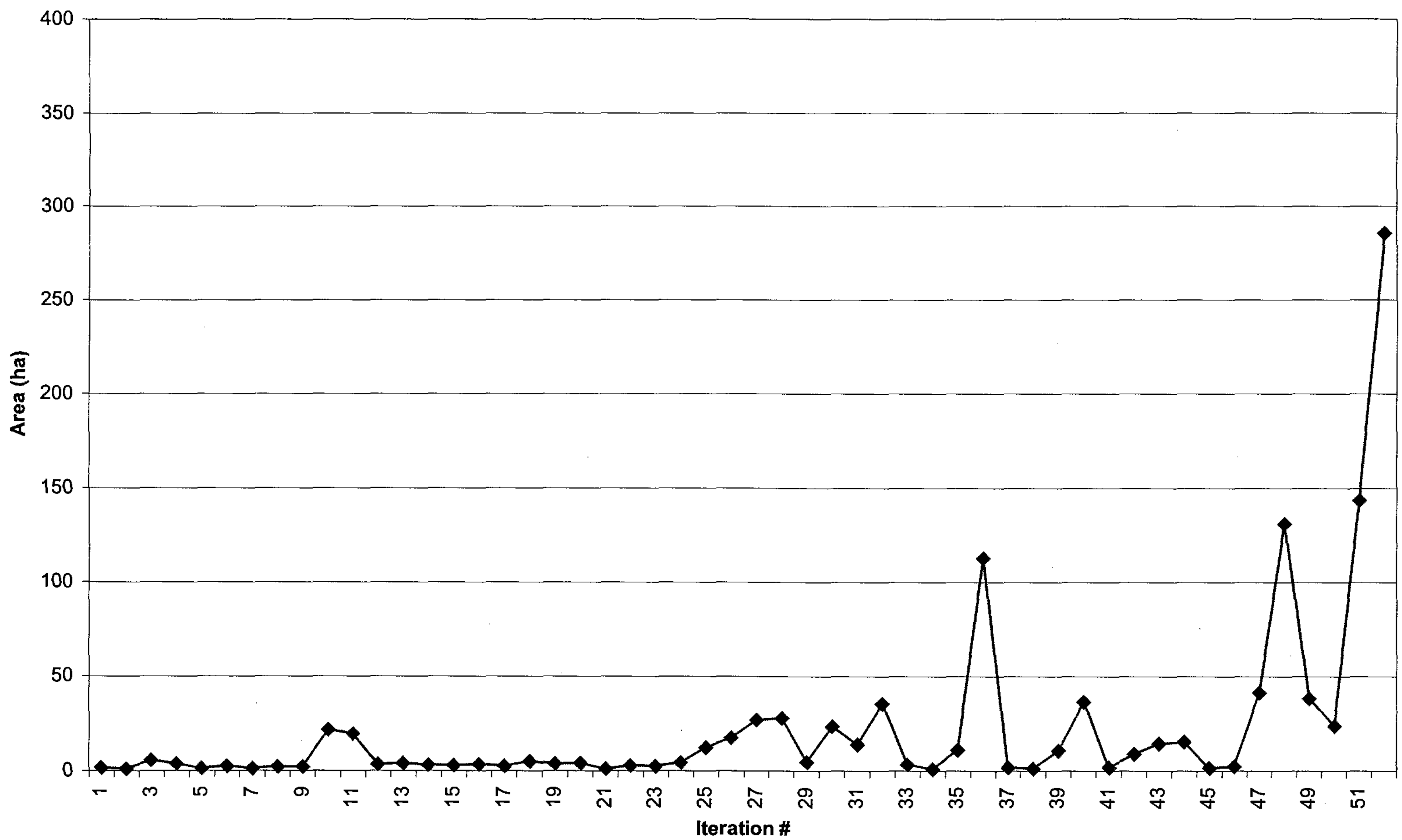


Figure 2-12 Classification analysis considering only the 1st & 2nd ecosystem component graph of patch area in relation to iteration removed during the simulation

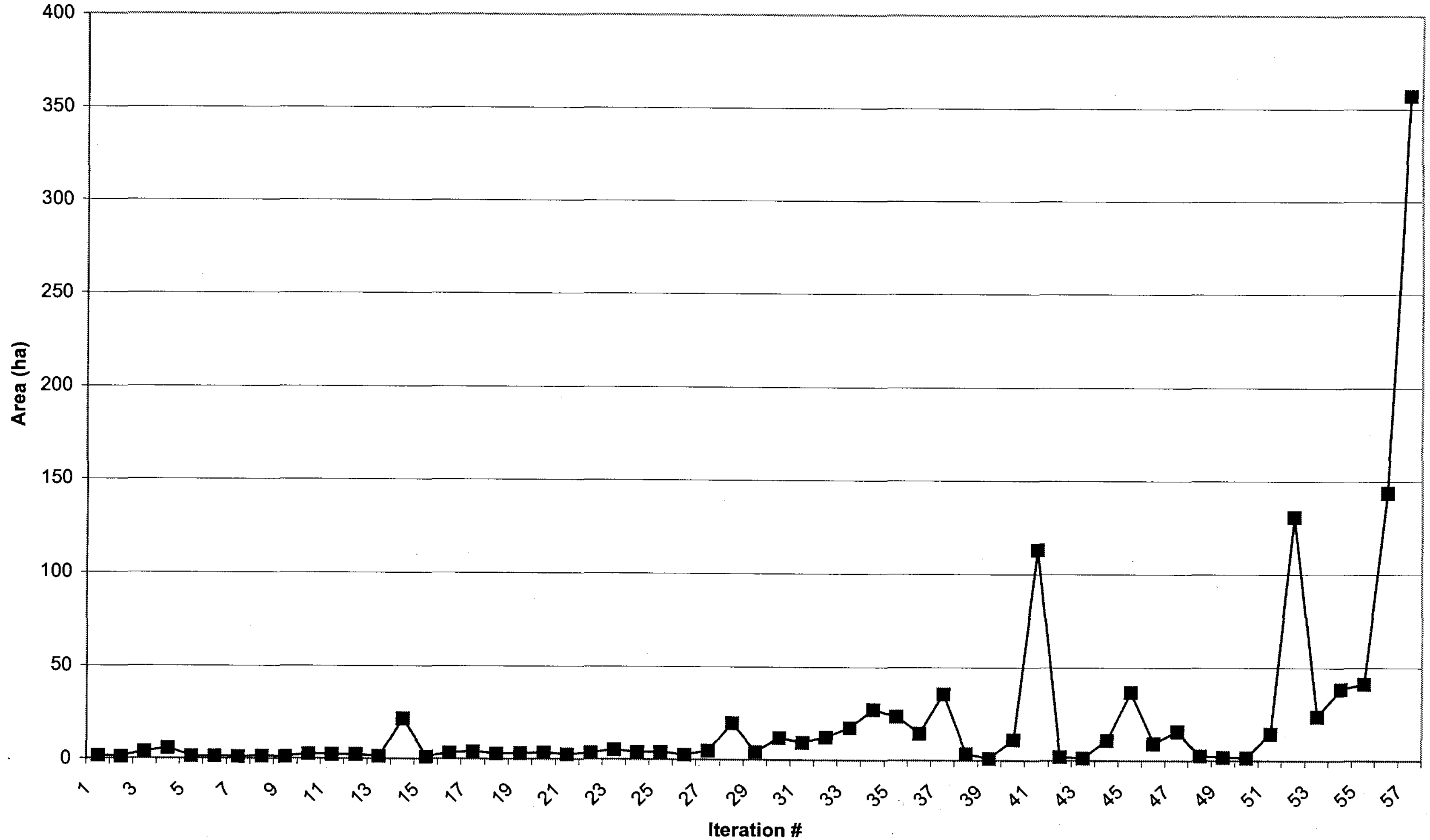


Figure 2-13 Classification analysis 1 (1st ecosystem component) map representing order of patch removal classified by 4 equal intervals

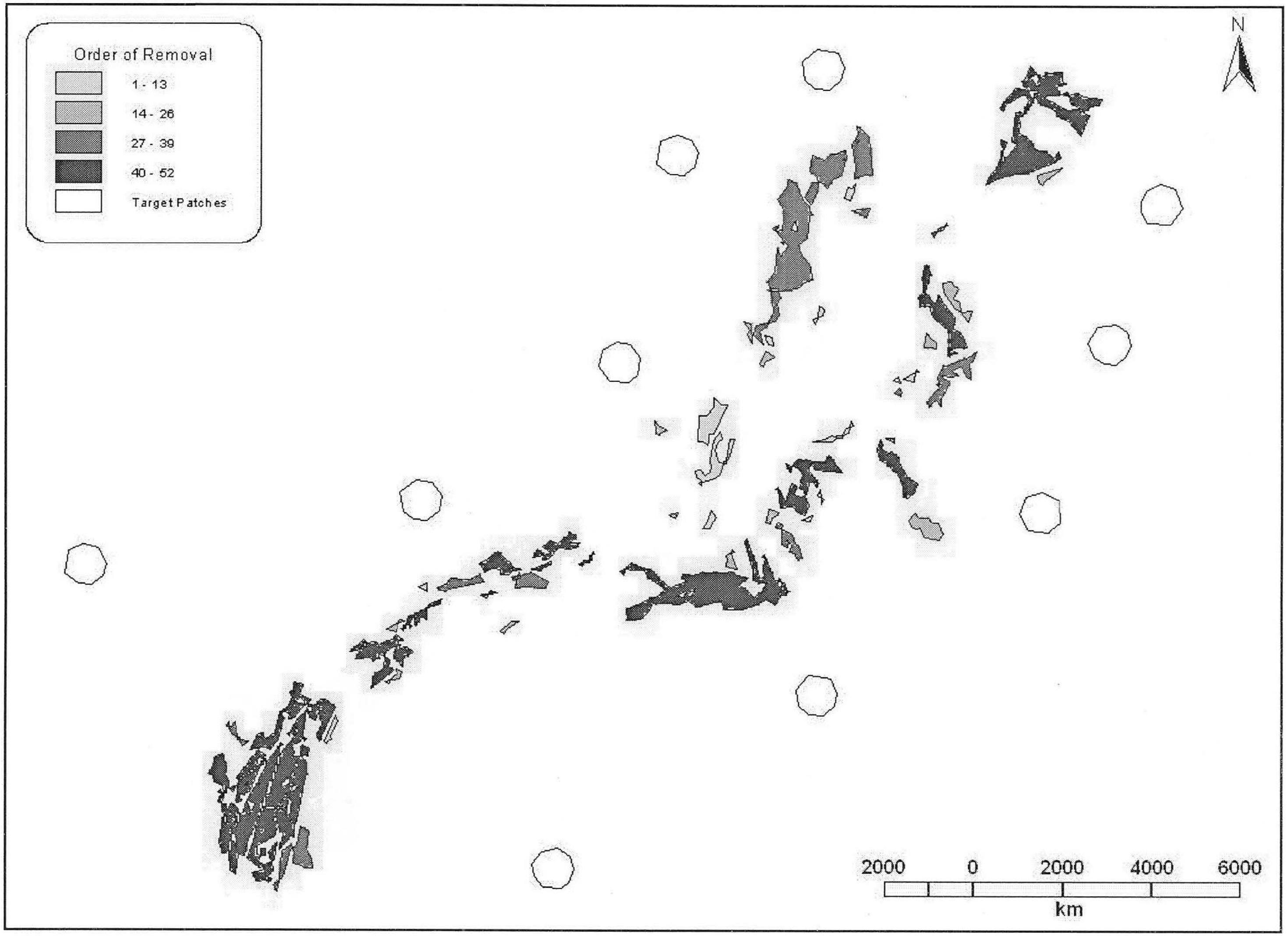


Figure 2-14 Classification analysis 2 (1st & 2nd ecosystem component), map represents order of patch removal classified by equal intervals

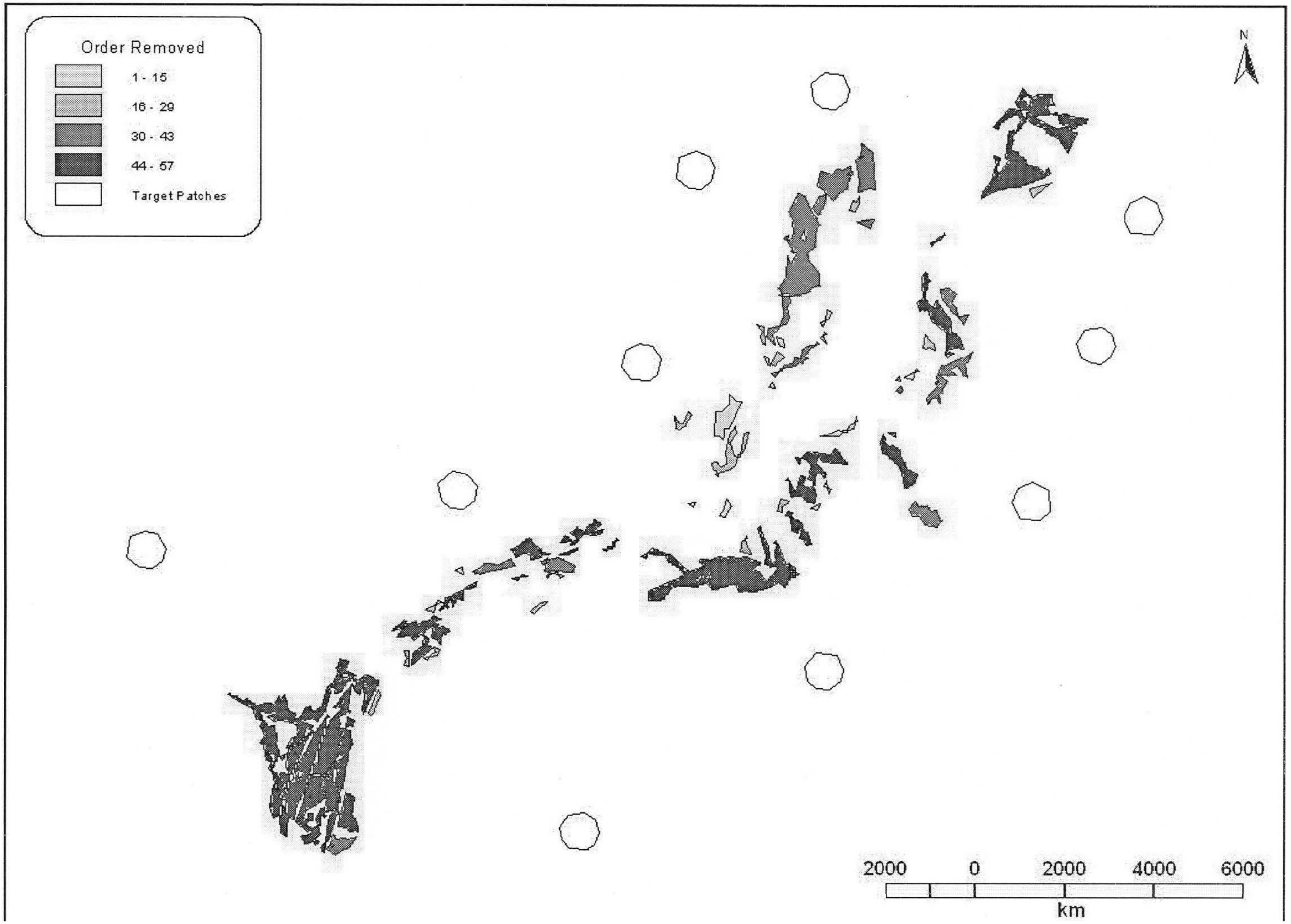


Figure 2-15 Patch retention option for classification analysis 1st ecosystem component. Highlighted patches indicate patches to be maintained

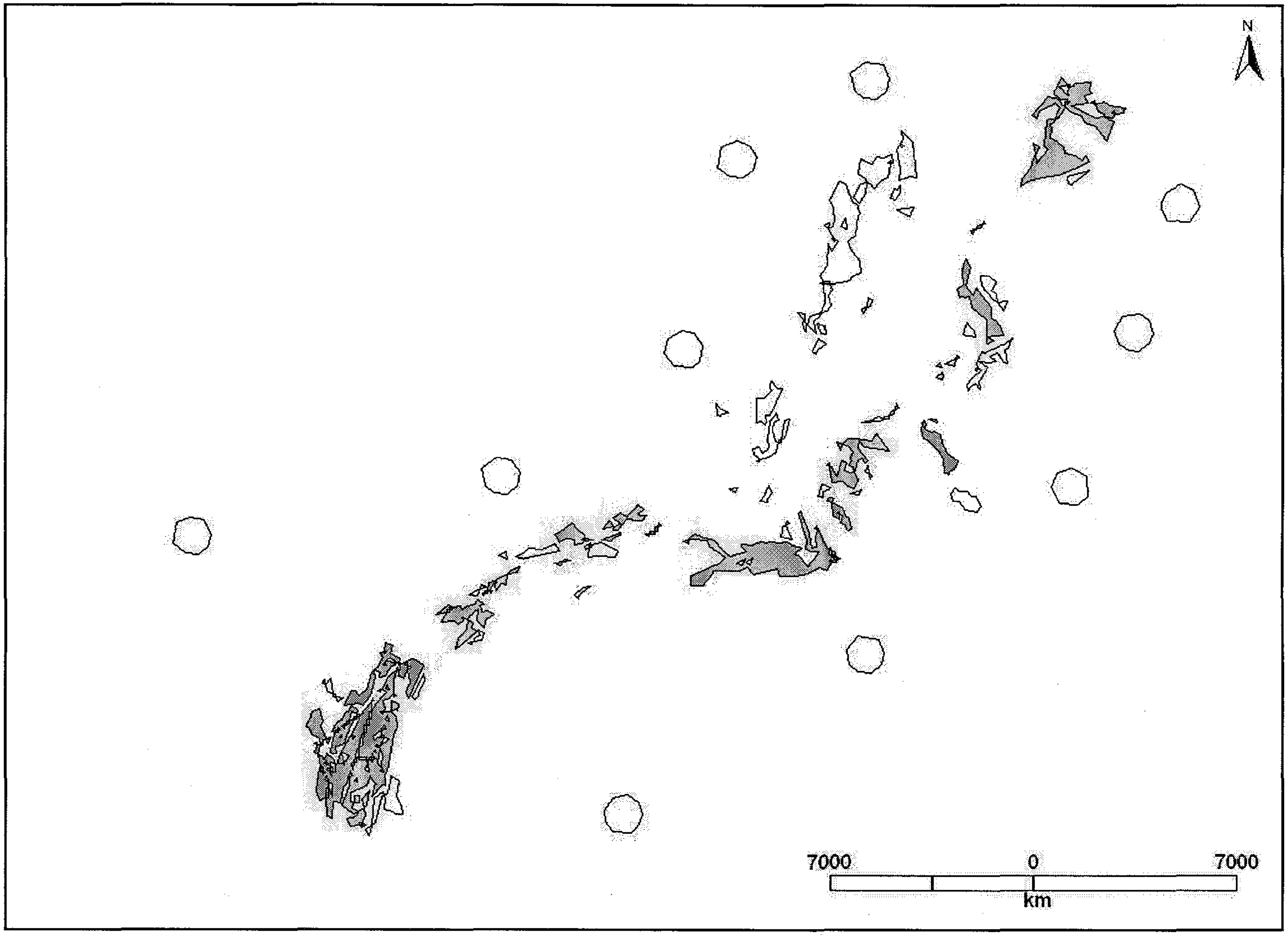
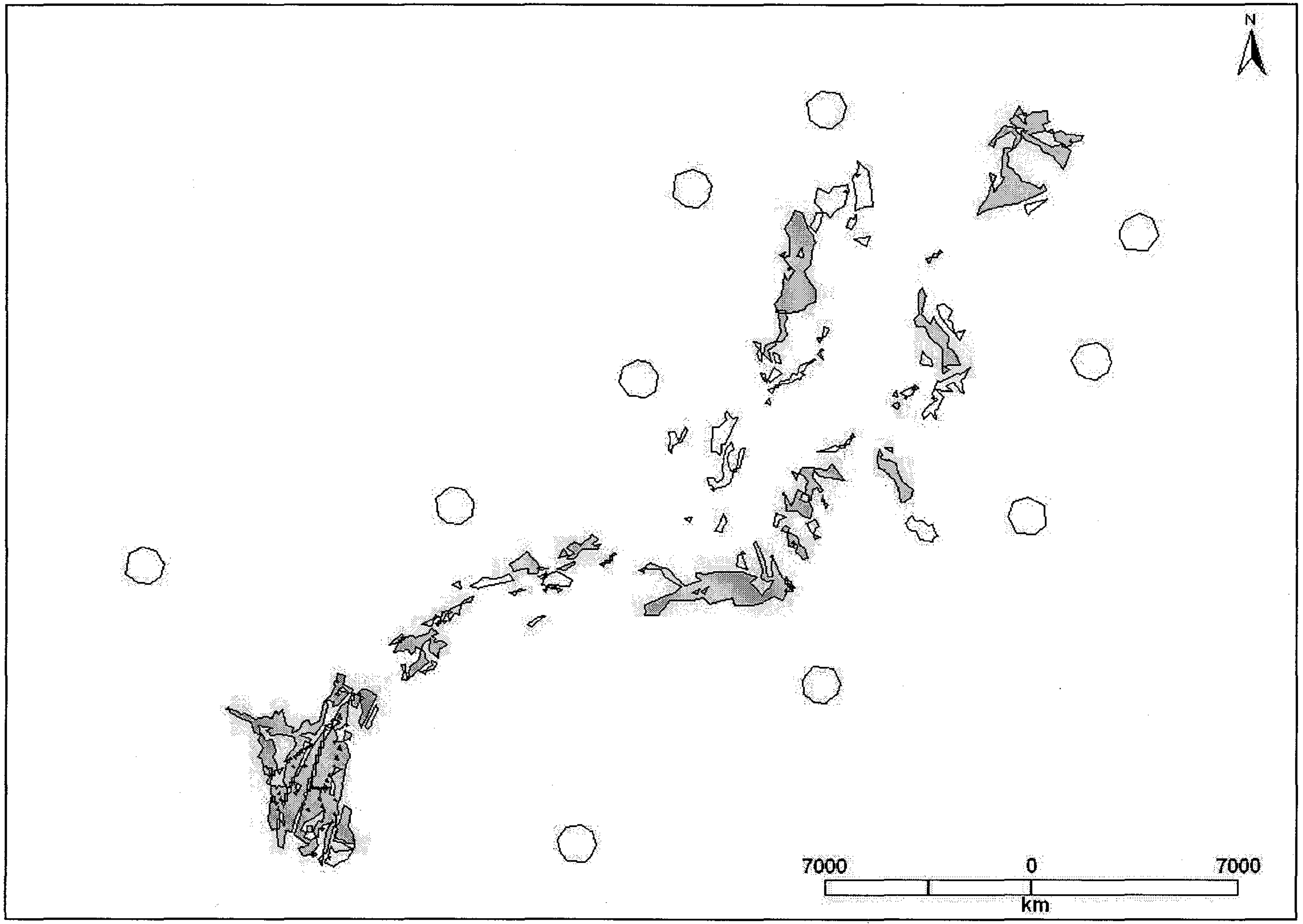


Figure 2-16 Patch retention option for classification analysis 1st & 2nd ecosystem component. Highlighted patches represent patches to be maintained



Cost Surface Uncertainty

Table 2-2 Comparison between landscape statistics of original analysis and cost analyses

	Original	Uniform	Slope	Hwy	Add-ons	Developed
Maximum Cost	182981	103724	106736	106951	114456	131288
Minimum Cost	50	250	50	50	50	50
Patch Area	1236	1236	1236	1236	1236	1236
Number of Patches	56	56	56	56	56	56

The patch landscape in the cost analysis was not varied from the original analysis, therefore the patch area and number of patches remained the same. The only difference that occurred was the cost for each organism to traverse the landscape, due to the variability in the cost surfaces used in each analysis. As illustrated in table 2-2, the minimum cost for an organism to travel between two patches was 50 for every analysis except the analysis, which applied the uniform surface. This suggests that the "cheapest" two patches to move between were most likely in the lower valley were the slope was minimal and there were no barriers to dispersal between them. Therefore when slope was removed from the analysis and replace by a uniform surface with cost values of 5, cost to move between these patches increased. The maximum cost to travel between two patches increased as barriers to dispersal were added to the landscape. It is interesting to note the difference between the maximum cost, for the slope cost surface and the slope/highway cost surface is only approximately 214 while the difference when the highway add-ons are considered is approximately 7720. This illustrates the impact that filling in holes within the highway, has on the analysis in regards to travel cost since there is now no cheap alternative routes to cross this feature.

The resultant area trade-off curves for the cost analysis have pronounced differences from the original analysis. For the uniform cost surface (Figure 2-17), the best and intermediate "c" dispersers experience a linear decline in connectivity as area is removed whereas in the original analysis (Figure 1-7) the area trade-off curves had a "step like" characteristic to the curve as patches were removed. In the uniform cost analysis, intermediate disperser "c" experienced a transition zone at approximately 800ha, where a steep drop in correlation length occurred, representing an approximate loss in connectivity of 19%. This loss in correlation length occurred over a smaller area removed than in the original analysis. A similar situation is evident for intermediate disperser "b", at approximately 700ha where correlation length drops sharply until an approximate area of 500 ha where a lower plateau is reached. Also, the connectivity of the landscape with all patches intact was substantially higher for all the intermediate dispersers for the uniform cost analysis, in comparison to the original analysis (Figure 1-7 & 2-17).

An interesting difference in the results occurs from the original analysis when slope is considered as the only barrier to dispersal (Figure 2-18). Instead of each of the dispersal classes viewing the initial connectivity of the landscape differently, all the intermediate dispersers view connectivity the same until reaching transitions zones around 575ha where intermediate dispersers "b" and "c" experience a distinct transition zone. Transition zones in the slope cost surface appear later, after a greater area is removed than in the original analysis. In the original analysis, the transition zone for intermediate dispersers "a" and "b" occurred at approximately 800ha (Figure 1-7) where in the slope cost surface analysis the transition occurred just above an area of 500ha. The "poor" disperser similarly to the original analysis did not perceive the landscape as connected at any area and therefore no change occurred for that disperser as patches were removed. The best and intermediate "c" dispersers did not display any distinct transition zones however, sharp losses occurred near the end of the analysis at approximately 500ha. This loss was not as distinct especially for the intermediate disperser "c" who experienced a sharp loss in correlation length at approximately 800ha in the original analysis.

The addition of the highway to the slope surface also results in the intermediate dispersers viewing the landscape connectivity similarly, however, over a shorter period than in the slope cost surface analysis (Figure 2-19). With the highway added to the slope surface the area trade-off curves for intermediate dispersers "a" and "b", similarly to the original analysis, experienced transition zones around 800ha. However, the loss in connectivity at this zone is much greater than in the original analysis, as the dispersers already viewed the landscape as having a greater connectivity when the landscape was completely intact than in the original analysis. The best and intermediate "c" disperser, however, do not experience a distinct transition zone with the highway added to the slope surface, experiencing instead a gradual drop in connectivity as area is removed. Suggesting that the loss of correlation length has a stronger relation to the area being removed from the landscape than the importance of the patch to these two dispersal abilities. This differs from the original analysis where intermediate disperser "c" experienced a sharp loss in connectivity at approximately 770 ha, suggesting that there are patches that are important to maintaining connectivity for this dispersal ability, when more costs to dispersal are considered. The "poor" disperser as in previous analyses had little change to the area trade-off curve when the highway was added to the cost surface.

This disperser still views the landscape as disconnected with little change to the level of connectivity as patches are removed.

The final two analyses, which respectively filled the "holes" in the highway (Figure 2-20) and added developed areas to the cost surface (Figure 2-21), had very similar area trade-off curves. Both of these analyses illustrated very little difference in the perception of the connectivity of the landscape by intermediate dispersers until they reached a transition zones at approximately 700ha. This zone was identical for the intermediate "a" and "b" dispersers in the two analyses, while the transition zone for the intermediate "c" disperser appeared later around 500ha for both analyses. In comparison to the original analysis these zones occurred at a landscape of smaller area than in the original analysis, where the transition zone for these dispersal abilities occurred closer to 800ha for all of the intermediate dispersers. The best disperser also experienced a sharper drop in connectivity than in the original analysis, which may be considered as a transition zone at approximately 400ha, as this zone is more distinct with a greater loss in connectivity than in the original analysis. The poor disperser as in other analyses illustrated little change in the area trade-off curves with the changes to the cost surface.

The area of patches removed at each iteration showed similar trends to the original analysis with smaller patches more likely to be removed early in the analysis while larger patches were often removed late in the analysis (Figures 2-22, 2-23,2-24,2-25,2-26). Smaller patches however, were still being removed late in the analysis, after larger patches, illustrating the importance of stepping stone in all the cost surface variations. The order of removal of the patches was different for each analysis as illustrated by figures 2-27-2-30. In the uniform cost analysis, very little importance was attributed to the northeast corner of the valley, which had no patches that were removed in the last quarter of the analysis (Figure 2-27). Patches from the last grouping removed were more concentrated in the southern half of the map, especially between the two middle target patches. However, when slope is considered as a barrier to dispersal (Figure 2-28) the distribution of the darkest classification (i.e. patches removed in the last quarter of the analysis) shifts to include the northeast corner, shifting from the central patches which were considered most important in the uniform cost analysis. As the highway is added to the cost surface, further importance is placed on the northeast corner as patches from the southwest portion of the map shift to this area (Figure 2-29). When the "holes" in the highway surface are fixed the northeast corner of the map again becomes less important as the patches removed in the last quarter of the analysis again shift to the southern half of the map (Figure 2-30). The final variation added the developed areas and this resulted in the patches in the middle of the map being removed from the analysis sooner. However, the southern half of the map again contains the greatest concentration of patches that were removed in the last group during the analysis (Figure 2-31). This differs from the original map, which tends to have a more even distribution of darker patches throughout the valley (Figure 1-9). However, the differences that occur in the order of removal maps do not appear to be substantial. Many of the core patches that were in the last grouping of patches remained the same in each of the cost analyses, with only a few patches being switched out of this group. It is difficult to determine how these changes in patch removal order will ecologically impact the landscape. Even though it appears that differences in removal order, results in very little visual change to the final patch removal order, the exclusion of a patch from the landscape may mean the difference between a connected and disconnected landscape for some organisms.

For the cost analysis, the suggested patch area to be retained for the reserve for the cost analyses are as follows: uniform cost 799.75ha (Figure 2-32), slope cost 554ha (Figure 2-33), slope and highway cost 860ha (Figure 2-34), slope and highway with add-ons 714.25ha (Figure 2-35), and slope, highway with add-ons, and developed areas 687.5ha (Figure 2-36). In the original analysis (Figure 1-11 & 1-12), 795.5ha was suggested to be maintained in the reserve. This only differs substantially from the slope cost analysis, which would include 241.5ha less area, and the developed area cost analysis that would include 108ha less area. The patch retention options were all developed from the transitions zones that were apparent for dispersers "a" and "b", as in the original analysis. However, the uniform cost analysis was an exception to this with the reserve being developed from just the transition zone attributed to intermediate disperser "a". The appearance of the patch retention option differs from the original analysis, especially in the analysis, which considered slope as the only barrier to dispersal. In this reserve only a few patches are maintained and these patches are all concentrated in the southwest portion of the map (Figure 2-33). Many of the suggested patch maintenance options had little emphasis on the northeast portion of the valley (Figure 2-32, 2-33, 2-35, 2-36), except for the cost surface considering only slope and highway, which include more patches in the northeast section of the valley (Figure 2-34). This option was most similar to the original analysis, where suggested patches for inclusion in the reserve were spread throughout the valley (Figure 1-11&1-12). Despite differences in the options presented there were many core patches, which were included in all of the alternative retention options.

Figure 2-17 Uniform cost area trade-off curve illustrating relation between correlation length and area as patches are removed from the landscape

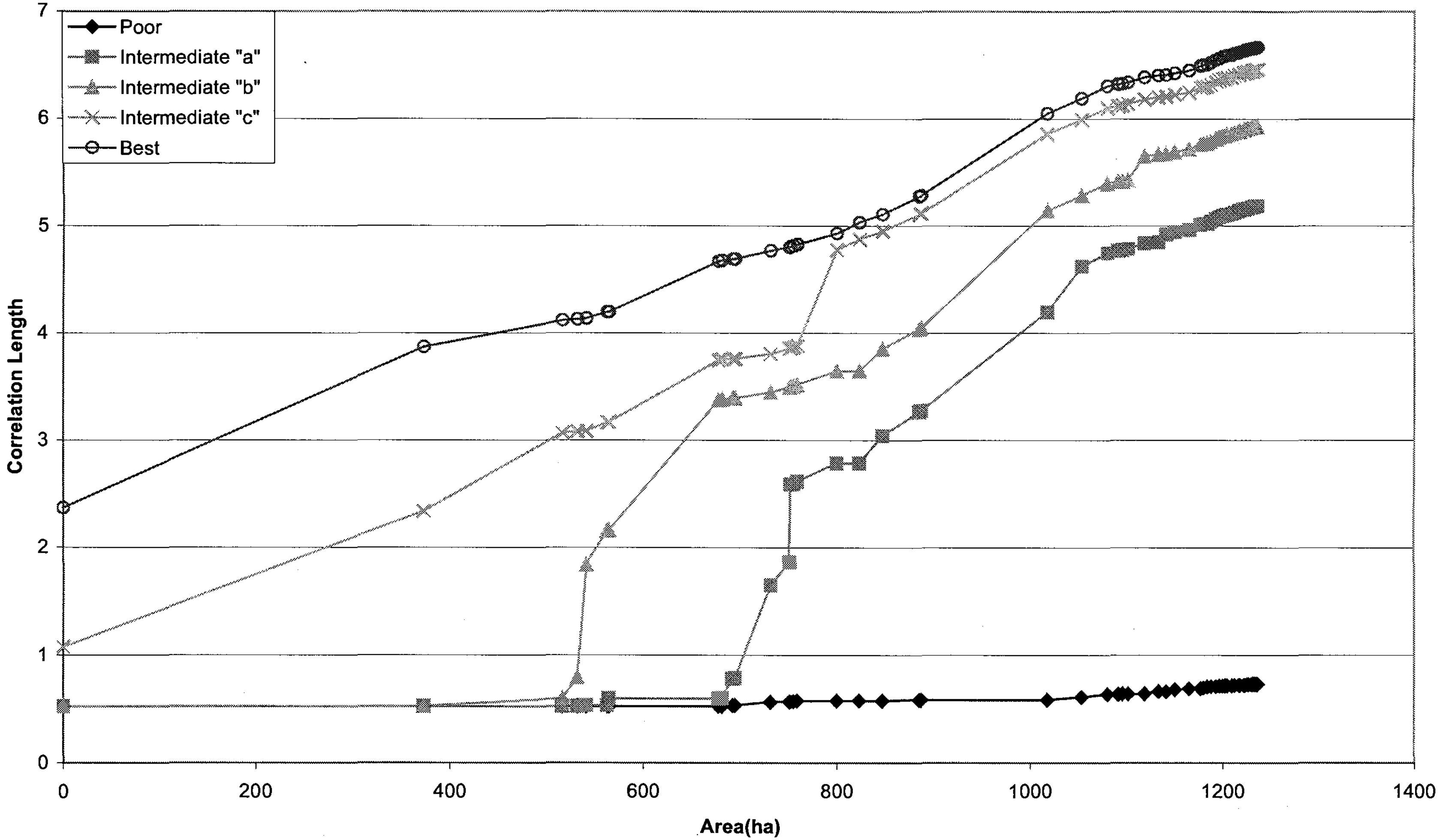


Figure 2-18 Slope cost surface area trade-off curve illustrating relation between correlation length and area as patches are removed from the landscape

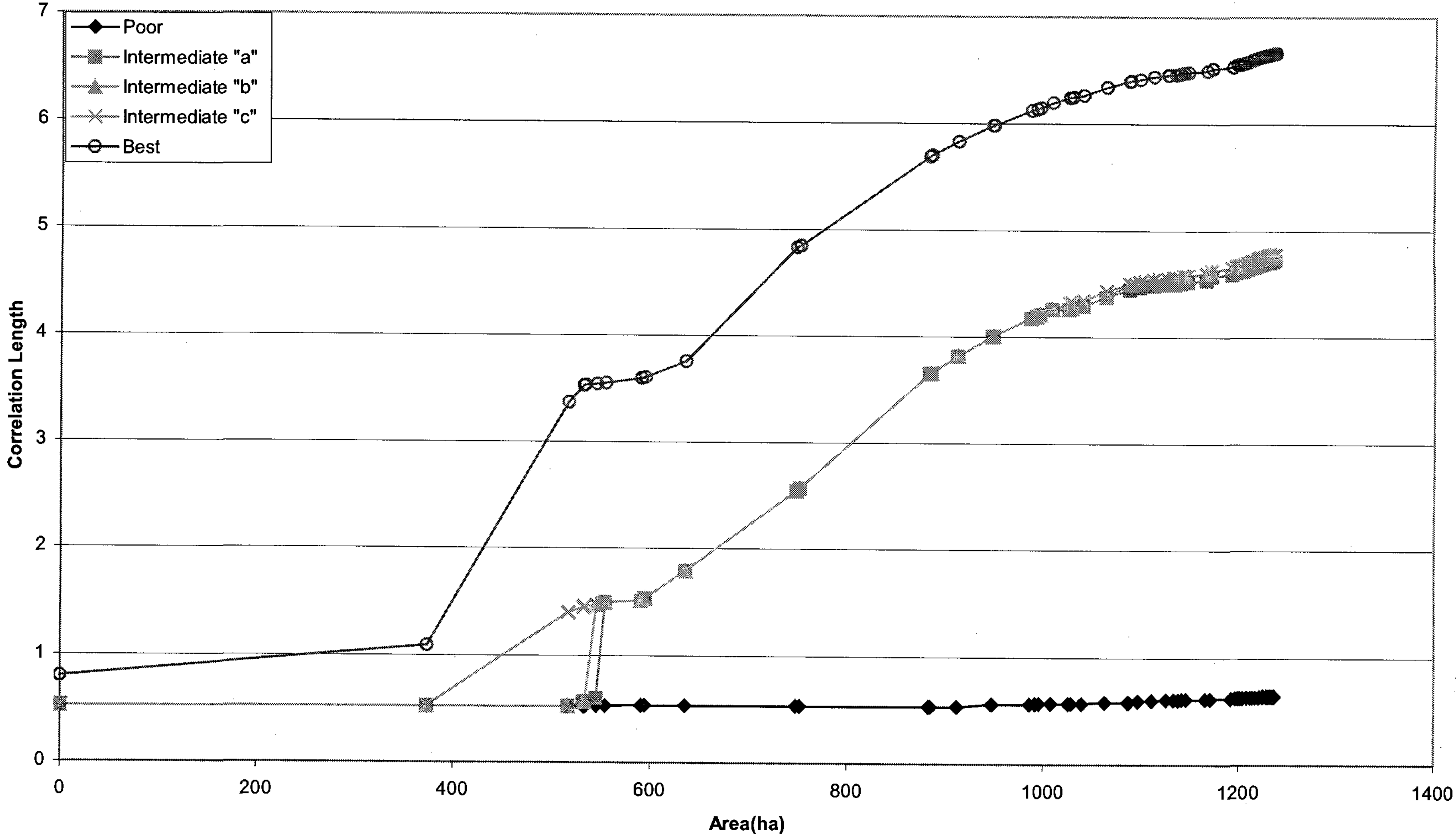


Figure 2-19 Hwy cost surface area trade-off curve illustrating relation between correlation length and area as patches are removed from the landscape

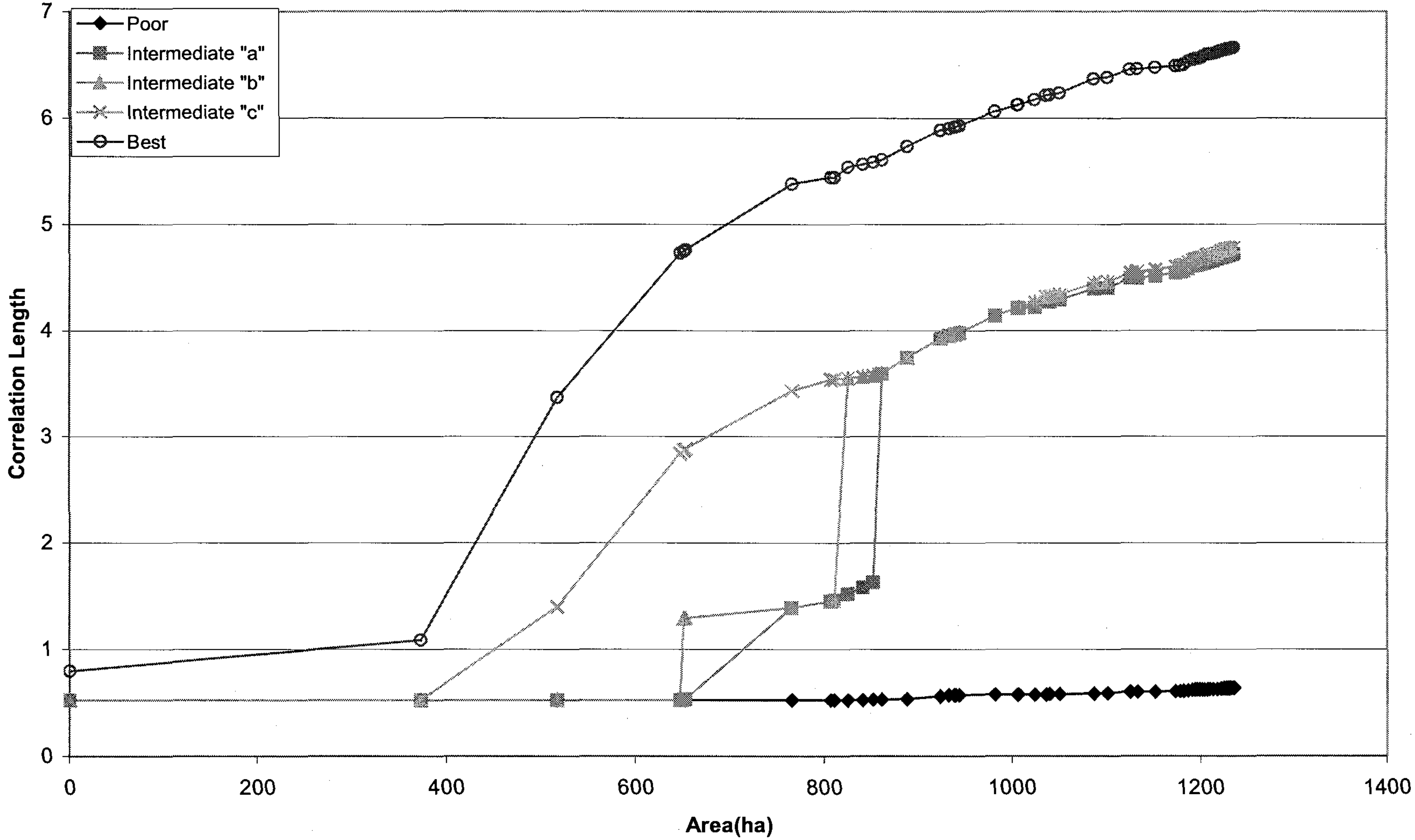


Figure 2-20 Highway add-ons cost surface area trade-off curve illustrating relation between correlation length and area as patches are removed from the landscape

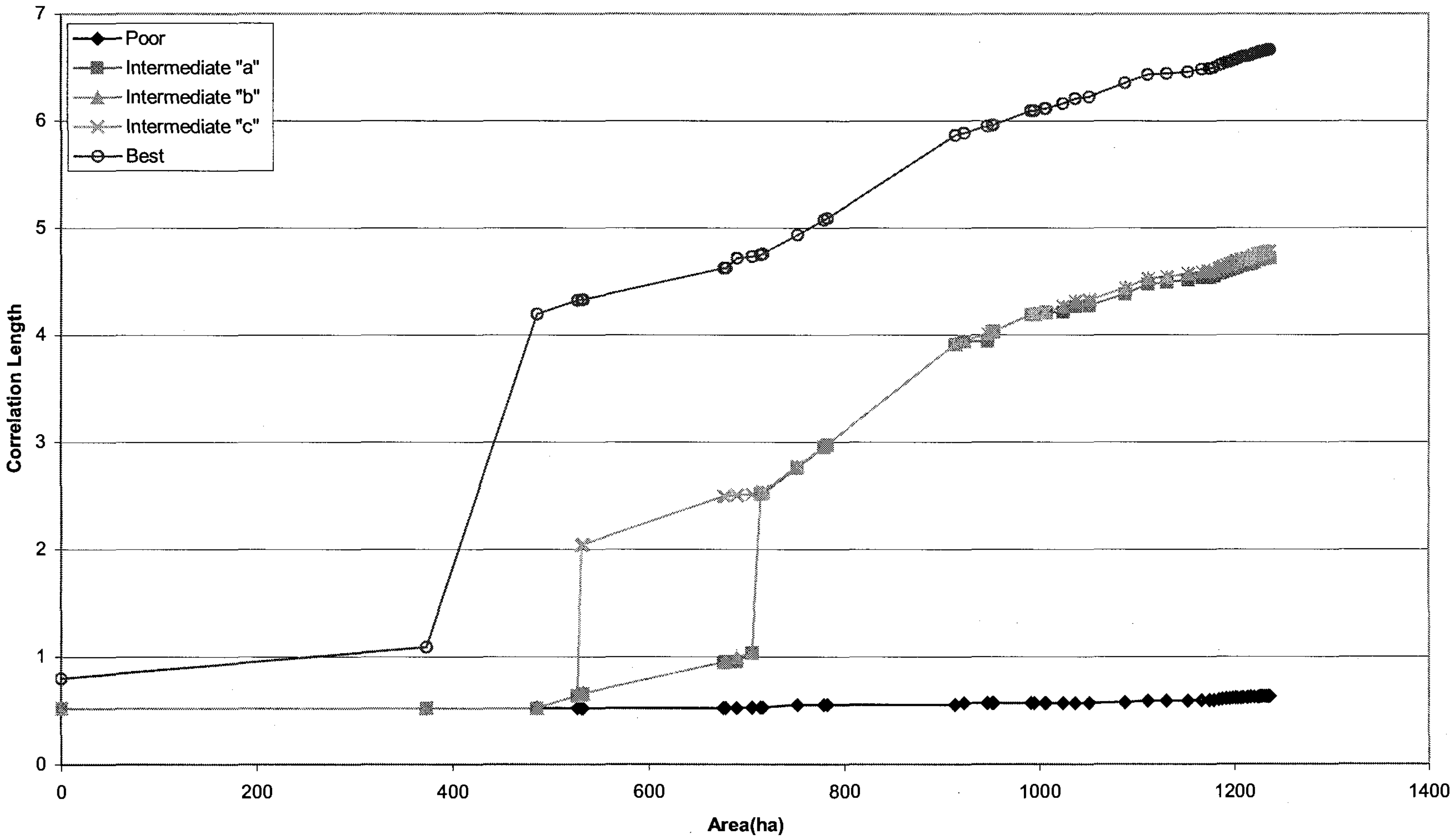


Figure 2-21 Developed areas cost surface area trade-off curve illustrating relation between correlation length and area as patches are removed from the landscape

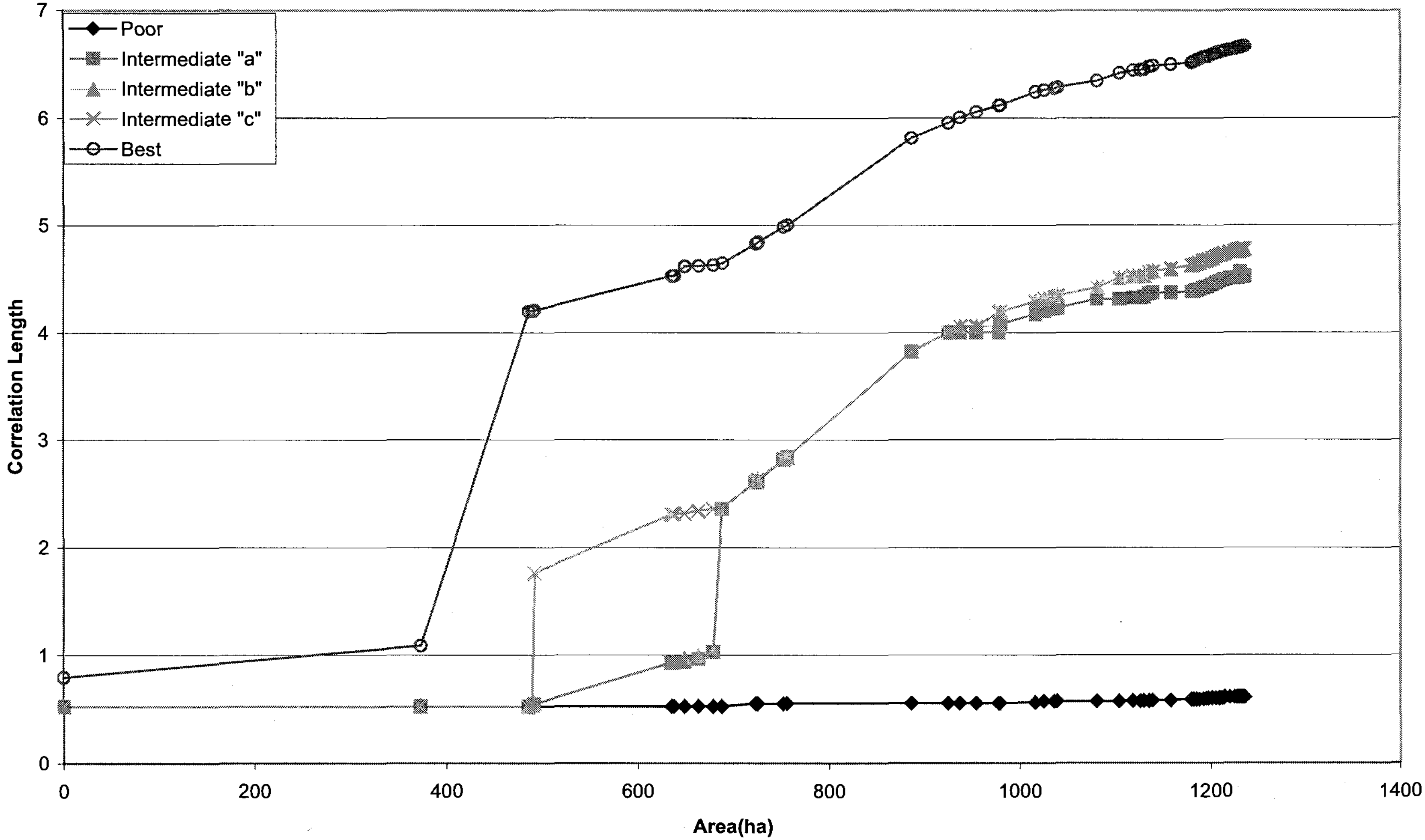


Figure 2-22 Uniform cost surface area of patch removed at each iteration of the simulation

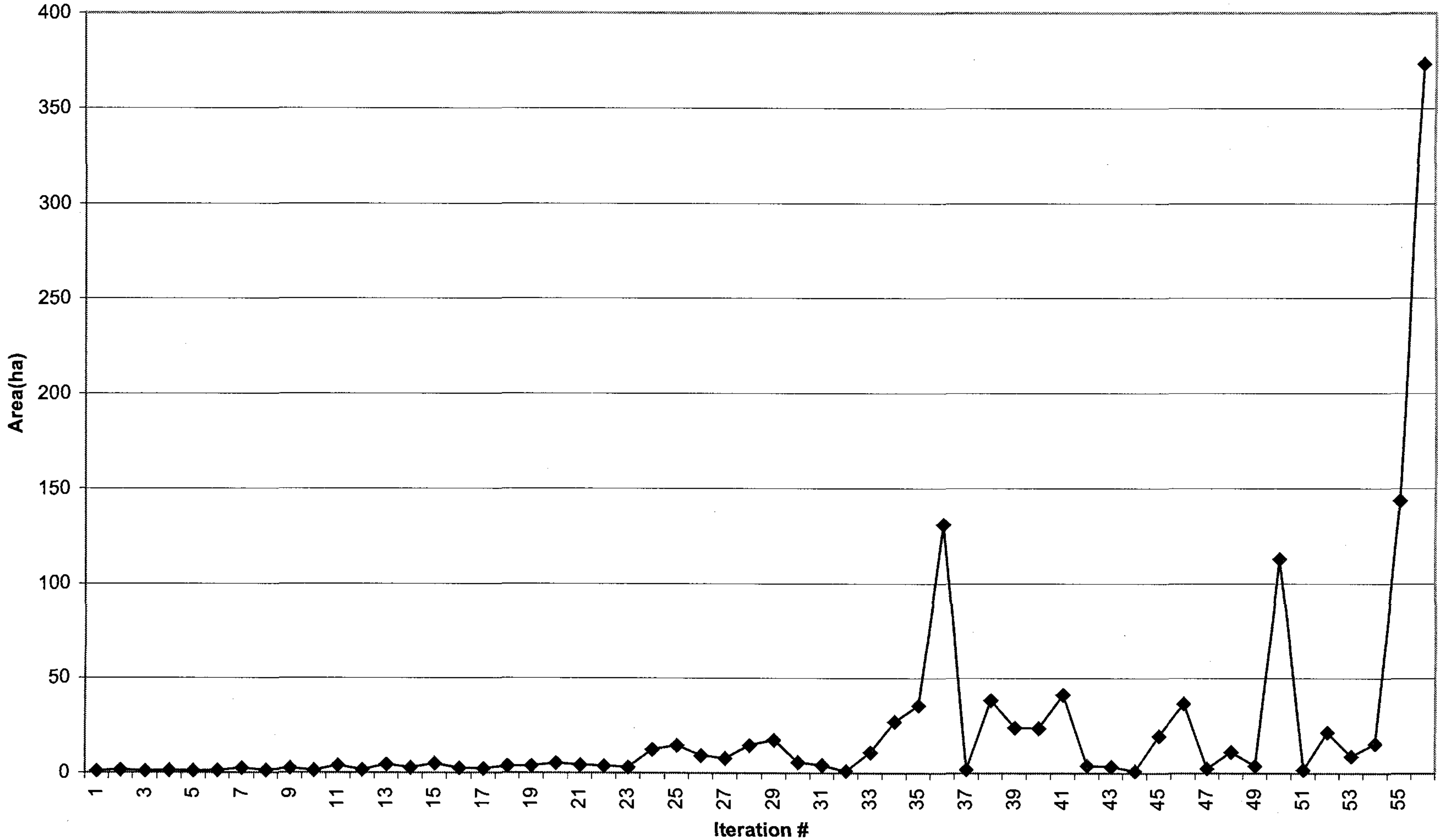


Figure 2-23 Slope cost surface area of patch removed at each iteration of the simulation

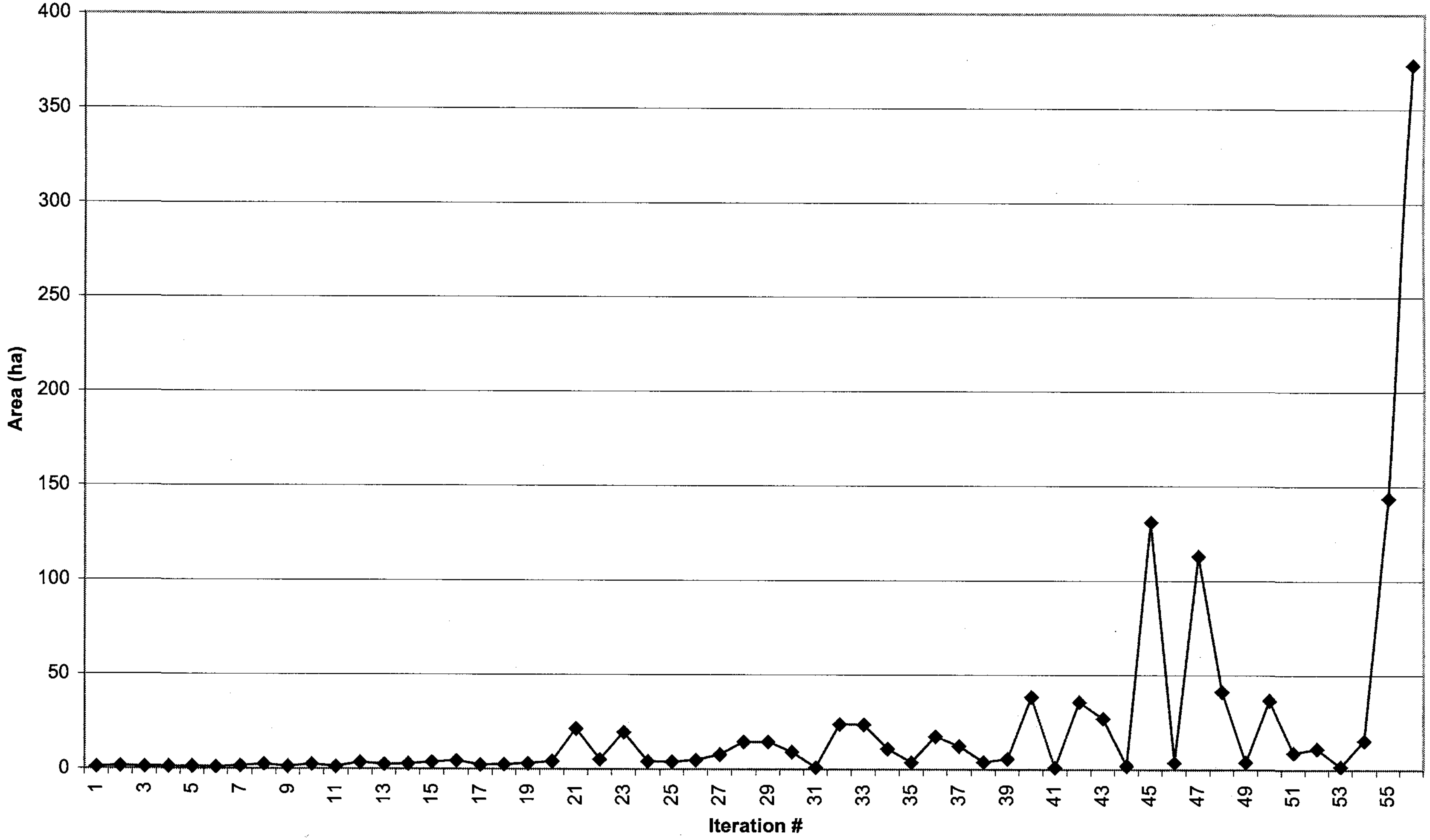


Figure 2-24 Highway cost surface area of patch removed at each iteration of the simulation

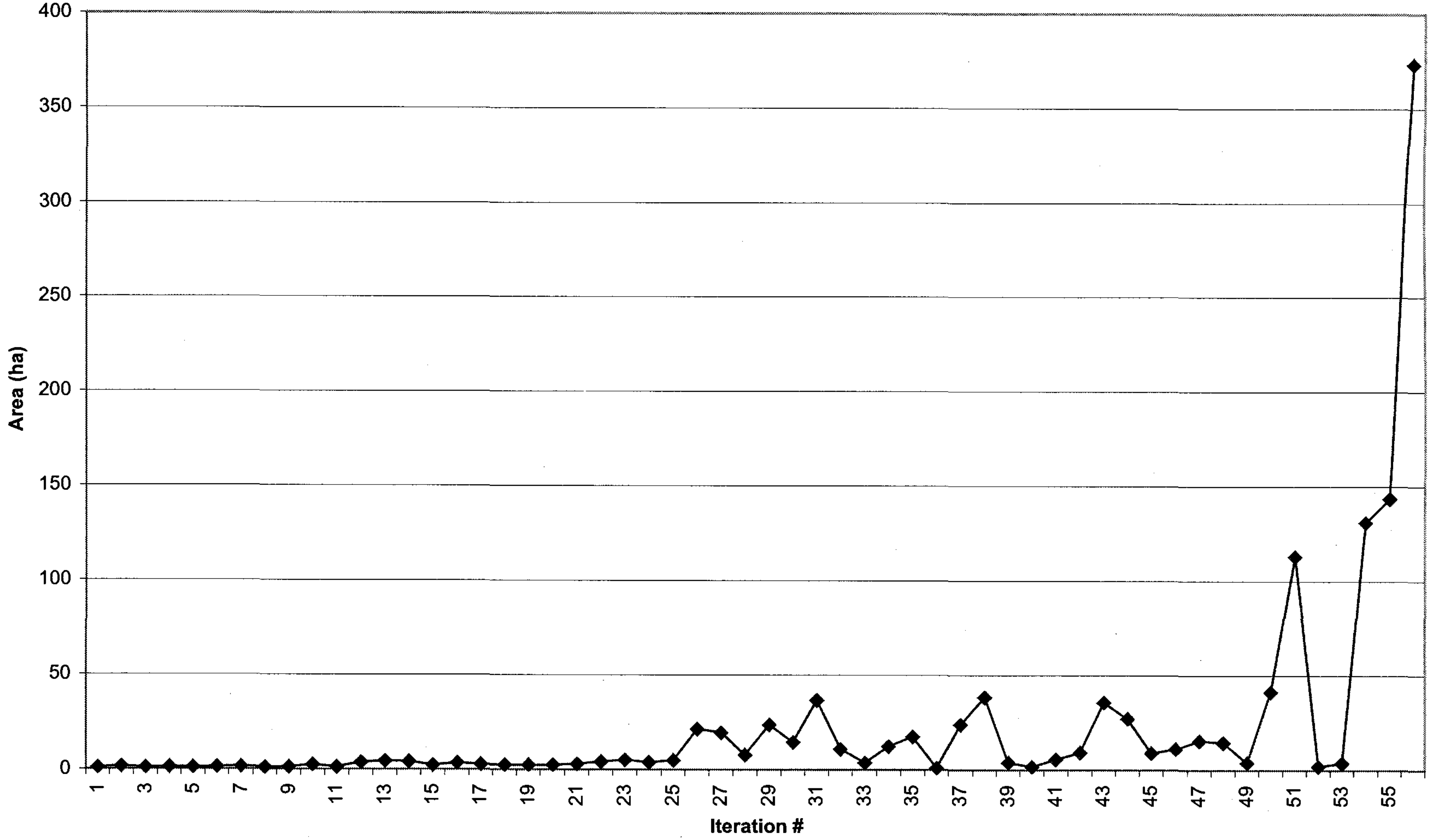


Figure 2-25 Highway add-on cost surface area of patch removed at each iteration of the simulation

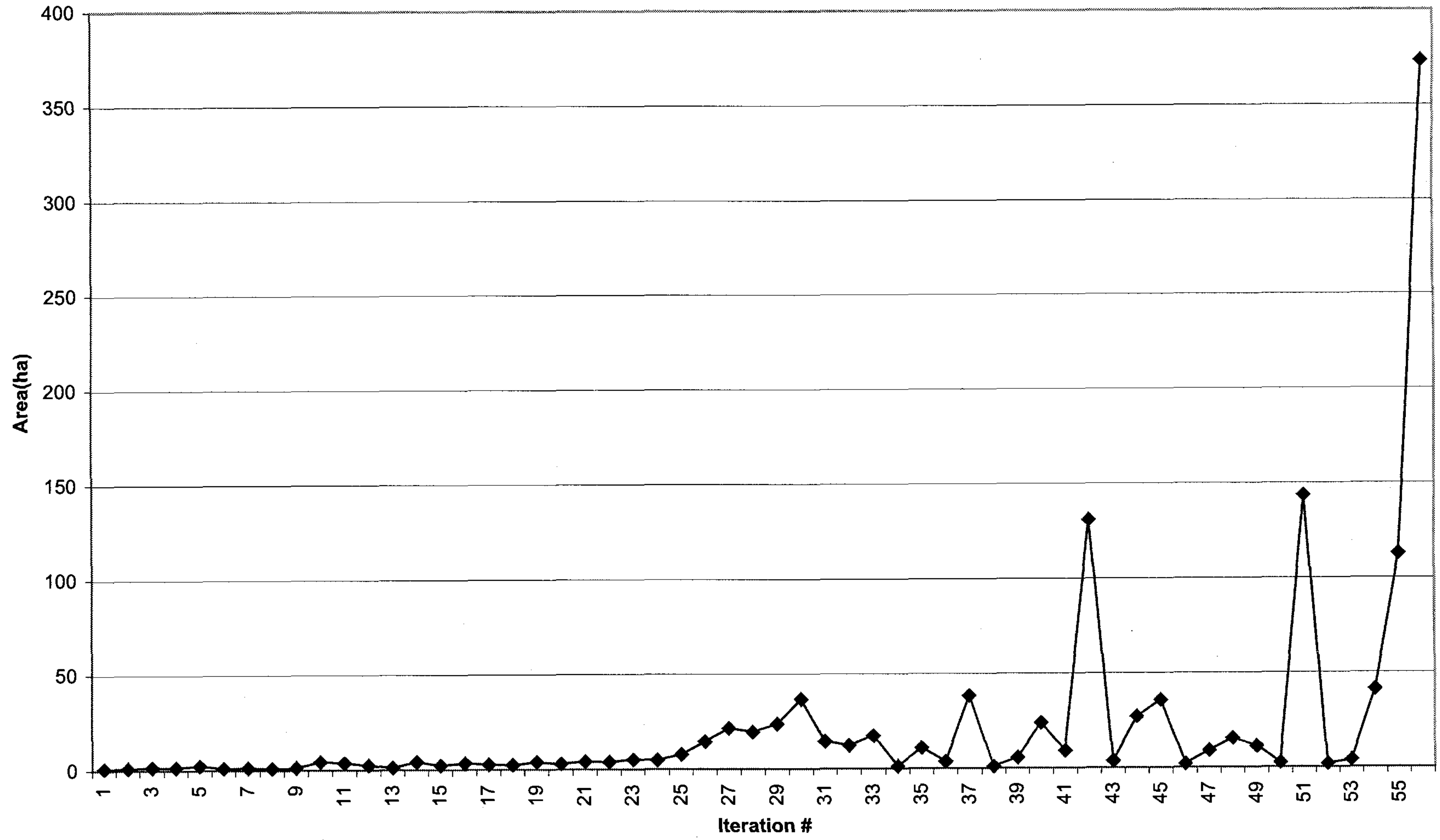


Figure 2-26 Developed area cost surface area of patch removed at each iteration of the simulation

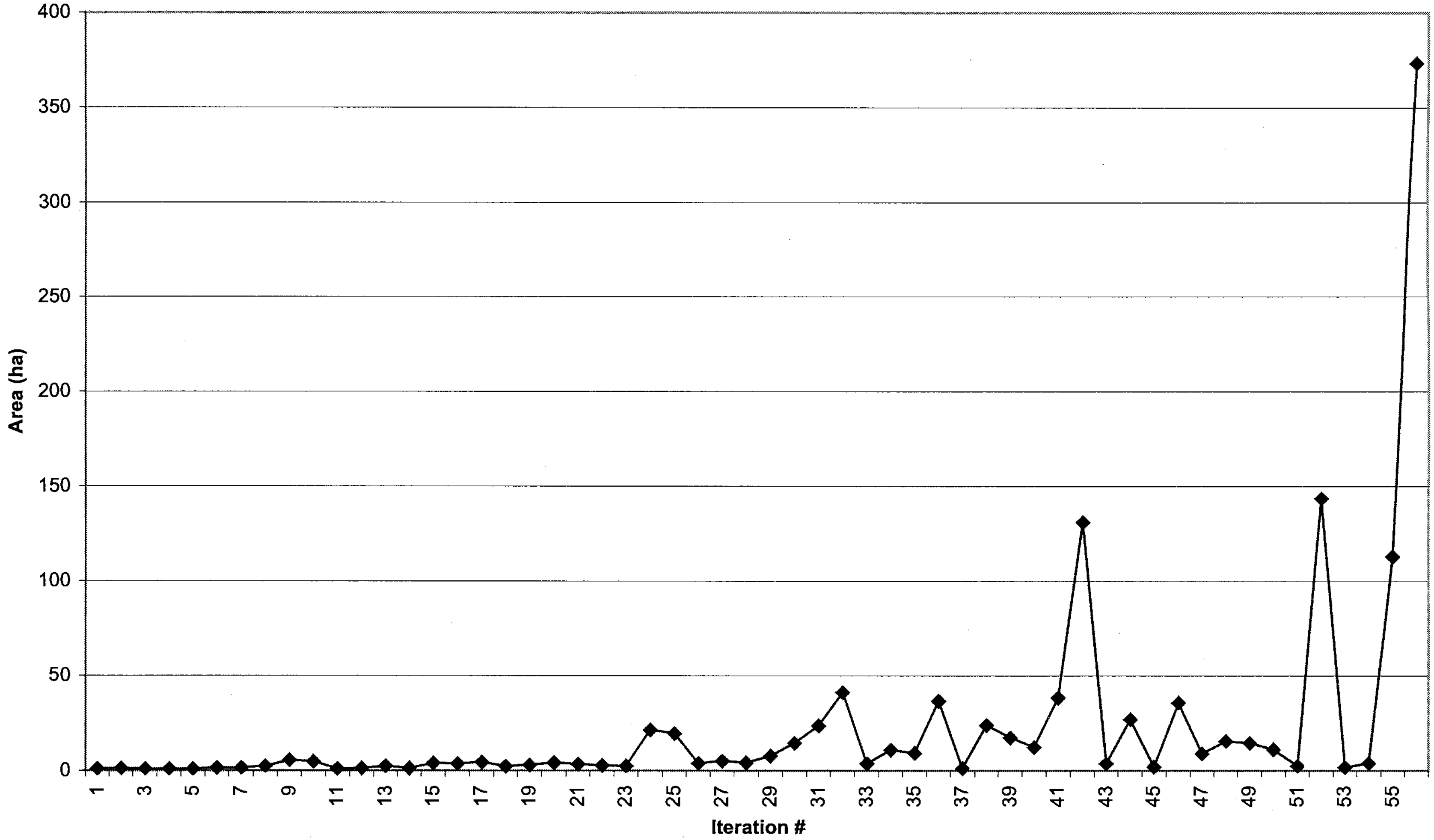


Figure 2-27 Uniform cost surface order of removal classified by four equal intervals

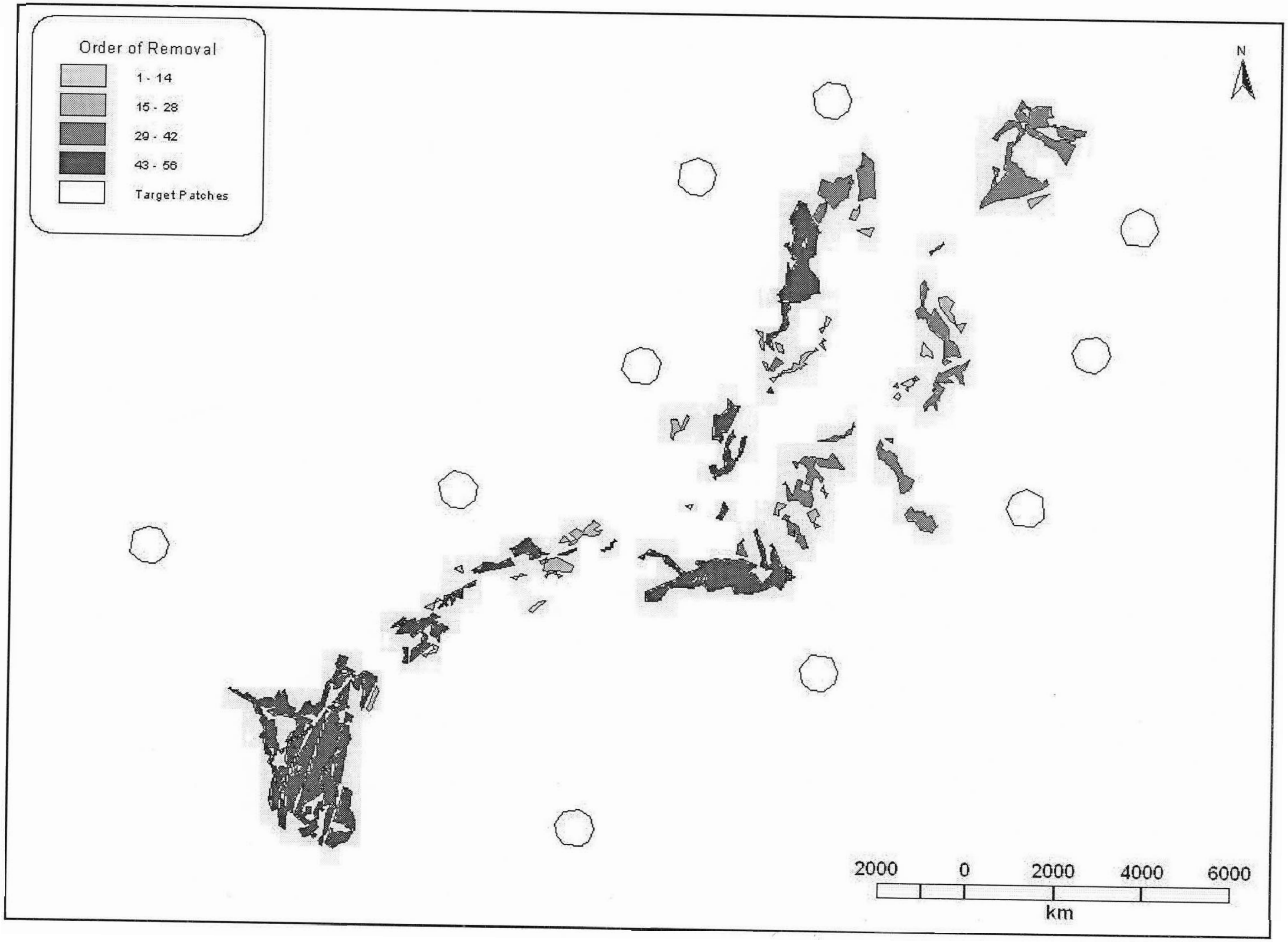


Figure 2-28 Slope cost surface order of patch removal classified by four equal intervals

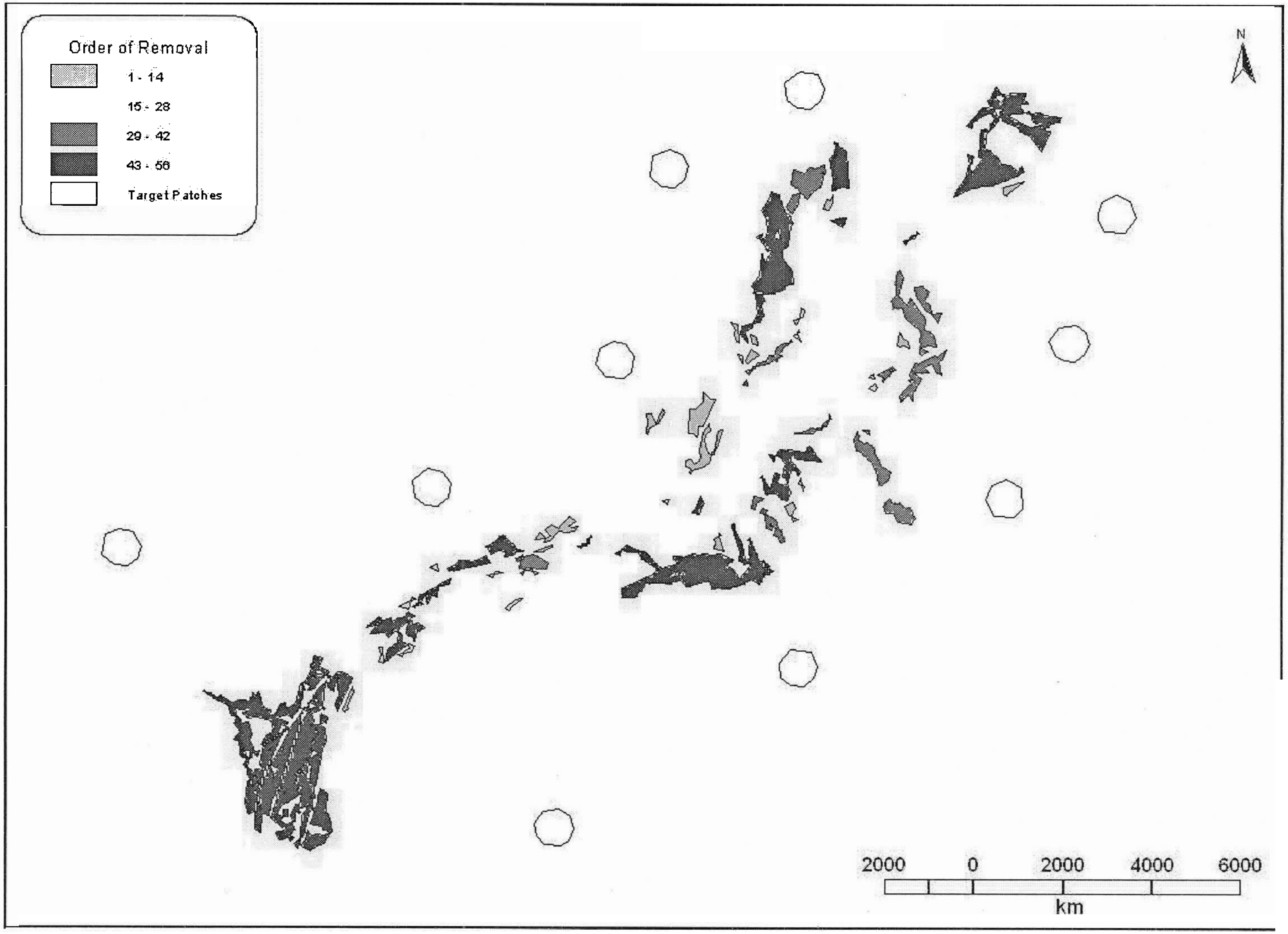


Figure 2-29 Highway cost surface order of removal classified by four equal intervals

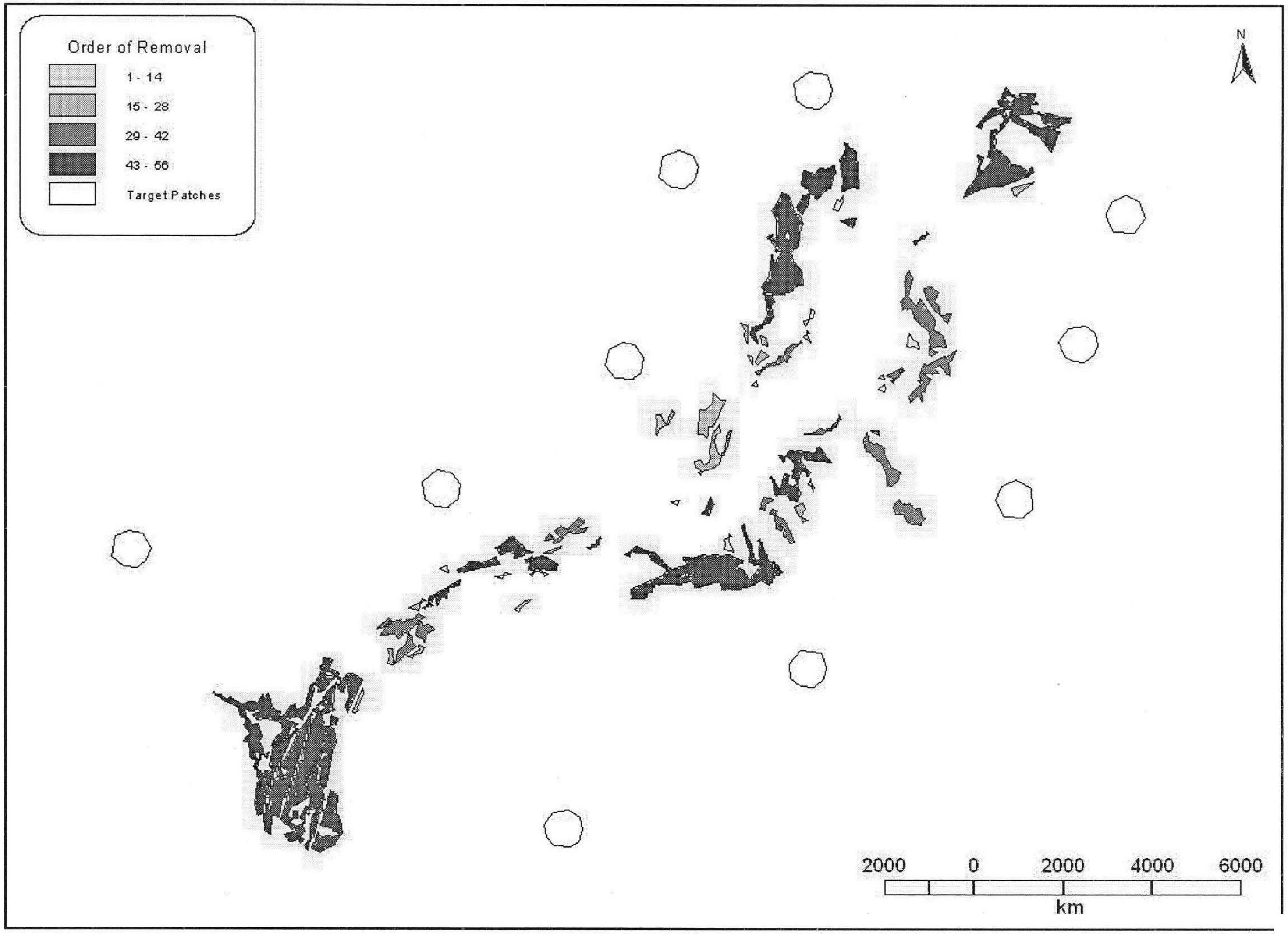


Figure 2-30 Highway add-on cost surface order of removal classified by 4 equal intervals

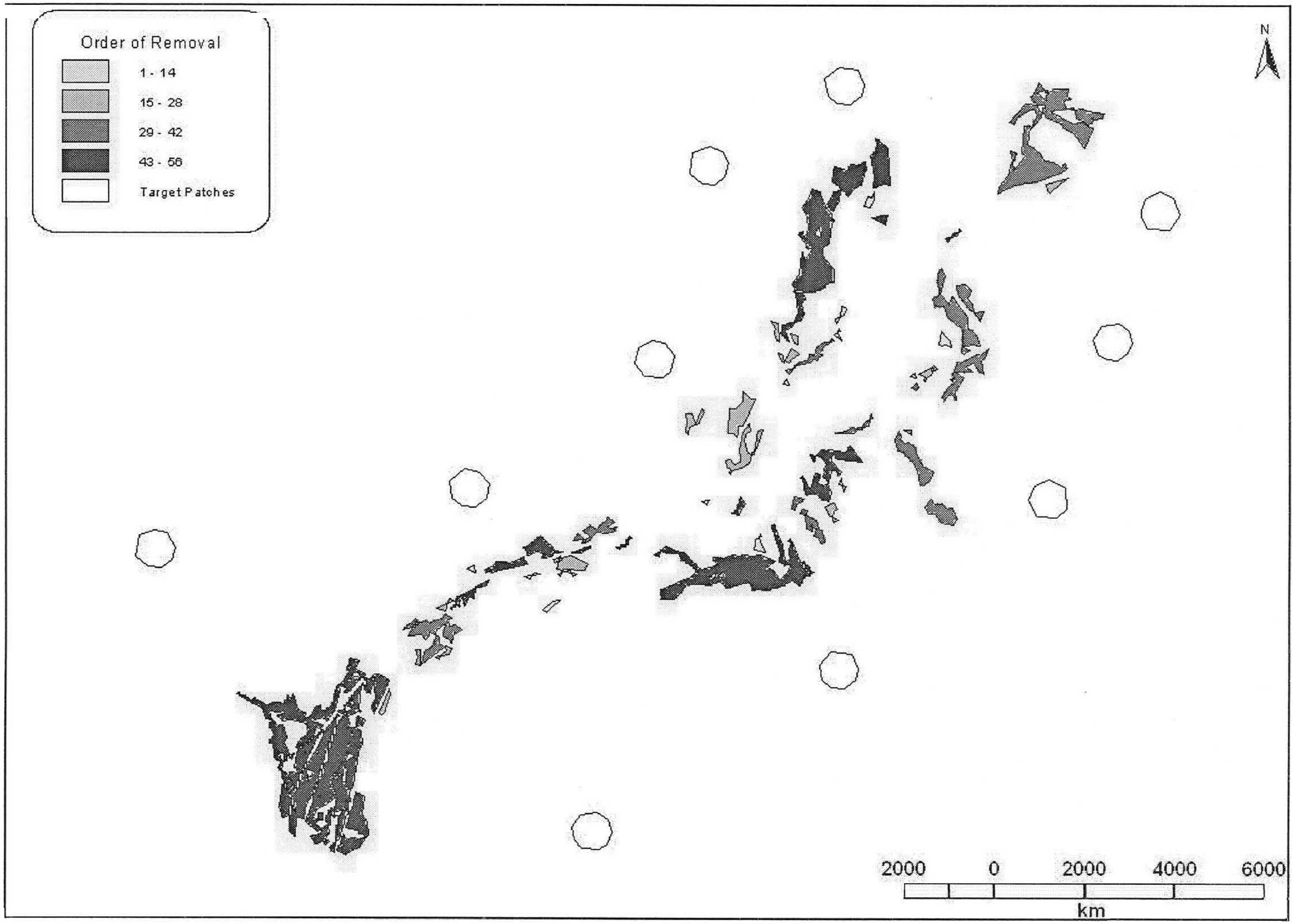


Figure 2-31 Developed area accost surface order of patch removal classified by 4 equal intervals

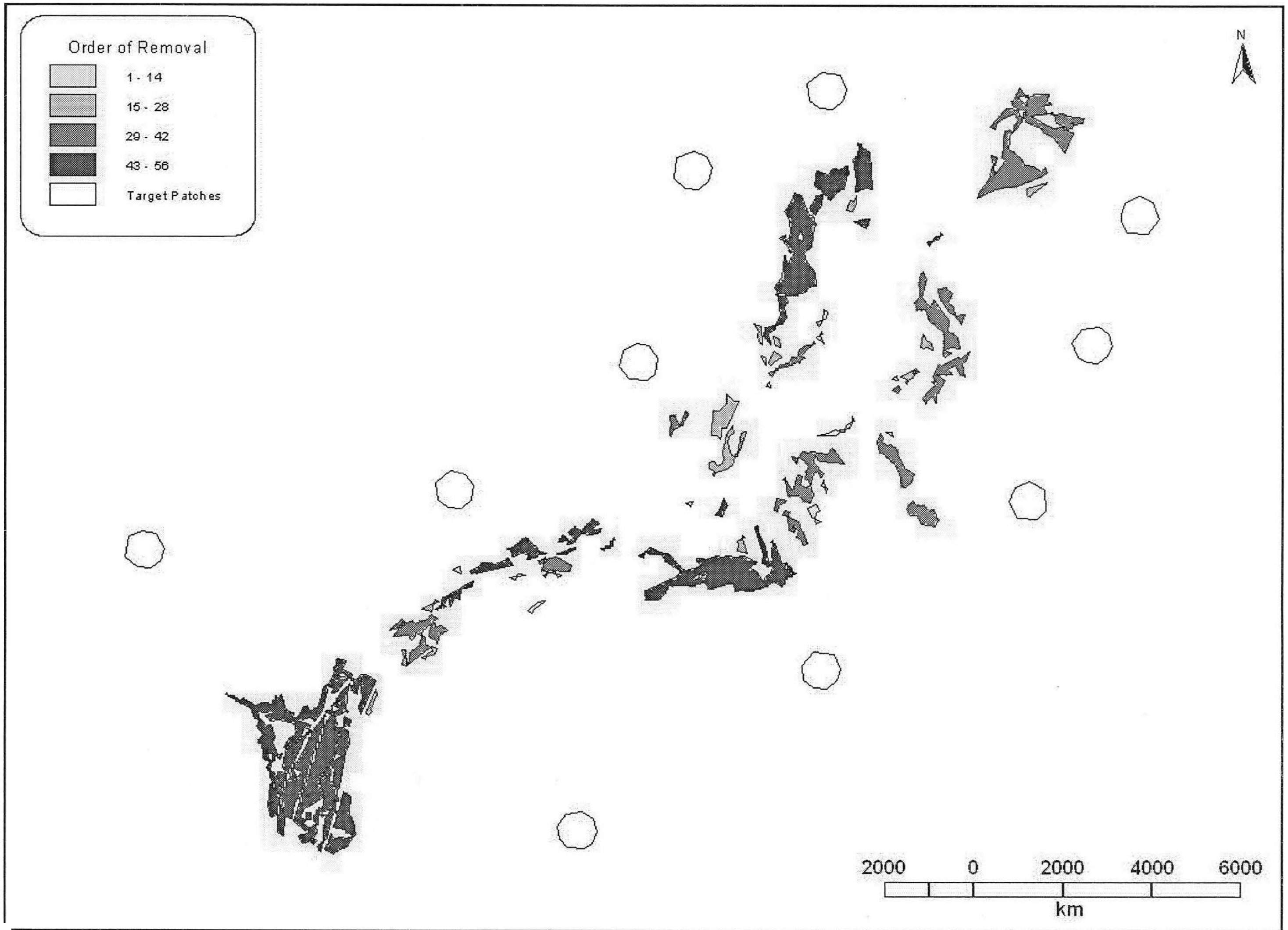


Figure 2-32 Uniform cost surface patch retention option. Highlighted patches represent patches to be maintained

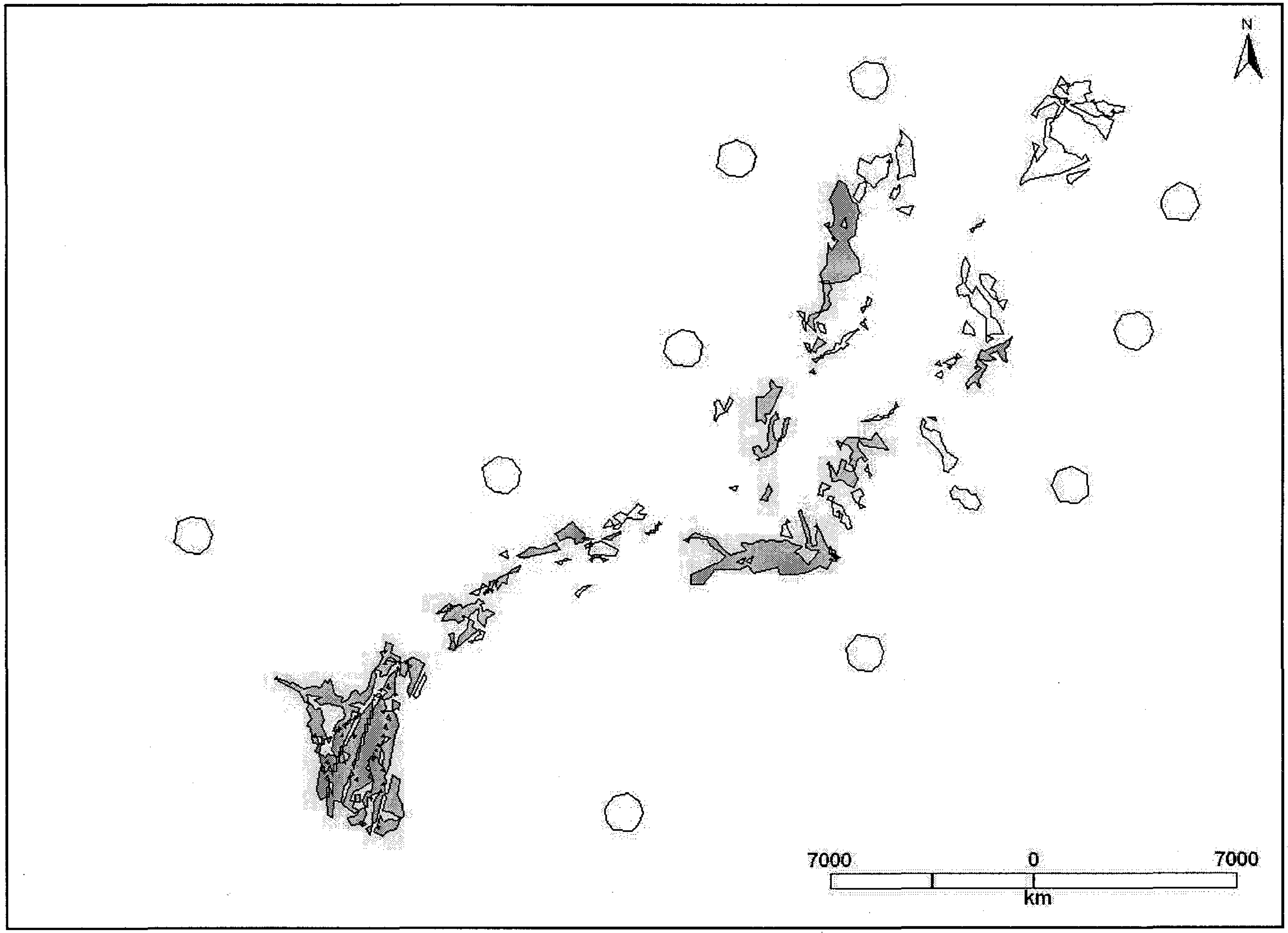


Figure 2-33 Slope cost surface patch retention option. Highlighted patches represent patches to be maintained

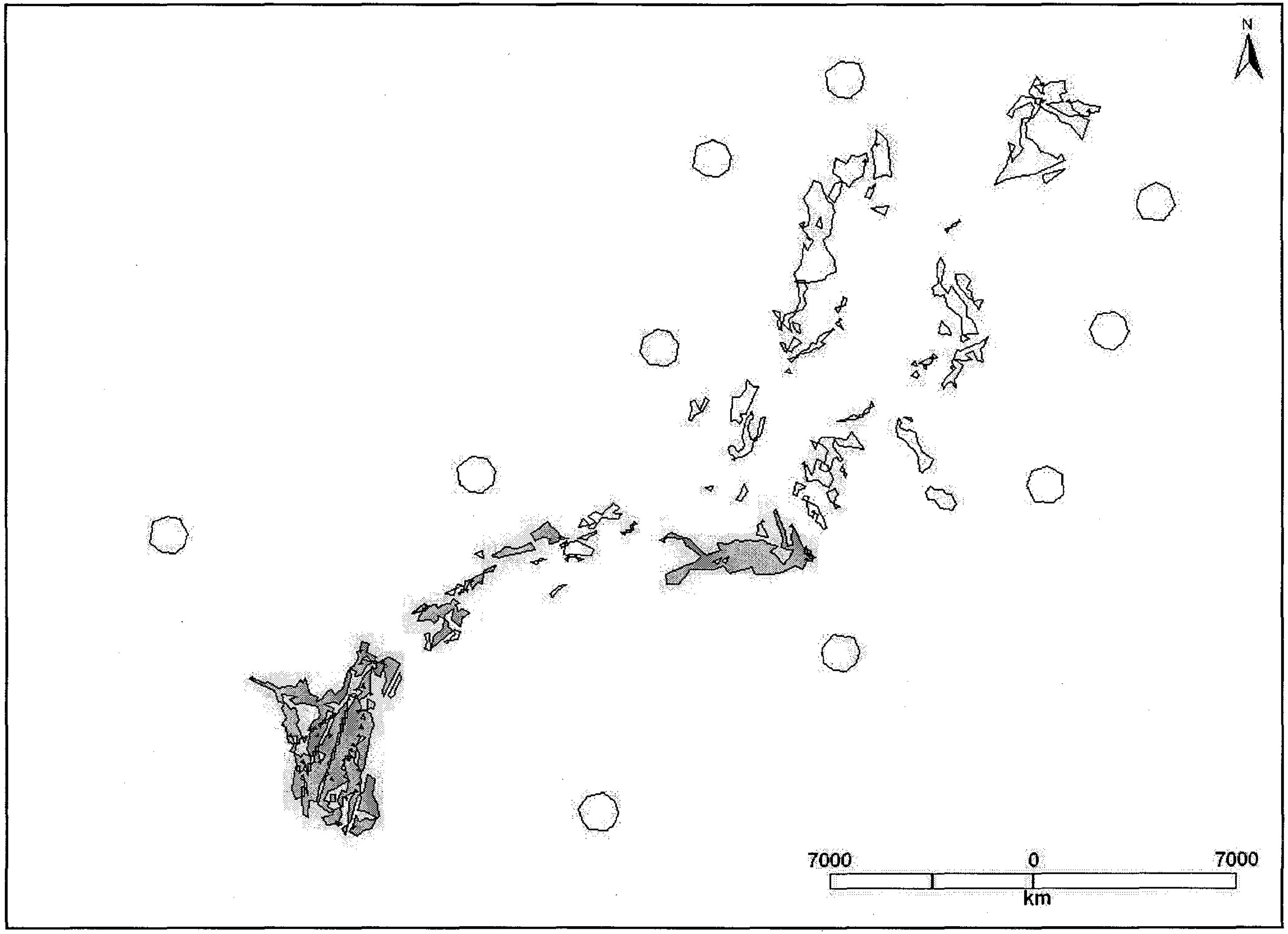


Figure 2-34 Highway cost surface patch retention option. Highlighted patches indicate patches to be maintained

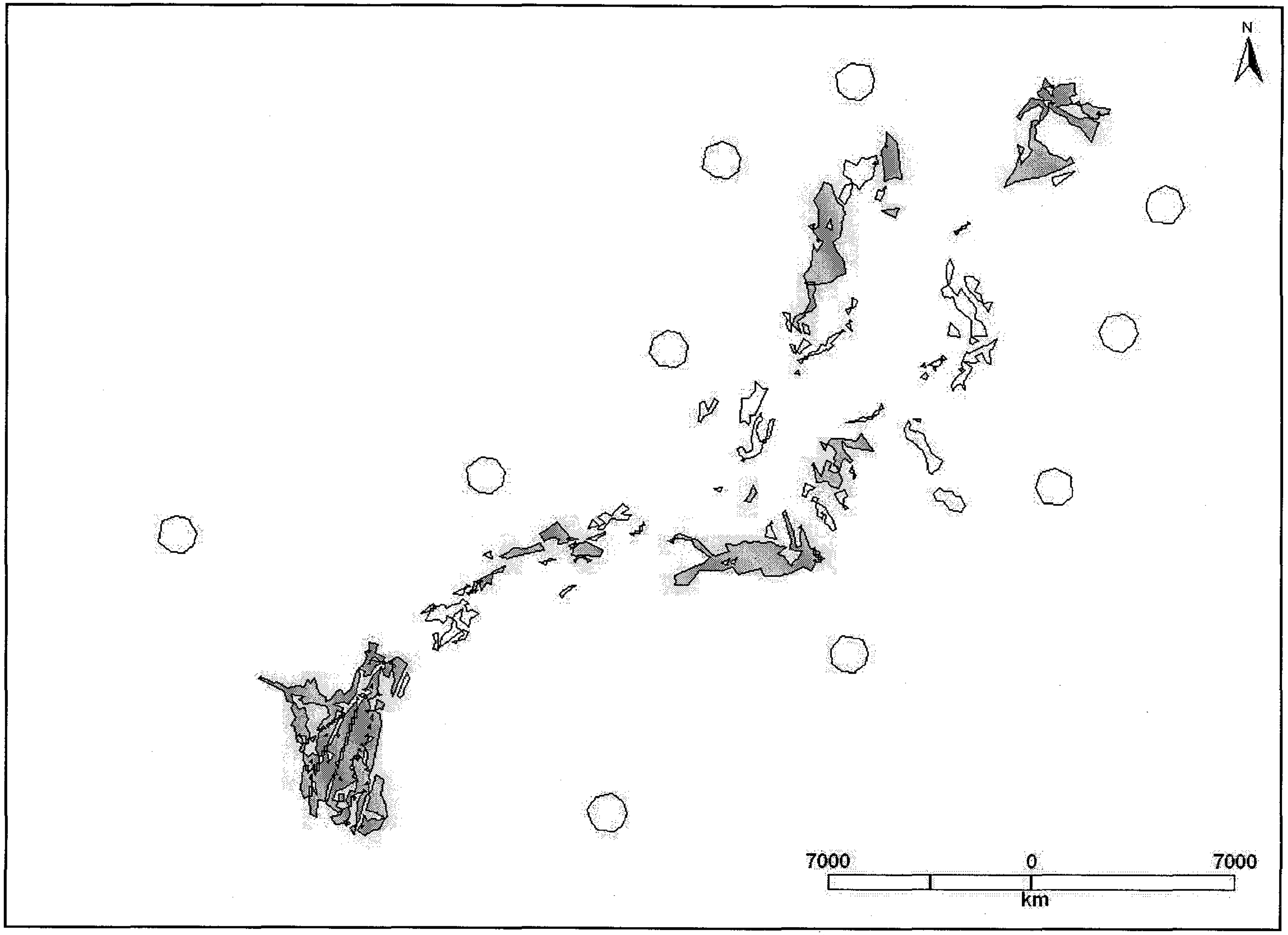


Figure 2-35 Highway add-on cost surface patch retention option. Highlighted patches represent patches to be maintained

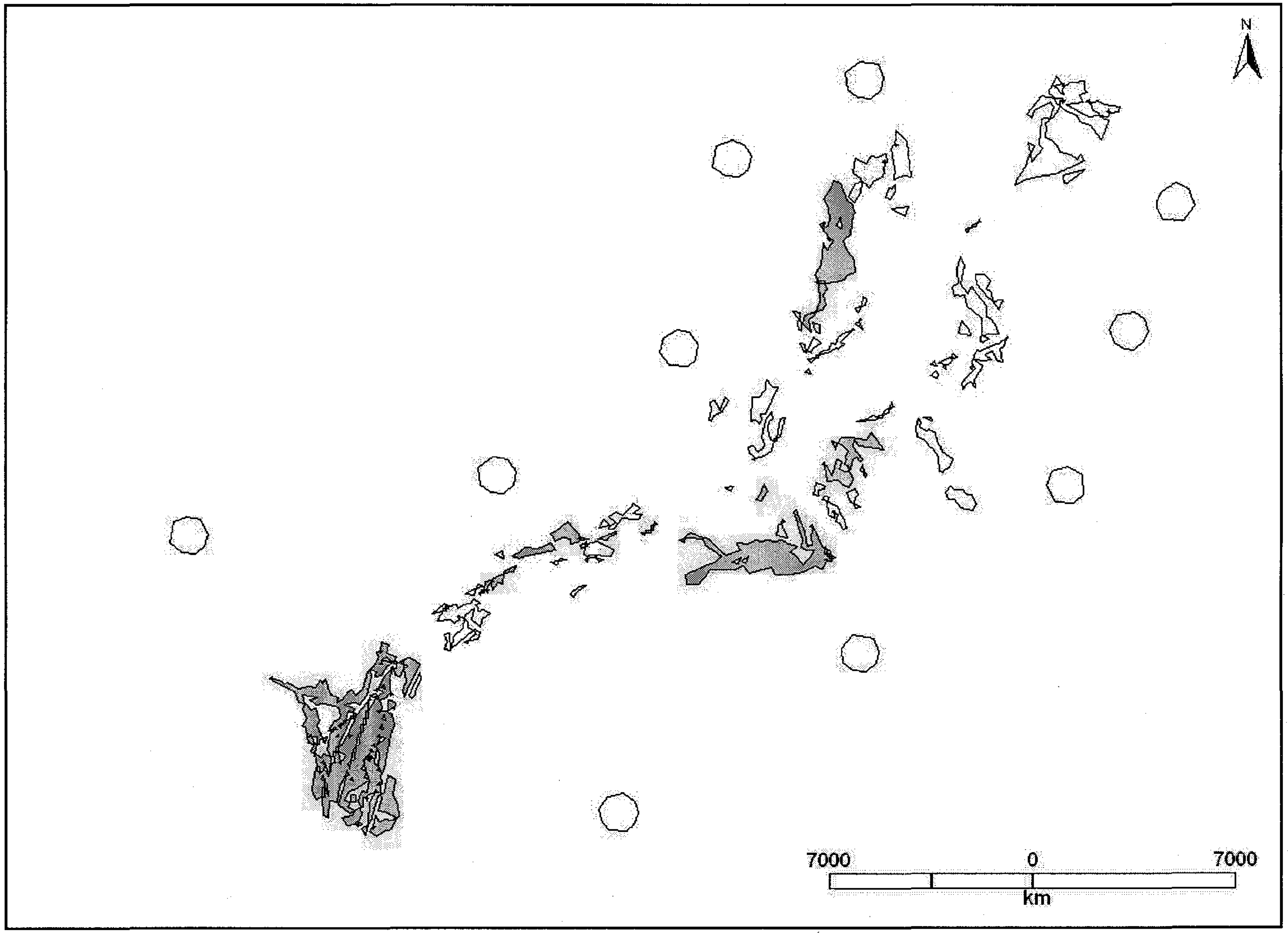
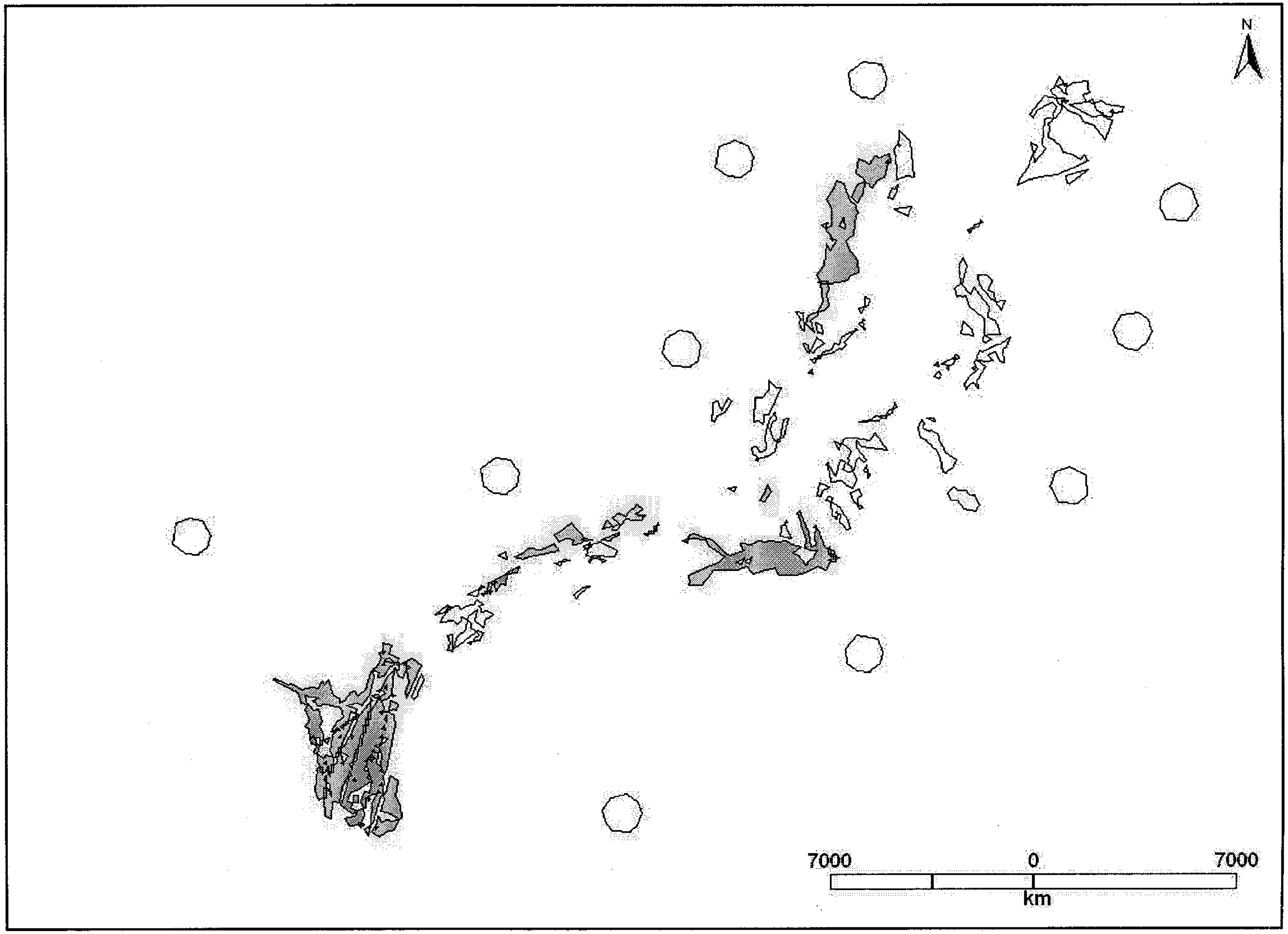


Figure 2-36 Developed area cost surface patch retention option. Highlighted patches represent patches to be maintained



Resolution Analysis

Table 2-3 Comparison between landscape statistics of original analysis and resolution analysis

	Original	10m	20m	30m	40m
Maximum Cost	182981	182620	179111	183609	175479
Minimum Cost	50	40	40	40	40
Patch Area	1236	1239.52	1235.68	1242.24	1233.6
Number of Patches	56	47	49	53	53

The spatial resolution of the input spatial data impacted both the cost surface and the patch map, altering all the landscape statistics (Table 2-3). The number of patches in the landscape increased, as the spatial resolution became coarser with a difference of 11 patches between the finest and coarsest resolution. However, the finest spatial resolution (resolution 10m), which had the fewest number of patches, did not have the least amount of total patch area. Resolution cell size 40m had the lowest patch area, while resolution 30m had the greatest amount of patch area. The area of the patch landscape only varied from the original analysis by 6.24ha between the greatest patch area, and 2.4ha from the lowest patch area observed, in this analysis. The minimum and maximum cost to travel between patches also varied between the original analysis and the resolution analysis. The minimum cost decreased from 50m in the original analysis to 40m in the finer resolutions, while the maximum cost both decreased and increased from the original analysis. Resolution 30m had the highest maximum cost distance while resolution 40m had the lowest cost distance between two patches. The cost to travel between the patches did not decrease or increase with the spatial resolutions, suggesting that the impact on cost is not simply related to the number of cells which comprise the landscape or the number of patches, rather it is more likely to be related to the change in the cell classification (i.e. slope value or barrier).

The resolution analysis also produces some variations in the area trade-off curves. One interesting difference occurs only in resolution cell size 40m (Figure 2-37), is that the lowest connectivity for the poor disperser is less than the lowest value for any of the other dispersers in the analysis. This differs from the original analysis (Figure 1-7) where the intermediate and poor dispersers reached the same lowest value of correlation length. It also appears that there is a slight decrease in connectivity for the poor disperser as patches are removed, however, again there is no distinct transition zone. The intermediate disperser "c" views the intact landscape (i.e. before patch removal) at resolution 40m as having greater connectivity than in the original analysis. There is no distinct transition zone that occurs for the best disperser at this spatial resolution, the loss of connectivity seems to be, as in past analyses, related to the loss in area rather than the removal of stepping stone or important patches. Transition zones for the intermediate dispersers occur after a greater amount of area is removed, in comparison to the original analysis. Intermediate disperser "c" experiences a transition at just less than 600 ha, in the original analysis however, at cell size 40m this transition zone occurred at 700ha. While intermediate dispersers "b" and "c" experienced a transition just above 600ha, compared to approximately 800ha as in the original analysis.

At resolution cell size 30m the most noticeable difference in the area trade-off curve (Figure 2-38) from the original analysis is the transition zone present at 1100ha for intermediate disperser "a", which occurred at just less than 800ha in the original analysis (Figure 1-7). The transition zones for the other dispersers remained similar to the original analysis with the best disperser having no distinct zone while intermediate disperser "b" and "c" remained at a similar location as in the original analysis at approximately 800ha.

As in previous analyses, there is no change in how the poor disperser views the landscape, as patches are removed.

Cell size 20m in the resolution analysis produced some interesting differences from the original analysis, as illustrated by the area trade-off curve (Figure 2-39). Intermediate disperser “b” views the intact landscape as more connected than in the original analysis while disperser “a” perceived the intact landscape as slightly less connected. Intermediate disperser “b” and “c” also perceive the landscape the same until the transition zone, contrary to the original analysis where all the dispersers perceived the landscape differently until the point of transition. Transition zones for the intermediate disperser “a” and “b” occur at approximately 900ha, in comparison to the original analysis where 100 more hectares of area were removed before a transition zone occurred. There is only a small shift in the transition zone for the intermediate “c” disperser from above 700ha to slightly less than 700ha at this resolution. However, a major drop in connectivity is evident for the best disperser in this analysis around 600ha, which was not apparent in the original analysis.

The final resolution analysis at cell size 10m produced some distinct differences from the original analysis, which are evident in the area trade-off curve (Figure 2-40). The intermediate dispersers “b” and “c “ initially view the landscape as less connected than in the original analysis. Also there was a distinct loss in connectivity for the best disperser that occurred at 600ha, differing from the loss in correlation length for this disperser in the original analysis, which occurred more gradually (Figure 1-7). Similar to other analyses there was little change in the area trade-off curve for the poor disperser. In regards to the intermediate dispersers, the transition zone for disperser "a" and "b" occurs around 700ha compared to approximately 800ha in the original analysis. Intermediate disperser "c" has two distinct transition zones, which occur at approximately 800ha and 600ha differing from the initial analysis where there was only a single transition at approximately 700ha.

The trends related to patch removal and area maintained were similar for all of the four spatial resolution analyses (Figure 2-41,2-42,2-43,2-44). Smaller patches tended to be removed first, while larger patches remained until later in the analysis. As in previous analyses, there were still small "stepping stone" patches that were removed late in the analysis. The order of removal was evidently different between the analyses, as illustrated by both the area vs iteration # graphs (Figure 2-41,2-42,2-43,2-44) and the order of removal maps (Figure 2-45,2-46,2-47,2-48). There were variations in the patches which were included in the final group of patches removed, however, these patches were not as distinctly associated with location on the landscape as in the cost analysis. There are a few key patches that varied between being included and not being included in the last category of patches removed. Two of these patches occurred in the northeast corner of the map, these two patches are the largest located in this area. However, these patches are not included in the final group of patches removed in all the analyses. The patch at the northeast corner of the map is included in the final group of patches removed in both resolution analysis 20m & 30m (Figure 2-46 & 2-47) and the original analysis (Figure 1-9), whereas this patch is considered less important at resolution 10m and 40m (figure 2-48 & 2-45). This is also true for another large patch located in the northeast section of the map, which is of considerable size and length. In resolution analysis 10m and 20m

(Figure 2-48 & 2-47), as well as in the original analysis (Figure 1-9) this patch was included in the final group of patches removed, whereas in resolution analysis 30m and 40m (Figure 2-46 & 2-45) this patch is excluded from the final group. This variation of patches included in the final group of patches was not just related to the number of patches in the landscape, as this variation was evident in the 30m and 40m resolution analyses, which had equal number of patches present in the landscape.

The number of patches to retain from this analysis to satisfy the needs of all the dispersal abilities, as previously mentioned, is the number of patches present before the first transition zone of any of the organisms. In the case, of the resolution analysis the area of the patches to be maintained is as follows: cell size 40m 694.08ha (Figure 2-49), cell size 30m 1114.72ha (Figure 2-50), cell size 20m 884.96ha (Figure 2-51), and cell size 10m 796.64ha (Figure 2-52). In comparison to the original analysis where 795.5ha were maintained (Figure 1-11&1-12) resolution cell sizes 20m & 30m would retain a greater amount of area in the reserve to maintain connectivity. However, cell size 10m would retain a similar amount of area to the original analysis, while cell size 40m would preserve less area. The appearance of the patches to be maintained in the reserve also differ from the original analysis, most noteable is the long patch present in the northeast corner of the patch map, which was discussed previously. This patch was not included as part of the reserve option in both the original analysis (Figure 1-11&1-12) and resolution cell size 40m (Figure 2-49). However, this patch was maintained in all the remaining resolution analyses (Figure 2-50,2-51,2-52). Similar to past analyses, there are core patches, which are maintained in all the reserve options despite the variations in the cell size.

Figure 2-37 Resolution analysis cell size 40m area trade –off curve illustrating relation between correlation length and area as patches are removed from the landscape

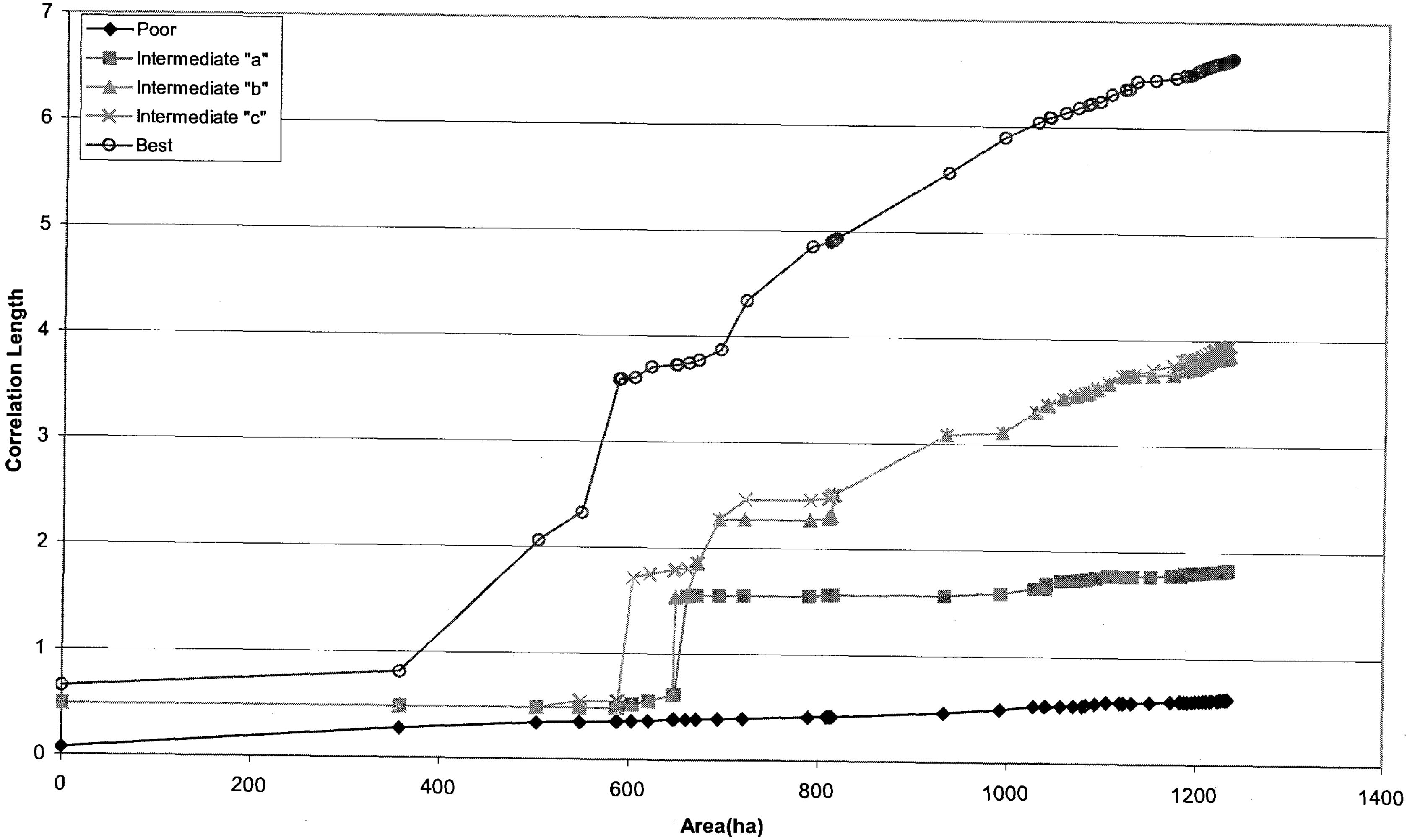


Figure 2-38 Resolution analysis cell size 30m area trade-off curve illustrating relation between correlation length and area as patches are removed from the landscape

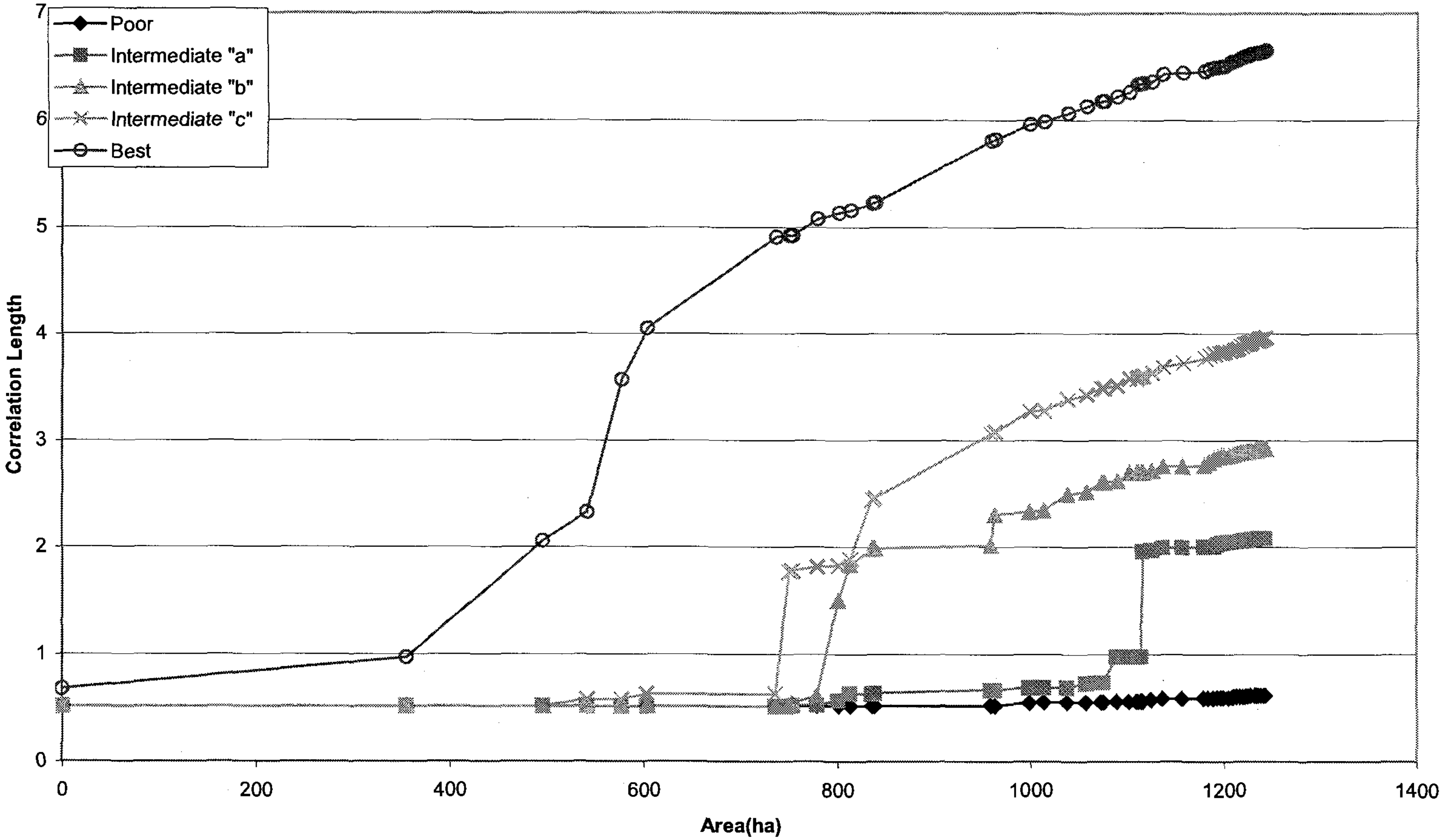


Figure 2-39 Resolution analysis cell size 20m area trade-off curve illustrating relation between correlation length and area as patches are removed from the landscape

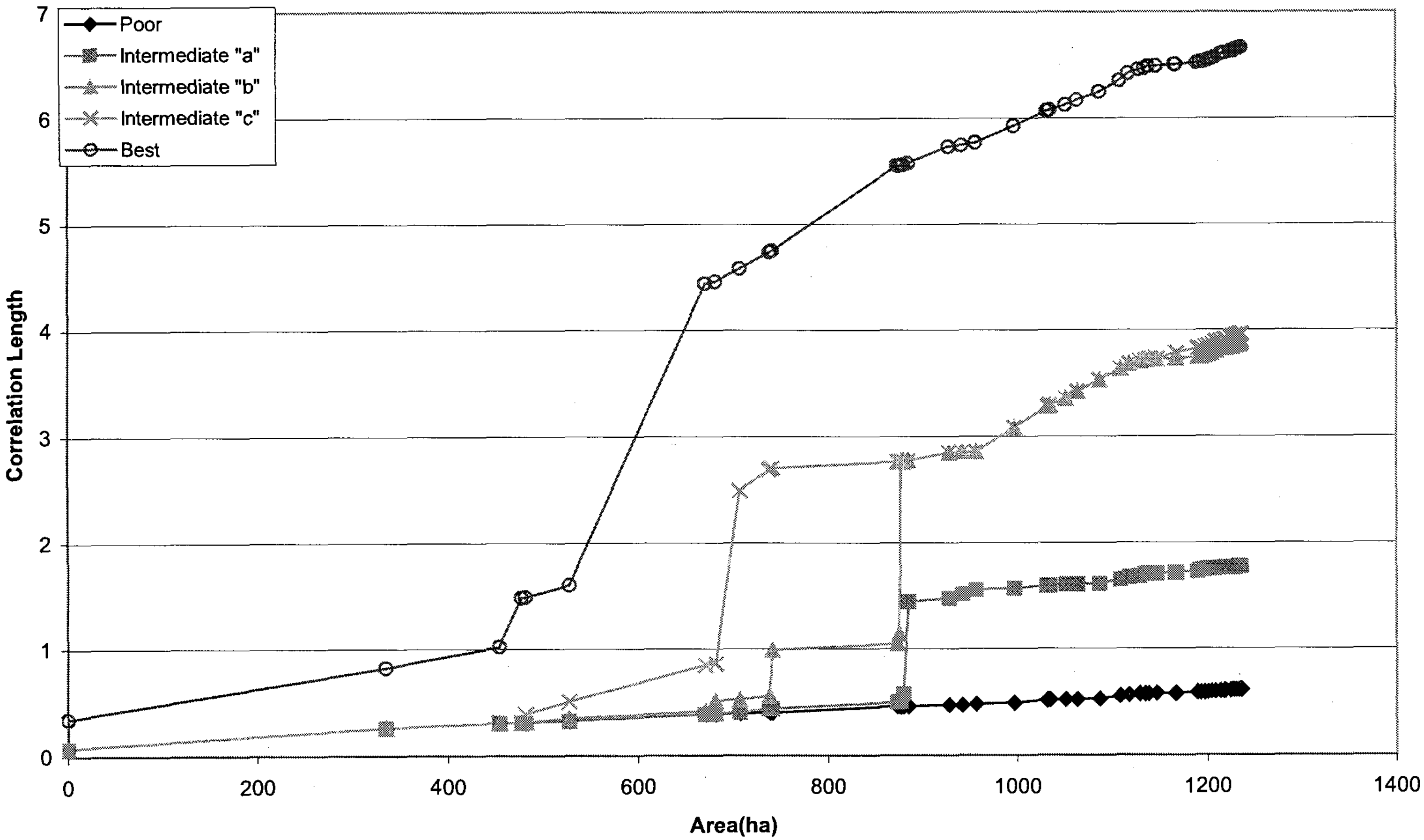


Figure 2-40 Resolution cell size 10m area trade-off curve illustrating relation between correlation length and area as patches are removed from the landscape

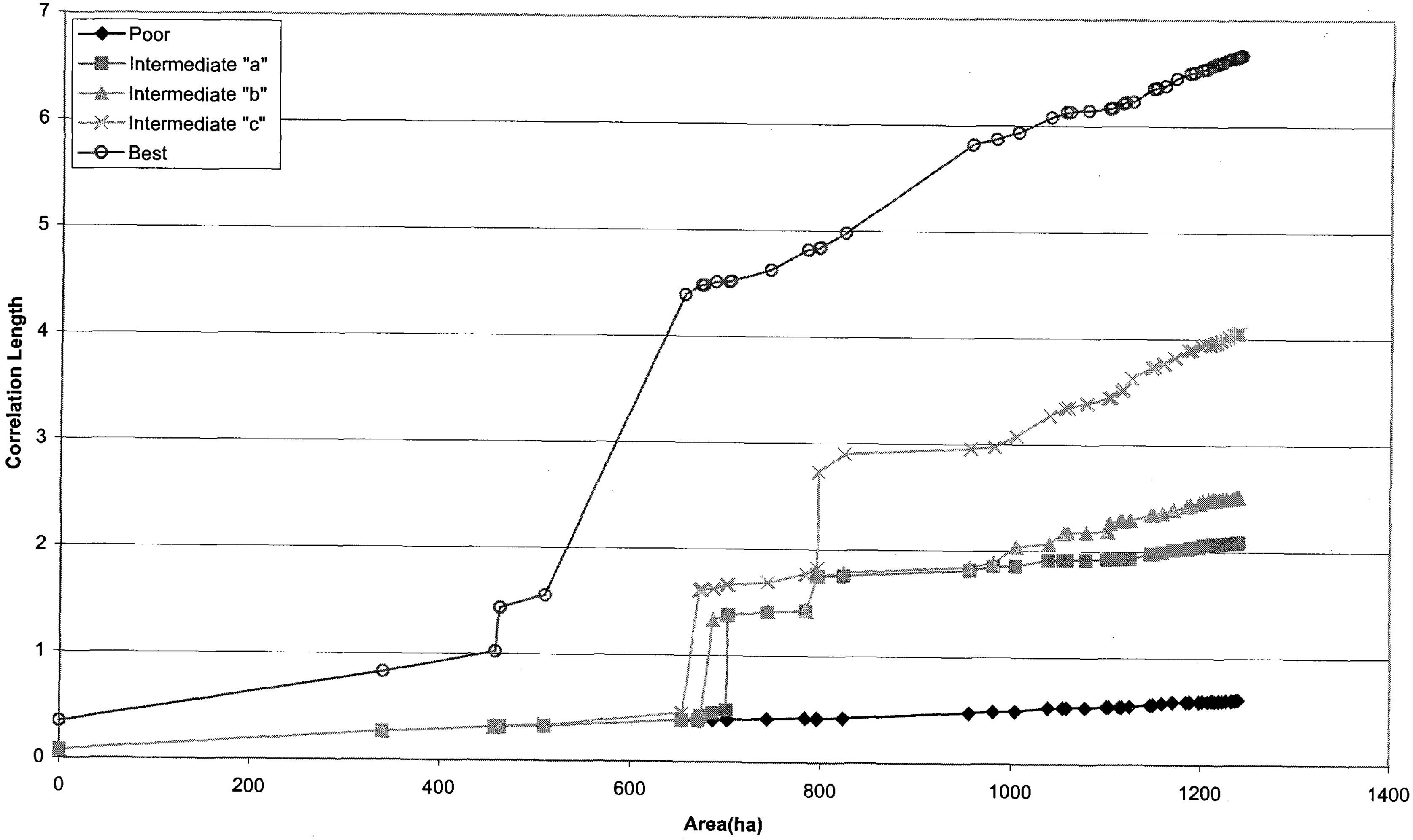


Figure 2-41 Resolution cell size 40m area of the patch removed at each iteration

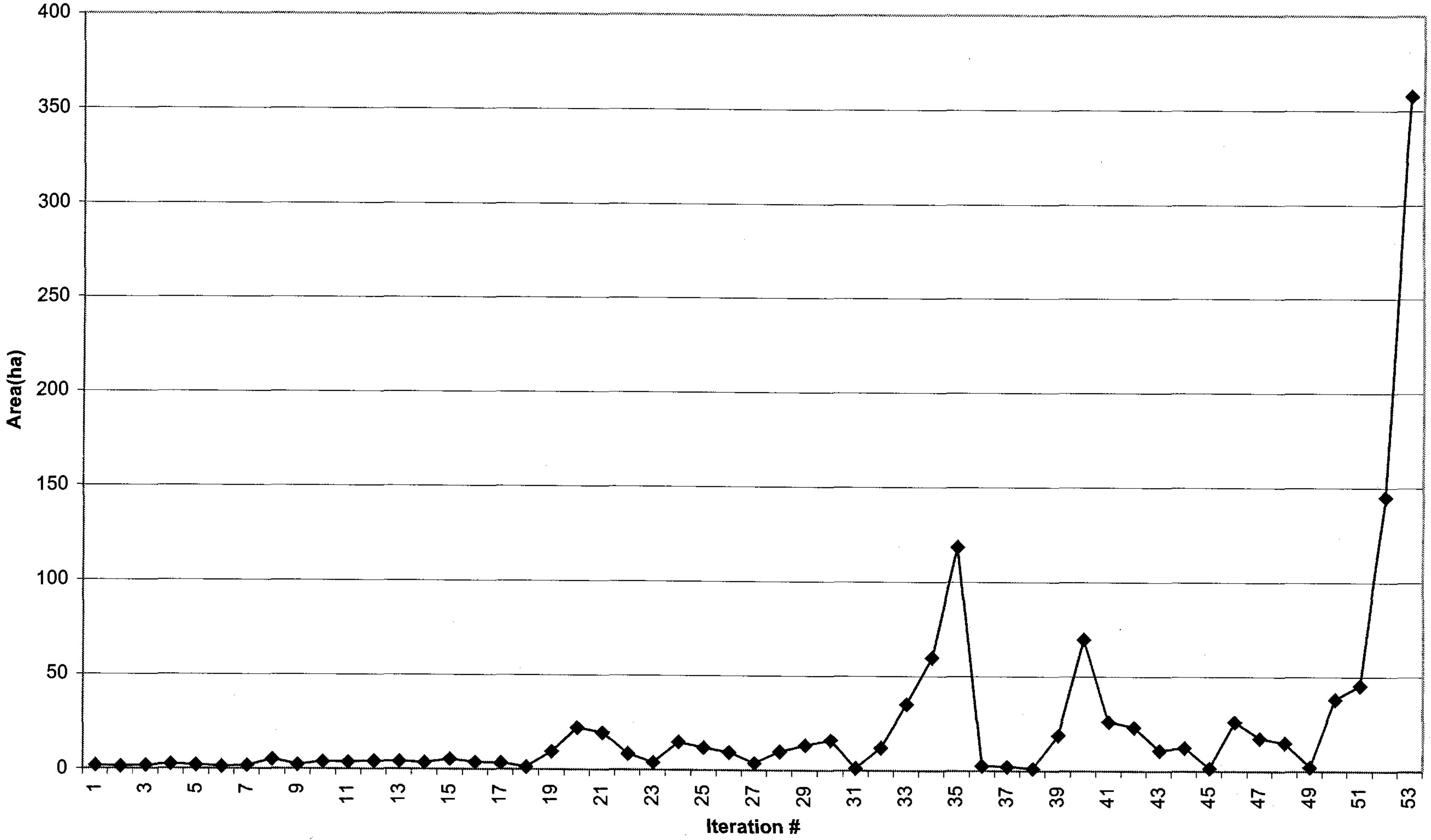


Figure 2-42 Resolution cell size 30m area of the patch removed at each iteration

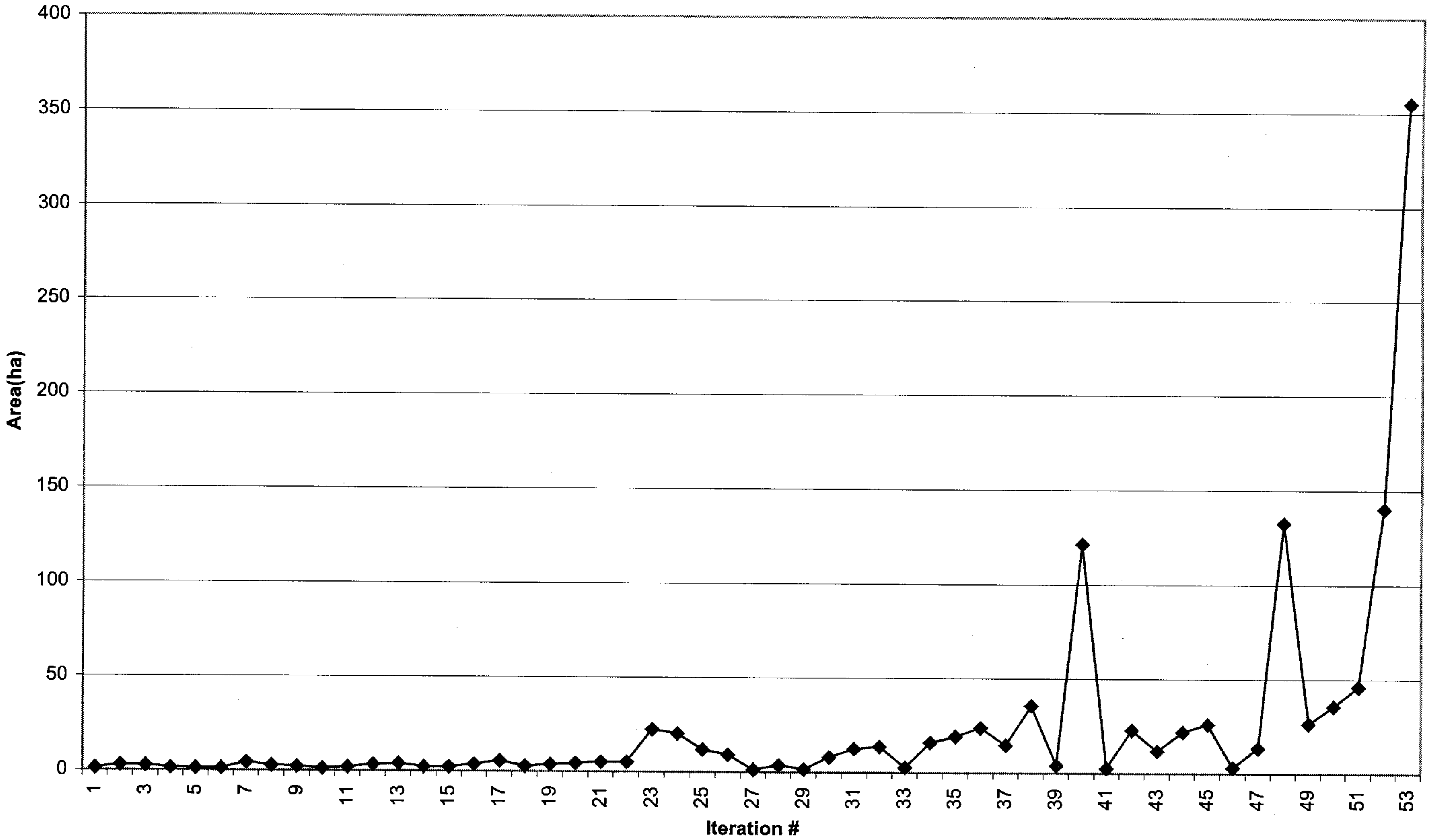


Figure 2-43 Resolution analysis cell size 20m area of the patch removed at each iteration

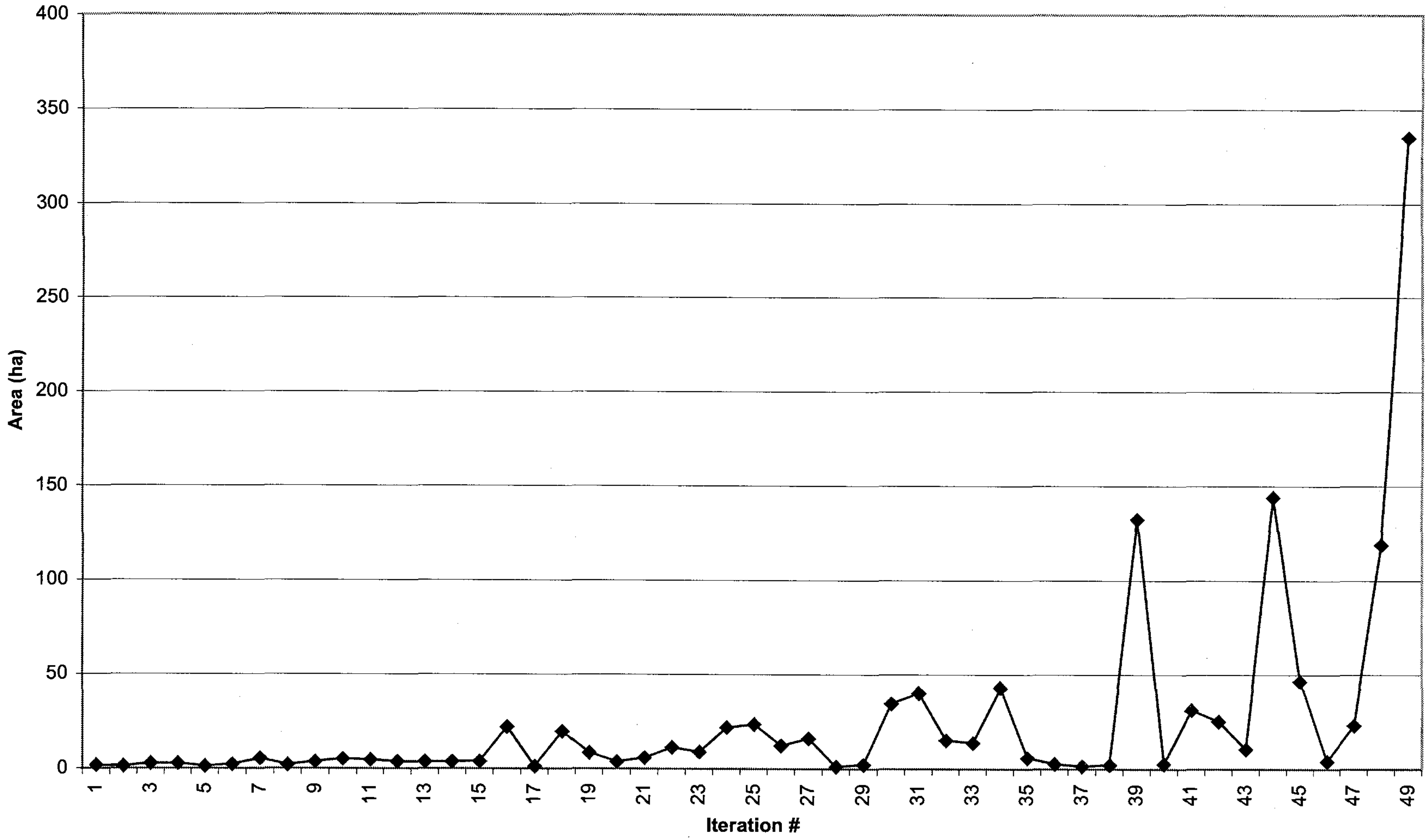


Figure 2-44 Resolution analysis cell size 10m area of the patch removed at each iteration

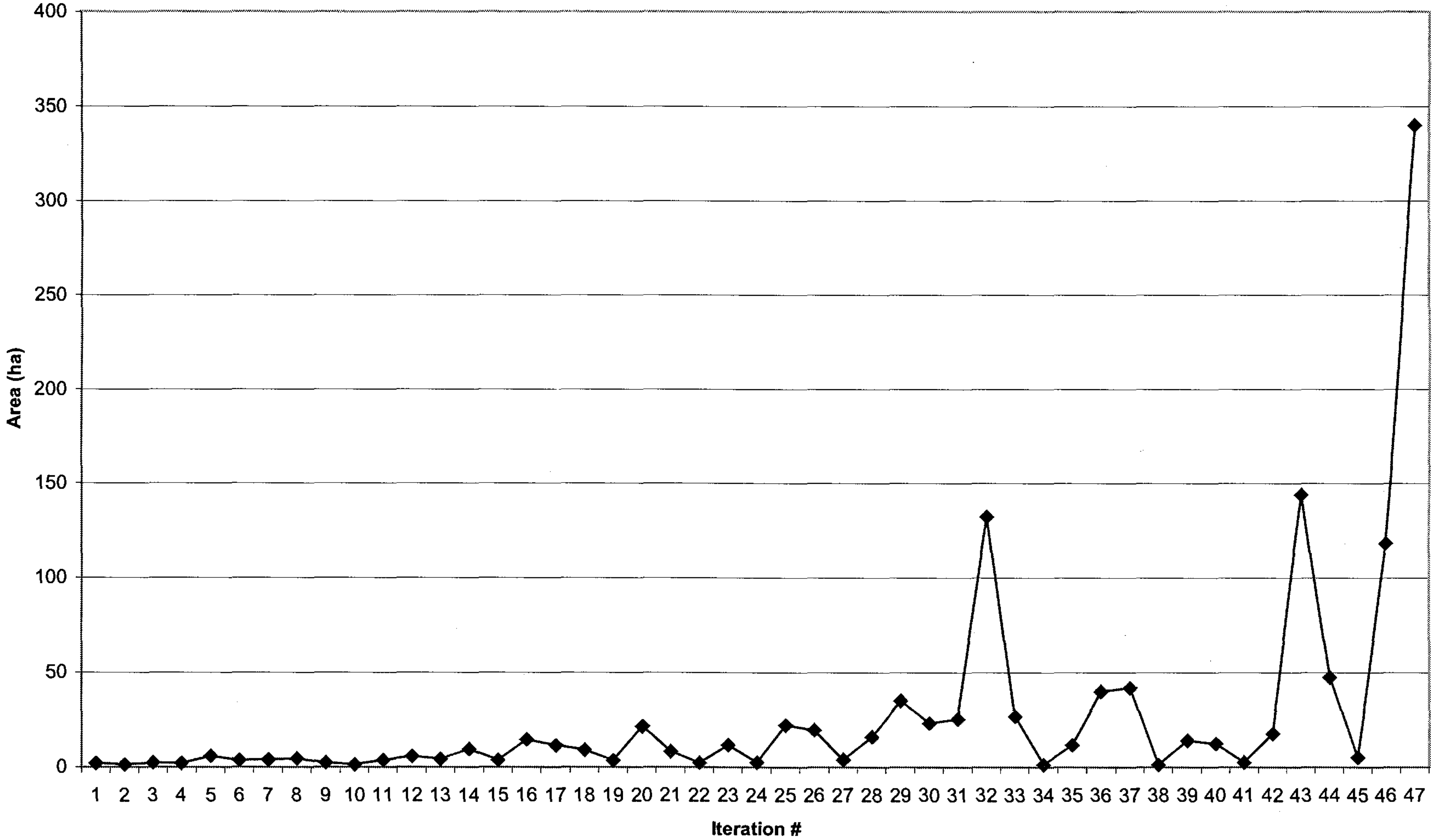


Figure 2-45 Resolution analysis cell size 40m order of patch removal classified by four equal intervals

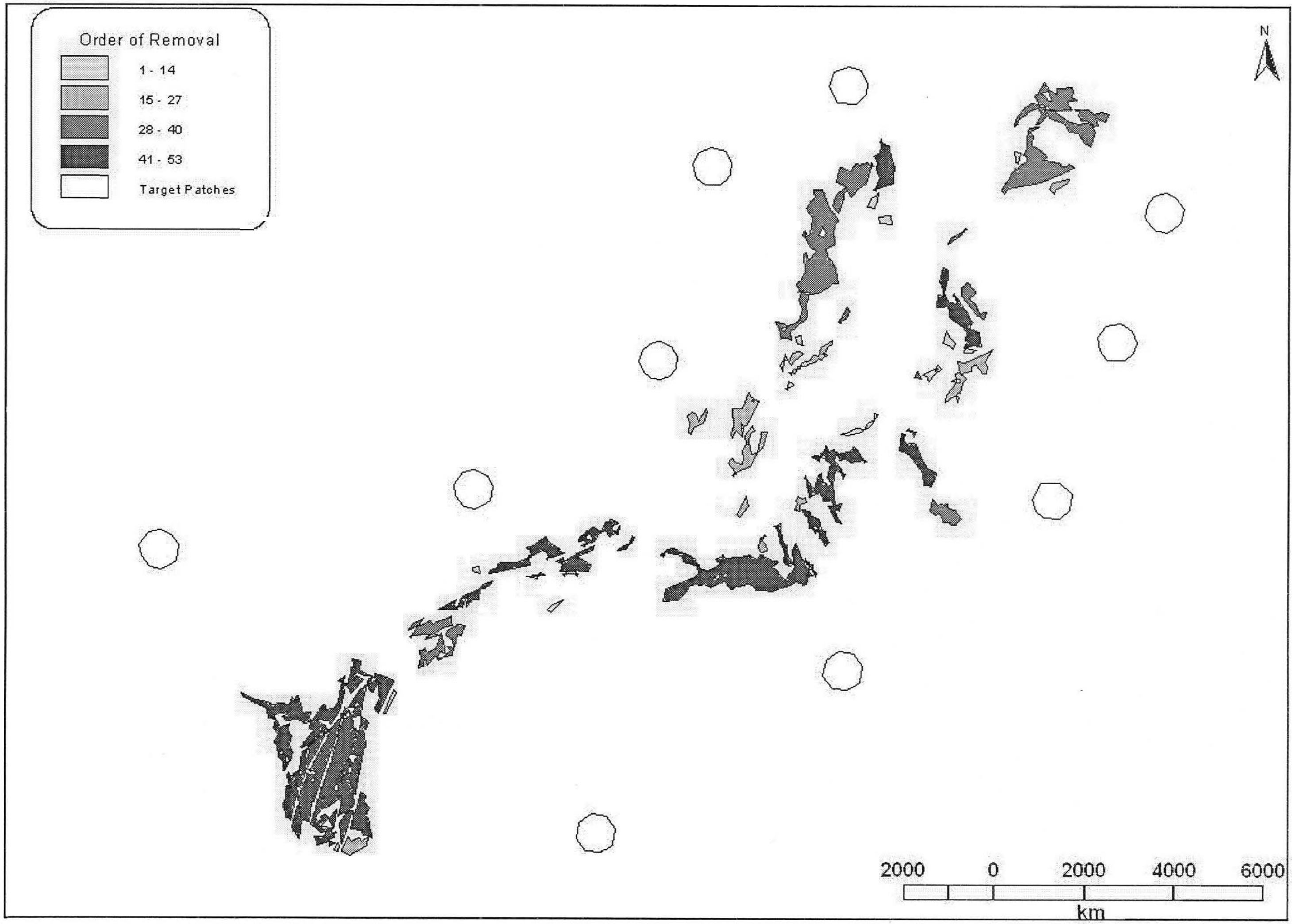


Figure 2-46 Resolution analysis cell size 30m order of patch removal classified by 4 equal intervals

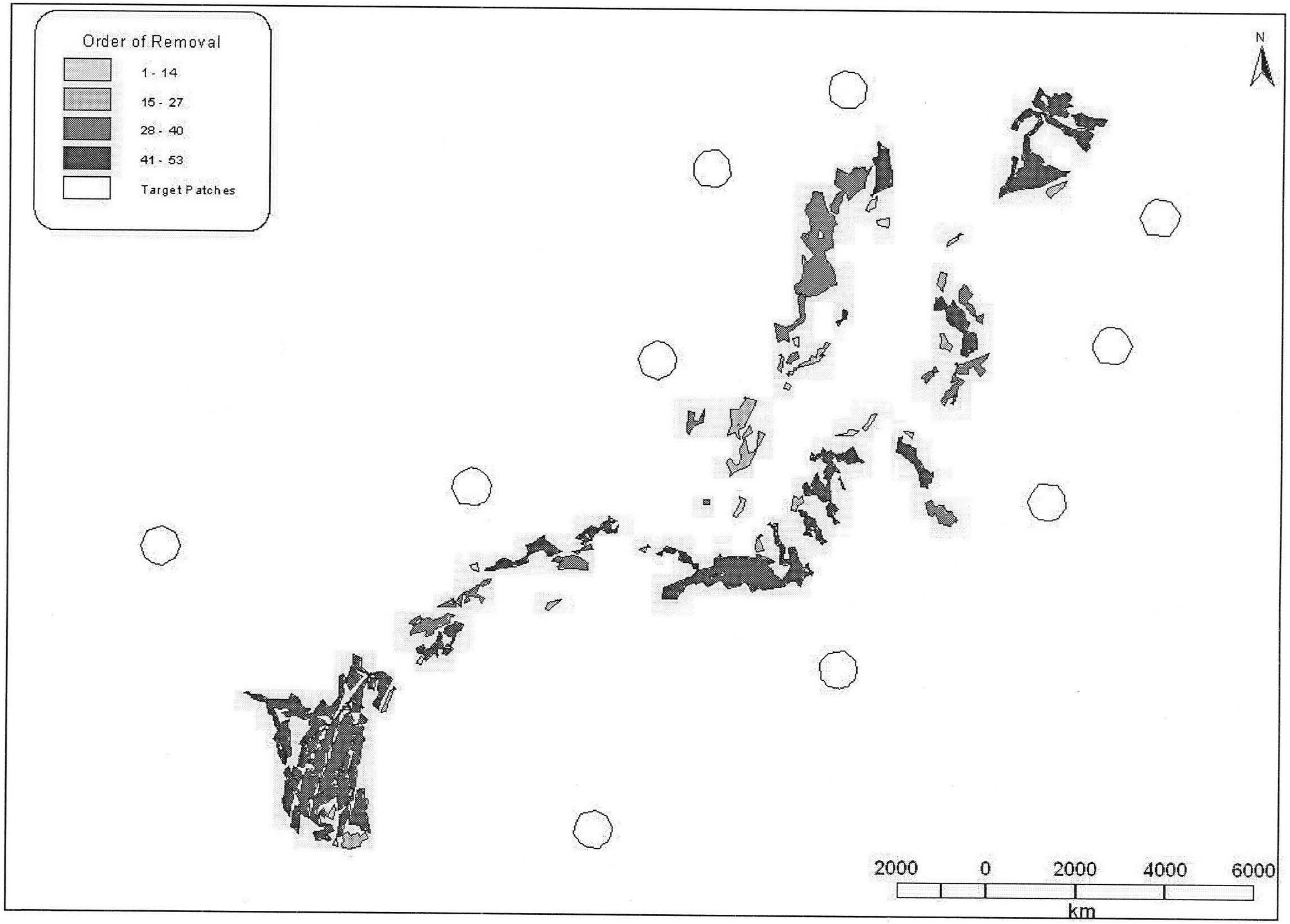


Figure 2-47 Resolution cell size 20m order of patch removal classified into four equal intervals

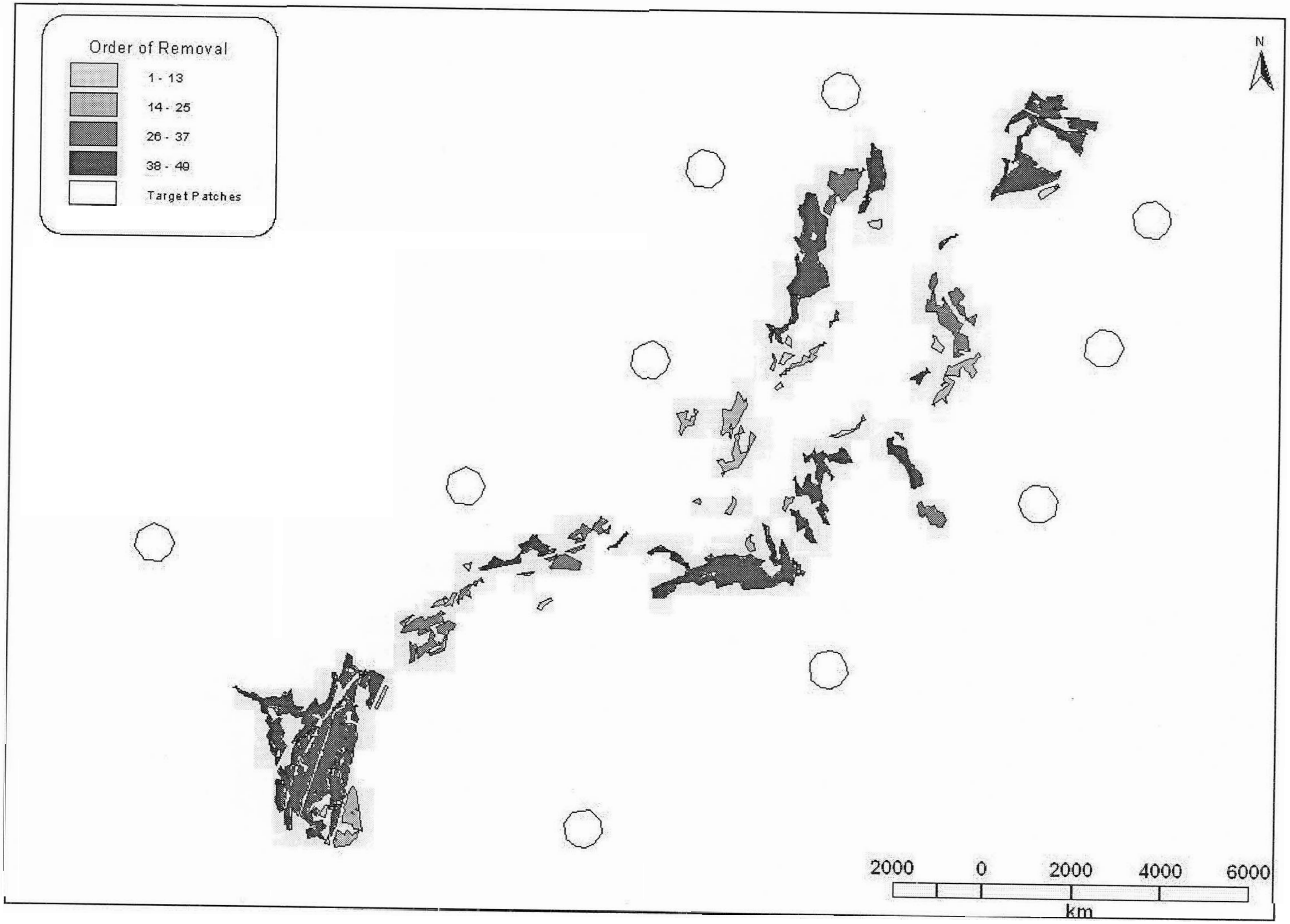


Figure 2-48 Resolution cell size 10m order of patch removal classified by four equal intervals

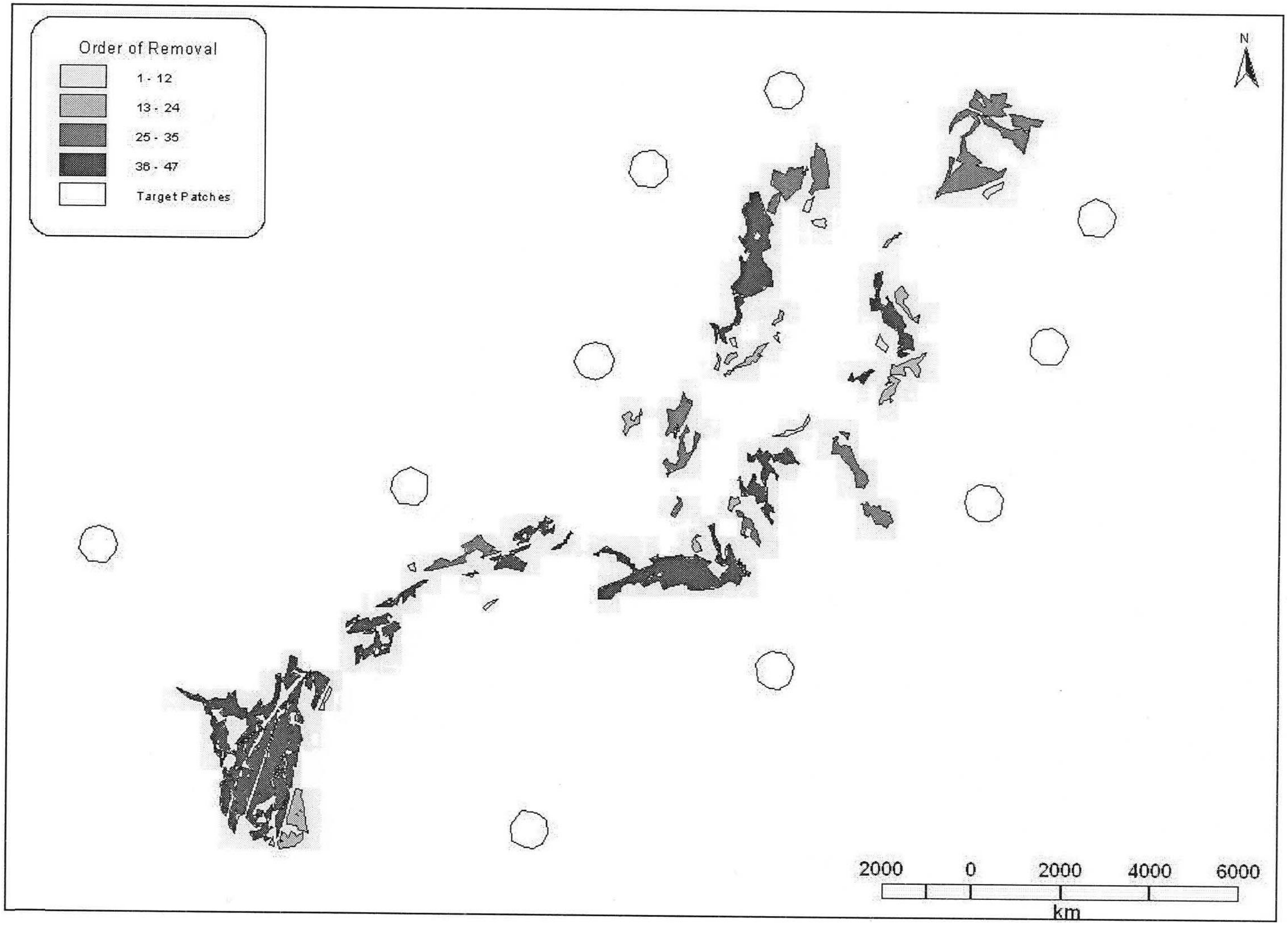


Figure 2-49 Resolution cell size 40m patch retention option. Highlighted patches indicate patches to be maintained

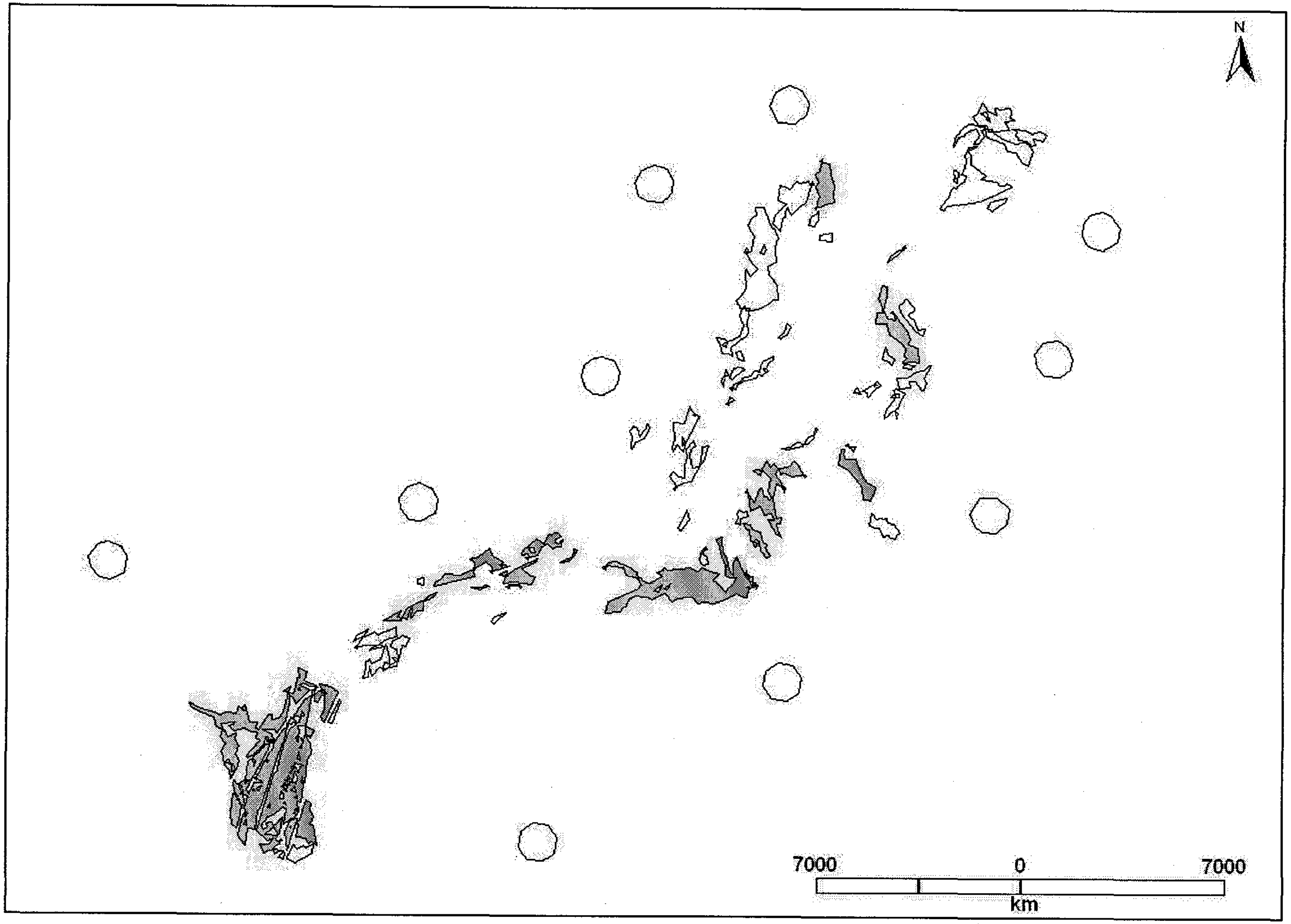


Figure 2-50 Resolution cell size 30m patch retention option. Highlighted patches indicate patches to be maintained

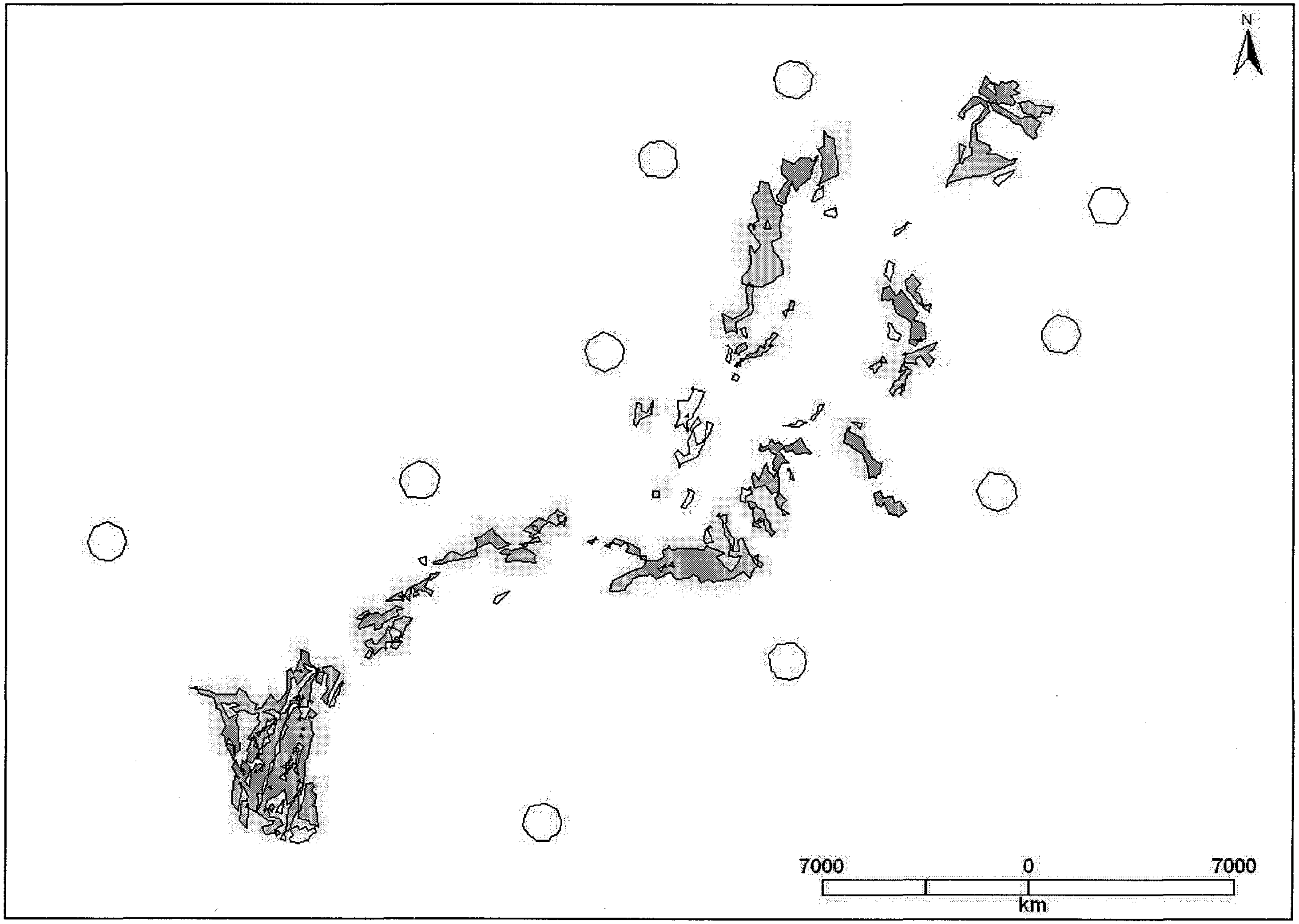


Figure 2-51 Resolution cell size 20m patch retention option. Highlighted patches indicate patches to be maintained

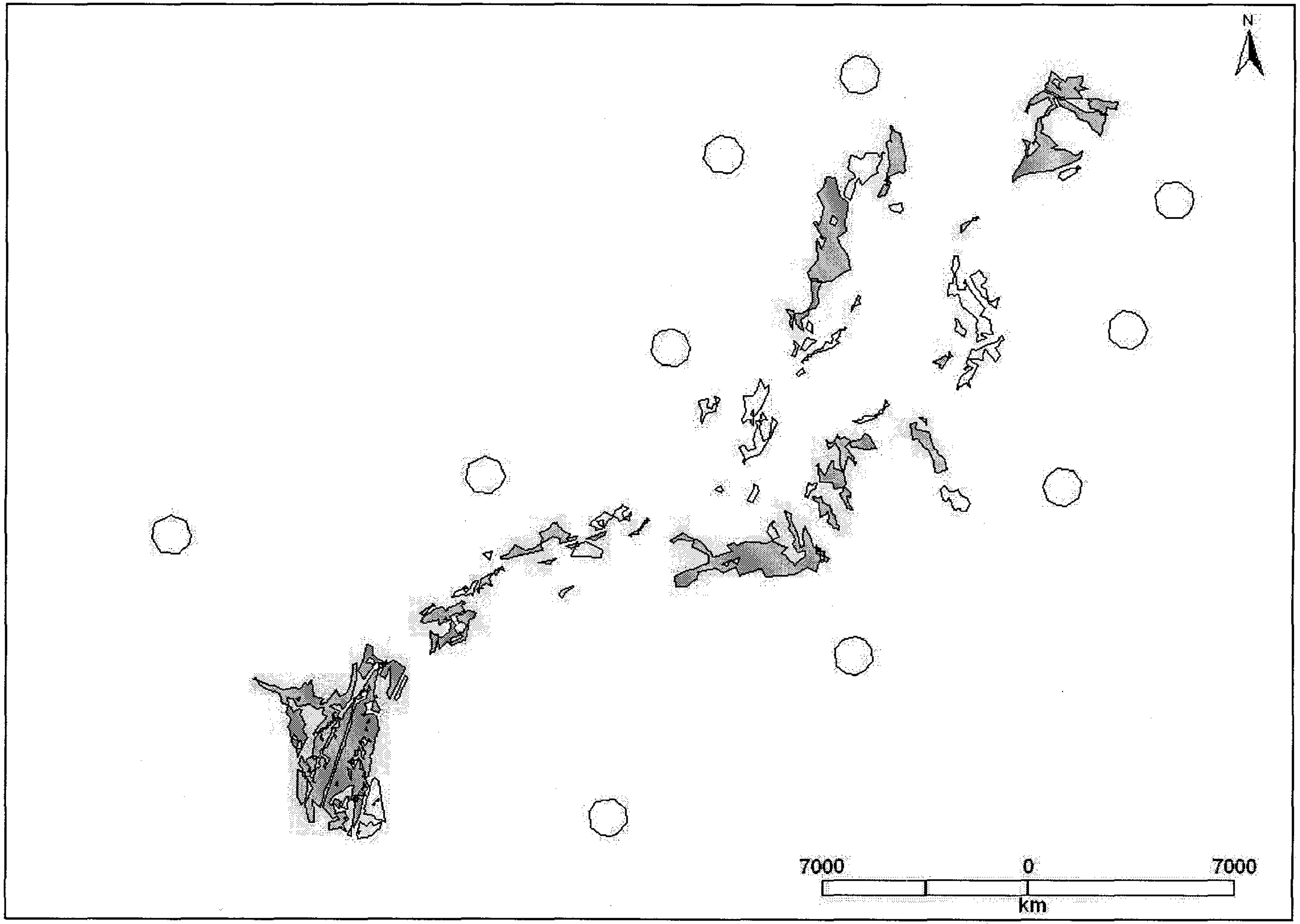
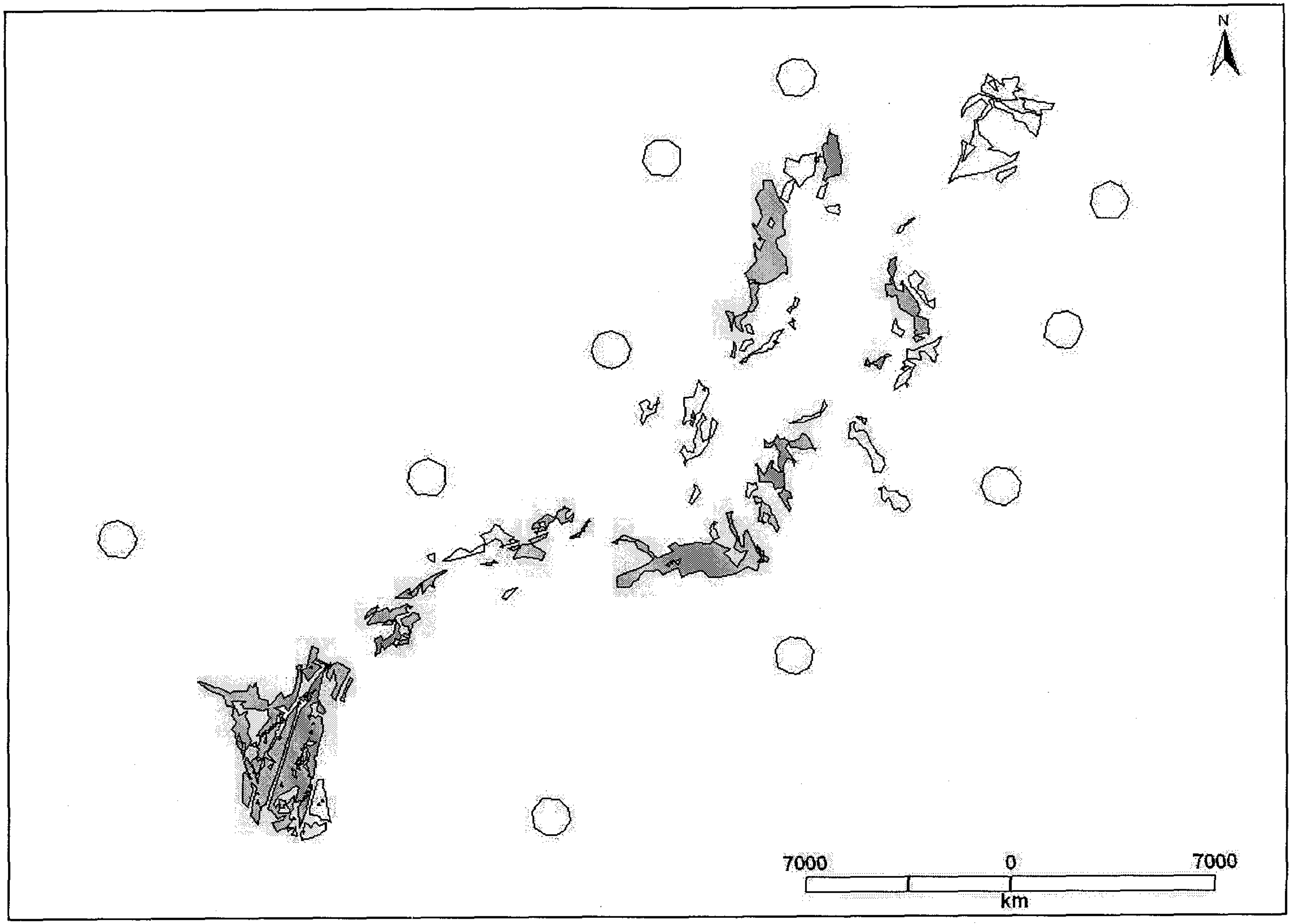


Figure 2-52 Resolution cell size 10m patch retention option. Highlighted patches indicate patches to be maintained



Discussion

The sensitivity analysis performed in this study illustrated how uncertainties in spatial inputs could potentially impact decisions made regarding land included in a protected area network. In general, the intermediate dispersers had the greatest sensitivity to changes within the spatial input data, while results attributed to the poor disperser were robust to variations within the spatial input data. Differences attributed to the dispersal ability were related to the spatial arrangement of patches in Whistler. Due to the spacing between patches the poor disperser already viewed the landscape as fragmented therefore removing patches had little impact on this disperser even with variations in the spatial data. The opposite was true for the best disperser, who could disperser over long distances without the need for stepping stone patches, and therefore could view the landscape as connected with very few patches. This is in contrast to organisms with moderate dispersal abilities, which required stepping stone patches to successfully traverse the valley between the target patches. Therefore differences created by the variations in spatial input data had the greatest impact on this group, due to the importance of the spatial arrangement and presence of patches to the successful dispersion of these organisms between the target patches. The different sensitivity of the dispersal groups to the analysis suggests that it is most important to fully understand the impact of variations in the spatial input data to mid-dispersal organisms, as these organisms are most likely to be impacted by the different alternatives produced by a sensitivity analysis. However, different spatial arrangements of patches on the landscape and the scale of the study could alter this result depending on how the organisms view the landscape.

The spatial resolution of the analysis and definition of the cost surface had the greatest impact on the final results of this methodology. It is quite evident from the results presented above that the spatial resolution of the analysis and cost surface have the potential to impact decisions made regarding the patches retained in the protected area network. In order, to mitigate against potential uncertainties as a result of the spatial resolution of the analysis, it should be ensured that the analysis is performed at the finest spatial resolution possible (Stokes & Morrison 2003). It is difficult to deal with the uncertainties related to the definition of cost, since this parameter could be defined in many different ways and in many cases there is little experimental data to support a definition. Therefore it is important to create alternative definitions and express the uncertainties surrounding the choice of definitions to those making the decisions.

Translating the impact of uncertainties in ecological modelling to real world systems and conservation efforts is difficult. When using model outputs in ecological decisions, a precautionary approach should be taken to account for uncertainties within the data. Alternative scenarios need to be created and the most conservative options should be recommended to decision-makers with emphasis on the potential uncertainties surrounding the outputs. Although, some of the differences observed in the area trade-off curves may have appeared small at a local scale, such as Whistler, even relatively small patches could potentially mean the difference between a connected and a disconnected landscape for some organisms. Also at larger scales, these variations may be more pronounced and with money at stake and dwindling resources available for conservation, these differences are very important to consider in the decision making process.

This research illustrates some important steps that Whistler can taken to ensure the most informed decision, regarding the retention of second growth forest patches in the protect area network. It was demonstrated above that there were core patches, which were present in all the reserve options presented from the various analyses. These patches should be considered important to maintaining connectivity due to the robustness to variations in the spatial data. Therefore, these patches could be confidently retained in the reserve due to their constant presence in all the presented patch retention options. Decision-making efforts can then focus upon those patches, which varied between the analyses. By presenting many possible scenarios, and allowing decision-makers to visually understand the uncertainties associated with methodologies that produce information, a more informed approach to decision making can occur. However, a balance must be maintained between presenting too many options, which may confuse decision-makers and lengthen the time required to make decisions. Failing to present other options does not allow the decision-makers to be aware of uncertainties in the data they are presented with and may lead to poor decisions.

Further research should focus on examining the impact of other forms of uncertainty on this selection technique, such as how considering patch characteristics impacts the measure of patch importance. Another issue, which should be examined is how sensitive this technique is to combinations of different types of uncertainties. This study only examined the issue of uncertainties within the spatial input data, and did not focus on the uncertainties of the model itself. The ecological assumptions of the model should be examined to ensure the robustness of this technique in reserve selection. It is also urgent that methodology be developed to express uncertainties in the output results to the decision-makers, so this issue maybe addressed in the decision making process.

Stoms et al. (1992) note that it is important not to tend towards the extremes of managing uncertainty, which are to disregard it or to look so critically at outputs that information is discarded due to potential uncertainties. Uncertainty will always be present in spatial modelling despite our best efforts to eliminate it. It is therefore important to acknowledge, study, and effectively communicate issues of uncertainty surrounding any output results of spatial modelling to the decision maker. More informed conservations decisions can be made, by simply making those with decision-making power aware of the potential uncertainty in the information they are being provided.

CONCLUSION

This research study presents a modeling method based on graph theory that can be applied to reserve design, by identifying patches important to maintaining connectivity. Further this research, illustrated how uncertainty in spatial datasets could potential alter output products of this methodology. The application of graph theory to protected area design can provide valuable information to decision-makers. However, one must consider the uncertainty that may surround the final results, due to the initial spatial data inputted into the analysis. Sensitivity analyses provide a useful tool to help identify and mitigate against uncertainty in the application of graph theory to protected area design. This type of analysis allows a range of alternatives to be presented to the decision-makers and helps to identify areas that are robust to variations in spatial input data. It also provides an opportunity to examine areas of uncertainty and revisit initial spatial inputs, spending more resources and time on those areas with the greatest potential to impact results. In regards to the case study in Whistler, where the cost surface was found to have the greatest impact on the results, further research could be performed to determine movement cost in the valley. This would aid in increasing the confidence level associated with this surface, thereby improving the information upon which final decisions would be based.

The results of this uncertainty analysis should not be applied to further applications of graph theory to ecology. The uncertainty surrounding the spatial inputs and the impact on results may change, when applied to different landscapes or scales of analyses. Sensitivity analyses should be incorporated into every similar application of graph theory, since the nature of the both the spatial input data and the analyses will change.

The methodology for determining an order of patch importance is limited by its inability to incorporate population information into the movement model. Therefore this methodology should not be applied to metapopulations, as it does not consider that patches may act both as sources and sinks to movement. Although, the lack of need for population information is also a strength due to the large amount of time and resources required to collect such information. Graph theory models can provide an overall understanding of the landscape and the ability of patches to contribute to movement, however, care should be taken to consider properties of patches that may effect an animal's movement across the landscape. The model may be able to incorporate this issue by assigning cost to patches, which are rich in resources that may lead to organisms remaining instead of just using the patch as a stepping stone.

Unfortunately, not all possible sources of uncertainty could be examined in this study. Further research should focus on studying the ecological aspect of uncertainty within the model, especially in regards to assumption made about animal movement. The uncertainty of GIS also needs to be further examined and should focus on examining how errors in boundary locations or conversion operations could impact results. It may also be interesting to determine how error propagates through the model used in this research and also determine how combined forms of errors and uncertainty impact final results.

The premise of this study applies to the broader application of GIS to ecology. The concerns addressed by Stokes & Morrison (2003), in regards to errors and uncertainty in GIS can be addressed through applying sensitivity analyses. The application of sensitivity analyses ensures that there is a consideration of uncertainty in the analyses and provides a means to communicate potential problems in the analyses to decision-makers. It is not only important to further research the application of GIS in ecology, but to also simultaneously research and develop methodology to quantify and express uncertainties associated with the use of spatial information and GIS. Uncertainty research in the application of GIS is an important topic. This type of research could improve information being provided to those making decisions in conservation and environmental management. Being aware of potential problems is the first step in mitigating against uncertainty. Once there is awareness, tools can be developed and applied to ensure more informed decision-making.

"If a man will begin with certainties, he shall end in doubts, but if he will be content to begin with doubts, he shall end in certainties." This quote from Sir Francis Bacon describes how to best deal with the seeming overwhelming uncertainties that can appear with the application of GIS. If we begin by doubting our methods we will end with greater certainty in our decisions than had we just ignored and not questioned our methods. We cannot eliminate uncertainty, as it is inherent in modeling, however, by addressing it we can bring greater certainty to our results and ultimately our decisions. It is therefore important to continue researching the issue of uncertainty in the ecological application of spatial models and GIS. Further research and awareness will help prevent situations such as those encountered by the Save the Redwoods League, where lack of awareness about GIS uncertainties almost led to poor conservation decisions (Stokes & Morrison 2003). GIS is a useful tool to environmental and natural resource management, as illustrated by this research, and through incorporating research from geographic information science to the ecological application of GIS we can improve the way in which we plan and implement environmental policy.

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