Factors Affecting the Invasion of *Pteronia incana* (Blue bush) onto Hillslopes in Ngqushwa (formerly Peddie) District, Eastern Cape.

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VINCENT KAKEMBO

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Abstract

The factors that influence the invasion of hillslopes by the shrub *Pteronia incana* in the communal rangelands of Ngqushwa (formerly Peddie) district were investigated. Mgwalana, one of the catchments in the district where encroachment by the species is widespread, was chosen. The study combined field observations with image analysis based on high resolution infrared imagery.

The catchment was flown and high resolution infrared images (1mx1m) were taken using a Kodak DCS420 digital, colour-infrared camera. The images were analysed using Idrisi32 and Kilimanjaro GIS versions. The ability of different vegetation indices to separate *P. incana* from the other cover types was investigated. Field observations of the degree of *P. incana* invasion in relation to, *inter alia*, soil surface conditions, slope angle and visible forms of erosion were made.

A Digital Elevation Model (DEM) of 20m spatial resolution was used to derive terrain parameters. The presence and absence of *P. incana* in relation to slope gradient and aspect were determined. The combined influence of the two terrain parameters and land use on the invasion was also investigated. The Topographic Wetness Index (WI), a component of the TOPMODEL was derived from the DEM and its relationship with the spatial distribution of *P. incana* was explored. Soil moisture dependencies for *P. incana* and grass species as well as surrogates for runoff under the shrub and adjacent bare areas were determined in the field.

A high level of classification accuracy confirmed the reliability of digital camera imagery for spatial analyses. Distinct spectral separability for the surface vegetation cover types was achieved by means of the Perpendicular Vegetation Index (PVI) as opposed to the ratio based vegetation indices (NDVI, SAVI and MSAVI). The absence/presence of *P. incana* was noted to be strongly influenced by slope angle and aspect. The probability for *P. incana* occurrence increased with slope steepness and southerly slope orientation. Abandoned and grazing lands were identified as the main invasion hotspots. Blanket invasion of the former signified the high susceptibility of abandoned land to *P. incana* invasion. The combined influence of land use, slope gradient and aspect was also noted to have promoted the invasion. This is borne out by the concentration of the invasion on abandoned steep slopes with a southerly orientation.

Local topographic variations were identified as having a strong bearing on *P. incana* spatial distribution. The topographically driven WI confirmed this relationship, such that *P. incana* was associated with the low WI values of convexities. Differences in the moisture dependencies between *P. incana* and grass species were demonstrated by the greater rooting depth of the former. During field surveys, soil surface crusting was noted as inherent to *P. incana* patchiness. The coupling between local topography and soil surface crusting underpins soil moisture variability at hillslope and patch scales respectively. This in turn determines the competition between *P. incana* and grass species and the eventual replacement of the latter by the former.

A close spatial correlation between fully established *P. incana* and severe forms of soil erosion was observed. Loss of patchiness and expansion of inter-patch bare areas promote runoff connectivity erosion. As most of the runoff becomes run out, hillslopes tend towards dysfunctional systems. Greater soil moisture storage after rainstorms under *P. incana* tussocks than the adjacent bare areas signifies the shrub’s water harvesting capabilities. The tussocks could thus serve as a starting-point-building- block for the rehabilitation of dysfunctional hillslope systems.

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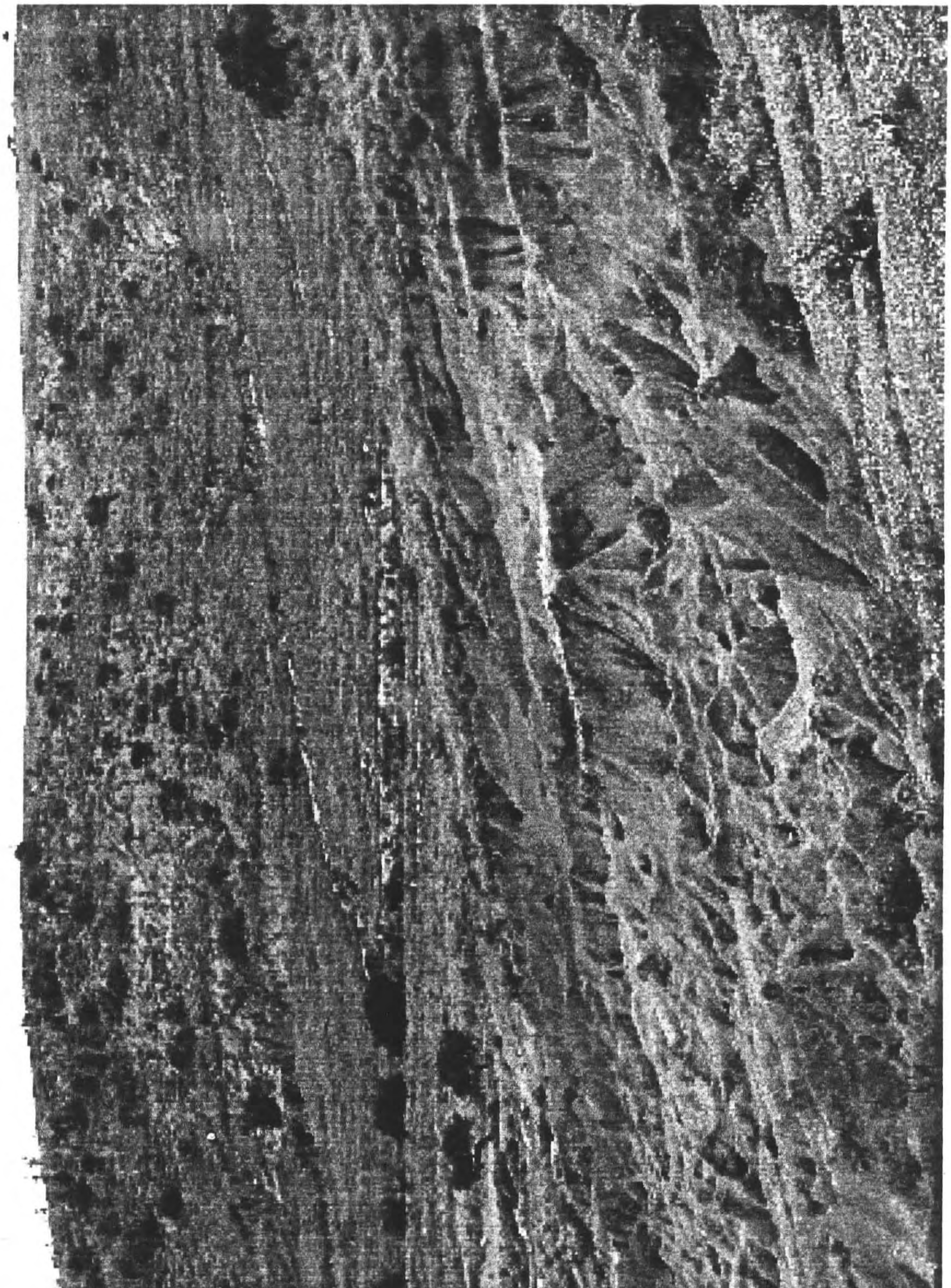
BLR -	Binary Logistic Regression
DEM -	Digital Elevation Model
DN -	Digital Number
FCC -	False Colour Composite
GCP -	Ground Control Point
GIS -	Geographic Information System
GPS -	Global Positioning System
HRI -	High Resolution Imagery
KIA -	Kappa Index of Agreement
MLE -	Maximum Likelihood Estimation
MSAVI-	Modified Soil Adjusted Vegetation Index
NDVI -	Normalised Difference Vegetation Index
NIR -	Near Infrared
P/B -	*P. incana* / Bare surface ratio
PCP -	Percent Correct Predictions
PVI -	Perpendicular Vegetation Index
RMS -	Root Mean Square
SAVI -	Soil Adjusted Vegetation Index
TIFF -	Tagged Image File Format
TSAVI-	Transformed Soil Adjusted Vegetation Index
VI -	Vegetation Indices
WI -	Wetness Index



Pteronia incana in badlands at one of the sites in the study area.

Chapter 1: Introduction

1.1 Introduction

Vegetation degradation is a widespread problem in vast areas of the Eastern Cape Province, particularly in the communally owned rangelands. A characteristic feature of these degraded rangelands is the replacement of perennial grasses by annual species and unpalatable dwarf shrubs. Surveys conducted in parts of the Eastern Cape by Palmer and Avis (1994) and Kakembo (1997) identified *Pteronia incana* (Blue bush), an unpalatable Karroid dwarf bush, as one of the most widespread invader species. In the latter study, the shrub was observed as spreading rapidly from stream valley bottoms onto grazing lands and abandoned cultivation fields. Whereas the extent of vegetation degradation has been established as well as its spatial distribution patterns (Palmer and Avis 1994), the factors that influence the invasion and the relationship of the invader species to soil biophysical properties and erosion potential are yet to be identified.

Alterations to vegetation cover may engender a host of changes in soil attributes which in turn affect erosion rates. According to Tongway and Hindley (1995), much of the change that occurs when grasslands move from one condition to another is associated with changes in soil surface characteristics. This view is supported by numerous researchers (cf. Trimble, 1990; Illius and O’Connor, 1999; Holm, 2000) who point out that changes to poorer vegetation cover and management may lead to a deterioration of the soil and its associated biophysical attributes. With the loss of perennial species, intershrub space increases and bare soil with lower nitrogen and organic matter levels increases, especially under high intensity grazing (Allsopp, 1999). Indisputable is the protective role of vegetation, particularly against raindrop impact and surface runoff. Ground cover between 20%-30% is considered to be critical for sheet wash (Rowntree, 1988), while a cover of 40% and greater provides protection from rill erosion (Govers, 1991).

That role notwithstanding, critical elements of vegetation, notably its structure, govern this protective role. Inferior vegetation species may create an impression of a healthy cover, yet they are highly ineffective in terms of erosion control. The ‘fetch’ length (distance between two plants), obstruction width and cover length, all of which vary with vegetation type, constitute vegetation structure. It is this spatial arrangement of vegetation elements that controls the hydraulic efficiency, particularly surface runoff amounts and flow lengths and paths. An increase in the fetch as a result of the loss of patches occupied by long-lived plants will alter the runoff/run-on and erosion/deposition dynamics so that each runoff event involves larger volumes of water (Snyman, 1999; Holm, 2000). In their study of water movement and vegetation patterns on shrubland and abandoned fields, Bergkamp *et al* (1996) noted rapid infiltration of locally generated overland flow near spatial structures in vegetation. According to Trimble (1990), as increasing proportions of a catchment are placed under poorer vegetative cover, surface runoff amounts and flow lengths become greater. Rills and gullies which are more efficient conduits of water conveyance may develop over time.

It is noteworthy that the relationship explained above is a bi-directional process. “C*hanges to soil quality by some degradation processes may alter the output of the environmental envelope so that the availability of water and nutrients over time is insufficient for some vegetation species to persist. An example is the loss of perennial grasses from a landscape*”. (Tongway and Hindley, 1995 p.4). The interactions between vegetation and soils bring about soil states which reinforce changes in vegetation (Allsopp, 1999). Given the typically unreliable rainfall and usually poor soils in rangelands, both vegetation and soil systems are therefore very vulnerable, especially to human manipulation. Against this background, an attempt is made in this study to establish the relationship between the encroachment of the veld by *P. incana*, the degradation of soil properties and soil erosion. The shrub is widespread in the communal lands of Ngqushwa (formerly Peddie) district, Eastern Cape where it is associated with severe erosion and the greatly reduced grazing capacity due to unpalatable nature of the shrub. These aspects are investigated within a catchment framework. Mgwalana, one of the catchments where encroachment by the invader species is tremendous, has been chosen. The findings within this framework could provide the basis for developing appropriate intervention strategies which could be extrapolated to a wider regional scale.

1.2 Research problem

Vast areas of the upper Mgwalana catchment, Ngqushwa district, have been invaded by *P. incana*. Blanket invasion has been observed on abandoned lands, most of which are severely eroded, as opposed to grazing lands where the invasion is gradual. The invaded parts of the catchment are characterised by intershrub bare surfaces, soil crusting and severe soil erosion. It is envisaged that the invaded areas, which comprise mainly hillslopes, have been converted into a non-resilient and dysfunctional run-out system where the loss of water, soil and nutrients into the adjacent stream channels has reached alarming proportions. The unpalatability of the shrub compounds the problem in that greater grazing pressure is mounted on the remaining natural grasslands. The grazing pressure is seen to exacerbate the invasion, providing *P. incana* with a competitive advantage over grass species.

The major research questions this study seeks to unravel are: (i) What are the factors that influence *P. incana* invasion? (ii) What is the relationship between *P. incana* and soil physical properties, particularly soil surface conditions? (iii) What are the implications of *P. incana* invasion for runoff generation and soil erosion?

After identifying the factors and understanding processes that underpin the invasion by *P. incana*, a protocol for developing the appropriate rehabilitation strategies will be recommended.

1.3 Research Aim and Objectives

The principal aim of this study is:

> To identify the relationship between the encroachment of rangeland by *P. incana* and rangeland degradation at the catchment scale.

The specific objectives by means of which this aim is achieved are:

1. To investigate the factors that influence the invasion of rangeland by *P. incana*.

To achieve this objective, land use changes in the study area are examined from sets of aerial photographs taken at earlier dates. The changes are compared to the present patterns of *P. incana* invasion. High resolution images and a Digital Elevation Model (DEM) of the study area are used to derive the relationship between terrain parameters and *P. incana* distribution patterns. Long-term rainfall records are also examined for evidence of climatic fluctuations. A combination of climatic fluctuations and land use changes could have provided a niche for the invasion. Field surveys are made to assess the present land use patterns and groundtruth the degree of invasion by *P. incana*.

2. To establish the relationship between the spatial patterning of the vegetation invader species, soil surface conditions, soil moisture, runoff connectivity and the degree of visible erosion.

This objective is achieved by means of field surveys to characterise the structure of the invader species. Comparisons are made of the variations in soil moisture and soil surface strength under *P. incana* cover and adjacent grass patches. The implications of *P. incana* pattern for runoff flowpaths are deduced in the field. Observations of visible forms of erosion are also made.

3. To develop a conceptual model depicting the relationship between *P. incana* invasion and the variables that influence its distribution.

A model to depict the interrelationships between the different variables creates a conceptual understanding of the cause-effect scenarios. The model is developed after identifying and teasing out the interplay between the critical variables and processes. The graphic layout of the interaction pattern could be used as a tool for devising and designing appropriate rangeland rehabilitation strategies.

4. <u>To recommend the appropriate rangeland rehabilitation strategies based on the findings on the study</u>.

Having established the dynamics of *P. incana* spatial distribution in relation to the various physical and human factors, recommendations for the rehabilitation of the degraded rangeland are made.

The results of this study are presented in stand-alone Chapters 4, 5 and 6 (see Chapter outline below). The methods and techniques used to obtain the results are tied to the respective chapters. For purposes of coherence, it is deemed befitting that a description of the relevant methodology precedes the presentation of the results. By implication, descriptions of and motivations for the methods and techniques used are incorporated into the relevant results chapters.

1.4 ***Chapter outline***

1.4.1 Chapter 1: Introduction

In Chapter 1, the environmental problem investigated in this study is highlighted and the research problem and main research questions are raised. The aim and the specific objectives of the study are also presented. A summary of the methods by means of which each objective is achieved and an outline of the chapters of the thesis are provided.

1.4.2 Chapter 2: Vegetation invasions: geoecological perspectives and geomorphic implications

Chapter 2 presents viewpoints from an ecological perspective regarding vegetation invasions. The geomorphic implications of vegetation patchiness resulting from the replacement of the original continuous plant cover are reviewed. Instances where this conversion reflects either system resilience or dysfunctionality are discussed.

1.4.3 Chapter 3: The study area and Pteronia incana invasion

Chapter 3 describes the physical setting of the study area and a background to human-induced land disturbances and climatic variations *vis-à-vis P. incana* invasion is provided. The Chapter also provides details on a broad catchment survey conducted to assess the extent of *P. incana* invasion.

1.4.4 Chapter 4: The Use of High Resolution Imagery to Characterise Pteronia incana Distribution

In Chapter 4, procedures used to process and analyse Digital Camera High Resolution Imagery (HRI) are described. The degree of *P. incana* invasion is characterised by means of image classification. Different vegetation indices are compared in terms of their ability to separate *P. incana* from other surface cover types.

1.4.5 Chapter 5: The Influence of Terrain Parameters on Pteronia incana Occurrence

Chapter 5 presents the calculation of terrain parameters from a DEM of the area. Values files extracted from the parameters are compared with the samples of *P. incana* obtained from the HRI. The combined influence of terrain parameters and land use on *P. incana* invasion is investigated. A wetness index based on the TOPMODEL, which is topographically driven is also developed from the DEM and a relationship with *P. incana* occurrence is sought.

1.4.6 Chapter 6: Interaction Between Pteronia incana, Soil Surface Conditions, Runoff Potential and Erosion

Relationships between the pattern of *P. incana*, soil surface conditions and visible forms of soil erosion are explored in Chapter 6. Surrogates for moisture dependencies for grass species and *P. incana* are examined. Potential feedback mechanisms between the pattern of *P. incana* and water flux are investigated.

Chapter 7: Discussion, Development of Conceptual Model, Conclusions

Chapter 7 discusses the results presented in Chapters 4, 5, and 6 within the framework of the perspectives and implications reviewed in Chapter 2. A synthesis of the findings is provided in the form of a conceptual model. Recommendations regarding the rehabilitation of the invaded and degraded landscapes are made. Final conclusions are also arrived at.

Chapter 2: Vegetation Invasions: Geoecological Perspectives and Geomorphic Implications

2.1 Introduction

Vegetation changes that are manifest in the invasion and replacement of native vegetation, usually by inferior invader species, are a problem that has attracted the attention of many researchers, particularly ecologists. Amongst other things, the factors that influence the invasibility of a given environment and invasiveness of an invader species have been the focus of ecologists' investigations. As a result, a sub-discipline of invasion ecology has emerged. Vegetation change in the form of invasion has tremendous implications for geomorphic processes. A gap still exists in the understanding of the linkages between vegetation invasion and the geomorphic processes that underpin these changes. This could in part be attributed to the fact that geomorphologists and ecologists have worked independently in this area of research. An attempt is made in this chapter to assemble the views and findings of ecologists on vegetation invasions on the one hand and the insights of geomorphologists on the interaction between vegetation change and geomorphic processes.

This chapter is divided into two main sections. In the first section, geoecological perspectives on vegetation invasions are presented. Firstly, the debate on equilibrium and non-equilibrium concepts as they relate to vegetation change is unravelled. This is followed by an examination of the factors that influence invasibility and invasiveness. An overview of the vegetation invasion scenario in South Africa is also made. In the second section, a review of investigations by different scholars of vegetation - geomorphic responses is made. This would then provide a conceptual framework for investigating the interaction between *P. incana* and a range of variables as spelt out in the objectives of this study.

2.2 Geoecological perspectives on vegetation invasion:

2.2.1 Equilibrium / non-Equilibrium debate

An extensive debate has been ongoing for the past three decades as to whether vegetation change could be explained in terms of either the equilibrium or non-equilibrium models (cf. Noy-Meir, 1973; Walker and Noy-Meir, 1982; Wiens, 1984; Westoby *et al*, 1989; Scoones, 1993; Sullivan, 1999; Illius and O'Connor, 1999; Sullivan and Rohde, 2002). Systems models, which the equilibrium concept draws on, have dominated approaches to the understanding of vegetation change throughout the middle decades of the 20th century (Head, 2000). For all that period, the equilibrium thinking has been the axiomatic explanation for a linear and predictable sequence of vegetation stages. The equilibrium concept lays emphasis on notions of gradual, linear change, homeostatic regulation of systems, stability equilibria (McIntosh, 1985; Cherrett, 1989; Pimm, 1991). The concept stems from the Clementian theory of plant succession which is, supposedly, an orderly and directional process whereby a community of plant species replaces another. Any form of disturbance is seen as pushing the successional sequence back to some form of sub-climax (Behnke and Scoones, 1993) while the reverse allows the vegetation to proceed towards the climax once again (Hoffman and Ashwell, 2001).

In South Africa, rangeland management efforts and strategies over the past 50 years have been dominated and influenced by the theory of ecological succession (Hoffman, 1988; O’Connor and Bredenkamp, 1997; Hoffman and Ashwell, 2001), despite the highly diverse nature of the region dominated by semi-arid conditions (Allsopp, 1999). The predictable succession thinking is also quite evident in Acocks' (1975 and 1988) highly acclaimed publications where the 'would be' climax vegetation for each of South Africa's veld types is indicated.

"The changes in vegetation which are occurring today are an artificial reversal of the evolutionary replacement of the southern scrub vegetation by sward grasses of tropical origin" (Acocks, 1988, p4).

The pertinent questions that arise from this approach are:

- Is vegetation change such a deterministic process?
- Does the equilibrium concept as it relates to vegetation change apply in all places, all of the time?
- Can the equilibrium thinking explain vegetation change and/or invasion in semi-arid areas?

Empirical challenges to the assumptions of the equilibrium framework emerged in the mid 1970s through the recognition of dynamism, non-linearity and variability in space and time (Head, 2000). Truly, vegetation change cannot be explained within the framework of deterministic and predictable patterns of linear succession, particularly in semi-arid environments which are basically non-equilibrial. Given their susceptibility to extremely variable disturbances such as droughts and stochastic rainfall events, vegetation change in semi-arid environments can hardly occur in a predictable and linear manner (Hoffman *et al*, 1999; Hoffman and Ashwell 2001). Such flaws gave rise to a paradigm shift to the non-equilibrium approach.

The non-equilibrium thinking drew on empirical evidence of change, instability and dynamism in ecosystems. It is a conceptual approach which accepts that variable abiotic parameters are central to ecosystem behaviour, with a remarkable degree of qualitative persistence in the face of stochastic abiotic events (Sullivan 1998). It is now understood that change is the norm and disturbance is integral to ecosystems rather than an external process (Head 2000). On the basis of empirical evidence, different scholars (cf. Walker and Noy-Meir, 1982; Scoones, 1993; Sullivan, 1996, 1998, 1999; Turner, 1999; Sullivan and Rohde, 2002) have observed that the abiotic component is highly variable in semi-arid rangelands and ecological conditions are driven more by climatic events rather than by the interaction of the biotic elements. This view could hold true for vegetation invasion in the present study given the semi-arid nature of the study area. Vetter (2003) points out that the equilibrium and non-equilibrium concepts are extremes along a continuum. Non-equilibrium dynamics predominate in more arid systems, while more humid systems display equilibrium characteristics, and many systems encompass elements of both. There is a need to strike a balance between the two paradigms along the common ground.

2.2.2 Invasibility and invasiveness:

Events and actions that render a given environment susceptible to invasion by non-native species have been examined by different scholars (e.g. Lonsdale, 1999; Davis *et al*, 2000; Buckland *et al*, 2001). Despite the research effort, the factors that influence the invasibility of environments and the invasiveness of certain plant species have proven elusive for ecologists (Davis and Pelsor 2001). As to why some biomes and regions seem more susceptible to invasions is also a long way from being clarified (Lavorel *et al*, 1999; Lonsdale, 1999; Stohlgren *et al*, 1999). Defined as the degree to which a community is susceptible to the establishment of external species (Lavorel *et al*, 1999), invasibility is the outcome of several factors. These range from changes in disturbance regimes (Crawley, 1987; Alpert *et al*, 2000; Prieur-Richard *et al*, 2000), climatic fluctuations in a given region (Lonsdale, 1999; Sax and Brown, 2000), fluctuating resources (Davis *et al*, 2000; Davis and Pelsor, 2001; Kolb *et al*, 2002) to species diversity (Palmer and Maurer, 1997; Lavorel, 1999; Lavorel *et al*, 1999; Prieur- Richard and Lavorel, 2000).

Invasiveness, which refers to the capacity of a species to successfully invade communities where it was previously absent (Prieur-Richard and Lavorel, 2000), is also influenced by various factors. These include: the traits of the invader, for example, low palatability or unpalatability (Buckland *et al*, 2001), propagule pressure (Davis and Pelsor, 2001), tolerances (Rejmanek, 1996; Sax and Brown, 2000), predation, mutualism and competition (D'Antonio *et al*, 1999; Richardson *et al*, 2000). However, it is deemed more practical at this stage to examine the characteristics of environments susceptible to invasion rather than those of the individual invasive species themselves. As pointed out by Prieur-Richard and Lavorel (2000), invasibility is a generic characteristic that does not discriminate *a priori* between different types of potential invaders. Besides, the difficulty of identifying the functional characteristics of invading species has been insinuated by Crawley *et al* (1996), Williamson and Fitter (1996) and Williamson (1996). Therefore, a description of the characteristics of the invader species investigated in this study in Chapter 3, section 3.3 will suffice.

2.2.2.1 Disturbance

A distinction is made here between human-induced disturbances and climatic fluctuations.

Human-induced disturbance

Invasion ecology scholars are unanimous that disturbances favour plant invasions. Empirical evidence from a number of experimental studies attests to the role of disturbed sites in facilitating invasion. According to Sax and Brown (2000), invading species are particularly successful in disturbed habitats, especially those altered by human activities. In the case of abandoned fields, the amount of bare ground created by soil disturbances has been shown to directly control the abundance of invading species (Burke and Grime 1996). In an experimental design on abandoned agricultural fields in Mediterranean France, Lavorel (1999) noted that species with early germination were intolerant of disturbance. It was also noted that disturbance favoured species with higher fecundity than in undisturbed vegetation, implying that the capacity to build a large seed bank is essential for colonisation of disturbances. Experimental manipulations by Buckland *et al* (2001) and Thompson *et al* (2001) in fertility-disturbance matrices demonstrated that the distribution of invaders was mainly dependent on disturbance, which entailed the creation of bare ground, and debilitation of the pre-existing vegetation. Hobbs and Atkins (1988) cited by Davis *et al* (2000) found that disturbance combined with eutrophication increased a community's invasibility. The combination entailed a reduction in resource uptake by resident vegetation.

Grazing is referred to by Landsberg *et al* (1999), Prieur-Richard and Lavorel (2000) as a complex disturbance which favours exotic invasions. Floristic changes induced by grazing were identified by Moretto and Distel (1999) as involving the replacement of palatable grasses by unpalatable ones. Selective defoliation of palatable species was noted in their study as conferring a competitive advantage to unpalatable species, ultimately leading to replacement in grasslands. According to Hobbs and Huenneke (1992), cited by Prieur-Richard and Lavorel (2000), an interaction between several disturbance types may promote the highest rates of invasion. As observed by Davis *et al* (2000), species invasions are one of the main ecological consequences of global changes in climate and land use.

Climatic fluctuations

Random temporal fluctuations in the extrinsic environment, including the effects of such things as severe storms and drought, can provide a niche for invaders to establish themselves (Sax and Brown 2000). The occurrence of such stochastic, shortlived events could reduce competition and increase invasibility (Davis and Pelsor 2001). These views are supported by studies by Buckland *et al* (2001) and White *et al* (2001) which confirmed that repeated disturbance from successive drought created extreme climatic events that suppressed the competitive dominance of the resident vegetation, promoting invasion. In the former study, a severe drought in 1995 created and sustained bare ground in plots, permitting invasion by *Thaliana* annual species with a persistent seed bank. As pointed out by Davis *et al* (2000), drought severe enough to cause mortality and create gaps in previously closed vegetation may increase invasibility either during drought itself or when it is over. The variation in soil moisture as a resource comes out clearly from the experimental studies alluded to above as the main underpinning in climatic fluctuation related invasions. The role of resource fluctuations in plant invasions is explored further below.

2.2.2.2 Resource fluctuation

The fluctuating 'resource availability theory' has been proposed by different scholars (cf. Burke and Grime, 1996; Davis *et al*, 1999; 2000; Stohlgren *et al*, 1999; Buckland *et al*, 2001; Davis and Pelsor, 2001) as the key factor controlling invasibility. According to Davis *et al* (2000), if an environment experiences a pulse of resource supply or a decline in resource uptake, an invading species may be able to exploit the unused resource, even though its ecology is not fundamentally different from that of the resident species. The fluctuating 'resource availability theory' is illustrated by Davis *et al* (2000) in Figure 2.1 below.

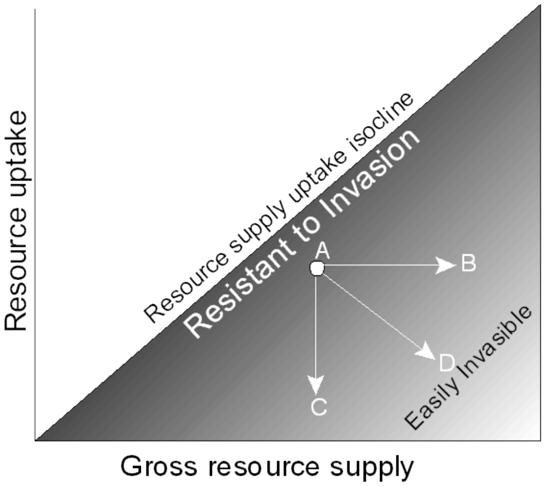


Figure 2.1: Theory of fluctuating resource availability (after Davis et al 2000).

According to Figure 2.1, resource availability could increase due to a pulse in resource supply (A B), a decline in resource uptake (A C) or both (A D). As the trajectory moves further right and or below the supply uptake isocline, resource availability and hence invasibility increases. They point out that meteorological fluctuations and site-specific events, such as large or small-scale disturbances, could cause resource fluctuations. In the same vein, MacGregor and O'Connor (2002) noted that during a severe drought, a shift in water availability beyond the threshold of stress of *C. mopane* species led to its dieback and replacement.

In a field experiment, Davis and Pelsor (2001) confirmed that fluctuations in resource availability of as short as a few weeks up to one year had a large impact on plant invasion success. Water and disturbance were identified as the main treatments that affected resource availability. The results of the study indicated that disturbance increased invasibility when resource availability was low; that is when plots were dry. This demonstrates the interaction between soil water and disturbance such that the latter eliminated resource uptake by resident vegetation, thus promoting invasion. The resource availability theory is corroborated by Thompson *et al* (2001) who confirmed that invasions are promoted by a fluctuation in the availability of resources either through addition of extra resources or a reduction of their use by the resident vegetation. Disturbances that may reduce the rate of resource capture by resident vegetation were also identified by Davis *et al* (2000) as promoting invasibility.

Findings from a study by Kolb *et al* (2002) supported the hypothesis that microhabitat types that differ in invasibility also differ in resource availability. A positive relationship between high nitrogen and high invasibility was identified. So was the relationship between water availability and invasibility. The reverse was true of acidity and invasion.

The ability of the 'resource availability theory' to integrate resource availability, disturbance and fluctuating environmental conditions can be deduced from this sub-section. As observed by Davis and Pelsor (2001), the theory may serve as a unifying concept that can integrate earlier ideas regarding invasibility. Successful invasion depends more on the spatial and temporal coincidence of resources (Buckland *et al* 2001) which, as has been noted, are impacted upon by among other things, human disturbance and climatic fluctuations.

2.2.2.3 Species diversity

Elton (1958), considered to be the father of invasion ecology, advanced species richness as one of the factors influencing invasibility. Subsequent scholars (cf. Palmer and Maurer, 1997; Chapin *et al*, 1997; Lonsdale, 1999; Naeem *et al*, 2000; Prieur-Richard *et al*, 2000) found a positive association between species richness and invasion. Lavorel (1999) points out that biological diversity plays an important role in the capacity of ecological systems to respond to changes related to climate, atmospheric and land use changes. The loss of biodiversity may lead to the loss of resistance to invasion by species external to the system. On the contrary, Rochette (1997), cited by Lavorel (1999) noted that the existence of diversity in Mediterranean lands is the key to rapid recolonisation of disturbances, as species capable of filling out disturbance gaps are readily present in the community. Such controversy highlights the uncertainty surrounding the role of biodiversity in vegetation invasions.

Besides the controversy above, conflicting empirical evidence has emerged regarding invasibility and community diversity (cf. McCloskey *et al*, 1996; Tilman, 1997; Lavorel *et al*, 1999). In contradiction to Lavorel’s (1999) work in the Mediterranean lands, Lavorel *et al* (1999) noted that there was no strong relationship between species richness and invasibility. They concede that diversity–

invasibility relationships are more subtle than studies thus far have acknowledged. As pointed out by Davis *et al* (2000), there is no reason to expect any consistent pattern between the species richness or diversity of a community and its susceptibility to invasion. They suggest that the positive association between species richness and invasion found by scholars like Lonsdale (1999) may arise from a tendency of diverse plant communities to be nutrient-poor. Stohlgren *et al* (1999) attribute the discrepancy in part to scale, given that opposite correlation patterns have been observed when working from local to regional scale. Burke and Grime (1996) suggest that future studies of diversity-invasibility should consider the confounding effects of fertility and disturbance. They noted in their study that resistance to invasion was found in communities with the highest number of species and moderate levels of disturbance.

All in all, it can be said with certainty that, other than resource availability which has been seen to integrate all factors, no factor can be singled out as solely determining invasibility. Nonetheless, the above discussion throws light on the role of individual factors in invasibility. As pointed out by Richardson *et al* (2000), the factors discussed above interact with each other and with the characteristics of individual species to promote invasibility.

A major shortcoming that can be gathered from the invasion ecology investigations reviewed above is the total disregard for the co-parameters of terrain and their implications for soil properties, geomorphic processes and, ultimately, the invasibility of a given environment. As pointed out in section 2.1 (Introduction) of this chapter, the lack of joint investigations between geomorphologists and ecologists engenders such critical omissions. There is a need to gain insights into the implications of geomorphic processes for invasibility. This would then provide an indepth understanding of the underpinnings of invasibility and the interaction between invader vegetation species and geomorphic processes. This interaction is reviewed in section 2.3. A brief review of the vegetation invasion scenario in South Africa is made below.

2.2.3 South African vegetation invasion scenario: an overview

A two-dimensional vegetation invasion scenario prevails in South Africa *viz*.: Alien species - from outside the country and indigenous species that invade biomes they do not belong to. All biomes in South Africa have been invaded to some degree by alien vegetation species (cf. Henderson and Wells, 1986; Macdonald and Richardson, 1986; Richardson *et al*, 1997; Versfeld *et al*, 1998; Hoffman *et al*, 1999; Hoffman and Ashwell, 2001). A recent national survey of alien plant invasions (Versfeld *et al*, 1998) indicates that 8.27% of South Africa is infested by a wide ranging alien woody vegetation species. In the semi-arid areas, invader shrubs and trees are reported by Versfeld *et al* (1998) and Hoffman *et al* (1999) as confined to riparian zones, particularly the Eastern Cape catchments of the coastal region from Kariega to Gamtoos where *Hakea* and *Acacia* species are documented as the dominant invaders. The main impacts of these species are manifest in their excessive use of surface water resources and hence reduction of stream flow, disruption of ecosystem processes and structure, and a threat to diversity (cf. Le Maitre *et al*, 1996; Richardson *et al*, 1997; Scott *et al*, 1998; Versfeld *et al*, 1998; Hoffman *et al*, 1999).

The invasion of terrestrial environments by indigenous species that are non-resident to certain biomes is another dimension of the vegetation invasion scenario. The grassland and savanna biomes in particular are reported by Henderson and Wells (1986) as invaded by shrubs indigenous to the Karoo, and unpalatable tussocky grass species. Wells *et al* (1983), cited by Henderson and Wells (1986), identifies three main categories of grassland invasion by indigenous species namely: the lower, moister grassland margins invaded by the adjacent savanna communities; the higher and wetter ridges invaded by unpalatable tussocky and poisonous grasses; and the higher and drier grassland margins invaded by xerophytic ‘karroid’ shrubs. The latter category, dubbed ‘the theory of the expanding Karoo’, has been questioned by different scholars (cf. Hoffman and Cowling, 1990; Behnke *et al*, 1993; Scoones, 1996; Parsons *et al*, 1997, Sullivan and Rohde, 2002). The concern that the grass sward was fast being irreversibly replaced by Karoo bushes is expressed in accounts by Acocks (1953 and 1979). Descriptions of the expansion of the unpalatable karroid shrubs into grassland by Bosch (1989), O'Connor (1995) and O'Connor and Roux (1995) followed suit. The conversion of landscapes in the invaded areas to arid Karoo conditions is the ultimate result of this expansion. Scholars disputing the 'expanding Karoo theory' argue that data to provide unequivocal support for it is lacking and that fluctuations in shrub and grass cover are a response to rainfall variability. The danger of extrapolating findings from small-scale studies to larger temporal and spatial scales is also highlighted (cf. Sullivan, 1999, Sullivan and Rhode, 2002). Notwithstanding this debate, it is quite clear that vast areas of the catchment that comprises the present study area, and many other catchments in the same district, are steadily being invaded by the karroid *P. incana*, with no indication of rainfall variability related fluctuations in the invasion rate.

2.3 The interaction between vegetation change and geomorphic processes

2.3.1 Introduction

In this section, perspectives on the interaction between vegetation replacement, the resultant patterning and geomorphic processes are examined. The influence of terrain parameters and the implications for soil surface conditions and moisture variations in this interaction are also explored. The focus of this review will be on arid and semi-arid areas, given that the present study is carried out in a semi-arid environment. In the overview of the equilibrium / non-equilibrium debate (sub-section 2.2.1), it was noted that the latter mechanism could explain vegetation change in semi-arid environments. Stochastic events which are integral to this thinking are a prominent feature of these environments. Geomorphic processes triggered by, among other things, disturbances and stochastic climatic events may promote vegetation change or invasion. By the same token, as observed by Sax and Brown (2000), when invader species become dominant in a given environment, they may lead to changes in ecosystem as well as geomorphic processes.

2.3.2 Vegetation - bare soil surface mosaics

The replacement of resident vegetation by invader or other species in arid and semi-arid, usually degraded environments may give rise to a mosaic of vegetation and bare soil surfaces which reflect the interplay between the species and geomorphic processes. The modification of soil surface conditions and water reallocation on and in the soil have, amongst other processes, been the focus of study by many a scholar (cf. Tongway and Hindley, 1995; Bergkamp *et al*, 1996; Bryan and Brun, 1999; Macdonald *et al*, 1999; Cammeraat and Imeson, 1999; and Valentin *et al*, 1999). The interplay is of great significance in explaining water redistribution on the soil surface, soil surface conditions and soil erosion patterns. Cammeraat and Imeson (1999) point out that at the hillslope or basin scale, the spatial patterning of vegetation determines the connectivity and concentration of overland flow. They conclude that vegetation spatial patterning can be used as an expression of soil properties important for water reallocation in and over the soil. According to Holm (2000), a change in vegetation structure which entails an increase in the 'fetch' could engender a degraded and dysfunctional landscape where water and nutrients are lost from the system. The development of patchy vegetation bands with alternating bare soil areas could be interpreted as an indication of the initial stages of recovery of a perturbed system (Bryan and Brun, 1999). Particularly intriguing are the feedback mechanisms that result from the interaction between vegetation, soil surface conditions, runoff and soil erosion. The interaction entails the redistribution of water on slopes inducing runoff run-on systems which are closely related to the vegetation patterns (Bergkamp *et al* 1999, Valentin *et al* 1999).

Vegetation patchiness could emerge when the original and often perennial plant cover is replaced. In arid and semi-arid areas, landscapes organised in a two-phase mosaic characterised by vegetated and bare components may develop (Valentin *et al*, 1999). In typically arid areas, the patterns vary from broad alternating vegetation bands and bare ground, to spotted or fuzzy patterns. Banded vegetation patterns are described in West Africa by Valentin *et al* (1999), Valentin and d’Herbès (1999), the Mediterranean by Bergkamp *et al* (1999), Cammeraat and Imeson (1999) and Australia by Ludwig *et al* (1999), Dunkerley and Brown (1999). In the semi-arid and more mesic regions, replacement of grass cover by shrubs characterised by considerable intershrub spaces may develop. This could gradually give rise to an alternating mosaic of shrubs and bare soil patches. Shrubs may also develop in parallel strips, a patterning that could have resulted from the inheritance of parallel contour ploughing micro-topography (Cammeraat and Imeson, 1999). The last two patterns are the forms *P. incana* (Blue bush); the shrub investigated in this study assumes. Not withstanding the differences in configuration between *P. incana*-bare patch patterns and banded vegetation in the more arid lands, vivid parallels can be drawn between the two with regard to their interaction with soil surface conditions, surface water redistribution on hillslopes, and erosion occurrence.

The origins and development of vegetation and bare soil surface mosaics remain subject to a number of hypotheses. Valentin *et al* (1999) suggest two main hypotheses to explain the origins and development of vegetation and bare soil surface mosaics in arid areas namely: the gradual degradation of an originally uniform plant cover due to climatic or human disturbances, and the colonisation of previously degraded bare areas under improving climatic or land use conditions. These hypotheses are supported by Dunkerley and Brown (1995), Cerdà (1997), Bryan and Brun (1999) and Leprun (1999) who point to rangeland deterioration resulting from the disruption of a formerly more continuous vegetation cover by overgrazing, trampling and precipitation decline. Abandoned fields in semi-arid SE Spain are reported by Cammeraat and Imeson (1999) as being invaded by *Stipa tenacissima* vegetation patterns under conditions of drought and grazing pressure. Runoff is then generated by the exposed and crusted soil with reduced infiltration capacity, promoting erosion. Leprun (1999) gives a more vivid hypothesis after his assessment of the formation of tiger bush patterns in West Africa:

"*A long drought period, as the Sahel suffered in the 1970s, may have destroyed most of the vegetation. On the sandy loam substrata, originated from more or less iron-crusted sedimentary or schistose rocks, with a gentle slope due to the monoclinal structure oriented NE-SW, thin soils are prone to erosion. Water erosion affects the bare crusted slopes and sorts out the eroded particles into textural fractions. ………………………… A part of the finer particles clogs the pores, seals the surface, fostering the subsequent colonisation by vegetation*" (Leprun, 1999, p41-42).

The hypotheses outlined above do highlight the prominent role of stochastic events and anthropogenic factors in the genesis of bare soil-vegetation mosaics. The presence of semi-natural and usually moisture-stressed vegetation is another underlying assumption. It is noteworthy though, that environments do exist where vegetation and bare soil surface mosaics are natural. Hence, temporal and spatial insights must be gained in order to identify the precise processes and controls involved.

The focus of this section is, however, not the description of the different vegetation patterns *per se*. In the main, it is meant to review the factors that influence the distribution of these mosaics and their interaction with water redistribution on and in the soil, soil surface conditions and soil erosion. These aspects are examined in the sub-sections that follow.

2.3.3 Factors and processes influencing vegetation-bare zone development

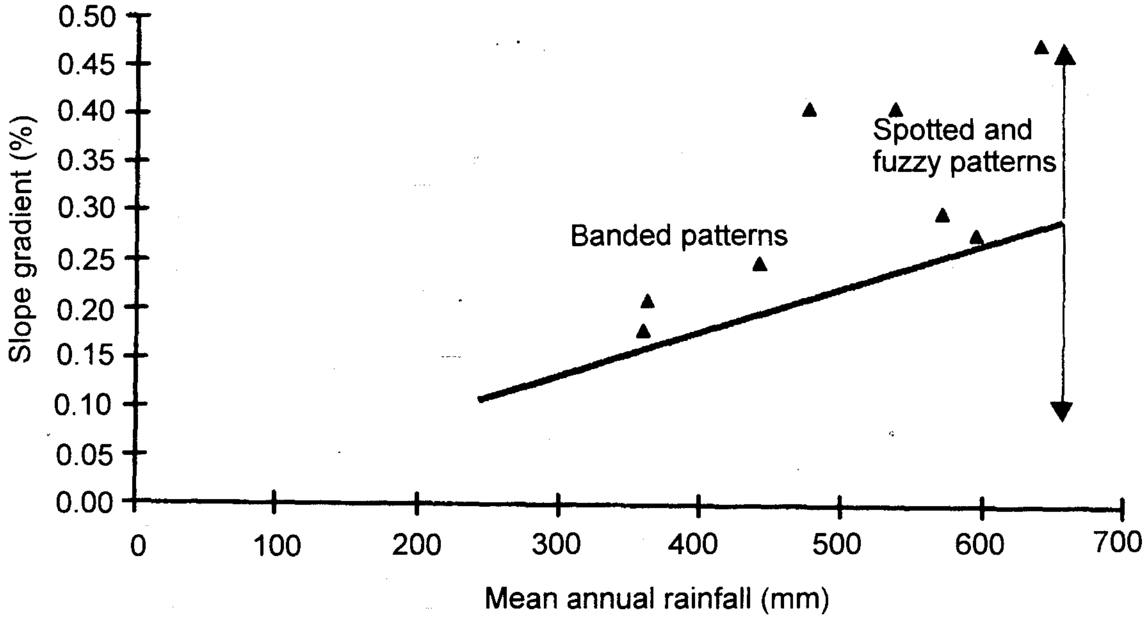
According to Cammeraat and Imeson (1999), vegetation structure ranging from polygon-like *Stipa tenacissima* structure in the dry Mediterranean region towards a more spotted and banded vegetation structure is influenced by a number of factors. These include local factors such as gradient and aspect, soil roughness (rock fragment cover), soil surface conditions (crusting and infiltration capacity), sediment delivery, and cover percentages of vegetation. More regional factors include climatological conditions as well as geological properties. These factors in turn influence the redistribution of major resources, notably soil moisture. This being a catchment based study, regional factors like geology are outside the scope of this investigation. The role of climate has been discussed in the section on geoecological perspectives. The influence of local factors namely; topographic parameters, soil surface conditions and soil moisture variations are examined below.

2.3.3.1 Slope gradient

Slope gradient is singled out by different authors (Dunkerley and Brown, 1995; Cerdà, 1997; Dunkerley and Brown, 1999; Eddy *et al*, 1999; and Zonneveld, 1999) as the fundamental variable influencing development of vegetation-bare zone patterns and water redistribution. The development of, for instance banded vegetation, vegetation arcs, and vegetation stripes in arid and semi-arid lands favour gentle to very gentle gradients (cf. Bromley *et al*, 1997; Eddy, *et al*, 1999; Bryan and Brun, 1999 and Janeau *et al* 1999). On steeper gradients, banded vegetation patterns give way to the elongated patch form (Dunkerley and Brown, 1995). It is only a few exceptions where banded vegetation patterns have been observed on slopes as steep as $10\text{-}30^0$ (Puigdefábregas and Sánchez, 1996) and $6\text{-}20^0$ (Bergkamp *et al*, 1999). According to Valentin *et al* (1999), slope gradient is the controlling factor for the occurrence of banded vegetation patterns. A slope gradient threshold of 11^0, below which the patterns cease to be banded but became spotted, was identified in Niger, West Africa. The same authors point out that there is a tendency for the slope gradient threshold to increase with mean annual rainfall (MAR) such that for a given slope gradient, vegetation pattern is likely to be banded under dry than under wetter conditions (See Figure 2.2 below). Their observations along a 200 km transect in West Africa indicated that areas receiving less than 550mm MAR were covered with banded vegetation while wetter zones had less contrasted patchy vegetation patterns.

Figure 2.2: Impact of MAR on the critical slope gradient value for the occurrence of vegetation patterns (after Valentin *et al*, 1999).

The growth of *Stipa tenacissima* patchy vegetation patterns in SE Spain was noted by Cerdà (1997) and Cammeraat and Imeson (1999) to be strongly influenced by slope gradient. Whereas the steep upper and middle hillslopes were noted as covered by open *Stipa tenacissima* tussocks, a decrease in their density with lower slopes was observed (Cammeraat and Imeson, 1999). The implications of slope gradient for vegetation pattern development are manifest in its interaction with water redistribution on the slope surface which takes the form of runoff run-on patterns. Slope angle, as the below which the patterns cease to be banded but became spotted, was identified in Niger, West Africa. The same authors point out that there is a tendency for the slope gradient threshold to increase with mean annual rainfall (MAR) such that for a given slope gradient, vegetation pattern is likely to be banded under dry than under wetter conditions (See Figure 2.2 below). Their observations along a 200 km transect in West Africa indicated that areas receiving less than 550mm MAR were covered with banded vegetation while wetter zones had less contrasted patchy vegetation patterns.



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2.3.3.2 Slope aspect

The influence of slope aspect on the development of vegetation patterns is subtler than that of the other terrain parameters. A catchment or regional scale study should unmask unequivocal links between slope aspect and vegetation patterns. Varying deductions have been made regarding the relationship between the two phenomena. A study by Leprun (1999) of the dynamics of tiger and dotted bush patterns in West Africa identified a NE-SW orientation of the tiger bush. This orientation happens to be the direction of the dominant wind (harmattan). Petrography was also noted as the limiting factor due to a gently sloping monoclinal lithology oriented NE-SW, giving the bedrock a low and constant slope of <5%, conducive to the development of tiger bush patterns. On the contrary, in a similar study of tiger bush patterns in Sokoto, northern Nigeria, Zonneveld (1999) recognised a NNW-SSE orientation of vegetation patterns, perpendicular to the prevailing ENE-WSW wind direction. He concluded that the orientation of the vegetation patterns irrespective of compass direction is determined by the terrain of slope and is perpendicular to the water flow, not the wind.

Studies by Puigdefábregas and Sánchez (1996) of *Stipa tenacissima* patchy vegetation species in SE Spain unravelled wide-ranging inclinations of the species to slope aspect. Increasing trends of plant cover with catchment area, wetness index and the slope length factor were noted to be more pronounced on the south-facing slopes than on the north-facing ones. The authors attribute these trends to water availability as the main control of plant cover, with topographic trends better expressed on the south-facing slopes where topography related local differences in soil moisture are expected to be more manifest. The shape of *S. tenacissima* tussocks in terms of the ratio between maximum and minimum diameters was also noted to show consistent variation trends on the south-facing hillslopes but not the north-facing ones. The development of *S. tenacissima* stripes parallel to contours was noted to show no relation with topography on the north-facing hillslopes. The reverse is true of the south-facing slopes where positive trends with local slope angle, wetness index and slope length factor were observed.

The bearing slope aspect has on resource availability, particularly soil moisture and hence plant productivity, should determine the degree and pattern of vegetation patchiness. As noted by Qiu *et al* (2001), slope aspect is one of the dominant controls on the spatial variability of soil moisture besides relative elevation, hillslope position and slope gradient. The direct role of soil moisture variations in the development of vegetation patchiness is examined fully in sub-section 2.3.3.4.

2.3.3.3 Soil surface conditions

The modification of soil surface conditions is central to vegetation patchiness. According to Dunkerley and Brown (1999), soil surface roughness and spatial variations in soil strength and bulk density are associated with microtopographic elements that compose vegetation mosaics. Surface roughness in particular promotes the opportunity time for infiltration, acting to retard flow and hence restricting scour. Soil surface crusting is one soil surface condition that is of particular relevance to the present study. The tendency to develop crusted surfaces is a phenomenon that has been noted in most vegetation-bare zone studies (cf. Dunkerley and Brown, 1995; Bergkamp *et al*, 1996; Bromley *et al* 1997; Cerdà, 1997; Bryan and Brun, 1999; Cammeraat and Imeson, 1999; Eddy et al, 1999; Galle *et al*, 1999; Malam Issa *et al*, 1999; Janeau *et al*, 1999; Valentin and d'Herbès, 1999; Valentin *et al*, 1999). It is usually the inter-shrub or inter-vegetated patch zones that are characteristically crusted. Romkens *et al* (1990) distinguish surface sealing and crusting, the former being the initial or wetting phase in crust formation and the latter as the hardening of the surface seal in the subsequent dry phase. It is deemed useful to provide an overview of the process of soil crusting and characterise soils that are susceptible to crusting as a background to gaining valuable insights into its implications for the development of vegetation patchiness.

Biological and physical crusts are identified by Malam Issa *et al* (1999) as the main types of soil crusts in arid and semi-arid areas. Biological or microbiotic crusts result from the development of communities of micro-organisms, particularly cyanobacteria. Their prominent role in the stabilisation and preservation of the soil surface against water and wind erosion has been highlighted (cf. Malam Issa *et al* 2001). Physical crusts on the other hand are formed mainly by the action of water and to some extent, wind. A typology of physical crusts which ranges from structural, erosion and deposition crusts is proposed by Valentin and Bresson (1992 and 1998). Raindrop impact plays an important role in the development of infilling, coalescing and sieving crusts which are all forms of structural crusts as designated by Valentin and Blesson (1992).

Slaking crusts, which are also a form of structural crusts, result from the physico-chemical dispersion of sodic soils. Such crusts predominate when soil is dry before rainfall (Le Bissonnais *et al*, 1989 cited by Valentin and Blesson, 1998), particularly due to age-long hardening after the disturbance of particle-to-particle bonds (Gerits *et al* 1990). Age-long hardening is influenced mainly by soil water content, clay content and time. It is accomplished by mechanisms like particle re-arrangement and cementation. The latter involves the formation and strengthening of structural bonds between structural units of primary particles. Cementing agents in the soil-carbonates and silica diffuse towards the region of particle to particle contact where they precipitate after saturation due to drying of the soil (Gerits *et al*, 1990). According to Romkens *et al* (1990) Exchangeable Sodium Percentage (ESP) or sodicity of the soil, electrolyte concentration in water and the presence of swelling clay (montmorillonite) may promote chemical dispersion of soil particles. A critical ESP of 15 (Mullins, 1998) or Sodium Adsorption Ratio of 13 (Miller and Donahue, 1995) is taken to define the boundary between sodic and non-sodic soils. Van Rooyen (2000) points out that most soils with a high clay content harden easily or become capped during degradation with the result of reduced infiltration.

Besides structural crusts, erosion crusts which also fall under the physical crust category, are of particular relevance to the present study. They consist of only one thin plasmic layer, 100μm to 1mm thick, and form when the loose coarse-textured upper layer is stripped away by overland flow, exposing the underlying clayey layer outcrops (Valentin, 1991; Valentin and Bresson, 1998). The exposed clayey layer is highly susceptible to crusting (Boardman, *pers.comm.*, 2000). Figure 2.3

below is a schematic illustration of the time sequence of soil crust formation.

The significance of soil crusts in the development of vegetation patchiness lies in runoff generation, reduced infiltrability, and inhibiting seedling emergence (Valentin and Bresson 1998). Erosion crusts promote runoff, which is reflected in the occurrence of runoff crusts (Valentin *et al*, 1999). The runoff thus generated may either sink in the adjacent downslope vegetated patch or coalesce with other runoff trajectories to form larger pathways. Thiéry *et al* (1995) described the runoff potential of the tiger bush zone of the Sahelian Niger as a function of its surface crusts. Bare zones covered by erosion and gravel crusts were noted as generating large amounts of runoff - more than half of the annual rain becomes runoff (Galle *et al*, 1999). Valentin and d'Herbés (1999) prove that a spatial relationship exists between soil crusts and soil moisture. They point out that given the role of crusts in hampering infiltration, a field survey of surface crusts appears to be a useful tool for predicting the spatial variation in soil moisture. This relationship will be explored further in the subsequent sub-section. The intriguing questions, however, that need to be unravelled in regard to crust formation are:

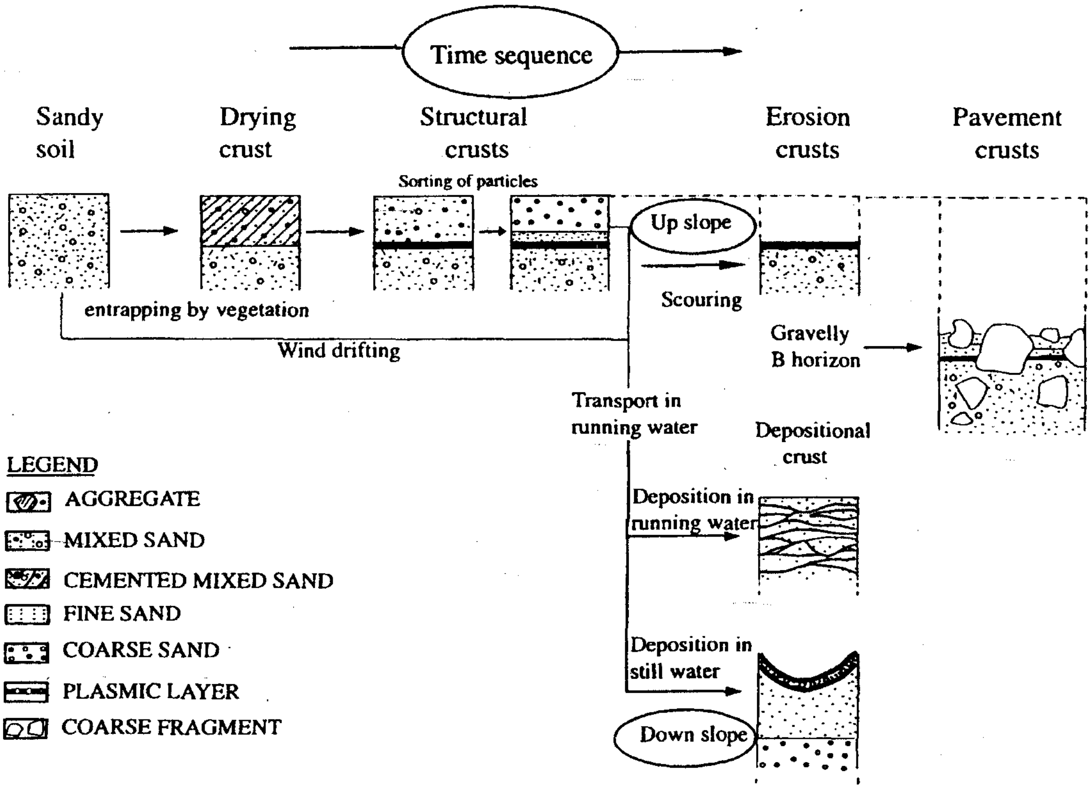
- Does crust formation predate the development of vegetation-bare surface mosaics? or
- Do vegetation-bare soil mosaics promote soil crust formation?

The processes of crust formation outlined above provide answers to these questions. As noted, disturbances and the attendant physico-chemical dispersion precede age-long hardening. Crusting therefore precedes the development of vegetation-bare zone mosaics in prior-to-disturbed environments. It is also true that the mosaics do promote soil crust formation as crusted zones are inherent to vegetation-bare zone patterns (Valentin *pers. comm*., 2003).

Figure 2.3: Schematic representation of the time sequence of soil crust formation (after Malam Issa et al, 1999).

2.3.3.4 Soil moisture variations

The relationship between soil moisture and vegetation patchiness is showcased by different studies (cf. Thiéry *et al*, 1995; Seghieri *et al*, 1997; Bromley *et al*, 1997; Galle *et al*, 1999; Macdonald *et al*, 1999; Valentin and d'Herbès, 1999). Vegetated patches in certain mosaics have been identified as water harvesting systems, giving rise to high soil moisture levels under them. Galle *et al* (1999) monitored soil moisture variations in four discrete zones of the Tiger bush patterns in Niger: the senescence zone (the downslope border of a vegetated patch), bare area, upslope border of the patch and the core of the patch. They found a progressive decrease in soil moisture from the vegetated patch, the upslope border, the senescence zone to the bare area at different dates (See Figure 2.4 below). The four zones are perceived as a unit that is an expression of water storage capacities induced by runoff redistribution, with three zones generating runoff and the vegetated patch as a run-on sink. Surface wash is generated on the bare zone, accumulates at the upslope edge and infiltrates in the vegetated patch where the resulting soil moisture promotes plant colonisation (Chappell *et al*,



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The senescence zone displays a decrease in tiger bush vigour because of the reduction in soil moisture and nutrients resulting from the capture of moisture and nutrients by the core of the vegetated patch (Thiéry *et al*, 1995). A study by Macdonald *et al* (1999) of the chenopod patterned shrubland in western New South Wales, Australia, demonstrates further the runoff redistribution dynamics on patched vegetation and their implications for soil moisture variations. They conclude that: "The v*egetation pattern of the chenopod shrubland is chiefly due to the difference in soil moisture which arises largely through the runoff of rainwater from bare, impermeable gilgai mound to the vegetated patch*" (Macdonald *et al* 1999, p102). Soil moisture variations in vegetated-bare zone mosaics are thus seen as a function of the water harvesting efficiency by vegetated patches, which is dependent on the type of patchiness (Ludwig *et al* 1999).

Figure 2.4: Variations in soil moisture across the four zones composing the Tiger bush patch (after Galle et al, 1999).

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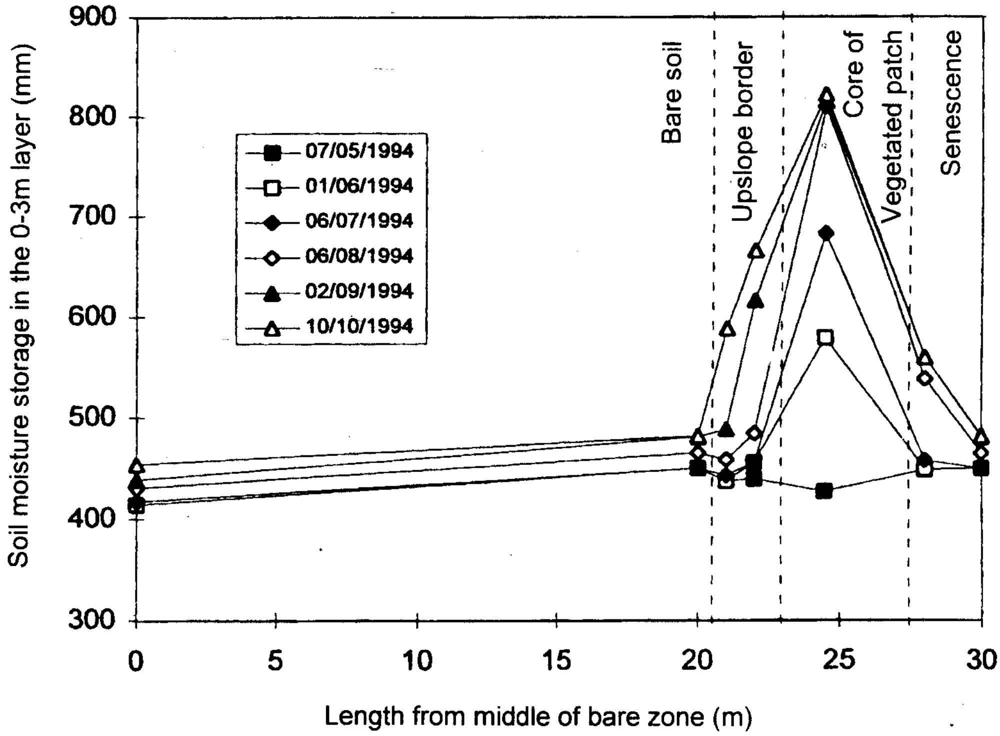


Figure 2.4: Variations in soil moisture across the four zones composing the Tiger bush patch (after Galle et al, 1999).

It is noteworthy, however, that heterogeneity associated with the local landscape has strong implications for soil moisture variability. Concavities in the local topography may force runoff to accumulate and to preferably infiltrate in hollows on hillslopes (Valentin and d'Herbès, 1999). The configuration of slope elements determines the pattern of divergence and convergence of runoff and shallow subsurface flow, so that saturation occurs preferentially in hollows, thereby localising the origin of surface drainage (Church and Woo, 1990). It is therefore important to unravel the implications of slope configuration induced soil moisture variability for patterns of vegetation patches. According to Anderson and Burt (1990), three zones may be identified where runoff accumulation and hence saturation is likely to occur: in hillslope hollows, since flowlines converge into them; at the base of any slope, since drainage area increases downslope; and in areas of reduced soil moisture storage. In the topography, zones of saturation develop typically in converging segments of the landscape where the ratio of the upslope contributing area to the length of the contour (A_I / W_I) is large. Such conditions are encountered along drainage lines and zones of water concentration in the landscape (Moore and Foster, 1990).

The realisation of the prominent role topography plays in soil moisture distribution has prompted the development and application of topographically driven models for the prediction of moisture variations in the landscape, particularly on hillslopes (cf. Anderson and Burt, 1990; Hutchinson and Dowling, 1993; Quinn *et al*, 1993; Quinn *et al*, 1995; Saulnier *et al*, 1997; Beven, 1997; Kirkby, 1997). Topographic indices are often derived from Digital Terrain Models (DTMs) to estimate the extent of zones of soil moisture concentration and surface soil saturation within the catchment. Consequently, topographic analysis has increasingly become useful as a means of predicting soil moisture variability and surface saturation zones (Anderson and Burt 1990).

The TOPMODEL and wetness index

The TOPMODEL parametrically relies on the preprocessing of digital terrain data to calculate the catchment distribution function of the topographic index $\ln(A_s / \tan\beta)$ (Quinn *et al*, 1993), where:

A_s = specific catchment area, and

β = local slope gradient

The local slope gradient (β) is a measure of the hydraulic gradient, which is the driving force behind water movement in a catchment. The specific catchment area (A_s) is defined by the local upslope contributing area to any point in a catchment. The TOPMODEL relates hydrological behaviour to the topographically derived variable $\ln(A_s / tan\beta)$. It predicts saturation excess and infiltration excess surface runoff and subsurface flow (Beven, 1997). It can also predict the expansion and contraction of variable source areas, with the areas having a high value of $\ln(A_s / tan\beta)$ saturating first (Anderson and Burt 1990). The Topographic Wetness Index (WI) as a derivation of the TOPMODEL is expressed as $\ln(A_s / tan\beta)$ (Quinn *et al*, 1995). The spatial distribution of the WI has been related to soil water variability and zones of surface saturation and runoff (Hutchinson and Dowling, 1993). The index reflects the tendency of water to accumulate at any point in the catchment (in terms of A_s) and the tendency for gravitational forces to move water downslope (expressed in terms of $tan\beta$ as an approximate hydraulic gradient) (Quinn *et al*, 1993).

The WI was originally intended for predictions of zones of saturation, but its use has now been extended to include predictions of soil moisture pattern (Asserup and Eklöf, 2000). In the present study, the topographic wetness index constitutes an important component of soil moisture variability modelling and its implications for the spatial distribution of shrub and grass species. Anderson and Burt (1990) caution that topographic control is minimal on slopes of less than 10^0. They point out that evidence from a number of studies is available to suggest that topographic WI methods may be appropriate for slopes of 10^0 and above, but inappropriate for slopes below this range or where soil piping is well developed. Hence, 10^0 is considered as the threshold for computing the WI and use of the TOPMODEL over any given catchment.

Soil moisture variations: Implications for species competition and replacement

The two layer-moisture theory has been postulated to explain different moisture dependencies between plant species, particularly between shrubs and grass (cf. Walker and Noy-Meir, 1982; Brown and Archer, 1990; Thornes and Brandt, 1994; Montana *et al*, 1995). The Walther model articulated by Walker and Noy-Meir (1982) relates the moisture dependencies bushes and grasses to soil depth, such that bushes are able to tap deeply draining surface water that the grasses cannot access, giving the former a competitive advantage. Brown and Archer (1990) associated the shallow rooting grass species with soil moisture in the upper soil layer (<30cm) and, by contrast, the deep rooting shrub species with the lower soil moisture layer (30-90cm and >90cm). A disturbance like overgrazing could change the competition scenario such that a reduction in grass biomass through overgrazing would reduce the competitive ability of the grasses (Moretto and Distel, 1999). The result could be a decrease in root production and resource uptake by grasses, allowing the use of more soil resources, particularly moisture, by the shrubs, eventually leading to a change to a more shrubby community (Montana *et al*, 1995).

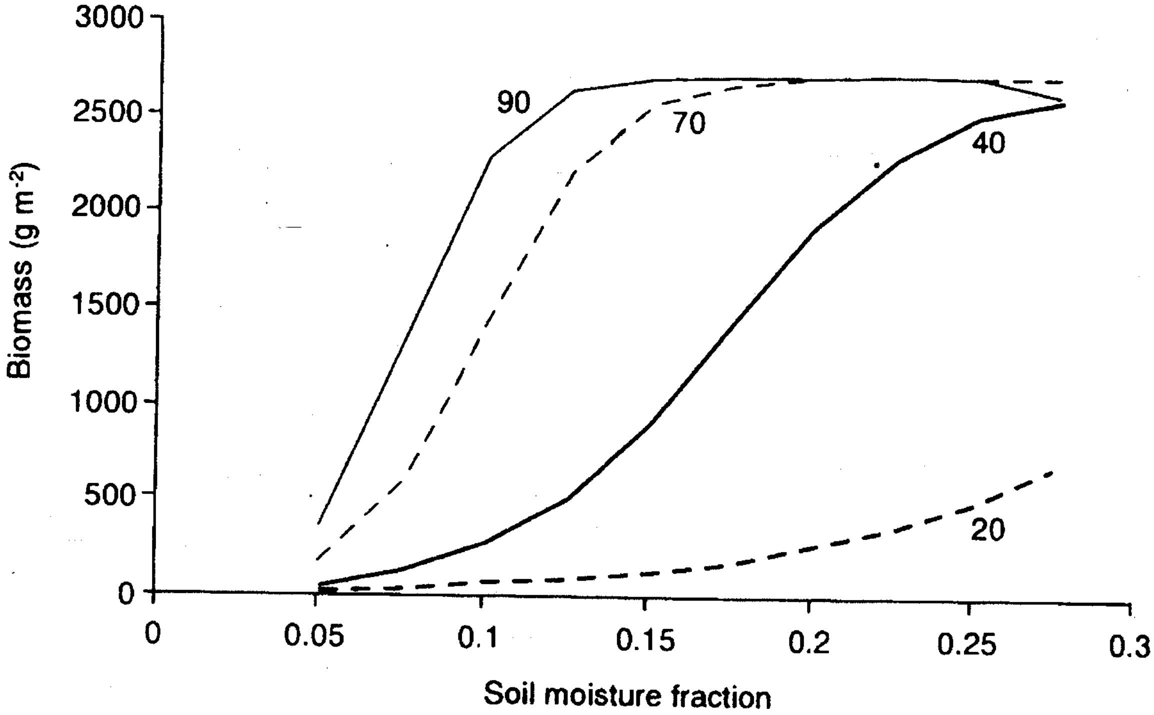
Thornes and Brandt (1994) demonstrate that vegetation capacity is conditioned by soil moisture, which itself varies with soil depth. Vegetation response in the form of biomass is a function of soil moisture and the depth of the soil as illustrated in Figure 2.5 below, based on parameters characteristic of SE Spain. The authors developed a competition model in a stochastic environment to compare two species whose growth parameters have different moisture dependencies. The deep rooting bush species, on the one hand, are supposedly resistant to drought as they can tap deeper, annually available and more reliable water resources. On the other hand are the shallow rooting grass species that are dependent on rainfall and die quickly under dry conditions. They conclude that competition between the plant species comes from the limit to total resource available after the deep-rooting species have satisfied their need.

Figure 2.5: Biomass response as a function of soil moisture and depth of soil (after Thornes and Brandt, 1994).

2.3.4 Bare and vegetated zone implications for runoff generation

That bare and often crusted zones between vegetated patches are characteristically runoff zones has been highlighted in sub-sections 2.3.3.3 and 2.3.3.4 above. Questions relating to vegetation patchiness as posed by Puigdefábregas and Sánchez (1996) are pertinent to this section and are summarised below:

- Do patchy mosaics of sparse vegetation affect the spatial distribution patterns of overland flow and sediment movement and, if yes, to what extent?
- Does the different soil moisture behaviour of vegetated and bare-ground patches explain their relative importance as runoff sources or sinks?
- Do bare-ground patches work as catchments that supply additional water to the downhill vegetated patches?



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- Do bare-ground patches work as catchments that supply additional water to the downhill vegetated patches?

- Do patchy vegetation mosaics show spatial patterns that can be associated with the downhill circulation of water and sediments?

As pointed out by Janeau *et al* (1999), a strong feedback relationship exists between bare surface crusts and vegetated zones, which in turn determines the local runoff pattern. Depending on their pattern, vegetated patches can either act as run-on zones or enhance runoff flowpaths, which could coalesce into larger pathways. According to Holm (2000), the loss of patches created by the death of by long lived plants will alter dynamics of runoff/run-on and erosion/deposition by increasing the fetch, resulting in each runoff event involving larger volumes of water. This view is supported by Cammeraat and Imeson (1999) who observed that the bigger the fetch between individual *S. tenacissima* tussocks, the larger the runoff flowpaths continuing over longer distances, while larger complexes of the tussocks experienced complete infiltration with no trace of overland flow observed below them.

Runoff generation from vegetation patches is largely a function of patchiness pattern. This is showcased by Ludwig *et al* (1999) in a simulation study where they compare the runoff response among four vegetated patch patterns: stipples - dispersed patches, stripes - larger, elongated patches, strands - long linear, basal patches; and one with no patches (see Figure 2.6 below). They observe that the stripe and strand patterns are 8% more efficient at capturing runoff than the stippled pattern. Run-out (runoff loss) from the no patch pattern is noted to be 25 % greater than for the patch patterns. The authors attribute the greater run-out from the stippled pattern to runoff flow between and around the smaller patches.

Such a pattern promotes runoff connectivity and, with much of the runoff becoming run-out, significant soil loss is inevitable. This is corroborated by observations of pedestal formation by Cammeraat and Imeson (1999) whereby the soil surface under less dense *S. tenacissima* patches is higher than the surrounding bare patches. Run-out amounts from the landscape could be exacerbated by the loss of vegetation patchiness, which could lead to the conversion to a dysfunctional system (cf. Ludwig and Tongway, 1995; Pickup *et al*, 1998; Holm, 2000), a condition whose characteristics are discussed in 2.3.5 below.

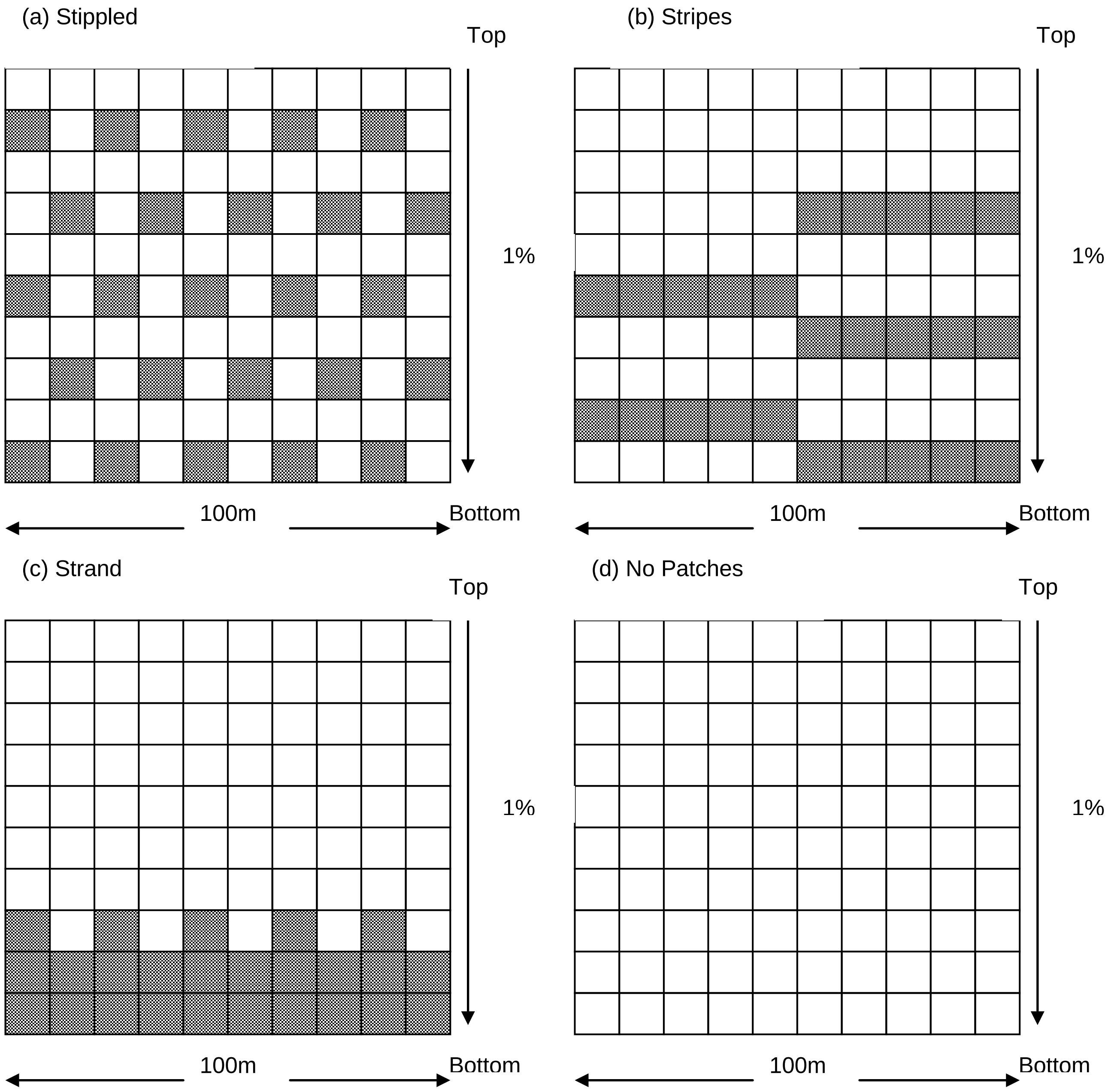


Figure 2.6: Top view schematics of vegetation patches (solid grid cells) and interpatches (open grid cells) for four simulated patchiness patterns (after Ludwig et al, 1999).

2.3.5 Vegetation-bare mosaics: Dysfunctional or resilient systems?

The loss of vegetation patchiness and the resultant expansion of bare areas may give rise to landscape dysfunction (Ludwig *et al*, 1999; Holm, 2000). According to Ludwig and Tongway (1995) and Ludwig *et al* (1999), when a landscape loses its patches, its ability to capture, store and recycle soil sediments and other materials washed into the system is lost. Such a system becomes leaky, runoff becomes run-out, with rainwater, sediments, organic matter and nutrients no longer captured and stored within the landscape. Such scenario may result into a non-resilient, eroded, resource poor landscape that is unlikely to recover from disturbance (cf. Ludwig and Tongway, 1995; Ludwig *et al*, 1999; Holm, 2000). The inability to recover arises from the impact of diverse interlinked processes such as the reduction in the soil infiltration or moisture holding capacity, soil erosion, loss of seed banks and increase in unpalatable woody shrubs (Pickup *et al,* 1998). MacGregor and O'Connor (2002) describe a sequence of events that led to the conversion to a dysfunctional landscape of a semi-arid savanna woodland in the Northern Province of South Africa. This began with degradation associated with early agricultural occupation, the invasion of hydromorphic grassland by woody species, a loss of perennial grasses and a rapid expansion of bare soil surfaces, soil sealing, runoff and erosion. An extended drought of the 1960s is said to have precipitated these changes. The authors suggest that indicators of irreversible change in system functioning are manifest such that productivity on this land has been seriously impaired, carbon sequestration has become minimal, and soil and nutrient loss continue unabated.

Vegetation-bare surface mosaics may be an expression of landscape resilience. The cases described in sub-section 2.3.3.4 (cf. Bergkamp *et al*, 1996; Galle *et al*, 1999; Macdonald *et al*, 1999) whereby vegetated patches act as run-on sinks and traps for organic matter, nutrients and other materials demonstrate the ability of a landscape to recover after its original vegetation has been replaced. As showcased by Ludwig *et al* (1999), the pattern of vegetation patchiness has a strong bearing on system resilience. Whereas some patterns promote run-out and leakage of other materials, others function as source-sink systems.

2.4 Conclusion

This review of geoecological perspectives has highlighted that an interaction between human-induced disturbance, fluctuations in climate and resource availability could favour vegetation invasions. It has also been gathered from the equilibrium / non-equilibrium debate that vegetation change in arid and semi-arid environments cannot be explained in terms of predictable and sequential patterns of linear succession. Regarding geomorphic implications, terrain parameters have a strong bearing on the development and pattern of vegetation-bare soil surface mosaics. In particular, spatial variations in soil moisture as a critical resource in arid and semi-arid environments are largely determined by local hillslope configuration. This has prompted the development of topographically driven moisture prediction models. Crusted soil surfaces, an inherent feature in the development of vegetation patchiness, influence the redistribution of water on and in the soil. Depending on their pattern, vegetation bare-surface mosaics could be an expression of either self-sustaining source-sink systems or leaky run-out systems that could become totally dysfunctional. In the present study, it is within the framework of the controlling factors reviewed in this Chapter that the invasion by *P. incana* is investigated.

Chapter 3: *The Study Area and Pteronia Incana Invasion*

3.1 Introduction

Mgwalana catchment showcases an environmental predicament that is rapidly and steadily unfolding in the communal lands of Peddie district. A considerable section of the district's communal lands bordering the Great Fish River is fast experiencing a rapid conversion of its vegetation to a uniform but undesirable state dominated by a karroid dwarf shrub, *P. incana*. The invasion is by no means unique to the communal lands *per se*. It is also reported in other parts of the Eastern Cape, for instance the Shamwari game reserve where remediation of land encroached by *P. incana* is underway (cf. O'Brien and Campbell 2002). A casual survey along the main Grahamstown-Peddie road also confirms that the invasion is not communal land specific. On the commercial game farms immediately south of the Great Fish river (see Figure 3.1), *P. incana* can be seen thinly interspersed among the Fish River Scrub (veld type 23c, Acocks, 1988). This co-existence with the natural vegetation is vividly distinct from the blanket invasion of vast areas of the communal lands across the Fish River. The invasion problem in the study area is compounded by its coupling with severe soil degradation such that simply clearing the invader as is the case at Shamwari would not be an appropriate remediation strategy.

As a fundamental unit of landscape and process studies, a catchment approach to the present study would provide an ideal framework for understanding the interrelated physical and human underpinnings of *P. incana* invasion and unravelling process-response systems related to the invasion. In this Chapter, an overview of the catchment's physical setting is provided. The origins of the invasion and the salient characteristics of *P. incana* as described in the catalogue of problem plants in southern Africa and other secondary sources are reviewed. Anthropogenic and climatic influences are contemplated to have provided a niche for the invasion. To this end, land use changes in the area are reconstructed and climatic fluctuations over the past century are analysed. A broad catchment survey is conducted to gain an understanding of the extent and pattern of *P. incana* invasion.

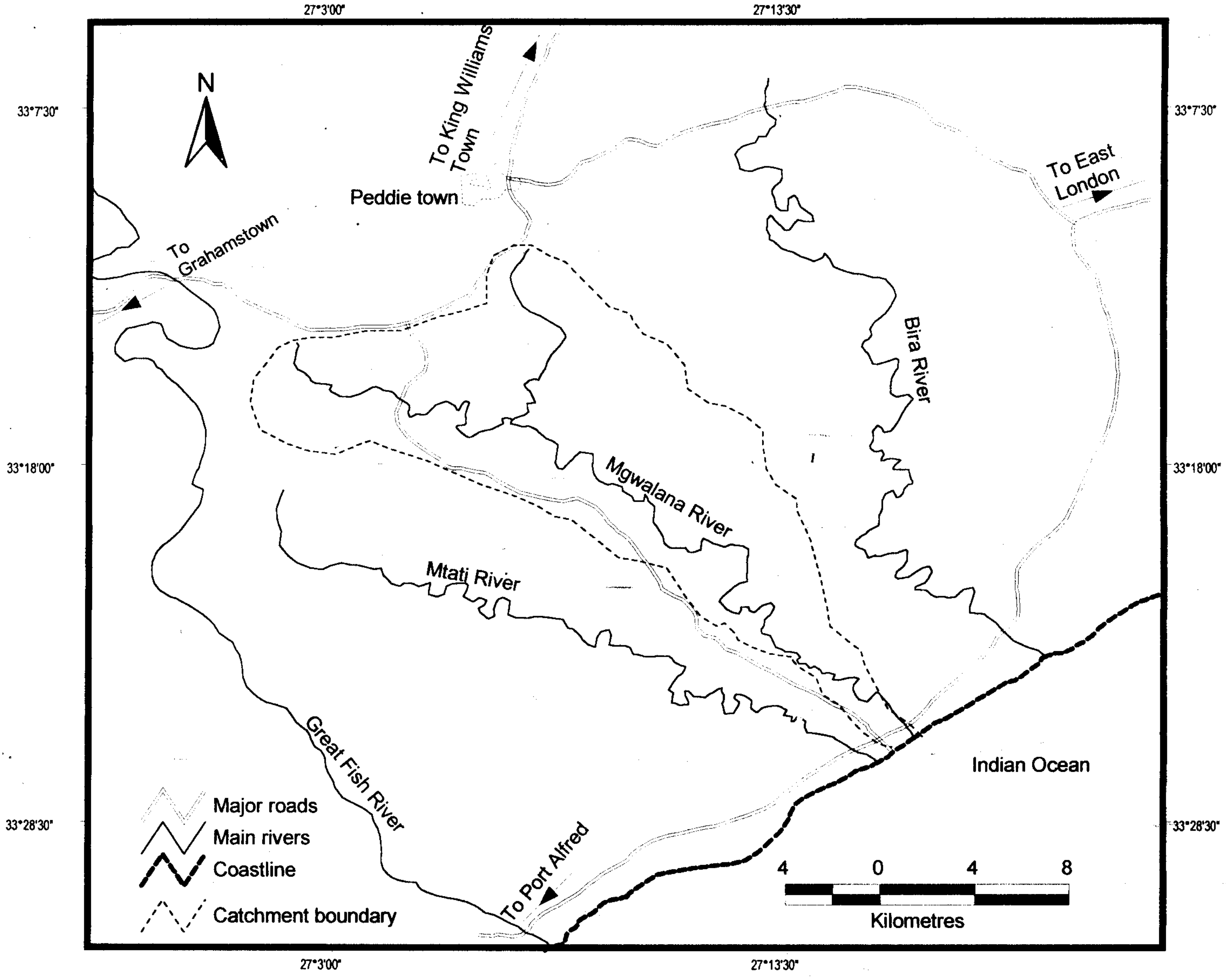


Figure 3.1 The regional setting of Mgwalana catchment.

3.2 Physical setting

Mgwalana catchment covers a surface area of 212km^2. With the highest elevation at 327 m above sea level, it shares a divide with the Great Fish River to the west and south-west. It is bordered by Bira and Mtati catchments to the east and west respectively (Figure 3.1). It is discernible from the geological map series covering the area that the highly fissile shales occurring in association with red mudstones of the Ecca group of the Karoo supergroup predominantly underlie the catchment. Pockets of limestone and calcareous sandstone pockmark the upper-middle catchment. The Ecca group rocks parent shallow litholic soils that belong to the Mispah form, characterised by stony or rocky phases. Swelling hydrous mica clays with a gross structure similar to that of montmorillonite were identified from analyses of soil samples taken from randomly selected sites as the dominant clay mineral compounds (see Table 3.1). Table 3.1 below shows the general characteristics of soil samples taken randomly across the communal rangelands.

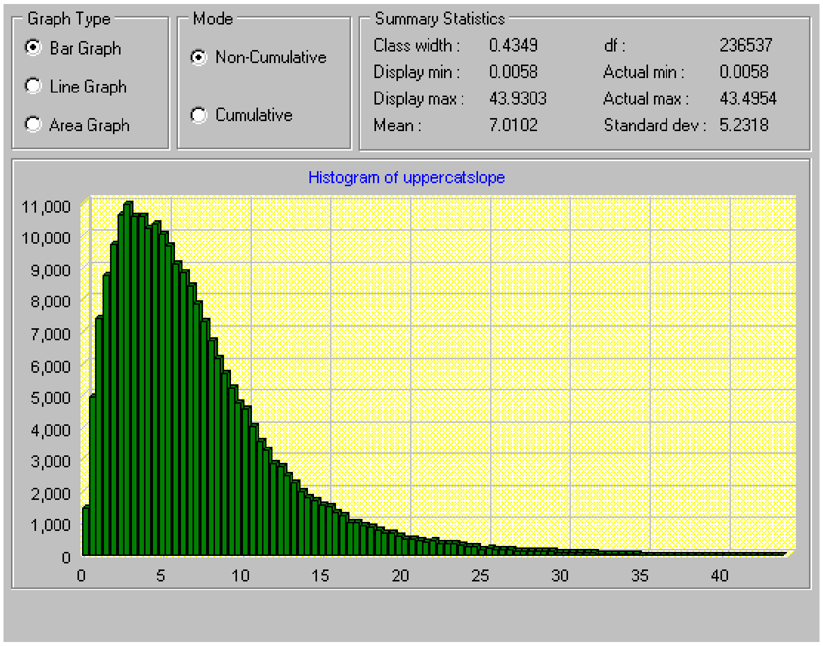
Table 3.1 General characteristics of soil samples taken from randomly selected sites in the study area. (Analyses conducted by Enviro-Quest consulting, Port Elizabeth).

Sample	% sand	% silt	% clay	% OM	SAR	Mineral compounds and Clay type (%)			
						Mica	Kaolinite	Quartz	Goethite
1	45.7	32.1	19.3	1.61	50.60	59	19	22	0
2	66.9	17.1	13.5	0.72	36.50	63	9	28	0
3	55.5	29.5	16.5	1.52	59.86	54	15	31	0
4	46.7	32.5	16.7	1.38	106.55	51	17	32	0
5	62.2	25.3	10.1	1.21	71.88	57	8	35	0
6	47.8	34.0	14.2	2.20	57.83	55	13	32	0
7	26.3	48.8	20.7	2.29	40.09	48	15	37	0
8	41.8	38.6	15.6	1.54	91.53	39	11	50	0
9	41.2	40.9	14.2	2.35	93.91	58	18	24	0
10	35.4	45.4	16.2	2.40	43.26	35	11	50	4
11	26.1	49.2	19.8	2.42	72.73	53	16	31	0

(OM = Organic matter; SAR = Sodium Adsorption Ratio).

The very low organic matter content in the soils gives an indication of how degraded the soils are. Trimble (1990) gives 3.5% as the organic matter threshold below which the soils would have highly unstable aggregates. It can also be gathered from the samples that the local soils are highly sodic as evidenced by the high sodium adsorption ratio. As discussed in Chapter 2, sub-section 2.3.3.3, such characteristics have serious implications for soil surface conditions, particularly crusting due to age-long hardening after disturbance. The implications are put in sharper focus in Chapter 6, section 6.6 and Chapter 7, section 7.8.

The topography of the area is depicted by a slope frequency histogram generated from the slope surface of the catchment in Idrisi Kilimanjaro GIS (Figure 3.2). The slope surface was calculated from a DEM with a resolution of 20m. Figure 3.3 is a summary of the slope frequencies in specific slope categories.



Graph Type
● Bar Graph
○ Line Graph
○ Area Graph

Mode
● Non-Cumulative
○ Cumulative

Summary Statistics

Class width :	0.4349	df :	236537
Display min :	0.0058	Actual min :	0.0058
Display max :	43.9303	Actual max :	43.4954
Mean :	7.0102	Standard dev :	5.2318

Figure 3.2 Slope frequency histogram of the study area generated from the slope surface. (Slope surface calculated from a 20m resolution DEM).

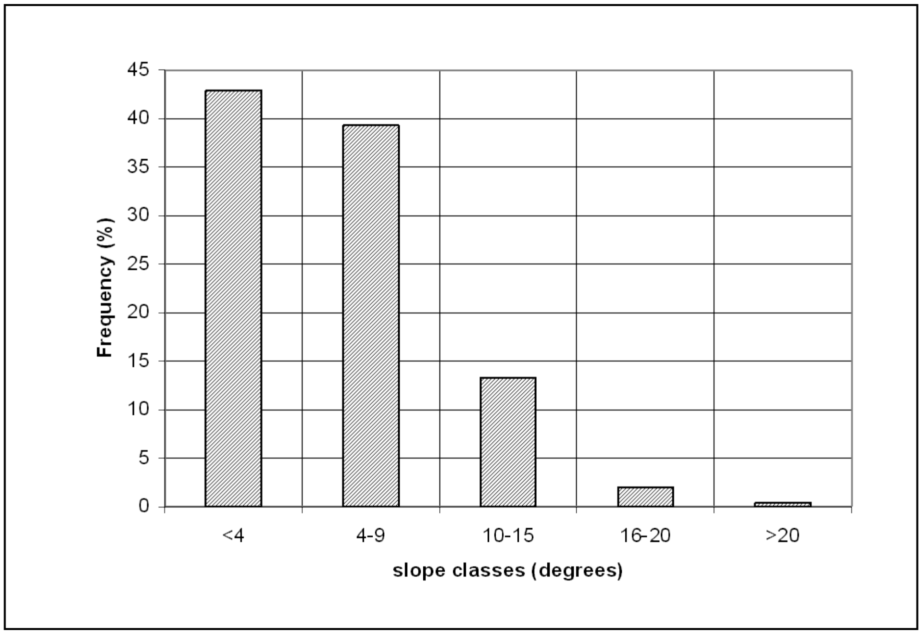
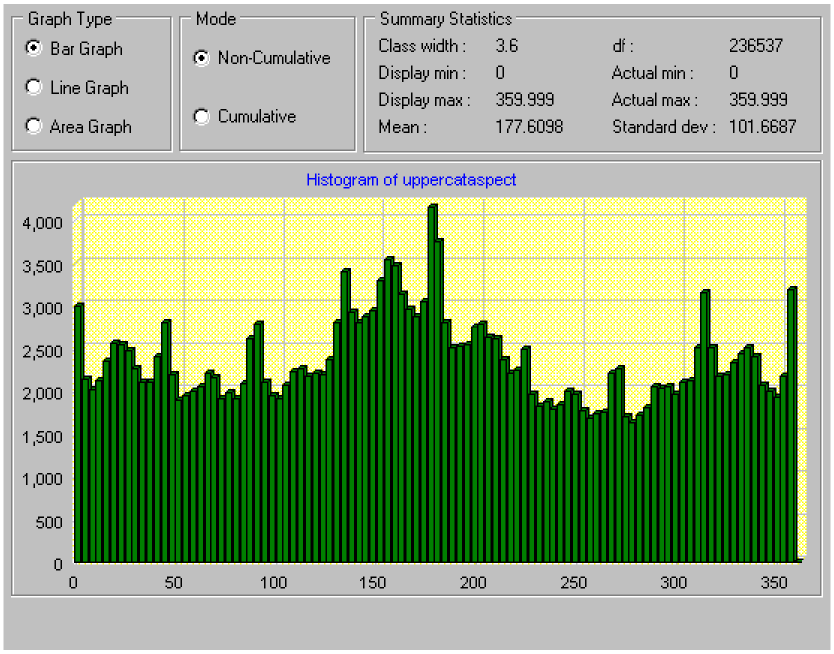


Figure 3.3 Summary of the slope frequencies in specific slope categories.

Whereas slopes below 9^0 constitute over 70% of the catchment, most slopes facing the channels of Mgwalana river and its tributaries characteristically rise steeply (10^0 and above) before they even out into gentle and extensive interfluves. It is these steeply rising slopes that are hotspots for *P. incana* invasion, particularly in the communal land section of the catchment.

The general topographic trends of the study area are such that the south facing slopes have a greater frequency than the rest of the slope directions (Figure 3.4). Overlays of slope and aspect images in the respective classes indicate that the land area occupied by slopes above 10^0 is greater on the south than the north facing slopes. In the slope category of 15^0 and above, the area is twice on the former than the latter slope orientation. Figures 3.4 and 3.5 are a histogram generated from the aspect surface and a summary of the cardinal slope directions.



Graph Type
● Bar Graph
○ Line Graph
○ Area Graph

Mode
● Non-Cumulative
○ Cumulative

Summary Statistics

Class width :	3.6	df :	236537
Display min :	0	Actual min :	0
Display max :	359.999	Actual max :	359.999
Mean :	177.6098	Standard dev :	101.6687

Figure 3.4 Aspect frequency histogram of the study area generated from the aspect surface. (Aspect surface calculated from a 20m resolution DEM).

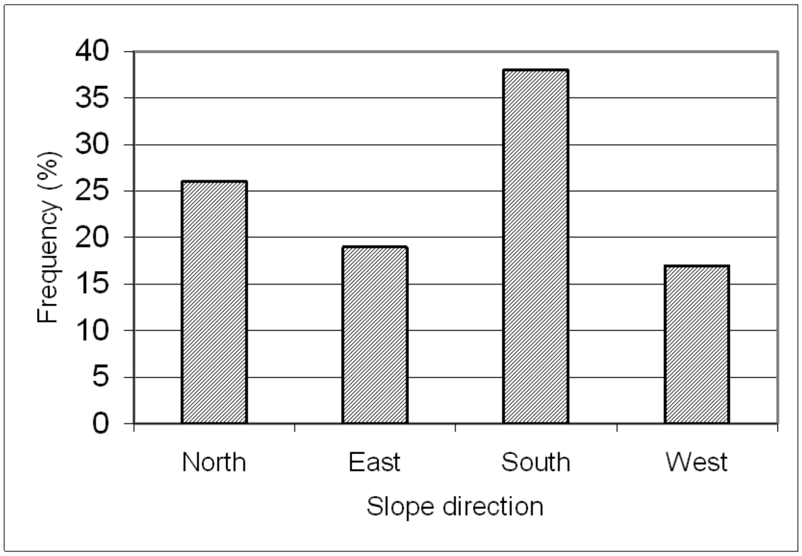


Figure 3.5 Summary of the cardinal slope directions as depicted by Figure 3.4 above.

3.3 *P. incana: The invader*

P. incana is formally documented as an undesirable invader in the catalogue of problem plants in southern Africa. The species is indigenous to the Succulent Karoo biome whose hallmark is the high diversity and strong dominance of dwarf to low leaf-succulent shrubs (Milton *et al*, 1997). It belongs to the *Asteraceae* family, has a perennial life cycle and is relatively long-lived (>20 years). The shrub has woody stems and semi-deciduous thin and sessile leaves with a hairy covering (see Figure 3.6 below). It is propagated via seeds that are mainly wind dispersed. Despite favouring mainly terrestrially dry habitats, the species can thrive in all-year rainfall, temperate winter rainfall and both temperate and sub-tropical summer rainfall regions. According to Wells *et al* (1986), the main undesirable characteristics of the plant are its unpalatability and competitiveness, leading to the replacement of preferred vegetation, particularly grasses.

Figure 3.6 *P. incana* invader plant patch.

P. incana is not new to the Eastern Cape and, as pointed out by Smith (1966), the species was collected in the Albany district in the middle 50s of the 19th century. He suggests that it appears to have spread from the south-western parts of the Klein Karoo. The invader was also reported as a weed in the Alexandria division as early as 1938. Smith attributes the overrunning of the natural grazing areas by the species to poor veld management. According to the terminology recommended by Richardson *et al* (2000), the invader species can best be described as a 'transformer' given that it has clear ecosystem impacts. The authors refer to a 'transformer' as an invasive plant which changes the character, condition and form or nature of the ecosystem over a substantial area relative to the extent of that ecosystem. One such characteristic is that the invader may act as an erosion promoter.

A broad catchment survey to assess the extent of the invasion was carried out (Figure 3.7). At intervals of 1km along the roads and tracks that traverse the catchment, observations of the presence or absence of *P. incana* were made on the slope elements adjacent to the routes. Each slope element – the lower, mid and upper - constituted a sampling point where a GPS reading, slope angle, aspect, soil surface conditions, erosion status, land use, and distance of *P. incana* from the stream channel were recorded. The relationships between these variables and *P. incana* occurrence are explored in Chapter 6.

It was gathered from the broad survey that the invasion is pronounced in the upper half of the catchment where the colonisation of entire hillslopes is a common feature. The vegetation of the lower part of the catchment down to the sea remains largely woodland, sections of which have been opened up for large-scale pineapple growing. This study therefore focuses on the upper half of the catchment which is essentially the affected part. Communal rangelands and the betterment villages of Nyaniso, Mgwalana, Celetyuma and Mhlekwa occupy the upper half of the catchment (Figure 3.8) while former white commercial farms constitute the lower one. Betterment (see section 3.4, p48 for description of betterment), a process introduced in the area in the 1960s ostensibly to control land degradation, is instead perceived as having exacerbated it (cf. Kakembo,

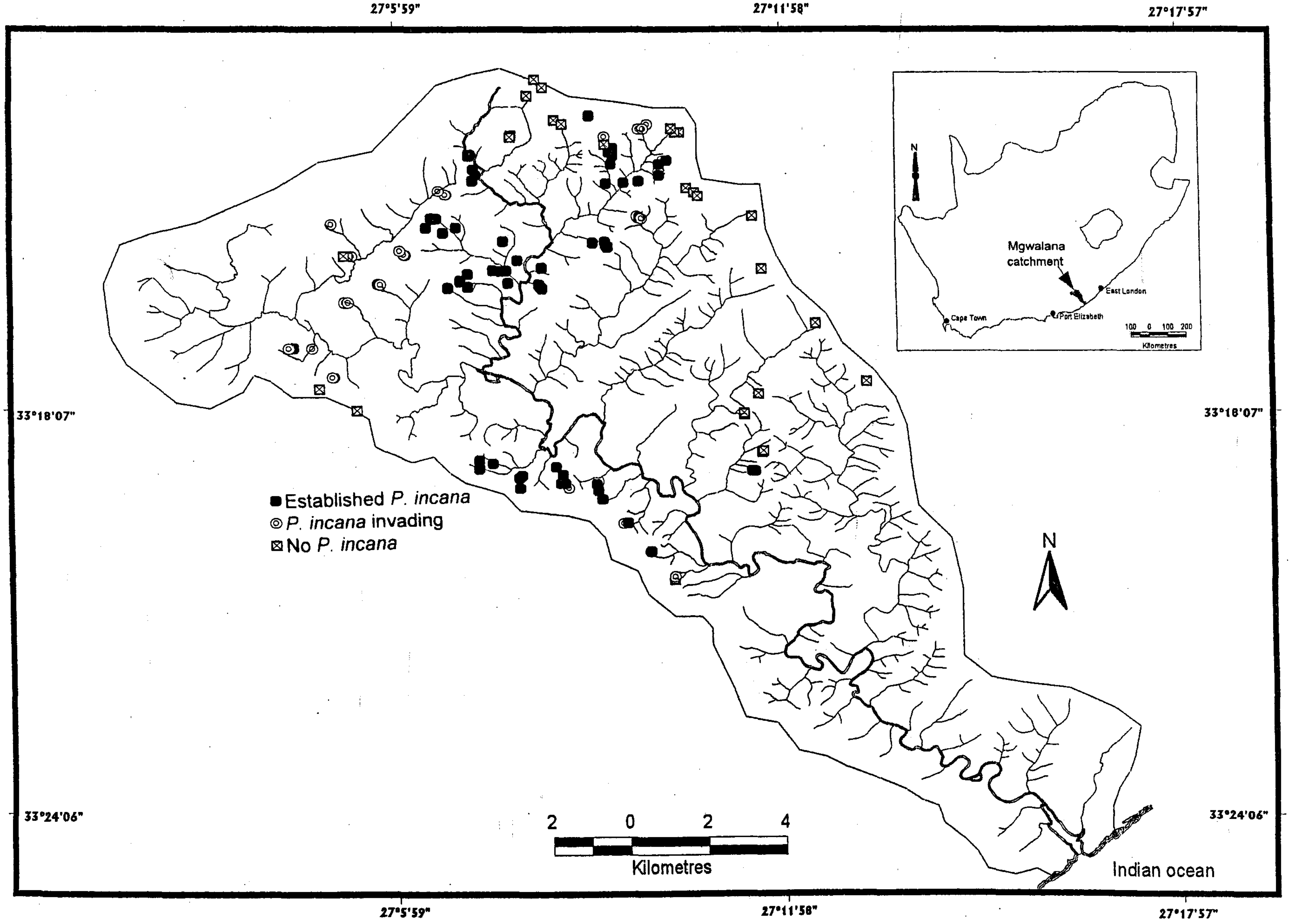


Figure 3.7 Broad catchment survey to assess the extent of *P. incana* invasion

1997; Hoffman *et al*, 1999; Hoffman and Ashwell, 2001; Duma, 2000). The process is seen by Hoffman *et al* (1999) as having fundamentally disrupted the people's ability to farm, and their commitment to the welfare of the land. This lack of commitment will have contributed to the land use changes discussed in section 3.4 below.

3.4 Land disturbance and land use changes

Evidence from past records in the form of aerial photographs indicates that many of the slopes facing Mgwalana river channels in the communal land section of the catchment were intensively cultivated. Outlines of bare cultivated fields can be discerned from the 1:15 000 aerial photographs of the catchment taken in 1975. It is also discernible from the same set of photographs that many of these bare fields are truncated by severe rills and in some instances gullies. This is a clear indication that, by then, they had been abandoned. Land abandonment when population is supposedly rising is obviously an unconventional trend. As pointed out by Kakembo and Rowntree (2003), the trend indicates a significant shift in population-land relationships in the area. The fact that the abandoned lands appear as bare patches on the photographs indicates that they had not been colonised by *P. incana,* as is the case presently. This provides a benchmark to the origins of the invasion. Incipient colonisation of the fields can be picked up from orthophoto maps of the area produced in 1988. Land abandonment can thus be said to be one of the main forms of land disturbance that has had serious implications for *P. incana* invasion in the catchment.

Besides the abandoned lands, grazing land has also been targeted significantly by *P. incana* invasion. Blanket invasion has been observed on some tracts of land, verified as grazing land from past aerial photographs. Invasion by *P. incana* of the existing grazing land is possibly measurable, although the rate has not been determined. The inevitable result of the loss of grazing land to *P. incana* is increased pressure on the remaining grass species whose biomass is often reduced to ground level. The overgrazing phenomenon has been noted from field observations as another major form of land disturbance. It is noteworthy however, that overgrazing *per se* cannot explain the invasion as there is veld still in excellent condition in close proximity to some villages.

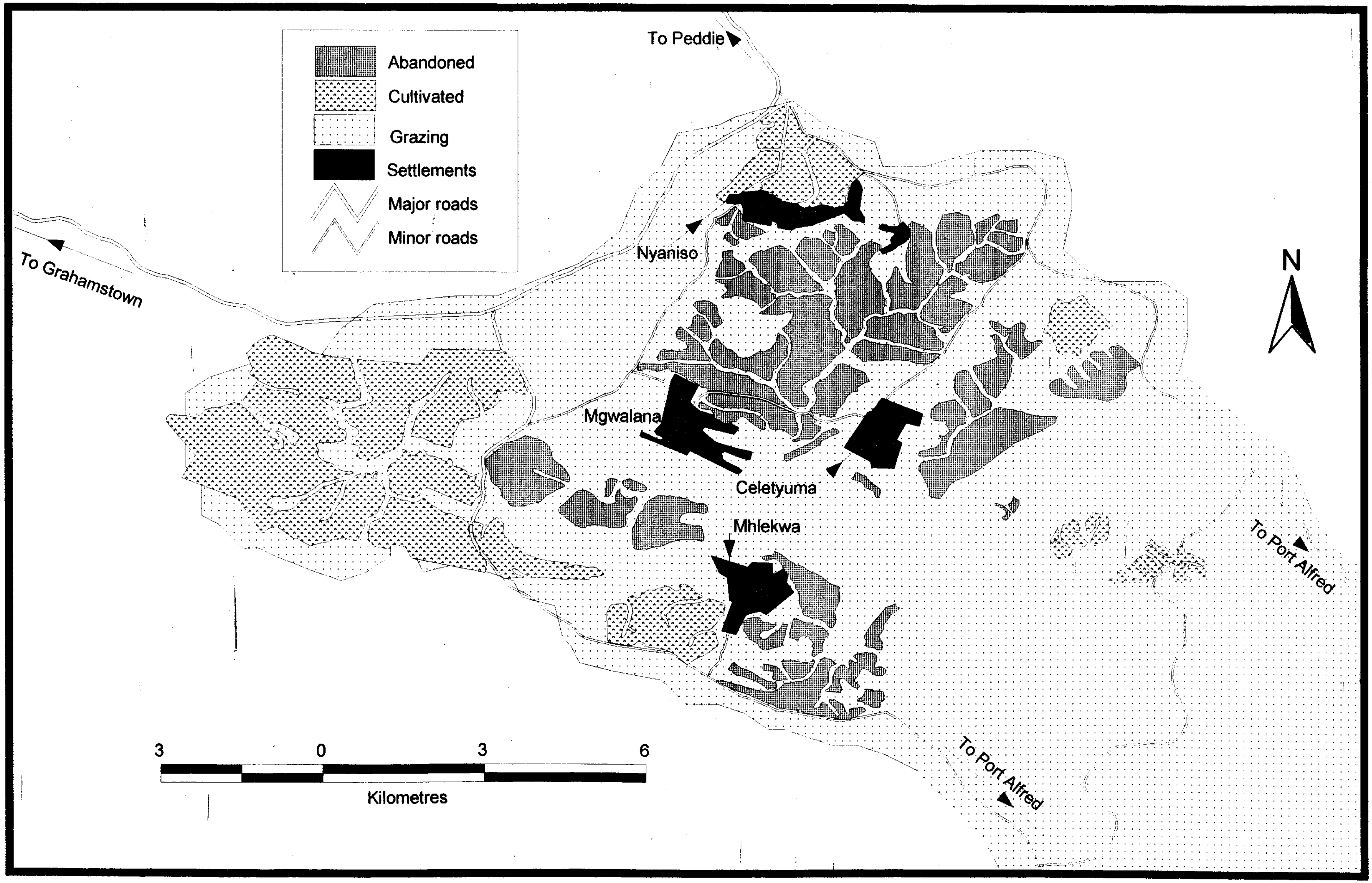


Figure 3.8 Land abandonment and other land use types in the communal lands of Mgwalana catchment. (Mapping based on 1988 orthophoto maps).

Figure 3.8 depicts the abandoned lands and the other land use patterns captured from the 1988 orthophoto maps in Arc/Info GIS format. The present land use distribution based on field observations remains largely the same as depicted by the figure, save the expansion of the invasion on much of the grazing land.

The dynamics of land use change and disturbance in the area are complicated further by the controversial betterment programme alluded to above, imposed on the people of the area in the 1960s. The programme, among other things, entailed the relocation of scattered homesteads into nucleated villages. According to a survey in the area by Hoffman *et al* (1999), betterment was bitterly resented and took root through coercion. The conversion of arable land to grazing and *vice versa* was one of the imposed changes. Arable holdings were forfeited and fields were allocated to residents elsewhere. This essentially meant that the cultivated fields allocated to them were distant from the villages. Many residents were not prepared to travel these distances. Hence, the abandonment of farmland and sound land stewardship could be perceived as resentment to such imposed land use policies. A combination of land disturbance and climatic fluctuations could have provided a niche for *P. incana* to establish itself in the area. These fluctuations in the form of long term rainfall data are examined in section 3.5 below.

3.5 Climatic fluctuations

Typically, the study area experiences a highly variable distribution of rainfall with a 32% coefficient of variation. A mean annual rainfall of 488mm, calculated over 70 years, confirms the semi-arid nature of the area. A summer rainfall peak between the months of October and April and dry winters with rainfall totals ranging between 50-100mm are experienced. Owing to the erratic nature of the area's rainfall, monthly rainfall peaks could be better indicators of episodes of intense rainfall than annual totals. As pointed out by Tyson (1986), indeed it is the frequency of occurrence of rainy spells of at least four consecutive days' duration that distinguish a wet from a dry year. That notwithstanding, for purposes of long term rainfall variability analysis, annual rainfall trends will suffice.

The annual rainfall distribution is positively skewed, implying that a larger proportion of the years receives annual rainfall below the mean than those receiving that above it. This is demonstrated by Figure 3.9 depicting the annual deviation of rainfall from the long-term mean, analysed using a 5year running average. Distinct drought and wet periods can be identified from the analysis. The significance of drought conditions in terms of suppressing the competitive dominance of the resident vegetation, hence promoting invasion was discussed in Chapter 2, sub-section 2.2.2.1.

The period between mid 1950s and 1970s (Kakembo and Rowntree, 2003) is characterised by a progressive abandonment of cultivated land in this area. The period is also persistently dry (Figure 3.9). Given the nature of the local soils alluded to in section 3.2 above, the coupling of drought conditions and land abandonment will have contributed significantly to the impairment of soil surface conditions. This must have inhibited the revegetation of the abandoned lands by resident species. It is thus hypothesised that the combination of drought conditions and land abandonment provided a niche for *P. incana* invasion and its gradual blanket establishment on abandoned and overgrazed land.

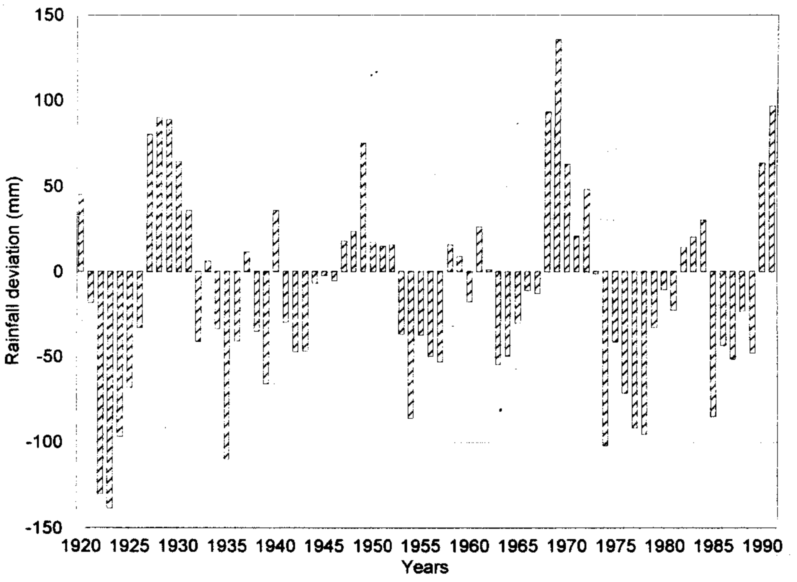


Figure 3.9 Annual deviation of rainfall from the long-term mean (5 year running mean).

3.6 Conclusion

P. incana and its invader traits have been characterised in this chapter. A broad survey of the catchment indicates a concentration of the invasion in the upper part of the catchment occupied by the communal villages. A unique history of coercive restructuring of settlements, land disturbances and climatic fluctuations in the form of abandonment and persistent droughts respectively, pervaded the area almost simultaneously. These factors are deemed to have interacted alongside natural physical ones and geomorphic processes to promote *P. incana* invasion.

Chapter 4: *The Use of High Resolution Imagery to Characterise Pteronia Incana Distribution*

4.1 Introduction

In this Chapter, High Resolution Imagery (HRI) is used to identify different land cover types in the study area. In particular, the spatial distribution *P. incana* and its variations in density are examined. Firstly, the process for image acquisition and exportation to GIS software is described. The procedure for geo-referencing, image enhancement and processing for the different land cover classes is explained. The limitations inherent in HRI and the remedial facilities available in the image processing software are reviewed. Vegetation indices are used to depict the spatial distribution of *P. incana* and other land cover types. Data for all the land cover types are extracted from the images in the form of values files and are graphically plotted. The suitability of vegetation indices for depicting land cover types is examined relative to the digital colour bands of the imagery. Inferences in regard to the efficacy of HRI for characterising *P. incana* distribution are drawn.

4.2 Image Acquisition and Geo-referencing

An investigation of a specific plant community like *P. incana* by remote sensing requires an appropriate spatial resolution. Different studies have concluded that the characterisation of individual phyto-sociological units cannot be obtained by Landsat-TM data due to the spatial resolution of the data (Spjelkavik, 1995; Nelsen *et al*, 1999). In the present study, high resolution infrared images with pixel array dimensions of 1012x1524 were acquired with a Kodak DCS420 digital, colour-infrared camera. The upper section of Mgwalana catchment was flown in a light aircraft in March 2001 at a height of approximately 780m above ground level. This yielded imagery with a spatial resolution of 1m×1m pixels. A total of 28 digital photographs of the catchment were taken. Ten images spanning approximately 1.5 km by 1.2 km each, depicting specific *P. incana* invasion scenarios were selected. The selection was based on earlier catchment surveys described and illustrated in Chapter 3, section 3.3 and Figure 3.3 respectively.

The local growing season is usually ideal for vegetation monitoring as this is the time growing vegetation is photosynthetically active and can have more prominent reflectance signatures (Bryant 1999). This would enhance the spectral separability of plant from soil. March, the month the catchment was flown, is during the local wet and growing season (October to April) and the rainfall conditions for the year 2001 wet season were above average (54.2mm as compared to the 70 year average of 49.3mm).

The imagery was visualised using Adobe PhotoShop 5.0.2 software where they were saved as Tagged Image File Format (TIFF) files. This made it possible to export them readily to the image processing software, Idrisi GIS. In the Idrisi environment, the images were converted into three digital colour band images, the infrared, red and green. The converted images were geo-referenced using real world Ground Control Points (GCPs) whose longitude-latitude co-ordinates were obtained in the field with a Trimble GeoExplorer 3 Global Positioning System (GPS). Clearly identifiable and spread out GCPs in the form of footpath, track and road junctions, and small isolated trees were chosen from the individual images (see Figure 4.1).

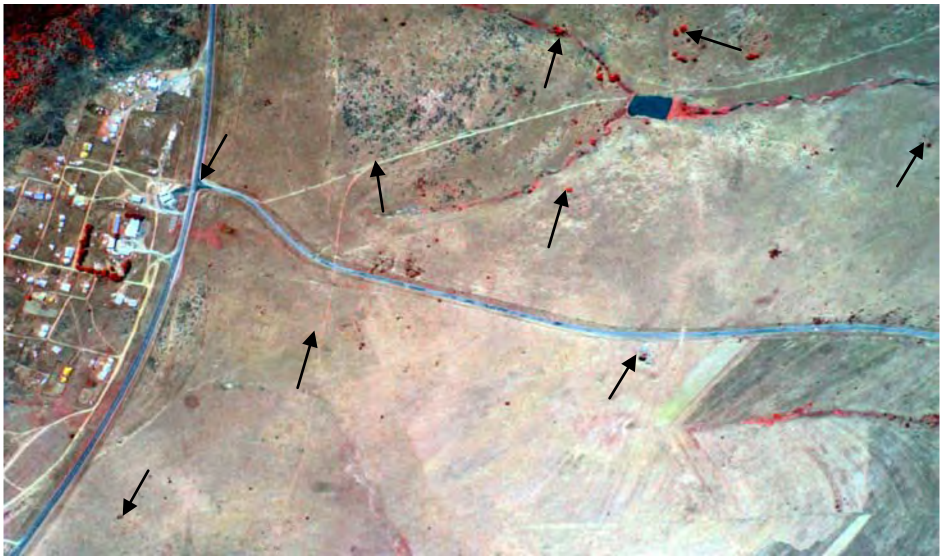
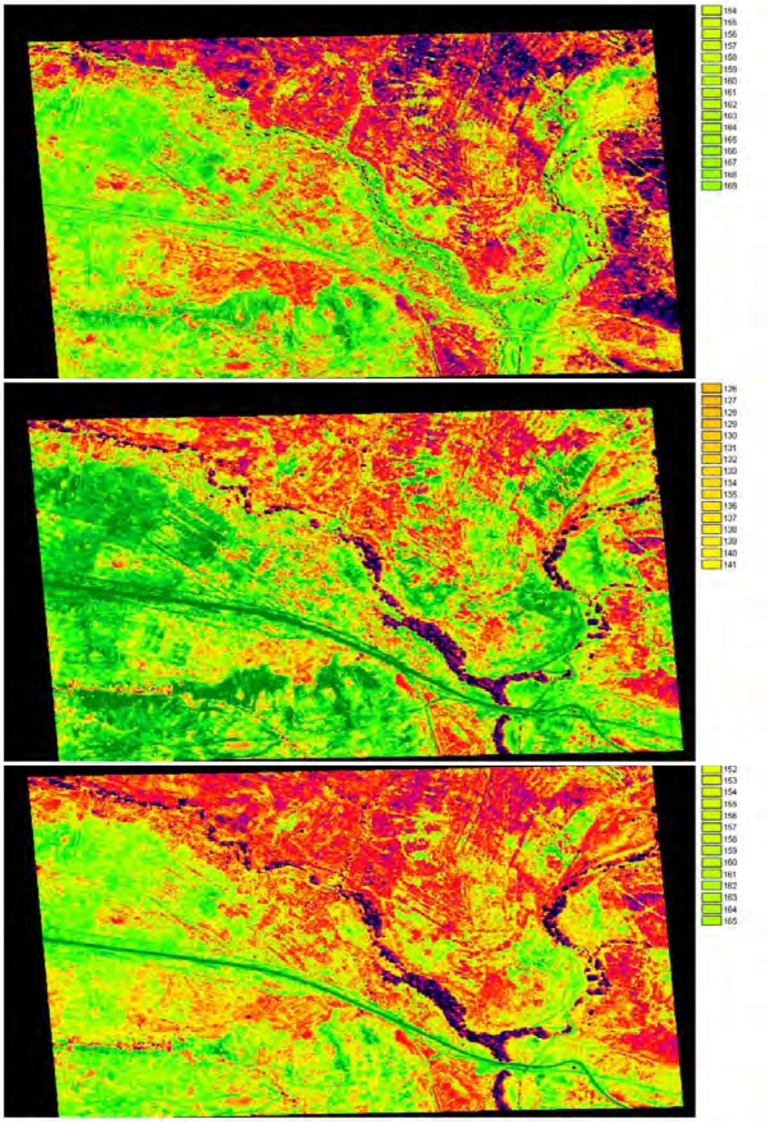


Figure 4.1 Examples of GCPs chosen from digital camera images.

The GPS readings were then differentially corrected using data from the East London base station (about 80km), provided by the Surveyor General at Cape Town. The Pathfinder Office Software provided by Trimble was used for the differential correction process. Twelve GCPs were used to resample each digital colour band image using a linear nearest neighbour resampling algorithm. The Root Mean Square (RMS) error for the resampling process was, in each case, as low as 0.07. Perfect overlays on the images of vector files, digitised from 1: 10000 orthophoto maps of the area, verified geo-referencing accuracy. Figure 4.2 is a sample screenshot of the three digital colour bands converted from one of the digital photographs: the infrared, red and green bands.



Figures 4.2a, b and c – The Infrared, red and green bands

4.3 Image enhancement, limitations and remedial measures

Colour-infrared digital camera imagery does have inherent problems. According to Tueller *et al* (2002), high spatial resolution imagery can be too noisy and hence have superfluous detail. There is a need to apply the filtering facility available in the image processing software for, among other things, noise reduction. Information from the Idrisi32 software help component indicates that the median filter is excellent for random noise removal by preserving useful detail in the image. The ‘adaptive box filter’ option is good for correcting "salt-and-pepper" random noise and also for noisy data where pixel brightness is related to the image scene, but has an additive or multiplicative noise factor. Spatial filters on the other hand are useful for numerous purposes such as image sharpening, smoothing, edge and texture detection (Tueller *et al* 2002). When the ‘adaptive box filter’ was applied to the imagery used in this study, more unexplained noise in the form of black spots with 0 Digital Number (DN) values appeared on the images. Spatial filters like the low-pass and high-pass options tended to distort and over-smooth the imagery. In the end, it is the median filter option that was applied for noise removal, as it was noted to enhance the imagery without any distortions and deviations from the original spectral characteristics. According to Fisher *et al*, (2000), the latter option allows a great deal of high spatial frequency detail to pass. It is quite effective when the noise produces extreme ‘outlier’ pixel values, for instance those corrupted with ‘salt and pepper’ noise.

Adjacent images taken in sequence with the DCS420 camera may not have the same spectral values in the overlapping areas. This is due to, among other things, the variation in the camera shutter speed and viewing angle. Owing to this inconsistency, individual images were treated as different scenes and no attempt was made to extrapolate values for spectral and vegetation indices between images. Spectral distortions, for instance brightness fall-off effects (cf. Dean *et al*, 2000; Mikkola and Pellikka, 2002) and bi-directional effects (c.f. Holopainen and Wang, 1998, Pellikka, 1998, Mikkola and Pellikka, 2002), have been reported in colour infrared images. In their assessment of the suitability of the DSC460c digital camera for quantitative remote sensing analysis of vegetation, Dean *et al* (2000) point out a significant decrease in brightness towards the corners of individual images, referred to as ‘vignette effects’. They attribute this to radial reduction in intensity to increasing obliquity in the view away from the nadir axis. In order to test for brightness fall-off in the imagery used in the present study, perpendicular and diagonal vectors were digitised across one of three of the colour bands for all ten images. Profiles over space were generated from the NIR, red and green bands along the digitised vectors. The profiles did not depict a decrease in DN values and hence brightness fall-off towards the edges of the imagery bands. A visual examination of the images did not show any traces of this phenomenon. Instead, the spectral characteristics of the individual bands were, in general terms, seen to depict the surface cover types as they are known from field surveys. Figure 4.3 shows an example of the trends in brightness values across the three bands of one of the images.

According to Mikkola and Pellikka (2002), bi-directional effects on colour infrared imagery are governed by land reflectance characteristics, atmospheric scattering and the sun-object-sensor angular relationships. The effect introduces brightness variations in the imagery, the front-lit side of the image being brighter than the back-lit one. This effect is more pronounced on vegetated surfaces and could result in inaccuracies in image classification and vegetation analysis (Pellikka, *Pers. Comm*. 2003). In the present study, the classification accuracy levels of the imagery presented in sub-section 4.4.1 of this chapter imply that bi-directional effects on the images used in this study were negligible.

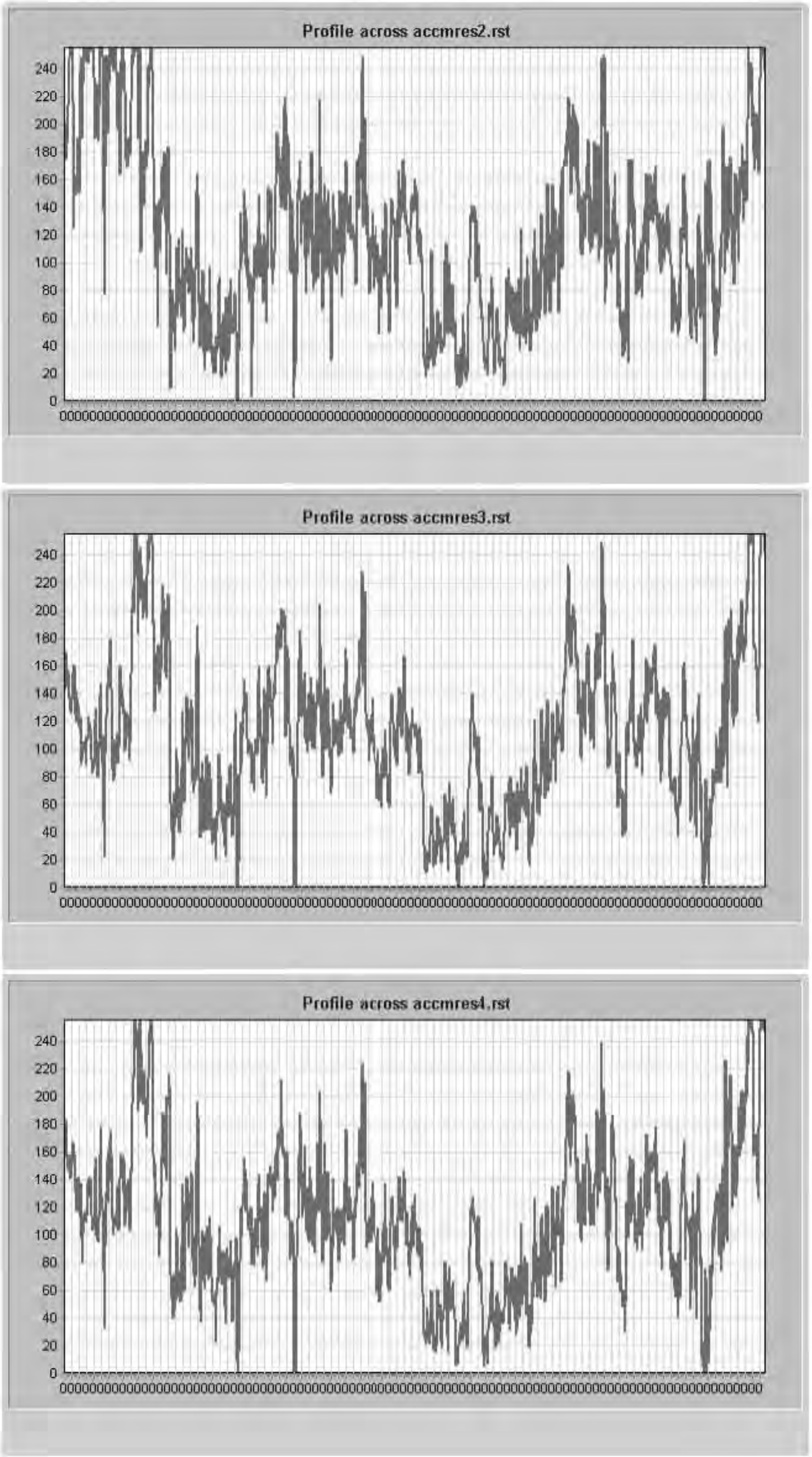
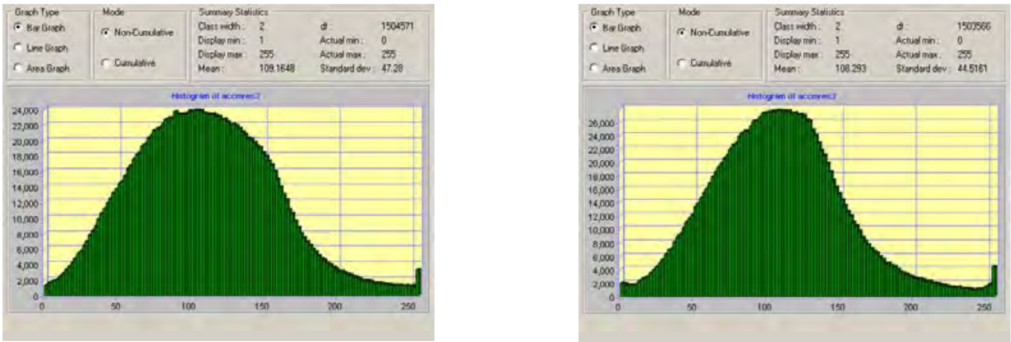


Figure 4.3 Profiles across the Infrared, red band and green bands of one of the images.

Notwithstanding the limitations of the DCS460 colour digital imagery referred to above, the images have been used successfully for quantitative remote sensing (cf. Harris *et al*, 1996; Stow *et al*, 1996; Mason *et al*, 1997a; Mason *et al*, 1997b; Tueller *et al*, 2002). As pointed out by Dean *et al* (2000), in general, quantitative spatial and spectral analyses can be carried out with the infrared colour digital camera images.

4.4 Image Processing and Analysis

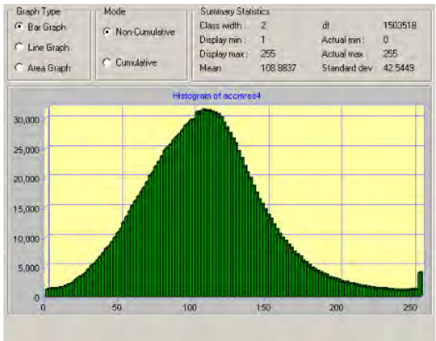
Figure 4.4 shows frequency histograms of the cell values for the three bands of one of the images after applying the median filter, with all three bands spanning a range of 0-255 DN values. The histograms reveal a normal distribution of the data among the three colour bands, typical of all the images whose histograms were plotted.



Class width : 2
Display min : 1
Display max : 255
Mean : 109.1648
df : 1504571
Actual min : 0
Actual max : 255
Standard dev : 47.28

Class width : 2
Display min : 1
Display max : 255
Mean : 108.293
df : 1503566
Actual min : 0
Actual max : 255
Standard dev : 44.5161

a) The infrared band b) The red band



Class width : 2
Display min : 1
Display max : 255
Mean 108.8837
df 1503518
Actual min : 0
Actual max 255
Standard dev 42.5449

c) The green band

Figure 4.4 Frequency histograms of the cell values for the three bands of one of the HRI.

It is noteworthy that digital cameras measure light as a digital number (DN). If the camera sensors were calibrated, the DN could be converted to reflectance values as is the case with satellite imagery. Surface reflectance could be calculated if the amount of radiance (light coming from a surface) and irradiance (incoming light) are known. This is achievable by dividing radiance by the irradiance plus compensation for atmospheric clarity at the time the image is acquired (Drake, *Pers. Comm*. 2003).

4.4.1 Image classification

Image classification was the first image processing task undertaken. Classification accuracy is of paramount importance to the present study, as it rests entirely on the reliable identification of the extent of *P. incana* invasion. A supervised classification using a maximum likelihood algorithm was applied, based on the *a priori* field surveys and data relating to surface cover conditions. GPS co-ordinates and the relevant data relating to the degree of *P. incana* invasion and other vegetation cover reliably informed the choice of training sites. Other cover types like riparian vegetation were readily identifiable from the false colour composite (FCC) created from the three colour bands (Figure 4.5).

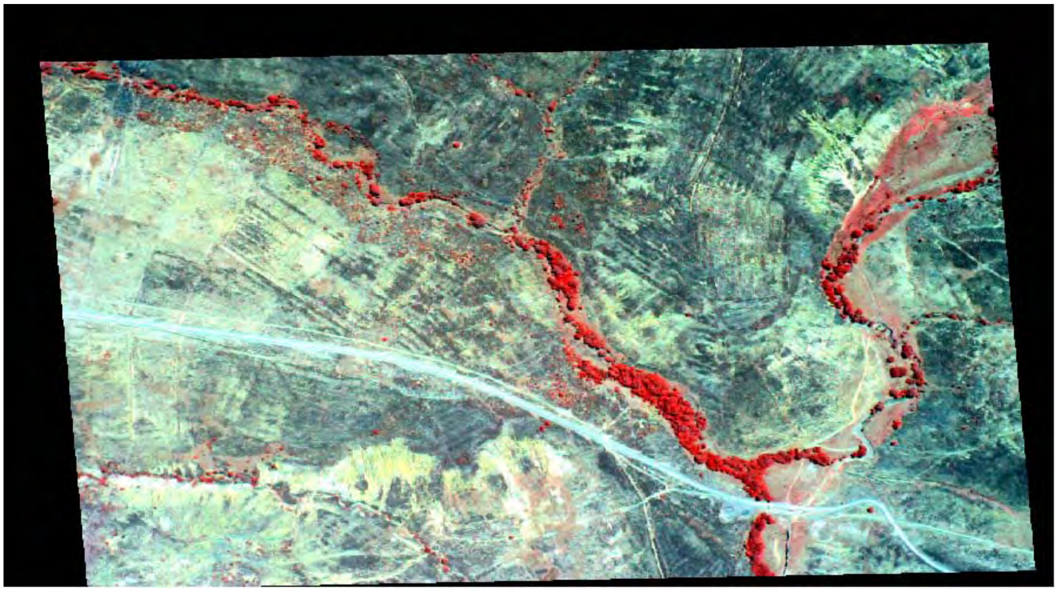


Figure 4.5: False colour composite of the scene classified in Figure 4.7 below.

Given that surface cover conditions are not point-based phenomena, the GPS co-ordinates provided precise nuclei around which the training sites were digitised on the FCC. The typical cover conditions of the area are predominantly advanced *P. incana* invasion usually among badlands, established *P. incana*, invading *P. incana* patches, grass cover, riparian vegetation, gullies and bare surfaces. In order to examine the details regarding the spectral response patterns of the respective cover types, their signatures were compared using SIGCOMP module in Idrisi. This facility also allows one to detect signatures that are similar (Eastman, 1999). Figure 4.6 below shows the minimum, mean and maximum DN values for each cover type extracted from the Infrared, Red and Green bands of one of the images.

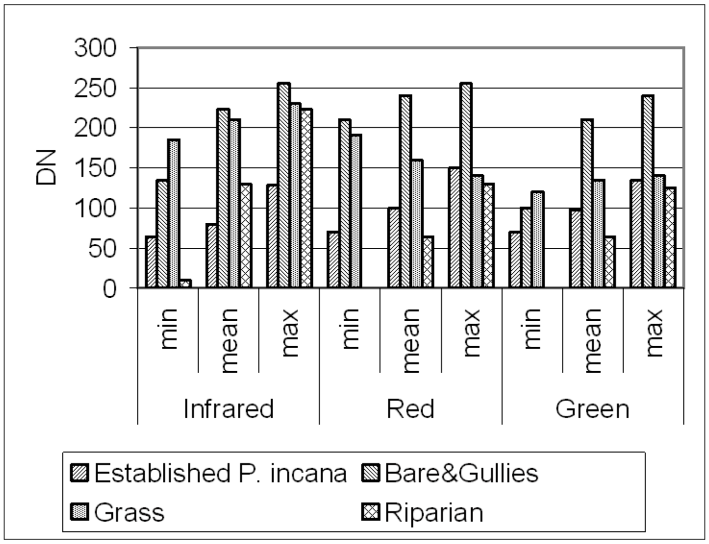


Figure 4.6 Comparison of signatures for the respective cover types.

As can be noted from Figure 4.6, the cover types are noted to assume unique spectral values and no overlap between them is detectable. Consistently different mean values for each of the cover types are noticeable. This confirms the distinctiveness of the signatures for the individual cover types.

Having verified the distinctiveness of the cover type signatures, classified images were produced. Figure 4.7 shows two scenes out of the 10 classified images depicting the cover characteristics outlined above. The scenario depicted in Figure 4.7a is typical of vast areas of the upper catchment while Figure 4.7b shows patches of *P. incana* gradually spreading and replacing the grass cover.

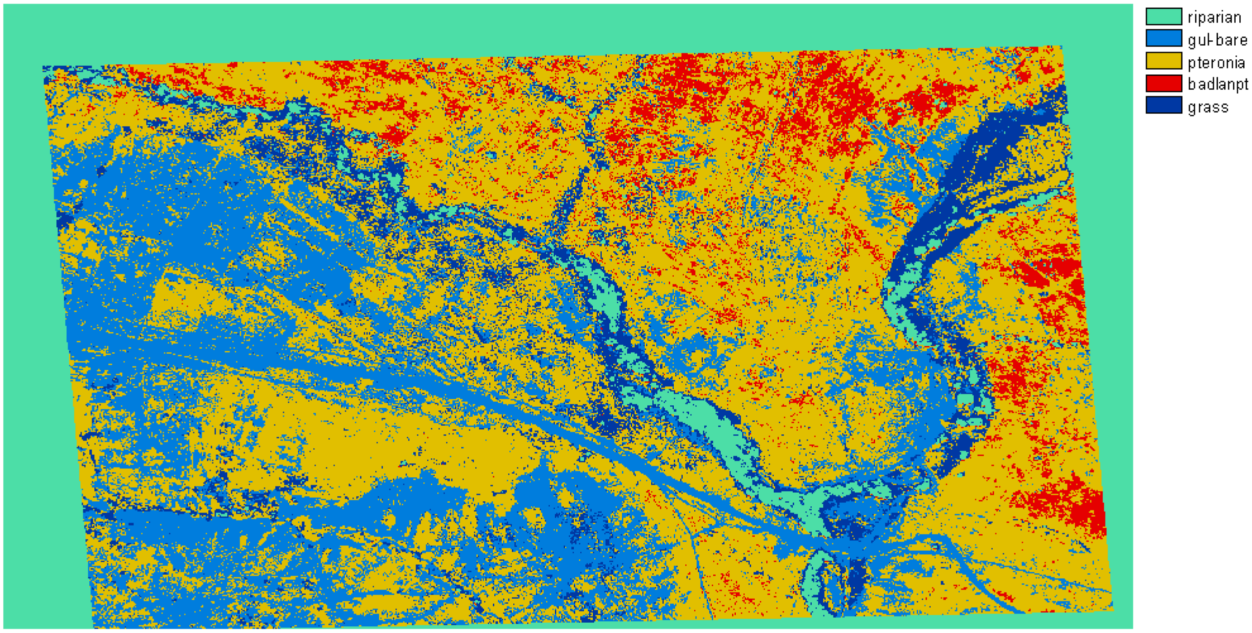


Figure 4.7a Classified image depicting advanced *P. incana* invasion and extensive gullied and bare surfaces (gulbare = gullies and bare areas, badlanpt = *P. incana* in badlands, *P. incana* = established *P. incana*).

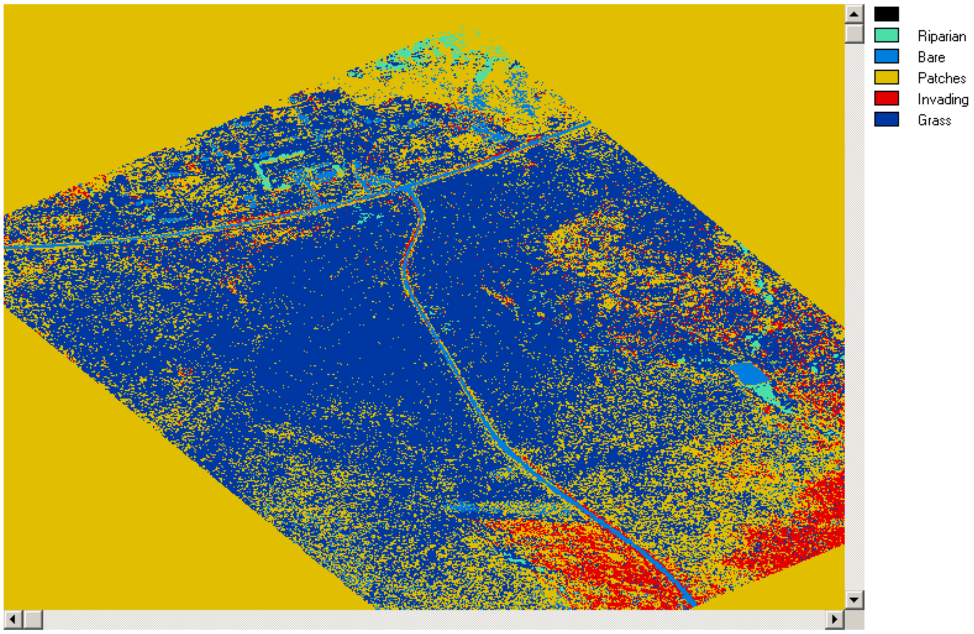


Figure 4.7b Classified image depicting nascent stages of *P. incana* invasion into grass vegetation (Bare surfaces include tar road, Patches = *P. incana* patches, Invading = Fairly established *P. incana* expanding).

4.4.2 Accuracy assessment

Notwithstanding the fact that image classification was ground data driven, errors may creep into the process. Factors including an element of subjectivity in determining the boundaries of the training site polygons, mixed pixel problem and image quality may influence the classification process. Therefore, there is a need to carry out an accuracy assessment of the classified images. The ground data on which the classification was based was split into two sets in order to carry out a post-classification quantitative accuracy assessment. All 10 sets of imagery were re-classified using the second data set after which the ERRMAT module in Idrisi32 was used to compare the second set of output images with the first one. The matrix generated by the module provides estimates of error *viz.*:

- Errors of omission - Representing the cases where sample points of a particular category were found to be mapped as something different.

- Errors of commission - Represents the proportion of sample points assigned to a certain category that actually belonged to another category (Eastman, 1999).

Values along the diagonal indicate cases that matched both classifications - supposedly the groundtruth and mapped images or datasets.

The Kappa Index of Agreement (KIA) built in ERRMAT provides a measure of reliability of the classification. It is an overall index based on all entries in the table, thus combining both the errors of omission and commission. The KIA ranges from 0.0 indicating no correlation to 1.0 indicating a perfect correlation. A KIA of >.70 is considered significant. The KIA values for all the classified images ranged from 0.78 – 0.96, denoting reliable classification accuracy. This level of accuracy also implies that the images have negligible bi-directional effects as explained by Mikkola and Pellikka (2002). The effects are more pronounced in thickly vegetated, usually forested environments. Table 4.1 illustrates an error matrix output for the image whose classification is depicted by Figure 4.7a. The cover surfaces are represented by numbers 1 to 5:

1= Riparian vegetation;
2= Gullies and bare surfaces;
3= Established *P. incana*;
4= *P. incana* in badlands;
5= Grass species.

Matrices for the rest of the images are provided in Appendix A.

Table 4.1 Error Matrix Analysis for one of the classified images. 1= Riparian vegetation; 2= Gullies and bare surfaces; 3= Established *P. incana*; 4= *P. incana* in badlands; 5= Grass species.

```
               (columns : truth) against (rows : mapped)

                  1         2         3         4         5     Total    ErrorC
         ----------------------------------------------------------------------
    1 |      334313         0         0         7        54 |  334374    0.0002
    2 |           2    189192      5204         0     10353 |  204751    0.0760
    3 |         919      8597    604950       700     28359 |  643525    0.0599
    4 |         862         0     29854     82250      1324 |  114290    0.2803
    5 |        2740         0       469         0    242139 |  245348    0.0131
          ----------------------------------------------------------------------
 Total |     338836    197789    640477     82957    282229 | 1542288
ErrorO |     0.0133    0.0435    0.0555    0.0085    0.1420 |             0.0580



         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0003    (0.0577 - 0.0583)
         95% Confidence Interval  =  +/-  0.0004    (0.0576 - 0.0584)
         99% Confidence Interval  =  +/-  0.0005    (0.0575 - 0.0585)


KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using GULCLAS4 as the reference image              GULCLAS3

    Category          KIA                          Category          KIA
------------ ------------
                                                   ------------  ------------


           1      0.9998                                  1      0.9830
           2      0.9128                                  2      0.9499
           3      0.8975                                  3      0.9048
           4      0.7037                                  4      0.9908
           5      0.9840                                  5      0.8311


           Overall Kappa =          0.9204
```

4.4.3 Vegetation indices (VI)

The classified images based on ground data provide a benchmark against which the relationship between the degree of *P. incana* invasion and a range of topographical and hydrological variables could be modelled. That notwithstanding, it would be useful to identify a specific vegetation index or group of indices that best characterise(s) the spatial distribution of *P. incana*. This would serve as a tool to readily discriminate *P. incana* from other cover types, determine the extent of the invasion and identify areas of land that are targets for rehabilitation. To achieve this, a range of vegetation indices is examined in the subsequent subsections of this chapter.

The first step was to calculate the Normalised Difference Vegetation Index (NDVI), the most commonly used basic measure of vegetation cover. The NDVI as a mathematical equation measures the difference between the reflectance of the infrared band and the absorption of the red band and expresses it as a value between –1 and +1. A realistic NDVI value for healthy green vegetation is +0.6. The more positive the NDVI value, the more the green vegetation is within the pixel (Wooster 1995). It is calculated thus:

$$\text{NDVI} = (\text{NIR} - \text{R}) / (\text{NIR}+\text{R}) \qquad (4.1)$$

where NIR = Near Infrared band, and
R = Red band.

According to Lillesand and Kiefer (2000), the NDVI helps compensate for changing illumination conditions, surface slope, aspect and other extraneous factors. The authors point out a number of factors can influence NDVI observations that are not related to vegetation condition. These include the variability in incident solar radiation, radiometric response characteristics of the sensor, atmospheric effects and off-nadir viewing effects. Given that the images were captured at a height of only 780m above the ground, some of the phenomena that tend to impair NDVI values were avoided. For instance, atmospheric interference in the form of radiation absorption was non-existent and the radiation scattering effect by water vapour and dust particles was negligible. Figure 4.8 below shows NDVI images of the same scenes whose classification is depicted in Figure 4.6 above.

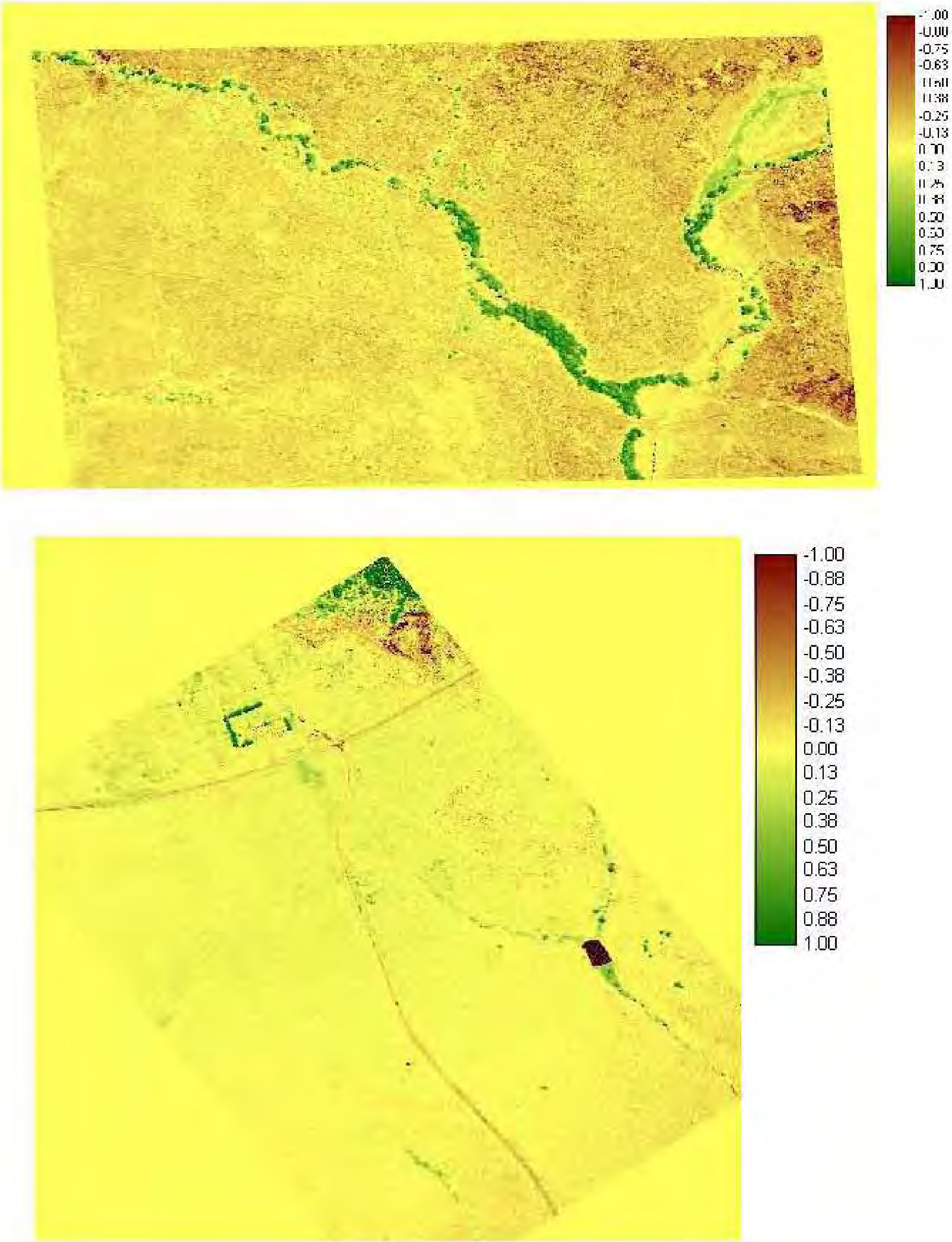


Figure 4.8a and b NDVI images of the classified scenes shown in Figure 4.6a and b above.

The NDVI images above reflect the vegetation condition of the areas they depict. Riparian vegetation is conspicuous, with high NDVI values, while grass cover is also distinguishable, with low but positive NDVI values. All areas invaded by *P. incana* have negative NDVI values but are indistinguishable from bare surfaces and gullied areas as the latter surfaces similarly have negative values (see Figure 4.8a and b). Figure 4.10a illustrates that the NDVI does not provide a clear spectral separation between established *P. incana* and bare surfaces. This is one of the major limitations of the NDVI such that, in cases where vegetation cover is interspersed with considerable bare areas, as is the case with *P. incana*, light reflected from the soil can have a significant effect on NDVI values, modifying them by up to 20% (Huete and Jackson 1988). According to Duncan *et al* (1993), different studies have found the use of greenness Spectral Vegetation Indices (SVIs) like the NDVI to be problematic in semi-arid regions due to low vegetation cover and highly reflective and variable soils. The authors further point out that the spectral response of vegetation in the green and infrared wavebands is less consistent than the red band, particularly where sparse canopies are contrasted with bright soils, a view supported by Ringrose *et al* (1989).

Huete (1988) cited by Duncan *et al* (1993) found that neither the NDVI nor the Perpendicular Vegetation Index (PVI, explained in section 4.4.5) to yield consistent estimations of vegetation biomass under conditions of incomplete canopy cover and variable soil background. Incomplete canopies scatter a portion of the incoming NIR radiation towards the soil, which is subsequently reflected back to the sensor and is thus dependent on soil optical properties. In their work undertaken in the Kalahari, Ringrose *et al* (1994) noted that during drought periods, when sparse green vegetation reflected against the bare soil background, Landsat MSS imagery appeared dark. They attributed this to the high soil reflectance, high proportions of stem to leaf area, and a low density cover. Ringrose *et al* (1989) also showed that in the Kalahari, semi-arid vegetation signatures do not give good spectral separation, as the infrared reflectance can be higher from the soil than from vegetation. Adaptive characteristics like the decreased leaf size and its cellular structure may be significant in reducing the effective infrared reflectance from plants in arid and semi-arid areas (Malthus *et al*, 1993).

A group of vegetation indices that attempt to reduce soil noise by altering the behaviour of iso-vegetation lines has been developed. These include The Soil Adjusted Vegetation Index (SAVI), developed by Huete (1988); the Transformed Soil Adjusted Vegetation Index (TSAVI) by Baret *et al* (1989), cited by Qi *et al* (1994); the Modified Soil Adjusted Vegetation Index (MSAVI_1 and MSAVI_2) by Qi *et al* (1994). Huete (1988) found that SAVI accounted for first-order soil–vegetation interactions better than the NDVI or PVI. The SAVI incorporates a correction factor into the NDVI equation that varies with the density of the vegetation.

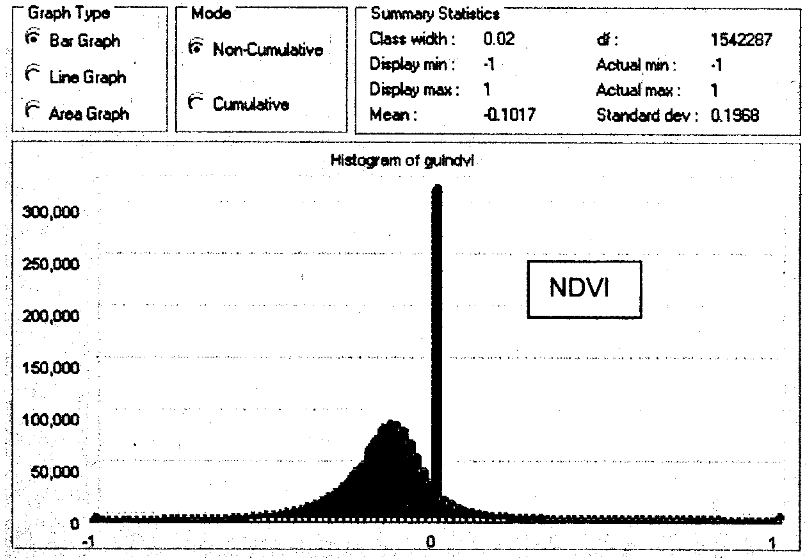
$$\text{SAVI} = [(\text{NIR}-\text{red})(1+\text{L})/(\text{NIR}+\text{red}+L)] \tag{4.2}$$

where: L is soil adjustment factor varying from 0.0 to 1.0.

The standard value typically used in most applications is 0.5, which is for moderate vegetation densities. It minimises soil brightness variations and eliminates the need for additional calibration for different soils (Huete *et al*, 1994). The TSAVI on the other hand takes into account the soil line slope and intercept, and used an adjustment factor of 0.08 to minimise the soil background effect. Since the TSAVI is in all respects similar to the SAVI (Qi *et al* 1994), it is not tested on the images in the present study. The MSAVI_1 and MSAVI_2 are also very similar in their sensitivity to vegetation and normalisation of soil noise (Qi *et al* 1994). The SAVI and MSAVI_2 are tested on the scene presented in Figure 4.8a above, which showcases a full-blown *P. incana* invasion scenario with pronounced bare surfaces and gullies. The latter vegetation index is calculated thus:

$$\text{MSAVI}_2 = \frac{2\text{NIR}+1-\sqrt{((2\text{nir}+1)^2 - 8(\text{NIR}-\text{RED}))}}{2} \tag{4.3}$$

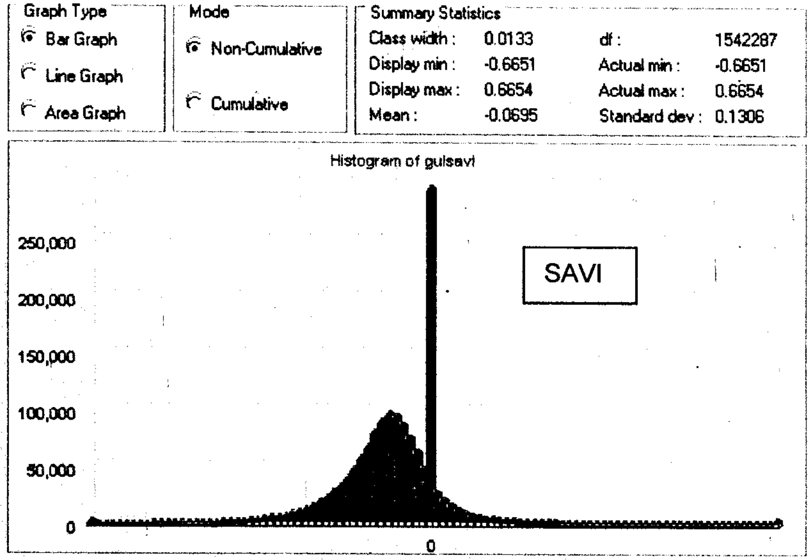
The frequency histograms for the NDVI, SAVI and MSAVI_2 images derived from the scene in Figure4.8a are compared in Figure 4.9. The mean, standard deviation, maximum and minimum values for each histogram are also provided.



Graph Type
● Bar Graph
○ Line Graph
○ Area Graph

Mode
● Non-Cumulative
○ Cumulative

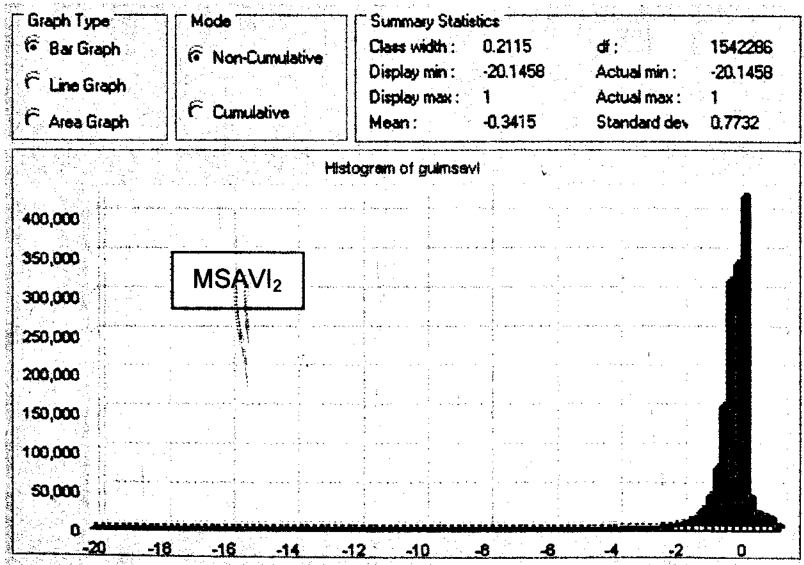
Summary Statistics
Class width : 0.02
Display min : -1
Display max : 1
Mean : -0.1017
df : 1542287
Actual min : -1
Actual max : 1
Standard dev : 0.1968



Graph Type
● Bar Graph
○ Line Graph
○ Area Graph

Mode
● Non-Cumulative
○ Cumulative

Summary Statistics
Class width : 0.0133
Display min : -0.6651
Display max : 0.6654
Mean : -0.0695
df : 1542287
Actual min : -0.6651
Actual max : 0.6654
Standard dev : 0.1306



Graph Type
● Bar Graph
○ Line Graph
○ Area Graph

Mode
● Non-Cumulative
○ Cumulative

Summary Statistics
Class width : 0.2115
Display min : -20.1458
Display max : 1
Mean : -0.3415
df : 1542286
Actual min : -20.1458
Actual max : 1
Standard dev 0.7732

Figure 4.9 Comparative NDVI, SAVI and $MSAVI_2$ frequency histograms .

A similar distribution of cell values for the NDVI and SAVI is noticeable. Their histograms are bell shaped as opposed to the negatively skewed cell values for the $MSAVI_2$. Extremely high frequencies for 0 cell values are evident in all three cases. This would signify widespread bare surfaces, depending on how well each of the three indices separates bare areas from other surfaces. The ability of the indices to separate the respective cover types is investigated in section 4.4.4 below.

4.4.4 Extraction of values from ratio based VI images

In order to determine whether the NDVI, SAVI and $MSAV1_2$ images could provide some form of spectral separability between different *P. incana* densities and the other surface cover types, values from all three images were plotted against the Infrared and red values. Using the sample classified image (Figure 4.7a), 250 points representing the respective surface cover types were digitised and rasterised. The EXTRACT module in Idrisi32 was used to extract values files from the three vegetation index images. Figures 4.10 and 4.11 below show the scatterplots.

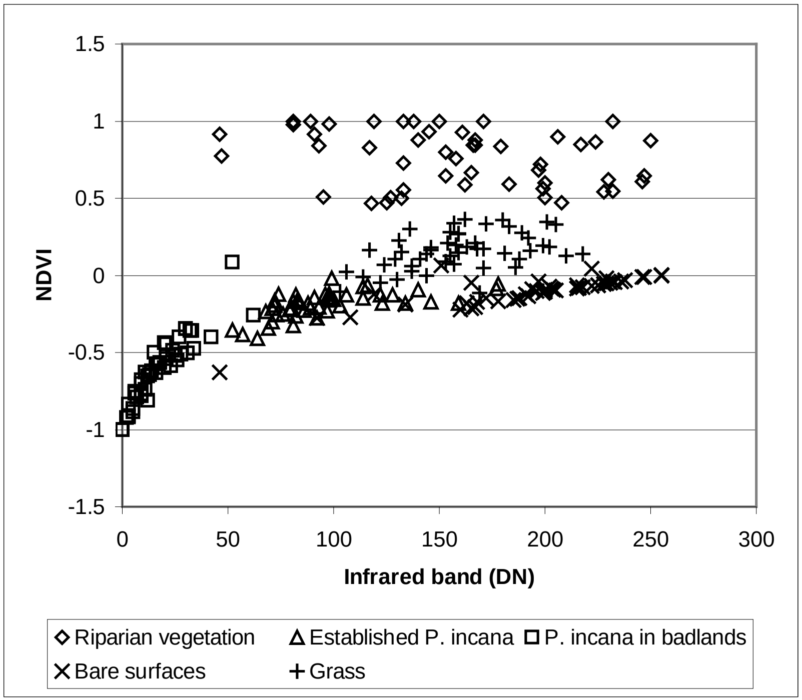


Figure 4.10a NDVI vs the Infrared band

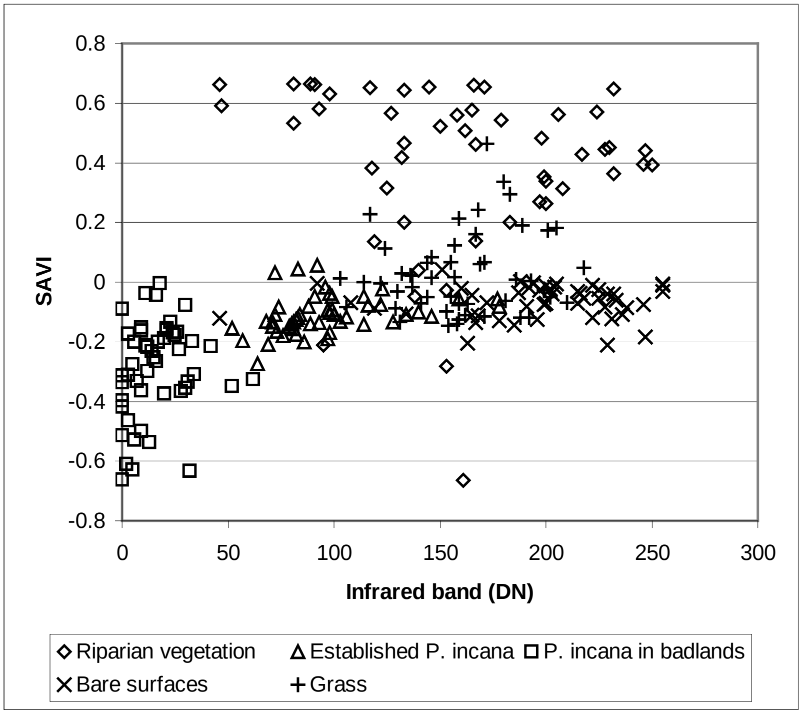


Figure 4.10b SAVI vs the Infrared band

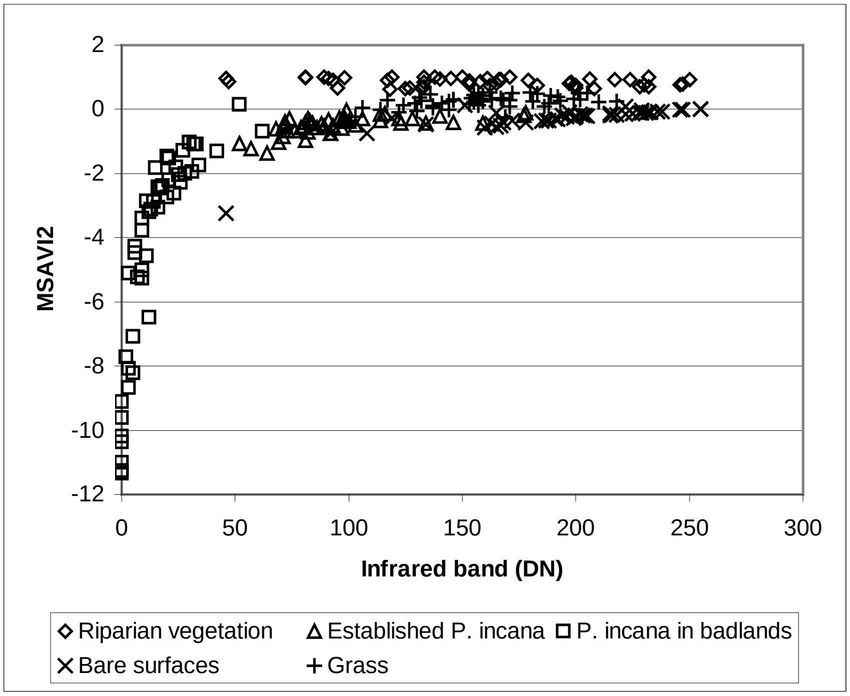


Figure 4.10c MSAVI2 vs the Infrared band

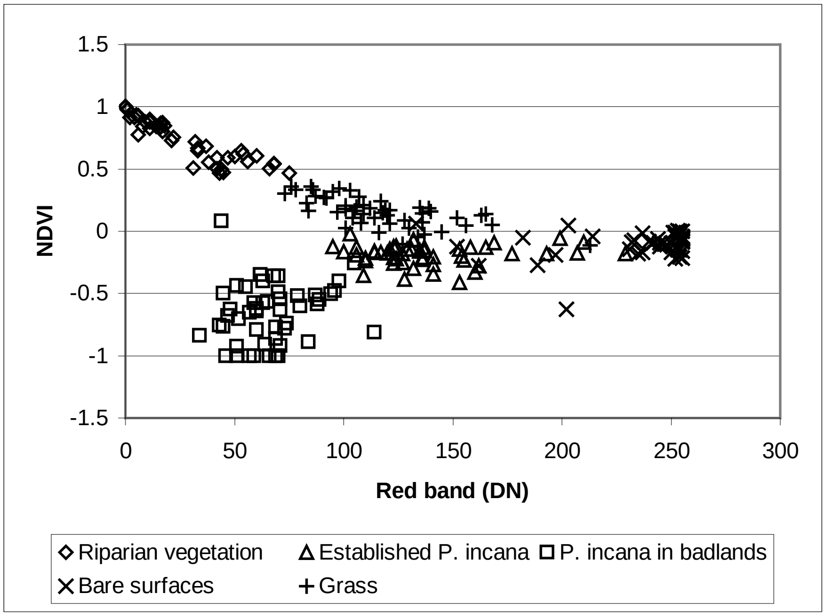


Figure 4.11a The NDVI vs the red band.

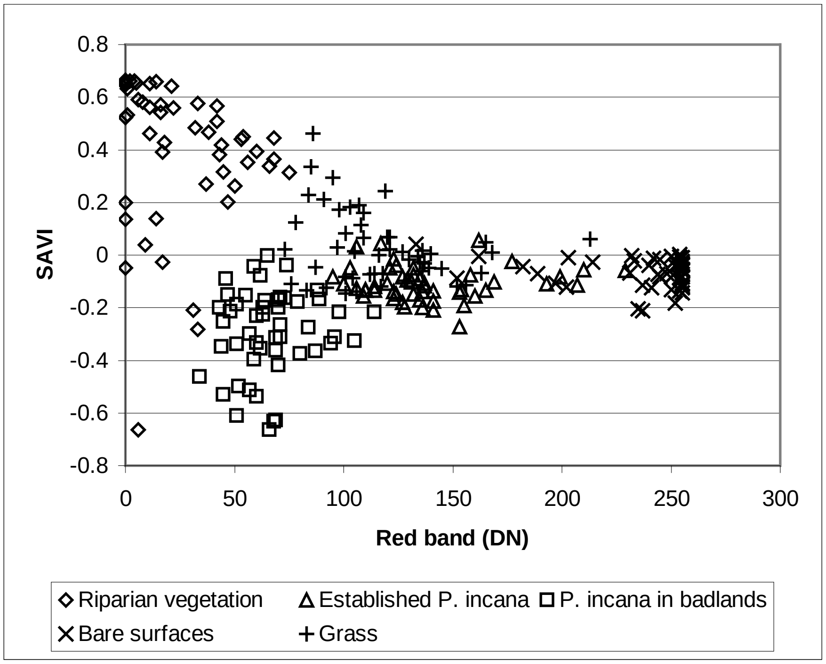


Figure 4.11 b SAVI vs the red band

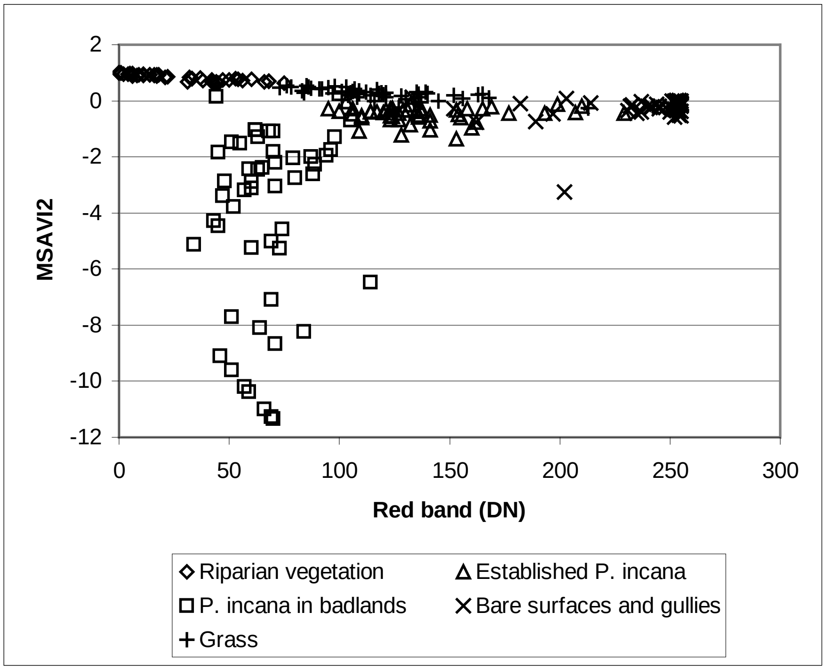


Figure 4.11c MSAVI2 vs the Red band.

It is discernible from Figures 4.10 and 4.11 above that the respective vegetation indices do not separate out the 'established *P. incana*' and 'bare surfaces' categories. A mix up of grass, bare surfaces and established *P. incana* is discernible particularly from the SAVI and $MSAVI_2$ scatterplots. The latter vegetation index does not clearly distinguish grass from established *P. incana*. The NDVI on the other hand separates grass from bare surfaces and established *P. incana*, but not established *P. incana* from bare surfaces. So, under the vegetation conditions investigated in the present study, the two vegetation indices fail to minimise soil noise. Overall, the NDVI fares better than the other two vegetation indices. This confirms the observation by Qui *et al* (1994) that over sparse or incomplete canopies, the NDVI produces higher Vegetation Indices (VI) values than the other indices. The authors suggest that is an artefact of the non-linear ‘convex’ response pattern of the NDVI to green cover. Regarding all three vegetation indices, earlier observations (cf. Musick, 1986; Ringrose, *et al* 1989; Duncan *et al*, 1993 and Ringrose *et al*, 1994) that the use of SVIs based on ratio of infrared/red reflectance has not been successful in arid and semi-arid areas are confirmed.

4.4.5 Infrared- red band relationships

The infrared and red bands are to some extent, seen to separate the land cover surfaces into distinct clusters in the Figures 4.11a-c. It is noticeable from the scatterplots that the DN values for both bands vary fairly distinctively along the abscissa with the different surface cover types. When plotted against each other, the two bands could provide some reliable measure of separability between the surface cover types. This is in line with the concept of the soil-line, which denotes the relationship between the red and infrared reflectances. A more or less linear relationship exists between the red and infrared reflectances from bare soils (Fox and Sabbagh, 2002). The relationship is expressed thus:

$$NIR = a\,R + b \qquad (4.4)$$

where ‘a’ and ‘b’ are the slope and intercept respectively.

When the values for the two bands are plotted, this yields the Perpendicular Vegetation Index (PVI). The PVI is the perpendicular distance of the measured point from the soil line and is defined as follows:

$$PVI = 1/\mathrm{sqrt}(a^2+1) \times (NIR - aR + b) \qquad (4.5)$$

where ‘a’ and ‘b’ are the slope and gradient/intercept of the soil line respectively. Figure 4.12 below illustrates this.

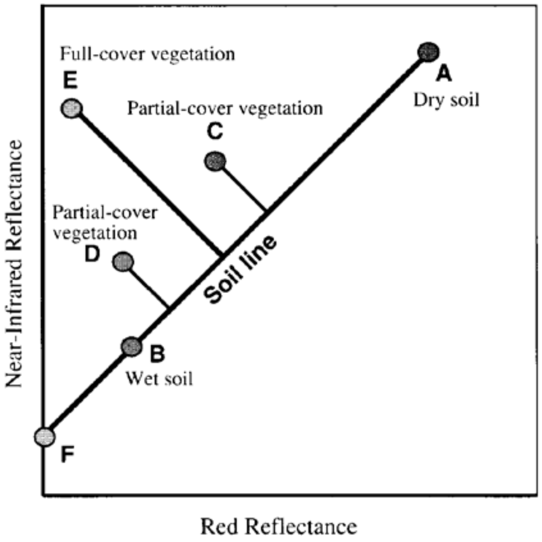


Figure 4.12 The relationship of vegetation to soil background using the PVI (Adapted from Jensen, 1996).

According to Figure 4.12, the PVI measures the changes from the bare soil reflectances caused by the vegetation. In this way it gives an indication of vegetative cover independent of the effects of the soil (Department of Meteorology website, The University of Reading, 2003). Points A and B represent dry and wet soils respectively. C and D are supposedly partially vegetated areas with relatively dry and wet soils respectively (Jensen, 1996).

In the attempt to achieve distinct spectral separation of *P. incana* cover from the other surfaces, the red and infrared band DN values for the respective cover types were extracted from the same sample image from which VI values were plotted. Figure 4.13 below illustrates this relationship.

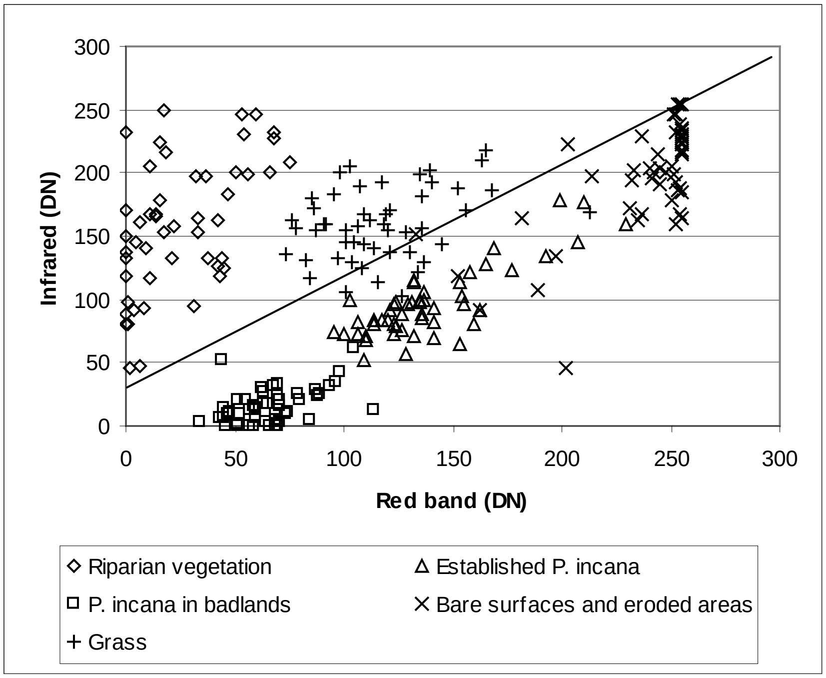


Figure 4.13 The relationship between the Red and Infrared bands for image 1 (sample image).

A clear spectral separability can be discerned from Figure 4.13 above. Distinct clusters of the respective surface cover types are unmistakably separated out. A clear progression from grass to riparian vegetation away from the soil line is noticeable. The few cases of grass spectral values that appear in the transition area of the soil line could be attributed to the 'mixed pixel' problem, where some bare patches intersperse grass areas. All the *P. incana* classes appear below the soil line owing to the considerable bare areas between individual *P. incana* plants. The slope and gradient of the soil line were calculated as 0.4376 and 74.27 respectively, such that:

$$NIR = 0.4376Red+74.27 \qquad (4.6)$$

Given the two values, the PVI was calculated using equation 4.5 above and Figure 4.14 below illustrates the resultant image.

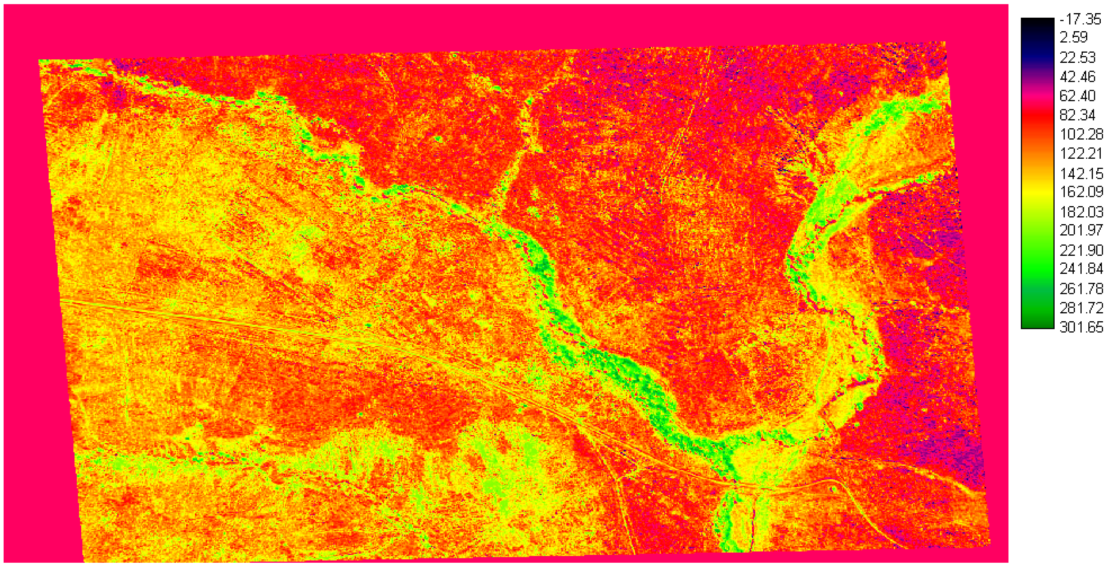


Figure 4.14 The PVI image of the classified scene, as shown in Figure 4.6 above.

When juxtaposed with the classified image (Figure 4.7a), the PVI image succeeds in discriminating the two densities of *P. incana*: the established *P. incana* and badlands *P. incana*, and bare surfaces and gullies. Contrary to the observation by Huette (1988), pointed out earlier, the index minimises soil noise significantly and gives an indication of vegetative cover independent of the effects of the soil. It can thus be said that under conditions of the present study area, where *P. incana* is interspersed with considerable bare surfaces, the PVI performs better than the other vegetation indices.

The selected images for analysis in this study represent different *P. incana* invasion scenarios in terms of density and stage of invasion. The efficacy of the red-infrared band relationship to discriminate *P. incana* densities, bare surfaces, gullies and other surface cover types cannot be determined on only one test image. There is a need to carry out a holistic analysis of the spectral characteristics of *P. incana* and the other surface cover types by way of examining the red-infrared band relationship across the range of *P. incana* invasion scenarios. This will give an indication of trends in spectral consistencies or inconsistencies of the HRV. To this end, the relationship was examined from the rest of the selected images representing a range of cover classes namely: Initial stages of *P.*

incana invasion in the form of isolated patches; invading *P. incana* with fairly extensive patches; established *P. incana* being full blown invasion of entire hillslopes; and *P. incana* in badlands, whereby gullied areas are invaded, such that there is advanced *P. incana* cover on the crests of gullied areas. Figure 4.15 and 4.16 below illustrate the details regarding the geographic locations of the 10 images and the red-infrared relationships respectively.

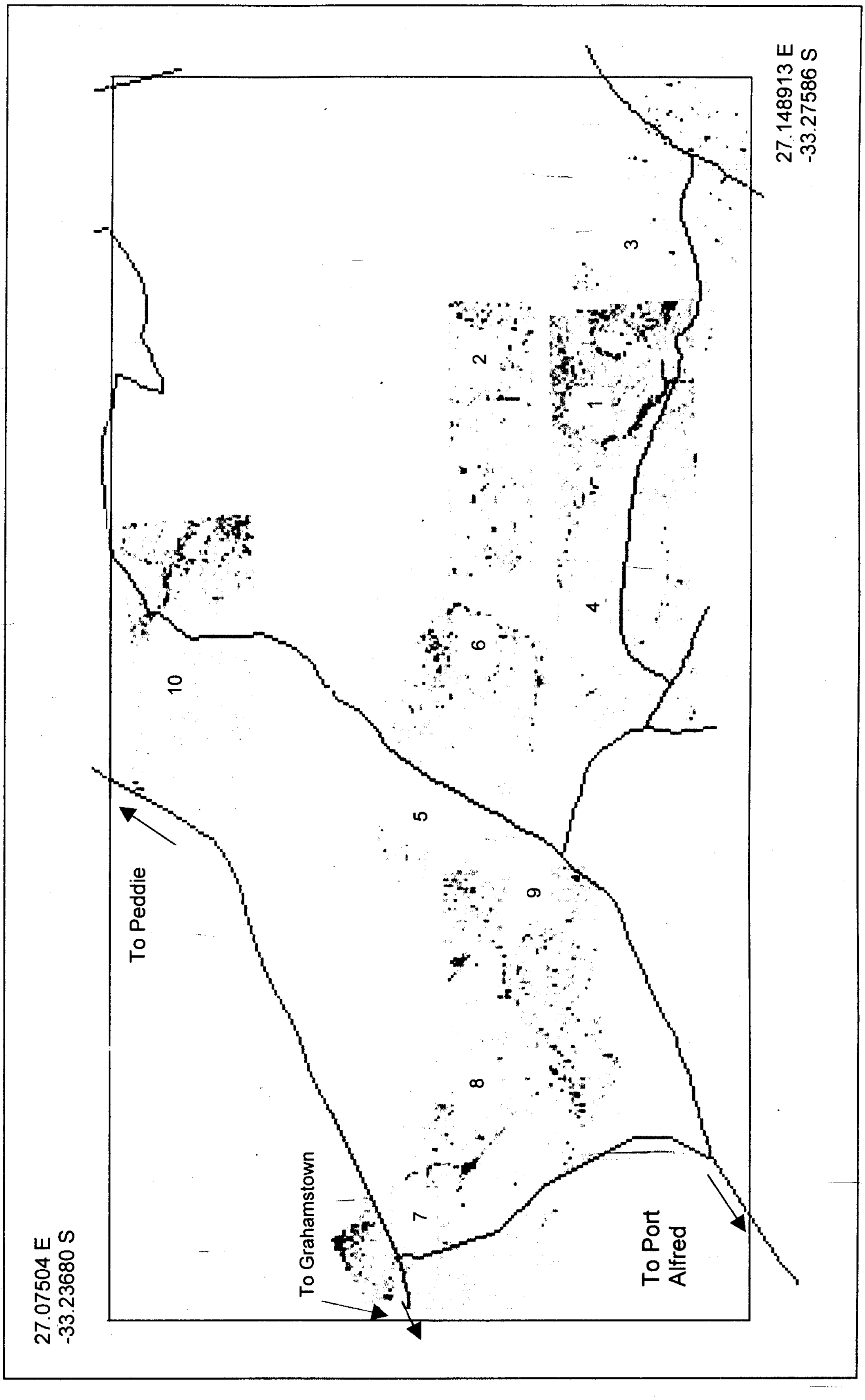
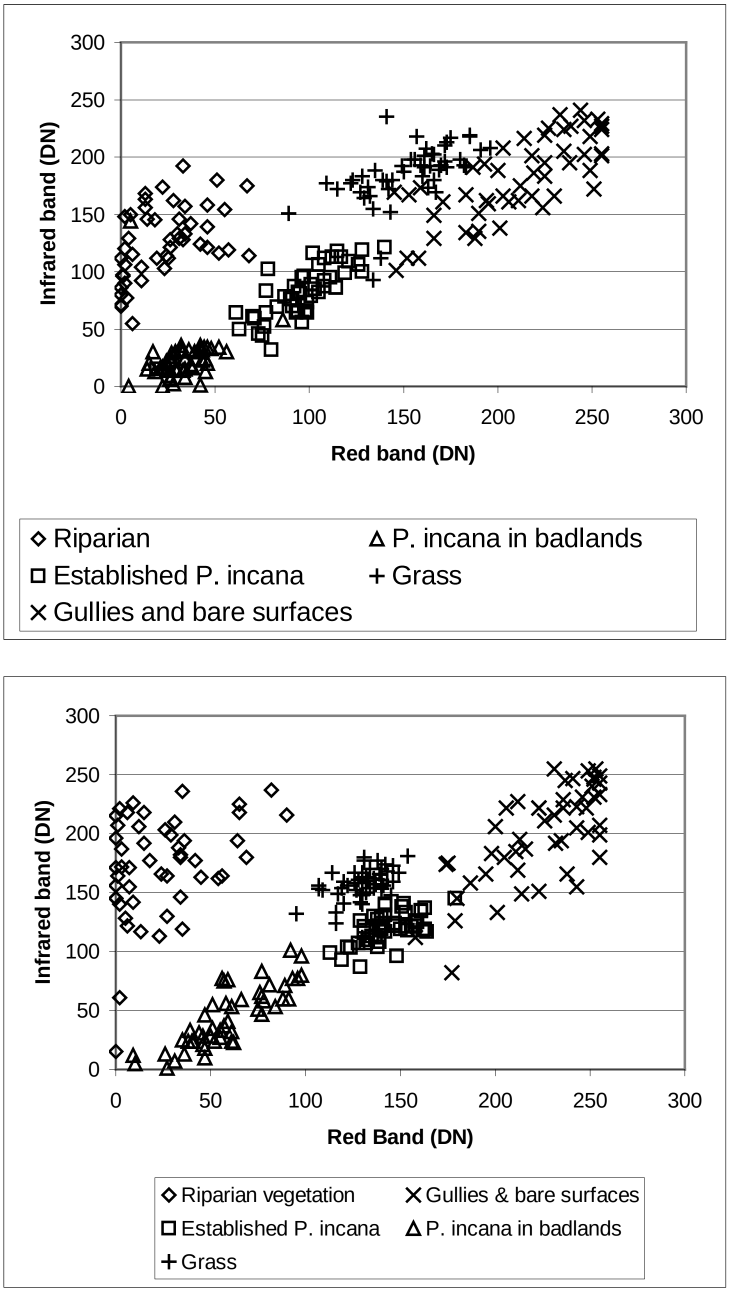
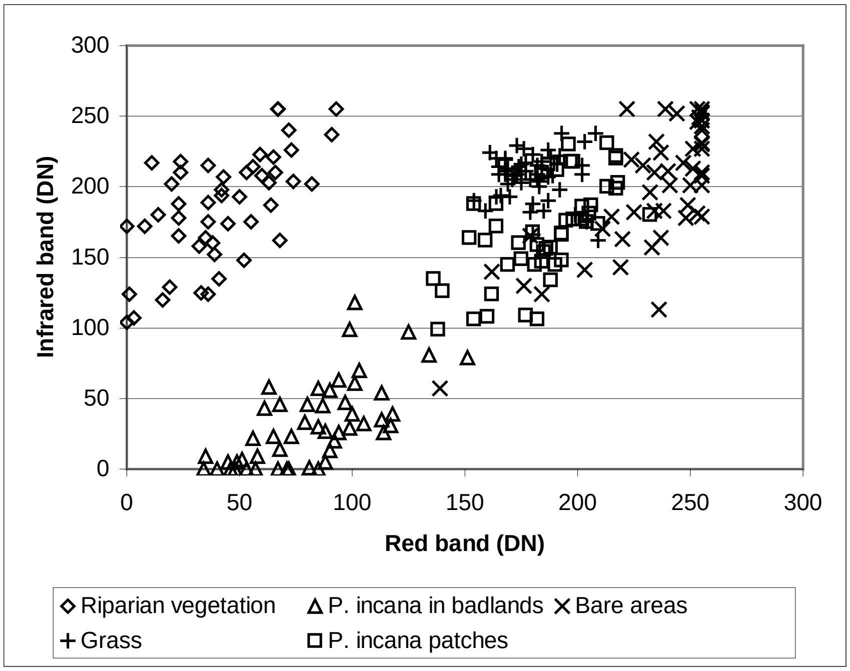
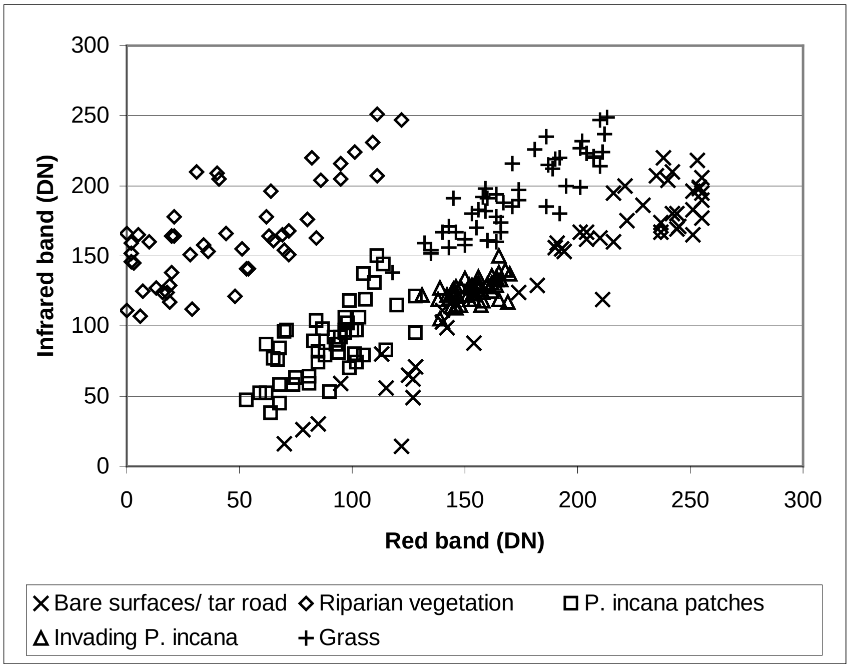


Figure 4.15 Geographic orientation of the 10 selected images – Numbered in the sequence of analysis. Roads vector file is overlaid.

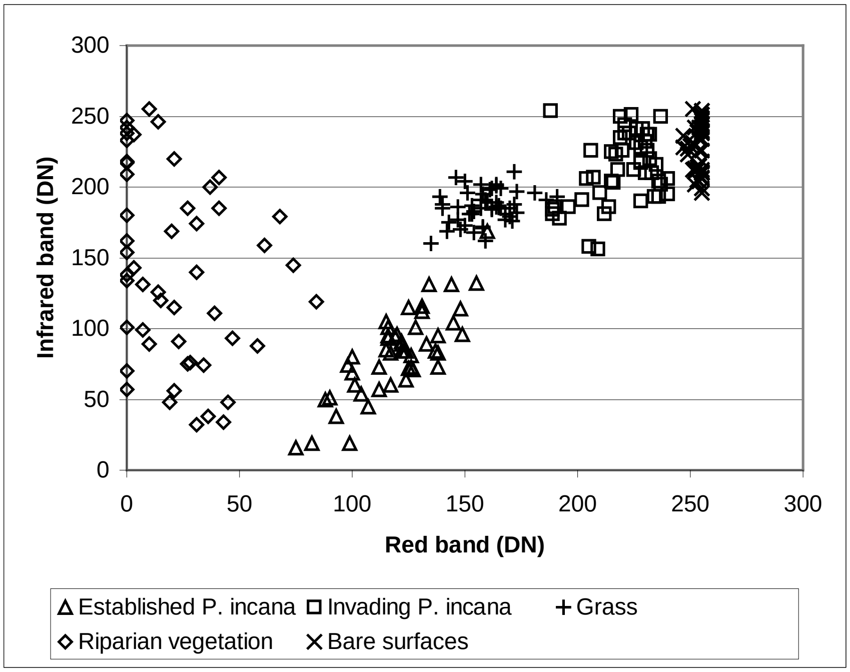


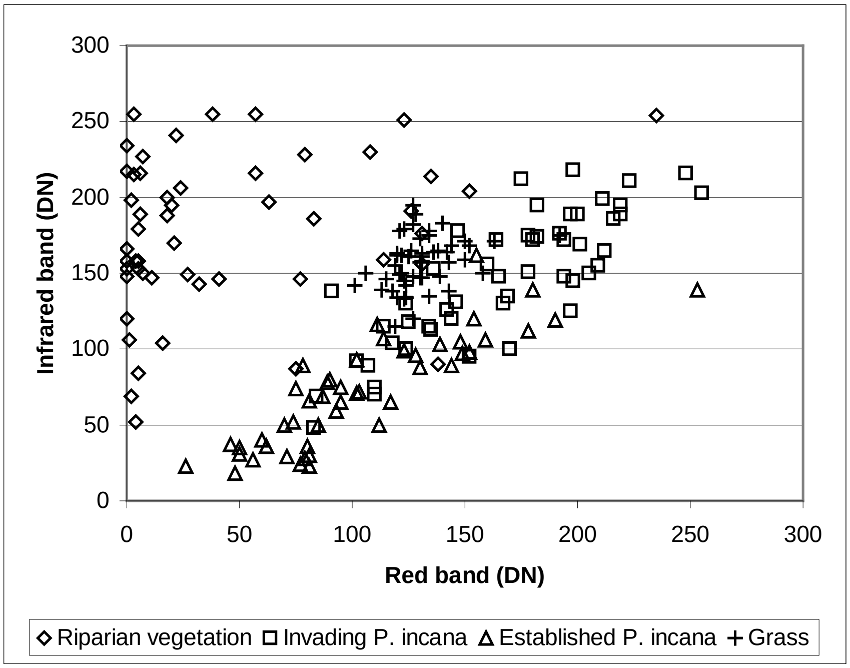
Figures 4.16a and b The Infrared – red relationship for images 2 and 3



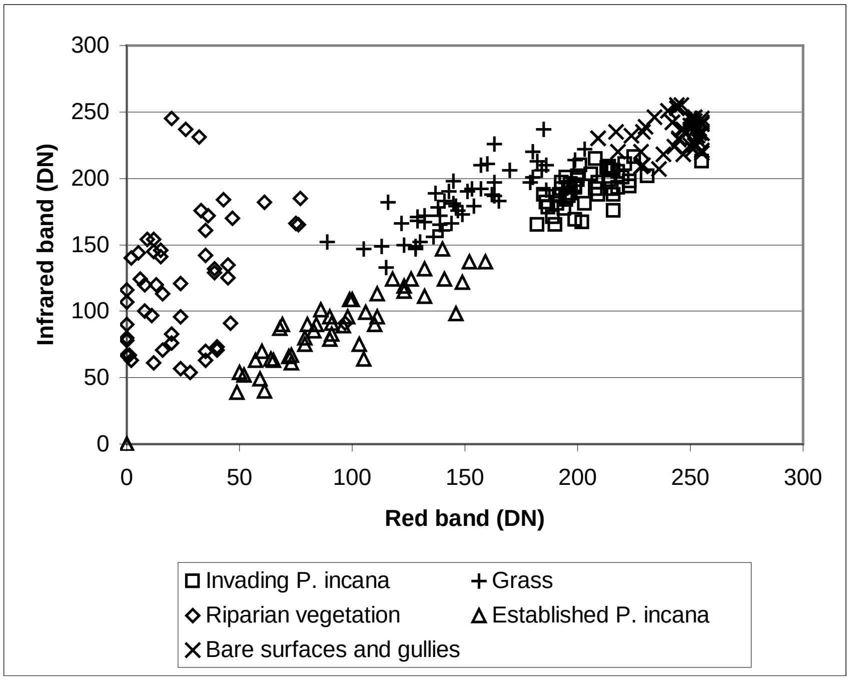


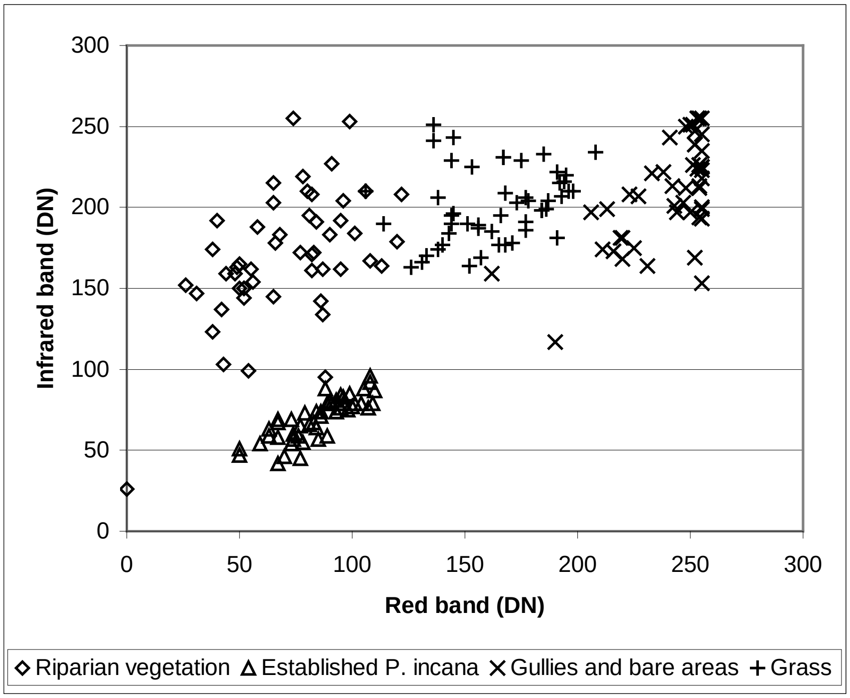
Figures 4.16c and d Infrared – red band relationship for images 4 and 5





Figures 4.16e and f Infrared – red band relationship for images 6 and 7





Figures 4.16g and h Infrared – red band relationship for images 8 and 9.

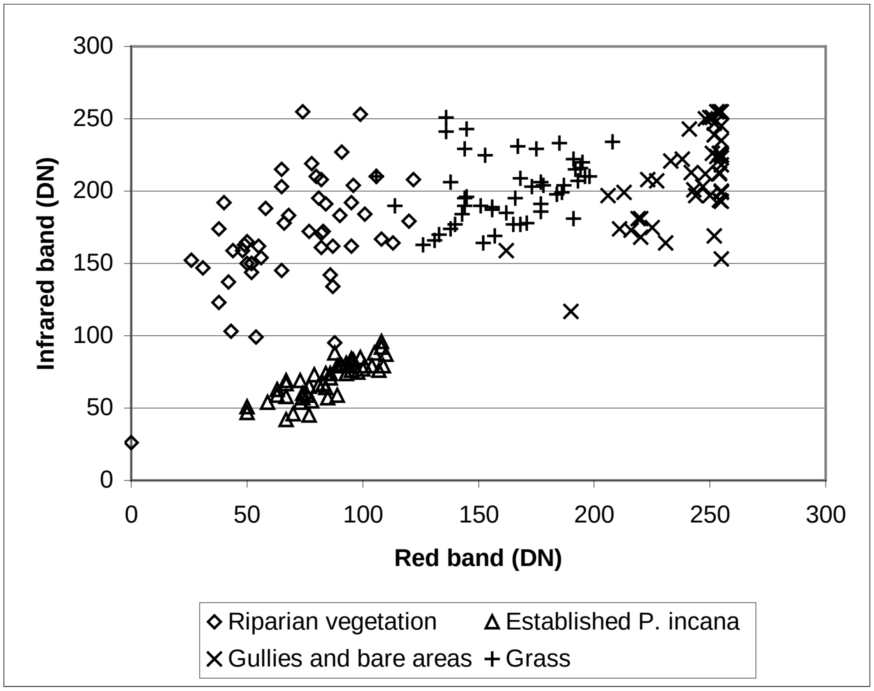


Figure 4.16i Infrared – red relationship for image 10

Judging from the trends in Figures 4.16a - 4.16i above, it can be said that red-infrared band relationships can be reliably used to discriminate surface cover types under the present and similar land cover conditions. The infrared band DN values for the respective cover types are typically consistent, such that riparian vegetation, grass and bare surface reflectancies are high. Riparian vegetation DNs are variable only in areas where this cover type is known to be in poor condition (e.g. Figures 4.16e and f). The low reflectance for the different *P. incana* densities is in keeping with earlier findings by Graetz and Gentle (1982) that perennial shrubs, for example saltbush in southern Australia, do not exhibit high infrared reflectance most of the time.

Consistent spectral separability of *P. incana* from the rest of the surface cover types is noticeable from all the scatterplots. In each case, unique spectral characteristics are associated with the different stages and nature of the invasion. Established *P. incana* assumes a more or less consistent position on the scatterplots and spectral range of 50-150DN, albeit the non-calibration of the images. *P. Incana* patches are noted to take variable positions on the relevant scatterplots. In some cases, an interface between the patches and grass exists (e.g. Figure 4.16c) while in the others, the two classes are separate (e.g. Figure 4.16d). This can be explained by the fact that in some areas, *P. incana* patches intersperse healthy grass cover, while in others, isolated patches do have some bare surfaces between them. The slight variability in spectral values of the other cover types can be attributed to a combination of factors ranging from veld condition, the choice of sampling points and the inherent Digital Camera inconsistencies explained earlier. Overall, distinct spectral separability exists in each case and consistent trends are discernible in terms of the separation of *P. incana* from the respective cover types as depicted by the scattergrams. Given the relative consistency of *P. incana*'s spectral characteristics, the images serve to provide a reliable geographical location relationship between the species and a range of topographical, geomorphic process and anthropogenic underpinnings.

In the next chapter, relationships between topographic variables, land use and *P. incana* occurrence are presented. After establishing the efficacy of the High Resolution Imagery to characterise *P. incana* distribution in this chapter, the selected images are then used in conjunction with the study area's Digital Elevation Model (DEM) to derive relationships between terrain and other related parameters and *P. incana*.

4.5 Conclusion

The HRI used in this study are noted to have limited constraints as evidenced by the classification accuracy assessment. Ratio based Vegetation Indices, including the NDVI, SAVI and MSAVI, do not to give a clear spectral separation between *P. incana* and bare areas and are therefore not appropriate for surface cover analysis in environments of incomplete vegetation cover. On the contrary, the PVI has been noted to provide distinct spectral separability between the different degrees of *P. incana* invasion, bare surfaces and the rest of the surface cover types. Despite a few limitations of the infrared images acquired with a digital camera, they can reliably be used to quantify and characterise general spatial and spectral analyses.

Chapter 5: *Terrain Parameters and Pteronia incana Occurrence*

5.1 Introduction

This chapter presents the use of a Digital Elevation Model (DEM) in conjunction with the images analysed in the preceding chapter to investigate the relationships between individual or a combination of terrain parameters and the occurrence and distribution of *P. incana*. This will provide a background to unravelling multi-tier relationships, including the role of topography-controlled variables like soil moisture in *P. incana* invasion. Terrain parameters are calculated from the DEM and statistical tests are carried out to determine the extent to which topographic variables predict the presence or absence of *P. incana*. The relationship between *P. incana* density, slope direction and angle is also analysed. Links between terrain parameters and land use as an important co-parameter in *P. incana* invasion are also investigated. Other topography related parameters deemed to have an influence on the spatial distribution of *P. incana*, particularly at the hillslope scale, are also calculated from the DEM. The implications of *P. incana* density for runoff and erosion are highlighted.

5.2 Implications of DEM resolution for topographic parameters

According to Zhang and Montgomery (1994), DEM resolution significantly affects the computed topographic parameters and related hydrological models. The representation accuracy decreases moderately at an intermediate resolution, but sharply at coarse resolutions. This observation is supported by Hickey (2000) who states that, with typically low resolution of DEMs, for instance the 30m for 7.5 minute USGS DEMs, micro-features critical for hydrological modelling are lost, resulting in significant error. As DEM resolution and accuracy increase, the landscape will be more accurately represented and topographically driven hydrologic models will approach actual values. If a DEM has a coarser resolution than the image used, it is necessary to resample the DEM to match the resolution of the image, especially in places of very steep relief

(Geosystems, 2002). In areas where slopes are not very steep, Geosystems recommend that the DEM resolution can stay much lower than 2-3 times the image resolution.

A distinction is made here between resolution and pixel size or grid cell, as there is a tendency to mistakenly refer the two concepts interchangeably. Resolution is approximately 2.83 times the pixel or grid cell size (NOAA Satellites and Information, 1999). Thus, the image pixel or grid cell must be multiplied by 2.83 for purposes of comparison with the DEM resolution. In the present study, an intermediate resolution DEM of 20m was used for the derivation of slope and aspect topographic parameters. This was the finest resolution available commercially. As can be noted from the slope frequency graph of the area (Figure 3.2), gentle slopes constitute 70% of the entire catchment. As per the recommendation above pertaining to gentle slopes, a DEM lower than 3 times the image resolution would suffice for slope and aspect calculation. In order to corroborate the appropriateness of the 20m DEM used in the present study, it was compared with field slope angle and aspect data collected during the broad catchment survey referred to in Chapter 3, section 3.3. Slope angle and aspect for all fifty sampling locations chosen from the field data were correctly predicted by the DEM.

It is noteworthy, however, that it is the work at hand that determines the selection of DEM size. When it comes to deriving micro-topographical features like small depressions, then manipulating the DEM to a finer resolution may become necessary. As pointed out earlier, finer topographic details may be lost at progressively larger DEM grid spacings.

5.3 Topographic Control of P. incana Occurrence

5.3.1 *P. incana* presence / absence

In order to establish a generic relationship between the occurrence of *P. incana* and slope angle and direction, 80 sampling points representing the presence or absence of *P. incana* in equal proportions were randomly selected and digitised from each of the 10 classified images. The sampling points were converted to raster image masks by means of the POINTRAS module in Idrisi. Windows of the same dimensions as the individual images were created from the DEM to facilitate the extraction of values files from them. Slope and aspect surfaces were calculated from the respective windows. Using the image masks as the feature definition files, values files representing the *P. incana* dichotomy were extracted from slope and aspect images. Out of the total of 800 sampling points digitised, Idrisi managed to successfully extract 783. The values files were readily exported to Excel spreadsheets where the data were assembled and *P. incana* absence and presence were coded as 0 and 1 respectively.

According to Quinn and Keough (2002), when a dependent model is categorical (with two levels 0 and 1), the Binary Logistic Regression (BLR) is best suited to model such response variables that are binary, e.g. presence / absence or dead / alive. This view is supported by SPSS Inc. (1999) who recommend the use of the BLR for situations in which one has to predict the presence or absence of a characteristic or outcome based on values of a set of predictor variables. It is suited to models where the dependent variable is dichotomous and hence discrete (0, 1). The BLR does not assume linearity of relationship between the independent variables and the dependent, and its success can be assessed by looking at, among other things, the classification table showing correct and incorrect classifications of the dichotomous dependent variable (Garson, 2002). There is a need to check for collinearity - an undesirable situation where the correlations among the independent variables are strong – before the BLR is run. A correlation of greater than 0.4 will lead to the inflation of the standard errors of the coefficients (Whitehead, 1998).

The SPSS version 10.0 statistical package was used to analyse the data. The *P. incana* dichotomy was entered as the dependent variable while slope angle and aspect as the independent variables. A very weak correlation ($R^2 = 0.0006$) between the two independent variables revealed non-collinearity between them (Figure 5.1). The point of departure was setting the null hypothesis (H_o) that there is no relationship between the absence or presence of *P. incana* and slope angle and aspect.

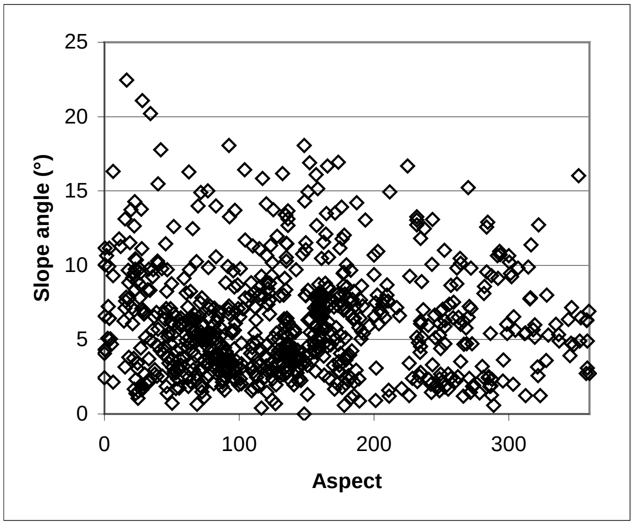


Figure 5.1 Non-collinearity between slope angle and aspect as independent variables.

In the present model, the "Percent Correct Predictions" (PCP) statistic is presented followed by the Maximum Likelihood Estimation (MLE) of the model parameters.

Table 5.1 below presents the PCP. The statistic assumes that if the estimated probability is greater than or equal to .5, then the event is expected to occur. The bigger the PCP, the better the model (Whitehead, 1998).

Table 5.1 Classification table for Absence and presence of *P. incana*.

***P. incana* absence / presence (cut off value is .50)**				
		Predicted		% Correct
		0	1	
Observed	0	291	100	74.4
	1	136	255	65.2
Overall percentage correct predictions				69.8

According to the classification table above, the model predicts 70% of the *P. incana* occurrences correctly. Implicit in this prediction is that slope angle and aspect do significantly influence the absence or presence of *P. incana*. The selected summary statistics of the MLE are further presented in Table 5.2 below.

Table 5.2 Summary statistics of the Binary Linear Regression.

Source: SPSS Output			
-2Initial LL	[1]	1084.082	
-2Ending LL	[2]	972.125	
Cox & Snell R Square	[3]	.13	
Nagelkerke R Square	[4]	.17	
	Chi-Square [3]	df	Significance
Hosmer & Lemeshow Test statistic (Chi-square)	32.742*	8	0.001
Overall model Chi-square Statistic = -2(I)LL - -2(E) LL	111.958[#]	2	0.001

* Critical value is 26.12, with 8 degrees of freedom at 0.001significance level.

[#] Critical value is 13.82, with 2 degrees of freedom at 0.001significance level.

The Model Likelihood statistic is the distributed Chi-square with degrees of freedom being the number of independent variables. The overall Chi-square is used to determine whether the overall model is statistically significant. As can be gathered from Table 5.2, the Chi-square is well above the critical value and hence statistically significant at the given level. Likewise, the Hosmer and Lemeshow Chi-square statistic is also well above the critical value, denoting statistical significance. The Cox & Snell and Nagelkerke R Square attempt to approximate the percentage variance explained in the regression. They do not have a sampling distribution and are therefore not amenable to testing. They are only descriptive measures of fit. On the strength of the Percent Correct Predictions and the Chi-square test statistics, the null hypothesis is rejected. Slope angle and aspect do have a strong influence on the presence and absence of *P. incana*.

In a dichotomous scenario, we model the probability that Y = 1 for a given value of X. The logistic regression predicts group membership as a sigmoidal probabilistic function for which the prediction switches from one group to the other at a critical value, specified as an inflection point where the probability (P) = 0.5 (Quinn and Keough, 2002). A least squares regression is not appropriate for binary data. Different authors (Whitehead, 1998; Garson, 2002; Quinn and Keough, 2002) instead recommend a logistic regression model. The logistic model is derived thus:

$$\pi(x) = (e^{\beta_0+\beta_1 x})/1+(e^{\beta_0+\beta_1 x}) \qquad (5.1).$$

Where β_0 and β_1 are the intercept and slope respectively. Given the categorical nature of the dependent variable (0,1), a log transformation of $\pi(x)$ (commonly known as the ‘logit’) is carried out thus:

$$\ln[\pi(x)/1-\pi(x)] \qquad (5.2).$$

The logit transformation is modelled against the predictor variable using the simplified equation:

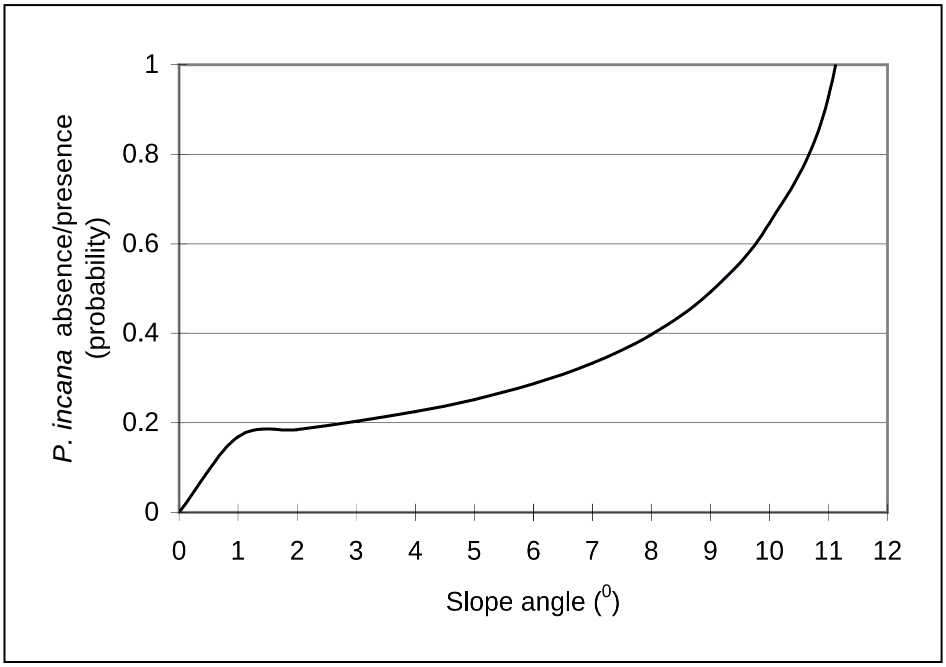
$$g(x) = \beta_0 + \beta_1 x_1 \qquad (5.3).$$

Where $g(x)$is the logit / natural log of the odds that $y = 1$ relative to $y = 0$. β_0 and $\beta_1 x_1$ are the constant (intercept) and the product of the regression coefficient (β_1) and the value of the independent variable. Whereas the $\pi(x)$ is constrained to between 0 and 1, the logit transformation values range between $-\infty$ and $+\infty$ (Quinn and Keough, 2002).

In the present model, the natural log of the odds that *P. incana* is present relative to being absent at a given slope angle and aspect was calculated. The logit coefficients in the SPSS output were provided as 0.178 for slope and 0.007 for aspect, with –1.982 as the intercept. The log odds ratios calculated using equation 5.3 were then translated back into predicted probability values using the equation:

$$P = 1/\ 1+e^{-(\beta_0 + \beta_1 x_1)} \qquad (5.4).$$

Figures 5.2 a and b below are logistic regression curves for slope and aspect respectively. The probability for *P. incana* presence or absence was calculated using formula 5.4 above.



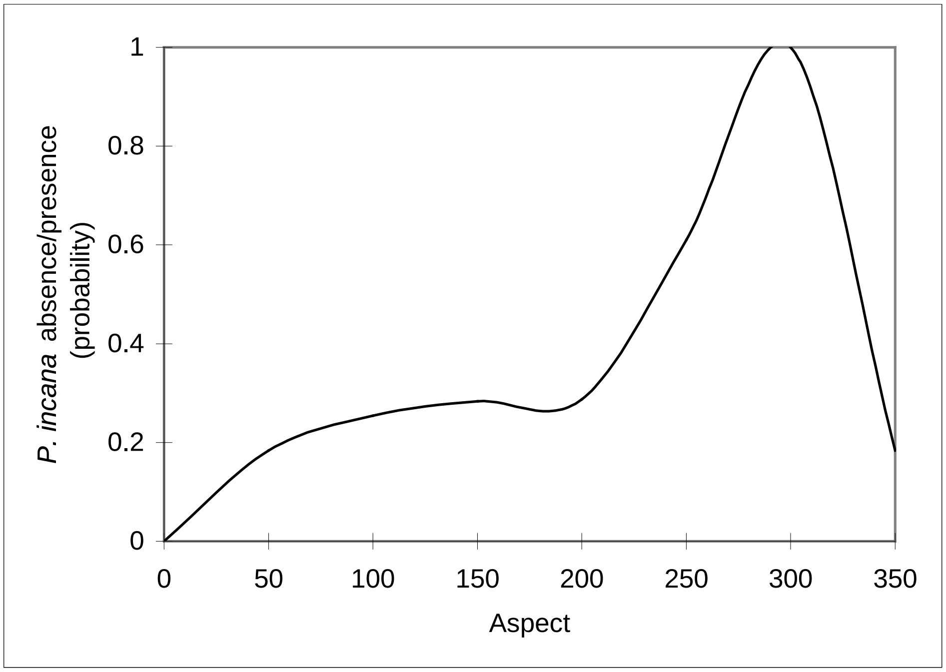


Figure 5.2 a and b: Logistic regression curves for slope and aspect.

As can be noted from Figure 5.2a above, the critical slope angle represented by the inflection point where P = 0.5 is about 9^0. This signifies the threshold slope for *P. incana* absence or presence. Likewise, south facing slopes (particularly SW oriented) are also identified in Figure 5.2b as the point at where the probability that *P. incana* is present is greater than that of its absence. There is a need however to obtain the finer details of the observed relationships as the regression model provides only a general overview of two topographical phenomena's influence. The subsequent subsections delve into the specifics of this relationship.

5.3.2 *P. incana* density variations with slope angle and aspect

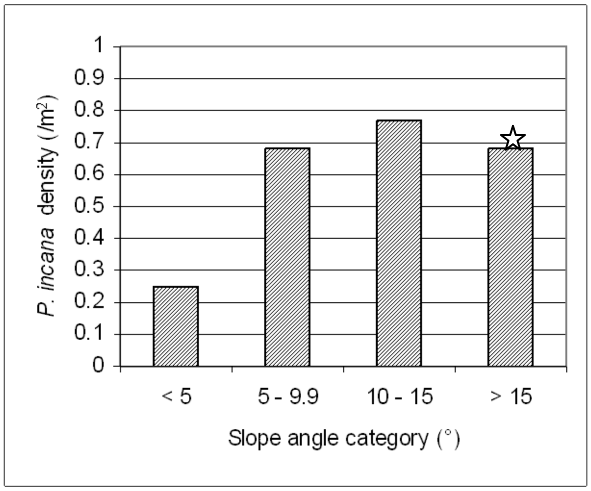
The presence/absence statistical test above only reveals that slope angle and aspect do have a statistically significant influence on *P. incana* occurrence. It thus provides a general picture of topographic controls on the species occurrence. It does not, however, reveal, let alone account for, the spatial variations in the density of the species with the different components of slope direction and categories of slope angle. There is a need to have a closer look at this relationship in order to establish whether the species do have preferential categories of slope angle and elements of slope orientation. Inferences can then be drawn as to the direct or indirect influence of the independent variables. To this end, *P. incana* density images were overlaid with slope angle and aspect surfaces following the procedure outlined below.

Boolean images representing the cardinal points of slope direction and categories of slope angle were prepared. Given that the DEM windows from which slope and aspect surfaces were calculated are not of the same rows and columns as the individual images, direct overlay of *P. incana* and terrain boolean images is not possible. Resampling the *P. incana* images to the DEM resolution was ruled out as it would lead to considerable loss of detail. This discrepancy was overcome by re-classing the slope and aspect surfaces into the desired categories using the RECLASS module in Idrisi. Polygons representing the outlines of the re-classed slope and aspect categories were digitised and rasterised using parameters copied from the *P. incana* images. It was therefore possible to directly overlay the rasterised aspect and slope masks with *P. incana* boolean images. The overlap zone of the *P. incana* and each image mask represented the area of a given slope category or aspect component covered by the species. The area in square metres of the overlap zones was calculated using the ‘Database query’ facility in Idrisi and recorded. This value was then divided by the area of the relevant image mask to obtain *P. incana* density. The term density is used here as a measure of the number of pixels with *P. incana* area rather than its degree of concentration. Given that the respective images represent varying *P. incana* invasion scenarios in relation to slope angle and aspect, *P. incana* density was calculated for all the 10 selected images. A *P. incana* density map was produced as a final product from each boolean image. Tables 5.3 and Figure 5.3 (histograms a-j) below present *P. incana* density variations with slope angle category.

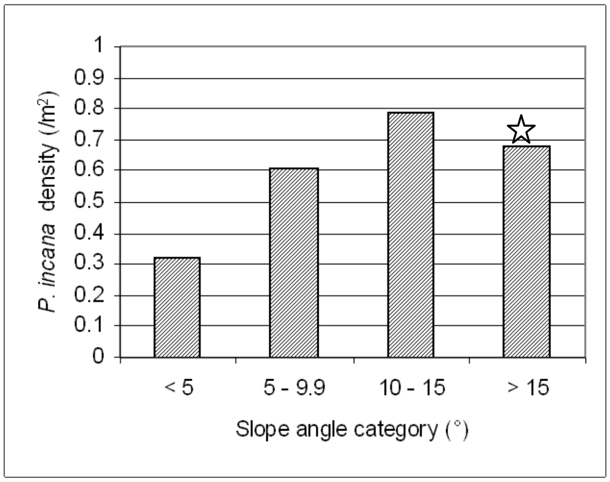
Table 5.3 *P. incana* density (m^2) variations with slope angle.

IMAGE	< 5	5 - 9.9	10 - 15	> 15
1	0.25	0.68	0.77	0.68
2	0.32	0.61	0.79	0.68
3	0.43	0.66	0.66	0.75
4	0.25	0.68	0.77	0.97
5	0.14	0.35	0.65	-*
6	0.46	0.5	0.6	0.89
7	0.27	0.37	0.5	-
8	0.28	0.37	0.52	-
9	0.34	0.38	-	-
10	0.004	0.014	0.36	0.51

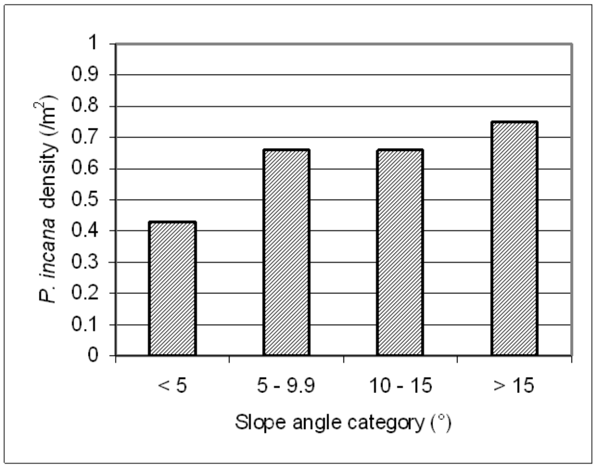
*Signifies absence of slope category.



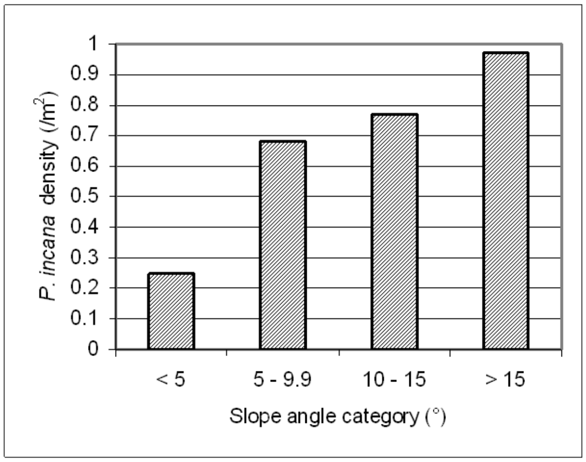
(a)



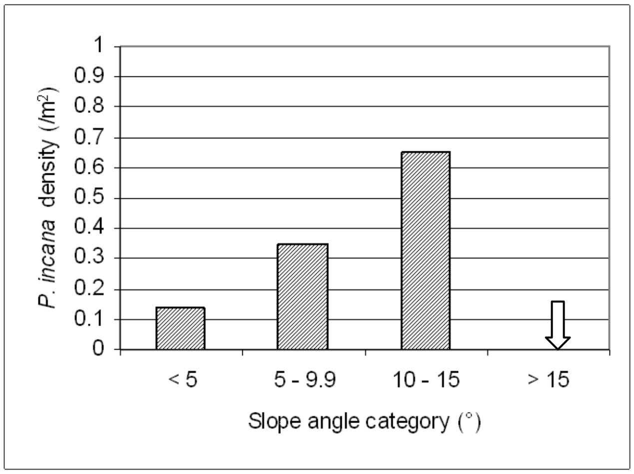
(b)


P. incana density (/m^2)
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
< 5
5 - 9.9
10 - 15
> 15
Slope angle category (°)

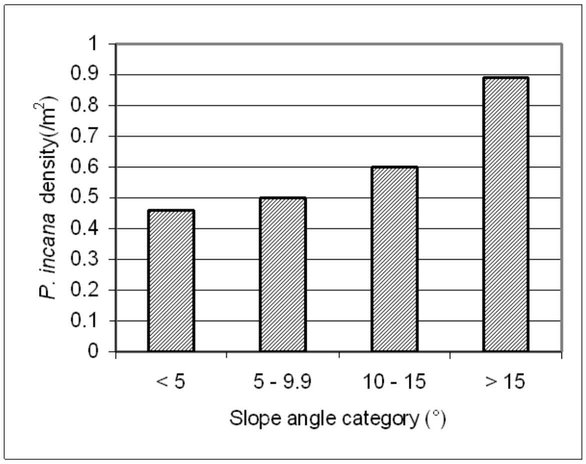
(c)


P. incana density (/m^2)
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
< 5
5 - 9.9
10 - 15
> 15
Slope angle category (°)

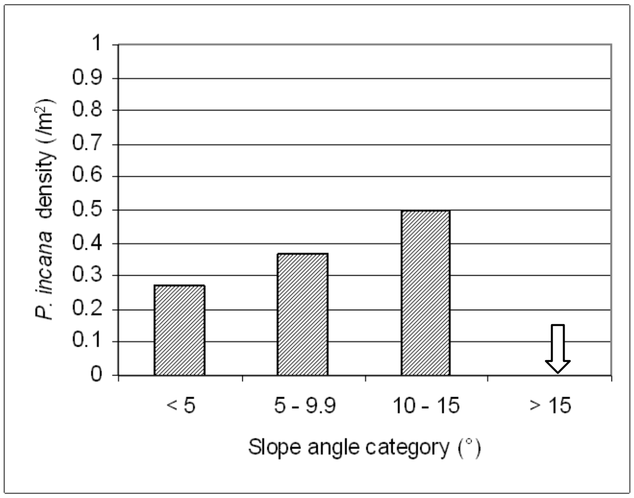
(d)


P. incana density (/m^2)
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
< 5
5 - 9.9
10 - 15
> 15
Slope angle category (°)

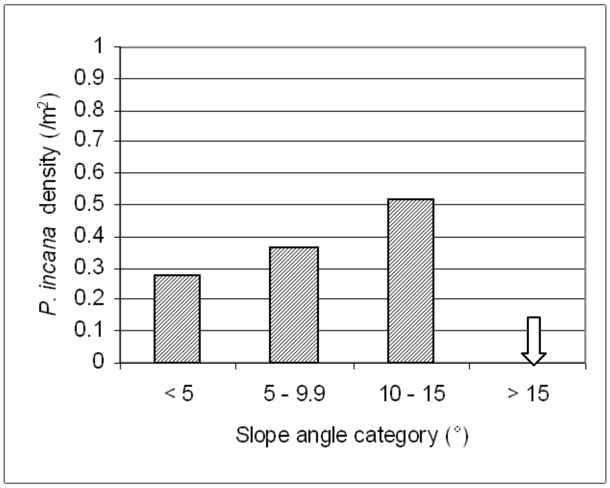
(e)


P. incana density(/m^2)
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
< 5
5 - 9.9
10 - 15
> 15
Slope angle category (°)

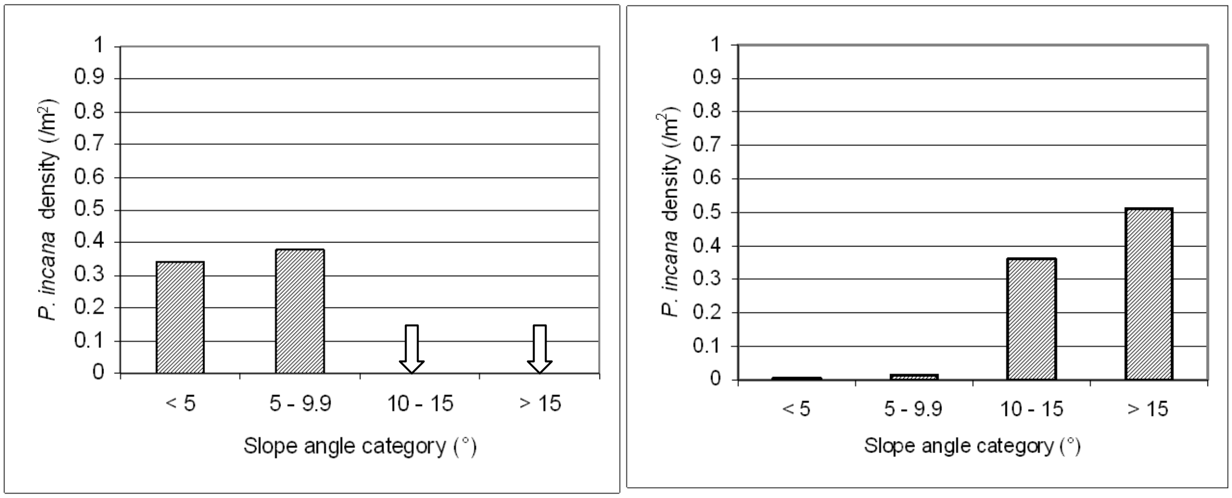
(f)


P. incana density (/m^2)
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
< 5
5 - 9.9
10 - 15
> 15
Slope angle category (°)

(g)


P. incana density (/m^2)
1
0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
< 5
5 - 9.9
10 - 15
> 15
Slope angle category (°)

(h)



(i) (j)

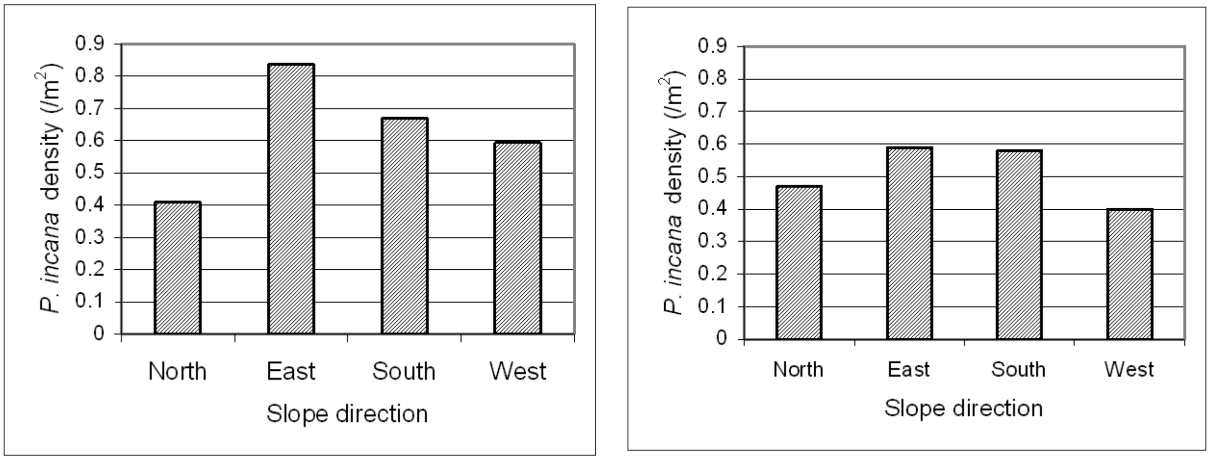
Figure 5.3 (a) – (j) Histograms showing *P. incana* density variations with slope angle categories for the respective images (1-10). Block arrows indicate absence of slope category in the area depicted by the image. ☆ Denotes considerable bare surfaces and erosion in slope category and hence lower *P. incana* density.

A clear trend that can be picked from the Figures above is that *P. incana* density increases with slope angle. The density is noted to be generally highest in the slope categories above 10^0. It can be gathered from Figures 5.3a and b (images 1 and 2) that the density in the $>15^0$ category is slightly lower than that of the preceding category. This is explained by the existence of severe forms of erosion and extensive bare surfaces at specific sites in this category for the two cases. The density is also noted to be extremely high for images 1 to 6 (Figure 5.3a to f). *P. incana* density of up to $0.9m^2$ /m^2 signifies cases of blanket invasion as opposed to images 7-10 (Figures 5.3g to j) where the density is just up to $0.5m^2$ /m^2 in the highest slope categories at these sites. The density is only between 0.3 – 0.4 in the lower slope categories for the latter group of images, except the $>15^0$ category for image 10. The absence of the $>15^0$ slope category is also noted in the latter group of images, save image 10. Field surveys and the slope surfaces derived from the DEM indicate that slopes of the latter group of images are predominantly gentle. A question that emerges from this observation is: does slope angle alone explain the observed differences in *P. incana* invasion and density? Certainly, there are sections of the catchment with slopes above 10^0 that are not invaded by *P. incana* or with a very low *P. incana* density. There exist site-specific invasion dynamics that cannot be explained in terms of only slope angle. It is necessary to examine its co-parameters in order to unravel its role in the *P. incana* invasion and density question. The influence of aspect as another slope parameter is examined next.

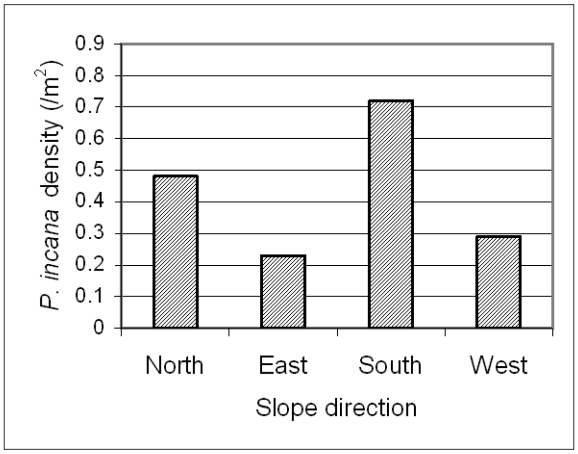
The slope direction for the hillslopes of the area was aggregated to represent the four cardinal directions (N E S W). Table 5.4 and Figure 5.4 below present *P. incana* density variations with slope aspect.

Table 5.4 *P. incana* density (m^2) variations with slope aspect.

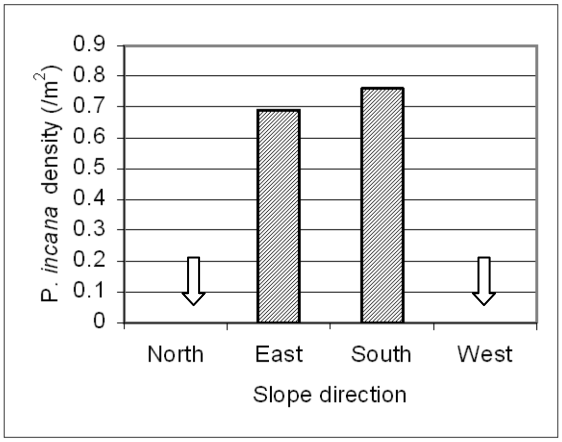
IMAGE	North	South	East	West
1	0.41	0.67	0.836	0.593
2	0.47	0.58	0.59	0.4001
3	0.48	0.72	0.23	0.29
4	-	0.76	0.69	-
5	0.25	0.52	0.07	0.12
6	0.25	0.561	0.62	0.12
7	0.34	0.36	0.09	0.47
8	0.35	0.35	0.46	0.46
9	0.38	0.23	0.47	0.47
10	0.14	0.48	0.7	0.12



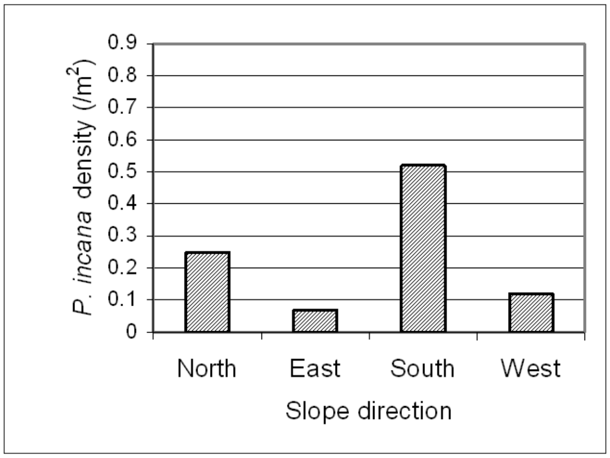
(a) (b)


0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
P. incana density (/m2)
North
East
South
West
Slope direction

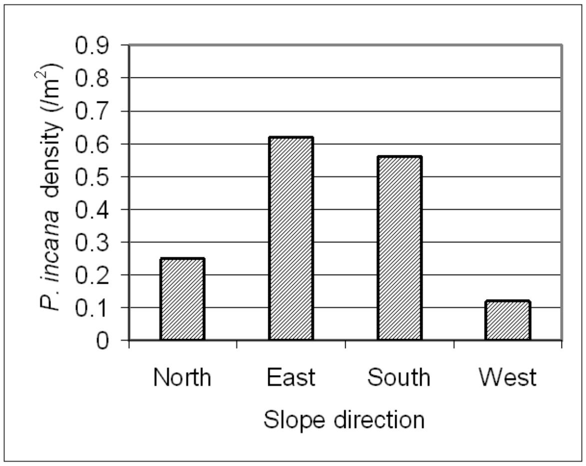
(c)


0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
P. incana density (/m2)
North
East
South
West
Slope direction

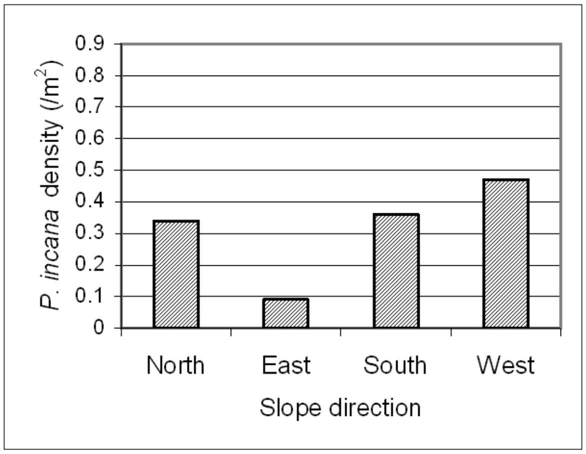
(d)


0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
P. incana density (/m2)
North
East
South
West
Slope direction

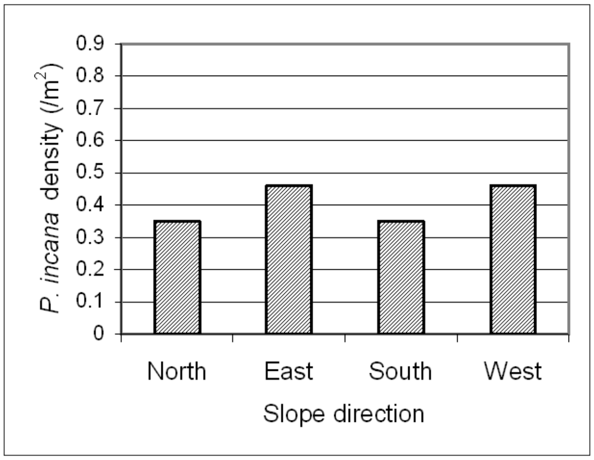
(e)


0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
P. incana density (/m2)
North
East
South
West
Slope direction

(f)


0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
P. incana density (/m2)
North
East
South
West
Slope direction

(g)


0.9
0.8
0.7
0.6
0.5
0.4
0.3
0.2
0.1
0
P. incana density (/m2)
North
East
South
West
Slope direction

(h)

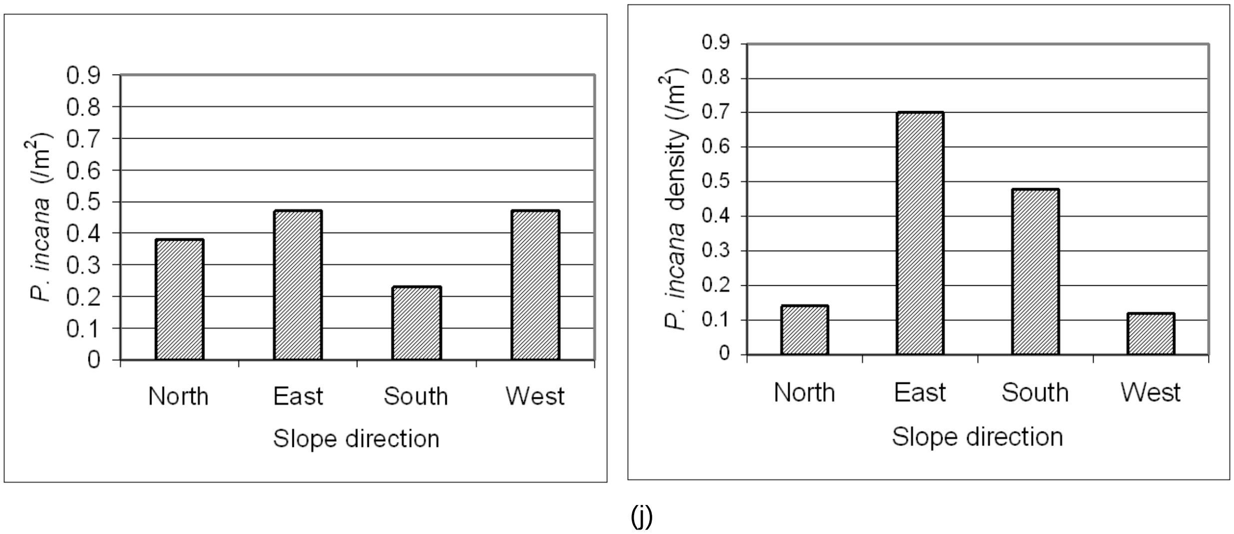


Figure 5.4 (a – j) Histograms showing *P. incana* density variations with slope direction for the respective images (1-10). Block arrows indicate direction of slope occupied by settlements.

It is noticeable that the areas depicted by Figures 5.4a - f have a higher *P. incana* density than is the case with Figures g - j. In the former group, slopes with a southerly orientation have the highest *P. incana* density (0.63/m^2 on average), followed closely by the easterly oriented ones (0.506/m^2). The latter group of Figures (g – j) with a lower density do not display a predominance of *P. incana* density in any slope direction. The area represented by Figure 5.4j stands out as having *P. incana* density on the east facing slopes that approximates that of the former group. The latter group of figures represents the area with gentle slopes referred to earlier in this section. Given the gentle nature of the slopes, topographic variations in terms of slope direction would not play a significant role in influencing *P. incana* invasion and density variations. Slope angle and aspect cannot be examined in isolation if *P. incana* invasion dynamics are to be clearly understood. There is a need to explore the relationship between the different co-parameters and the spatial variations in *P. incana* invasion. The combined influence of land use and slope parameters is examined in section 5.3.4.

5.3.3 *P. incana* invasion and land use

As pointed out in Chapter 3 section 3.4, preliminary field surveys indicated that the invasion by *P. incana* has strong links with land use change in the form of land abandonment and present grazing activities. In order to highlight and quantify the link between *P. incana* and past and present land use practices, shapefiles of abandoned and grazing land digitised from the 1988 orthophoto maps of the study area were imported into Idrisi from ArcView in vector format. When added as a layer to each of the *P. incana* boolean images, the imported vector files served to provide outlines for on-screen digitising of individual vector files for abandoned and grazing land on each image. The digitised polygons were rasterised and overlaid on the exclusively *P. incana* images. It was thus possible to determine from each image the area of abandoned and grazing land invaded by *P. incana*. This would not necessarily suggest a causal relationship, but it would give an idea as to the relationship between *P. incana* invasion and land use, and provide a background for further investigation from different standpoints. Table 5.5 and Figure 5.5 below illustrate this relationship.

Table 5.5 Extent of *P. incana* invasion on abandoned and grazing land on selected images.

Image no.	**Abandoned area (ha)**	**Abandoned area invaded**		**Grazing area (ha)**	**Grazing area invaded**	
		(ha)	**%**		**(ha)**	**%**
1	111	90	81	21	4	19
2	97	74	76	16	3.8	24
3	70	52	74	12	3.1	26
4	105	71	68	27	11.3	42
5	65	52	80	41	8.2	20
6	94	72	77	17	3.9	23
7	32	29	91	72	6.5	9
8	-	-	-	113	52	46
9	-	-	-	97	32	33
10	-	-	-	120	42	35

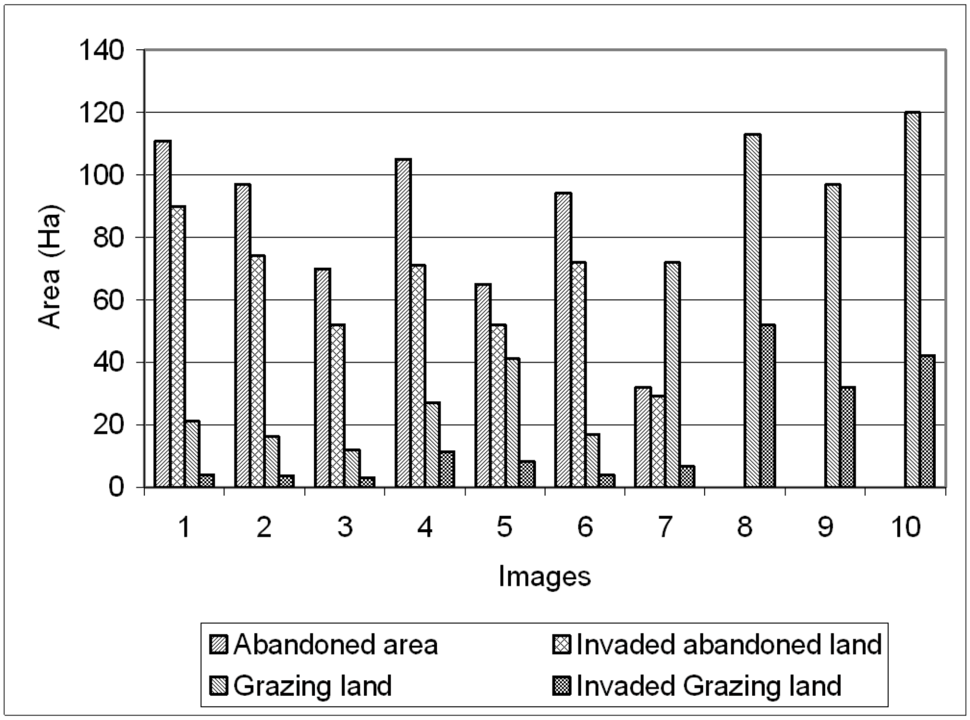


Figure 5.5 Area in hectares of abandoned and grazing land invaded by *P. incana*.

Figure 5.5 confirms that blanket invasion of abandoned lands by *P. incana* is a prominent phenomenon in the area. The small percentage of these lands not under *P. incana* cover is in most instances bare and gullied land. On the other hand, partial invasion on the grazing lands is noticeable from the figure. Abandonment *per se* may not explain the invasion. The interaction between land use, slope angle and aspect and the implications for *P. incana* invasion is examined in section 5.3.4 below.

5.3.4 Land use – slope angle and aspect interaction: Implications for *P. incana* invasion

Land use is an important co-parameter whose relationship with terrain parameters would go some way to unravel the invasion equation. Partial and blanket invasion of grazing and most of the abandoned lands respectively are noted in section 5.3.3 above. The strong spatial relationship between abandoned land and the invasion is highlighted. It would be useful to establish how much of the invaded abandoned and grazing lands are within the respective slope angle categories. Given that *P. incana* density within each slope category has already been established, a multi-pronged relationship would be identifiable. The relationship was readily achieved by overlaying the invaded abandoned and grazing lands with slope category boolean images as illustrated in Figure 5.6 below. The area of invaded abandoned and grazing land in each slope category was aggregated for all the images to obtain a broad picture of the land use slope relationship. Figure 5.7 depicts the acreage of invaded and remaining abandoned and grazing land in the respective slope categories.

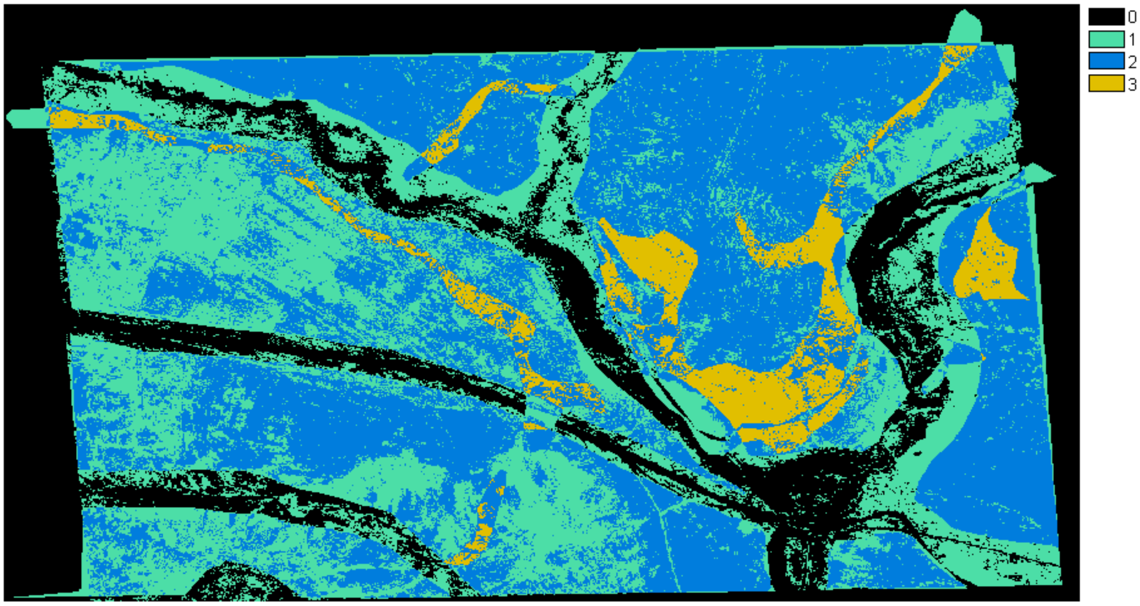


Figure 5.6 Overlaid layers: 2 = Invaded abandoned land; 3 = Invaded abandoned land in slope category 5^0 - 9^0; 0 & 1 = other surface cover types excluded from overlay.

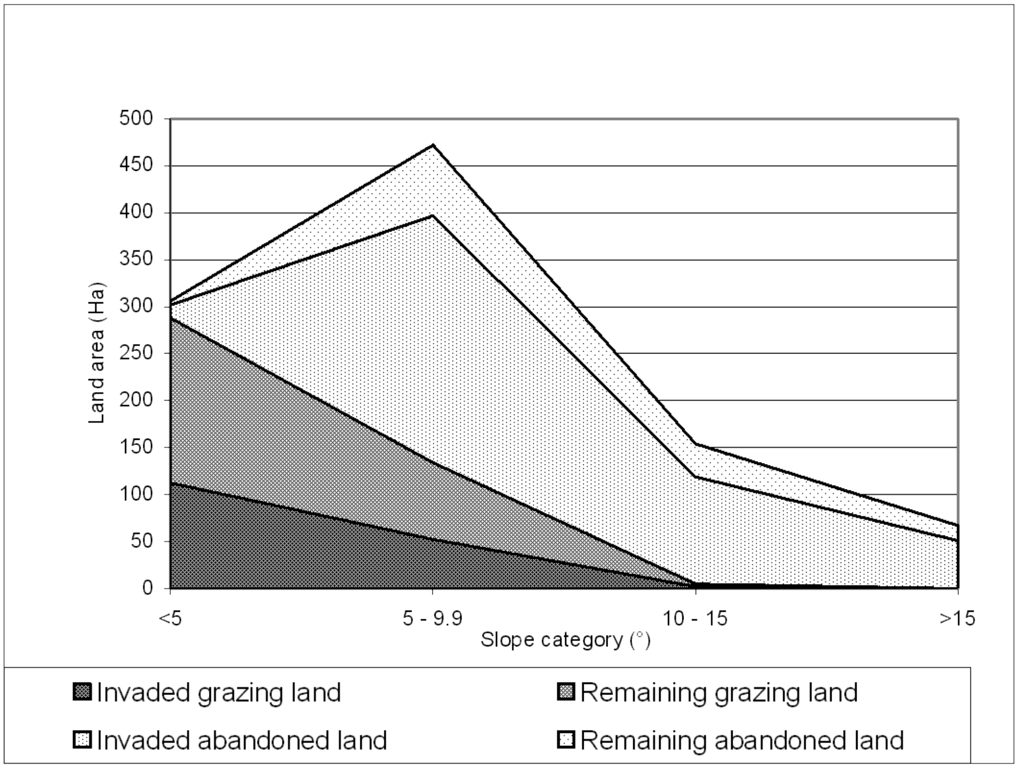


Figure 5.7 Invaded and remaining abandoned and grazing land per slope category.

Most invaded abandoned land area is in slope category 5^0-9.9^0 (Figure 5.7). A considerable land area is also noted in the 10^0- 15^0 and >15^0 categories especially considering that their frequency in the entire catchment is only 13% (see Figure 3.3, Chapter 3). It has been established that *P. incana* invasion is more pronounced in the latter slope categories as evidenced by the 'slope angle-*P. incana* density data'. By implication, the land abandonment – slope angle interaction has a strong bearing on *P. incana* invasion. Most of the grazing is carried out in the <5^0 category, particularly in the areas represented by images 7 –10 with low-density partial *P. incana* invasion (see Tables 5.3 and 5.4). This dichotomy invokes land use based area-specific management strategies, which will be discussed in Chapter 7. It also makes the abandoned lands - slope angle - *P. incana* invasion relationship all the more important in regard to geomorphic process implications. It is therefore these high-density *P. incana* areas that are the focus of process implications examined later in this chapter. The link between the terrain co-parameter of aspect and land use is explored next.

Figure 5.8 Invaded and remaining abandoned and grazing land in relation to slope direction.

Figure 5.8 shows the area of invaded abandoned and grazing land in relation to slope direction. It is evident that 71% of the invaded abandoned land is located on the south facing slopes. It also follows that the south facing slopes were by far the most farmed before abandonment, given that 77% of the previously farmed land is already invaded by *P. incana* (see Table 5.5 above). Invaded grazing land is distributed among the north, south and west slope directions. From the facts presented above, the following observations pertaining to slope angle and aspect on the one hand and the invasion of abandoned and grazing land can be made:

- Steep slopes in the category 10^0-15^0 and >15^0 have the highest *P. incana* density (Figure 5.3).
- *P. incana* density is highest on the south facing slopes (Figure 5.4).
- Invaded abandoned land with high *P. incana* density is predominantly located on steep south facing slopes with slope angles above 5^0 (Figures 5.7 and 5.8).
- Invaded grazing land with low *P. incana* density is located mainly on gentle slopes below 5^0 with a north, south and west orientation (Figures 5.7 and 5.8).

The cumulative influence of slope angle and aspect on the one hand and the impact of land abandonment and grazing on the other has promoted *P. incana* invasion. On the basis of the above findings, it can be said unequivocally that, in the area, disturbed lands in the form of mainly abandonment, particularly on steep south facing slopes, are highly susceptible to invasion by *P. incana*.

5.3.5 Local topographical control of *P. incana* spatial distribution and density

Local topographical variations were alluded to in Chapter 2, sub-section 2.3.3.4 as having strong implications for runoff redistribution. Concavities and convexities were noted as determining runoff convergence and divergence respectively. This in turn influences soil moisture distribution and vegetation pattern. Field observations of *P. incana* distribution from different vantage points indicate that its density is high on convexities and less or completely absent in concavities. The link between local topography and *P. incana* distribution and density is examined in this section.

The TOPOSHAPE and CURVATURE are surface analysis modules in Idrisi32 and derive local topographical attributes from a DEM. Before deriving such surface attributes at the finer hillslope scale, it is important to create a “depressionless” DEM by way of removing sinks or pits from it. Sinks in elevation data are most commonly due to errors in the data. The errors are often due to sampling effects and the rounding off of elevations to integer numbers (Singh and Engel, 1999). Naturally occurring sinks in elevation data with a cell size of 10 meters or larger are rare except for glacial or karst areas, and generally can be considered errors (Mark, 1988). Sinks were removed from the DEM windows of the same dimensions as the images using “PIT REMOVAL”. This is a new facility in Idrisi32 release 2 and Idrisi- Kilimanjaro and is an important addition to the software Hickey (2000).

TOPOSHAPE, which derives a classification of 11 surface shape features, was used to compare different surface shapes with *P. incana* occurrence. Initially, 100 point-features representing *P. incana* presence and absence were digitised from False Colour

Composites of the imagery and overlaid on the local topographical surfaces. Figure 5.9a below is one of the local topographical surfaces with the digitised point features and vectors for rivers and roads imported from ArcView overlaid. It is compared with Figure 5.9b, a false colour composite of the same area.

As can be discerned from the local topographical surface, *P. incana* is more established on convexities – ridges and convex hillslopes. It is less dense or completely absent from the concave and saddle sections of the hillslope. This corresponds well with slope configuration and *P. incana* spatial distribution on the False Colour Composite. A similar distribution pattern was noted from the rest of the imagery where the same exercise was conducted. Field observations corroborated this relationship. *P. incana* has a tendency to colonise spurs or hillslope convexities as opposed to the hollows or concave sections which remain as grass patch remnants. The underpinnings of this relationship will be examined in the discussion Chapter 7.

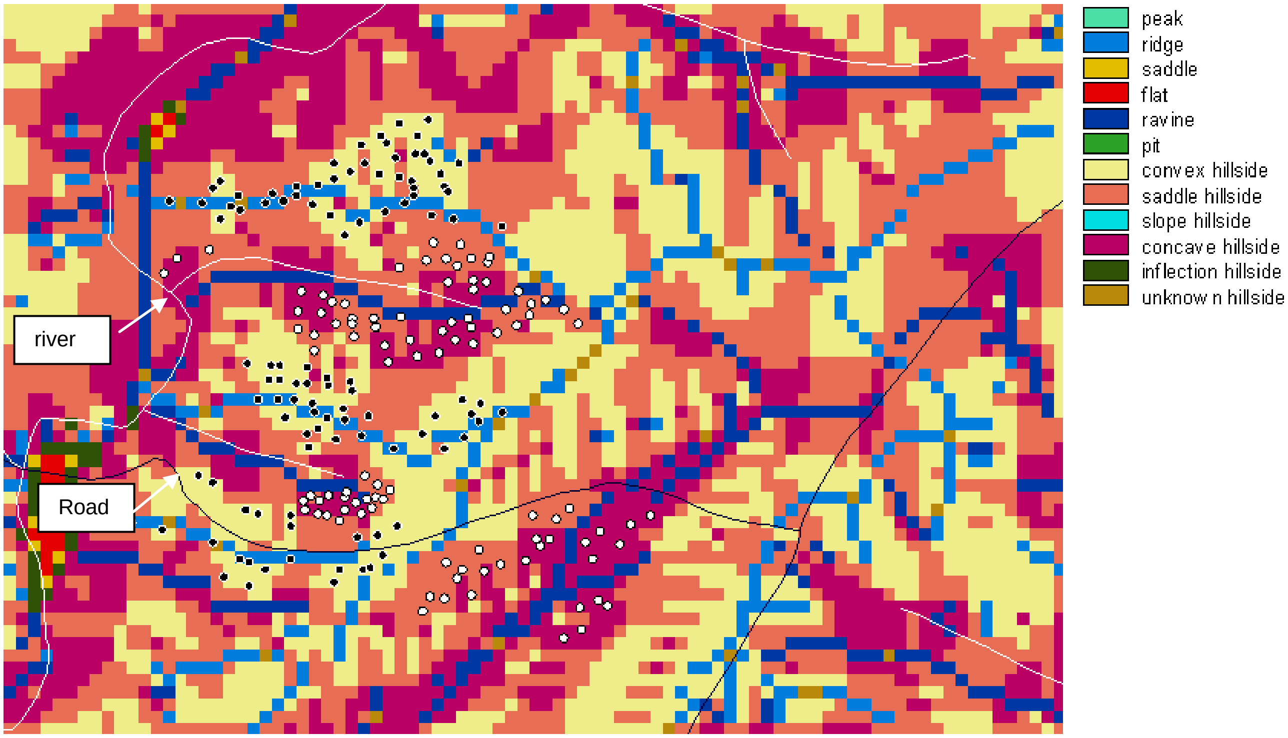


Figure 5.9a Local topographic surface derived using TOPOSHAPE. White dots (digitised from *P. incana* Boolean images) denote absence of *P. incana*. The reverse is true with black dots.

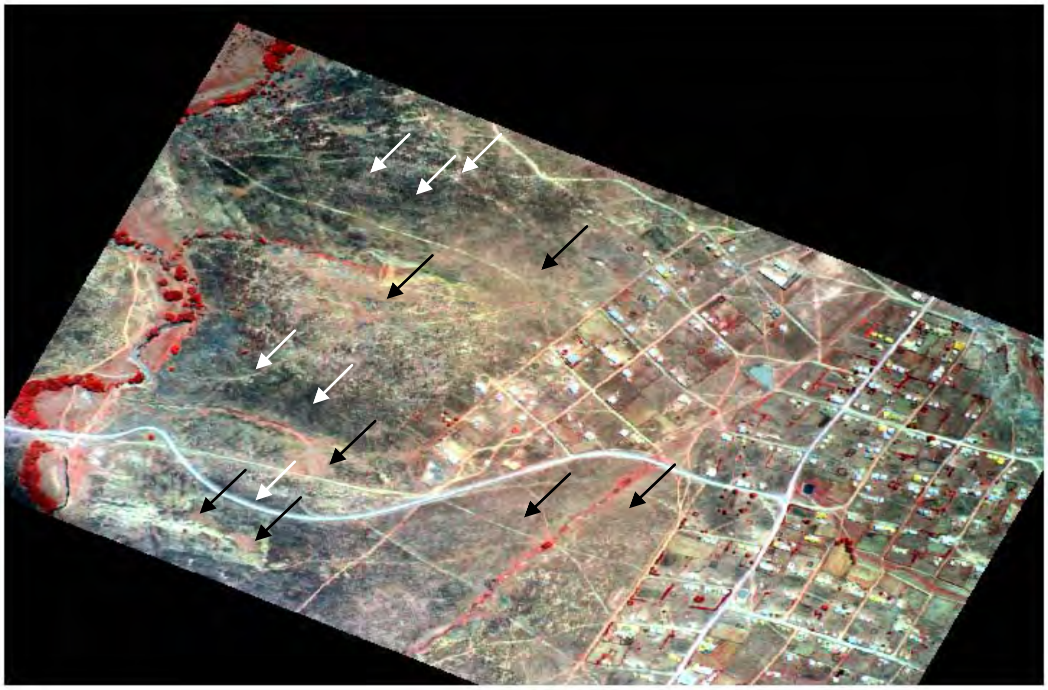


Figure 5.9b One of the False Colour Composites of the imagery. White arrows show convexities with a concentration of *P. incana*. Black arrows show concavities with grass patches appearing on the FCC as light red surfaces.

Quantifying the observed relationship using TOPOSHAPE derived surfaces would however, be a futile exercise as such surfaces are just classes without values. An attempt is made here to investigate this relative spatial correlation further by way of quantifying the local topography using "CURVATURE" and comparing it with *P. incana* density. The "CURVATURE" algorithm in Idrisi32 makes it possible to quantify for each pixel the maximum rate of change of a curve fit through the pixel along the direction of aspect (Idrisi 2001). It is evaluated in two orthogonal directions: the profile curvature measures how rapidly the surface is changing in the direction of the aspect while plan curvature measures this orthogonal to the aspect (Herrington and Pellegrini, 2000).

Curvature surfaces were derived from the DEM windows with the same dimensions as the individual images. Hillslope units with varying curvature, particularly conspicuous convexities, concavities, ridges and hollows were identified from the curvature surfaces and enclosed in polygons. The overall curvature of the enclosed units was determined by averaging the curvature values extracted from a number of digitised points deemed representative of the enclosed areas. The same polygons were then overlaid on *P. incana* density maps prepared as described in section 5.3.2. The corresponding density values were recorded.

Figure 5.10 shows the relationship between *P. incana* density and slope curvature as a surrogate for convexities and concavities in the local topography. The "CURVATURE" facility in Idrisi Kilimanjaro returns a negative value for spurs, ridges and convex elements of slope while the reverse is true for concavities or hollows. Despite the non-linear nature of the relationship, an unmistakable dichotomy is discernible between low density or no *P. incana* areas and high density areas. The grass patches and other areas with limited *P. incana* invasion are clearly distinguished by their positive eigenvalues, which denote concavities or hollows. On the other hand, *P. incana* clusters identified by their high density, tend to occupy convexities signified by their negative values. Some areas with *P. incana* density as low as a few isolated patches (see arrow in Figure 5.10 showing a case of just above 0.2m^2) are also noted to originate from convexities. These are perceived as some form of cradle from which *P. incana* spreads into the surrounding terrain, a phenomenon that has come to be known as patch-dynamics in the field of vegetation invasions (Thornes, *Pers. comm.*, 2000). Areas of greater concavities such as the riparian zone are noted from the images as largely devoid of the invasion. The implications of the local topography for particularly moisture distribution which in turn influences the pattern of *P. incana* invasion and the persistence of some grass species are examined further in section 5.3.7 where an index for wetness which is strongly linked to local topography is derived from the DEM.

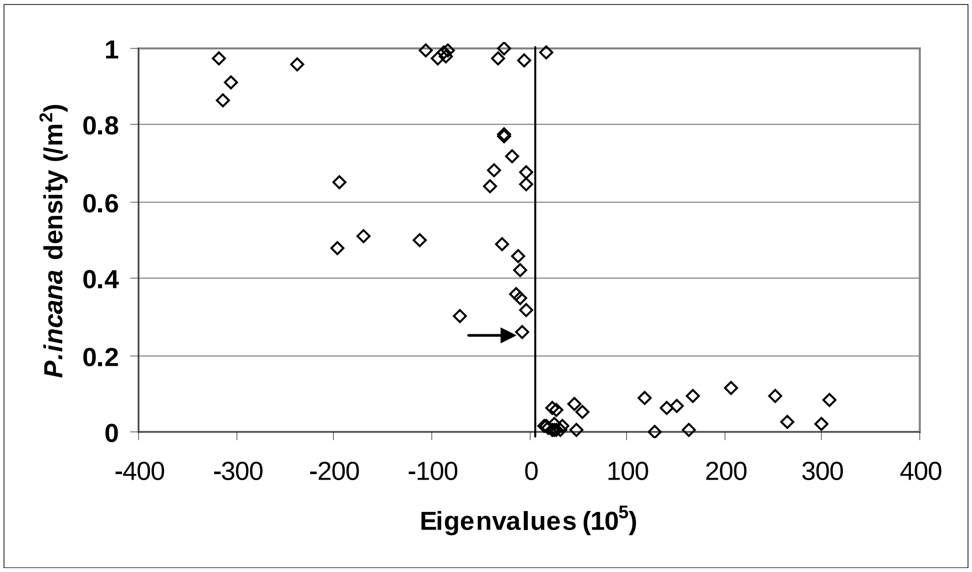


Figure 5.10 The relationship between *P. incana* density and slope curvature.

5.3.6 Derivation of the Wetness Index (WI)

The Wetness Index (WI), which is one of the major topographically driven derivations strongly related to spatial soil moisture variations, was calculated for all the images. The 'RUNOFF' module in Idrisi Kilimanjaro generates a 'flow direction grid' from each cell to its steepest downslope neighbour. Pit removal from the input DEM is an important pre-requisite. The one-directional D8-algorithm in the software (D for deterministic) determines flow to neighbouring cells on the basis of the steepest downhill slope within a 3X3 grid cell surface. The details of this procedure are explained by Singh and Engel (1999) and Asserup and Eklöf (2000). A Multi-directional flow algorithm incorporated in the 'Topoform' programme has also been developed by Pilesjö *et al* (1998). This option is not yet available in Idrisi. The output from the flow direction process is used to prepare a 'flow accumulation grid' where each cell is assigned a value equal to the number of cells that flow to it. Flow is traced from every grid cell and output cells with a high flow accumulation are the areas of concentrated flow, used to identify stream channels (Singh and Engel, 1999). The flow accumulation grid generated by 'RUNOFF' is the final product displaying a drainage network on the land surface, which is used to calculate the specific catchment area grid. The RUNOFF algorithm in Idrisi Kilimanjaro has been enhanced to incorporate precipitation and permeability data if available. The specific catchment area is calculated thus:

$$A_s = \text{flowaccumulation} * 20 * 20 / 20 \text{ (for cell size = 20 m)} \qquad (5.5)$$

The specific catchment area grid is combined with the slope surface to produce a WI image according to the formula:

$$\text{WI} = \ln(A_s / \tan \beta) \qquad (5.6)$$

where A_s is the specific catchment area, being the local upslope contributing area to a specific point in the catchment, and β is the local slope. Figure 5.11 below is a WI image derived from the same DEM window as Figure 5.9 above.

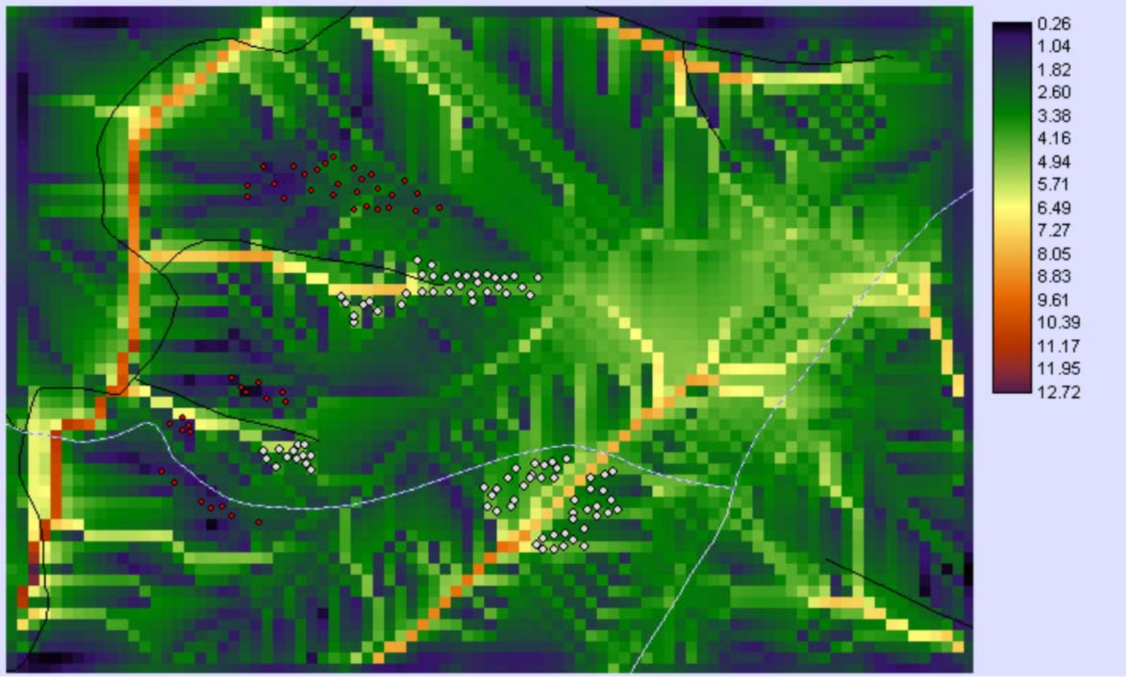


Figure 5.11 Wetness index surface derived from one of the DEM windows of the same size as the digital camera images. Red and white dots overlaid are *P. incana* and grass/'no *P. incana* clusters respectively. A road and main river vector files digitised from an orthophoto map are superimposed.

As can be gathered from the legend for Figure 5.11, the topographic WI is highest in the concavities bordering stream channels and decreases progressively towards spurs and other convexities. Wetness variations in micro- topographical features like hollows and other concavities are also brought out quite clearly. Having carried out extensive soil moisture surveys at a number of local topographical sites in the area depicted (see below), the WI image is deemed a fair representation of the local wetness conditions. Therefore, the need to manipulate the DEM to a finer resolution in order to derive more representative WI values falls away. More often than not, DEM manipulation generates the desired number of rows and columns, but does not add any details in terms of elevation values to the DEM (Drake, *Pers. Comm.* 2003).

In order to test whether the Topographic WI mirrors field moisture variability, WI values were compared with soil surface moisture conditions. Soil moisture data up to the depth of 7cm were collected in the field using a Delta-T Theta Probe ML2 moisture meter from 38 adjacent, relatively homogeneous grass patches and *P. incana* clusters identified from the images. Soil moisture conditions for the sites were determined by averaging moisture readings taken at each site on 8 field visits. The visits were conducted during the month of March 2002. Multiple visits were made for purposes of getting a consistent picture of the moisture variability. An additional 4 visits were made to the sites between the months of June and September of the same year, which received considerable amounts of winter rain. The latter visits provided data for the soil moisture variations during fairly wet conditions (see Chapter 6, section 6.7). The comparison presented in Figure 5.12 below is based on the data collected during the first phase of the visits.

A strong spatial relationship ($R^2 = 0.73$) between the WI and soil moisture (%) based on selected field sites can be noted from Figure 5.12. A clear divide between the moisture and wetness values between the grass and *P. incana* patches is readily noticeable. The latter can be identified in the lower cluster with low wetness and soil moisture values. The reverse is true of the upper cluster representing grass patches. This confirms that *P. incana* occupies dry areas as opposed to grass whose remanent patches persist in the wetter convexities and hollows. On the basis of this relationship, it can be inferred that, to a reasonable extent, the WI index is a proxy for soil moisture variability.

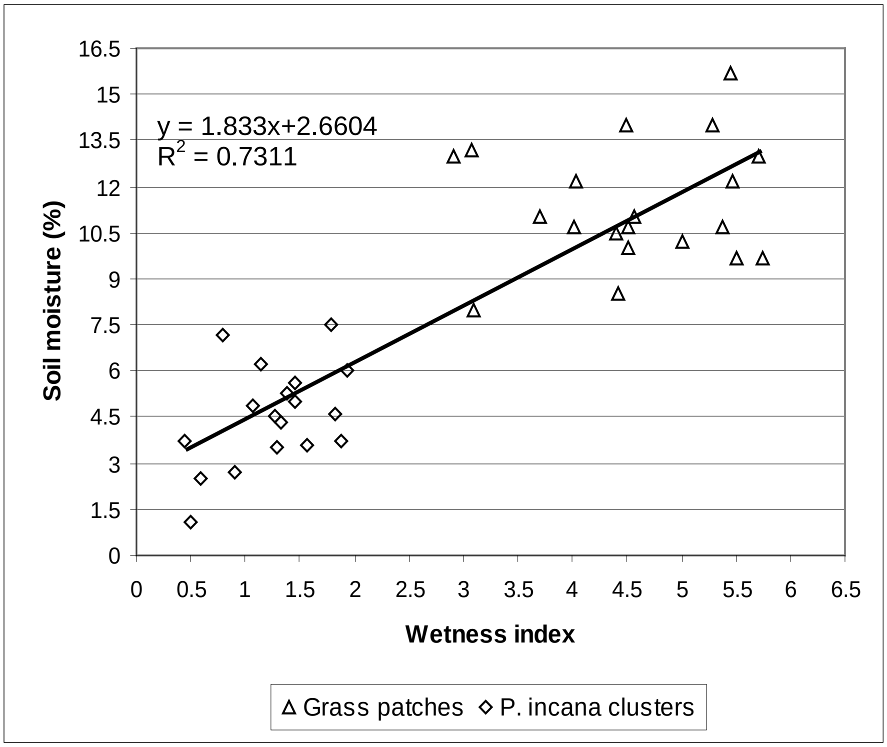


Figure 5.12 The relationship between the Topographic Wetness Index and soil moisture (%).

Initial tests of the potential relationship between the WI and *P. incana* density entailed digitising point features dichotomously on the species' boolean images, such that high and low density (usually grass patches) *P. incana* areas were separated. When the point features were overlaid on WI images (Figure 5.11), a dichotomy was discernible with *P. incana* clusters and 'no *P. incana*' or grass patches occupying low and high wetness areas respectively. This relationship was explored further by comparing the WI images generated from all 10 DEM windows of the same size as the digital camera images with *P. incana* density maps prepared for each image. Point features, 30 in number, were digitised on each of the WI images. Hillslope units with visibly low WI values on the one hand and high ones on the other were targeted. Obvious riparian areas were excluded. The points were rasterised and values files were extracted from the image masks. The same points were overlaid on the respective *P. incana* density maps and the density values of the polygons in which the points lay were recorded. Borderline cases were excluded from the analysis. Figure 5.13 is a scatterplot of the wetness values and *P. incana* density.

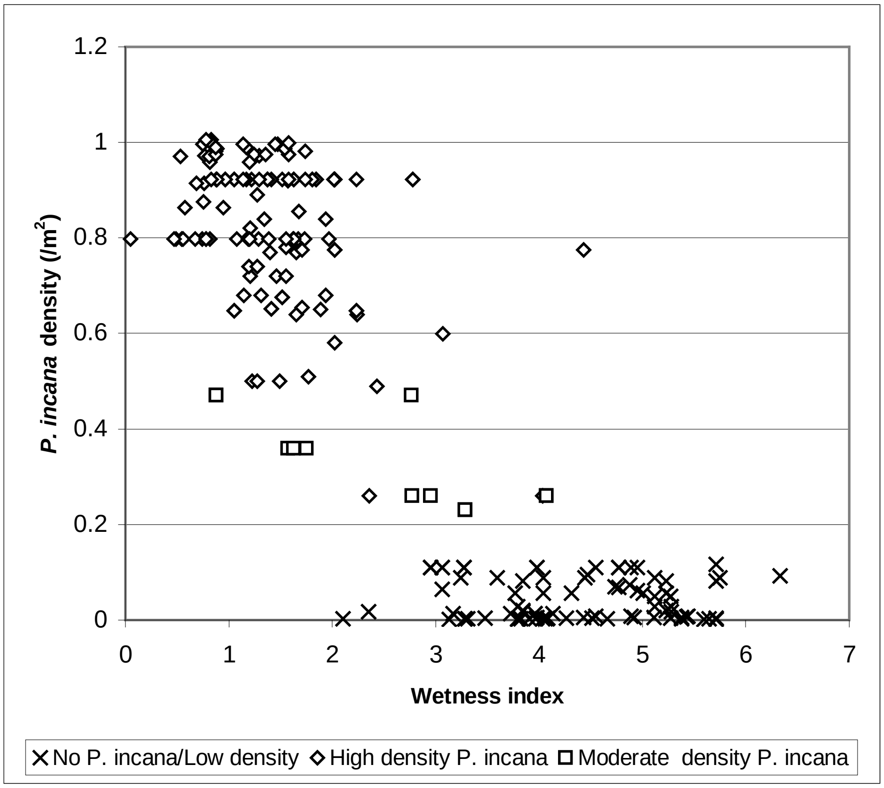


Figure 5.13 Scatterplot depicting the relationship between the WI and *P. incana* density.

A clear pattern is discernible from the Scatterplot, with high *P. incana* density (0.5 – 1 /m^2) associated with low WI values, concentrated between 0.2 and 2. The reverse is true of the low density or '*no P. incana*' areas, with WI values concentrated between 2 and 6.5. Areas with moderate *P. incana* density, have fairly high WI values of 1.5 – 4. In general terms, it can be said that the topographically controlled wetness as a surrogate for soil moisture variability has a strong influence on the spatial distribution of *P. incana* and that it is a strong indicator of the environments conducive for its invasion. It is certain that the shrub tends to avoid areas of high wetness, particularly hollows which are usually saturated areas. Little wonder that the shrub is absent in the riparian zones. A few cases of low wetness values with no *P. incana* and high *P. incana* density areas with fairly high wetness values are also noticeable. This indicates that besides the topographically driven

WI, some other factors, for instance soil properties, do play a role in determining the shrub's spatial distribution. The role of soil surface conditions, which in turn also influence soil moisture variability, will be examined in Chapter 6.

5.4 P. incana patchiness loss and implications for runoff

It was noted in section 5.3.2 that in the invaded areas *P. incana* density increases with slope angle. However, the question of density relates to the degree of invasion per unit area and dominance of *P. incana* over other species rather than percentage cover. Patchiness is one of *P. Incana's* inherent properties. The loss of patchiness and expansion of bare areas is an aspect of concern and has to be brought to light if the implications for processes like runoff and soil erosion are to be understood. To investigate this phenomenon, exclusively *P. incana* and bare surface images were prepared using images with a predominance of *P. incana* and bare surface mosaics. Figure 5.14 below is a zoomed in window to illustrate the mosaics. It can be noted from the Figure that bare surfaces (red blocks) appear as not interconnecting in some places. This is just because *P. incana* tussock canopies (yellow blocks) interlock, obscuring the connectivity of bare areas.

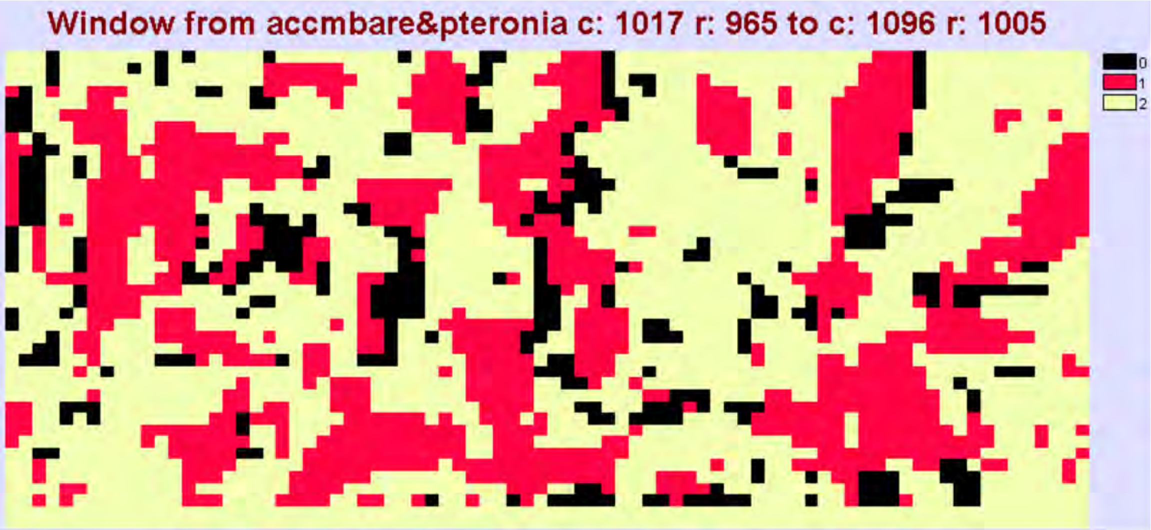
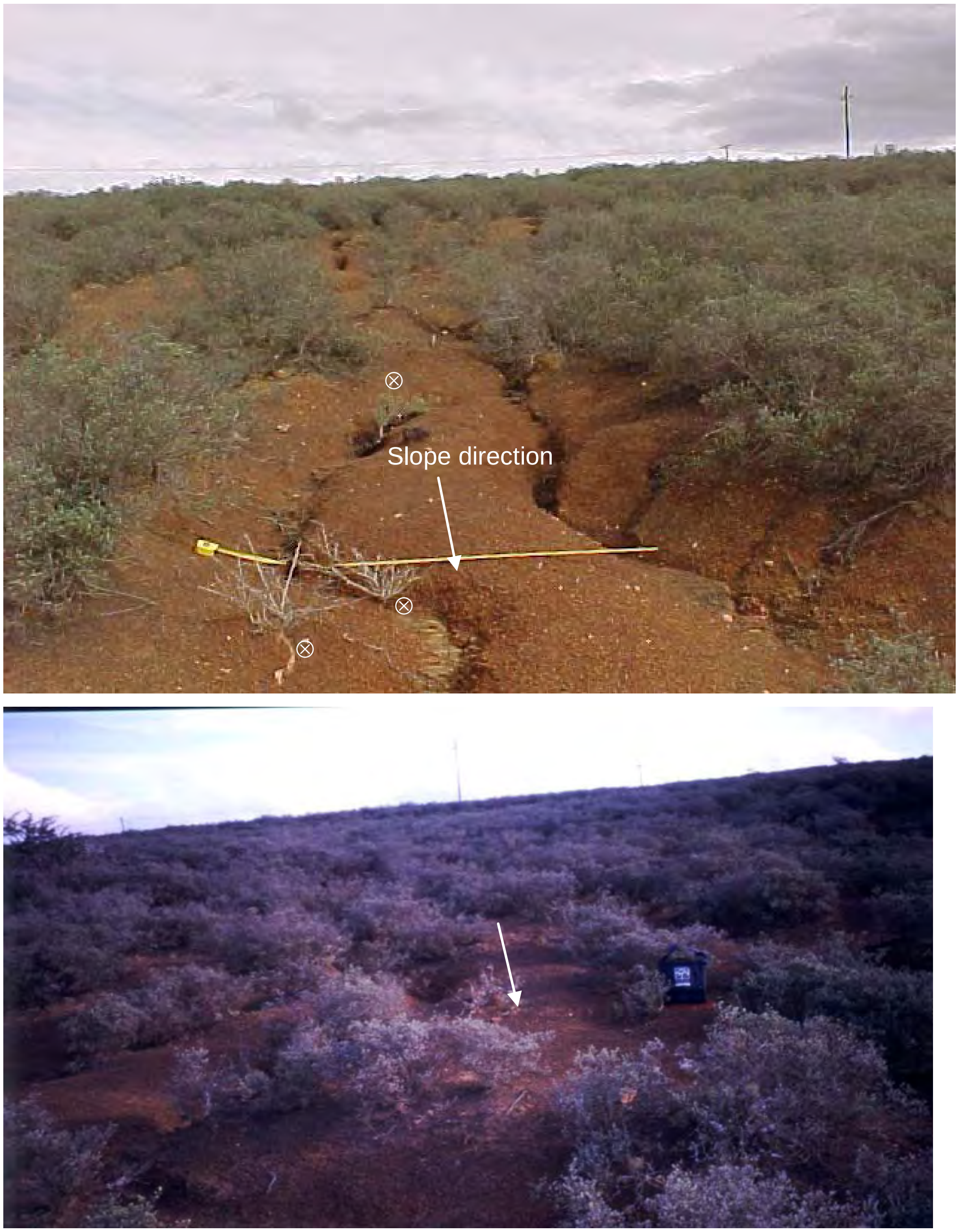


Figure 5.14 *P. incana* and bare surface mosaic window from a boolean image. 1 = bare surfaces; 2 = *P. incana*; 0 = other surfaces excluded from the boolean image.

Figure 5.15a below illustrates the loss of patchiness by *P. incana* under field conditions. Taken in June 2002, the picture displays considerable expansion of bare areas and erosion intensification as compared to Figure 5.15b, taken at the same spot two years earlier (July 2000).



Figures 5.15 (a) *P. incana* patchiness at one of the field sites. ⊗ Symbols show patchiness loss to erosion, (b) Picture of same site taken two years earlier.

The question that arises here is: how does *P. incana* patchiness loss manifest itself, particularly with slope angle variability? To answer this question, the ratio of *P. incana* to bare surfaces (P/B ratio) as an indicator of patchiness loss is worked out in the respective slope angle categories. The ratio was arrived at by extracting windows of 10mX10m from images representing areas where *P. incana* loss of patchiness is known to be a prominent feature. The slope category for the windows was determined by comparing them with slope images. For each window, the area in square meters covered by *P. incana* and bare surfaces was obtained using the 'Database Query' facility in Idrisi, making it possible to calculate the ratio. Figure 5.16 shows the P/B ratios of the image windows extracted from different slope categories.

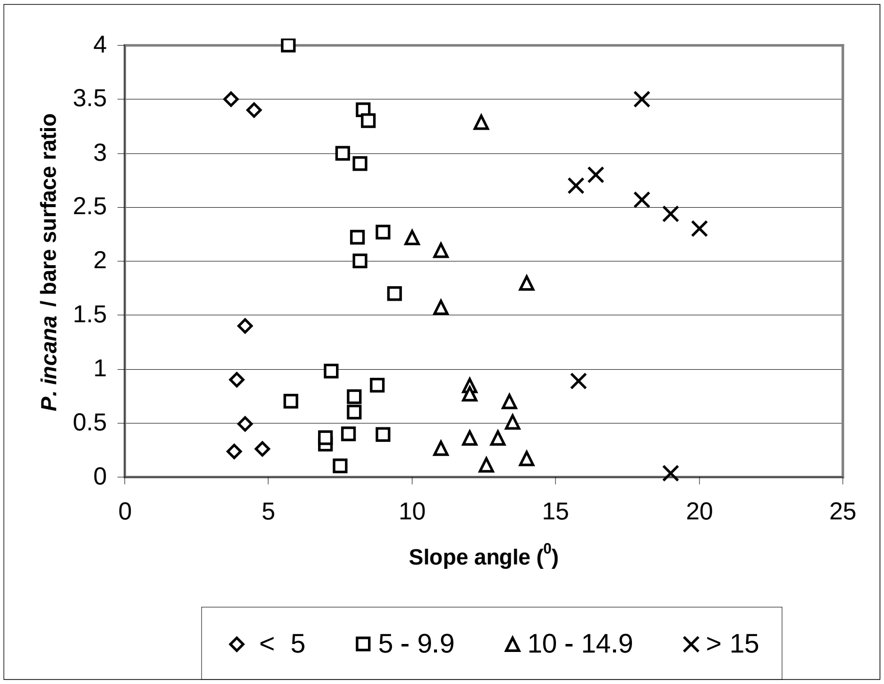


Figure 5.16 *P. incana* /bare surface ratio in different slope categories.

As can be noted from Figure 5.16, P/B ratio sizes are not unique to any of the slope categories. At least in each category, P/B ratios vary across the whole range of sizes. By implication, there is no relationship between *P. incana* patchiness and slope angle at these sites. It also suggests that bare surface patches are interspersed almost evenly among *P.*

incana tussocks in the affected areas. In all the slope categories, some cases of the P/B ratios are noted to be below 1, implying that bare surfaces cover larger areas than *P. incana*. In many instances, such ratios (below 1) have been identified as an indicator of severe erosion forms, usually gully and severe rill erosion. As a surrogate for runoff connectivity, it follows that the smaller the P/B ratio, the greater the potential for runoff connectivity and severe erosion especially at steep slope angles.

Over extensive areas, the bare surfaces do interconnect randomly as can be noted in Figures 5.14 and 5.15a. *P. incana* tussocks display an inconsistent stippled pattern characterised by a considerable fetch length due to patchiness loss. Deductions regarding the observed pattern of *P. incana* and bare surface mosaics will be presented in the discussion Chapter 7.

The P/B ratio could be perceived as a dividing line between trajectories that will either take the system to a state of irreversible degradation, characterised by severe erosion or recovery. Increased loss of vegetation patchiness to erosion and hence the expansion of bare areas (5.15a) is one of the indicators of former trajectory. The conditions illustrated by the Figure and even worse ones are widely prevalent in the area.

Despite assuming an inconsistent stippled pattern on most hillslopes, in a few areas, *P. incana* can be seen to display some form of orientation to the local micro-topographic features like contouring on abandoned fields. It tends to develop in some form of strips on slopes with such structures (see Figure 5.17 below).

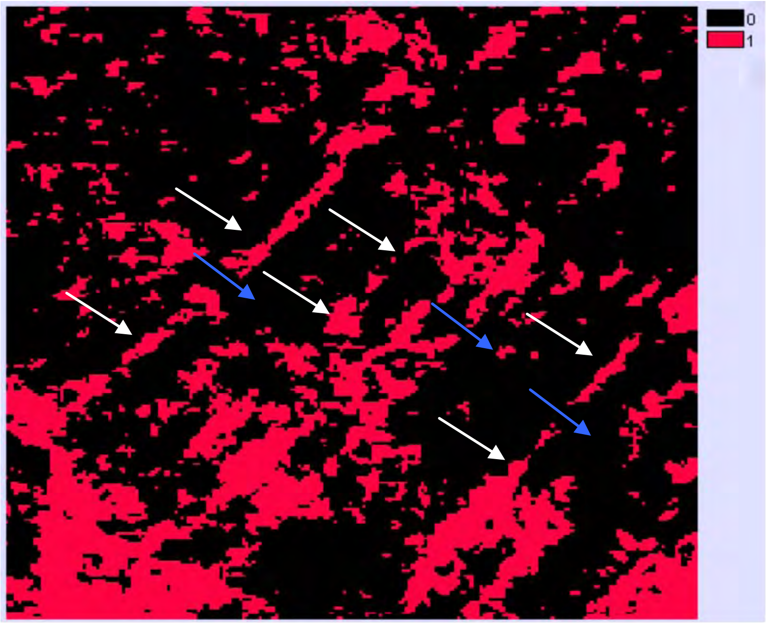
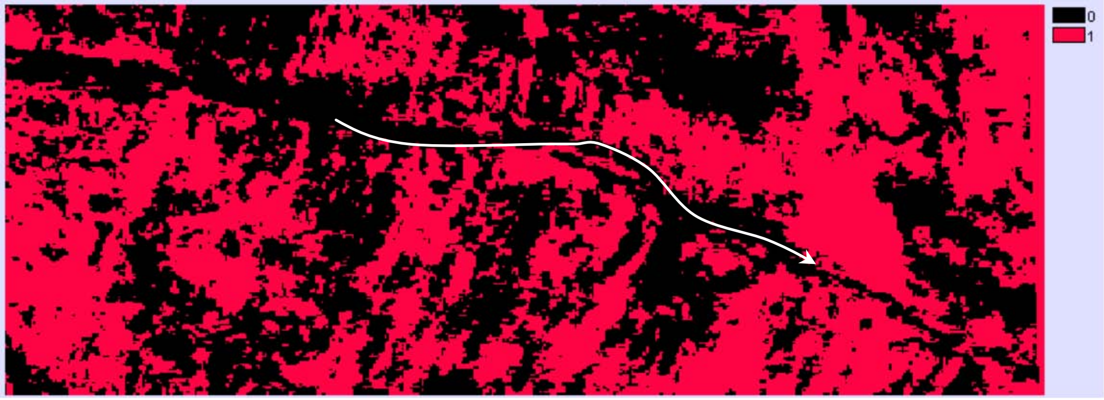


Figure 5.17 *P. incana* boolean image window showing stripes developing along contours. White arrows show direction of slope. Blue arrows show runoff loss between stripes. 1 = *P. incana*, 0 = Bare and other surfaces.

Under such striping conditions along contours, *P. incana* is seen to act as a sink, as it captures runoff and other materials. The success rate in runoff capture under these conditions is limited as flow divergence may lead to runoff concentration at conduits between individual tussocks, leading runoff loss to downslope stripes (see blue arrows in Figure 5.17 above). This loss has been noted in the field to lead to the development of elongated rills and gullies as illustrated in Figure 5.18 below.



0 = bare and other surfaces; 1 = *P. incana*.

Figure 5.18 *P. incana* boolean image window showing run-out between *P. incana* stripes.

The vegetation structure of *P. incana* is examined further under field conditions in Chapter 6 where more light is thrown on the species' structural dynamics.

5.5 Conclusion

Slope angle and aspect do significantly influence the presence or absence of *P. incana*. Steep slope categories and south facing slopes have greater *P. incana* densities. The combined influence of terrain parameters and land disturbance, particularly land abandonment, is noted to have promoted *P. incana* invasion. The topographically driven Wetness Index as a surrogate for soil moisture variability is identified as having a strong relationship with the spatial distribution of *P. incana* such that the higher the WI, the greater the likelihood of *P. incana* absence and vice versa. *P. incana* loss of patchiness is perceived as an indicator of increased runoff generation such that, the greater the patchiness loss, the higher the potential for runoff connectivity and erosion.

Chapter 6: *Interaction Between Pteronia Incana, Soil Surface Conditions, Runoff Potential and Erosion*

6.1 Introduction

Using survey data, the degree of invasion of *P. incana* and its relationship with hillslope conditions, visible erosion forms, soil physical properties and their implications for runoff potential are explored in this chapter. The structural attributes of *P. incana* are described and soil surface conditions are examined under field conditions. On the basis of photographic evidence and field observations, scenarios of *P. incana* invasion dynamics and the envisaged trajectories of how it establishes into the terrain are developed. Projections of the potential feedback mechanisms between vegetation pattern and water flux are also described.

6.2 ***Broad Catchment Surveys***

50 hillslopes were surveyed at 1km intervals across Mgwalana catchment between 8th February and 12th November 2000. As described in Chapter 3, section 3.3, general observations were made relating to the presence or absence of *P. incana*, degree of invasion in relation to slope position and angle, soil surface conditions (particularly crusting), erosion status, land use and distance from stream channel were recorded. Each hillslope constituted three sampling points: the lower, mid and upper slope elements, giving a total of 150 field sampling points (Appendix B). After identifying the invasion hotspots within the catchment, more detailed observations of the pattern of *P. incana* and deductions regarding the implications for hillslope processes were made along transects and other sampling units as will be described in the relevant sections of this chapter.

6.2.1 Typical *P. incana* invaded hillslope

Typically, hillslopes fully invaded by *P. incana*, particularly on abandoned land, have a number of characteristics in common. Most transects along such hillslopes display an absence of *P. incana* for a distance between 3 and 6 metres from stream channels, mainly tributaries of Mgwalana river. The mainstream itself has an extensive riparian zone totally devoid of *P. incana*, with a boundary lined by grass species. *P. incana* absence in this zone is unequivocally linked to high wetness as was noted previously (Chapter 5, sub-section 5.3.6). Severe erosion forms, usually gullies, are mostly located on the lower element of slope (65%, n=50). The mid slope is characterised by mosaics of *P. incana* and considerable bare areas, which seem to be expanding with time (Figure 5.14b). Distinct runoff pathways hallmarked by pedestalling in the inter *P. incana* tussock areas are common features on this slope element. Tongway and Hindley (1995) explain the process of pedestalling as resulting from the removal of soil by erosion of an area to a depth of at least several cm. This leaves the butts and exposed roots of surviving plants on a column of soil above the new general level of the landscape. The conditions on the upper slope elements are in many respects similar to those of the mid slope. Wide crusted bare surfaces in the inter- *P. incana* patch areas, incipient rilling and a few cases of gully erosion (15% of the observations) were noted on these slope elements. Severe sheet erosion in the bare patch areas is most prevalent on this section (60% of the observations). Figure 6.1 overleaf is a sketch profile that attempts to typify hillslope conditions fully invaded by *P. incana*.

6.2.2 Spatial relationships between *P. incana* and selected variables

The geo-coded observations recorded in the surveys were entered in an Excel spreadsheet and then exported to ArcView 3.2 GIS as a compatible Dbase file. The 'Query Builder' facility in the ArcView database was used to interrogate the data, deriving spatial correlations between field descriptions of the degree of *P. incana* invasion and the range of variables singled out in section 6.2 above. The degree of *P. incana* invasion was divided into four classes *viz*.: 1= No *P. incana*; 2= *P. incana* patches; 3= *P. incana* invading (i.e. sections of hillslopes colonised) and 4= Established *P. incana* (where entire hillslopes are invaded). The rest of the variables were also divided into specific classes as shown in Table 6.1. The frequency of each class of *P. incana* invasion in relation to the

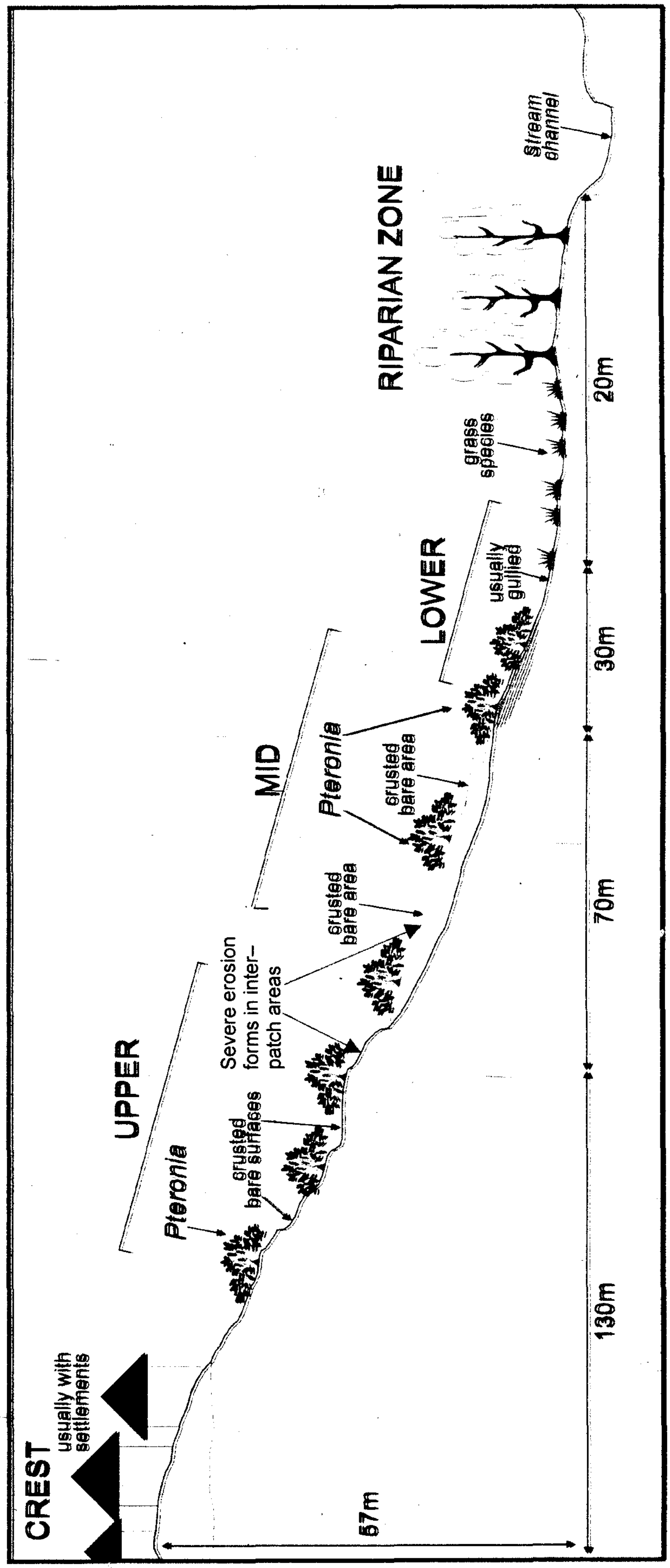


Figure 6.1 Sketch transect of a typical Pteronia invaded hillslope.

respective variables was also determined in the ArcView database. It was thus possible to carry out a Chi-square test to determine whether the differences in the observed frequencies occurred by chance or were significant enough to reflect the influence of a specific variable on the degree of *P. incana* invasion or vice versa. The null hypothesis formulated in each case, e.g. for slope angle, was: There is no significant difference in the degree of *P. incana* invasion in relation to slope angle categories. The results of the non-parametric Chi-square test statistic are presented in Table 6.2.

Table 6.1 Description of variables and their categories. *SARCCUS (1981) erosion classification.

Variable		**Class or description**			
1	Slope angle category (^{0})	< 5	5 - 9	10 - 15	>15
2	Hillslope position	Lower	Mid	Upper	-
3	Land use	Cultivation	Grazing	Abandoned land	-
4	Erosion class	No Erosion (S1)*	Sheet erosion (S2, S3)*	Severe sheet and incipient rill erosion (S5, R2)*	Severe rill and gully erosion (R4, G2 - 4)*
5	Soil surface condition	Not crusted	Crusted soil surfaces		-
6	Distance from stream channel (m)	<5	5 - 50	51 - 100	>100

Table 6.2 Chi –square test results – *P. incana* classes vs selected variables. *DF = Degrees of Freedom.

***P. incana* class vs**	**DF***	**Critical value**	**Chi- square**	**Significant? (level 0.001)**
Slope angle category	9	27.88	182.59	Yes
Hilllope position	6	22.46	5.15	No
Land use	6	22.46	41.48	Yes
Erosion class	9	27.88	365.39	Yes
Soil surface condition	3	16.27	214.8	Yes
Distance from channel	9	27.88	12.31	No

As can be noted from Table 6.2, differences in the observed frequencies of *P. incana* invasion class in relation to slope angle category, erosion class, soil surface condition and land use are significant enough to reflect the influence of the respective variables. Hillslope position and distance from the channel were not significant. The null hypotheses in respect of the former set of variables are rejected as opposed to the latter. Against the background of the identified significant relationships, there is a need to establish the specific variable descriptions that contribute strongly to the relationships. The statistically significant relationships are examined more closely in the sub-sections below.

6.2.2.1 *Degree of P. incana invasion and slope angle categories*

The frequency of occurrence of *P. incana* invasion classes in relation to the respective variables was expressed as a percentage of the total observations (n = 150). Figure 6.2 illustrates the relationship between *P. incana* invasion and slope angle categories. It is evident from the Figure that whereas the gentle slope categories are associated with no *P. incana* and a low degree of *P. incana* invasion, the frequency of the 'established *P. incana*' class increases with slope angle. The strong relationship is in keeping with the findings from remotely sensed data regarding the relationship between *P. incana* density and slope angle. The prominence of slope angle in *P. incana* invasion warrants, among other things, an investigation of specific hillslope processes that could explain this relationship. This will be attempted in the later sections of this chapter.

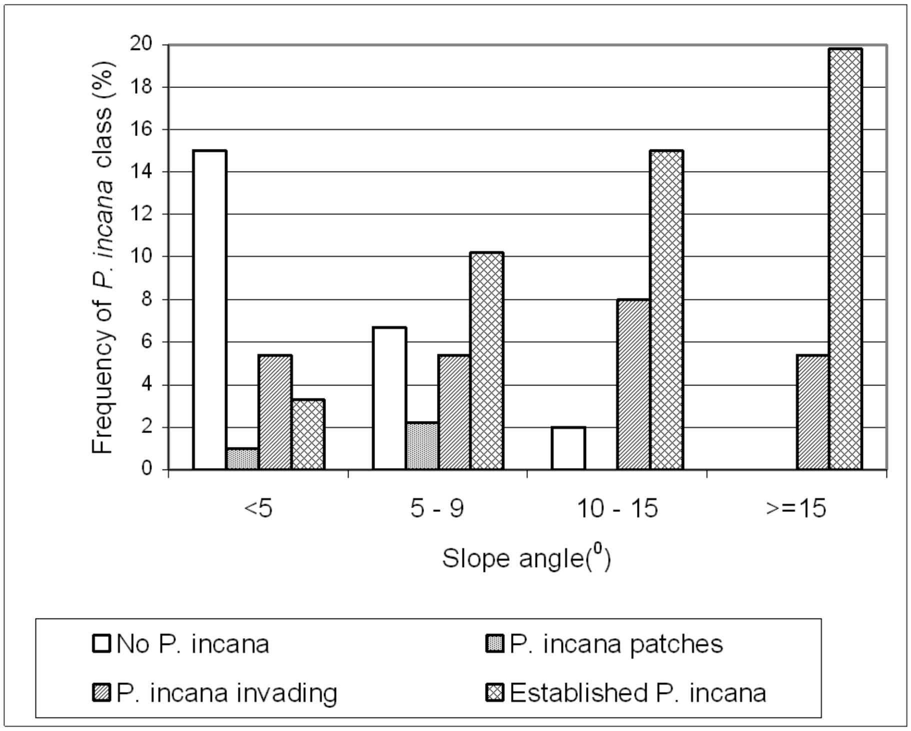


Figure 6.2 Relationship between degree of *P. incana* invasion and slope angle categories.

6.2.2.2 P. incana invasion and soil surface condition

A statistically significant Chi - square value between *P. incana* class and soil surface condition signifies a close association between the two variables. The relationship is depicted by Figure 6.3 below.

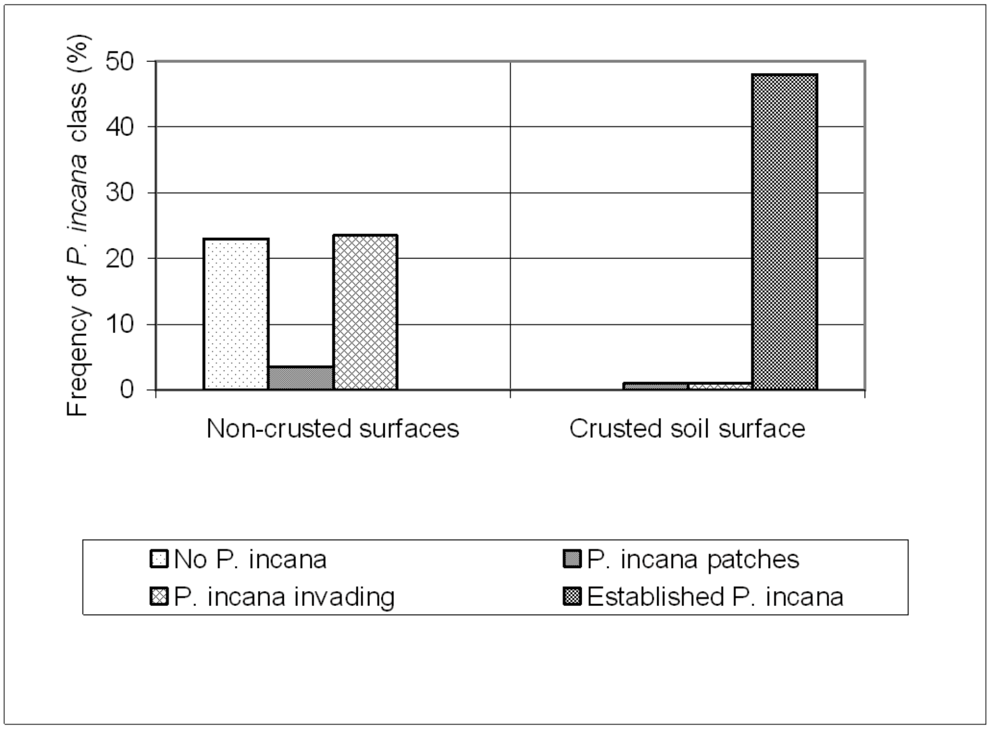


Figure 6.3 The relationship between soil surface condition and *P. incana* invasion.

A clear dichotomy is evident from Figure 6.3 such that areas where *P. incana* is fully established have an extremely high frequency of crusted soils as opposed to the uninvaded areas and to some extent, the partially invaded ones. Most of the bare patches between *P. incana* tussocks (96% of the observations) have crusted soil surfaces. Given *P. incana's* loss of patchiness and the expansion of bare areas as was noted earlier (section 5.4), a combination of such surfaces and steep slopes would undoubtedly have strong implications for runoff generation and erosion. The relationship between the observed soil erosion forms and *P. incana* invasion is examined below.

6.2.2.3 P. incana invasion and soil erosion

It is imperative to bring out the finer details regarding the statistically significant relationship between the degree of *P. incana* invasion and specific erosion class as defined in Table 6.1. Figure 6.4 below illustrates the frequencies of the respective *P. incana* invasion descriptions in relation to erosion classes.

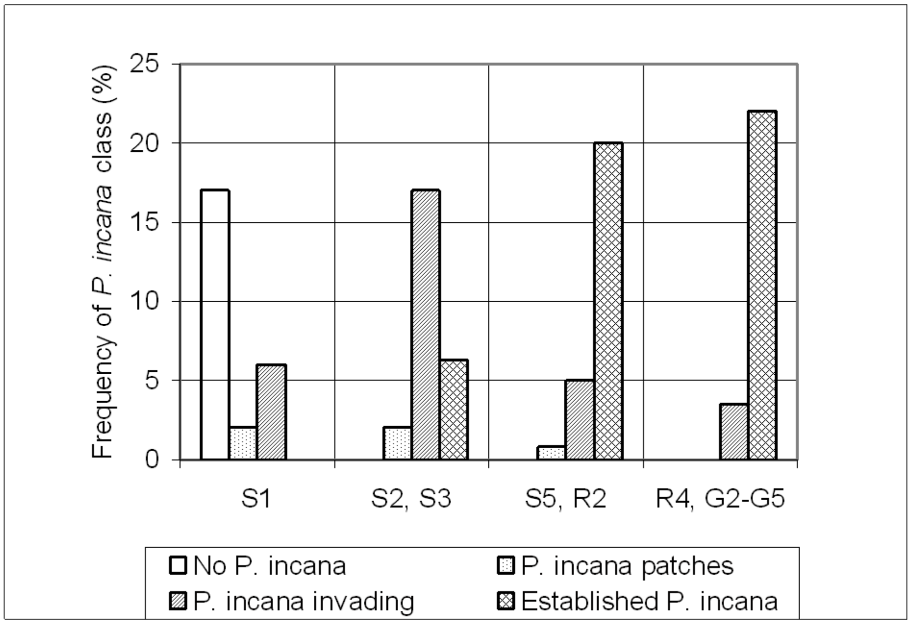


Figure 6.4 Relationship between erosion class and *P. incana* invasion.

Clear linkages between specific erosion classes and the degree of *P. incana* invasion are noticeable from Figure 6.4 above. Whereas the no visible erosion class (S1) dominates areas with 'no *P. incana*, severe forms of erosion *viz*. severe sheet, rill and gully erosion are associated with 'established *P. incana*' and to a reasonable extent, '*P. incana* invading' (partially colonised areas). The close association between *P. incana* and soil erosion raises a question as to whether *P. incana* invasion and patchiness predate erosion or vice versa. The discussion Chapter 7 puts the erosion and *P. incana* invasion scenario in perspective.

6.2.2.4 P. incana invasion, land use and soil surface condition

The statistically significant Chi-square value between land use and *P. incana* invasion supports the results from remotely sensed data (Chapter 5, section 5.3.3), that abandoned land is highly susceptible to *P. incana* invasion. A graphic illustration of this relationship will therefore, not be repeated here. A selection by the 'Query Builder' in ArcView of all abandoned land sites which at the same time have crusted surfaces returned 100% of the sites, as opposed to 41% of the grazing and none for the land still in cultivation. The query facility also indicated 73% of the abandoned land sites as having extensive bare patches between *P. incana* tussocks as compared to 14% on grazing land. The implications of such relationships for erosion are highlighted by close spatial correlation between abandoned land and erosion (cf. Kakembo, 1997; Kakembo and Rowntree, 2003).

6.3 P. incana pattern

The pattern of vegetation on a landscape is an important indicator of the control that vegetation exerts on water flux. The spatial arrangement of vegetation elements reflects its role in filtering out sediment, arresting the flow of runoff water, and control on erosion (Tongway and Hindley, 1995). It also increases surface roughness as a result of stems projecting into the flow, retarding runoff flow velocity. The level of roughness will largely depend on the morphology of the plant and its density of growth (Obando, 1996). It is against this background that the pattern of *P. incana* was surveyed in the field.

Tongway and Hindley (1995) single out three parameters as indicators of the functional status of a given landscape, *viz*.:

- The number of obstructions to overland flow per unit length of transect.
- Obstruction width per plant and number of plants across a transect.
- The distance between obstructions (fetch) per unit length of transect.

The parameters are illustrated in Figure 6.5a and b below.

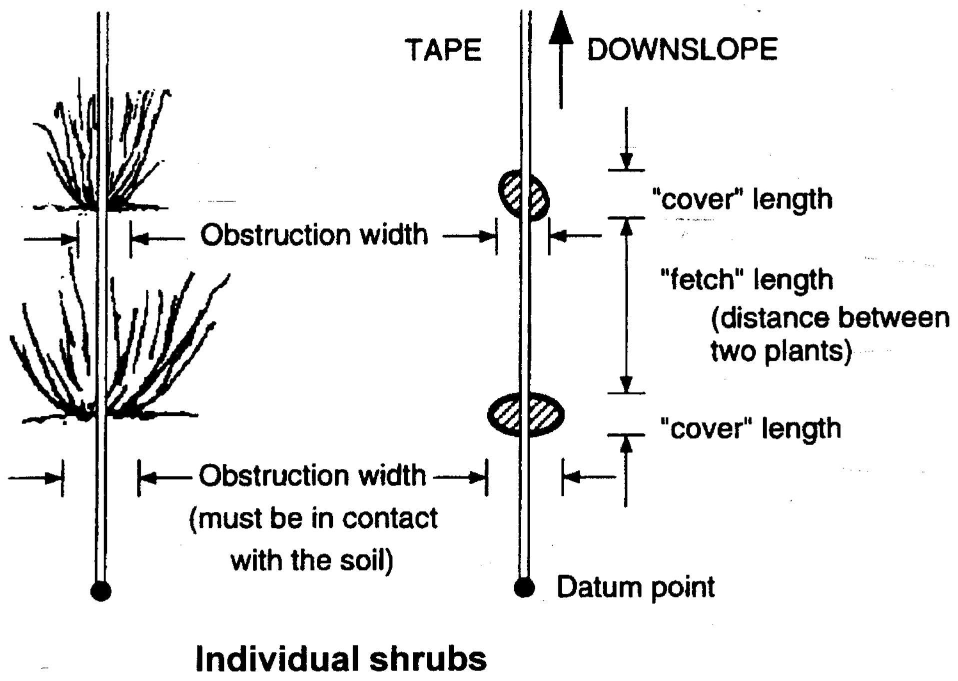


Figure 6.5a An illustration of vegetation pattern parameters for individual shrubs (Adapted from Tongway and Hindley 1995).

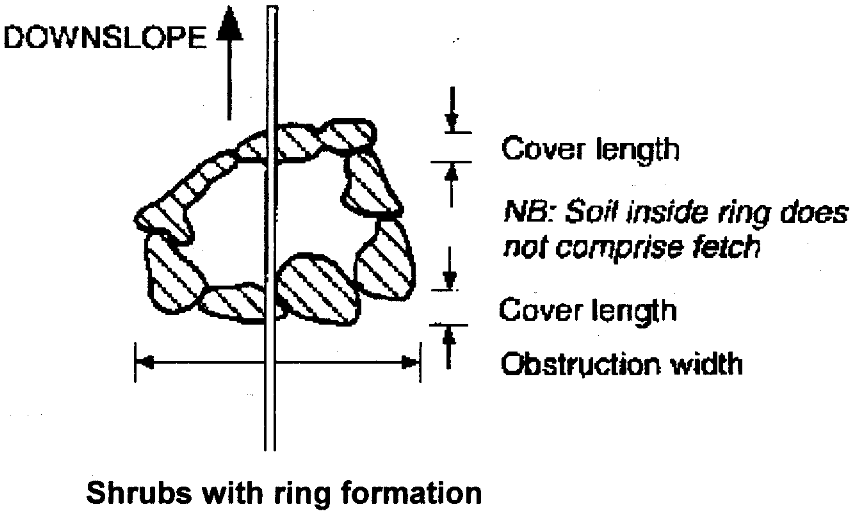


Figure 6.5b An illustration of vegetation pattern parameters for shrubs with a ring formation (Adapted from Tongway and Hindley 1995).

6.3.1 Sampling design

Pattern parameters which are considered critical in the interaction between *P. incana* and water flux were measured along transects aligned with the maximum slope. *P. incana* pattern is noted to vary with the degree of invasion. It ranges from isolated patches within grassland, usually in a ring formation (designated as category 1) as a cradle from which it spreads into the terrain, to fairly regularly alternating stippled patches interspersed by bare soil surfaces (category 2). Extreme cases characterised by considerably wide bare surfaces between tussocks (category 3) are also widespread. These three categories of *P. incana* pattern (illustrated by Figure 6.6a, b, and c) were therefore treated separately during field measurements. Regarding the isolated patches, a group of shrubs in ring formation as illustrated by Figure 6.5b were chosen randomly as individual sampling points. Pattern measurements were taken from 30 of these formations.

Three hillslopes apiece were selected for *P. incana* pattern measurements for the regularly alternating patches and wide bare-patch tussocks. The transect lengths ranged between 80 to 185 metres for the latter category and 86 to 125 metres for the former. *P. incana* pattern parameters were measured along three transects per hillslope in quadrats of 5mx5m. In each quadrat, the number of stems, height of stems, obstruction width (the width axis), cover length, and fetch (distance between individual plants) were measured. The measurements were then averaged for the hillslopes of either category.

The period during which the broad catchment surveys described in section 6.2 were conducted (February - November 2000) spanned the wet rainy season between February and April and the dry winter months of June to October. Over this period, very small changes were noted in *P. incana's* aboveground biomass accumulation in response to rainfall seasonality. Individual shrubs become moribund during the dry winter period and sprout during the wet season. In particular, fully grown shrubs seem to achieve a stable state at a particular height of about 65 cm on average. Against this background, it was deemed unnecessary to vary data collection regarding *P. incana* pattern between the wet summer and dry winter periods. It was conducted as a once off exercise over four days in

October 2002. The average height given above is in keeping with observations in semi-arid Spain and most Mediterranean areas where shrub cover is usually less than 100 cm. By implication, it is not likely to have a great influence on the increased energy of raindrops, which could coalesce and cause splash erosion (Obando 1996).

6.3.2 Pattern parameters

Table 6.3 below is a summary of the parameters as measured along the respective transects and sampling points for the three *P. incana* pattern categories. It can be noted from the table that the averages of the parameters vary between the categories. As per the description of the categories 2 and 3, the average fetch length and number of stems per quadrat are expected to be bigger and smaller respectively. The obstruction width and cover length also tends to be bigger in the former category. The wider fetch in category 3 signifies loss of patchiness.

Table 6.3 Summary of *P. incana* pattern parameters. 1= Isolated patches; 2= Alternating patches and bare soil; 3= Wide bare surfaces between *P. incana* tussocks.

***P. incana* pattern category**	**Average Number of stems***	**Average obstruction width (cm)**	**Average cover length (cm)**	**Average height (cm)**	**Average fetch length (cm)**
1	n/a®	110	82	68	n/a®
2	44	48	79	65	45
3	33	39	74	61	107

* As per 5X5m quadrat
® Parameter not applicable to category (see Figure 6.5b).

Figure 6.6 *P. incana* pattern categories: a = category 1, b = category 2 and c = category 3. (See Table 6.3 caption for category descriptions).

Relationships between structural parameters for category 3 are illustrated by Figure 6.7a, b and c.

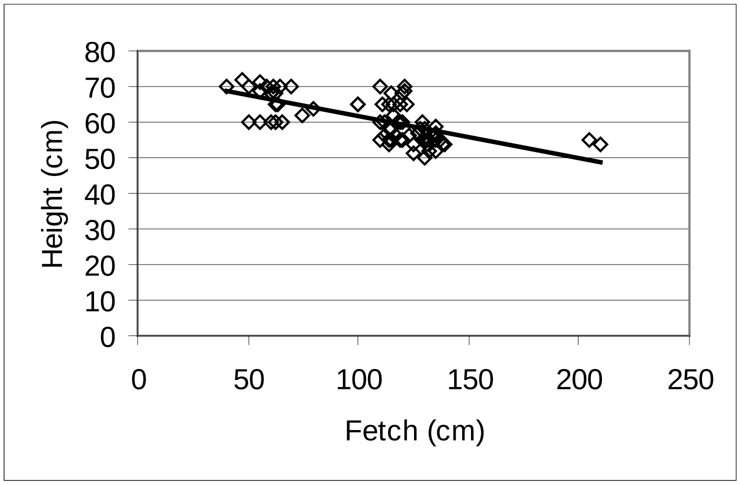


Figure 6.7a Relationship between fetch and height.

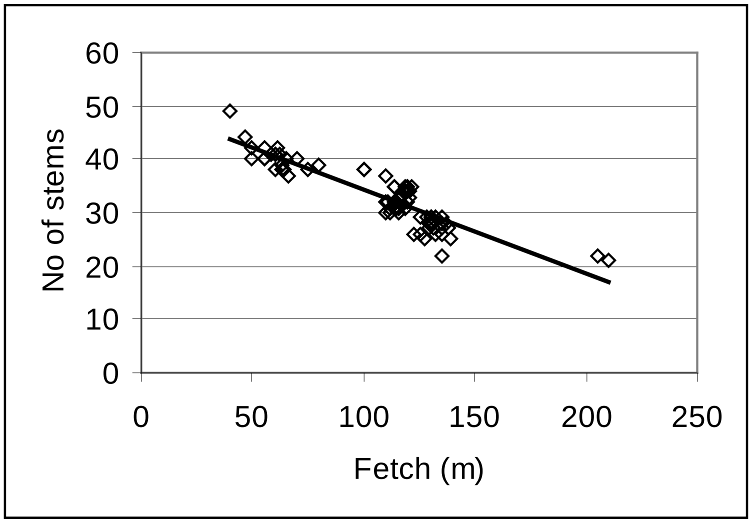


Figure 6.7b Relationship between fetch and no of stems.

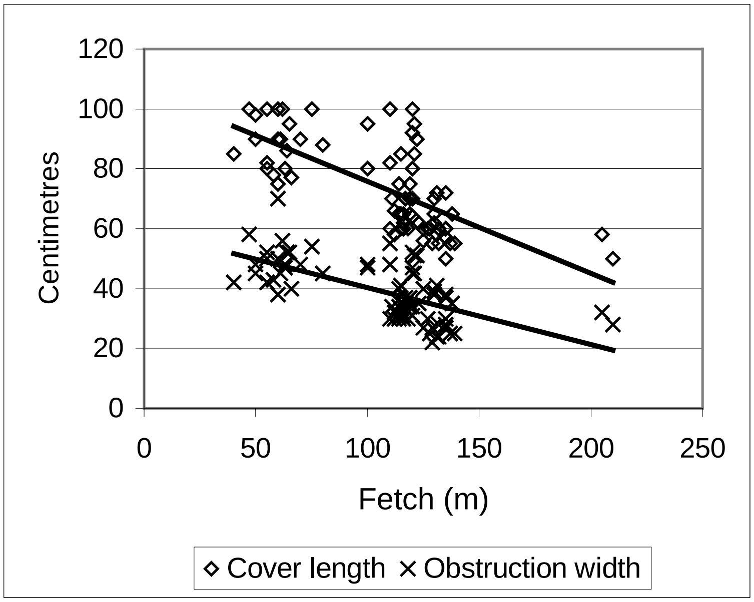


Figure 6.7c Relationship between Fetch, cover length and obstruction width.

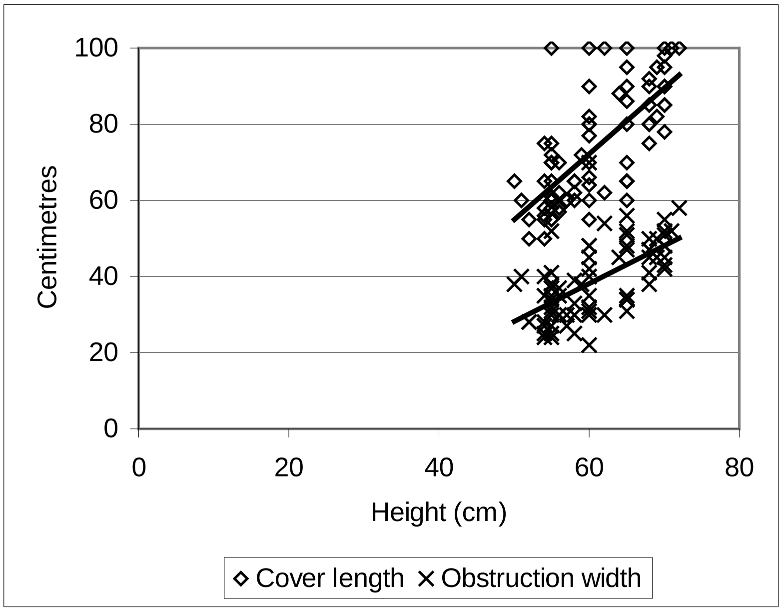


Figure 6.7d Relationship between Height, cover length and obstruction width.

Clear trends can be picked from Figure 6.7 above. It can be noted from Figure 6.7a, b, and c that *P. incana* tussock height, number of stems, cover length and obstruction width decrease with increasing fetch. Two distinct clusters are discernible with each parameter, such that a higher value of each parameter is associated with a smaller fetch and *vice versa*. The reduction in the value of each parameter with increasing fetch could be interpreted as tending to loss of patchiness which is preceded by stunted growth of individual shrubs, characterised by lower shrub height, number of stems per quadrat, cover length and obstruction width. This phenomenon is illustrated by Figure 5.15a, section 5.4 of Chapter 5. Cover length and obstruction width are also illustrated by Figure 6.7d as increasing with height. Such variable dynamics in *P. incana* pattern parameters will have a bearing on raindrop interception and runoff / run-on dynamics. These aspects will be put in perspective in the discussion Chapter 7.

6.4 How does P. incana establish itself into the terrain?

The process by means of which *P. incana* establishes itself into the surrounding terrain is a multi-pronged one, the understanding of which would require long term monitoring. Sequential aerial photographs and satellite imagery would provide some useful information. It would, however, require satellite images of very fine resolution to separate the shrub from other vegetation cover. Even then, determining the trajectories of the invasion would necessitate sequential imagery in close succession, which implies prohibitive costs. Regarding aerial photographs, it would be very difficult to distinguish *P. incana* from say grass species, even at a fine scale of photography.

Evidence assembled thus far indicates that *P. incana* invasion parallels land use change and patterns. Other minor disturbances like small termite mounds and other burrowing animal activity have been observed in the field as points where *P. incana* patches carve a niche into the terrain. In scale terms, the role of such isolated points is exceeded by far by the impact of large scale disturbances, *viz.*, cultivation, abandonment and grazing activities. Aerial photography would provide land use change information against which the invasion would be benchmarked. That blanket invasion of abandoned lands could be pegged against 1975 aerial photographic scene was alluded to in Chapter 3, section 3.4. Many of the abandoned lands are noticeable from this photography set as eroded bare areas. Against this background and field observations, possible case scenarios of *P. incana* invasion trajectories are postulated below.

The mode of invasion is perceived to vary between abandoned and grazing lands. In abandoned cultivated lands, bare crusted surfaces had become resource poor as severe erosion preceded the invasion (see section 6.2.2.3). Such conditions will have favoured the invasion of the resilient *P. incana* with very limited competition from grass species. Unfortunately, the invasion rate cannot be ascertained owing to the reasons given in the foregoing paragraph. That notwithstanding, the trajectory described here is envisaged to have rapidly created a single species dominance scenario, with high shrub density and patchiness. As the abandoned lands became resource poorer by way of erosion intensification, patchiness loss and expansion of bare areas gave rise to the presently observed high *P. incana* dominance, with wide bare areas. Soil crusting and erosion on steep slopes could also have given rise to a direct conversion to high *P. incana* dominance with wide bare areas. Figure 6.8 illustrates this trajectory.

The invasion trajectory of grazing lands is based on field observations. Instant dominance as is the case on abandoned land is not contemplated in this instance. As was pointed out in Chapter 3, section 3.4, the rate of invasion on grazing land is measurable, but has not been determined in the present study. Personal field observations over a period of three and half years of *P. incana* patch dynamics on specific hillslopes have shown that individual shrubs emerge and eventually assume a ring formation (see Figure 6.6a). Grass species in the centre and immediate vicinity of the ring formation die off, leaving a bare crusted soil surface. Owing to the wind dispersed nature of the seed bank, single stands of the shrub are gradually seen to intersperse the local hillslope. Adjacent ring formations coalesce with time, finally colonising the entire hillslope. The invasion trajectory described here is illustrated by Figure 6.8. From a position of low dominance, low patchiness and low density, the process leads to coalescence of *P. incana* tussocks and development of bare areas. This gives rise to high dominance, with a mosaic of *P. incana* tussocks and bare areas. Patchiness loss and erosion in the inter-patch bare areas could ultimately lead to widening of bare areas and low tussock density (see Figure 6.8).

It is deemed appropriate to point out here that at some of the observed sites, the apparent competitive advantage *P. incana* gains on grass species is in part attributed to the continual low above-the-ground biomass for latter species. Grass species interspersed by *P. incana* patches have been seen to remain at ground level, owing to relentless grazing activities. This phenomenon is, however, site-specific and cannot be generalised as veld still in excellent condition has been observed near some villages. There is a need gain an understanding of other factors at play, particularly the local community regulatory frameworks.

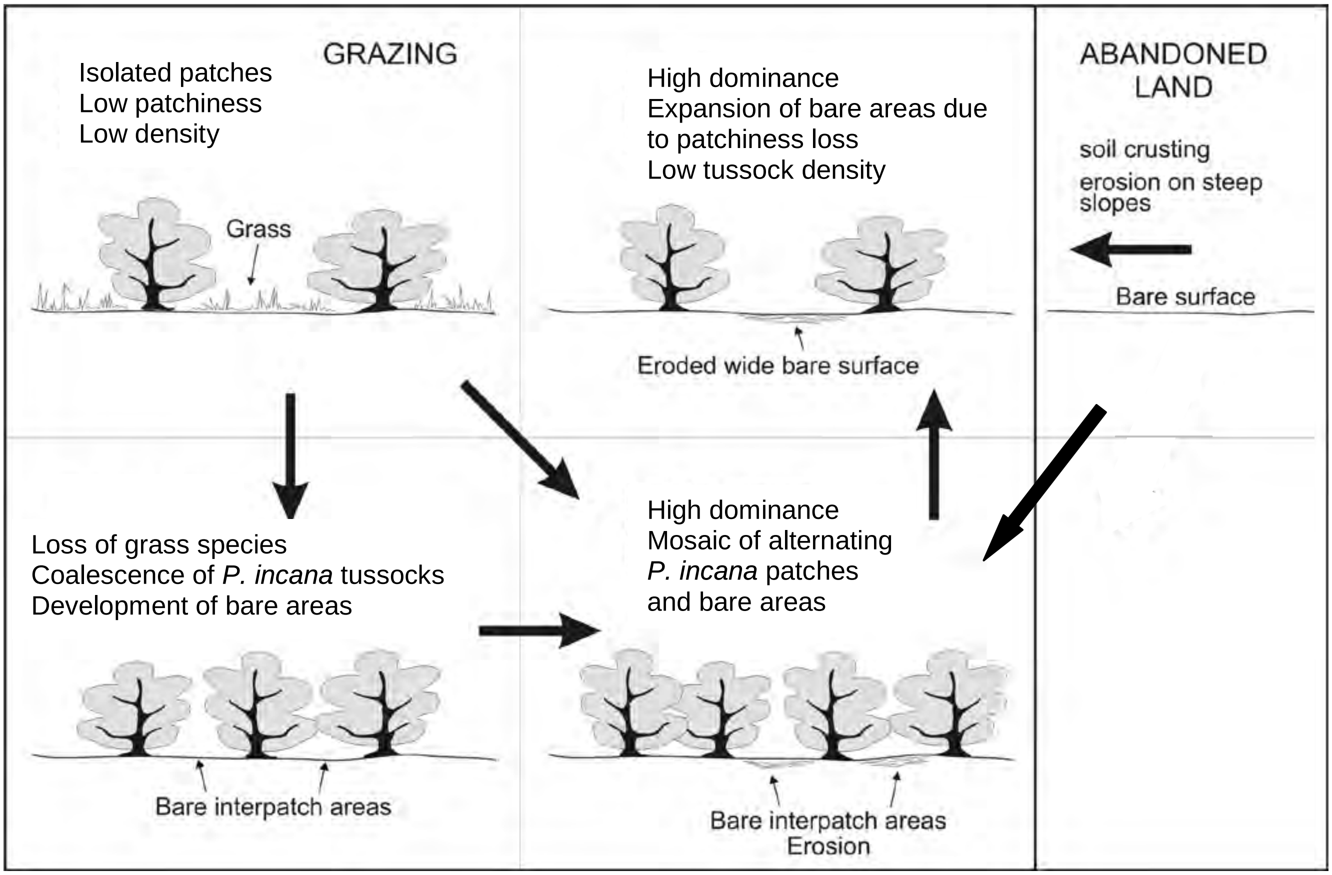


Figure 6.8 Theorised *P. incana* invasion trajectories.

6.5 *Moisture dependencies for P. incana and grass species and implications for competition and replacement.*

The two- layer moisture theory was elucidated in Chapter 2 section 2.3.3.4 whereby it was confirmed that shrub and grass species have different soil moisture dependencies that vary with depth of the soil. This has strong implications for competition and replacement.

As was pointed out in section 6.4 above, grass species adjacent to *P. incana* patches die off as the latter species gradually colonise entire hillslopes. In order to investigate the differences in moisture dependencies between the two species, their rooting depth was examined at a site with fairly deep soil where *P. incana* and grass cover are adjacent. Two trenches of up to 1 and 0.6 metres of depth respectively were excavated under either species (see Figure 6.10). The species' roots were exposed with a fine spray of water (after Hobson, 1989 and Montana *et al*, 1995). For both species, root numbers within a 10 cm by 10 cm frame were counted and averaged every 20 cm of depth. Figure 6.9 shows the average number of roots with depth for the two species.

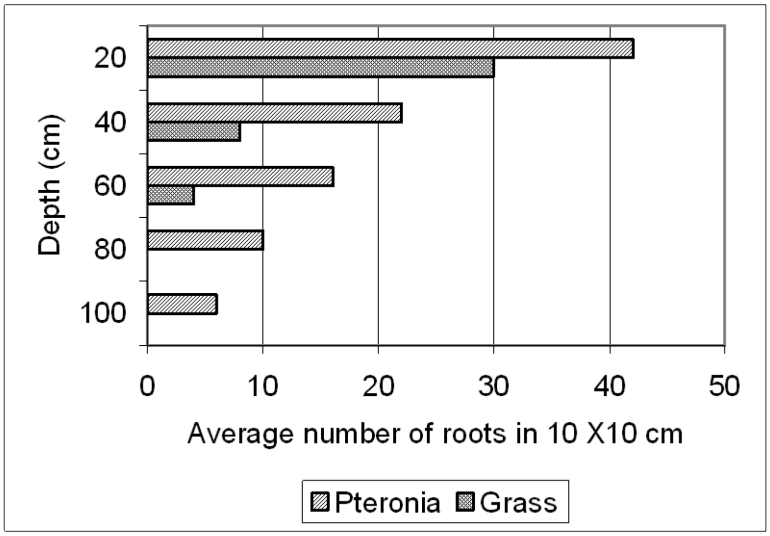


Figure 6.9 Average root numbers for *P. incana* (up to 1m depth) and grass species (up to 60 cm).

Figure 6.10 Trenches excavated under *P. incana* and grass species to expose roots.

It can be noted from Figure 6.9 that while root distribution for both species is abundant in the top 20 cm, the average root number decreases more rapidly with grass species than with *P. incana*. The rooting depth for the latter ranges between 90 cm to 1m as opposed to 40 cm and to a limited extent, 60 cm for grass. It is noteworthy, however, that the soils of the study area are predominantly shallow and that deep soils similar to the site where the trenches were excavated are a rare occurrence. The soil moisture budget under shallow soil conditions is compounded by the removal by erosion of the top soil layer, which the grass species are largely dependent on, and the formation of erosion crusts, the implications of which are examined in section 6.6 below. Under such conditions, the two-layer soil moisture theory is largely constrained. Surface soil moisture is limited and it is only the deep rooting shrubs which are also able to capture some of the surface runoff generated on crusted soil surfaces that will survive (see section 6.7). Such case scenarios in part explain the competitive advantage *P. incana* gains over grass species. This phenomenon is illustrated by Figure 6.11 below which illustrates a distinct divide between soil moisture dependencies for *P. incana* and grass species. Soil moisture readings were taken from grass patches and *P. incana* clusters using the Delta-T ML2 meter (section 5.3.6, Chapter 5) which provides soil moisture values for surface soil up to 7cm. NDVI values corresponding to the respective grass patches and *P. incana* clusters

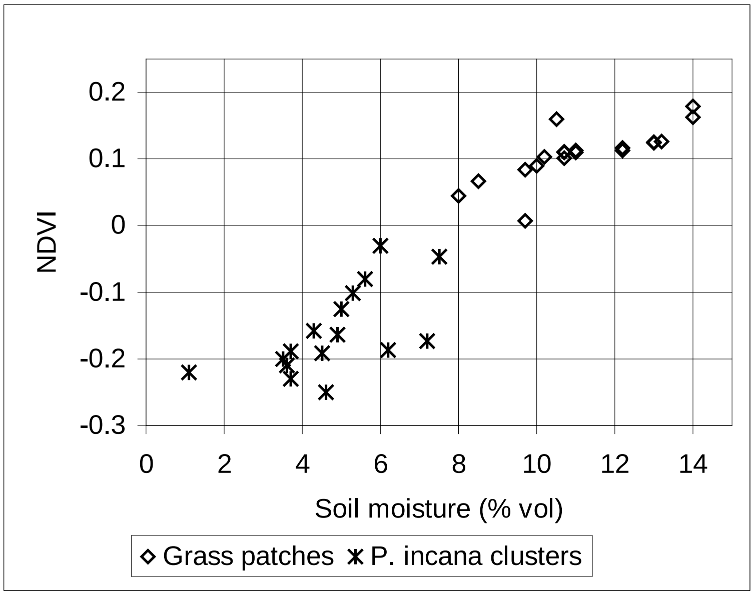


Figure 6.11 Divide between soil moisture dependencies for *P. incana* and grass

were extracted from the relevant images. Grass patch NDVI values are distinctly positive and coincide with high soil moisture as opposed to the negative values for *P. incana*. As can be noted from Figure 6.11, grass species are confined to areas where surface soil moisture volume is on average 10 % and above. 8% is noted as the threshold separating the two species, other things being equal.

6.6 Comparative soil surface conditions under P. incana and grass species

The development of soil crusts and their significance in the development of vegetation patchiness was highlighted in Chapter 2, section 2.3.3.3. So was their role in reduced infiltrability and runoff generation. The high susceptibility to crusting of the soils of the study area was noted from Table 3.1 in Chapter 3 showing the general characteristics of the local soils. The Sodium Adsorption Ratio of the soil samples analysed (see section 3.2, Chapter 3) ranged from 36.50 to 106.55, values that are well above the critical value 13, which according to Miller and Donahue (1995) defines the boundary between sodic and non-sodic soils. The dominance of swelling hydrous mica clays (see Table 3.1) compounds the crusting susceptibility problem.

Soil crusts of up to 1mm thick, perceived as erosional crusts according to the description by Valentin (1991) and Valentin and Blesson (1998), are a widespread phenomenon in the study area, particularly in the inter *P. incana* bare patch area (see Figure 6.12). In order to determine whether soil crusting conditions are significantly greater under *P. incana* cover than grass, a comparative investigation of the soil surface conditions was conducted from the same adjacent, relatively homogeneous grass patches and *P. incana* clusters described in Chapter 5, section 5.3.6. Simultaneous measurements of soil moisture and soil crusting were made at the sites. Soil surface strength as a surrogate for soil crusting was assessed using a hand-operated cone penetrometer. Soil surface strength readings taken from 20 points at each site were averaged to obtain a single reading for each of 30 homogeneous areas chosen for this purpose. Figure 6.13 below is a histogram of the mean soil surface strength values under *P. incana* and grass species.

Figure 6.12 Soil surface crust (shown by white arrows) in the inter-*P. incana* bare patch area.

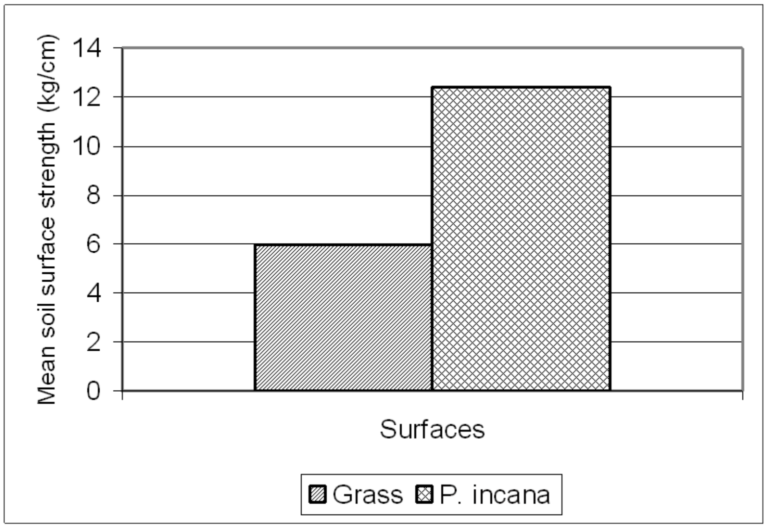


Figure 6.13 Comparative soil surface strength between grass species and *P. incana*.

A T- test value of 11.09 calculated for the 30 sites, significant at 0.001 level, confirms that soil surface conditions in terms of crusting under *P. incana* are significantly greater than under grass. This implies significantly greater runoff generation under *P. incana* cover than grass. That the crusts are erosional by type also indicates significantly greater erosion under the former than the latter.

6.7 Runoff generation under P. incana cover.

Greater potential for runoff generation on crusted soil surfaces under *P. incana* has been borne out in section 6.6 above. That notwithstanding, it is noteworthy that, to some extent, *P. incana* plays some role in so far as effecting interception and influencing infiltration. A runoff experimental set-up would be a useful mechanism to determine variations in the magnitude of runoff volumes under *P. incana*, adjacent zones and other cover surfaces. This would also give an indication as to the runoff generation dynamics and the extent of interception and infiltration functions as influenced by *P. incana*. An understanding of these dynamics would serve to inform the design of appropriate rehabilitation strategies. Due to the unavailability of equipment to carry out such experimental set-up, surrogates for runoff were examined in the field. Kirkby *et al* (2002) single out the several elements that the runoff threshold consists of *viz*.:

- Above-ground interception on plant leaves and stems
- Interception by litter
- Storage within the soil
- Depression storage within surface roughness features.

The third element on the list would undoubtedly be an expression of the first two, especially where plant height does not promote the coalescence of raindrops. This is true of *P. incana* (see section 6.3 on *P. incana* pattern) and grass cover. Ascertaining soil moisture variations under different cover components after a series of rainstorms would give a clear signal regarding storage within the soil and runoff. This would also help to answer two of the critical questions as posed by Puigdefábregas and Sánchez (1996) that are also relevant to the present study *viz.*:

- Do patchy mosaics of sparse vegetation affect the spatial distribution patterns of overland flow and sediment movement, and if yes, to what extent?

- Does the different soil moisture behaviour of vegetated and bare-ground patches explain their relative importance as runoff sources or sinks?

To this end, soil moisture readings under *P. incana* and grass were taken again from the same 38 sites in July and September 2002, this time after a series of heavy recharge rainstorms. The moisture readings on the two occasions were averaged. Under *P. incana* cover, the exercise was modified such that separate moisture readings were taken on the bare zone upslope of individual stands or a group of shrubs and directly under *P. incana* plants. Figure 6.14 shows the moisture variations under the respective surfaces.

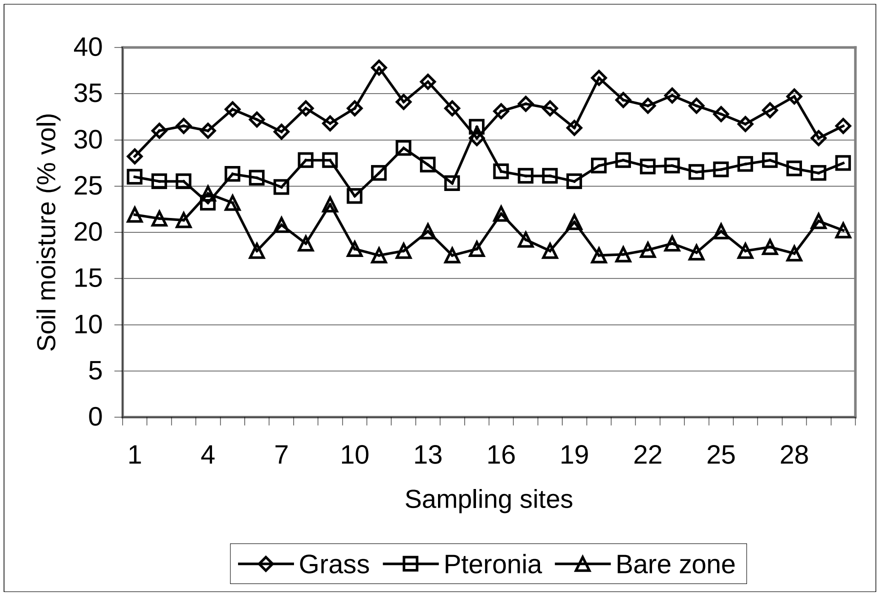


Figure 6.14 Soil moisture variations under grass, *P. incana* and bare surfaces after a series of recharge rainfall events.

Distinct differences in surface soil moisture storage characterised by a progressive increase from the bare zones upslope of *P. incana*, *P. incana* tussocks to grass cover are noticeable in Figure 6.14. Soil moisture storage is significantly greater under *P. incana* than under the bare zones (T-test value = 15.12, statistically significant at 0.001 level). By implication, much greater runoff is generated on the bare surface than under *P.*

incana. This difference signifies the extent to which *P. incana* tussocks act as sink areas for some of the runoff generated on the bare zones. This is also coupled with the aboveground and litter interception. These functions are largely influenced by site-specific structural attributes of *P. incana*. It was noted during the process of taking soil moisture readings that the wider the fetch and hence bare area upslope of individual or a group of *P. incana* tussocks, the lower the moisture value and vice versa. This implies that the bigger the fetch length, the greater the runoff volume and vice versa. The implications of these runoff dynamics will be put in a wider perspective in the next chapter which is the discussion chapter.

6.8 Conclusion

A strong spatial relationship between *P. incana* distribution, crusted soil surfaces and visible erosion forms has been identified in this chapter. The potential for runoff generation on crusted soil surfaces is in part governed by the *P. incana* pattern. Envisaged trajectories of how *P. incana* establishes itself into the surrounding terrain have been presented. Different soil moisture dependencies for *P. incana* and grass have also been examined. While grass is largely dependent on the near soil surface moisture, *P. incana* can tap deep soil moisture as well as runoff generated on crusted surfaces. The limited role played by *P. incana* by way of above-ground interception and runoff capture must be taken cognisance of when rolling out rehabilitation strategies.

Chapter 7: *Discussion, Development of Conceptual Model, and Conclusions*

7.1 Introduction

In this Chapter, the results presented in Chapters 4, 5, and 6 are discussed within the framework of the perspectives and implications reviewed in Chapter 2 and other findings cited in the results chapters. The efficacy of the Kodak DCS420 Digital Camera Imagery for quantitative spatial and spectral analyses is contextualised. The findings regarding the ability of vegetation indices *viz*.: NDVI, SAVI, MSAVI and PVI to discriminate *P. incana* from other cover types are reviewed. An appraisal of the interaction between land use, the co-parameters of terrain and hillslope processes to influence *P. incana* invasion is provided. A conceptual model is developed to explain the linkages between variables that underpin *P. incana* invasion. Recommendations regarding the rehabilitation of the invaded and degraded landscapes are made. Directions for future research are given and final conclusions are then drawn.

7.2 Efficacy of the Kodak DCS420 Digital Camera Imagery

The suitability of the Kodak DCS420 Digital Camera Imagery for spatial and spectral analyses has been borne out by the image classification accuracy and relative consistency of the spectral characteristics of *P. incana* and other cover types. The levels of classification accuracy and distinct spectral separability for the respective vegetation cover types indicate that, by and large, the imagery suffers limited spectral distortions (section 4.3). This is in contrast with Kodak DCS460c digital camera whose imagery is plagued by considerable brightness fall-off reported by Dean *et al* (2000). Such differences between the two camera versions could in part be attributed to the fact that while the DCS460c uses full field of view, the DCS420 uses the central field of view (Palmer, *Pers. Comm.*, 2003).

The spatial resolution of the imagery (1mx1m) is well suited for analysing finer details. In the present study, specific aspects pertaining to the spatial distribution of *P. incana* are reliably identified, making the imagery a useful tool for rehabilitation targeting and monitoring. Notwithstanding the strengths of the DCS420 camera, inconsistencies in the spectral values in the overlapping areas of adjacent imagery are a major shortfall. The shortcoming is, however, not insurmountable as calibration services for this and a range of other digital cameras are now available (cf. Department of Geomatic Engineering, University College, London, 2001). The range of spectral values between images captured as a once off exercise is fairly consistent. This has been demonstrated by the relative consistency of spectral values for the different stages of invasion of *P. incana*. Other cover types displayed a typically consistent range of values in the red and infrared bands. The digital camera is thus a reliable instrument for capturing and mapping spatial phenomena at a very high resolution.

7.3 Suitability of vegetation indices to discriminate P. incana

The inability of Spectral Vegetation Indices based on the ratio of infrared/red reflectance to separate *P. incana* from bare soil surfaces was noted (section 4.4.4). SAVI and $MSAVI_2$ designed specifically to cancel the effect of soil noise in cases where pixels contain a mixture of green vegetation and bare soil (Eastman, 1999) fail to provide clear separability between grass, bare surfaces and the Established *P. incana* category. The NDVI response to green cover is borne out by its clear separation of grass from bare surfaces, albeit its inability to separate the latter from *P. incana*. The success of the PVI to distinctly separate out bare soil surfaces from the different stages of *P. incana* invasion and other cover types is in keeping with Jensen's (1996) view that the effectiveness of the PVI increases with a decrease in canopy cover. Under wet soil conditions, the PVI tends to underestimate the leaf area index. The effectiveness of the PVI under the present study area conditions is, however, in direct contrast to the observation by Huete (1988, quoted by Duncan, 1993) that both the NDVI and PVI do not give a consistent estimation of vegetation under sparse vegetation conditions. The divergent observations and standpoints suggest that the effectiveness of a vegetation index in a given environment cannot be directly extrapolated to another. As pointed out by Eastman (1999), the use of any of the vegetation transformations depends on general geographic characteristics of the application area.

In Chapter 4, section 4.4.5, parallels were drawn between *P. incana* and the saltbush of southern Australia in terms of their uncharacteristically low infrared reflectance. This contrasts with normal vegetation, which, supposedly, reflects strongly in the infrared wavelengths (Lillesand and Kiefer, 2000). However, this pattern of spectral response is based on observations of single leaves. The main factors controlling canopy reflectance include the size and orientation of leaves, the physiology of the component species, the nature of the underlying soil and the amount of vegetation cover, which control the contribution of the background to the spectral response (Armitage *et al*, 2000).

Malthus *et al* (1993) observe that the cellular structure of the leaf, and therefore leaf area, largely determine infrared reflectance. *P. incana's* physiology characterised by semi-deciduous thin and sessile leaves with a hairy covering should influence the shrub's unique spectral response. When dormant and photosynthetically inactive during dry spells, the aboveground biomass for *P. incana* appears as dead material. Graetz and Gentle (1982) show that certain shrubs are not usually reflective in the infrared band and appear only slightly different from dead litter as opposed to the distinctive infrared signature of green vegetation. This view is supported by the observation by Ringrose *et al* (1989) that semi-arid vegetation signatures do not give good spectral separation, as the infrared reflectance can be higher from the soil than from vegetation. The scatter effect of the infrared reflectance by soil is shared between shrubs and bare surfaces. The poor response of *P. incana* to ratio-based vegetation indices (e.g. the NDVI) could be ascribed to the shrub’s phenological attributes. The influence of seasonal variations on the spectral characteristics of *P. incana* was not assessed. Further research is required to gain an understanding of the influence of the variations on the shrub's spectral response. Against the background of the effectiveness of the PVI, it can be said that, depending on the local geographical characteristics, the PVI is particularly suited for the identification of perennial shrubs with characteristics similar to *P. incana,* which are usually interspersed with considerable bare zones.

7.4 Land disturbance and P. incana invasion

Abandoned and grazing lands have been identified as the main sites targeted by *P. incana*. Blanket invasion of most tracts of the former (77% of the total) as opposed to 39% of the latter was highlighted in Chapter 4, section 4.4.6. This constitutes a dichotomy of *P. incana* invasion trajectories as described in Chapter 6, section 6.4 which play themselves out with strong leanings against the two land use backgrounds. Perspectives on the invasion of the former prong are presented first.

7.4.1 Invasion of abandoned lands

The first question to be answered is: What makes abandoned lands so susceptible *P. incana* invasion? An understanding of the factors that underpin the susceptibility of abandoned lands to invasion would necessitate working back in time. The temporal component would then provide some background as to the sequence of events, which synergistically promoted the invasibility of these lands. The factors that influence the invasibility of a given environment were reviewed in section 2.2.2 of Chapter 2. Notable among them are changes in disturbance regimes and climatic fluctuations in a given region. The high success rate of invading species in disturbed environments was also alluded to. On abandoned lands, the creation of bare ground through cultivation, and removal of the pre-existing vegetation (Thompson *et al*, 2001) is the first in the sequence of events that render the lands vulnerable to invasion.

In the present study, a background to the timing of land use changes and climatic fluctuations was provided in sections 3.4 and 3.5 of Chapter 3. Progressive land abandonment between 1950 and 1970 coincided with a persistent drought period, particularly in the 1960s. Perspectives on the role climatic fluctuations are likely to have played are given in sub-section 7.4.1.1 below.

7.4.1.1 The influence of climatic fluctuations

The hypothesis that "a combination of drought conditions and land abandonment provided a niche for *P. incana* invasion and its gradual blanket establishment on abandoned land" is elucidated here. The persistence of bare abandoned plots as discerned from the 1975 aerial photographs is most likely to have been perpetuated by drought (see Figure 3.10). Age-long hardening, a process explained by Gerits *et al* (1990), resulting from earlier disturbance of particle-to-particle bonds was one of the effects that drought has on the bare soils. The process is influenced mainly by soil water content, clay content and time. On the basis of the values provided (Table 3.1), the local soils exceed the critical Exchangeable Sodium Percentage. Their high susceptibility to chemical dispersion and cementing was established. This condition is aggravated by the dominance of hydrous mica, characteristically highly prone to desiccation cracking.

Parallels can be drawn between the present study and the one by Buckland *et al* (2001) (see section 2.2.2.1) whereby a severe drought in 1995 is said to have created and sustained bare ground plots that promoted the invasion by *Thaliana* annual species with a persistent seed bank. Likewise, a combination of disturbance, drought and age-long hardening of the soil surface may have created less competitive conditions for other species and rendered the lands highly vulnerable to blanket invasion by *P. incana*. The impairment of the soil surface conditions has strong implications for resource fluctuation, a concept examined below.

7.4.1.2 Resource fluctuation

Davis *et al* (2000) articulates the theory of 'fluctuating resource availability', such that meteorological fluctuations and site-specific events, such as large or small-scale disturbances, could cause resource fluctuations. In the present study, drought, soil physical and chemical properties and disturbance in the form of land abandonment are envisaged to have interacted to cause a shift in water availability beyond the uptake threshold for grass species. This scenario tallies with the two-level moisture theory which is discussed in section 7.6.3. This shift must have augured well for replacement of the resident grass species by *P. incana*.

Resource availability as it pertains to the present study differs from that articulated by Davis *et al* (2000). The current situation in Mgwalana catchment cannot be explained in terms of either a reduction in resource uptake by the resident species, an increase in resource availability due to a pulse in resource supply, or both. Impaired soil surface conditions are seen to have reduced infiltrability, promoted runoff and reduced soil moisture beyond the rooting depth of grass species. Therefore, gross resource loss (both water and nutrients) rather than an increase in gross resource supply occurred. This concurs with Davis and Pelsor (2001) (see section 2.2.2.2) where disturbance increased invasibility when the gross resource supply was low when plots were dry. The invasion of abandoned lands depends on the spatial and temporal coincidence of resources, which are impacted upon by human disturbance, soil physical and chemical properties and climatic fluctuations. Further appreciation of the invasion of abandoned lands can be achieved when examined in conjunction with terrain parameters (see section 7.6).

7.4.2 Invasion of grazing land

Perspectives on grazing land, the second prong of the invasion by *P. incana*, are presented in this section. Unlike the case with abandoned lands, the invasion of grazing land is largely a competition and replacement scenario. For as long as there is no shift in soil water resource availability below the uptake threshold for grass species, *P. incana* is unlikely to gain a competitive advantage over grass species. Site-specific factors that are likely to cause this shift range from soil loss, soil surface condition, particularly crusting, and continual low grass biomass. The ArcView GIS database of field observations was used to interrogate grazing sites. All sites where *P. incana* had fully established itself had severe erosion forms. Sites where *P. incana* was in the process of invading all had mild erosion forms. *P. incana* invasion was noted as non-existent at all the sites with no visible erosion. By implication, soil loss triggered by (*P. incana*) invasion (see section 7.9) gives rise to the shift referred to above, tilting the balance of available resources in favour of *P.*

incana. The strong spatial correlation between erosion and crusted soil surfaces was established (sub-section 6.2.2.2). This would further aggravate the resource shift, and inevitably, competition will give way to replacement.

Site-specific personal observations have shown a consistent reduction to ground level of grass biomass due to continuous grazing by domestic herbivory. This would reduce the competitive ability of grasses by decreasing root production and resource uptake, allowing the use of more soil resources, particularly moisture, by shrubs (Montana, 1995). This phenomenon should explain the replacement of palatable grasses by unpalatable shrubs observed by Moretto and Distel (1999) on grazing land. As was pointed out in section 6.4, the impact of overgrazing cannot be overemphasised as the relationship between *P. incana* invasion and grazing activities is not a linear one. Tracts of rangeland still in pristine condition can still be seen in close proximity to villages. *P. incana* also seems to respond to any form of disturbance that is likely to disrupt the balance of available resources. For instance, the role of minor disturbances, e.g. termite mounds and mole rat colonies cannot be underestimated. These could act as nuclei from which *P. incana* patches may spread and coalesce into the terrain.

7.5 The Equilibrium / non-equilibrium debate as it applies to P. incana invasion

The divergent standpoints adopted by proponents of either the equilibrium or non-equilibrium systems were presented in section 2.2 of Chapter 2. The question that arises here is *viz*.: could *P. incana* invasion be explained in terms of either the equilibrium or non-equilibrium systems? The replacement of grass species by *P. incana* cannot be explained in terms of predictability, stability, and sequential succession as expounded by the equilibrium concept. The paradigm implies a linear, predictable response of the vegetation to grazing pressure and range condition. It is interpreted as a stage in plant succession, which can be manipulated predictably with stocking rate (Vetter, 2003). It is in this vein that Fynn and O'Connor (2000) attribute vegetation compositional changes and local extinction of grass species such as *Themeda triandra* to heavy grazing. Proponents of the non-equilibrium concept on the other hand (cf. Sullivan, 1999; Sullivan and Rohde, 2002) perceive grazing as having a negligible influence on vegetation cover and composition. They contend that arid rangelands appear to be resilient to long-term intensive grazing. The grass sward is dominated by annual grasses which grow from a seed bank in subsequent wet years, during which biomass production is more or less proportional to the amount of rainfall. During drought periods, there may be literally no grass to overgraze.

Such observations cannot be extrapolated to all semi-arid environments and the belief that overgrazing is totally innocuous to rangeland condition is extreme. In competition and resource fluctuation environments like the one in the present study, the dynamics presented above may not apply. More resilient species like *P. incana* will gain an upper hand, resulting in replacement. This standpoint may also not be valid in environments of restricted mobility of livestock. Limited access to grazing land will undoubtedly engender grazing pressure and decimate the competitive ability of grass species.

The non-linearity of the relationship between grazing and vegetation change has been alluded to in section 7.4.2 above. There is also a need to consider stochastic variations that can lead to unpredictable jumps in the behaviour, for example from grass to bush as articulated by Thornes (in press). For instance, the response of area-specific soils to stochastic events as discussed in section 7.4.1 above and the coupling with local topographic variations, which in turn control soil resource availability and flux need to be understood. As suggested by Vetter (2003), more research is needed to gain a predictive understanding of the response of different types of vegetation communities to continuous high-intensity grazing.

Competition and replacement as observed in the present study can, to some extent, be explained in terms of the non-equilibrium paradigm which *inter alia*, takes cognisance of the implications of rainfall cyclicity and resource fluctuations, particularly soil moisture, as the main underpinnings. Within such a stochastic environment, topographic and edaphic controls on resource availability largely determine competitive advantage. The two-level moisture theory examined in section 7.6.3, which has a strong bearing on *P. incana's* competitive advantage, is a culmination of these controls.

As pointed out by Vetter (2003), the two paradigms of equilibrium and non-equilibrium are extremes along a continuum. Despite some controversial standpoints propounded by either paradigm that have become scientifically and logically questionable, common ground does exist. A balance between the two paradigms would be the most realistic path to pursue as a means of gaining an understanding of the intricate rangeland dynamics that give rise to vegetation change.

7.6 Topographic factors and P. incana occurrence

The joint influence of slope angle and aspect on *P. incana* presence or absence was established (section 5.3). The threshold slope and aspect for *P. incana* occurrence were identified. The logistic regression used to establish this dichotomy, however, neither assumes a linear nor a causal relationship. A closer look at the two topographic variables indicated that the steeper the slope angle, the greater the density of *P. incana*, with categories above 10^0 having the greatest density. A high frequency of the 'established *P. incana*' class was observed during field surveys as most pronounced in slope categories above 10^0. Slopes with a southerly orientation were identified as having a greater *P. incana* density than the other slope directions. However, the two topographic variables cannot single-handedly explain *P. incana* distribution. It is necessary to tease out the underpinning attributes of terrain and co-parameters that contribute to this relationship. *P. incana* density variations with slope angle are examined first, followed by aspect.

7.6.1 ***P. incana*** **density variation with slope angle**

The relationship between slope angle and *P. incana* invasion can be explained in terms of processes that operate on slopes. It is noteworthy that all the assumptions made in this sub-section are presented against the background of the aggravating circumstances that pervade the study area, *viz.*:. land disturbances and the attendant forms of soil degradation. The implications of slope angle for the *P. incana* invasion are manifest in the interaction with surface and sub-surface water flux and redistribution, with slope gradient as the controlling factor. The steeper the slopes, the greater the potential for surface and sub-surface water movement. Steep slopes have lower runoff thresholds, thinner soils, reduced soil moisture storage and consequent sparse vegetation (Kirkby *et al*, 2002). Little wonder, therefore, that steep slopes in the categories 10^0-14.9^0 and $>15^0$ were identified (section 5.3.3) as having the highest *P. incana* density.

In view of the above, steeper elements of slope experience greater resource fluctuations, particularly soil moisture. The fluctuations are likely to be greater under aggravating circumstances of land disturbance and soil surface crusting as is the case in the present study. The 'Walther model' and its implications for competition between grass and shrubs will therefore also be manifest. Deep-rooted shrubs like *P. incana* will gain a quicker competitive advantage and eventual replacement of grass species, particularly in crusted soils. Regarding surface/lateral water redistribution, the propensity for runoff generation and stripping away of the humic layer is greater on steep slopes than gentle ones. The resultant formation of erosional crusts and accompanying shift in available resources will have provided favourable conditions for the more rapid invasion of steep slope elements. Runoff / run-on systems which in turn act as source-sink areas that sustain vegetation patchiness are primarily controlled by the slope angle of contiguous slope elements.

The above review of topographic control of *P. incana* invasion is principally premised on a broad catchment scale. It is worth mentioning that *P. incana* invasion and patchiness dynamics operate at the slope scale, particularly the first order basin divides. It is at this scale that local topographic control on soil moisture fluctuations which determine the competition between grass species and shrubs becomes predominant. As pointed out by Rowntree *et al* (in press), the distribution of grass and bush is linked to the strong coupling between local topographic variations and soil depth, topography and moisture movement. The Wetness Index, which is a derivation of the TOPMODEL, is based largely on local topographical variations. That the index is central to moisture variability modelling and its implications for the spatial distribution of *P. incana* was demonstrated (section 5.3.6). Local topographical control on *P. incana* distribution is examined in section 7.6.3. *P. incana* density variations with aspect are discussed in section 7.6.2 below.

7.6.2 *P. incana* density variation with aspect

The question to be answered in this section is: why are the south facing slopes more invaded by *P. incana* than the other slope directions? Naturally, topographic trends of the study area were noted to be more expressed on slopes with a southerly orientation whose frequency is greater than the rest of the slope directions (see section 3.2). The land area in the steep slope categories was noted as greater on the south than the north facing slopes. However, a greater southerly topographic orientation alone cannot explain the predominance of *P. incana* invasion. If it were to be so, then the north facing slopes, whose frequency is second to the south facing ones would be the second most invaded slope direction. On the contrary, easterly oriented slopes followed the southerly ones (section 5.3.2). In view of the fundamental control of slope steepness on the soil moisture budget and other slope processes, the higher occurrence of steep slopes on the south facing slopes could to a reasonable extent explain the greater invasion by *P. incana*. Indeed, the prevalence of the steepest slope categories of 10^0-14.9^0 and $>15^0$ is twice as much on the south facing slopes as the north facing ones.

The role of aspect in *P. incana* invasion can be better appreciated when viewed in combination with land use and slope angle. The link between aspect and land use was explored in section 5.3.3 of Chapter 5. The facts identified should account for this relationship, viz.:

- Most of the abandoned land is located on the south facing slopes.
- South facing slopes were by far the most farmed before abandonment.
- Invaded abandoned land with high *P. incana* density is predominantly located on the south facing slopes.

- The acreage of invaded abandoned land is mostly in the slope category 5^0-15^0, with a considerable acreage in the above 15^0 category as well.

The facts above highlight the interaction between land disturbance, slope angle and aspect. As noted previously, the role of the first two factors in *P. incana* invasion is more direct than that of aspect, which plays an indirect role as an independent medium in this interaction. The abandonment of land on steep, mostly south facing slopes made it more vulnerable to resource fluctuation, impairment of soil biophysical properties, formation erosion crusts and erosion, all of which promote *P. incana* invasion. It can thus be summed up that disturbed lands in the form of abandonment on steep south-facing slopes are highly susceptible to invasion by *P. incana*.

The fundamental control that slope aspect has on the spatial variability of soil moisture explains why south facing slopes were the most farmed. The slopes are supposedly shadier than the sunny north-facing ones and, given the geographical location of the study area, in close proximity to the east coast, the dominant onshore moist winds make the slopes moister and thus more suitable for farming. This should, however, not be seen as a contradiction with the concept of the Wetness Index which is examined in section 7.6.4 below. The moist conditions referred to here are meteorology driven as opposed to the topographically based Wetness Index. Whereas the former operates at a regional scale, the latter plays itself out at the local slope scale.

Given the wind-spread nature of *P. incana* seeds and the dominance of the south east onshore winds, south oriented slopes could be more predisposed to seed propagation than the other slope directions. The identification by Leprun (1999) of orientation of the tiger bush to be the direction of the dominant harmattan winds was referred to earlier (sub-section 2.3.3.2). Further research as to whether *P. incana* itself has an inherent inclination to slope aspect needs to be done.

The review in sections 7.4 and 7.6 above highlights the synergy between anthropogenic and topographic underpinnings of vegetation invasion. The implications of the latter for *P. incana* distribution are discussed in the subsequent sections. The disregard for terrain parameters was identified as a critical omission from the invasion ecology investigations (section 2.2.2). Cognisance must be taken of the role of such spatially variable and scale related factors as a prerequisite to the understanding of the dynamics of invasibility and invasiveness of a given environment and species.

7.6.3 The role of local topographic heterogeneity in *P. incana* distribution

The broader topographical components of slope gradient and aspect and their influence on *P. incana* occurrence have been examined above. However, as pointed out in section 7.6.1 above, *P. incana* invasion and patch dynamics operate at the local topographic scale. It is at this level that the fine details regarding the underpinnings of its distribution, from single patch to entire hillslope scale, can be unravelled.

The analysis of local topographic variations in relation to *P. incana* distribution using TOPOSHAPE and CURVATURE in Idrisi Kilimanjaro indicated that *P. incana* is more established on convex than concave elements of slope. The reverse is true of grass species, which have in many cases been restricted to hollows and other localised points of origin of surface drainage. Such a distribution is a clear expression of the kingpin role played by local topography in a number of processes that underpin resource distribution, setting the scene for a competitive environment between *P. incana* and grass species. This is essentially the scale at which most hillslope processes are initiated. These processes include:

- The redistribution of both surface and sub-surface flow, which in turn determines soil moisture distribution.
- Soil removal from convexities and its accumulation in concavities. This in turn determines soil depth, which draws a line in the competition between shrubs and grasses.
- Modification of soil surface condition, particularly formation of soil surface crusts.
- The interaction between vegetation patchiness and runoff flow paths.

The strong implications the local landscape has for soil moisture variability were highlighted (sub-section 2.3.3.4). That local topographical configuration determines the pattern of runoff convergence and divergence, as well as shallow sub-surface flow, was confirmed. Runoff divergence on convexities like spurs and ridges translates into reduced soil moisture storage, particularly in the topsoil layers. The convergence of flowlines into hillslope hollows makes them zones of water concentration in the landscape. The different soil moisture dependencies of *P. incana* and grass species, such that the former can tap deep soil moisture as opposed to the latter which is dependent on surface soil moisture, explain their preferential locations. The limited soil moisture budget on slope convexities enables *P. incana* to gain a competitive advantage over grass, while the pronounced moisture storage element in concavities enables grasses to persist.

The prominence of convexities in *P. incana* invasion dynamics is demonstrated by the isolated *P. incana* patches, which were identified (section 5.3.5) as originating from spurs and ridges. Given their soil moisture status, they act as the starting point from which *P. incana* establishes itself into the terrain. On abandoned lands, many of which were noted as having remained bare for long periods, convex slope elements must also have facilitated rapid blanket *P. incana* invasion as they must have been drier and non-conducive for grasses to re-establish themselves.

With regard to soil thinness and accumulation, it is at the scale of local topography that the 'Walther model' as it applies to soil depth differences plays itself out. Convex and steep elements of slope tend to have thinner soils (Kirkby *et al*, 2002) as opposed to concavities, particularly hollows where soils accumulate, owing to hillslope transport processes. The model articulates that soil depth determines the competitive advantage between grasses and shrubs or bushes. In the same vein of different moisture dependencies explained above, *P. incana* will access the limited moisture resources of the thin soils on the convexities while grass persistence will be possible in the deep, usually saturated soils of the hollows.

The differences in moisture dependencies between *P. incana* and grass species as determined by soil depth are illustrated by the root distribution analysis (section 6.5). The greater rooting depth of *P. incana* (90cm – 1m) than grass (40cm-60cm) demonstrates the former's competitive advantage in line with the Walther model. The rooting depth differences serve to confirm the observation by Thornes and Brandt (1994) that competition between the plant species comes from the limit to total resource available after the deep-rooting species have satisfied their need. However, the two-level moisture scenario was noted to fall away under shallow and crusted soil conditions, particularly on abandoned lands where the topsoil layers have been removed by erosion. It is under such circumstances that the deep rooted *P. incana,* which is also able to tap some of the surface runoff (see section 7.7), gains outright competitive advantage, resulting in blanket invasion.

The modification of soil surface conditions, particularly soil crust formation, is supposedly more pronounced on convex elements of slope than the concave ones. Owing to the greater soil removal from the former than the latter, the susceptibility to the formation of erosion crusts is greater on convexities than concavities. Crusting on convexities in turn gives rise to low infiltrability and hence reduced moisture storage within the soil, a setting conducive for *P. incana* to establish itself. The role of soil crusting and its influence on *P. incana* distribution is examined fully in section 7.8 below.

The pattern of *P. incana* patchiness and its interaction with runoff flow paths on convexities manifests itself in wider inter-patch areas on the steep sections of convex slopes. As pointed out by Kirkby *et al* (2002), the natural response of vegetation to the thin soils of steep slope sections is producing a sparser cover. Wider inter-patch areas give rise to runoff connectivity, which may in turn lead to the expansion of bare areas as was noted in section 5.4 of Chapter 5.

As noted, the overriding control of local topography on *P. incana* spatial distribution is rooted in soil moisture variability. The Topographic Wetness Index, which is based on the local topographic heterogeneity, is examined below.

7.6.4 The Topographic Wetness Index and *P. incana* distribution

The topographically driven Wetness Index (WI) derived from the respective DEM windows (section 5.3.5) augments the findings regarding the influence of local topography on soil moisture variability and the implications for *P. incana* distribution. The strong correlation between the WI and field moisture values, illustrated by Figure 5.12, confirms the reliability of the latter as a tool to predict slope configuration induced moisture variability. A distinct divide between *P. incana* and grass clusters is illustrated by the figure. *P. incana* is associated with low WI values as opposed to grass, demonstrating that their moisture dependencies are largely controlled by the local topographical variations.

The question to be answered here is: to what extent does the WI predict soil moisture variation and hence *P. incana* distribution? Anderson and Burt (1990) recommend 10^0 as the threshold slope for computation of the WI due to the minimal topographic control on soil moisture variability on slopes less than 10^0. Understandably, such a limit should signify well defined local topographic variations which clearly determine both surface and subsurface flow. The recommendation notwithstanding, the WI was computed from DEM windows representing all the selected High Resolution images regardless of slope angle differences. WI values computed from DEM windows representing slopes ranging between 4^0 and 9^0 correctly predicted *P. incana* and grass clusters. The locations of either vegetation type were identified as the characteristically low and high soil moisture sites surveyed during the series of field visits. This implies that, despite the minimal topographic control on soil moisture conditions on slopes less than 10^0, to some extent, moisture conditions which determine *P. incana* and grass distributions could be still predicted by the WI with some degree of reliability on less steep slopes.

The WI, however, may not fully explain soil moisture variability and hence *P. incana* distribution. The test of whether the WI reflects field moisture values (section 5.3.6), was based on *a priori* field selection of homogeneous *P. incana* clusters and grass patches, most of which happen to be located on spurs and hollows respectively. The predictive ability of the topographically driven WI for such distinct topographic units should be high. The test, however, identified some *P. incana* clusters supposedly having high WI values. Likewise, other areas with low WI values were also identified as devoid of *P. incana*. An appraisal of slope angles for these areas indicates that they are steep enough for the topographic WI to take effect. Such an atypical relationship implies that the WI and therefore *P. incana* presence are not controlled by local topographic variations alone. Other factors other than the WI are important. This is more so at the more localised patch scale as compared to entire slope configuration units. As pointed out by Hornberger *et al* (1998), the following presuppositions underlie the application of the WI:

- The water table is parallel to the soil surface
- The saturated hydraulic conductivity decreases exponentially with depth
- Steady state conditions prevail in the catchment
- Groundwater recharge rate is uniform

Whereas the first two presumptions are realistic, the last two are controversial. Heterogeneity may exist even within the same topographic unit due to varying soil properties, underlying bedrock and outcrops of bedrock, causing the conductivity profile to vary greatly (Rhode and Seibert, 1999, cited by Asserup and Eklöf, 2000). The steady state assumption inherent to the WI does not consider such small scale variations. Sampling strategy also may influence the predictive ability of the WI. The weak correlations identified by Asserup and Eklöf (2000) between the WI and field moisture values could be explained by the use of point samples. Such samples are not representative of spatial organisation, as wetness of a given place cannot be determined using only a single point.

In the present study, the soil moisture disparity and *P. incana* distribution scenario is compounded by, *inter alia*, the coupling between land disturbance cycles and the heterogeneous soil surface conditions characterised by crusting and erosion. Such forms of soil degradation, particularly on abandoned land, will have altered soil moisture conditions. Gully erosion in particular targets deep soils in hollows or concave lower slope elements, which are supposedly topographically high wetness areas. This will induce “inverted topographic wetness conditions” by drying out potentially high wetness areas. This explains the suggestion by Anderson and Burt (1990) that topographic Wetness Index methods may be inappropriate for slope sections that are gullied or where soil piping is well developed.

Low WI on convexities that are not invaded by *P. incana* may indicate that favourable soil surface and moisture conditions for grass survival still exist, regardless of the local topographic configuration. This also points to the likelihood that the observed strong spatial correlation between *P. incana* and the local hillslope configuration reflects the contribution by other factors to the influence of topographic variability on soil moisture distribution. These factors include land disturbance, particularly abandonment, which more than anything else engendered alteration of soil surface conditions. Vegetation patchiness, which is an integral component to the moisture redistribution dynamics on hillslopes, is examined in the next section.

7.7 Implications of P. incana patchiness loss for landscape function

Patchiness as an attribute of *P. incana* pattern was examined (section 5.4). *P. incana* and bare surface mosaics were identified as a prominent feature from boolean images and field surveys of *P. incana* pattern (section 6.3) in areas with a predominance of *P. incana* cover. Evidence of expanding bare surfaces with time due to patchiness loss was also presented by means of time series photographs and an assessment of *P. incana* pattern parameters (sections 5.4 and 6.3). The analysis of *P. incana* / bare surface ratios (Figure 5.16) indicated that the ratio sizes are not related to slope angle and that bare surfaces are interspersed randomly among *P. incana* tussocks.

It was pointed out in section 2.3.4 that vegetation pattern is an expression of its influence or lack of it on water redistribution on the land surface. A consistent pattern of vegetation and bare surface mosaics would reflect a self-regulating negative feedback mechanism characterised by distinct runoff/run-on and erosion/deposition zones (Cammeraat and Imeson, 1999; Holm, 2000). On the contrary, the lack of a consistent pattern characterised by loss of patchiness, as is the case with *P. incana*, indicates a positive feedback mechanism whereby enhanced runoff flowpaths could coalesce into larger pathways (Janeau *et al*, 1999). The only exception where *P. incana* displays some consistent pattern is its orientation to contouring on abandoned fields (Figure 5.16). This is in keeping with the observation by Cammeraat and Imeson, (1999) that shrubs may develop in parallel strips, a patterning that could have resulted from the inheritance of parallel contour ploughing micro-topography. This was observed to be the case with *Stipa tenacissima* patchy vegetation patterns in SE Spain.

The decrease in *P. incana* tussock height, number of stems, cover length and obstruction width with increasing fetch as observed under steeply sloping field conditions (section 6.3) confirm the patchiness loss phenomenon. The bearing this phenomenon will have on runoff / run-on dynamics is articulated thus: "the loss of patches occupied by long lived plants will alter dynamics of runoff/run-on and erosion/deposition by increasing the fetch, resulting in each runoff event involving larger volumes of water" (Holm, 2000, p23).

In view of the above, the question posed is: What are the implications of *P. incana* pattern for landscape function? Ludwig *et al*, (1999) (see section 2.3.4) demonstrate the bearing of vegetation patchiness pattern on runoff generation and ultimately landscape function. In the present study, bare surface mosaics on many hillslopes interconnect randomly while tussocks of *P. incana* display a stippled pattern with considerable fetch lengths between individuals or groups of them due to loss of patchiness (section 5.4). As was pointed out in section 2.3.4, such pattern would imply runoff connectivity and with much of the runoff becoming run-out, tremendous soil loss would be inevitable. Run-out amounts are exacerbated by the loss of vegetation patchiness, as is the case with *P. incana*. As observed by Ludwig and Tongway, 1995; Pickup *et al*, 1998 and Holm, 2000, the loss of vegetation patchiness and an increase in landscape heterogeneity would give rise to the conversion to a dysfunctional system.

Deductions that can be made from the spatial distribution of *P. incana* and bare surface mosaics are *viz.*:

- *P. incana* patches have limited capacity to trap nutrients, sediments and runoff on hillslopes.
- Given the steeply sloping conditions of most of the affected areas, the bare ground zones can be seen as an expression of runoff connectivity and inevitable erosion.
- A ‘leaky’ non-resilient system where runoff readily becomes run-out prevails under such vegetation–bare surface mosaic conditions.
- The potential for conversion to a dysfunctional system is high as the system progressively loses more of its patches due to erosion pressure.

If *P. incana* patchiness bears hallmarks of a dysfunctional system, another question to answer is: to what extent is *P. incana* pattern an expression of dysfunctionality? Despite the manifestation as a leaky system, it is noteworthy that *P. incana* and bare surface mosaics are not run-out systems in absolute terms. Some semblance of resource capture exists which could, with appropriate intervention strategies enable the system to flip to recovery. Post-recharge rainstorm soil moisture readings as surrogates for runoff indicated significantly greater moisture storage under *P. incana* tussocks than under bare zones upslope of the tussocks (section 6.7).

The variation in soil moisture is in keeping with observations by different scholars (Thiéry *et al*, 1995; Seghieri *et al*, 1997; Bromley *et al*, 1997; Galle *et al*, 1999; Macdonald *et al*, 1999; Valentin and d'Herbès, 1999) who conclude that soil moisture variations in vegetated-bare zone mosaics are a function of the water harvesting efficiency by vegetated patches. In the present study, the coupling between *P. incana*’s aboveground and litter interception creates a sink area for a component of the runoff generated on the bare zones immediately upslope of the *P. incana* patch. This runoff redistribution pattern has important implications for strategies to be adopted in the rehabilitation of highly degraded *P. incana* and bare zone mosaics, as cognisance must be taken of the latter’s runoff harvesting function. Soil crusting, which is inherent to vegetation-bare zone patterns, is considered in the section below.

7.8 The relationship between soil crusting and P. incana patchiness

Soil crusting was identified during field surveys as a prominent feature of bare surfaces between *P. incana* tussocks. The frequency of crusted soil surfaces was noted as extremely high in ‘established *P. incana*’ areas, moderate in partially invaded areas and isolated *P. incana* patches, as opposed to no visible crusting in the uninvaded areas. Penetrometer measurements of soil surface strength (a surrogate for crusting) also indicated that soils in the inter-*P. incana* bare zones are highly crusted (Figure 6.13). Chemical analyses of soil samples confirmed that the local soils have a very low crusting threshold (section 3.2). The tendency to develop crusted soil surfaces is a phenomenon unique to vegetated-bare surface mosaics (section 2.3.3.3). A twofold question that needs to be addressed is *viz*.:

- What are the implications of soil crusting for *P. incana* patchiness and vice versa?

In the review of the literature regarding crusting, it was established that in prior-to-disturbed environments, soil crusting precedes the development of vegetation patchiness. In the present study, this is applicable mainly to abandoned lands, which, as photographic evidence suggests, must have undergone considerable resource depletion, as they lay bare for long periods. The following conditions will therefore have taken effect:

- Physico-chemical dispersion of the soil, especially during the dry season, and the attendant age-long hardening - owing to the high sodicity and swelling mica content of the local soils.
- Conversion of the lands to resource poor status, particularly drastic reduction in soil moisture and soil loss on sloping ground. Severe erosion forms are, indeed, noticeable on aerial photographs.
- Inhibition of seedling emergence, particularly for grass species.
- Patchy colonisation by the more resilient *P. incana*, given the constrained soil moisture budget.

The set of conditions outlined above constitutes the implications of soil crusting for *P. incana* patchiness on the one hand. On the other hand, individual *P. incana* tussocks emerge and intersperse grass species, especially on grazing land. Individual shrubs expand into a ring formation (see Figure 6.6a, Chapter 6) and coalesce as *P. incana* gains competitive advantage. Crusted bare areas are noted to develop in the inter-patch area, confirming Valentin's (*pers. comm.*, 2003) observation that crusted zones are inherent to vegetation patchiness. In this instance, *P. incana* patchiness is seen to promote soil crust formation.

The significance of soil crusting in the inter-*P. incana* patch area can be summed up thus:

- Crusting is the main control on soil moisture variation at the patch scale. This stems mainly from the reduced infiltrability it engenders. As concluded by Valentin and d'Herbès (1999), "Due to the prevailing role of crusts in hampering infiltration, field survey of surface crusts appears to be a useful tool for predicting the spatial variation in soil moisture as shown by the consistency between simulations and field measurements" (Valentin and d'Herbès 1999, p250). It could thus be said that the coupling between the WI and soil crusting would be a good indicator of soil moisture variability.
- The competition between *P. incana* and grass species, and the eventual replacement of the latter, is therefore underpinned by soil surface crusting inherent to the former. This competition environment in part explains the dying off of grass species often seen in the vicinity of *P. incana* patches.
- Runoff generation occurs on the crusted bare surfaces, which may coalesce to create larger pathways. This is exacerbated by the loss of *P. incana* patches and the attendant expansion of bare areas, promoting runoff connectivity and erosion. To some extent, *P. incana* patches act as sink areas for some of the runoff generated on bare surfaces (see section 7.7 above). Extreme case scenarios have been reported in a study of the tiger bush by Galle *et al* (1999) whereby bare zones covered by erosion and gravel crusts were noted as generating large amounts of runoff, such that more than half of the annual rain becomes runoff.

The relationship between *P. incana* patchiness and erosion is addressed in section 7.9 below.

7.9 The relationship between P. incana patchiness and soil erosion.

A close spatial correlation between *P. incana* patchiness and varying degrees of soil erosion severity was identified during field surveys. Erosion severity was noted to increase with the extent of *P. incana* invasion such that severe rills and gullies are more pronounced where *P. incana* is fully established as opposed to no visible erosion in most of the remaining indigenous grasslands. Erosion occurrence in the invaded areas is no surprise as runoff generated on the exposed and crusted inter-*P. incana* bare patches with reduced infiltrability would beget erosion. This condition is exacerbated by the loss of patchiness, giving rise to the expansion of bare areas. The close spatial association between the degree of *P. incana* invasion and erosion raises questions *viz.*:

- Does erosion predate *P. incana* invasion, such that the propensity for the invasion of eroded areas is high?
- Does *P. incana* invasion predate erosion, such that the invasion induces the observed erosion?
- Or, do both scenarios apply?

Answering these questions would necessitate time series observations and interpretations of the invasion process, particularly from sequential photography and manipulative experiments. *P. incana* invasion was observed in the field in conjunction with baseline data obtained from two available sets of photography *viz*. 1975 aerial photographs and 1988 othophoto maps of the scale 1:15 000 and 1:10 000 respectively. Using these sources, it is possible to contribute some perspectives on the process. As pointed out in section 3.4, cultivated fields are discernible from the 1975 aerial photographs as bare surfaces, many of which are truncated by severe rills and, in some cases, gullies. The same erosion forms are noted to have worsened from the 1988 orthophoto maps. These fields had clearly been abandoned by this stage. By implication, much of the erosion on abandoned land predates *P. incana* invasion. Given the expansion of bare areas as illustrated (Figure 5.15a and b), erosion conditions may worsen under *P. incana* cover. However, cognisance must be taken of the runoff harvesting, aboveground and litter interception roles of *P. incana*. To some extent, *P. incana* does provide some ground cover without which the situation could have been far worse.

On the basis of personal field observations over a period between 2000 and 2003, the erosion initiation scenario on grazing land invaded by *P. incana* is envisaged as different from that of abandoned land. As described in section 7.8 above, *P. incana* patches establish themselves gradually within grazing land, usually grassland. The trajectories of how *P. incana* establishes itself into the surrounding terrain are fully mapped out in section 6.4, Chapter 6. Given *P. incana's* patchiness attribute, the inter-tussock bare areas are seen as promoting runoff connectivity and ultimately, erosion. The erosion conditions worsen as entire hillslopes are colonised. Under such circumstances, *P. incana* invasion is seen as predating and literally promoting erosion. This pattern of erosion, whose trajectories are largely land use based, has strong implications for rehabilitation strategies. These implications will be examined in section 7.11 where recommendations based on the findings of this study are made. A synthesis of the inter-linkages identified between variables that underpin *P. incana* invasion is provided in section 7.10 below.

7.10 Development of Conceptual Model

A conceptual model depicting the interplay between the critical variables and processes identified in this study as underpinning *P. incana* invasion is developed in this section. An understanding of the interaction between sets of independent and dependent variables would provide the basis for designing appropriate landscape rehabilitation measures. The linkages are depicted by Figure 7.1 below.

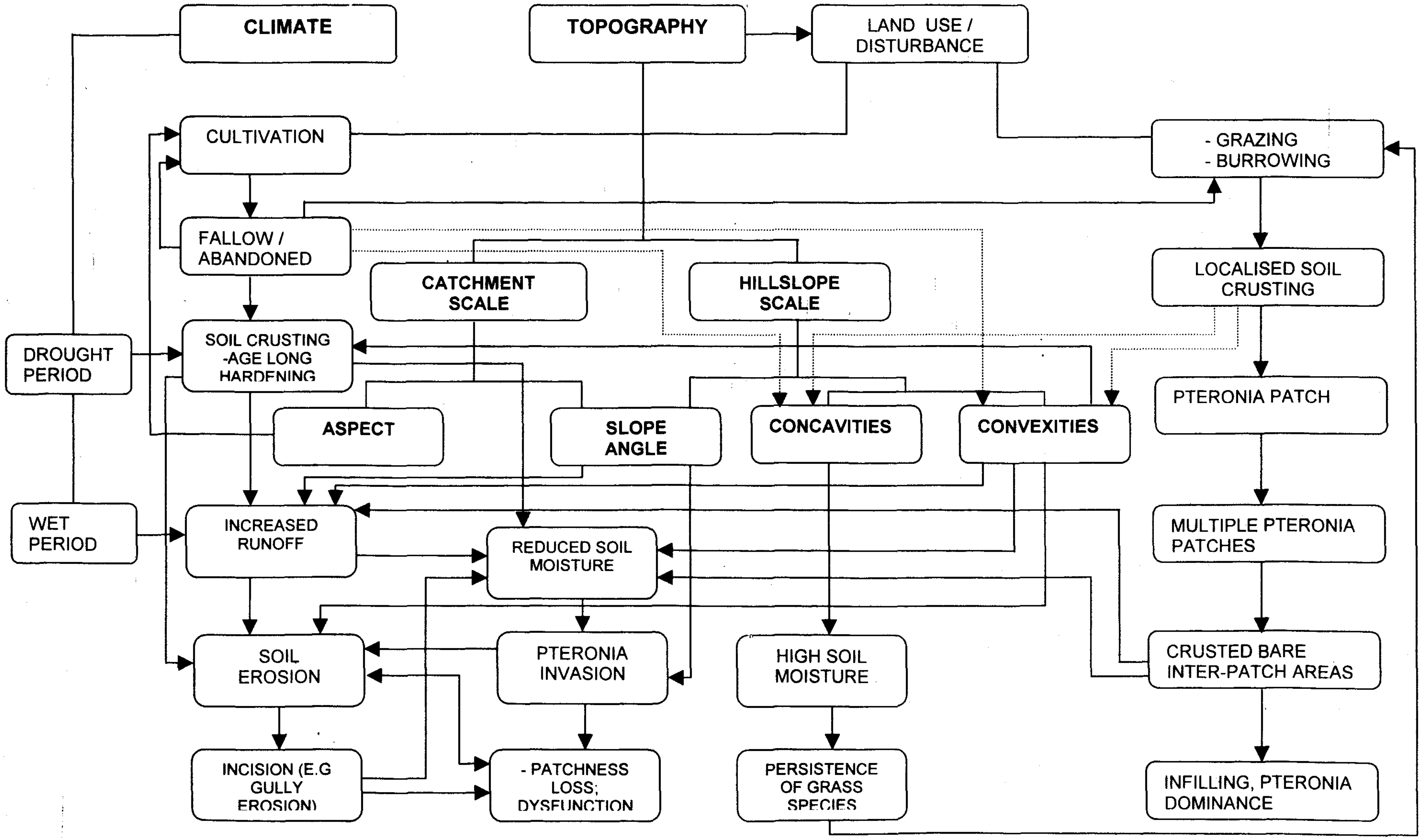


Figure 7.1 Conceptual Model depicting inter-linkages between variables that underpin *Pteronia* invasion. Independent physical factors are bolded. Dotted lines indicate variable dependency rather than influence.

7.11 Recommendations for rehabilitation

The implications of *P. incana* patchiness for landscape function have been elucidated in the present Chapter as well as the results Chapters 5 and 6. Having established the dynamics of *P. incana* spatial distribution in relation to the various human and physical factors, recommendations for the rehabilitation of the degraded landscapes are provided in this section. A dichotomy of invasion trajectories between grazing and abandoned land in terms of gradual invasion of the former as opposed to blanket invasion of the latter was identified. The dual invasion prongs would invoke land use based area-specific rehabilitation strategies.

On grazing land, *P. incana* could be manually cleared depending on the degree of invasion. Patches of the shrub that coexist with grass could be uprooted to restore the latter's competitive advantage. Due to the persistent seed bank *P. incana* possesses, regular follow-up clearing exercises would be necessary to prevent the shrub from re-establishing itself. This strategy would serve to restore some of the grazing land lost to *P. incana* and ease the grazing pressure from the rapidly shrinking grassland remnants. Burning is another viable alternative to manual clearance. Fire plays a major role in eradicating *P. incana*, as the shrub is intolerant of even a single fire event. The constant reduction of grass biomass to ground level due to continuous grazing was alluded to in section 7.4. This would make *P. incana* more predisposed to fire than grass. Besides, the latter would sprout easily after a fire as opposed to the former, which is fire intolerant.

The invasion scenario on much of the abandoned land constitutes a landscape function crisis. It was noted that the expansion of bare areas due to *P. incana* patchiness loss has created an extremely leaky run-out landscape that has been converted to a dysfunctional system. The restoration of such landscapes is inevitably a labour-intensive undertaking. Clearing and burning of *P. incana* would not be an appropriate approach, as they would exacerbate erosion conditions on the extensive crusted bare surfaces. The starting point should be the recognition of the fact that *P. incana* provides some ground cover without which erosion conditions would be far worse. Its interception role and limited enhancement of runoff infiltration under *P. incana* patches was noted. Patchiness pattern is another important consideration. As pointed out by Ludwig *et al* (1999), knowledge of the patchiness pattern can contribute to the formation of appropriate rehabilitation strategies and programmes. The pattern of *P. incana* has been identified as inconsistently stippled, with expanding bare areas.

The focus of rehabilitation efforts on degraded abandoned land should be on trapping of sediment and litter, and moisture retention on the inter-patch bare areas. This is achievable by laying structures that trap and store the limited soil resources (Tongway and Ludwig, 1993). Moisture deficiency has been identified as the main underpinning in the competition between grass species and *P. incana*. Trapped sediments will restore the two-level moisture conditions conducive for grass to re-establish itself. The coexistence of the two has been observed where moisture conditions are favourable. Once grass is re-established in the hitherto crusted bare areas, *P. incana* tussocks could then be systematically cleared and replaced by more sediment traps to encourage grass growth. The success of such an exercise would necessitate grazing control, such that animals are kept out of the areas where rehabilitation is underway. The total participation of the local communities would have to be ensured.

In line with the technique outlined above, Ludwig *et al* (1999) recommend the building of piles of brush oriented along contours. According to the experimental work done on banded vegetation in eastern Australia, soil sediments and litter were noted to have significantly accumulated within brush piles after three years. Water infiltration rates increased 10-fold and perennial grasses and forbs had re-established themselves within the brush piles as opposed to the controls. All this was achieved with some moderate grazing pressure. The rehabilitation of dysfunctional hillslopes in the present study would necessitate site-specific designs of runoff and sediment traps in the interconnecting eroded bare surfaces. Sites characterised by extremely wide bare areas and incision in the form of severe rills on fairly steep slopes would require larger piles of brush laid perpendicular to slope direction. These would be effective in trapping large runoff volumes generated over wide bare surfaces and incision areas. Brush could be selectively obtained from the woodland zone of the lower catchment. *P. incana* could also be selectively uprooted and used to form brush strips on hillslopes sections where bare areas are not very wide. Cognisance should be taken of the local topographic configuration when laying structures to trap sediment as it governs water redistribution on hillslopes. Due to the limited moisture retention capacity, thinner soils and less likelihood of seepage on spurs and other convexities, trapping structures should be laid more closely and aligned to the slope configuration. Other micro-topographic features like depressions would act as runoff convergence and moisture concentration areas, thus enhancing a more rapid re-establishment of grass species. Runoff escape routes that are not natural drainage lines should be identified and closed off.

Sites with regularly stippled pattern of *P. incana* tussocks could have thinner brush piles laid in the inter-patch bare areas given their relatively better water harvesting capability. With time, sediment and litter accumulation within the piles will enhance moisture retention in the inter-patch areas, encourage grass growth, and eventually lead to the elimination of crusted bare areas. With regard to the badland areas, gullying at most of the sites predates *P. incana* invasion. An element of protective, albeit ineffective cover provided by *P. incana* in badlands is borne out by the existence of clusters of *P. incana* tussocks on overhanging gully crests. Conventional gully rehabilitation rather than *P. incana* control techniques would therefore apply at such sites.

7.12 Directions for future research

Potential underpinnings of *P. incana* invasion that lie outside the scope of the present study require further investigation. These include:

- The role of regional factors - Does *P. incana* have an inherent inclination to regional factors such geology?
- Influence of specific climatic phenomena: Do phenomena such as *El Nino* oscillations explain the shrub’s spread to biomes it is not resident to?

- It would be helpful to examine the phenology of *P. incana* and explore its potential for enhancing the shrub's invasion success. Do the unique phenological attributes contribute to the shrub's competitive advantage?
- The influence of seasonal variations on the spectral characteristics of *P. incana* needs to be assessed. Details of the shrub's temporal spectral response pattern would be useful for monitoring purposes using Remote Sensing techniques.

7.13 General Conclusion

In the present study, Remote Sensing methods and Geographic Information Systems have been used to identify the spatial distribution of *P. incana*, an invader shrub species that has been strongly linked to land degradation. It has been demonstrated that the quantitative spatial and spectral characteristics of *P. incana* can reliably be derived using Infrared Digital Camera High Resolution Imagery. Spatial analyses at a fine resolution of 1mx1m have revealed vivid and unique relationships between the shrub and anthropogenic and physical factors of terrain at the catchment, hillslope and patch scales.

A supervised classification based on *a priori* field surveys provided a consistent basis for determining the spatial distribution of *P. incana* and other cover types. The high classification accuracy levels achieved confirmed that the Infrared Digital Camera Imagery used in this study had minimal spectral distortions. *P. incana's* unique spectral response, characterised by low infrared reflectance, contrasts with normal photosynthetically active vegetation whose infrared signature is distinctive. A combination of this unique spectral characteristic, bare soil surface noise and its phenological attributes largely explains the shrub's poor response to ratio-based vegetation indices, e.g. NDVI, SAVI and MSAVI. In contrast to the ratio-based indices, the Perpendicular Vegetation Index distinctly separates out *P. incana* at different stages of invasion from bare soil surfaces and other cover types. This confirms its suitability for the detection of shrubs with a spectral response similar to *P. incana*, particularly in environments with substantial bare zones.

The use of High Resolution Imagery in conjunction with a Digital Elevation Model has enhanced our understanding of the influence of topographic variations on the spatial distribution of *P. incana*. At the catchment scale, slope angle and aspect significantly influence the shrub's absence or presence. The probability for *P. incana* occurrence increases with slope steepness and the southerly slope orientation. Whereas the influence of the former on *P. incana* occurrence is manifest in the interaction with surface and sub-surface water redistribution, the bearing of the latter is indirect, as it is explained in terms of the greater land disturbance due to cultivation on the south facing slopes than the other slope directions.

The interaction between land disturbance, slope angle and aspect to promote *P. incana* invasion is borne out by the high density of the shrub on abandoned land located mostly on steep south facing slopes. Against the background of the timing of land use changes and climatic fluctuations that prevailed in the study area, the combined role of land abandonment and stochastic climatic fluctuations like drought and the attendant age-long hardening of the crusting prone soils may have enhanced the invasibility of the abandoned plots. The hypothesis that "a combination of land abandonment and stochastic climatic fluctuations provided a niche for *P. incana* invasion and its predominance on abandoned land" is thus accepted.

At the hillslope scale, the preferential location of *P. incana* on convexities as opposed to grass species in concavities is an expression of the control of local topography on resources that underpin the competition between shrub and grass species. The spatial relationship between vegetation type and the topographically driven Wetness Index confirms that the soil moisture dependencies of the two species are largely determined by local topographic variations. Implicit in the topographically controlled soil moisture variability is the two-layer moisture theory as illustrated by the Walther model. Soil depth variations between convexities and concavities demonstrate the coupling between local topographic variations and soil depth, which in turn draws the line in the competition between shrubs and grasses.

Soil moisture dependencies and their implications for competition and replacement are complicated further by soil crusting, which is the main control on soil moisture variation at the patch scale. Localised soil surface crusting inherent to *P. incana* patches underpins the competition between the shrub and grass species, and the ultimate replacement of the former. Much of the *P. incana* invasion on grazing land takes this trajectory.

The implications of *P. incana* patchiness for landscape function are manifest in its limited capacity to trap sediments, nutrients and runoff on hillslopes. The loss of patchiness and the resultant expansion of bare areas is an expression of runoff connectivity, erosion intensification and eventual conversion to a dysfunctional system. The semblance of resource capture evidenced by greater moisture storage under *P. incana* tussocks than on bare zones portrays the above-ground and litter interception role played by the shrub's tussocks. These could serve as starting point building blocks for the rehabilitation of run-out hillslopes by way of laying run-on structures in the inter-patch areas.

The main research questions raised in section 1.2, which constitute the specific objectives of this study *viz*.:

- What are the factors that influence *P. incana* invasion?
- What is the relationship between *P. incana* and soil physical properties, particularly soil surface conditions?
- What are the implications of *P. incana* invasion for runoff generation and soil erosion?

have all been addressed.

This study has highlighted, among other things, the central role played by topographic parameters and geomorphic processes in vegetation invasions. That ecologists tend to ignore the wider topographic and geomorphic ramifications in their investigation of vegetation invasions was identified in section 2.2.2 of Chapter 2. There is a need for joint investigations between geomorphologists and ecologists in order to gain a full understanding of vegetation invasion dynamics.

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APPENDIX A

ERROR MATRICES FOR IMAGES 2 - 10

Error Matrix Analysis of for image 2: 1= Riparian vegetation; 2= Bare surfaces and gullies; 3= Established *P. incana*; 4= *P. incana* in badlands; 5= Grass.

```
                  1         2         3         4         5     Total    ErrorC
         ----------------------------------------------------------------------
     1 |     110904      4429       173     15695     23757 |  154958    0.2843
     2 |        433     49372      6228       767         0 |   56800    0.1308
     3 |        193      3370    394925      9034      9328 |  416850    0.0526
     4 |       4655         0      8432    570426      3392 |  586905    0.0281
     5 |         19         0         4        29    326723 |  326775    0.0002
          ----------------------------------------------------------------------
 Total |     116204     57171    409762    595951    363200 | 1542288
ErrorO |     0.0456    0.1364    0.0362    0.0428    0.1004 |            0.0583


         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0003    (0.0580 - 0.0586)
         95% Confidence Interval  =  +/-  0.0004    (0.0579 - 0.0587)
         99% Confidence Interval  =  +/-  0.0005    (0.0578 - 0.0588)


KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using ACCMCLASS4 as the reference image ...

    Category          KIA
------------     ------------

           1      0.6925
           2      0.8642
           3      0.9284
           4      0.9542
           5      0.9998


ACCMCLASS3

    Category          KIA
------------ ------------

           1      0.9493
           2      0.8584
           3      0.9504
           4      0.9309
           5      0.8726


Overall Kappa =         0.9193
```

Error Matrix Analysis of for image 3: 1= Riparian vegetation; 2= Bare surfaces and gullies; 3= Established *P. incana*; 4= *P. incana* in badlands; 5= Grass.

```
                  1         2         3         4         5      Total    ErrorC
         -----------------------------------------------------------------------
     1 |     221483       468     41864      4934      2312 |   271061    0.1829
     2 |        466     18135       575         0         0 |    19176    0.0543
     3 |       3659      2804    329195       364      6173 |   342195    0.0380
     4 |         90        15       179    560705         0 |   560989    0.0005
     5 |      10362         0     19241         0    319264 |   348867    0.0849
          -----------------------------------------------------------------------
 Total |     236060     21422    391054    566003    327749 |  1542288
ErrorO |     0.0618    0.1534    0.1582    0.0094    0.0259 |              0.0606


         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0003    (0.0603 - 0.0609)
         95% Confidence Interval  =  +/-  0.0004    (0.0603 - 0.0610)
         99% Confidence Interval  =  +/-  0.0005    (0.0601 - 0.0611)


KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using CELCLASS5 as the reference image ...

    Category          KIA
------------     ------------

           1      0.7840
           2      0.9449
           3      0.9491
           4      0.9992
           5      0.8922


CELCLASS4

    Category          KIA
------------ ------------

           1      0.9251
           2      0.8446
           3      0.7967
           4      0.9853
           5      0.9665


Overall Kappa =         0.9175
```

Error Matrix Analysis of for image 4: 1= Riparian vegetation; 2= Bare surfaces and gullies; 3= Established *P. incana*; 4= Grass.

```
                1         2         3         4     Total    ErrorC
         ------------------------------------------------------------
     1 |   247955         0         0     13218 |  261173    0.0506
     2 |     5047    265559         0     57851 |  328457    0.1915
     3 |    91652       851    387348         0 |  479851    0.1928
     4 |    53824         0         0    418983 |  472807    0.1138
          ------------------------------------------------------------
 Total |   398478    266410    387348    490052 | 1542288
ErrorO |   0.3777    0.0032    0.0000    0.1450 |            0.1442


         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0005    (0.1438 - 0.1447)
         95% Confidence Interval  =  +/-  0.0006    (0.1437 - 0.1448)
         99% Confidence Interval  =  +/-  0.0007    (0.1435 - 0.1450)


KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using AFMGCLASS FINAL4 as the reference image ...

    Category          KIA
------------     ------------

           1      0.9318
           2      0.7685
           3      0.7426
           4      0.8331


AFMGCLASS FINAL1

    Category          KIA
------------ ------------

           1      0.5452
           2      0.9959
           3      1.0000
           4      0.7909
```

Overall Kappa =   0.8061

Error Matrix Analysis of for image 5: 1= Riparian vegetation; 2= Bare surfaces and gullies; 3= Established *P. incana*; 4= Grass.

```
                  1         2         3         4     Total    ErrorC
         ------------------------------------------------------------
     1 |     510355         0         0         7 |  510362    0.0000
     2 |          0     14313        93        47 |   14453    0.0097
     3 |      15140        62    341831      3860 |  360893    0.0528
     4 |       3368      3224     28482    621506 |  656580    0.0534
          ------------------------------------------------------------
 Total |     528863     17599    370406    625420 | 1542288
ErrorO |     0.0350    0.1867    0.0771    0.0063 |             0.0352


         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0002    (0.0350 - 0.0354)
         95% Confidence Interval  =  +/-  0.0003    (0.0349 - 0.0355)
         99% Confidence Interval  =  +/-  0.0004    (0.0348 - 0.0356)


KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using GPCLASS5 as the reference image ...

    Category          KIA
------------     ------------

           1      1.0000
           2      0.9902
           3      0.9305
           4      0.9101


GPCLASS3

    Category          KIA
------------ ------------

           1      0.9477
           2      0.8115
           3      0.8993
           4      0.9891


Overall Kappa =         0.9465
```

Error Matrix Analysis of for image 6: 1= Riparian vegetation; 2= Bare surfaces and gullies; 3= Established *P. incana*; 4= Grass.

```
                  1         2         3         4     Total    ErrorC
         ------------------------------------------------------------
     1 |     140694       643     43535     12446 |  197318    0.2870
     2 |         25     64957         0      4382 |   69364    0.0635
     3 |          0         0   1014631       503 | 1015134    0.0005
     4 |        497      8317      1781    249877 |  260472    0.0407
          ------------------------------------------------------------
 Total |     141216     73917   1059947    267208 | 1542288
ErrorO |     0.0037    0.1212    0.0428    0.0649 |            0.0468


         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0003    (0.0465 - 0.0470)
         95% Confidence Interval  =  +/-  0.0003    (0.0464 - 0.0471)
         99% Confidence Interval  =  +/-  0.0004    (0.0463 - 0.0472)

KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using GP2CLASS FINAL2 as the reference image ...

    Category          KIA
------------     ------------

           1      0.6841
           2      0.9333
           3      0.9984
           4      0.9508


GP2CLASS FINAL1

    Category          KIA
------------ ------------

           1      0.9958
           2      0.8731
           3      0.8749
           4      0.9220


Overall Kappa =         0.9073
```

Error Matrix Analysis of for image 7: 1= Riparian vegetation; 2= Bare surfaces; 3= P. incana patches; 4= Established *P. incana*; 5= Grass.

```
                  1         2         3         4         5     Total    ErrorC
         ---------------------------------------------------------------------
     1 |     647870       198       230       669       686 |  649653    0.0027
     2 |          3     22781         0       332         0 |   23116    0.0145
     3 |       1322         2     11229      8333      3252 |   24138    0.5348
     4 |        512      2448       342    161874      3357 |  168533    0.0395
     5 |      13175     13142         5      5578    644948 |  676848    0.0471
          ---------------------------------------------------------------------
 Total |     662882     38571     11806    176786    652243 | 1542288
ErrorO |     0.0226    0.4094    0.0489    0.0844    0.0112 |             0.0347


         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0002    (0.0345 - 0.0350)
         95% Confidence Interval  =  +/-  0.0003    (0.0345 - 0.0350)
         99% Confidence Interval  =  +/-  0.0004    (0.0344 - 0.0351)

KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using JUNCLASS5 as the reference image ...

    Category          KIA
------------     ------------

           1      0.9952
           2      0.9851
           3      0.4611
           4      0.9554
           5      0.9183


JUNCLASS4

    Category          KIA
------------ ------------

           1      0.9609
           2      0.5844
           3      0.9503
           4      0.9053
           5      0.9801


Overall Kappa =         0.9440
```

Error Matrix Analysis of for image 8: 1= Riparian vegetation; 2= Bare surfaces; 3= P. incana patches; 4= Established *P. incana*; 5= Grass.

```
                  1         2         3         4         5     Total    ErrorC
         ----------------------------------------------------------------------
     1 |     735048        21         0      1092         2 |  736163    0.0015
     2 |          0     23201      1195       432      1531 |   26359    0.1198
     3 |        705      1225    504121      4845      9653 |  520549    0.0316
     4 |          0         0        25      5753         0 |    5778    0.0043
     5 |        290      1173     10791         0    241185 |  253439    0.0484
          ----------------------------------------------------------------------
 Total |     736043     25620    516132     12122    252371 | 1542288
ErrorO |     0.0014    0.0944    0.0233    0.5254    0.0443 |              0.0214


         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0002    (0.0212 - 0.0216)
         95% Confidence Interval  =  +/-  0.0002    (0.0212 - 0.0216)
         99% Confidence Interval  =  +/-  0.0003    (0.0211 - 0.0217)


KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using JUN2CLASS4 as the reference image ...

    Category          KIA
------------     ------------

           1      0.9971
           2      0.8782
           3      0.9526
           4      0.9956
           5      0.9422


JU2CLASS3

    Category          KIA
------------ ------------

           1      0.9974
           2      0.9039
           3      0.9649
           4      0.4726
           5      0.9470


Overall Kappa =         0.9662
```

Error Matrix Analysis of for image 9: 1= Riparian vegetation; 2= P. incana patches; 3= Established *P. incana*; 4= Grass.

```
                1         2         3         4     Total    ErrorC
         -----------------------------------------------------------
     1 |     234724         0      1591         0 |  236315    0.0067
     2 |       2659    471663     25133      2072 |  501527    0.0595
     3 |       4073      4744     79880         0 |   88697    0.0994
     4 |      12195    158348     16827    528379 |  715749    0.2618
          -----------------------------------------------------------
 Total |     253651    634755    123431    530451 | 1542288
ErrorO |     0.0746    0.2569    0.3528    0.0039 |            0.1476


         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0005    (0.1471 - 0.1481)
         95% Confidence Interval  =  +/-  0.0006    (0.1470 - 0.1482)
         99% Confidence Interval  =  +/-  0.0007    (0.1469 - 0.1483)


KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using JU3CLASS5 as the reference image ...

    Category          KIA
------------     ------------

           1      0.9919
           2      0.8988
           3      0.8919
           4      0.6010


JU3CLASS3

    Category          KIA
------------ ------------

           1      0.9119
           2      0.6192
           3      0.6256
           4      0.9927


Overall Kappa =         0.7819
```

Error Matrix Analysis of for image 10: 1= Riparian vegetation; 2= Bare surfaces and gullies; 3= Established *P. incana*; 4= Grass.

```
                  1         2         3         4     Total    ErrorC
         ------------------------------------------------------------
     1 |     385315        11       428      1069 |  386823    0.0039
     2 |          0    144659     20637     45932 |  211228    0.3152
     3 |        729      9875    147853      5270 |  163727    0.0970
     4 |          4     45077     13535    721894 |  780510    0.0751
          ------------------------------------------------------------
 Total |     386048    199622    182453    774165 | 1542288
ErrorO |     0.0019    0.2753    0.1896    0.0675 |           0.0924


         ErrorO     = Errors of Omission   (expressed as proportions)
         ErrorC     = Errors of Commission (expressed as proportions)


         90% Confidence Interval  =  +/-  0.0004    (0.0921 - 0.0928)
         95% Confidence Interval  =  +/-  0.0005    (0.0920 - 0.0929)
         99% Confidence Interval  =  +/-  0.0006    (0.0918 - 0.0930)


KAPPA INDEX OF AGREEMENT (KIA)
------------------------------


Using NYANCLASS3 as the reference image ...

    Category          KIA
------------     ------------

           1      0.9948
           2      0.6380
           3      0.8900
           4      0.8492


NYANCLASS

    Category          KIA
------------ ------------

           1      0.9975
           2      0.6810
           3      0.7878
           4      0.8633


Overall Kappa =         0.8584
```

APPENDIX B

FIELD DATA BASED ON BROAD CATCHMENT SURVEY

Longitude	Latitude	Slope no.	position	dist.from	slope	cover type	erosion	comment	land use	tenure area
				chann. (m)	angle (0)	% cover	status			
27.13427	-33.22532	1	lower	1	3	Ind. 80%	S1	no pteronia	grazing	communal
27.13852	-33.22341	1	mid	250	2	Ind. 80%	S1	no pteronia	grazing	communal
27.13624	-33.22122	1	upper	800	2	Ind. 80%	S1	no pteronia	grazing	communal
27.14325	-33.23215	2	lower	2	7	Ind. 80%	S1	no pteronia	grazing	communal
27.14121	-33.23115	2	mid	422	4	Ind. 80%	S1	no pteronia	grazing	communal
27.44335	-33.23032	2	upper	522	2	Ind. 80%	S1	no pteronia	grazing	communal
27.15422	-33.23721	3	lower	2	12	riparian	S1	no pteronia	grazing	communal
27.15061	-33.23012	3	mid	12	10	pt. 100%	S3	est.pt., crust	grazing	communal
27.15422	-33.23501	3	upper	57	8	pt.45%	S2	pt. Invading	grazing	communal
27.15562	-33.23841	4	lower	2	10	pt. 100%	S3	est.pt., crust	grazing	communal
27.15622	-33.23832	4	mid	32	9	pt. 100%	S3	est.pt., crust	grazing	communal
27.15566	-33.23911	5	lower	4	14	pt. 100%	G2	est.pt., crust	ab.cult	communal
27.15621	-33.23953	5	mid	36	8	pt. 100%	S3	est.pt., crust	ab.cult	communal
27.15615	-33.24052	5	upper	115	10	pt. 100%	S3	est.pt., crust	ab.cult	communal
27.16438	-33.23353	6	lower	3	13	pt.+ ind.	G2,S3	pt. Invading	grazing	communal
27.1628	-33.23304	6	mid	54	9	ind. 70%	S1	pt. Invading	grazing	communal
27.165056	-33.23253	6	upper	82	3	ind. 60%	S1	pt. Invading	grazing	communal
27.1713	-33.23342	7	lower	3	5	ind. 60%	S1	no pteronia	grazing	communal
27.1722	-33.23455	7	mid	47	4	ind. 60%	S1	no pteronia	grazing	communal
27.1734	-33.23436	7	upper	95	5	ind. 60%	S1	no pteronia	grazing	communal
27.16835	-33.24278	8	lower	4	11	pt. 100%	S4, G1	est.pt., crust	grazing	communal
27.17082	-33.24132	8	mid	56	8	pt. 100%	S4, G1	est.pt., crust	grazing	communal
27.16941	-33.24122	8	upper	101	4	pt. + ind	S2	pt. Invading	grazing	communal
27.17843	-33.25014	9	lower	6	4	Ind. 80%	S1	no pteronia	grazing	form. Comm.
27.17725	-33.24962	9	mid	58	5	Ind. 80%	S1	no pteronia	grazing	form. Comm.
27.17534	-33.24834	9	upper	159	2	Ind. 80%	S1	no pteronia	grazing	form. Comm.
27.19211	-33.25541	10	n/a	n/a	2	woodland	S1	no pteronia	grazing	form. Comm.
27.1944	-33.26842	11	n/a	n/a	2	woodland	S1	no pteronia	grazing	form. Comm.
27.21833	-33.28136	12	n/a	n/a	2	woodland	S1	no pteronia	grazing	form. Comm.
27.19377	-33.29837	13	n/a	n/a	2	woodland	S1	no pteronia	pinezone	form. Comm.
27.22157	-33.29541	14	n/a	n/a	4	woodland	S1	no pteronia	pinezone	form. Comm.

27.23326	-33.27505	15	n/a	n/a	5	woodland	S1	no pteronia	pinezone	form. Comm.
27.08453	-33.25721	16	lower	4	11	ind. 60%	S1	pt. Invading	grazing	communal
27.08432	-33.15248	16	mid	24	8	ind.+ pt	S2	pt. Invading	grazing	communal
27.08341	-33.15218	16	upper	69	3	ind. + pt.	S2	pt. Invading+crust	grazing	communal
27.08722	-33.26529	17	lower	3	6	ind. 60%	S1	no pteronia	grazing	communal
27.08841	-33.26525	17	mid	106	7	ind. 70%	S1	no pteronia	grazing	communal
27.08935	-33.26582	17	upper	151	7.5	pt. + ind	S5	pt. Invading	grazing	communal
27.08821	-33.27656	18	lower	3	7	ind. + pt.	S2	pt. Invading+crust	grazing	communal
27.08806	-33.2759	18	mid	44	7.5	ind. + pt.	S2	pt. Invading+crust	grazing	communal
27.08725	-33.27631	18	upper	75	5	ind. + pt.	S5	pt. Invading+crust	grazing	communal
27.0789	-33.28722	19	lower	5	9	th.bush+pt.	S5	pt. Invading+crust	grazing	communal
27.07311	-33.2873	19	mid	29	11	th.bush+pt.	S5	pt. Invading+crust	grazing	communal
27.07425	-33.28721	19	upper	56	10	ind. + pt.	S5	est.pt., crust	grazing	communal
27.08466	-33.29462	20	lower	5	9	ind. + pt.	S4	pt. Patches	grazing	communal
27.0843	-33.29451	20	mid	40	9	ind. 60%	S1	pt. Patches	grazing	communal
27.0805	-33.29701	20	upper	108	9.5	Ind. 80%	S1	no pteronia	fall+contour	communal
27.09651	-33.3024	21	n/a	n/a	7	ind. 90%	S1	no pteronia	fall+contour	communal
27.12138	-33.3173	22	lower	4	14	pt. 100%	G2,R4	est.pt., crust	grazing	communal
27.12104	-33.31725	22	mid	42	14.5	pt. 100%	G2,R4	est.pt., crust	grazing	communal
27.12156	-33.31752	22	upper	64	8	pt. 100%	G2,R4	est.pt., crust	grazing	communal
27 1313	-33.3195	23	lower	3	14	pt. 100%	S4	est.pt., crust	grazing	communal
27.13221	-33.31965	23	mid	76	10	pt. 100%	S4	est.pt., crust	grazing	communal
27.1314	-33.3221	23	upper	106	7	pt. 100%	S4	est.pt., crust	grazing	communal
27.14236	-33.32191	24	lower	5	14	pt. 100%	S3	est.pt., crust	grazing	communal
27.14328	-33.32156	24	mid	50	8	pt. 100%	R1, S4	est.pt., crust	grazing	communal
27.14436	-33.32225	24	upper	97	1	pt. + ind	S1	pt. Invading	grazing	communal
27.1526	-33.32561	25	lower	4	16.5	pt. 20%	G4, RB	pt.btwn gullies	grazing	communal
27.1525	-33.32589	25	mid	29	14	pt. 70%	S4, R2	est.pt., crust	grazing	communal
27.15237	-33.3257	25	upper	55	11	pt.+bare pat	S4, R2	est.pt., crust	grazing	communal
27.15922	-33.3309	26	lower	8	18	pt.+bare pat	G3	est.pt., crust	grazing	communal
27.15953	-33.3316	26	mid	46	15	pt.+bare pat	R5, S4	est.pt., crust	grazing	communal
27.15861	-33.33162	26	upper	78	10	ind. 60%	S1	pt. Invading	grazing	communal
27.1649	-33.33821	27	lower	6	22	pt.+bare pat	G3	est.pt., crust	grazing	communal

27.1649	-33.33868	27	mid	28	18	pt.+bare pat	S4	est.pt., crust	grazing	communal
27.1648	-33.33862	27	upper	64	10	pt. + ind	S1	pt. Invading	grazing	communal
27.17156	-33.34421	28	lower	5	9	ind. 60%	S1	isol. Pt.	grazing	form. Comm.
27.17186	-33.3448	28	mid	25	8	Ind. 80%	S1	no pteronia	grazing	form. Comm.
27.17199	-33.3449	28	upper	40	8	Ind. 80%	S1	no pteronia	grazing	form. Comm.
27.0962	-33.2718	29	lower	3	16	ind. 40%	S4	pt. Invading	grazing	communal
27.09596	-33.2716	29	mid	21	14	ind. 40%	S2	pt. Invading	grazing	communal
27.0956	-33.27155	29	upper	52	5	ind. 40%	S2	pt. Patches, crust	grazing	communal
27.1013	-33.2636	30	lower	3	6	pt. + ind	S2	pt. Invading	grazing	communal
27.10226	-33.2646	30	mid	32	5	pt. + ind	S1	pt. Invading	grazing	communal
27.10268	-33.2647	30	upper	70	4	ind. 60%	S1	isol. Pt.	grazing	communal
27.22758	-33.2701	31	lower	3	23	pt.+bare pat	S4, R3	est.pt., crust	grazing	communal
27.1166	-33.27093	31	mid	48	15	pt. 70%	S4	est.pt., crust	grazing	communal
27.11641	-33.2712	31	upper	74	8	pt. 70%	S4	est.pt., crust	grazing	communal
27.12831	-33.26861	32	lower	3	8	pt.+bare pat	G4, R5	est.pt., crust	ab.cult	communal
27.12696	-33.26871	32	mid	70	12	pt.+bare pat	R3, R4	est.pt., crust	ab.cult	communal
27.012635	-33.2693	32	upper	130	9	pt.+bare pat	R3, R4	est.pt., crust	ab.cult	communal
27.13668	-33.2719	33	lower	3	11	pt.+bare pat	S5	est.pt., crust	ab.cult	communal
27.13701	-33.2724	33	mid	40	9	pt.+bare pat	S4	est.pt., crust	ab.cult	communal
27.1375	-33.27275	33	upper	78	4	pt.+bare pat	S4	est.pt., crust	ab.cult	communal
27.1539	-33.2615	34	lower	5	13	pt.+bare pat	G4, RB	est.pt., crust	ab.cult	communal
27.15415	-33.26215	34	mid	52	10	pt.+bare pat	S4	est.pt., crust	ab.cult	communal
27.1546	-33.26275	34	upper	107	6	pt. 70%	S3	est.pt., crust	ab.cult	communal
27.16343	-33.2557	35	lower	2	10	pt. + ind	S3	pt. Invading	grazing	communal
27.16285	-33.2555	35	mid	42	8	pt. 70%	S3	est.pt., crust	grazing	communal
27.16223	-33.25511	35	upper	86	7	pt. + ind	S2	pt. Invading	grazing	communal
27.16803	-33.24481	36	lower	2	14	pt.+bare pat	2	est.pt., crust	grazing	communal
27.16795	-33.2439	36	mid	30	10	pt.strips+ind	S2	pt. Invading	grazing	communal
27.16813	-33.24335	36	upper	71	5	pt.strips+ind	S2	pt. Invading	grazing	communal
27.12915	-33.2633	37	lower	3	11	Ind. 80%	S1	no pteronia	cont.grazing	communal
27.1296	-33.2353	37	mid	18	8	Ind. 80%	S1	no pteronia	cont.grazing	communal
27.1298	-33.2348	37	upper	57	7	Ind. 80%	S1	no pteronia	cont.grazing	communal
27.1189	-33.24	38	lower	5	16	pt.+bare pat	R2, S4	est.pt., crust	grazing	communal

27.11928	-33.2397	38	mid	14	11	pt.+bare pat	R2, S4	est.pt., crust	grazing	communal
27.11945	-33.23948	38	upper	45	9	pt.+ ind.	S3	pt. Invading	grazing	communal
27.1111	-33.24865	39	lower	2	4	ind. 70%	S1	pt. Invading	grazing	communal
27.11291	-33.24958	39	mid	43	4	ind. 70%	S1	pt. Invading	grazing	communal
27.11308	-33.2497	39	upper	130	4.5	ind. 70%	S1	pt. Invading	grazing	communal
27.11055	-33.2558	40	lower	5	7	pt.+bare pat	R2, S4	est.pt., crust	ab.cult	communal
27.10975	-33.25585	40	mid	41	5	pt.+bare pat	R2, S4	est.pt., crust	ab.cult	communal
27.1092	-33.25571	40	upper	73	4	pt.+bare pat	R2, S4	est.pt., crust	ab.cult	communal
27.18926	-33.30275	41	lower	3	18	pt.+wood	S4	deg. Woodland	grazing	communal
27.18928	-33.30285	41	mid	18	15	pt.+wood	S4	deg. Woodland	grazing	communal
27.18943	-33.30321	41	upper	53	14	pt.+wood	S4	deg. Woodland	grazing	communal
27.19418	-33.31248	42	lower	5	12	pt.+bare pat	S3, G2	deg. Woodland	grazing	communal
27.19408	-33.31263	42	mid	24	13	pt.+wood	R2, S4	deg. Woodland	grazing	communal
27.1939	-33.3129	42	upper	46	14	pt.+wood	G3	deg. Woodland	grazing	communal
27.19113	-33.31757	43	lower	2	17	pt.+bare pat	R2, S4	est.pt., crust	grazing	communal
27.19158	-33.31761	43	mid	27	16	pt.+bare pat	S5	est.pt., crust	grazing	communal
27.19198	-33.31768	43	upper	51	12	pt.+bare pat	R2, S4	est.pt., crust	grazing	communal
27.12335	-33.24441	44	lower	8	14	badlands	G4, R5	est. pt, s.erosion	ab.cult	communal
27.12048	-33.24264	44	mid	36	14	badlands	G4, R5	est. pt, s.erosion	ab.cult	communal
27.121353	-33.23991	44	upper	72	12	badlands	G4, R5	est. pt, s.erosion	ab.cult	communal
27.11315	-33.24246	45	lower	7	4	Ind. Grass,90%	S1	no pteronia	grazing	communal
27.13461	-33.27121	46	upper	52	11	pt. +bare pat	S5, G2	est. pt, crust	ab.cult	communal
27.137983	-33.27382	45	mid	32	8	Ind. Grass,90%	S1	no pteronia	grazing	communal
27.100239	-33.26445	47	lower	7	7	pt. +bare pat	S3, S4	est. pt., crust	grazing	communal
27.09741	-33.26261	47	mid	40	6	pt. +bare pat	S3, S4	est. pt., crust	grazing	communal
27.10161	-33.26659	47	upper	80	5	pt. +bare pat	S3, S4	est. pt., crust	grazing	communal
27.11669	-33.27192	48	lower	8	10		R2, S4	est. pt., crust	ab.cult	communal
27.11784	-33.26873	48	mid	28	11	pt. +bare pat	G2, S4	est. pt., crust	ab.cult	communal
27.12285	-33.26889	48	upper	78	13	pt. +bare pat	R3, R4	est. pt., crust	ab.cult	communal
27.1272	-33.27048	49	lower	9	15	pt. +bare pat	G4, G5	est. pt, s.erosion	ab.cult	communal
27.12724	-33.26885	49	mid	28	10	pt. +bare pat	R3, R4	est. pt., crust	ab.cult	communal
27.13002	-33.27088	50	lower	6	12	pt. +bare pat	R4, S5	est. pt., crust	ab.cult	communal
27.13175	-33.26993	51	mid	42	9	pt. +bare pat	R3, R4	est. pt., crust	ab.cult	communal

27.13044	-33.26706	51	upper	82	16	badlands	G5	est. pt, s.erosion	ab.cult	communal
27.13279	-33.26443	51	lower	10	15	badlands	G5	est. pt, s.erosion	ab.cult	communal
27.12698	-33.26861	52	lower	6	11	pt. +bare pat	R5, S4	est. pt., crust	ab.cult	communal
27.13015	-33.26429	52	mid	25	16	pt. +bare pat	G5	est. pt, s.erosion	ab.cult	communal
27.13253	-33.26137	52	upper	72	10	pt. +bare pat	R5, S4	est. pt, crust	ab.cult	communal
27.13543	-33.25887	53	lower	8	12	pt. +bare pat	R5, S4	est. pt, crust	ab.cult	communal
27.12123	-33.26083	54	lower	5	8	pt. + ind	S2, S3	pt. Invading, crust	grazing	communal
27.12512	-33.25834	54	mid	35	5	Ind. Grass,90%	S1	no pteronia	grazing	communal
27.12209	-33.25797	54	upper	70	6	Ind. Grass,90%	S1	no pteronia	grazing	communal
27.10832	-33.25775	55	lower	4	4	Ind. Grass,90%	S1	no pteronia	grazing	communal
27.11213	-33.25713	55	mid	30	5	Ind. Grass,90%	S1	no pteronia	grazing	communal
27.11493	-33.25795	56	lower	7	11	pt. +bare pat	R4, S5	est. pt, s.erosion	ab.cult	communal
27.11575	-33.25987	56	mid	26	11	pt. +bare pat	R5, S4	est. pt, crust	ab.cult	communal
27.11498	-33.26334	56	upper	53	9	pt. +bare pat	R5, S4	est. pt, crust	ab.cult	communal
27.11229	-33.25496	57	lower	5	12	pt. +bare pat	R 4	est. pt, crust	ab.cult	communal
27.12102	-33.26373	58	upper	75	14	pt. +bare pat	R4, S5	est. pt, crust	ab.cult	communal
27.11877	-33.26148	58	mid	36	13	pt. +bare pat	R 3	est. pt., crust	ab.cult	communal
27.11084	-33.26077	58	lower	8	8	pt. +bare pat	R2, S4	est. pt., crust	ab.cult	communal
27.11066	-33.26194	59	lower	6	9	pt. +bare pat	R2, S4	est. pt., crust	ab.cult	communal

pt. +bare pat - *P. incana and bare patches*
est. pt, crust - *Established P. incana and crusted surfaces*
pt. + ind - *P. incana and indigenous grass species*
ab.cult - *Abandoned cultivation*
pt.+wood - *P. incana in woodland*
pt. Invading - *P. incana invading*
Ind 80% - *Indigenous grass species cover 80%*