The potential costs and benefits of addressing land degradation in the Thukela catchment, KwaZulu-Natal South Africa

Report of the NCAVES Project

United Nations | System of Environmental Economic Accounting

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PREFACE AND ACKNOWLEDGEMENTS

This project was commissioned by the United Nations Environment Programme (UNEP) as part of the South African component of the international, European Union (EU)-funded, Natural Capital Accounting and Valuation of Ecosystem Services (NCAVES) project. The NCAVES project is being carried out as a collaboration between UN Environment, United Nations Statistics Division (UNSD), the South African National Biodiversity Institute (SANBI) and Statistics South Africa (Stats SA). UNEP commissioned Dr Jane Turpie to assist with the compilation of pilot monetary ecosystem accounts at regional scale, as one of five test countries, which concluded in June 2020. Dr Turpie's company, Anchor Environmental Consultants, was also involved in the compilation of the land and terrestrial ecosystem accounts at national scale, under contract to SANBI. All of this related work has been a collaborative effort involving a number of players.

The purpose of the study was to demonstrate that natural capital accounts can be useful in informing scenario analysis for policy and decision-making. The scope of work for this scenario-based study was devised on the basis of inputs from a range of government and non-government stakeholders. The key environmental issues of KwaZulu-Natal were discussed with stakeholders at an NCA Forum meeting in Pretoria in 2018. The participants of the NCAVES stakeholder workshop held in Pretoria in March 2018 were asked to list what they considered to be the top three environmental issues in KwaZulu-Natal. From this exercise, it was clear that urbanisation, water pollution and ecosystem degradation were the issues of most interest. In ensuing discussions with the Department of Forestry, Fisheries and the Environment (DFFE, Dr Christo Marais, Chief Director of Natural Resource Management Programmes at the DFFE, and Ryan Brudvig, Implementation Head of the Working for Water programme in KwaZulu-Natal), the provincial nature conservation authority Ezemvelo KZN Wildlife (EKZNW) and SANBI, it was agreed that the study should focus on land degradation.

The work was carried out using spatial land cover data supplied by EKZNW. We are grateful to William Speller of UNEP, Julian Chow and Bram Edens of UNSD for their direction and inputs. The study benefitted from the inputs of Amanda Driver and Aimee Ginsburg of SANBI, Gerhardt Bouwer and Robert Parry of Stats SA, and Jeanne Nel of the University of Wageningen, Netherlands.

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SUMMARY FOR POLICYMAKERS

Scope of the study

- This study forms part of the Natural Capital Accounting and Valuation of Ecosystem Services (NCAVES) project and builds on the compilation of pilot physical and monetary ecosystem services accounts for KwaZulu-Natal for 2005-2011. The analysis demonstrates the utility of ecosystem accounts – consistent with the System of Environmental Economic Accounting – Ecosystem Accounting (SEEA EA) framework – to inform policymaking.
- This study investigates the economic case for ecosystem management and restoration interventions in the Thukela river basin over the period 2021-2030 to achieve or exceed land degradation neutrality relative to a 2015 baseline. This study weighs the benefits of ecosystem restoration – in terms of the monetary value of improved provision of selected ecosystem services – against the costs of interventions.
- The catchment area is 2.91 million ha and occupies about a third of the province of KwaZulu-Natal.
- It was estimated that almost 555 000 ha or 26% of the remaining natural area of the Thukela catchment (which is mostly grassland and savanna) was degraded in 2015, the base year for South Africa's land degradation neutrality targets.
- The catchment provides a range of ecosystem services which contribute to benefits used by the economy and to the provision of sustainable livelihoods. The study includes the following ecosystem services: **water retention (regulation of water supply); sediment retention (erosion control); carbon sequestration; provisioning of livestock products; provisioning of wood products; provisioning of non-wood products; and nature-based tourism.**
- Various human activities and decisions have had an adverse effect on the region's ecosystem services over time: overharvesting of resources; overgrazing; bush encroachment; the spread of invasive alien plants; and the loss of natural habitat due to expanding cultivation, human settlements and other activities such as mining. The grassland and savanna biomes, which dominate the landscape, were particularly badly affected.
- The study compares estimates how these services change by 2030 under a business-as usual scenario with low intervention levels and continued degradation; and then compares this to ecosystem service delivery under two possible intervention scenarios - (a) a land degradation neutrality scenario, and (b) a full restoration scenario - taking the costs of the interventions into account.

Key messages

- **Halting and reversing ecosystem degradation has positive net economic benefits.** The degradation of natural landscapes and ecological infrastructure has significant social, economic and environmental costs. Measures to slow, halt and reverse the net impacts of poor land management are in most estimations expected to have positive net benefits: addressing land degradation can be justified in economic terms.
- **Preventing degradation now is more cost effective than fixing it later.** Achieving land degradation neutrality involves a combination of avoiding degradation, reducing the rate of

further degradation of land (ideally to negligible levels) through sustainable land management, and offsetting the unavoidable degradation through restoration of an equivalent amount of land. The estimated costs of achieving sustainable land management were lower than those of rehabilitation or restoration. As such, preventing future land degradation is generally far more cost-effective in the long run than aiming to reduce or reverse past degradation.

- **The sooner restoration begins, the better.** Although previous investments in maintenance, rehabilitation and restoration efforts have not reversed or even stopped ongoing degradation, they have slowed the rate of degradation. Further investment to slow the rate of degradation now entails less need for costly future restoration.

- **While ecosystem restoration can be expensive, requiring significant and sustained investments in order to be effective, the potential benefits are much higher than the costs.** The benefits are potentially much higher when the level of restoration ambition is increased above the international commitments. That is, full restoration of ecosystems yields higher benefits (net of restoration costs) than interventions that are merely consistent with meeting South Africa's land degradation neutrality targets.

- **Different restoration interventions have different impacts, some yielding more benefit than others.** The results suggest that the protection and restoration of grassland areas could yield significantly more benefit than the benefits gained through addressing bush encroachment.

- **Including other economic benefits in the analysis, such as employment, would further strengthen the case for restoration and sustainable land management.** The study focuses on tangible ecosystem services and does not factor in other economic benefits associated with restoration, such as job creation. If these are included, the net benefits of halting land degradation and achieving restoration would be even greater.

- **For each Rand invested in full restoration there is a return of at least 1.7 Rand.** This is an underestimate. It includes only selected ecosystem services, and does not include the local multiplier effects of job creation, nor the intangible aesthetic and cultural values associated with restoring landscapes. The estimate of carbon benefits is very conservative; it does not include the global public good benefits of mitigating carbon emissions -- only local benefits.

- **Investment in restoration needs to be maintained through sustainable land management.** Restored areas need to be maintained; otherwise future costs will be incurred to restore again once land becomes degraded.

- In comparison of business-as-usual (BAU) to the full restoration option, key findings per ecosystem service include:

    - **Regulation of water supply** is significantly improved by the clearing of invasive woody plant species such as *Eucalyptus sp.* Natural habitats regulate the overall amount of streamflow and its variation throughout the year. In Thukela a major concern is reduction in water yield from the spread of these invasive species. The avoided losses of yield that accompany restoration were costed using the municipal sales price of water. The full restoration scenario is estimated to lead to avoided losses of R709 million in 2030.

    - **Erosion and sedimentation within watersheds** can impose costs as it causes structural damage to dams, flooding, affects the quality of drinking water, and increases water treatment and maintenance costs at water treatment work. Natural ecosystems help to prevent this by controlling soil erosion and retaining sedimentation. The annual value of erosion control provided by ecosystems was 2% higher under the full restoration scenario.

- Restoration is associated with large increases in natural grassland vegetation with high below-ground storage potential and high soil organic content which could **increase carbon storage** from an estimated 357 Tg C total ecosystem carbon to 363 Tg C.

- **Livestock farming** is an important livelihood activity, both on private and communal land, across large areas of the Thukela catchment. The results demonstrate how even a small loss in grazing land can have a significant impact on livestock production. On commercial land, grazing capacity was 2.2% higher under restoration compared to BAU, resulting in a gain in the value livestock provisioning service of R92 million per year in the catchment.

- Increased indigenous woody biomass ("**bush encroachment**") has been recognised in South African policy as a form of land degradation. Bush encroachment not only has a significant impact on wildlife numbers and densities but also impacts on visitor numbers to protected areas and game reserves, and their overall satisfaction. If efforts are made to restore the catchment over the next decade then there is great potential to expand the wildlife sector in the Thukela catchment. The value of nature-based tourism was expected to increase by 20% to about R95 million in 2030 under the restoration scenario. The converse is also true: degradation constrains growth to the **wildlife economy** and the ability to achieve goals set out in South Africa's National Biodiversity Economy Strategy.

- **This application shows the usefulness of ecosystem accounting for policymaking**. It made use of SEEA Ecosystem Accounting statistics to account for land cover and ecosystem service flows in a comprehensive and consistent manner. The results contribute to policy work on land degradation neutrality and support the potential for the massive economic contribution that ecosystem restoration can make to livelihoods and biodiversity. The next proposed steps would be 'ground-truthing'; while the remote sensing data used in this study is useful, physical measurements on-the-ground are required for verification and calibration.

ABBREVIATIONS AND ACRONYMS

BAU	Business-as-Usual
BLR	Binary Logistic Regression
BSU	Basic Spatial Unit
BTR	Boosted Tree Regression
CA	Cellular Automata
CARA	Conservation of Agricultural Resources Act (No 43 of 1983)
con ha	Condensed Hectare/s
DAFF	Department of Agriculture, Forestry and Fisheries
DFFE	Department of Forestry, Fisheries and the Environment
DALRRD	Department of Agriculture, Land Reform & Rural Development
EEA	Experimental Ecosystem Accounting
EI4WS	Ecological Infrastructure for Water Security
EKZNW	Ezemvelo KwaZulu-Natal Wildlife
EPCPD	Environmental Planning and Climate Protection Department
FAO	Food and Agriculture Organization (of the United Nations)
FEGS-CS	Final Ecosystem Goods and Services Classification System
GDP	Gross Domestic Product
HRU	Hydrologic Response Unit
IAP	Invasive Alien Plant
IOCB	Indian Ocean Coastal Belt
IPCC	Intergovernmental Panel on Climate Change
KZN	KwaZulu-Natal
LCM	[TerrSet] Land Cover Modeler
LDN	Land Degradation Neutrality
LSU	Livestock Unit
LULC	Land Use / Land Cover
MCA	Markov Chain Analysis
NAP	National Action Plan
NDVI	Normalised Difference Vegetation Index
MUSLE	Modified Universal Soil Loss Equation
NCAVES	Natural Capital Accounting and Valuation of Ecosystem Services
NIAPS	National Invasive Alien Plant Survey
NLC	National Land Cover
NPP	Net primary production
NPV	Net Present Value
NRM	Natural Resource Management (Directorate of DFFE)
OOAO	One Out, All Out
RLE	Red List of Ecosystems
SA	[Republic of] South Africa
SA	South Africa
SANBI	South African National Biodiversity Institute
SDG	Sustainable Development Goal
SEEA	System of Environmental-Economic Accounting
SLM	Sustainable Land Management
SNA	System of National Accounting
SOC	Soil Organic Carbon
Stats SA	Statistics South Africa

SWAT	Soil and Water Assessment Tool
SWSA	Strategic Water Source Area
TerrSet	TerrSet Geospatial Monitoring and Modelling suite
TIFF	Tagged Image File Format (.tiff file)
UN	United Nations
UNCCD	United Nations Convention to Combat Desertification
UNEP	United Nations Environment Programme
UNSD	United Nations Statistics Division
WMA	Water Management Area
ZNLD	Zero Net Land Degradation

EXECUTIVE SUMMARY

Introduction

This study forms part of the UN's Natural Capital Accounting and Valuation of Ecosystem Services (NCAVES) project, and builds on the compilation of pilot monetary ecosystem accounts for the province of KwaZulu-Natal, South Africa, for 2005-2011. That study showed that the supply and value of many ecosystem services had been negatively affected by changes in ecosystem extent and condition. This study aimed to use a scenario-based approach to explore an issue relating to existing targets, programmes and interventions and is of particular relevance to government stakeholders. Following consultation and exploration of spatial data, the focus of the study was narrowed down to that of land degradation, specifically in relation to South Africa's land degradation neutrality (LDN) commitments. In KwaZulu-Natal, land degradation mainly takes the form of (a) loss of biomass cover leading to bare areas and erosion, (b) increased indigenous woody biomass ("bush encroachment") and (c) encroachment of invasive alien plants (IAPs). The aim of the study was to estimate the costs and benefits of achieving or surpassing land degradation neutrality (LDN) in 2030 relative to 2015, based on a scenario analysis which takes some of the uncertainties into account. Secondary aims were to provide insights into conceptualising and planning for LDN, and to demonstrate the potential usefulness of natural capital accounting in this regard.

Study Area

This study focuses on the catchment area of the Thukela River system (Figure I), one of the least modified catchment areas (in terms of conversion to agriculture or urban land uses) in KwaZulu-Natal, but one in which land degradation is extensive. The Thukela River Catchment is 2.91 million ha in extent, and occupies about a third of the province. It is largely under grassland and savanna vegetation, with much of this natural land being under communal tenure. The catchment also has several large water supply dams in its higher lying areas.

Government is addressing land degradation through IAP clearing, erosion rehabilitation and farming and livestock management interventions, mainly in the upper and middle reaches. However, these efforts have not been enough to have any measurable effect.

Land degradation policy and action in South Africa

The first coordinated response to land degradation in South Africa dates as far back as 1861. Through much of the 20th century, extension services were provided to advise commercial farmers.[1] In the 1940s to 1980s, "Betterment Schemes" attempted to improve rangeland management practices in communal areas, but were heavily resisted, and subsequent extension services were also unsuccessful. In 1983, the Conservation of Agricultural Resources Act was promulgated to promote

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[1] Extension services, in the context of agriculture, involve entities - often large agricultural bodies or public institutions, but usually involving someone resident in the area - providing up-to-date information, knowledge, advice and technical support to through farmer education. Extension services are often seen as an intermediary between evolving agricultural science and farmers on the ground.

better agricultural practices. The subsequent LandCare programme, which focuses on sustainable land management (SLM) as well as restoration and rehabilitation of degraded areas in communal land areas, has reportedly also had a low level of success. Recent reintroduction of extension services in commercial and communal areas has also reportedly not been well implemented.

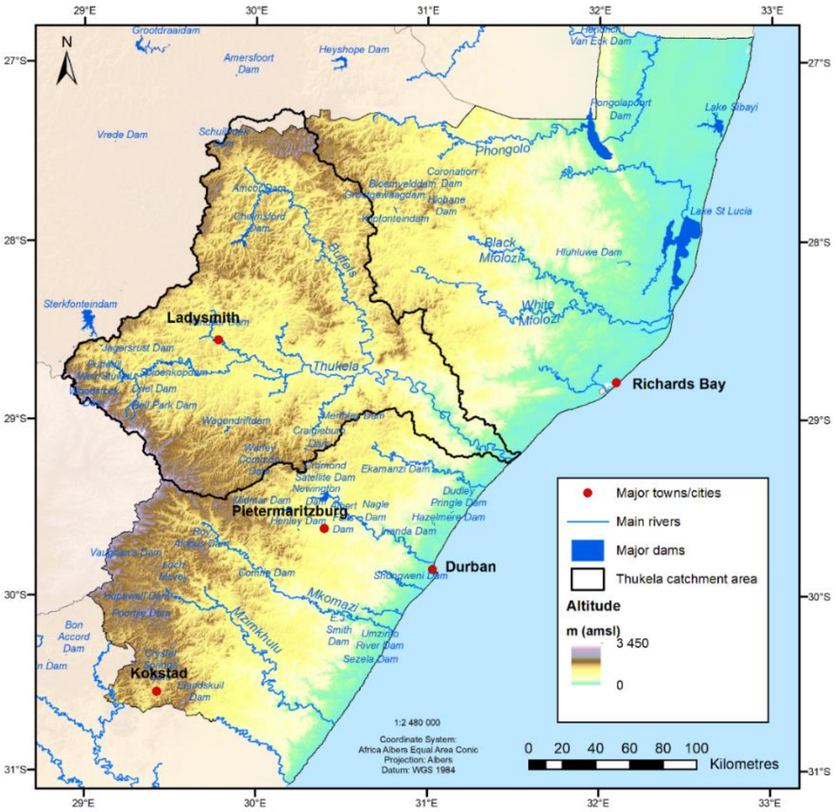

Figure I. Location of the Thukela catchment in KwaZulu-Natal showing topography, drainage and reservoirs.

Since 1995, the Department of Forestry, Fisheries and the Environment's (DFFE) National Resource Management (NRM) programmes have been one of the key contributors of the South African government's land degradation interventions. These form part of the stable of "expanded public works programmes" which are designed to achieve both social and environmental objectives, with employment being a key objective. Since the inception of "Working for Water", this has grown to include a number of other programmes, with an annual budget of almost R2 billion in 2017. Nevertheless, a much greater financial investment will be needed to meet the country's environmental targets.

Working for Water is still the largest of the NRM programmes, and focuses on clearing IAPs. About 200 000 ha of IAPs are cleared annually, in addition to periodic follow-up clearing. While over R1.5 billion is spent annually on dealing with invasive alien organisms (mostly plants), capacity constraints have meant that many of the areas have only been partially treated, which is ineffective. Outside of protected areas and some areas cleared by private landowners, IAPs continue to spread. The disappointing environmental outcomes can be attributed to the myopic focus from investors and planners which have subsequently relegated to the importance of environmental objectives as secondary to social objectives. However, South Africa finds itself in a window of opportunity to prioritise the development of formal policies and approaches to address IAPs as well as other forms of land degradation through their obligations under various international agreements.

Following the ratification of the United Nations' Convention on Biodiversity (UNCBD), Convention to Combat Desertification (UNCDD) and Framework Convention on Climate Change (UNFCCC), and the development of a new constitution in the 1990s, South Africa developed updated policies that increased monitoring and protection of the environment. A national survey of land degradation was undertaken in 1999, which has informed policy since. South Africa has a Land Degradation National Action Plan (NAP; 2014, and under revision) which includes the restoration of degraded ecosystems to contribute to ecosystem services delivery, climate change adaptation and mitigation. Since 2018, bush encroachment, has also been recognised in South African policy as a form of land degradation.

In 2015, South Africa adopted the UN's 17 Sustainable Development Goals (SDGs). This includes the obligation of achieving "land degradation neutrality" (LDN) whereby "the amount and quality of land resources necessary to support ecosystem functions and services and enhance food security remain stable or increase within specified temporal and spatial scales and ecosystems". UNCCD signatory countries, including South Africa, have established voluntary targets to achieve LDN by 2030, with respect to a 2015 baseline. South Africa's LDN targets are given as areas (in ha) of croplands, wetlands and eight biomes, that will be rehabilitated and sustainably managed by 2030, as well as area targets for clearing IAPs (1 063 897 ha) and bush encroachment (633 702 ha). Measures suggested to achieve these targets include SLM practices, erosion control, clearing alien species and bush clearing.

In order to extend the work being carried out under the DFFE programmes, there are various potential mechanisms for incentivising ecosystem restoration, including stewardship programmes, payment for ecosystem services, biodiversity offsets, certification, subsidisation of favourable land management activities and taxation of damaging activities. Opportunities for financing are presented by South Africa's Green Fund, Drylands Fund and Jobs Fund, as well as initiatives under the Biodiversity Economy programme.

Conceptual and analytical framework

The UN defines land degradation in terms of reduction in productivity of cultivated lands and rangelands. Achieving LDN involves a combination of avoiding degradation, reducing the rate of further degradation of land (ideally to negligible levels) and offsetting the unavoidable degradation through restoration of an equivalent amount of land. Designing measures to achieve LDN will require that countries have a good understanding of existing degradation (the baseline), of the drivers, process and rate of degradation, and the costs and effectiveness of alternative measures. The notion of "LDN targets" tends to focus the discussion on restoration, but countries should actually be outlining an LDN *strategy* that prioritises prevention as far as practicable. Ideally, the combined SLM-plus-restoration strategy would minimize the cost of achieving no net loss over a fixed period of time. Assumptions about the effectiveness of SLM investments will therefore be crucial in determining the appropriate area to restore in order to offset the residual expected degradation. It is also important to note that as countries delay the planning and implementation of LDN, so they will need to increase the restoration requirement, which will likely raise the costs (Figure II).

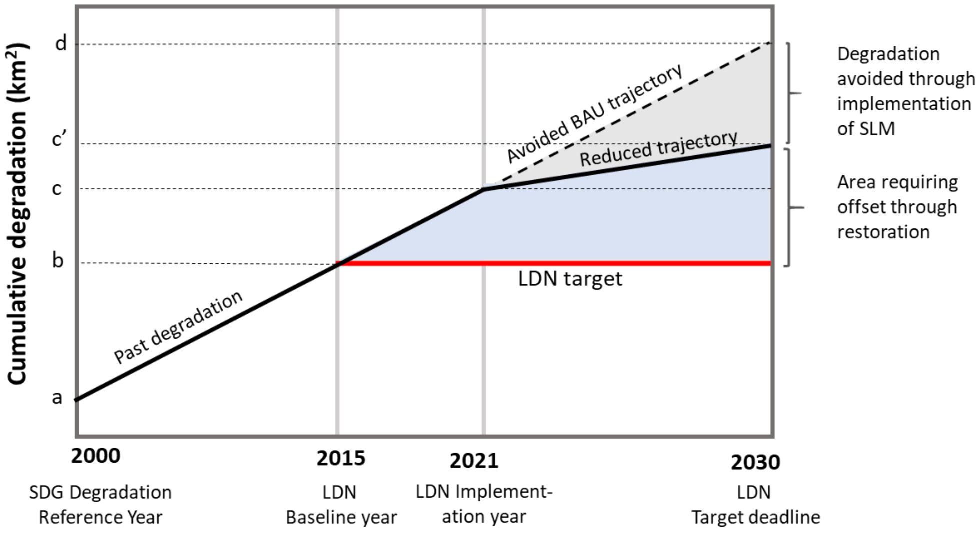

Figure II. *Conceptual diagram showing the consequences of delaying taking action towards achieving LDN. Increasing delay leads to a higher restoration requirement, which usually raises costs.*

The UN has developed guidelines for the measurement of degradation based on positive, stable or negative change in land cover, land productivity dynamics or soil organic carbon (SOC) relative to a base year of 2000, using the "one out, all out" principle. This approach, for which tools to make use of satellite data, such as trends.earth, have been developed, does not adequately describe rangeland degradation in the study context because both bush encroachment and IAPs *increase* vegetative cover and productivity. Therefore, in this study we used a more manual, rule-based approach to estimate degradation in the study area.

South Africa is committed to LDN irrespective of the cost. This should be seen as a minimum goal. Indeed, the country has been addressing land degradation for decades, but the goalposts appear to have moved from ambitious national targets to keeping problems under control in priority areas, as a result of financial realities. A clear understanding of the costs and benefits of reversing and preventing land degradation is needed to inform South Africa's targets (possibly exceeding the UN's LDN targets) and financing strategy.

Research objectives

This study explored the potential costs and benefits of achieving LDN in the Thukela catchment (i.e. restoring and maintaining levels of degradation at 2015 levels by 2030), and also compared this with the potential costs and benefits of a much more ambitious hypothetical target of close to full restoration of remaining natural areas, followed by SLM thereafter. To do this, we determined the level of degradation as at 2015, and estimated the expected level of degradation by 2030 under a **Business-as-Usual (BAU) Scenario**. This scenario had low levels of intervention and continued land degradation though spread of IAPs, bush encroachment and loss of vegetative cover and erosion. Next, we estimated what it would cost to achieve LDN by 2030. Assuming a start of action in 2021, this required restoration of the degradation that had already occurred from 2015-2021, followed by the implementation of SLM measures. For this **LDN Scenario**, we used a lower and upper bound estimate of costs, based on assumptions about the efficiency of SLM in halting future degradation. Under the lower bound (optimistic) cost estimate, it was estimated that SLM measures would be highly effective. Under the upper bound (pessimistic) cost estimate, it was assumed that SLM would be ineffective, necessitating restoration of an area equivalent to all projected degradation from 2015-2030. The reality was expected to lie between these two estimates. Finally, we estimated the costs and benefits of a **Full Restoration Scenario** in which interventions are implemented from 2021-2030 that restore all degraded natural areas (as at 2021) to a healthy condition. This assumed that SLM measures would stem further degradation.

Methods

This study was based on the latest (2017) output of the KZN land cover (LC) data series since this was closest to the 2015 baseline year for South Africa's LDN targets. The KZN land cover data series has information on condition relating to the loss of vegetative cover (albeit with limitations), but does not include information on IAPs or bush encroachment. IAPs cannot be distinguished from indigenous woody vegetation in satellite data, making it difficult to estimate the extent of IAPs or bush encroachment. One can only estimate the change in extent of woody vegetation cover. We estimated the extent of IAPs from national dataset for 2010, adjusted to 2017 levels based on spread rates from the literature. We also estimated the extent of *all* woody vegetation expansion since 2005 based on a comparison with the earliest available KZN LC data. The estimated IAP extent was subtracted from the latter area to estimate the area of indigenous bush encroachment. This was rendered spatially to create the 2017 baseline using a rule-based modelling approach. More detailed LC classes were created in the process.

To create the LC for the BAU Scenario, we projected IAP extent to 2030, projected change in woody vegetation based on the rates observed for 2005-2017, and intersected the two to distinguish the extent of IAPs and bush encroachment, as well as loss of vegetative cover and erosion as at 2030. The LC for the LDN scenario as at 2030 was the same as the 2017 baseline. To create the LC for the Full Restoration Scenario, all degraded areas (classified as IAPs, recent bush encroachment, reduced biomass cover or eroded) were reverted to their estimated former LC class. Urban and cultivation extent were held constant under all the scenarios.

We then estimated the impacts of the scenarios on hydrology and ecosystem services, drawing on approach and estimates of ecosystem service values developed for the KZN ecosystem accounts pilot (Turpie et al 2021). Focusing on natural land cover types for this study, these included spatially-explicit estimates of the exchange value of provisioning services (ecosystem inputs to livestock production, harvested wild plant and animal resources net of human inputs), cultural services (nature's contribution to the income derived from tourism expenditure), and regulating services (the costs saved as a result of hydrological and climate regulation, and the contribution of natural areas to agricultural production by supporting pollinators). The analysis took into account the effects of ecosystem condition on capacity to supply ecosystem services, as well as expected changes in the demand for services as a result of population and income growth.

In estimating the cost of interventions, it was assumed that most efficient pattern of clearing IAPs and dense bush encroachment would be to take a phased approach involving a relatively constant cash flow and level of employment. In order to prevent further degradation as a result of overgrazing (e.g. loss of grassland productivity, erosion or bush encroachment), as well as to maintain the condition of areas where bush encroachment had been cleared, it was assumed that SLM could be achieved with a combination of improved extension services and non-monetary and monetary incentive measures including stewardship programmes and payments for ecosystem services (PES).

Results

It was estimated that almost 555 000 ha or 26% of the remaining natural area of the Thukela catchment was degraded in 2015. Areas impacted by IAPs accounted for at least 3.5% of the catchment area, and roughly 9.5% of the catchment had undergone a change to a less woody or more woody LC class. Erosion accounted for about 2% of the catchment area.

Under a BAU Scenario, it was estimated that by 2030, invaded areas would increase to 4.2% of the catchment, there would be further encroachment of indigenous woody vegetation in 3.5% of the catchment area, and there would also be an increase in eroded areas. The area of healthy grassland would decline from 42% in 2017 to 36.3% of the catchment area, and the area of healthy woody vegetation would decline by 25% to 12.2% of the catchment area. Under the hypothetical full restoration scenario, healthy grassland would be restored to its former area of over 56% of the catchment, and healthy woody vegetation cover would be restored to 18.6% of the area.

The estimated changes in the biophysical supply and total monetary value of ecosystem service benefits are presented in Table I below. Degradation is worst under BAU, maintained at 2015 levels under LDN, and largely eliminated under Full Restoration. Accordingly, most ecosystem services were expected to improve with increasing restoration. However, wood products decrease because of the removal of encroaching alien and indigenous woody vegetation. There are major gains in carbon in restored grassland areas, but these are cancelled out to some extent by decreases in carbon due to woody vegetation removal. Because of this, carbon retention is decreased under the LDN scenario, but is increased under the full restoration scenario, which includes the restoration of grasslands that were degraded prior to 2015.

Table I. The total biophysical supply and total value (R millions in 2030) of ecosystem services per scenario.

	BAU	LDN Scenario	Full Restoration Scenario
Biophysical supply			
Yield increase (Mm^3 relative to BAU)		16	64
Sediment retained (t/ha/y)	10	10	10
Ecosystem carbon (Tg C)	357	354	363
Livestock production (LSU/y)	496 590	534 161	571 425
Wood products (m^3)	410 932	370 057	352 165
Non-wood products (t)	22 136	24 232	28 477
Nature-based tourism value (R million)	85	95	104
Value (R millions in 2030)			
Water supply (relative to BAU)		171	709
Erosion control	287	289	291
Carbon storage (global)	261 317	259 093	266 006
Carbon storage (national)	2 064	2 047	2 101
Livestock production	826	865	918
Wood products	689	616	584
Non-wood products	22	23	22
Nature-based tourism value	85	95	104

To achieve LDN, the minimum total investment in IAP clearing would be R514 million (present value of expenditure over 25 years), while to achieve full restoration this would increase to R2 355 million.

Addressing bush encroachment involves clearing in more densely-encroached areas and introducing SLM practices in all affected areas. Under the LDN Scenario, it would be necessary to clear the 44 781 ha that had already undergone major encroachment from 2015 to 2021. Under a phased treatment approach to spread costs, clearing these areas would require an estimated investment (PV) of R238 million. For the lower-bound LDN costing, it was assumed that further bush encroachment could be contained through SLM. Under the upper bound LDN costing, it was assumed that SLM would be completely ineffective, necessitating further clearing to offset the projected increase in bush encroachment, increasing the cost to R507.2 million. The Full Restoration scenario required clearing the 130 241 ha that had undergone major encroachment since 2005, at an estimated cost of R691 million.

Under the LDN Scenario, SLM measures were applied to the 53 771 ha threatened with degradation between 2021-2030, at a cost over 25 years of R2.0 billion. Under the Full Restoration Scenario, SLM measures were applied to much larger area of 335 309 ha, at a cost of R6.1 billion.

The results of the cost-benefit analysis suggest that the implementation of restoration interventions and SLM in the Thukela catchment could result in a net benefit under LDN scenario, provided that SLM measures are at least partially effective. Furthermore, they suggest that Full Restoration would have a significantly better outcome than just meeting LDN targets. Using a discount rate of 3.66%, the net present value over 25 years of undertaking LDN interventions could be as high as R435.5 million, but for full restoration, this increases to R6 389.6 million (Table II). Note that this does not include the non-use or intangible values associate with the restoration of habitats and biodiversity, which are likely to be considerable.

Table II. Present value of the costs of interventions and ecosystem service benefits relative to BAU under the LDN and Full Restoration scenarios (2020 R millions, 3.66% discount rate, 25 years).

	Present value (R millions) base estimate		
	LDN Scenario		**Full Restoration Scenario**
Costs relative to BAU	Upper bound costs	Lower bound costs	
Clearing IAPs	514.4	514.4	2 355.2
Addressing Bush Encroachment	507.2	237.6	691.1
Active restoration of grasslands, erosion	2 623.6	-	-
Sustainable land management	-	1 981.02	6 093.62
Total present value of costs	**3 645.18**	**2 733.09**	**9 139.98**
Benefits relative to BAU			
Water supply	2 591.4	2 591.4	10 757.2
Sediment retention	38.9	38.9	63.1
Tourism	121.8	121.8	243.6
Carbon storage (avoided national cost)	-274.91	-274.91	597.5
Harvested resources	70.6	70.6	2 391.3
Livestock production	620.7	620.7	1 476.9
Total present value of benefits	**3 168.6**	**3 168.6**	**15 529.6**
Net Present Value	**-476.6**	**435.5**	**6 389.6**
BCR	**0.9**	**1.2**	**1.7**

Conclusions and recommendations

Implementing LDN in South Africa is already an imperative at any cost. Implemented timeously, it simply means halting further degradation and undertaking some restoration to make up for where degradation has not been completely halted. This would be expected to be far less costly than restoring past degradation. In South Africa, where there has been some delay in stepping up action to stem degradation, achieving LDN will now also involve restoring the degradation that has taken place in the six years since 2015. Nevertheless, the benefits of achieving LDN in 2030 relative to 2015 are likely to outweigh the costs, especially if SLM measures are implemented in an effective manner. Moreover, it would be even more beneficial to go above and beyond the minimum requirements of LDN, to address degradation that has taken place before 2015. In the Thukela catchment, much of the degradation of grassland areas took place well before this. These areas have the potential to make a valuable contribution to biodiversity, ecosystem services and livelihoods. Thus, in the case of the Thukela catchment, restoration above and beyond the obligations under LDN would be desirable.

These results are conservative, in that they do not include all the values associated with healthy ecosystems, and do not consider the increasing scarcity value of intact natural systems. In particular, we have not included estimates of the non-use or intangible values associate with the restoration of habitats and biodiversity, which are likely to be considerable and increasing over time.

The conclusions were also supported by the sensitivity analysis, which was necessary due to some of the uncertainties involved in the mapping and projection of land cover and ecosystem condition, modelling and valuation of ecosystem services, and the costs and effectiveness of measures to address the drivers and effects of degradation. These uncertainties could be reduced with further monitoring and research to advance natural capital accounting as a monitoring tool and to inform national strategy and priorities to address land degradation, including:

- Developing high resolution time series data of IAPs at national scale using field data and technological innovation;
- Defining the reference condition for terrestrial ecosystems, and determining ecosystem condition from a defined base year and going forwards that takes all forms of degradation into account;
- Undertaking further research to improve modelling of the effects of changes in ecosystem condition on hydrological and other ecosystem services; and
- Undertaking scientifically rigorous research into the efficacy of different interventions addressing land degradation in different biophysical and social contexts.

For example, even with the very important and increasing contribution of remote sensing, some of the challenges faced in this study have emphasised the importance of collecting ground and/or aerial baseline and monitoring data to improve the accuracy and consistency of classifying land cover and ecosystem condition. This includes improving the techniques for assessing and mapping the extent of IAP invasions in South Africa to generate a dataset that is consistent with the Standard for National Land Cover structure and resolution. The compilation of more accurate ecosystem accounts over a series of years will be invaluable in guiding future interventions and monitoring their outcomes. Indeed, the existing ecosystem accounts were already a very useful starting point for this study.

Ecosystem accounting will also be better able to inform such policy decisions through further baseline research on the supply of ecosystem services. For example, there are few data on the stocks of resources in different habitats or how they vary with ecosystem condition. One of the key challenges was modelling the hydrological effects of complex land cover changes at large scale. This did not yield satisfactory results within the time available, and we instead relied on existing estimates derived with simpler models. Future studies will need to start by exploring these complexities at smaller scales.

While availability of information allows for relatively robust estimates of restoration costs, there is still considerable uncertainty regarding the methods, costs and effectiveness of achieving SLM. Land degradation in the study area occurs as a result of poor regulatory frameworks, planning and implementation around land tenure and farming activities, as well as socio-economic drivers such as high population densities, market access, cultural norms and poverty. Without addressing the institutional issues behind land degradation, SLM efforts carry a risk of failure. For this reason, our study included upper and lower bound estimates of costs to encompass the range of possibility in this regard. Strategies for achieving LDN should assume a middle road, and place strong emphasis on addressing the underlying issues.

Finally, this study has also shown that, while addressing land degradation can be shown to be worthwhile from a societal point of view, it still comes at a considerable financial cost. Innovative and integrated funding solutions will be needed to achieve LDN or more ambitious restoration goals. Public investment should therefore include a focus on providing the enabling conditions that foster the private sector investment needed.

1 INTRODUCTION

1.1 Project background

This study forms part of the Natural Capital Accounting and Valuation of Ecosystem Services (NCAVES) project, which was launched in 2017 by the United Nations Statistics Division (UNSD) and United Nations Environment Programme (UNEP) with funding from the European Union (EU). The NCAVES project aimed to assist five participating partner countries, including South Africa, to advance the knowledge agenda on environmental and ecosystem accounting and initiate pilot testing of System of Environmental-Economic Accounting: Experimental Ecosystem Accounting (SEEA EEA), with a view to improving the management of natural biotic resources, ecosystems and their services at the national level as well as mainstreaming biodiversity and ecosystems in national level policy, planning and implementation.

Under the NCAVES project, pilot monetary ecosystem accounts (hereafter pilot accounts) were compiled at sub-national scale, for the province of KwaZulu-Natal (Turpie *et al.*, 2020b). These accounts built on the physical ecosystem extent and condition accounts for 2005 to 2011 (Driver *et al.*, 2015), adding estimates of the supply and use of ecosystem services in physical and monetary terms, and estimates of the value of ecosystem assets in the province, taking sustainability of past and present uses into account. Spatial models were developed for ten categories of ecosystem services in order to quantify and value the supply of ecosystem services from the ecosystem assets across the province. The results highlighted the spatial variability in the supply and use of different ecosystem services, costs of environmental degradation and loss of natural ecosystem functioning. The ecosystem monetary asset account shows the changes in the value of ecosystem assets for 2005 to 2011.

Although there is still work to be done to complete the coverage of ecosystem services and refine the estimates made in the pilot accounts, the results from the pilot study provided interesting insights and highlighted important environmental issues that require urgent policy intervention. Across the ten broad groups of services assessed, the combined value of the annual flow of ecosystem services was R34.2 billion in 2011 or 7.6% of the provincial GGP. While this is a significant contribution, it is apparent that the values of many of the services have decreased over time. The monetary asset account showed reductions in ecosystem values of some R22.8 billion over the six-year period (note that all values were expressed in constant 2010 Rands). Most of what was lost was through degradation and losses in the grassland and savanna biomes. This was a common finding across almost all the services. The value of harvested wild resources decreased by almost R500 million in these two biomes, carbon by close to R400 million and the value of livestock production by just over R300 million. These losses in value were primarily due to loss of vegetative cover driven by overgrazing in rangeland areas and a loss of natural habitat due to the expansion or densification of urban, peri-urban and rural settlements. Nature-based tourism was the only service where the value associated with grasslands and savannas increased. Unsurprisingly, much of this value was in protected areas where harvesting of wild resources and grazing of livestock is either prohibited or restricted.

This scenario analysis study followed on from the pilot accounts with the aim of demonstrating the usefulness of the accounts and the datasets and models developed in their compilation for identifying environmental issues and evaluating alternative management and policy options. The study was tasked to tackle a set of key environmental issues that can be addressed at local scale using interventions that also have the potential for scaling up, and that are aligned to existing targets, programmes and interventions and therefore of particular relevance to government stakeholders. Following stakeholder consultation and exploration of spatial data, the focus of the study was narrowed down to that of land degradation, specifically in relation to South Africa's land degradation neutrality (LDN) commitments.

Land degradation has been highlighted as one of the major causes of the loss of ecosystem services in the pilot accounts, and is also resulting in the decline of many indigenous plant taxa in South Africa. Land degradation in KwaZulu-Natal is mainly in the form of:

- Loss of vegetation and soil cover in rangelands and cultivated lands, as a result of poor management practices;
- Bush encroachment (increases in woody vegetation) in grassland and savannas as a result of poor grazing and fire management practices; and
- The spread of invasive alien plants.

The issue of land degradation is particularly relevant in light of South Africa's international commitment to achieve Land Degradation Neutrality (LDN) relative to 2015 by 2030 (von Maltitz *et al.*, 2019) [2]. LDN is defined as "a state whereby the amount and quality of land resources necessary to support ecosystem functions and services and enhance food security remain stable or increase within specified temporal and spatial scales and ecosystems" (UNCCD, 2016). The years 2021 to 2030 have also been declared the United Nation (UN) Decade on Ecosystem Restoration.[3]

1.2 Aim of the study

The main aim of the study was to provide insights into the consequences of land degradation and the costs and benefits of investing in measures to address this problem in KwaZulu-Natal. The study addresses two main questions:

1. What will be the cost and return on investment (ROI) of achieving land degradation neutrality (relative to 2015) by 2030? and
2. Is it enough to aim for LDN targets, or is further restoration justifiable?

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[2] LDN targets are voluntary targets that parties to the United Nation's Convention to Combat Desertification (UNCCD) have formulated. South Africa went about setting these targets between 2016 and 2018 (von Maltitz *et al.*, 2019).

[3] This unites the world behind a common goal of halting, preventing and reversing the degradation of ecosystems. This includes restoring 350 million ha of degraded landscapes globally by 2030.

2 STUDY AREA

2.1 Geographic context

The study focused on a smaller geographic area than the provincial-scale pilot accounts, since the land cover data needed to be augmented with additional information on land degradation, and the hydrological modelling needed to be updated accordingly. This study focused on the Thukela River catchment area (Water Management Area "V"), which is one of the least modified catchment areas in KwaZulu-Natal (in terms of conversion to agriculture or other land uses), and one in which the geographic extent of the land degradation problems discussed above is greatest. The Thukela catchment, which extends from an altitude of 3282 m to sea level, falls almost entirely within the province and at over 2.91 million hectares, occupies about a third of its area (Figure 2.1). It is an important water source area, with several important water supply reservoirs located in the higher lying parts of the basin. This project did not include areas in the catchment area that fall outside of KwaZulu-Natal as no consistent land cover products could be utilised for those areas.

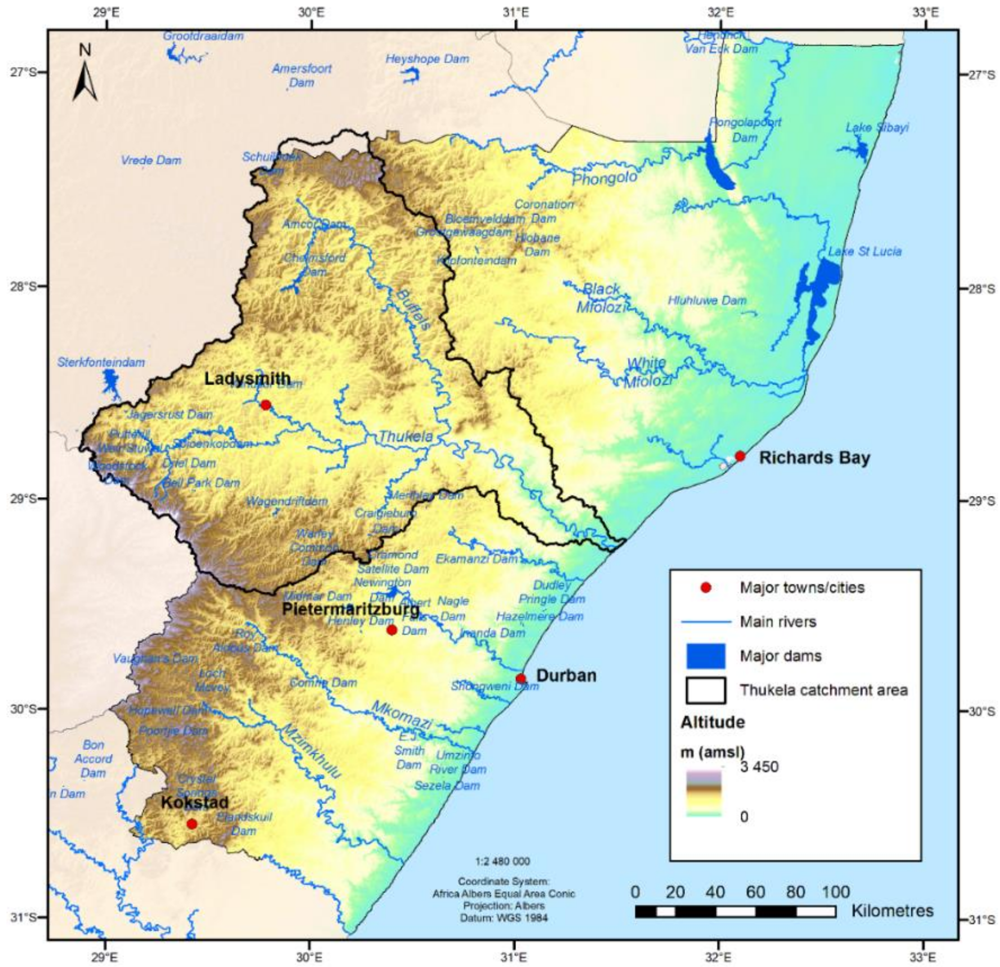

Figure 2.1. Location of the Thukela catchment in KwaZulu-Natal showing topography, drainage and reservoirs.

The Thukela catchment falls within six of South Africa's nine biomes (including azonal vegetation such as wetland, riparian and estuarine vegetation). Grassland is the dominant biome, covering over 73% of the total area, followed by Savanna with just over 21%. Forest and the Indian Ocean Coastal Belt (IOCB) make up just 1.5% of the total area, with the remainder comprising waterbodies and Azonal Vegetation (Figure 2.2, Table 2.1). The catchment contains almost 6% of South Africa's Grassland area.

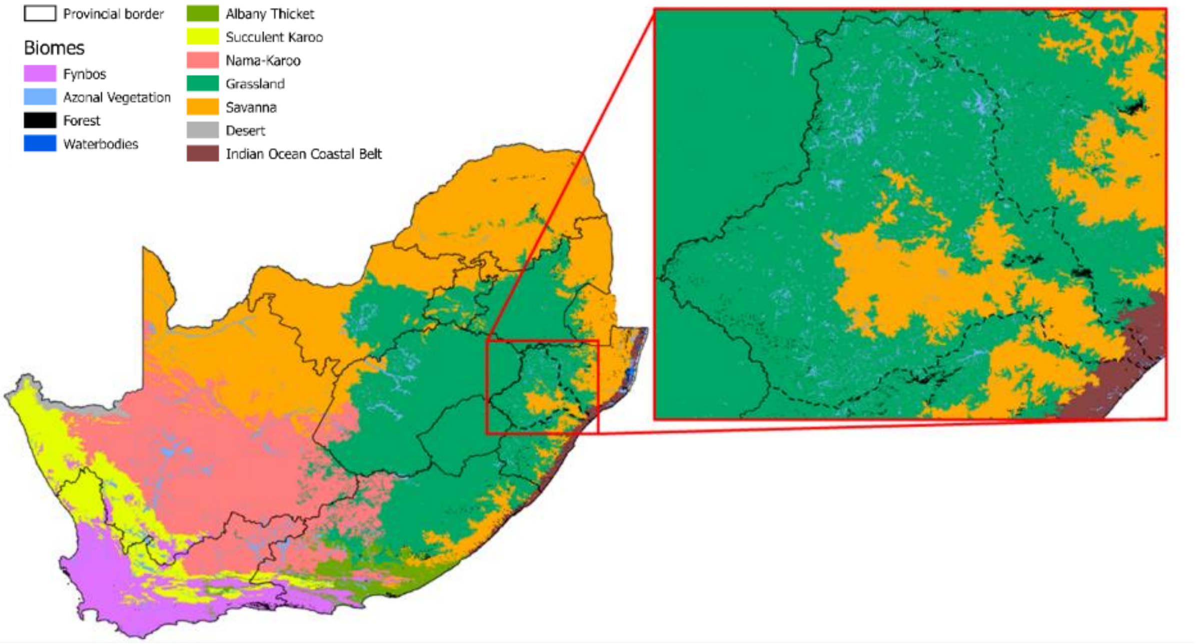

Figure 2.2. The biomes of South Africa and the Thukela catchment (indicated by the dotted outlined) in the inset map.

Table 2.1. The area (ha) and proportion of each biome in the Thukela catchment relative to the whole of the South African mainland.

Biome	Area in South Africa (ha)	Area in Thukela catchment (ha)	% of catchment	% of national area within Thukela catchment
Forest	515 896	18 895	0.7%	3.7%
Grassland	36 253 335	2 124 414	73.1%	5.9%
Savanna	40 423 102	616 425	21.2%	1.5%
Indian Ocean Coastal Belt	1 144 916	20 126	0.7%	1.8%
Azonal Vegetation*	3 208 242	125 295	4.3%	3.9%
Waterbodies*	65 296	479	<0.1%	0.7%

*Wetlands (not included as a biome), as per the SANBI National Wetland Map 2018, amount to 16 923 ha in the Thukela catchment area, substantially less than the area of the Azonal Vegetation area

Much of the basin is communal land under the control of the various tribal authorities[4] under the Ingonyama Trust, and most of this is in the savanna biome, apart from pockets of communal lands in the uppermost catchment areas (Figure 2.3, Figure 2.4). With almost 80% of the basin (2.32 million ha) still under natural vegetation, it contains the bulk of remaining undeveloped land outside of protected areas in the province (Figure 2.4). However, much of this land has become degraded and is under threat of further degradation, as discussed in the following sections.

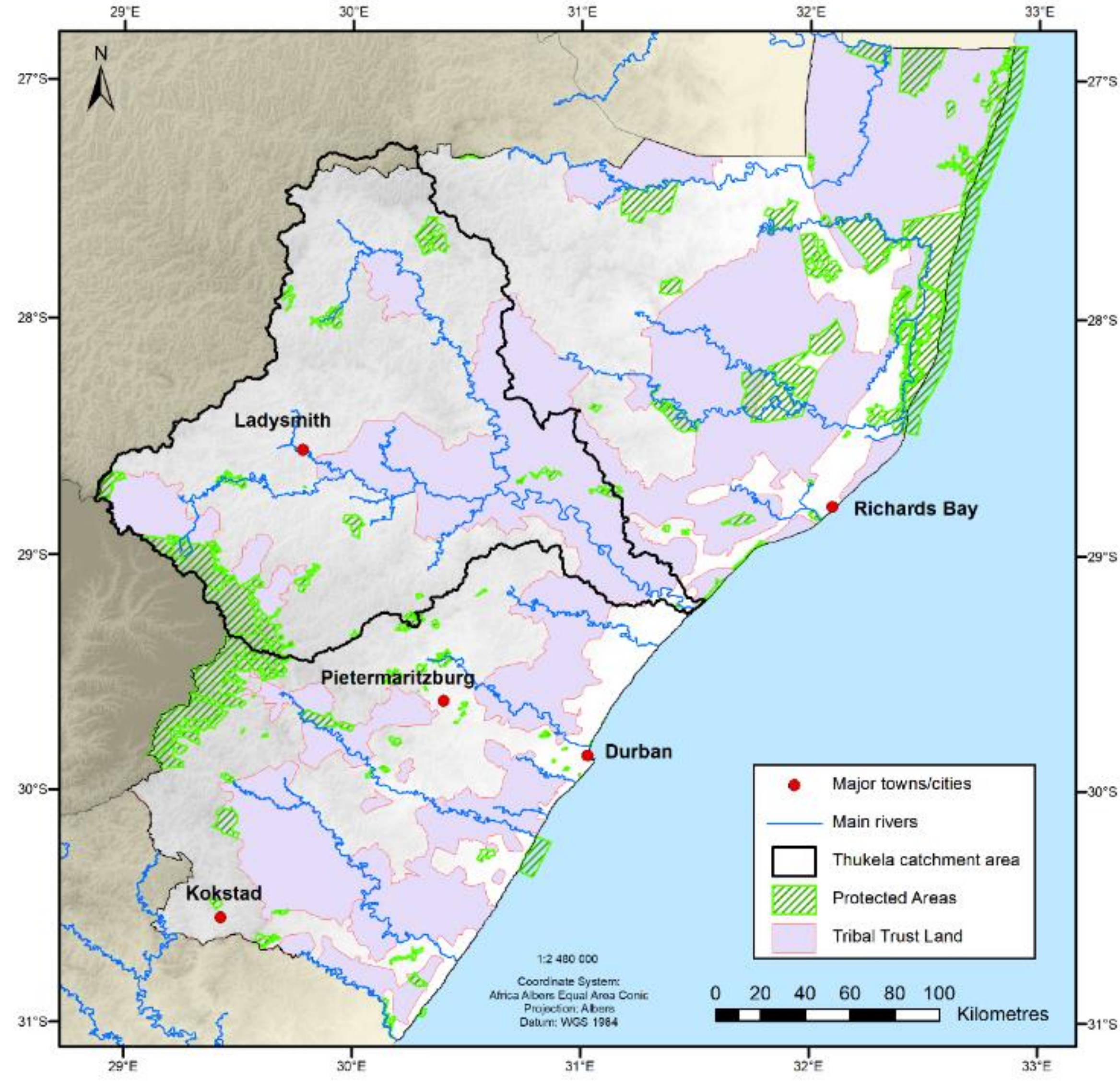

Figure 2.3. Land tenure of the Thukela catchment in the context of KwaZulu-Natal.

[4] Examples are the aMazizi and aMangwane in the mNweni region near the source of the Thukela River in the upper catchment.

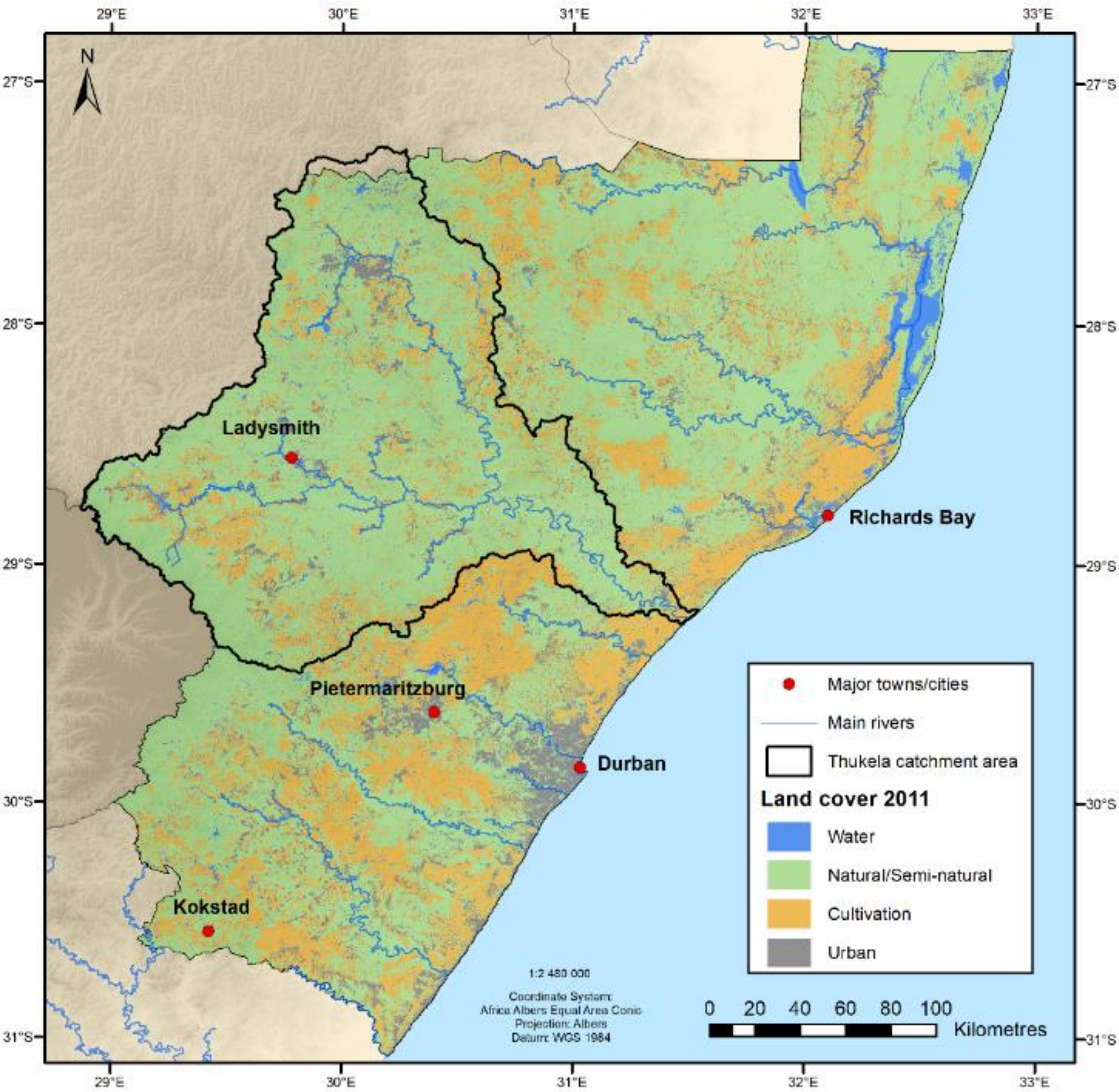

Figure 2.4. Broad categorisation of land cover of the Thukela catchment in the context of KwaZulu-Natal.

2.2 Land degradation

2.2.1 Invasive alien plants

The spread of IAP species is well-studied and is a well-known driver of degradation worldwide. IAPs are considered to be the second biggest pressure on ecosystem functioning, accounting for 25% of biodiversity loss (Poona, 2008; van Wilgen & Wilson, 2018), including having resulted in 40% of the increases in threat status of indigenous plants in South Africa (Republic of South Africa, 2018) and the biggest driver of land degradation in the country (von Maltitz *et al.*, 2019). While several invasive taxa occur in South Africa, most of these are relatively well controlled and pose manageable risks, whereas IAPs pose a serious threat with by far the most prolific spread and number of species in South Africa (van Wilgen & Wilson, 2018).

While some IAPs are relatively benign, certain species, once established, can negatively impact ecosystem functioning, of which streamflow reduction is one of the most serious impacts. In KwaZulu-Natal, IAPs include a suite of species such as gums *Eucalyptus spp.*, Australian wattles *Acacia spp.* and pines *Pinus spp.* that tend to spread from existing plantations or stands and invade water courses, affecting water flows and reducing ecosystem functioning and provisioning (Richardson & Van Wilgen 2004; Figure 2.5). There are a host of other species which have impacts on a range of ecosystem services and habitats. In the catchments of KwaZulu-Natal, IAPs are estimated to reduce water flows by 2.3-5.0% (Le Maitre *et al.*, 2016). The Thukela catchment has been estimated to have the highest reduction in water yield as a proportion of the registered water use in the catchment, of not only KwaZulu-Natal, but the entire country, with an estimated loss of 2.6% of its total runoff due to IAPs. This could rise to 16% if left unmanaged (Cullis, Görgens & Marais, 2007; van Wilgen & Wilson, 2018). The impacts of IAPs also include diminished agricultural productivity with impacts on both livestock and crops, with total losses in South Africa estimated to be in the billions of Rands (Poona, 2008).

Combatting IAPs requires a multi-layered approach including identifying areas of IAP invasions and developing technologies to improve detection, environmental education, assistance in declaring and clearing invasion, developing legislation and policy, and finally, clearing/removal of IAPs and following up clearing. South Africa has made significant strides in all of these aspects. Most notably successes include clear legislation on management and listing of IAPs into various categories[5] in terms of propagation and growing of IAPs, and undertaking the use of biocontrol agents for IAP

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[5] The Conservation of Agriculture Resources Act 1983 (Act 43 of 1983) (CARA) lists 198 Invasive Alien Plants (IAPs) in four categories: Category 1a: Invasive species which must be combatted and eradicated. Any form of trade or planting is strictly prohibited, Category 1b: Invasive species which must be controlled and wherever possible, removed and destroyed. Any form of trade or planting is strictly prohibited. Category 2: Invasive species, or species deemed to be potentially invasive, in that a permit is required to carry out a restricted activity. Category 2 species include commercially important species such as pine, wattle and gum trees. Plants in riparian areas are Category 1b. Category 3: Invasive species which may remain in prescribed areas or provinces. Further planting, propagation or trade, is however prohibited. Plants in riparian areas are Category 1b.

containment, which may be especially useful for controlling high water use IAPs. Clearing alien vegetation is more cost effective[6] than developing further water augmentation infrastructure and can yield significantly increased runoff and water supply per area and at the same time avoiding opportunity costs (Marais & Wannenburgh, 2008; Morokong *et al.*, 2016; The Nature Conservancy, 2019).

Figure 2.5. Google Earth images from the upper catchment taken in 2005 (top) and 2017, showing spread of IAPs.

Despite the wealth of literature on IAPs in South Africa, compiling detailed spatial data has been challenging (CommonGround, 2003; von Maltitz *et al.*, 2019). IAPs are difficult to detect from satellite imagery and show similar spectral characteristics to native vegetation. Their spread within a landscape is also difficult to model. While Shezi & Poona (2010) noted the challenges of detecting and accurately assessing IAP distribution and spread, they also commented on the great potential

[6] Actual clearing costs and overhead costs.

using increasingly sophisticated remote sensing technology and classification methods to refine IAP maps. However, there appears to be very little done since then in South Africa, especially over a large scale. Rather techniques akin to habitat suitability modelling have been applied which predict where IAP species *may* and are *likely* to occur, which can assist in their management and control prioritisation but do not help in more refined accounting of degradation and determining spread. The most detailed data available are from the National Invasive Alien Plant Survey (NIAPS) project (Kotzé *et al.*, 2010), which has produced a 250 m^2 resolution raster layer detailing the density of several key IAP species using a statistical approach combined with low level aerial surveys.

Based on the NIAPS dataset, the Thukela catchment area had just under 81 000,[7] condensed hectares (con ha) of IAPs in 2010, which at the time was over 5% of the IAPs in the country (Van Wilgen & Wilson 2018; Figure 2.6). The dataset shows that four species are the main IAP culprits in the Thukela catchment in terms of condensed area. They are (in descending order) Australia wattles *Acacia spp.*, gums *Eucalyptus spp.*, pines *Pinus spp.* and prickly-pears *Opuntia spp.* Wattles are by far the most prolific, and thus they and possibly one or two of the other species would form priority species for being assessed. According to the NIAPS data, there are three key areas of high IAP infestation; the quaternary catchments north-west of Newcastle (quaternary catchments V31C, H and J), around the Rosetta/Mpofana (V20D and B) and the Oqaqeni area (northwest of KwaDukuza, V50B and C; Figure 2.6).

2.2.2 Loss of vegetative cover and erosion

Land degradation in the form of natural vegetative cover loss involves the removal of the biological productivity of the topsoil. It is caused by human activities and is exacerbated by poverty and the adverse impacts of climate change, amongst many other aspects. South Africa's land degradation has been compounded not only by being susceptible due to climatic (water-scarce, low rainfall, semi-arid) and geological (highly erodible in many parts) factors, but also political and social factors which have created a dual system of land tenure, placing enormous strain on select parts of the country where overgrazing and erosion are rife (Hoffman & Ashwell, 2001; Clover & Eriksen, 2009). Roughly 720 000 ha of sheet and gully erosion covers South Africa (Department of Environmental Affairs, 2017a).

In spite of its relatively high rainfall, KwaZulu-Natal has a high provincial veld degradation index and one of the highest provincial indices of soil degradation and susceptibility to donga formation (Figure 2.7). This undermines the productive potential of land and water resources in this area and presents serious challenges in terms of resilience to drought. Soil erosion is a particularly serious problem in the upper catchment areas of the province and communal land (Figure 2.6 – note only gully erosion is displayed), mainly linked to poor grazing management and abandoned agricultural fields (Hoffman & Todd, 2000; Peden, 2005; Sonneveld, Everson & Veldkamp, 2005). That said,

[7] Of which 63 446 condensed ha were wattle, gum and pine within the KwaZulu-Natal portion of the catchment area.

there is little evidence to suggest these areas (and similar areas in South Africa) have declining production (von Maltitz *et al.*, 2019).

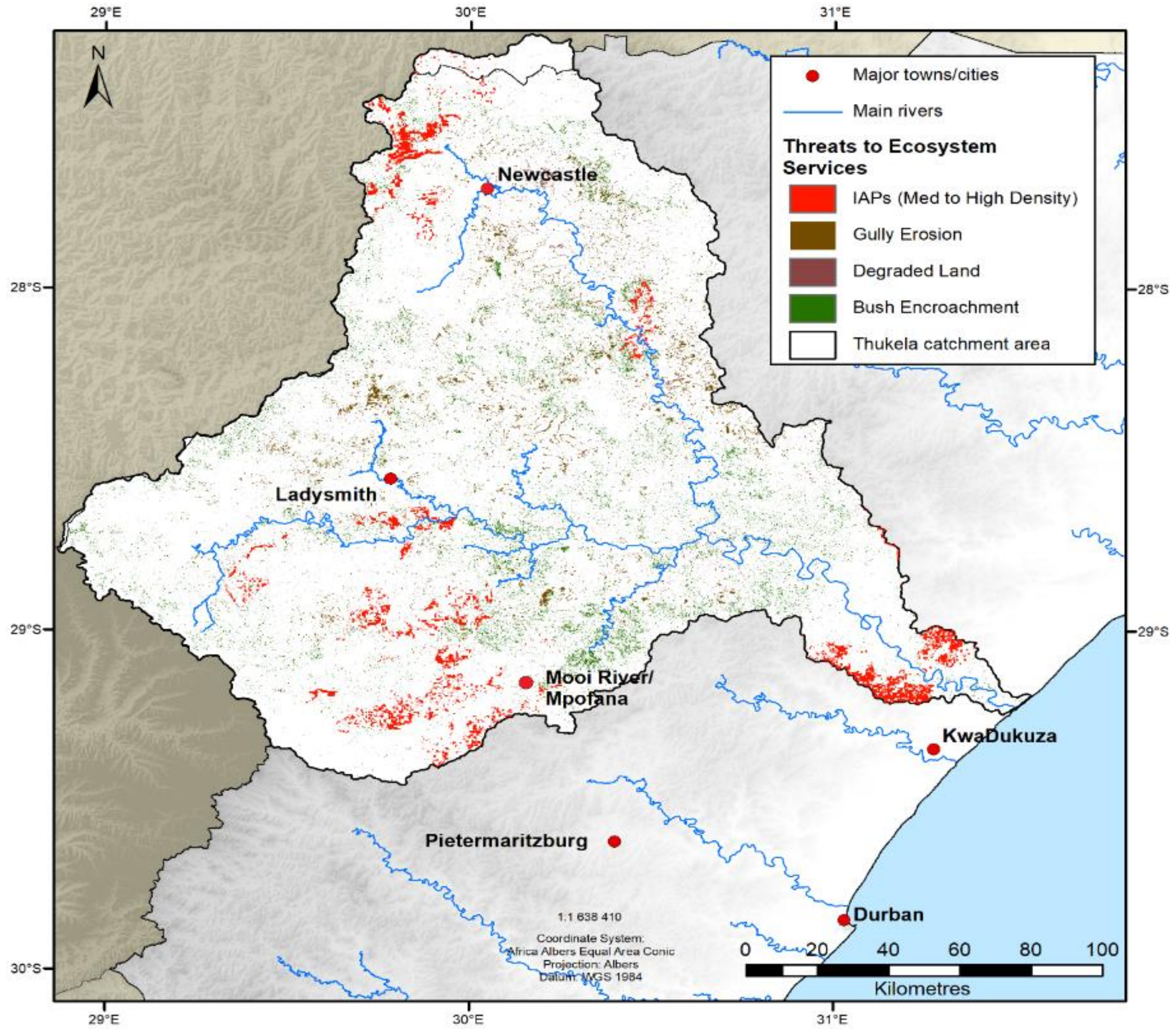

Figure 2.6. Locations of the main forms of land degradation in the Thukela River catchment area ca. 2011. (Data sources: IAPs - Kotze et al. *(2010), Gully erosion, degraded land, bush encroachment - KZN Land Cover, EKZNW and GeoTerra Image (2005, 2011).*

The steep topography combined with sedimentary geology in many parts of the catchment area make the soil particularly susceptible to erosion resulting in diminished vegetation cover. This is exacerbated by poor land management practices and natural hazards such as intense thunderstorms and wildfires. The erosion of fertile soil has widescale implications for food security, as well as for water infrastructure (e.g. siltation of reservoirs) and the costs of water treatment. Erosion is often linked to poor livestock and crop management,[8] especially in areas of steep

[8] Such as overgrazing, short fallow periods, removal of indigenous vegetation and poor irrigation and planting practices (Hoffman & Ashwell, 2001).

topography which characterise the Thukela catchment. The reduction in crop production potential and livestock capacity increase the impacts on livelihoods, especially considering the importance of livestock for supplementing cash flow (Stronkhorst *et al.*, 2010b). Declining crop potential is exacerbated by abandonment of cultivated land, both where crop potential has declined enough and where people have abandoned areas and emigrated elsewhere. These areas fall fallow and although the often recover to some point, often remain degraded to some level, rarely returning to a truly healthy natural state. This can have a positive feedback in increasing overall levels of degradation in a region. While degradation and loss of vegetative cover is extensive, there are numerous areas where there is recovery, predominantly attributed to reduction of livestock (see section 0).

Figure 2.7. Severe gully erosion near Mabhulesini in the Upper Thukela Catchment (Photo: Joshua Weiss).

2.2.3 Bush encroachment

Bush encroachment involves the increase and spread in the abundance of a single or few dominant indigenous woody vegetation species, particularly in the grassland and savanna biomes. In South Africa, where these biomes make up 27.9% and 32.5% of the land surface area, respectively, there has been a significant increase in tree cover since national-scale aerial photography was first undertaken in the 1940s, with estimates between 5.7 to 13 million hectares, and an increase in woody plant land cover of over 2.7 million hectares since 1990 (Skowno *et al.*, 2017; Turpie *et al.*, 2019).

Bush encroachment occurs as a result of poor land management practices, mainly overgrazing and active reduction in the intensity or frequency of fires, and is also facilitated by increased carbon in the atmosphere (Rohde *et al.* 2006; Turpie *et al.* 2019). It may also be due to historical reductions in megaherbivores which play an important ecological role through browsing and damaging trees (Russell & Ward, 2016; Norris *et al.*, 2020), which are now exclusively restricted to protected areas and game ranches. Bush encroachment is distinct from alien invasions in that it is largely a result of *in situ* management actions, unlike in the case of invasive alien plants which spread onto land as a result of past introductions elsewhere in the landscape. Bush encroachment is not caused by particular species but is rather a change in balance of the types of plants occurring in ecosystems. That said, over 40 species have been listed as problematic species in terms of bush encroachment, and new species are being added (Turpie *et al.*, 2019). The same species are likely to be benign and useful at their natural densities.

Bush encroachment changes the structure and composition of ecosystems. It impacts on biodiversity and ecosystem functioning, and hence the supply of ecosystem services. It reduces commercial and communal rangeland productivity, since grassy biomass is reduced, and may lead to further damage from overgrazing in a negative feedback as suitable grazing areas shrink. It changes the availability of woody and non-woody resources in the landscape (Russell & Ward 2016, Turpie *et al.* 2019), and aboveground carbon sequestration (Ward, 2005). It can affect soil infiltration rates, groundwater recharge and surface runoff. The increase in woody cover in protected areas and game reserves can have a significant negative impact on ecotourism as the game viewing experience is affected through poor visibility (i.e. dense bushy vegetation prevents sightings of wild animals, Arbieu *et al.*, 2017).

Up to a point (about 45% tree cover; O'Connor, Puttick & Hoffman 2014), bush encroachment can be reversed by improving land management practices such as rotating livestock and ensuring adequate fire regimes are put into effect (Turpie *et al.*, 2019). However, beyond that point, the only choice is active clearing which can be labour-intensive and costly.

Bush encroachment in KwaZulu-Natal appears to have been taking place since the early 20th century, and the province now has some of the highest levels of bush encroachment in the country. Most bush encroachment has occurred in grassland and open woodland habitats, affecting primary productivity in these areas and thus degrading land (Russell & Ward, 2016). The relatively new presence of some woody species is a threat to soil stability in many parts of the province (Grellier *et al.*, 2012). Researchers are still grappling with how to identify areas where bush encroachment has occurred and rates of change (O'Connor *et al.*, 2014). Based on changes in land cover to woodier classes, bush encroachment has been particularly severe in the middle reaches of the Thukela Catchment (Figure 2.6).

2.3 Existing activities to address land degradation

There has been a fair amount of management effort by government (DFFE) to address land degradation problems in the Thukela catchment. In 2018/19, nearly 4 000 hectares of IAPs were cleared, employing just over the equivalent of 1500 people (C. Marais, DFFE, *pers. comm.*). In the 2019/20 financial year, 1007 hectares were initially cleared and 1863 hectares underwent follow-up clearing. Currently, IAP clearing interventions are being carried out mostly in the upper and middle reaches of the Thukela catchment. The upper parts of the catchment are seen as a priority, although the steep topography makes tackling the problem more challenging due to reduced accessibility. No government interventions are taking place in the lower third of the catchment area. The current level of effort is not nearly enough to stay ahead of the problem, however (C. Marais, *pers. comm.*).[9]

Most erosion rehabilitation is focused on the upper catchment in the uThukela District Municipality. Similarly, there have been efforts to address farming and livestock management practices by DALRRD but these have not been successful, partly due to the extent of degradation, and partly due to lack of attention to the social context or a lack of continued funding to ensure sustainability and desired results (J. Botha, KZN DARD, *pers. comm.*). There has not been any dedicated action against bush encroachment at this stage, but it does now have the attention of DFFE as a form of land degradation that needs to be addressed.

9 In a conversation with DFFE, it was estimated that a budget four to seven times the existing budget for land rehabilitation (focusing on IAP clearing) would be required to effectively deal with the issues in the Thukela catchment. Van Wilgen *et al.* (2020) argue that the national figure could be up to 8.6 times higher than existing for national level and that priorities in terms of area need to be undertaken. They also state that monitoring outcomes rather than the inputs is needed.

3 LAND DEGRADATION POLICY AND ACTION IN SOUTH AFRICA

3.1 Early efforts at addressing land degradation

The first formal policies aimed at curbing land degradation in any form are said to have been initiated as early as 1861, when the Colony of the Cape of Good Hope promulgated the Xanthium Spinosum Act (Lukey & Hall, 2020). Intervention policies in the 20th century were primarily aimed at commercial farmers, through subsidies and the provisioning of extension services who advised on cultivation practices and advisory councils on soil conservation. These measures proved very successful until the mid-1990s, but because they had almost exclusively benefited white commercial farmers, they were largely withdrawn after South Africa became a democracy in 1994. While extension services have since been reinstated to an extent, the quality of engagement with commercial farmers is reportedly not satisfactory.

In the communal land areas, “Betterment Schemes” were introduced from the 1940s to the 1980s. These attempted to introduce practices such as rotational grazing and to reduce livestock numbers but were actively resisted by communities and were not successful (Peden, 2005). Very little assistance was given to these areas thereafter, until more recently. However, these too have failed to achieve many successes. Extension services have also been introduced in these areas, but have all but collapsed due to a lack of technical and institutional expertise or poor knowledge transfer to communities and farmers, and a focus on production rather than the traditional use of livestock (Peden, 2005; J. Botha, *pers. comm.*). Communal areas suffer from governance mismatches between local municipalities and tribal authorities with little planning, minimal funding and little direction in terms of management. Within the communal land areas, there has often been little progress after the introduction of initiatives, due to several reasons, both internal and external to the communities. In general, communal land areas are still beset by a lack of implementation of sustainable farming and rangeland management practices, leading to increasing land degradation. Some more remote regions have very little interaction with any initiatives or departmental efforts (Peden, 2005; Stronkhorst *et al.*, 2010a), and are often the most critical owing to their location within strategic water source areas. In some areas, rangeland degradation has been so severe as to be classed as “beyond repair” (J. Botha, *pers. comm.*).

Measures have also been in place in South Africa for a number of decades to protect lands from the spread of IAPs and mitigate their effects on water supply. In the 1970s, the Mountain Catchment Act came into effect in order to ensure adequate protection of high water-yield areas. In 1983, the Conservation of Agricultural Resources Act (No 43 of 1983; CARA) was promulgated to ensure agricultural assets (and thus economic interests) were better protected, particularly with respect to soil conservation (Lukey & Hall, 2020).

3.2 Policy development after 1990

South Africa's environmental policies improved dramatically after becoming a signatory to the UN conventions on biodiversity (UNCBD), to combat desertification (UNCDD) and climate change (UNFCCC) in the 1990s and following the development of a new constitution and overhaul of all policies after 1994. Since then, the country has undertaken regular studies to report on the state of biodiversity, land degradation and its response to climate change, as required by these conventions. A national survey of Land Degradation in South Africa was undertaken in 1999 (Hoffman & Todd, 2000). This remains one of the most comprehensive undertakings to date and has informed public policy since.

In the last decade, land degradation policy has largely been driven by the global environmental agenda. South Africa is obliged to develop targets to combat land degradation, carry out action to do so, and report on the status of degradation. The Land Degradation National Action Plan (NAP) was completed in 2004. An updated NAP for the period 2017-2030 is still under review (see Department of Environmental Affairs, 2017a). The NAP includes seven key outcomes. Outcome 6 is of particular relevance; "By 2030 South Africa to ensure that degraded ecosystems are restored whilst contributing to ecosystem services delivery, climate change adaptation and mitigation". The details include:

1. Protect and conserve ecosystems and their services (through sustainable land management and ecosystem-based adaptation), to increase drought resilience (R2bn p.a.);

2. Strengthen communities' ability to adapt to the effects of desertification, land degradation and drought;

3. Identify, and where relevant, manage and control areas under bush encroachment and IAPs (R2 bn p.a.); and

4. Identify communities and landscapes at high risk of desertification, land degradation and drought (R0.5 bn p.a.).

Outcome 7 was to formulate a policy and targets for achieving land degradation neutrality (LDN).

The problem of bush encroachment and its seriousness has only recently been recognised, and both the most recent NAP and the LDN targets (discussed below) include dealing with bush encroachment. However, there was some debate as to whether bush encroachment should be allowed as a carbon sequestration measure or treated as a form of land degradation. Based on a consideration of multiple ecosystem services, the latter has been recommended (Turpie *et al.*, 2019). Legislation will need to catch up with this. Bush clearing within an important biodiversity area or affecting listed species can require authorisation under the National Environmental Management Act (NEMA), National Forestry Act (NFA) or Biodiversity Act (NEMBA). There is some level of ambiguity as to the treatment of bush thinning, as opposed to complete clearing, e.g. with a bulldozer (Turpie et al. 2018).

3.3 Land Degradation Neutrality

In 2015, 17 sustainable development goals (SDGs) were adopted by world leaders as a roadmap for the next 15 years. One of these, Goal 15 "Life on Land" tackles the widespread problem of land degradation through target 15.3: "*By 2030, combat desertification, restore degraded land and soil, including land affected by desertification, drought and floods, and strive to achieve a land degradation-neutral world*." This requires a commitment to avoid degradation, to move towards sustainable land management (SLM) and at the same time to massively scale up the rehabilitation of degraded land and soil. It also constitutes a response to climate change action, food, energy and water security, forced migration and resource-driven conflict. By reaching land degradation neutrality, communities everywhere should be able to thrive by building on a healthy and productive foundation. The indicator proposed for SDG target 15.3 is: "*Proportion of land that is degraded over total land area*" (referred to as SDG indicator 15.3). The three sub-indicators are: land cover and land cover change; land productivity; and carbon stocks above and below ground.

The UN defines land degradation neutrality (LDN) as "a state whereby the amount and quality of land resources necessary to support ecosystem functions and services and enhance food security remain stable or increase within specified temporal and spatial scales and ecosystems (UNCCD, 2016).

As part of South Africa's obligations in this regard, the LDN targets have been established, which guides the NRM programmes and assists the country in meeting its SDG targets and obligations. Between 2016 and 2018, the country set its LDN targets for 2019, with support from the Global Mechanism (Department of Environmental Affairs, 2017b; von Maltitz *et al.*, 2019), which included:

- 20 integrated interventions in each of five key rural Strategic Water Source Areas;
- A total of 1 370 600 ha of land restored, with 3 230 271 ha of follow up treatment[10]; and
- 695 wetlands rehabilitated.

New voluntary LDN targets have since been set for 2030, which include clearing and following up on nearly 1.7 million ha of alien species and addressing land degradation in over 7.2 million ha of grassland, shrubland and sparsely vegetated areas (Department of Environmental Affairs, 2017a, 2017b; Republic of South Africa, 2018). These are determined with respect to a 2015 baseline extent and state. The LDN response strategy revolves around avoiding degradation, reducing degradation and restoring degraded lands. A range of measures have been suggested including improved grazing management, erosion control, clearing alien species, bush clearing and SLM practices. South Africa already has several laws and policies aimed at preventing land degradation, and DFFE has the mandate to promote the conservation and sustainable management of natural resources. Of particular relevance of DFFE's already existing National LandCare Programme (LandCare) and Natural Resource Management (NRM) programmes, which are discussed in more detail below.

South Africa has committed to achieving LDN (no net loss) by 2030 as compared to 2015 and to bring about an improvement in an additional 5% of the national territory (net gain). It specifies that

[10] In accordance with meeting Aichi Targets.

LDN will be achieved in the Grassland and Thicket Biomes. The specific targets to avoid, minimize and reverse land degradation are given in Table 3.1. There is no further information on the spatial aspects of this plan.

Table 3.1 South Africa's LDN area targets, and the percentage of the published baseline degraded area that is targeted for treatment

	Degraded area (ha)	**Action**	**Target area (ha) by 2030**	**%**
Cultivated areas	6 000 000	Improve productivity and SOC stocks	6 000 000	100%
Forest (FAO definition)	5 006 400	Rehabilitate and sustainably manage	1 809 767	36%
Savanna (<5m)	24 047 900		2 646 069	30%
Thicket			87 621	
Grassland			2 436 170	
Succulent Karoo			149 877	
Nama Karoo			528 632	
Fynbos			1 349 714	
Desert			76 525	
Wetlands	107 400	Rehabilitate	61 900	58%
IAPs*	1 510 600	Clear	1 063 897	70%
Bush encroachment	5 719 400#	Clear	633 702	11%
Artificial areas	353 900	Rehabilitate	350 000	99%

* It is assumed that this is affected area (at any density), not condensed ha, and that this is an estimate of the area infested since a recent baseline date (not given).
# Based on an estimated change from grassland to woodland cover from 1990 to 2014 (Skowno *et al*. 2019).

Very little detail is provided on how the LDN targets were arrived at, and crucially, it is not clear as to whether the biome and IAP or bush encroachment targets are additive or not. The biome targets are associated with "areas of declining productivity", which could mean reduction in NDVI only, or it could include areas affected by bush encroachment and IAPs. The figures quoted do not seem to tie up with the baseline estimates of degradation in the same report. Interestingly, the NRM programme has specified a much larger IAP clearing target of 8 069 456 ha for the same period (DEA 2017b). Their clearing targets cover 8% of the Savanna Biome, 6% of the Grassland Biome and 45% of the Indian Ocean Coastal Belt over the period 2016-2030.

3.4 The LandCare Programme

LandCare, initiated in 1997, is one of DFFE's flagship programmes that focuses on SLM as well as restoration and rehabilitation of degraded areas in rural areas to enhance productivity and food security (Department of Agriculture, 1999a; Mulder & Brent, 2006). The key components of the LandCare programme centre around conservation agriculture (farming that encourages minimal soil disturbance, crop rotation and permanent soil surface cover) and include soil care, water care, veld care, conservation agriculture and junior care. The LandCare programme funds community-based projects that include support for farmers and landowners, fencing, and IAP clearing.[11] Among its principles are fostering group or community based resource management in a participatory way, empowering sustainable livelihoods, building capacity within communities and between state actors at various levels, private entities, NGOs and communities, and blending together the appropriate upper level policy using bottom-up feedback mechanisms. The LandCare model involves partnerships that result in project plans with specific processes and outcomes which can be implemented and financed. These are managed by local communities, who also provide several of their own resources, primarily labour and materials. Assistance in the form of writing project proposals and on-site technical support is usually derived from provincial government. Private entities may assist with specific services or products (Department of Agriculture, 1999b; Mulder & Brent, 2006). While there have no doubt been successes countrywide, by 2005 it appeared to have to have achieved minimal results in communal rangelands (Peden, 2005), with poor uptake and a very low budget relative compared with the NRM programmes.

3.5 The NRM programmes

While many private and public initiatives have been put in place to address land degradation, the largest and best-known of these are the DFFE's **Natural Resource Management (NRM) programmes**, also known as the "expanded public works programmes". The different NRM programmes are as follows (Republic of South Africa, 2018: 89–90):

- **Working for Water** aims to prevent, contain and reduce the density and distribution of established, invasive alien species in order to reduce their negative effects on the environment.
- **Working for Wetlands** aims to rehabilitate wetlands to restore hydrological functions that underpin water flow and quality regulation.
- **Working for Forests** aims to improve the management of woody biomass resources to reduce the risks of invasions, increase biodiversity and deliver socio-economics benefits.
- **Working for Land** aims to ensure that degraded ecosystems are restored to their formal or original state wherein they are able to maintain or support the natural species of that system.
- **Working on Fire** addresses the prevention and control of Wildland fires to enhance the sustainability and protection of life, poverty and the environment through the implementation of Integrated Fire Management (IFM) practices.
- **Working on Waste** is a proactive preventative measure that recognises that inadequate waste services may lead to litter which is not only visual pollution but may lead to health hazards and environmental degradation.
- **Working for the Coast** aims to create access to pristine beaches and a well conserved coastline through regular coastal clean-ups, as well as the removal of invasive alien vegetation.
- NRM value-added activities which aim to create work opportunities and deliver socio-economic benefits through the optimal use of cleared invasive alien plants.

Of the NRM programmes, the largest by far is Working for Water (WfW), and as a result, most of the response in terms of land degradation since the 1990s has been focused on clearing IAPs. Since

[11] There is no indication that this programme assists commercial landowners.

2010 An average of 200 000 “condensed ha”[12] of IAPs have been cleared per year with around 600 000 ha per year receiving follow-up treatment (clearing usually requires three follow-ups; van Wilgen *et al.*, 2020). These programmes involve partnerships with local communities and other government departments such as the DARDLR, and Department and Water and Sanitation (DWS).

The NRM programmes primarily strive to mitigate land degradation through social upliftment via the creation of jobs and livelihood benefits to marginalised South Africans (Magadlela & Mdzeke, 2004; Turpie, Marais & Blignaut, 2008; Giordano, Blignaut & Marais, 2012). The programmes have managed to attract and redirect substantial funding, mainly owing to its social development aspect. Since their establishment in 1995, as Working for Water, over R15 billion has been invested into the NRM programmes. The annual budgets have increased significantly over time, particularly since 2010, reaching almost R2 billion by 2017 (Republic of South Africa, 2018; van Wilgen *et al.*, 2020). This makes the NRM programmes the largest national investment in sustainable environmental management in post-apartheid South Africa (Giordano *et al.*, 2012).

The Natural Resource Management (NRM) directorate proposed a target of clearing 8 069 456 con ha of IAPs between 2016 and 2030, over seven times the national LDN target (see below; Department of Environmental Affairs, 2017b). Giordano *et al.*, (2012) estimated that between 192 000 and 494 000 jobs could be created over 15 years, depending on the extent to which clearing takes place. This could create up to US$15 billion per annum in income for poor rural households.

Despite the benefits of the programme, funding and management constraints have meant that many of the invaded areas have only been partially treated, which is ineffective. Thus, IAPs continue to spread and densify, remaining an enormous threat to many ecosystems (Wilson & Henderson, 2017; van Wilgen & Wilson, 2018; van Wilgen *et al.*, 2020). Currently DFFE spends over R1.5 billion annually on containing alien species invasions (not limited to plants), a figure that has risen in real terms, while other departments at all levels of government also spend substantial amounts. The consensus is that while targets have been achieved through focused efforts in protected areas, the clearing efforts outside of protected areas have only slowed down the spread, and the problem is still increasing (van Wilgen & Wilson, 2018; van Wilgen *et al.*, 2020).

A limiting factor from an environmental perspective has been the focus of the programmes on employment targets rather than on achieving restoration in the most efficient way. In many cases, a lack of attention to the environmental objectives has resulted in weak implementation and monitoring, failure to follow up, and areas reverting to invasion by IAPs. This is a major issue that affects all of the NRM programmes. For example, when presenting the Working for Wetlands programme with an efficient technological solution that would radically speed up the process and reduce the costs of wetland degradation, the programme managers requested that the technology be regressed so that it would take a longer to implement and keep people employed (anonymous but verified source, *pers. comm.*).

[12] IAPs occur at varying densities, so measures are standardized to the equivalent area at 100% density

There are many ways in which the IAP clearing efforts could become more efficient. Based on modelling studies and the experiences of WfW, a fair amount is already known about the logic of how to approach IAP clearing. According to van Wilgen & Wilson (2018) and J. Botha (*pers. comm.*), clearing IAPS that are at medium levels of density/infestation is more cost-effective than attempting to reduce dense infestations which are often beyond a 'point of no return.' In practice, there has been what has been described as an 'identify and direct' approach[13]. In this approach, problem species are identified and certain people are selected to deal with them in a specific way (Lukey & Hall, 2020). Given the low level of success, there is consensus in the scientific community that future policy must be evidence-based, using technical interventions proposed by biological invasion scientists and managers. In addition, consensus on target species and prioritisation is key in forming a dedicated biological invasion policy that is more focused on the environmental outcomes than on the direct social outcomes of the programme. It is possible that the greater number of policies relating to climate change and meeting targets under the global Sustainable Development Goals (SDGs) may provide the perfect 'policy-development window' for the development of formal policy on biological invasion in South Africa (Lukey & Hall 2020).

Estimates by Giordano *et al.* (2012) suggest that enormous financial investment is required to meet both biophysical and socio-economic targets as set out by the NRM programme such as increased employment and national IAP clearing targets under the Working For programmes. Whatever approach is taken, the most successful approach is likely to be one that is integrated, that accounts for fire, IAPs, veld rehabilitation and often overlooked, livestock numbers and movement.

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[13] All historic approaches appear to have followed this, not only since 1994 after which these have potentially been more progressive.

4 CONCEPTUAL AND ANALYTICAL FRAMEWORK

4.1 Defining LDN and LDN targets

Land degradation has become one of the foremost concerns on the global agenda, to the extent that the UN has declared this as the decade of restoration. Goal 15 of the UN's Sustainable Development Goals (SDG) is to "Protect, restore and promote sustainable use of terrestrial ecosystems, sustainably manage forests, combat desertification, and halt and reverse land degradation and halt biodiversity loss", and the associated target 15.3 urges countries to combat desertification, restore degraded land and soil, including land affected by desertification, drought and floods, and strive to achieve a land degradation-neutral world by 2030. To this end, countries are required to report on the proportion of their total land area that is degraded (Indicator 15.3.1) on a regular basis. In addition, under the United Nations Convention to Combat Desertification (UNCCD), countries are required to define their LDN Targets to achieve no net degradation relative to 2015, by 2030.

The concept of LDN was introduced by Lal, Safriel & Boer (2012). This clarified that in the context of LDN, degradation refers to the reduction in productivity of cultivated lands and rangelands. They suggested that achieving LDN (or zero net land degradation – ZNLD) would involve a combination of avoiding degradation, reducing the rate of further degradation of land (ideally to negligible levels) and offsetting new degradation by restoring already-degraded lands (see Figure 4.1, Cowie *et al.*, 2018). Avoidance and reducing the rate of further degradation would be best achieved by the introduction and promotion of SLM practicies, but given that this is not always successful, some restoration will also be required to achieve LDN. Indeed, Lal *et al.* (2012) argue that it is technically feasible to exceed LDN through restoration. Since restoration can be used to offset degradation in different areas, a decision needs to be made as to how to balance these two responses in order to minimuse the costs (or maximise the net benefits) of achieving LDN.

Degradation is a relative concept and requires a baseline. Technically, LDN could be based on measurement of degradation relative to a 2015 baseline, with a view to achieving no change relative to the baseline over time. However, given that LDN will realistically require offsetting of degradation in some areas through restoration in other areas, setting LDN targets requires that countries are able to measure how much land has been degraded at the 2015 baseline (relative to an earlier reference period) as well as subsequently. This also aligns with reporting for SDG Indicator 15.3.1. Ideally, the current level of degradation should be measured against a reference period before any measurable degradation. However, information dating back to those times is usually scarce. Moreover, satellite remote sensing data, which significantly expedite the methods for measuring degradation (Giuliani *et al.*, 2020; Reinermann, Asam & Kuenzer, 2020), only date back to the 1980s, which means that areas that were already degraded by then could be recorded as stable or improving rather than degraded. In South Africa, much land degradation is believed to have occurred before the 1980s (von Maltitz *et al.*, 2019), including in the Thukela catchment. In addition, while the SDG indicators do not require it, we suggest that it will be important to identify the type of degradation, since different forms of degradation require different approaches, and so there needs to be differentiation in the monitoring of treatment effects.

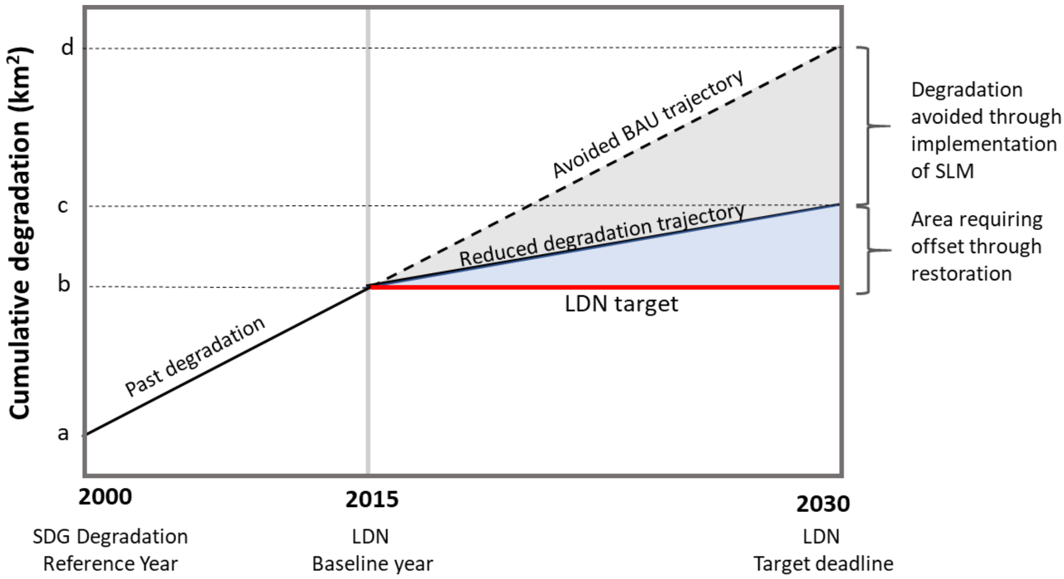

Figure 4.1. Schematic to explain the rationale behind LDN and the setting of restoration targets. Source: Authors.

While quantifying existing degradation helps to identify opportunities for offsetting further potential degradation, this does not address the question of how to identify areas of potential degradation and the interventions required to prevent this as far as possible. The former seems to have dominated the setting of LDN targets. Indeed, the mere notion of target-setting suggests a strong emphasis on restoration rather than prevention. In South Africa, the LDN targets are somewhat vague, stating that LDN "is achieved" in certain biomes (grassland and thicket), and that specified areas of each biome are "rehabilitated and sustainably managed". South Africa's LDN targets are focused on rehabilitating and sustainably managing and controlling IAPs, bush encroachment, wetland degradation and rehabilitating artificial areas, and have been set as area targets at biome level as at 2018 (Department of Environmental Affairs, 2017a, 2017b, 2018; von Maltitz *et al.*, 2019, see section 3.3). Certainly, setting targets for restoration is easier than laying out a plan for the prevention of degradation, but this could lead to an inefficient outcome, since the restoration literature repeatedly asserts that in most cases preventing degradation (without the costs of formal statutory conservation) is more cost effective than restoration (Benayas *et al.*, 2009; Holl & Aide, 2011; Gnacadja & Wiese, 2016; Pandit *et al.*, 2018).

Designing measures to prevent further degradation will require that countries have a good understanding of the drivers, process and rate of degradation in order to devise strategies to reverse and halt it. Potential future degradation can be estimated on the basis of statistical analysis of past trends (e.g. de Souza & de Marco, 2018), although projecting into the future always involves a high level of uncertainty (Alexander *et al.*, 2017; Hamad, Balzter & Kolo, 2018; El-Tantawi *et al.*, 2019). Potentially more challenging is assessing the type and level of investment required to prevent this future degradation (Reed a*et al.*, 2011; McElwee *et al.*, 2020). This point is seemingly lost in the setting of "LDN targets", since targets tend to be expressed as area targets for restoration. In fact, the assumptions about the effectiveness of SLM investments are crucial in determining the appropriate area to restore in order to offset the residual expected degradation. In an optimistic scenario, the degradation trajectory would be reduced to close to zero, requiring very little or no restoration to offset any residual. In a pessimistic scenario, e.g. where SLM measures have a high probability of failure due to institutional reasons, one might expect to offset all the expected degradation with restoration. Ideally, countries should be outlining an "LDN strategy" that outlines the interplay between SLM and restoration, based on their realistic costs of successful intervention. In order to do this, however, there are many practical challenges to be overcome – particularly the quantification of land condition, the determination of the rates of different forms of degradation in different areas, and the costs and impacts of different types of SLM and restoration interventions.

To further complicate matters, many countries, including South Africa, have set their targets after 2015, and will only have embarked on implementation a few years after the LDN baseline year. Increasing delays will lead to an increasing area that must first be restored before being sustainably managed (Figure 4.2). This may be expected to increase the costs of LDN since the cost of restoration is typically higher than the cost of prevention. However, delayed action may be attractive, depending on the rate of time preference. Thus, countries also have to optimise the timing of action, taking the dynamics of the situation into account.

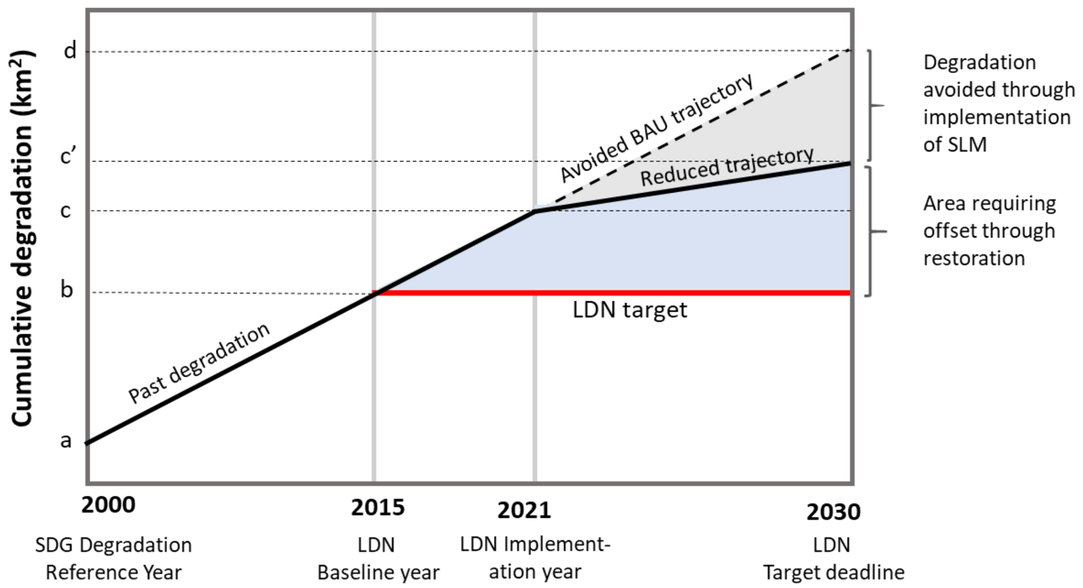

Figure 4.2. Schematic showing the consequences of delaying taking action towards achieving LDN. Increasing delay leads to a higher restoration requirement, which usually raises costs.

In restoration interventions, the problem is usually conceptualised as maximising the returns to effort through directing effort to the areas where restoration efforts yield the greatest returns, up to the point where the marginal gains from restoration are equal to the marginal costs or within the limits of a budget constraint. For LDN, the objective would be to minimize the cost of achieving no net loss over a fixed period of time.

4.2 Measuring degradation

The UN has developed guidelines for the measurement of degradation (Indicator 15.3.1). The degradation indicator is a binary indicator (degraded vs not-degraded) derived from three sub-indicators which are based on changes in land cover, land productivity dynamics, and SOC, respectively. If any one of these sub-indicators is negative or declining for a land unit, or stable relative to an earlier negative trend, then the land unit would be considered degraded, following the “one out, all out” (OOAO) principle. The methodology for computing these indicators, detailed in the UN metadata for this index, has between very thoroughly developed and was endorsed by the UNCCD’s governing body in 2017. Probably as a result of data considerations, the year 2000 has been suggested as the baseline for the identifying degraded areas as at 2015. Unless they have better datasets of their own, countries are able to make use of recommended global datasets on landcover, net primary production and SOC, as well as purpose-built online tools such as Trends.Earth. Trends.Earth uses several data sources, including LULC and NDVI derived from satellite data, soil moisture and precipitation to generate the 2015 baseline (degradation relative to 2000), as well as allowing up-to-date comparison with the baseline, using statistical approaches to control for rainfall variation. The output is a raster grid indicating positive, stable or negative change.

One might expect that the three sub-indicators could generally be substitutes, and hence in combination help to obtain a robust indication of degradation. For example, degradation that is detectable as a change in land cover (e.g. deforestation) could also lead to a reduced LDP and SOC. However, in South Africa, some major forms of degradation (IAP and indigenous woody plant densification and spread) result in increased land productivity dynamics (Venter *et al.*, 2020), and some forms of degradation would not be expected to be reflected in terms of a change in net primary production (NPP, e.g. invasion of thicket by alien cactus). Therefore, for natural terrestrial areas (rangelands and protected areas), NVI trends alone cannot reliably be used to indicate degradation, and the change in land cover would be the key sub-indicator for these types of degradation in the OOAO index. Furthermore, a decrease in NDVI can indicate an area where restoration has taken place, so an OOAO approach could provide a false indication of degradation. Thus, in South Africa, a rule-based approach using ground or aerial data as well as satellite data would provide a more reliable estimation of areas and types of degradation (Figure 4.3). In this model, land cover can be used to identify areas of signifcant degradation that has resulted in major structural vegetation changes. Changes in NDVI can then be used to suppement this by identifying areas where changes in land productive dynamics have occured within a land cover class (i.e. smaller changes, such as reduced grassland productivity).

In this study, we focused on the larger changes detectable as changes in land cover in areas under natural vegetation.

In this study, we estimated the area of degradation, based on land cover and NDVI changes from 2005 to 2017. The period 2005 to 2017 was used instead of 2000 to 2015, since these aligned with the existing, ground-truthed KZN LC datasets. The 2017 LC was considered to be a reasonable approximation of the 2015 baseline required for LDN. Our main deviation from the UN method was that all areas infested by IAPs were counted as degraded, not just the area that has become infested since 2000 or 2005. The UN base year of 2000 is sensibly based on the practical difficulties of obtaining earlier monitoring data on many of the elements of the degradation indicators. While it is not difficult to estimate the extent of IAPs at a base year such as 2000, it does not make sense to ignore areas infested by this time as not degraded. A similar argument may be made for rangeland areas that have been identified as degraded through botanical studies, should this information be available.

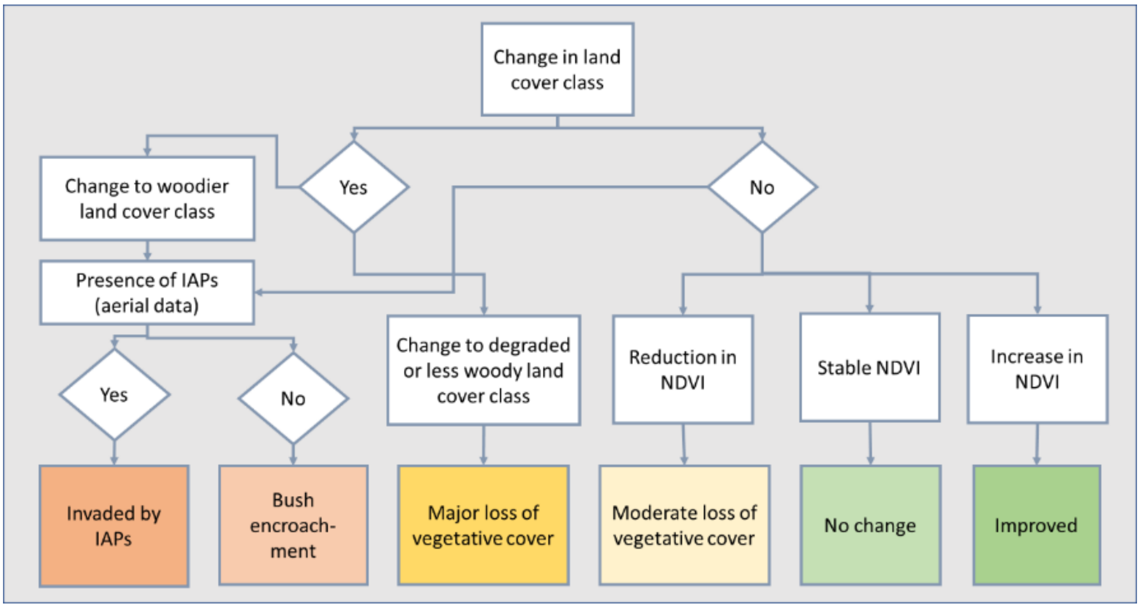

Figure 4.3. Rule-based model for assessing degradation of natural areas and rangelands outside of protected areas in South Africa for the 2015 Baseline relative to 2000.

4.3 Scenario approach

Scenario analysis is useful for interrogation of potential alternative policy options, particularly where such policies are actively under development, and where policy makers would benefit from the additional information. Such an analysis needs clear distinctions between different options, including a business as usual option. In order to develop the business as usual scenario, forecasting is typically based on past trends.

Scenarios can be conceived as static or dynamic. Static scenarios have a clear objective for a future year, and the analyst identifies what needs to be done to get there. This can be analysed as an optimisation problem (using a model to find the optimal way to achieve the objective) or more simply using estimates based on the best available information. Dynamic scenarios involve implementing a policy at the start and then using modelling with feedback effects to arrive at some point in the future. This usually involves some kind of simulation modelling, such as agent-based models or causal descriptive models. In this study, we used a static scenario approach, which was aligned with alternative objectives in terms of the degree to which degradation is addressed. The scenarios were compared using a cost-benefit analysis framework.

The key questions of the study were addressed through an analysis of alternative intervention scenarios for the period from 2021 to 2030. This corresponds to the remaining period for South Africa's LDN commitment, and also coincides with the UN's Decade of Restoration. We worked from the land cover data set for 2017, the closest available to the Reference year of 2015 for South Africa's LDN commitments.

We determined the level of degradation as at 2015, and estimated the expected level of degradation by 2030 under a **Business-as-Usual (BAU) Scenario**. This scenario had low levels of intervention and continued land degradation though spread of IAPs, bush encroachment and loss of vegetative cover and erosion. Next, we estimated what it would cost to achieve LDN by 2030. Assuming a start of action in 2021, this required restoration of the degradation that had already occurred from 2015-2021, followed by the implementation of SLM measures. For this **LDN Scenario**, we used a lower and upper bound estimate of costs, based on assumptions about the efficiency of SLM in halting future degradation. Under the lower bound (optimistic) cost estimate, it was estimated that SLM measures would be highly effective. Under the upper bound (pessimistic) cost estimate, it was assumed that SLM would be ineffective, necessitating restoration of an area equivalent to all projected degradation from 2015-2030. The reality was expected to lie between these two estimates. Finally, we estimated the costs and benefits of a **Full Restoration Scenario** in which interventions are implemented from 2021-2030 that restore all degraded natural areas (as at 2021) to a healthy condition. This assumed that SLM measures would stem further degradation.

The implications of the different scenarios were assessed at a high level, based on estimated costs of the interventions, and the avoided losses of ecosystem services, taking time into account. Changes in the value of ecosystem services were based on changes in ecosystem extent and quality, by building on the baseline data, methods and tools which were developed as part of the pilot accounts for KwaZulu-Natal.

4.4 Natural capital assessment approach

Most studies on land degradation in South Africa have focussed on a narrow dimension of the problem. For example, the majority of studies of IAPs have focussed on their impacts on stream flows, while studies on bush encroachment have tended to consider impacts on rangeland carrying capacity (Le Maitre, Versfeld & Chapman, 2000; Dzikiti *et al.*, 2016; Preston *et al.*, 2018). Given that the methods to address these problems tend to be costly, this can limit policy responses in areas where the more well-known problems are not as severe.

Land degradation in South Africa involves fundamental changes in the nature of ecosystems, changing both their extent and condition. This affects the delivery of a whole array of ecosystem services. The analysis of options to address land degradation should therefore take a more holistic approach. In this study, a range of ecosystem services affected by the policy options were included in the analysis, namely: ecosystem contribution to baseflows, sediment retention, carbon storage, inputs to livestock production, supply of harvested resources, and tourism value.

Natural capital accounting therefore provides an excellent starting point for assessing the implications of different policy options. In this study, we built on the work carried out in compiling the pilot Ecosystem Service and Asset Value Accounts for KwaZulu-Natal (2005-2011). However, the focus of this study on meeting the UN's LDN requirements necessitated updating parts of this work to a more recent land cover dataset, and the refinement of the classification of ecosystem condition. Thus, in addition to the policy analysis, this study also provides further input to the development of ecosystem accounts in South Africa.

The study was carried out at the same spatial resolution as the pilot accounts, based on a basic spatial unit (BSU) of 1 ha (100 x 100, Eigenraam, 2019; Statistics South Africa, 2019). The baseline landcover was based on the latest (2017) output of the KwaZulu-Natal land cover data series since this was closest to the 2015 Reference year for South Africa's LDN targets.

4.5 Cost-benefit analysis framework

The scenarios were evaluated using a cost-benefit analysis framework. Cost-benefit analysis involves the comparison of alternatives in terms of their expected costs and benefits over a defined period of time. It involves the estimation of future values in present value terms through the process of discounting at a rate which reflects the rate of time preference. This places greater weight on values occurring closer to the present, which means that the future benefits of restoration projects will be down-weighted compared with the upfront investment costs, and have to be substantial in order for a project to be viewed positively. The sum of the discounted costs and benefits reduces the value of an alternative to a single measure - the net present value (NPV). For a project to be considered viable, the net present value (NPV) must be positive.

Cost-benefit analysis can be applied as a purely financial analysis (e.g. by a private investor) or as an economic analysis to guide policy and decision making. In the latter case, applicable here, the costs and benefits to society as a whole should be taken into account, including consumer surplus and the intangible or non-use benefits associated with biodiversity, which is typically quantified in terms of society's willingness to pay. A relatively low rate of discounting is used, known as the social rate of discount, which takes a longer term view than a financial analysis. For this analysis we used the social rate of discount of 3.66% taken from Addicott *et al.*, 2020) over a time period of 25 years (2021-2045).

In practice, consumer surplus associated with nature-based activities or the willingness to pay for biodiversity is difficult to estimate, and are often omitted. This is also the case in this study, which was based on the detailed valuation of ecosystems undertaken for the pilot natural capital accounts for KwaZulu-Natal. The latter did not include the estimation of consumer surplus or non-use values, since such values are not included in accounting. Ideally, these values need to be included when moving from the accounts to policy applications in order to have a more complete estimate of the welfare implications.

Due to all the data limitations and uncertainties around estimates of costs and benefits, cost-benefit analysis usually incorporates some form of sensitivity analysis. In this study we included a worst-case scenario (using upper bound estimates of costs and lower bound estimates of benefits) and a best-case scenario (using lower bound estimates of costs and upper bound estimates of benefits)[14]. We also tested the results using alternative discount rates.

In addition to comparing the net present value of the alternative scenarios, we also computed the benefit-cost ratio (BCR) or return on investment (ROI) of the LDN versus the Restored Scenario. This was based the difference in benefits of these scenarios from the BAU scenario divided by the costs of restoration interventions in each case.

[14] If a range of costs was available from the literature then these were used to generate best-case and worst-case estimates, e.g. for SLM were extracted from UNCCD, 2015, but if data were not available then a general 25% change in costs was used to generate a lower and upper bound. For ecosystem benefits a range was estimated based on assumptions around, for example, stocks of resources under different levels of woody encroachment or for tourism, trends in visitor numbers to South Africa and the province of KwaZulu-Natal which provided the upper and lower bound estimates.

5 METHODS

5.1 Estimation of land cover under different scenarios

5.1.1 Overview and general limitations

The KwaZulu-Natal land cover data series has information on condition relating to the loss of vegetative cover (albeit with limitations), but does not include information on IAPs or bush encroachment, as IAP and bush encroachment involve woody plants that are integral to the classification of natural land cover types. Based on information in the literature we modelled the increase of IAPs from a 2010 baseline dataset, and integrated the modelled 2017 extent into the 2017 LC, creating new land cover classes in the process. We also estimated the extent of recent bush encroachment based on a comparison with the earliest (2005) KwaZulu-Natal LC data. The latter information did not alter the baseline land cover but was used in estimating the restoration, LDN and BAU scenarios. For the restoration scenario, areas affected by IAPs, recent bush encroachment, degraded or eroded were reverted to their estimated former land cover class. For the BAU scenario we used projected IAP extent using the same model and the analysis of 2005-2017 rates of land cover change to project bush encroachment, loss of vegetative cover and erosion as at 2030. For the LDN scenario, we used modelled estimates of the extent of degradation as at 2015 for estimating costs and benefits as far as possible, but the differences did not warrant creation of a new land cover for the hydrological modelling. Urban and cultivation extent were held constant under all the scenarios.

The key forms of land degradation that needed to be considered in estimating land cover changes were bush encroachment, alien invasive plant spread, loss of vegetative cover and erosion. Land degradation is well studied in South Africa but few spatial products exist which have captured the quantified extent of degradation, often due to the difficulty of obtaining true indications of degradation and condition from satellite imagery products (Turpie, Forsythe & Thompson, 2020a), but also due to the demanding nature of obtaining ground-truthed data over large areas (Turpie *et al.*, 2020a). There are also issues relating to how well many products or tools (i.e. Trends.Earth) capture true condition of and decline in natural ecosystems and their integrity. Many spatial products include shapefiles at district or catchment scale with only a percentage or single value of degradation measure, which are unsuitable for fine-scale analysis.

While the best available data and tools were used for this study, working at the scale of the Thukela catchment meant that it was not possible to carry out any further ground truthing than has been carried out in the development of the EKZNW landcover datasets. At a smaller scale, such as a quaternary catchment, it is more feasible to obtain better estimates of LULC features and map them at a high resolution (i.e. Pringle *et al.*, 2015). This would still require local expert knowledge and ground-truthing for verification purposes, which was beyond the scope of this study. Additionally, the large scale of the study area prevented accurate incorporation of ecosystem services from wetlands. These are comprehensively mapped in South Africa at a fine scale, often at 1:50 000 or less. Many of the wetlands would not be large enough to be integrated into land cover or the SWAT hydrological modelling approach.

The complexity of the problem, inaccuracy of the spatial data and resource limitations of the study required a fairly simple narrative-based modelling and GIS integration approach (see Swetnam *et al.* 2011) to modelling changes in land cover for the BAU scenario. Scenario analysis involving changes in land cover are increasingly using sophisticated tools such as Markov Chain Analysis (MCA), Cellular Automata (CA), Cellular Automata-Markov Model (CA-Markov), Boosted Tree Regression (BTR) and Binary Logistic Regression (BLR). Several software products exist to make this easier, the most commonly used being the TerrSet Geospatial Monitoring and Modelling suite (TerrSet, formerly IDRISI[1], Clark Labs, 2020),[15] which includes a Land Change Modeller (LCM). Future work should explore these options.

5.1.2 The KwaZulu-Natal land cover datasets

EKZNW has produced land cover datasets for 2005, 2008, 2011 and 2017. The satellite imagery used to create LULC maps improves with technological advances over time, which have been particularly rapid in the last decade. As such, the 2017 LC is a vastly improved product compared to the previous issues. The 2017 KwaZulu-Natal land cover dataset (hereafter 2017 LC; GeoTerra Image, 2018) is the first provincial land cover product that has been derived from true multi-seasonal imagery, incorporating all seasonal dynamics for natural vegetation and waterbodies and correcting for these (GeoTerra Image, 2018). Most natural vegetation classes were not influenced by the 2011 classification and were derived purely from 2017 Sentinel 2 imagery.

The original intention was to build the scenarios from a 2011 baseline (corresponding with the pilot accounts). However, after the release of the 2017 LC, it was possible to develop a more accurate estimate of the 2015 Reference land cover for the LDN target.

The 2017 LC has 48 LULC classes, of which all but one are present in the Thukela catchment (Appendix 1). The dataset has a nominal spatial resolution of 20 m and was mapped from Sentinel-2 imagery. Imagery was derived from 1 January 2017 to 31 December 2017, using cloud-based data processing capabilities and desktop digital classification procedures. Users accuracy is very high, mostly above 95%, with the lowest being for clear-felled plantation forests (GeoTerra Image, 2018), which transition between clear-felled, regrowth and mature over a short time period. The only additional class compared to the 2011 LC was that of wetlands which was split into 'wetlands' and 'wetland – drainage / riparian'. The original dataset was re-projected from WGS84 Universal Transverse Mercator (UTM) 36S to the South Africa Basic Spatial Unit Albers 25E projection (see Eigenraam, 2019; Statistics South Africa, 2019), keeping the same resolution. This is the same projection as was used in the pilot accounts and for the forthcoming National Terrestrial Land and Ecosystem Accounts (Statistics South Africa, 2020).

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[15] This appears the case *anecdotally*, based on a concise literature search using Google Scholar.

The KwaZulu-Natal land cover data series includes limited information on degradation in that it includes degraded grassland, bushland and forest, as well as erosion dongas. However, it does not include information on woody IAPs, whose presence is integral to the classification of natural land cover types but cannot be discerned from other woody vegetation via satellite imagery. Furthermore, identifying the total degraded area at any point requires analysis of changes in land cover in order to detect the areas where woody cover has increased (bush encroachment), and areas where woody cover has decreased (e.g. due to firewood harvesting or burning practices), other than where identified in the existing "degraded" land cover classes. These trends from the land cover data series, in combination with estimates of IAP densities, were used to produce the Baseline 2017 land cover and the future (2030) land covers under the different scenarios.

5.1.3 Estimating the extent of the existing degraded land cover types

The degraded forest, degraded bushland, degraded grassland LC classes, which have been ground-truthed, are largely based on the presence of bare patches (M. Thompson, GeoTerra Image, *pers. comm.*) and indicate areas of lower vegetation cover than expected. Nevertheless, it should be noted that the 2017 LC dataset showed a reduction in degradation in nearly all the tertiary catchments of the study area relative to the 2011 LC. This was attributed to improved mapping and the ability to discern these areas from the multi-seasonal imagery used, and the 2017 values were deemed to be more accurate (GeoTerra Image, 2018; Jewitt, Thompson & Moyo, 2019).[16] This created a challenge for future projections, since it may be incorrect to interpret this as an improvement in land condition. In fact, the degraded grassland class in particular is thought to be an underestimate of degradation, since overgrazing can result in a change to less palatable species but still have a level of vegetative cover that does not affect classification, and this remained a limitation for this study. The existing degradation classes were retained for the 2017 Baseline LC, but were augmented by information on which areas had changed from woodier to less woody classes (see below).

For the BAU land cover, we projected the extent of degraded forest, degraded bushland, degraded grassland and erosion based on the rates of change in their extent from 2005 to 2017.[17] The total area of each of these classes in each tertiary catchment was ascertained for 2005, 2008, 2011 and 2017. These were then analysed to determine the best model fit. In the case of degraded grassland, there was a consistent downward trend which was best fit by a logarithmic function. The area under erosion had a very gradually increasing trend, which was also best described by a logarithmic function. These functions were used to determine the rate

[16] Degraded areas for both grassland and bushland were both created by "intersecting a generic mask generated on the basis of a pre-determined 2017 maximum annual NDVI value threshold with the 2017 mapped grassland and woody-cover vegetation class extents " (GeoTerra Image, 2018, 11). These areas represent observed areas with substantially lower vegetation cover than the surrounding areas.

[17] We did not consider old fields (previously grassland or bushland), old plantations and rehabilitated mines in the analysis.

of expansion of the existing areas in GIS[18], which resulted in marginal increases in erosion area in each tertiary catchment from 2017 extents and marginal increases in degraded grassland area in eight of the 12 tertiary catchments. There was no consistent trend for degraded forest or degraded bushland, however. There was a substantial increase in area in both land cover types from 2005 to 2011, followed be major decline to 2017 (Figure 5.1).

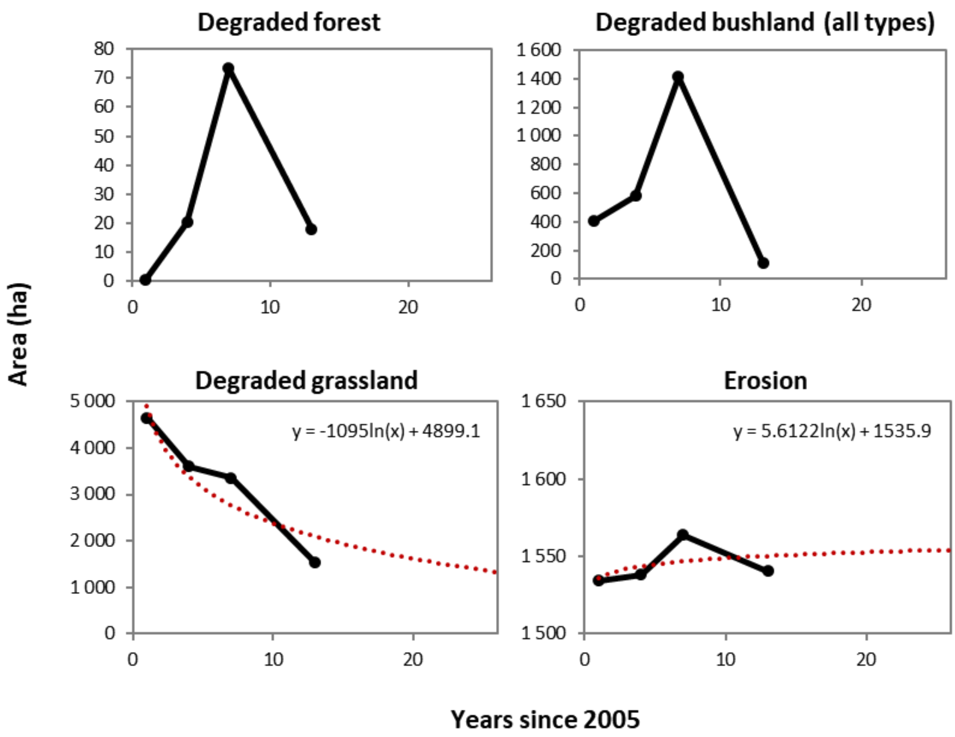

Figure 5.1. Example (tertiary catchment V13) of the different trends in extent of four degraded land cover classes from 2005 to 2017.

This is likely to be primarily due to improved mapping capabilities used in the compilation of the 2017 LC as well the use of multi-seasonal imagery. Both land cover classes were mapped with significantly more accuracy in the 2017 LC than in the 2011 LC. The degraded bushland had a user accuracy nearly 70% higher in the 2017 LC (GeoTerra Image, 2018). Thus, the areas of these two land cover classes were not changed from the Baseline for the BAU 2030 land cover. The erosion LC class had a 98.75% user's accuracy in the 2017 LC edition[19] and thus it

18 The areas were vectorised and then adjusted using the Buffer by Percentage tool in QGIS by the appropriate factors

19 The User's Accuracy is the accuracy from the point of view of a map user, rather than the producer of the map. It essentially explains how often the class depicted on the map is actually present on the ground.

was not deemed necessary to supplement with additional datasets (such as the DAFF 2011 gully erosion shapefile). Erosion generally showed a slight increase year-on-year from previous datasets. For these LC classes, it was assumed that the 2017 LC provided a reasonable approximation of land cover as at 2015 (the Reference year for LDN).

5.1.4 Estimating changes in woody cover

It was assumed that the changes to woodier classes were indicative of degradation due to bush encroachment and/or IAP spread, whereas the opposite trends were indicative of degradation due to resource use or excessive burning. In reality, some of the latter could be recovery due to better management or active restoration such as bush clearing or IAP clearing, but for the purpose of this study this was assumed to be negligible. Some proportion of the changes in both cases may have been artefacts of the data, particularly in more complex parts of the landscape, but this was also assumed to be a small proportion.

In order to estimate the extent of these types of degradation in the Baseline 2017 land cover, changes in woody cover were ascertained for the period 2005 to 2017, by overlaying the oldest and latest KwaZulu-Natal land cover datasets. Any area that had changed from a land cover class with lower woody cover to one with higher woody cover was identified as an area of increased woody cover, or vice versa. This was based on the ordering of land cover classes shown in Figure 5.2. For the areas that had increased in cover, these were separated into bush encroachment and IAP spread based on the IAP modelling (see below).

The rates of change in these LC classes were used to estimate the BAU 2030 land cover. From the resulting matrix between the six main natural vegetation classes common to both the 2005[20] and 2017 LC (see results), we estimated the rates at which different land cover classes were changing into other land cover classes. The average annual change was calculated for each of the 30 possible pathways[21] in each tertiary catchment and converted to an expected proportional change over the period 2017 to 2030. Although, from a longer term perspective, it is likely that both bush encroachment and desertification are in an exponential phase (e.g. see O'Connor, Puttick & Hoffman 2014), for the purpose of this study, we conservatively assumed that the rate of change over the next decade would be similar to the last decade. We did however assume that there would be some change in density in certain areas, which cannot be predicted from a trend with two points in time. To ensure a more accurate value of total Dense thicket & bush (densest natural vegetation class other than indigenous forest), the areas that changed from grassland to medium bush and grassland / bush clumps mix to medium bush were reclassified as Dense thicket & bush for the 2030 BAU land cover. This ensured a more realistic

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[20] Wetland – drainage / riparian as a class was not present in the 2005 LC classification scheme and forest glade and alpine heath-grass were excluded, the former due to its tiny extent in the catchment and the latter due to its location, primarily along the border with Lesotho in the uKhahlamba Drakensberg Park where it is well protected.

[21] Seven pathways accounted for 84% of the changes.

densification of woody plants based on the previous 12 years' change. Based on these, the areas that had experienced woody encroachment and woody vegetation loss since 2005 would be expected to approximately double from 2017 to 2030.

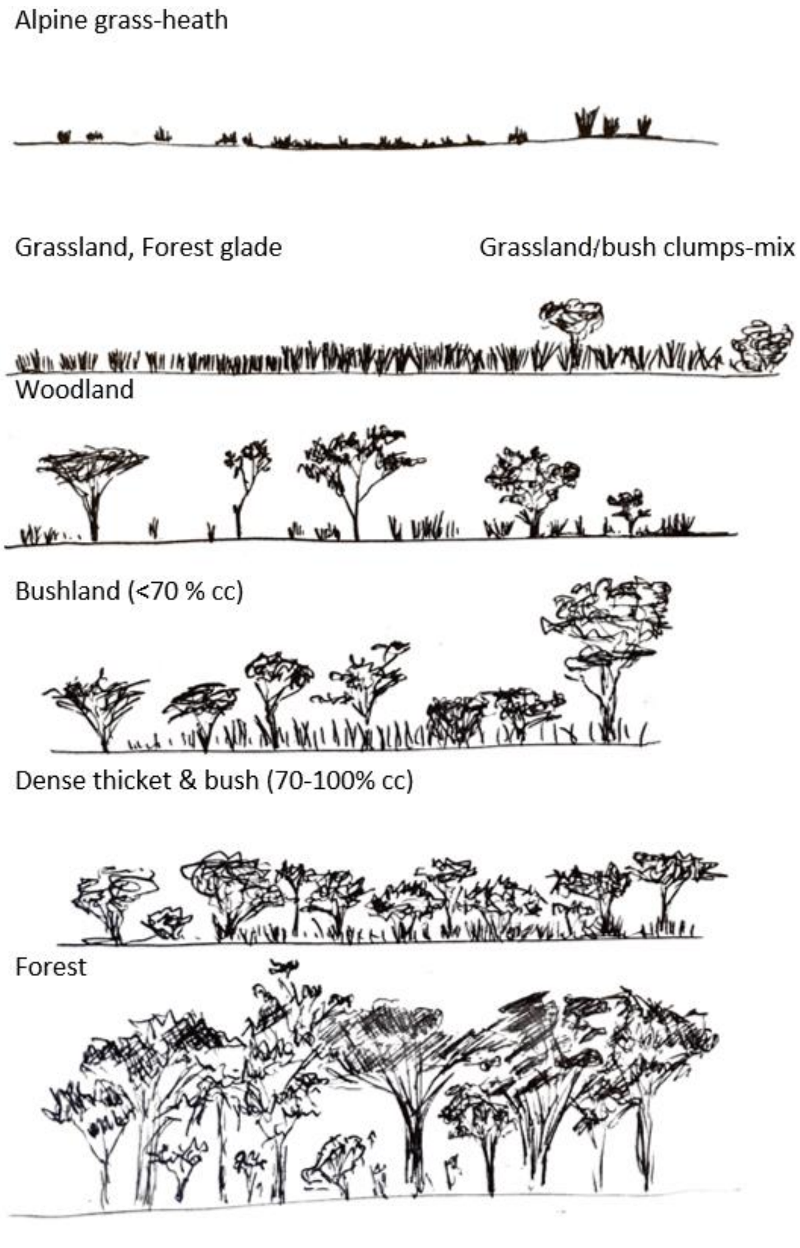

Figure 5.2. Schematic diagrams depicting the general structure of the natural land cover classes (hand drawn in black) as per the EKZNW land cover classification.

To model the changes spatially for creation of the 2030 BAU, the 2017 land cover data were converted from a raster to a multi-polygon shapefile. The areas that had experienced a change in woody cover were then expanded into adjacent natural areas based on the expected proportional change calculated above, using the "Buffer by Percentage" plugin in QGIS (Figure 5.3). In order to deal with the resulting areas of overlap, the pathways were prioritised in descending order of extent. Using this method meant that the effective rate of change was curbed by spatial limitations in addition to being limited to natural vegetation areas (i.e. not encroaching into cultivated areas, for example). The resulting output was reconverted back into a raster and integrated into the 2017 LC.

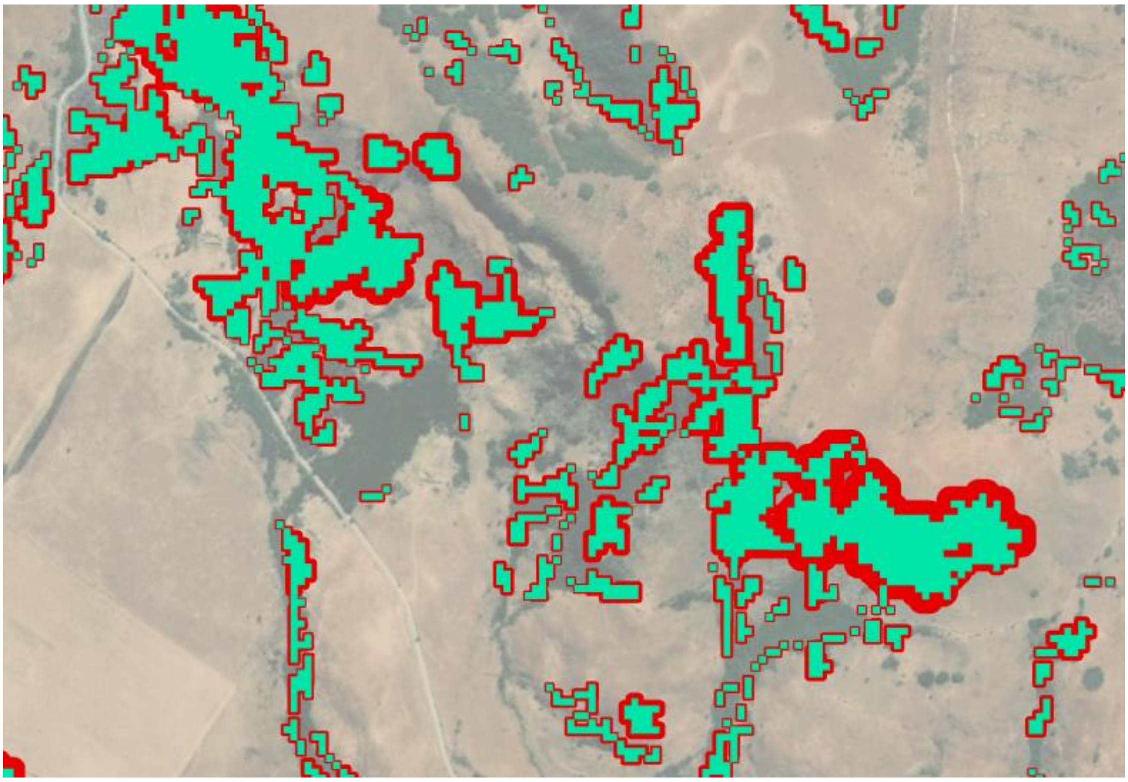

Figure 5.3. Example of areas with a specific change in woody cover from 2005 to 2017 shown in green and the expected further extent of this change by 2030 (under BAU) shown in red, using a linear projection.

After accounting for the estimated density of IAPs (see below), the comparison of the 2030 estimated LC with the 2017 Baseline LC provided estimated areas of vegetation where woody cover had increased by more than 40 percentage points. This was used in determining the type of action required for the treatment of bush encroachment (O'Connor *et al.*, 2014).

5.1.5 Estimating the extent of IAPs

We based our estimates of IAP densities on the most recent and comprehensive available spatial data for the study area, the NIAPS dataset based on low level aerial surveys and statistical interpolation (Kotzé *et al.*, 2010). The NIAPS dataset provided information on the average density of each IAP species at a 250 m^2 resolution as at 2010. While the NIAPS database contains information on many IAPs, we focussed only on the three most prolific woody species in the catchment – wattles *Acacia spp.*, gums *Eucalyptus spp.* and pines *Pinus spp*. The limited ability to integrate the remaining species into a land cover class was too low for the spatial scale of the analysis and the three species chosen constituted a large proportion (82%) of the total condensed area of IAPs in the catchment.

Because the NIAPS layer was published in 2010, the IAP densities first needed to be updated to 2015 levels for the LDN Reference Year, 2017 for the land cover baseline, 2021 for our modelled interventions and 2030 for the BAU scenario. There is substantial literature detailing work aimed at determining different rates of spread of IAPs. Different species proliferate at varying rates in different environments, and observed rates vary considerably within a species (van Wilgen & Le Maitre, 2013). No reliable methods exist for assessing invasion trends in South Africa (Van Wilgen & Wilson 2018). Factors such as growth requirements, vulnerability to fire, extent of overgrazing, drought and degree of natural vegetation degradation all play a major role in the rate of spread, often acting in concert to facilitate spread of IAPs (Le Maitre *et al.*, 1996, 2000; Enright, 2000; van Wilgen, 2010; Preston *et al.*, 2018). Their review suggested that rates ranged from 7.4-15.6%, with the exception of *Acacia spp.* in riparian environments. Other studies in South Africa show rates of as little as 1.8% per annum to up to 10%, with 3-8% being considered a typical range (Table 1, Moeller, 2010; Rebelo *et al.*, 2013; van Wilgen *et al.*, 2016). Scale of study is also an important factor in determining the drivers and extent of spread, which differs markedly from a few hectares (i.e. a small study site) to quaternary catchments or biomes (Rouget & Richardson, 2003; Pyšek & Hulme, 2005). While van Wilgen & Le Maitre (2013) developed a useful model for determining spread, one key variable was the number of years after introduction, which was unknown in this study. Using similar density categories, Versfeld, Le Maitre & Chapman (1998b) determined the annual percentage change for different species. However, this would not work for different density classes as was the case in the study.

The most recent publication on the status of biological invasions in South Africa reaffirms that the densification and spread of invasion generally follows a typical sigmoidal population growth curve, where the rate of expansion increases at first and then slows down as competition for space increases (van Wilgen & Le Maitre, 2013; van Wilgen & Wilson, 2018). The rapid expansion phase usually lasts several decades and even beyond a century (van Wilgen & Wilson, 2018). Invasion involves densification of existing areas of invaded land as well as spread to new areas (Preston *et al.*, 2018). In this study, most areas already had at least low levels of infestation, and we only modelled densification.

For this study, we applied a simple densification model to each pixel with a maximum growth rate (*r*) of 7.5% which was reached at a density of 37.5%. This was given by the equation $y = -5 \times 10^5 x^2 + 0.004x - 0.0033$, where x is the density in the previous year. While this did not allow for spread into areas with zero IAPs in 2010, this was not considered to be a major limitation within the scope of the study. This was then used to determine the rate of densification from the known starting density in 2010. This equation was used to estimate the density (and con ha) for each of the three woody IAP species for each raster value (arranged as species per quaternary catchment) over the period from 2010 to 2030.

Table 5.1. Different rates of spread of invasive alien plants in South Africa in the literature surveyed. The value of the spread per annum percentage or single mean is either a mean calculated, or a single figure provided in by the authors.

Species	Spread per annum (%) or range	Spread per annum (%) mean or single value	Source
A. dealbata, A. decurrens	10%		Versfeld *et al.* (1998)
A. mearnsii	5-10%	8.13%	
A. melanoxylon	5%		
Eucalyptus spp., Pinus patula	5-10%	7.5%	
General	4-7%	4.7%	Enright (2000)
Pinus spp.	3-8%	5.5%	Higgins, Richardson & Cowling (2000)
General	8.5-17%		Cullis *et al.* (2007)
Pinus spp.	-	3.75%	Moeller (2010)
Acacia spp.	2.1%, 4.8%, 1.8%	2.6%	Rebelo *et al.* (2013)
General	-	5%	Working for Water
Pinus spp.	-	15.6%	van Wilgen & Le Maitre (2013)
Acacia spp.	-	10%	
General	7.4-15.6%	11.5%	
General	5-10%	7.5%	van Wilgen & Wilson (2018)

*Not a figure or spread *per se*, but rather an assumed rate of spread *at which IAPs could be contained* at a relatively low cost. This was later found to be too low in reality (van Wilgen *et al.*, 2012).

5.1.6 Integrating IAPs into the Baseline (2017) and BAU (2030) LC

The IAP modelling outputs were used to update the NIAPS raster with estimates for 2017 and 2030. While the NIAPS data is the most complete and detailed data at a large-scale for South Africa, the 250 m^2 resolution was substantially coarser than the 20 m^2 resolution of the KwaZulu-Natal LC, which made it challenging to integrate (Figure 5.4). The average density across a 250 m^2 pixel could come from an even or a clumped distribution. Furthermore, the density is often an average over a larger area involving several pixels again making integration difficult. This meant that decisions had to be made as to how to allocate the condensed area in a 250 m^2 pixel into the 20 m^2 pixels, which would inevitably alter the total invaded area in our results. Without this integration we would struggle to follow the principles of NCA of using *spatially explicit* data.

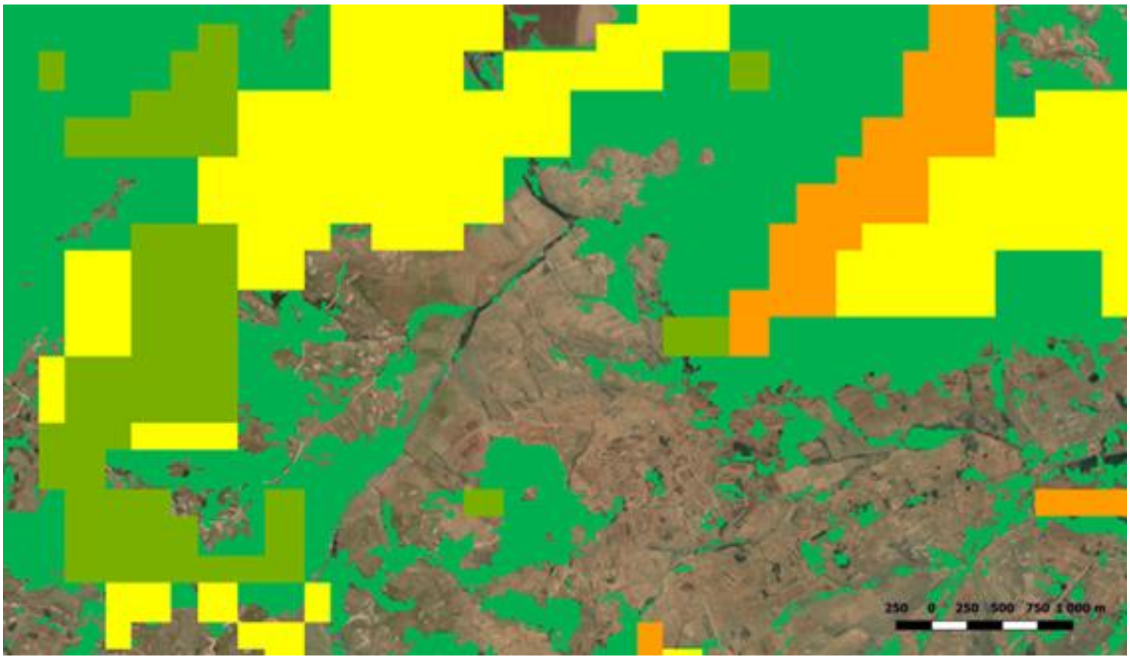

Figure 5.4. Example of original NIAPS 250 m² data (olive, yellow and orange) and natural vegetation areas (green) in the 2017 LC 20 m² resolution, highlighting the disparity in resolution. (Basemap source: Esri World Imagery).

The data for the target species were split into separate rasters and resampled to 20 m² using a bilinear resampling method (Figure 5.5). This technique calculates the value of each pixel by averaging (weighted for distance) the values of the surrounding 4 pixels. It is suitable for continuous data (which the NIAPS data are) and avoids creating values outside the range when increasing the resolution of the data. It was then snapped to the 2017 LC, to ensure accuracies in later calculations and all cell values of 0 (no presence) were removed. IAP values were only assigned to natural or degraded natural vegetation cells in the 2017 LC, i.e. avoiding cultivated land or anthropogenic LULC, much like the creators of NIAPS did through their methods but not reflected in the final spatial data due to the resolution.

The canopy cover densities of IAPs in the Thukela Catchment ranged from 0% to 87%. To create new land cover classes, densities were categorised as follows (based on Versfeld, Le Maitre & Chapman 1998; Marais, Wilgen & Stevens 2004):

- 0–4.99% = Negligible invasion
- 5–24.99% = Light invasion
- 25–49.99% = Medium invasion
- ≥ 50% = Heavy invasion

Owing to the above challenges of integration, areas of less than 5% density, were treated as uninvaded which reduced the overall invaded area in our dataset compared to published data on IAPs in the Thukela catchment. To account for this difference, we assumed that the residual

IAPs predicted in the NIAPS dataset that were not integrated into our baseline land cover were located along drainage lines.

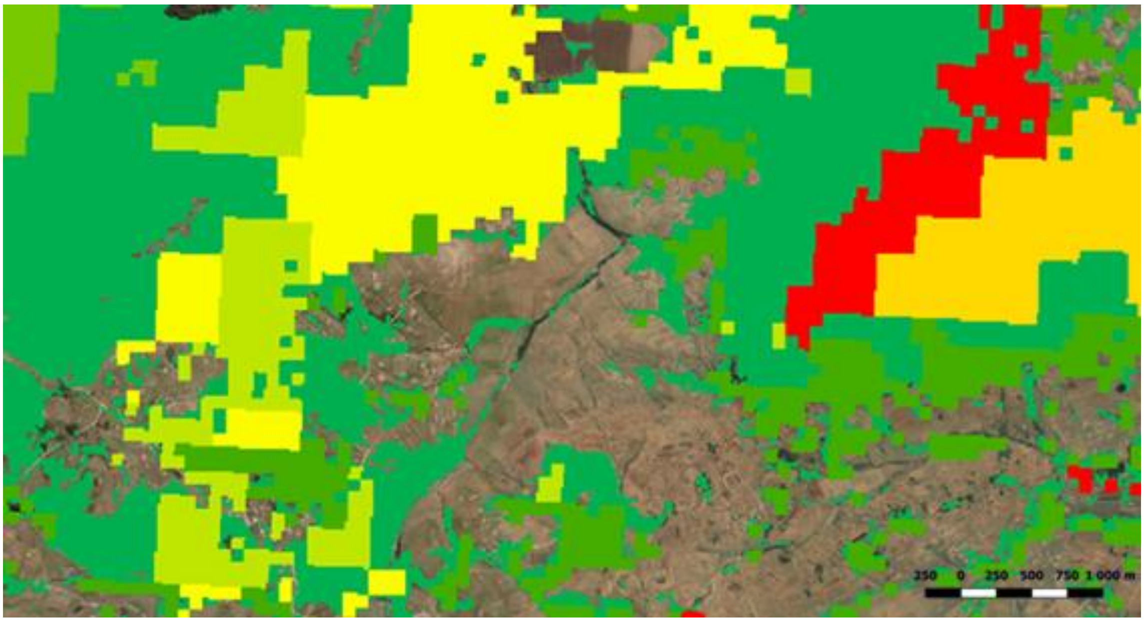

Figure 5.5. Example of the resampled and snapped NIAPS data with 20 m^2 resolution. (Basemap source: Esri World Imagery).

The three key species were split by the density classes. In many areas, there was an overlap of different species. To reduce unnecessary complexity, the information on the dominant species was used. For example, if a cell had medium wattle density and light gum density, it was classified as medium wattle. However, where two species were in the same density class, these were classified as a mixture (i.e. medium gum and pine invasion).

The next step was to integrate the IAP information into the 2017 LC to create the Baseline LC (2017). Integrated LC classes were created based on the natural KwaZulu-Natal LC classes and the density classes of IAPs (see rules applied in Appendix 2: Rules for developing the integrated classes in the land cover datasets). If LC classes that contained woody vegetation overlapped with areas of IAPs, then based on the density class of the IAPs, the original vegetation class could be estimated. For example, if an area classified as "grassland with bush clumps" turned out to be an area with moderate IAP infestation, then it might be reclassified as "grassland with medium IAP infestation". In this case restoration would hypothetically reinstate its status as a grassland. Forests in the study area are not typically invaded by the species considered, so unless the infestation was dense, the overlay of IAPs on forest areas were taken to be an artefact of the course scale of the IAP data, except for dense infestations, in which case the forest was assumed to have been classified as such because the IAPs were dense. For LC classes that have low levels of woody vegetation, denser infestations of IAPs were taken to be an artefact of the course scale of the IAP data, since the satellite data are based on actual levels of woody vegetation.

For the BAU 2030 LC, we first projected changes in woody cover based on past trends (see section 4.2.4), before integrating this with the modelled IAP densities for 2030. Where woody cover had increased in extent, the change attributable to IAPs was taken into account in determining the change due to bush encroachment. Only changes in the natural environment were modelled, while the area under waterbodies, urban / built-up and cultivated classes were held constant. The 2030 areas of the latter would therefore be somewhat underestimated. For example, there was a net increase in built-up dense settlements of over 6 500 ha per year and in annual commercial dryland crops of over 15 000 ha per year between 2005 and 2017 in the Thukela catchment. The schematic in Figure 5.6 shows the general process followed for integrating IAPs into the 2017 LC.

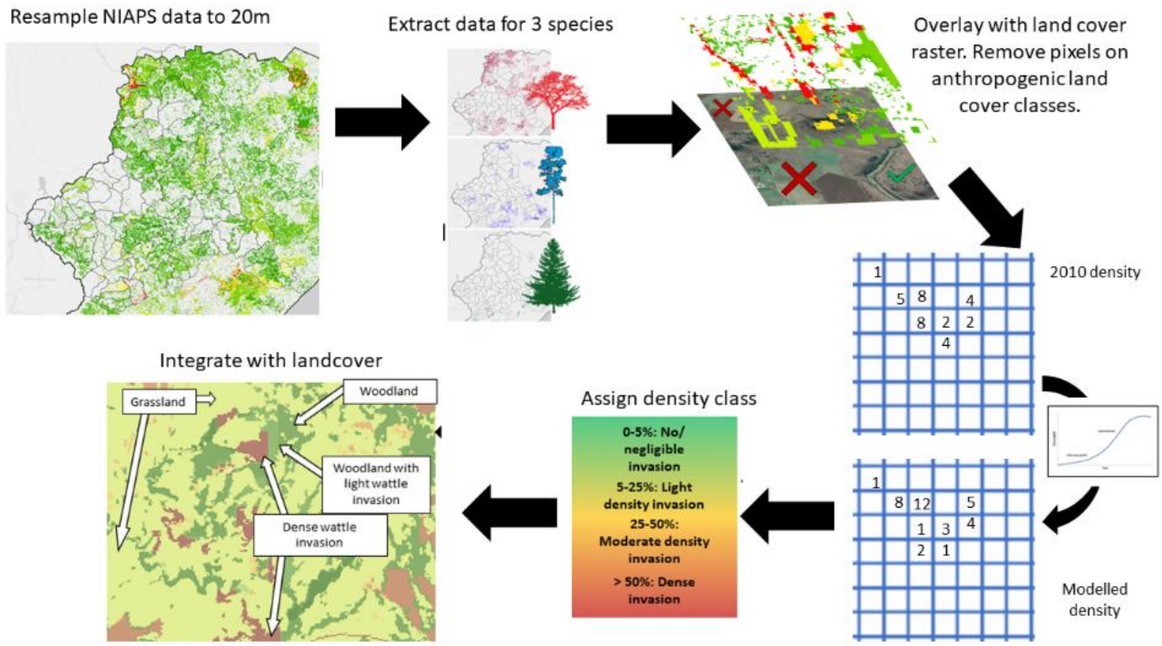

Figure 5.6. Schematic showing the process of integrating the NIAPS data into the land cover.

5.1.7 Creation of the Restored LC (2030)

The restored landscape represented the opposite extreme of the BAU scenario, and involved eliminating all IAPs infestations of >5%, 'reversing' bush encroachment that had occurred since 2005, and restoring areas that had reduced vegetation cover, including gully erosion.

To create the land cover dataset for this scenario, the densities of IAPs in infested areas were set to the minimum class, and the land cover was set to the estimated original class before IAPs, by removing IAPs from the integrated class. To effect the reversal of bush encroachment, the pixels that had increased woody cover in the 2017 LC than in the 2005 LC were returned to the 2005 LC class.

To simulate the restoration of lost vegetative cover, the pixels that had reduced woody cover in the 2017 LC than in the 2005 LC were returned to the 2005 LC class. The degraded land cover classes were treated as follows. Degraded forest was reclassified as forest. However, the remaining degraded LC classes are ambiguous – degraded bushland could be Dense thicket & bush, medium bush and woodland, and degraded grassland could be grassland or grassland / bush clumps mix. Thus the spatial data on the intact natural classes were extracted into separate layers and used to guide the choice of restored bushland or grassland class based on adjacency to the relevant degraded pixels, using the "Nibble" tool in ArcGIS (Figure 5.7). Similarly, the erosion dongas were returned to the same class as their surrounds (e.g. grassland or cultivated area; Figure 5.8).

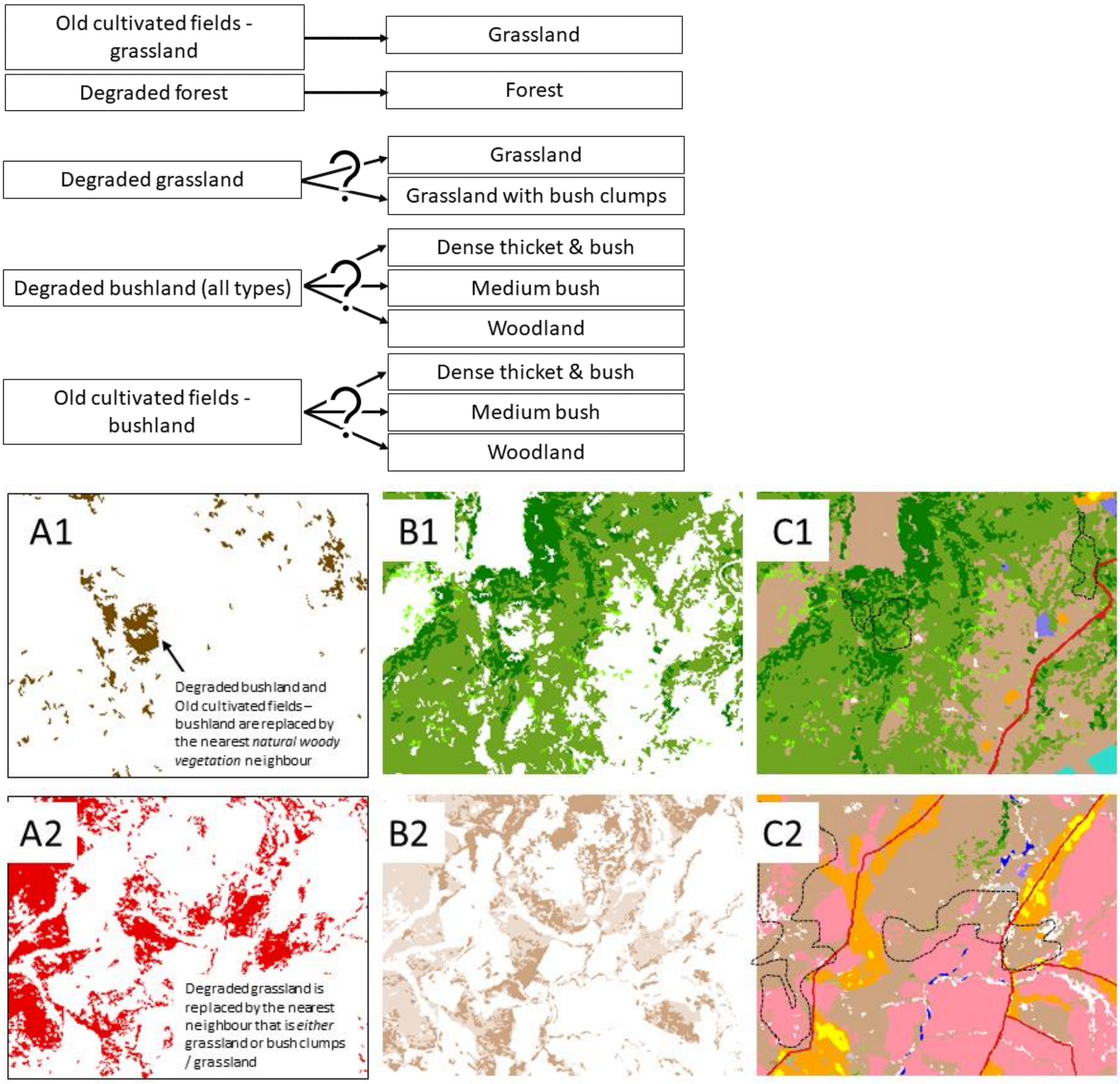

Figure 5.7. Diagram showing the process for 'restoring' degraded bushland and old cultivated fields – bushland (A1-C1) and degraded grassland. Degraded bush and old cultivated fields – bushland (A1) are replaced only by the nearest natural woody vegetation neighbouring pixel, regardless of the distance using the Nibble tool (B1) to form the restored scenario (C1) Eroded areas (just east of Colenso, Thukela catchment) as per the 2005 LC shown in reddish-brown. Degraded grassland (A2) are replaced only by the nearest natural grassland vegetation (grassland / bush clumps mix or grassland) neighbouring pixel, regardless of the distance using the Nibble tool (B2) to form the restored scenario (C1).

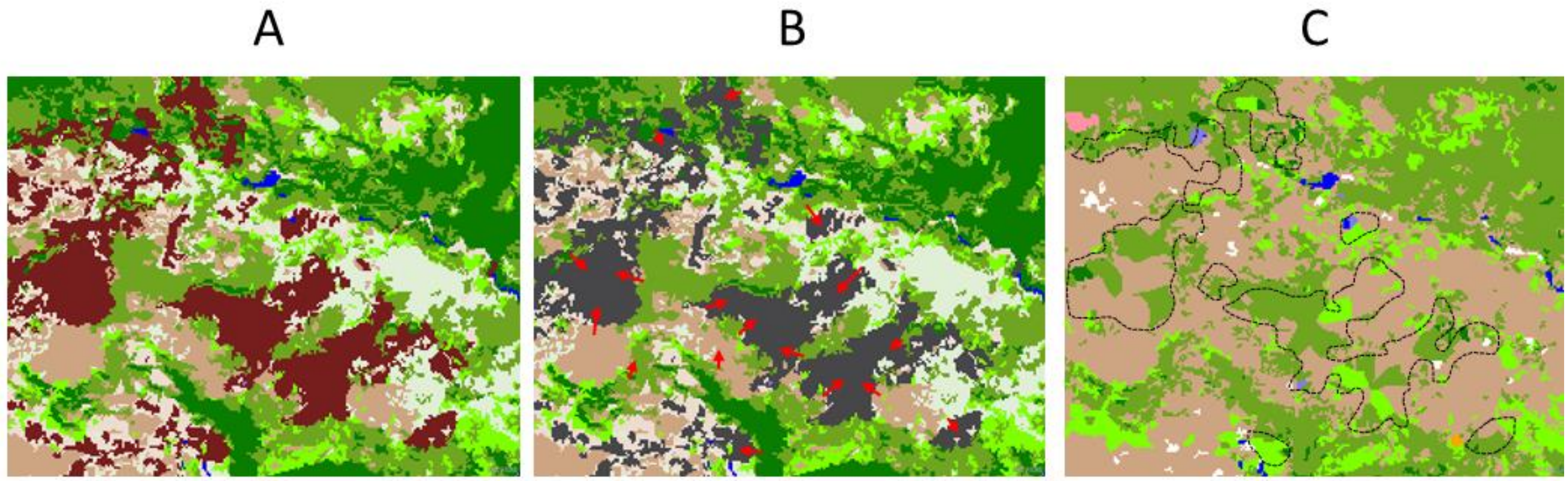

Figure 5.8. Diagram showing the process for removing erosion to mimic restoration for the restored scenario. A) Eroded areas (just east of Colenso, Thukela catchment) as per the 2005 LC shown in reddish-brown. B) The eroded areas are removed from the 2005 LC raster. The red arrows show the general direction that the land cover 'encroaches' into the set value (erosion) when the Nibble tool is run, whereby the nearest neighbour to each blank cell replaces the blank cell. C) The 'restored' areas once the Nibble tool has run. The previously eroded areas are roughly shown by the dotted outline polygons. Note that some areas (pale green and brown) are also replaced as these represent degraded areas in A and B, and are transformed as per the steps for reducing degraded land.

5.2 Impacts on ecosystem services and benefits

5.2.1 Sediment retention

In order to estimate changes in sediment retention, we built on the Soil and Water Assessment Tool (SWAT) model (Arnold *et al.*, 1998; Arnold & Fohrer, 2005) that was developed for the pilot accounts for KwaZulu-Natal (Turpie *et al.*, 2020b). The model was modified to include the more complex 2017 land cover dataset that was developed for this study.

SWAT is a physically-based, semi-distributed hydrological model that can operate on a daily, monthly, or annual time step and is designed to predict the impact of management on water, sediment, and agricultural chemical yields in ungauged or sparsely observed watersheds. Major model components include weather, hydrology, soil temperature and properties, plant growth, nutrients, pesticides, bacteria and pathogens, and land management. The model is set up for a catchment area or basin (= watershed in USA). The catchment is divided into multiple sub-catchments, which are further subdivided into hydrologic response units (HRUs) that consist of homogeneous land use, management, and soil characteristics. The HRUs represent percentages of the sub-catchment area and are not identified spatially within a SWAT simulation. There are approximately 40 parameters set for each LULC class in the SWAT structure.

The SWAT model provided estimates of changes in sediment yields (in tonnes) in each sub-catchment under the different scenarios. This was converted to volume using a density of 1.35 t/m^3 (Rooseboom, 1992; Haarhoff, J. & Cassa, 2009). Due to the potentially large and costly damages of sedimentation, we assumed that the service would be fully demanded, and we used the replacement cost of lost storage capacity (e.g. through raising the dam wall, constructing a substitute dam at a new site to make up the reduction in capacity or constructing check dams) to estimate its value. This was done by estimating the amount of storage that would have to be constructed to prevent a similar amount of sediment from reaching downstream aquatic environments, using an average capital replacement cost of R9.90 per m^3 (2020 Rands, Preston, 2015).

5.2.2 Water supply

The SWAT model also estimated changes in catchment runoff under the different scenarios. This required the parameterization of 16 new LC classes of invaded vegetative land class/ IAP type/ density combinations. Being a desktop study, the parameters identified as most likely to be affected by IAPs were estimated based on recent work in the study area (Scott-Shaw, 2018; Scott-Shaw, Hill & Gillham, 2020). Scott-Shaw (2018) and Scott-Shaw *et al.* (2020) developed and calibrated a SWAT model based on three years of research on species specific tree water-use within a riparian forest along the Buffleljags River in the Western Cape of South Africa, as well as in a summer rainfall region in the upper uMgeni catchment of KwaZulu-Natal in South Africa. The site in the upper uMgeni consisted of an indigenous stand of eastern mistbelt forest that had been invaded by *Acacia mearnsii*, *Eucalyptus nitens* and *Solanum mauritianum*. The outputs of the SWAT model were provided for each sub-catchment and HRU, at a monthly time step. These included evaporation and percolation, surface flows, lateral flows and groundwater contribution to streamflow, sediment yields and nutrient loads.

The estimation of effects on runoff were considerably lower than expected and were not considered to be reliable. This was likely related to the ambitious scale of the modelling (much larger than usual for SWAT), as well as the difficulty of replicating the complexity of the land cover data in model. Given that the flows are primarily affected by IAPs, which have disproportionately high evapotranspiration rates, we relied on existing estimates of the impacts of IAPs on flows in the Thukela catchment (van Wilgen & Wilson 2018, Cullis *et al*. 2007, Le Maitre *et al.* 2016), but adjusting for changes in IAP extent under the different scenarios. These estimates are based on simpler hydrological models. A unit-area flow reduction of 1243 m^3/condensed ha for IAPs in the Thukela catchment was used to estimate the total annual reduction in surface water runoff (Le Maitre *et al.* 2016). This was converted into a reduction in yield based on the predicted changes in yield associated with MAR estimates based on Cullis *et al.* (2007). The impact of IAPs and bush encroachment on flows was valued as the avoided losses in catchment yield costed using the municipal sales price of water as provided in the South African Water Accounts for the Thukela WMA and inflated to 2020 Rands (~R11.03/m^3).

5.2.3 Carbon storage

Using the South African National Carbon Sink Assessment (Department of Environmental Affairs, 2015), total ecosystem carbon in the Thukela catchment was estimated for the 2017 Baseline and for the 2030 Scenarios. The physical mean carbon value (g C/m_2) for each natural, cultivated and degraded LULC class was extracted from the National Carbon Sink Assessment Total Ecosystem Organic Carbon map and multiplied by the area of each land cover type within Thukela catchment in 2011 (the closest year to the date of data release 2010) to get a total ecosystem carbon value for each land cover type in each year.[22]

The carbon storage value of these stocks was valued in terms of the avoided losses of economic output by South Africa as well as the rest of the world, using recent published estimates of the global and disaggregated country-specific damage effects of climate change. Estimates of the social cost of carbon (SCC) vary greatly, ranging from US$10 to US$1000/tCO_2 (Ricke *et al.*, 2018). Recently, Nordhaus (2017) provided an updated estimate of global SCC as US$31/t$CO_2$ and Ricke *et al.* (2018) produced a far higher estimate of US$417/ t$CO_2$. This was disaggregated to country-level, with the estimated cost to South Africa being US$3.31, which is 0.8% of their global SCC estimate. The global SCC net of the South African portion can be considered as an exported service in the form of cost savings to the rest of the world. These damages are expressed as US$ per tonne of CO_2 emissions. Thus, carbon stocks were first converted to the

22 Values for dense *Acacia spp.* Stands were obtained from Kalita *et al.* (2016), while areas *Eucalyptus spp.* and *Pinus spp.* values were obtained from Guedes *et al.* (2018) who obtained total ecosystem carbon estimates in plantations in savanna areas of Mozambique

equivalent quantity of CO_2 using molecular weight of CO_2/molecular weight of carbon and then converted into 2020 South African Rands.

In this study, we used an average SCC value based on the recent estimates of Ricke *et al.* (2018) and Nordhaus (2017) to estimate the total value of carbon storage in the catchment from both a South African perspective and a global perspective (Table 5.2). It is important to note that the value of SCC is expected to increase over time as populations and per capita incomes grow, and thus it is strictly correct to see the estimate being specified in terms of the year of emission. For example, using the DICE model, Nordhaus (2017) provided updated estimates of the SCC for a ton of CO_2 emitted in 2015 (US\$31.25/tCO_2 in 2010 US\$) and also for CO_2 emissions in a range of future years. These values increased at a real growth rate of 3% per year. Carbon retained in the environment will increase in real value over time. Thus, we adjusted the Ricke *et al.* (2018) and Nordhaus (2017) SCC estimates at a rate of 3% per year to derive estimates for 2030 (Table 5.2, in 2020 Rands).

Table 5.2. The estimates of the Global and South African SCC values per tCO_2 used in this study based on values from Nordhaus (2017) and Ricke et al. (2018), all in 2020 South African Rands.

	Base estimate (average)	**Worst-case (Nordhaus 2017)**	**Best case (Ricke *et al.* 2018)**
Global SCC per tCO_2 in 2030	5 303.5	953.1	9 653.9
South Africa SCC per tCO_2 in 2030	41.9	7.53	76.3

The SCC is a net present value of avoided costs, typically over 100 years. However, for accounting purposes, values must be determined for the year in question. Thus, the annualised social cost of carbon (ASCC) was then estimated as:

$$ASCC = \frac{(\delta * SCC)}{(1 - (1 + \delta)^{-t}}$$

where δ is the discount rate, and t is the time period of the SCC calculation in years. For this study, we assumed t = 100 years, and we used a social rate of discount of 3.66%.

5.2.4 Livestock production

The ecosystem service is the land's contribution to production, which includes fodder provision and other inputs. Fodder provision was not quantified, but as a proxy, we quantified the amount of production supported in terms of large stock units. Changes to livestock production as a result of changes in the extent of rangeland areas under each scenario were estimated using livestock statistics as calculated in the pilot accounts (see Turpie *et al.*, 2020b) where the annual production was valued in terms of resource rent, which is the economic rent that accrues in relation to environmental assets, including natural resources and ecosystems. Labour costs, user costs of fixed capital and intermediate inputs are deducted from the market value of the outputs (benefits). The ratio of intermediate expenditure, labour costs and capital expenditure to gross income was taken from the 2007 Agricultural Census. The annual capital expenditure was taken as a proxy for cost of capital.

For commercial livestock, production was estimated at the magisterial district level and for communal livestock at the municipal ward level. Using DAFF quarterly statistics on the number of communal and commercial livestock in KwaZulu-Natal from 1996-2018 we estimated the livestock production under the 2017 Baseline Scenario. We then estimated the change in grazing land[23] under the LDN and Restored scenarios relative to the BAU scenario within the communal and commercial land at the district and ward level and adjusted the production values for each scenario based on this percentage point change, assuming that livestock are being stocked at maximum levels in the catchment.

5.2.5 Harvested Resources

To estimate the impact of different scenarios on the value of harvested resources, we used the same spatial modelling approach as (Turpie *et al.*, 2020b) focusing on those resources whose availability was likely to be affected, namely: woody resources (firewood, poles and timber), thatching grass, wild plant foods and medicines, and wild meat. We estimated the annual subsistence or small-scale harvest of resources under each scenario, based on spatial estimates of their supply in relation to the estimated demand. All values were converted to 2020 prices.

Demand for each natural resource group was estimated at the census sub-place level. Data from the 2011 Census and 2016 Community Survey Census were used to calculate adjustment factors at the District Level, which were then applied to obtain estimates of natural resource demand at the sub-place level for 2017. Total demand per sub-place was obtained from data on the number of households in each subplace using a particular natural resource, and the average amount of the resource demanded per household.

Estimates of stocks of natural resources across various land cover types were based on the values used in the pilot accounts (Turpie *et al.*, 2020b). These were derived from information in the literature relating to different ecosystem and vegetation types, as well as species distributions, where appropriate. For this study, the estimates needed to be extended to the more detailed land cover classes used, which took IAP densities into account. This required adjusting resource stock estimates for invaded land cover classes, where appropriate. In cases where a natural vegetation type already had high woody biomass, stock estimates were not altered for invaded classes. For example, forest invaded by wattle had the same firewood stock estimate as uninvaded forest. Conversely, stocks of non-woody resources were reduced for invaded land cover classes. This is because we assumed indigenous plant foods and medicines would decline, and suitability of the habitat for wildlife would decrease. Similarly, grass stocks

[23] These were all areas deemed suitable for grazing based on land cover, and without accounting for grazing capacity. They included wetland and drainage lines, woodland, grassland (all types), degraded grassland and old cultivated fields (grass) including invaded areas up to a medium density. Studies conducted at a finer scale should seek to incorporate location-specific values for carrying capacity.

were reduced where invasion occurred in a land cover with little to no woody vegetation in its natural state (e.g. grassland). Once stock estimates had been made for all land cover classes, they were then modified to account for the varying accessibility of natural resources across the catchment. For all private land and protected areas, we conservatively assumed only 10% of natural resources were available for harvesting. For land under communal tenure, we assumed all resource stocks were fully available for harvesting.

For the BAU, full restoration and LDN scenarios, we assumed that there would be very little change in demand from 2017 to 2030. The justification for this is that increases in population would roughly be balanced by ongoing declines in traditional natural-resource-based livelihoods. Indeed, demand for natural resources between 2005 and 2011 decreased by 12% across all resource groups. This was the result of a decrease in the number of traditional houses in the catchment and therefore less dependence on poles and thatching grass for construction, as well as a significant decrease in the number of households using wood for cooking and heating. The change in demand between 2011 and 2017 was not as noteworthy with a decrease of only 1% across all resources. This was due to the increase in population in the catchment and a much smaller decrease in the number of traditional houses, and in some instances an increase. The use of wood for cooking and heating did decrease substantially as more households attained access to electricity. Hence, as demand has flattened out with very little change between 2011 and 2017, changes in natural resource use under the different scenarios were estimated solely on the changes in stocks that were a result of changes in land cover.

5.2.6 Tourism

Tourism value was estimated following the methods of Turpie *et al.* (2020b) who used a combination sub-national tourism data and the density of geotagged photographs uploaded to the internet to map tourism value to ecosystems and other attractions across KwaZulu-Natal.

Tourism expenditure by foreign and domestic tourists in KwaZulu-Natal in 2017 was extracted from South Africa’s State of Tourism Report 2016/2017. The proportion of tourism expenditure attributed to visiting attractions, as opposed to activities such as visiting family and friends, attending conferences religious events, or receiving medical treatment was estimated for each category of tourists (holiday, visiting friends and relatives, business and other) based on information collated from the SA Tourism annual performance reports and from data collected in regional tourist offices. Tourists whose main purpose is either visiting friends or family, or business tend to spend much less of their money on visiting attractions than holiday/leisure tourists. These types of tourists do, however, make up a large proportion of the total tourism spending and so these contributions are not insignificant.

The ecosystem contribution to tourism was valued as resource rent generated by nature-based tourism, which is the residual of the total output after all costs for capital and labour have been subtracted. The gross operating surplus was calculated using a conversion factor for 2017 extracted from the South African Tourism Satellite Accounts, and then user costs of fixed capital were subtracted to derive the resource rent (see Turpie *et al.* 2020b for more detail on this approach). This value was then converted into 2020 prices.

The spatial distribution of tourism value was mapped based on the density of geotagged photographs uploaded on the website *flickr.com*. These densities were obtained using the InVEST Recreation Model 3.5.0 (www.naturalcapitalproject.org) which uses an API to get data from the website into a grid specified by the user. Densities of geotagged photographs provide a means of mapping value to tourism attractions, rather than to the places where tourists spend their money (e.g. at their accommodations), so is more accurate in assigning the tourism value to the actual attractions that caused the expenditure. The model calculates the average annual photo-user-days (PUDs) for each grid cell (1 km x 1 km) across the period 2005-2017. The model used the latitude/longitude data from photographs as well as the photographer's username and photo date to calculate PUDs. One PUD is one unique photographer who took at least one photo in a specific location on a single day. The tourism value for was then spatially allocated in proportion to photo density in order to determine the value of tourism within the Thukela catchment. These values were then apportioned based on land cover data using the Baseline 2017 Land Cover to determine the proportion of attraction-based tourism value attributable to natural ecosystems (i.e. nature-based tourism).

We used tourism statistics on the number of visitors and tourism expenditure within KwaZulu-Natal and in South Africa over the period 2008-2019 to make a set of simple assumptions on how tourism might change over the next decade under a BAU management approach and a restoration focused management approach. While tourism expenditure in South Africa has increased over the past decade, it has shown a declining trend in KwaZulu-Natal over the same period. We used both sets of statistics to develop a "Best Case" and "Worse Case" scenario for how tourism might change in the future. The statistics were used to generate a simple relationship between tourism expenditure over time which we then modelled into the future up until 2030 (Figure 5.9). The median of the Best- and Worst-Case scenarios was used as the most likely tourism forecast.

We assumed that tourism under the Baseline (LDN) Scenario would follow current trends. Under the Restored Scenario we assumed that tourism would follow the same trend as under current conditions but taking into account the growth goal set out in South Africa's National Biodiversity Economy Strategy (NBES) which states that by 2030 the South African biodiversity economy will achieve an average annualised GDP growth rate of 10% per annum. Therefore, we increased tourism in the catchment relative to the Baseline incrementally reaching a 10% annual growth by 2030. Under the BAU Scenario we assumed that tourism would decrease relative to the Baseline (LDN) as a result of the impacts of a degraded landscape on visitor satisfaction and the loss of suitable wildlife areas for developing the Wildlife Economy strategy.

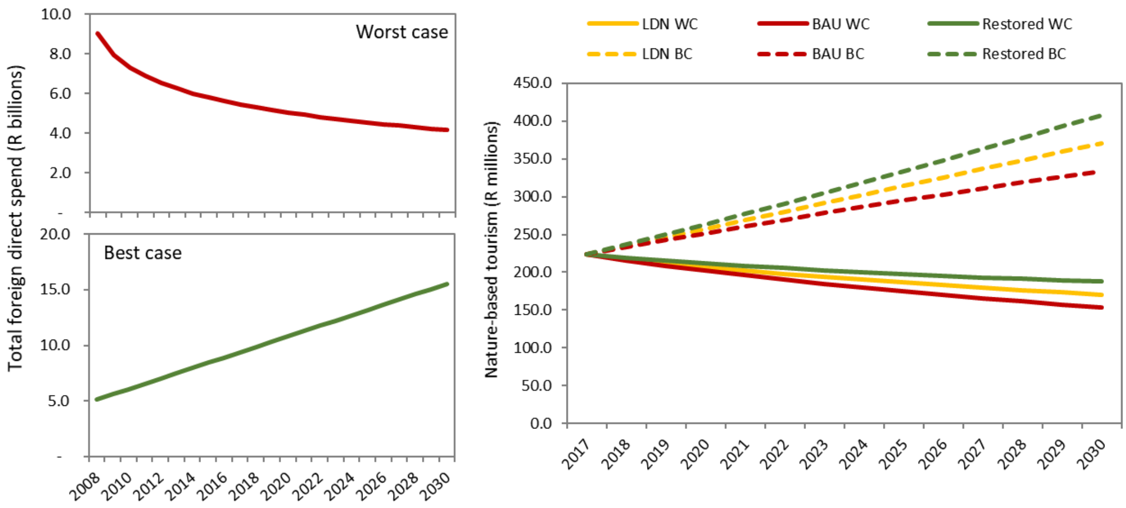

Figure 5.9. (Left) Actual (2008-2019) and forecasted (2020-2030) total foreign direct spend (R billions) in KwaZulu-Natal (worst case) and South Africa (best case) used to generate (Right) a simple forecast model to determine changes in tourism value in 2030 under the Baseline (LDN), BAU and Restored Scenarios. The median of the best- and worst-case scenarios was used as the most likely forecast of nature-based tourism value in the catchment.

5.3 Design and cost of interventions

The design and cost of the interventions were estimated based on data collated from DFFE, particularly for IAP clearing in the study area, as well as from the literature. The analysis took into consideration estimates of the costs of interventions and their management, including engineering costs of active rehabilitation; management, regulation and enforcement costs; IAP clearing costs; and per hectare costs associated with measures to improve rangeland management. Given the timing of the study, the analysis assumed that the additional interventions as designed in this study would begin in 2021. In fact, the LDN targets were only published in 2018, and the work undertaken in the catchment since then was incorporated into the analysis.

To evaluate costs of restoration and rehabilitation, we derived cost per hectare estimates for clearing IAPs, addressing dense bush encroachment, and addressing erosion, land degradation and denudation through SLM, based on information from the literature.

5.3.1 Clearing of invasive alien plants

Most estimates of the IAP clearing costs in SA are based on the Working for Water programmes' person-day estimates. These person-day estimates are built on data pertaining to costs of invasive alien plant clearing developed over the programmes' lifetime and give an indication of the number of person days required for clearing different groups of invasive plants (aquatic, cactus, herbaceous, sprouting and non-sprouting trees) at different age classes and for landscape and riparian infestations for a number of different treatment methods. The Working for Water information system contains data on clearing efforts throughout South Africa, including person-day estimates, total cost and initial density estimates. These data, along with the spatial extent of each management unit for 2001-2012 (Wannenburgh, 2015) were used by Turpie *et al.*, 2018 to develop a set of regression models which estimated the person days required to clear a hectare of infestation. These showed that the costs per ha of clearing rise exponentially with increasing density. The models were used to estimate the clearing costs per ha of *Acacia*, *Eucalyptus* and *Pinus*, respectively, based on their density.

5.3.2 Restoration of areas affected by dense bush encroachment

The clearing or thinning of woody cover can be undertaken manually or mechanically. Manual clearing is labour intensive and involves the removal of woody plants through felling or stumping using tools such as handsaws, axes, mattocks, chainsaws or hand-held revolving blades (Turpie *et al.*, 2019). The height at which the woody plant is cut and the season in which it is cut are important factors that affect regrowth. Woody cover can also be cleared using heavy machinery such as bulldozers, tractors and custom-built bush cutting machines. While this approach can be faster than manual clearing it can cause substantial soil disturbance, which favours the establishment of woody seedlings and usually requires aftercare treatments (Turpie *et al.*, 2019). In most cases, especially on communal land, mechanical clearing can be prohibitively expensive. Therefore, it is usually more feasible to clear manually than mechanically, owing to the lower cost. Manual clearing can also contribute towards job creation, which is an important consideration, particularly in communal areas.

Cost estimates for indigenous bush encroachment control in South Africa are scarce relative to IAP clearance costs. For this study, cost estimates of manual clearing were taken from Turpie *et al.* (2019), converted into 2020 prices. The cost for manual clearing was assumed to be R4410 per ha for manual clearance where a 25-75% reduction in woody cover was required, and R2646 per ha where restoration can be achieved with a less than 25% reduction in woody cover. Areas mapped as bush encroached were assigned the higher or lower restoration cost estimate, based on the level of clearance that would be required according to the encroachment pathway. For example, where dense bush would need to be restored to medium bush, we used the lower cost estimate, since we assumed a <25% reduction in woody cover would be sufficient. However, restoring dense bush back to grassland would require removal of >25% of woody cover, hence the higher cost estimate was used for this bush encroachment pathway. For the full restoration scenario, these cost estimates were applied to all areas where indigenous bush encroachment had occurred between 2005 and 2030. For the LDN scenario, we costed the reversal of all indigenous bush encroachment which was estimated to occur between 2015 and 2021.

5.3.3 Rehabilitation of erosion gullies

The rehabilitation of erosion gullies is an incredibly expensive undertaking that is labour intensive, requiring structural and management measures. Rehabilitation is undertaken to protect the bare soil and reduce the energy of runoff water, with the aim of enhancing vegetation cover and reducing sediment export. This can be achieved through a number of structural means, such as through reshaping gully banks, fitting gabions, constructing silt fences, using soil erosion blankets, and reseeding or replanting vegetation as well as the use of brush packing[24]. The cost of rehabilitation depends largely on the level of degradation and the extent to which the above structural measures need to be implemented.

As part of the Maloti Drakensberg Transfrontier Project in the Upper Thukela catchment the cost of rehabilitating gully erosion through the construction of gabions and other structural interventions[25] was estimated to be in order of R55 500 per ha (in 2007 prices, R112 110 in 2020 prices) in year one. Follow up costs were required in year three, five and seven at 70%, 30% and 10% of the cost in year one (Mander *et al.*, 2007). In the Mapungubwe National Park in the Limpopo Province a rehabilitation project was implemented in 2017 to address gully erosion due to overgrazing[26]. The rehabilitation included reshaping of gullies through the integration of silt fences, erosion blankets and brush packing. However, the project focused on a small area of less than one hectare, with costs of construction materials (soil erosion blankets and silt fences) exceeding R150 000 per ha. While the use of these materials could be effective in rehabilitation, their implementation at a large scale is not economically feasible. Finding cost-effective measures for gully rehabilitation remains a challenge (Liu *et al.*, 2019).

Indeed, the literature suggests that it is more economical to undertake rehabilitation in the early stages of gully formation or to prevent the formation of gullies from starting through improved rangeland management than trying to rehabilitate once the gully has formed (Milton, Dean & Richardson, 2003; Valentin, Poesen & Li, 2005; Carey *et al.*, 2015). Furthermore, the use of structural materials to control vast areas of gully erosion is unpractical, requires significant expense, is difficult, often ending in failure, and rarely achieves benefits in terms of an increase in land productivity (Valentin *et al.*, 2005; Carey *et al.*, 2015; Gomez *et al.*, 2020). Therefore, given the high cost and low success rate of gully rehabilitation and that in most cases gully erosion can be prevented through SLM practices, it was assumed that these areas would be targeted through SLM initiatives as opposed to active structural rehabilitation.

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[24] Brush packing is the method of covering degraded soils with packed woody branches and/or other organic material, to simulate the protective cover effect of vegetation. Brush packing provides a micro-climate for grass seed to germinate and establish and also protects the area from grazing if thorny vegetation is used.

[25] Limited detail was provided on the structural materials used and their implementation.

[26] Information about this project was extracted from the Global Database on Sustainable Land Management (SLM) of WOCAT (the World Overview of Conservation Approaches and Technologies); https://qcat.wocat.net/en/wocat/technologies/view/technologies_3359/.

5.3.4 Sustainable land management

It is well recognised that the costs of restoration increase exponentially as the level of degradation progresses and vegetation cover is lost (Milton *et al.*, 2003). Improving land management is one of the most cost-effective interventions to prevent and halt degradation. Furthermore, in order to address lower levels of bush encroachment it is important to introduce more effective land management practices to maintain the gains made through any restoration or rehabilitation efforts. Appropriate management of grazing and fire are crucial for preventing bush encroachment from reoccurring and for allowing vegetation recovery. Improving land management practices in private and communal rangeland areas requires strengthening extension services and institutions for promoting sustainable rangeland management practices, which include sustainable stocking rates and rotational grazing practices that provide adequate rest for grazing areas.

As outlined by Turpie *et al.* (2019) improving land management in order to maximise productivity and ensure resilience to climate change is the mandate of government. Improving land management requires expanded deployment and improved effectiveness of agricultural extension services, as well as implementing appropriate incentives and institutional changes to facilitate improved management in communal and commercial rangeland areas (Abdu-Raheem, 2014; Turpie *et al.*, 2019). However, the deployment of extension services needs to be a permanent and sustained intervention. To date, efforts by government to improve land management have largely been either insufficient or unsuccessful, partly due to the extent of degradation in the catchment, and partly due to lack of attention to the social context or a lack of continued funding to ensure sustainability and desired results. The ratio of extension staff to farmers was reported to be 1:878 in 2005 (Williams *et al.*, 2008; Greenberg, 2010). With extension staff numbers only decreasing over time, this ratio has likely worsened. In 2007 there were a total of just 2155 extension officers in the country with 360 of these (16%) situated in KwaZulu-Natal, with less than 25% of these having been exposed to technical training programmes since joining the public service (Williams *et al.*, 2008). In 2008 it was reported that KwaZulu-Natal had one of the highest shortfalls of extension personnel in the country due to the higher number of communal farmers in the province as well as the higher number of projects emerging as a result of the land reform programme.

Improving extension services, particularly in communal land areas, requires deploying more staff and improving training of current staff (Abdu-Raheem, 2014). Using information on the number of extension officers in the province and the ratio of officers to farmers from Williams *et al.* (2008) we estimate that 234 extension officers are needed in the Thukela catchment (twice current numbers) to achieve adequate and effective extension outreach at a cost of R74 million per year.

In addition to extension services, in order to halt any further degradation and to achieve the outcomes as laid out in the Land Degradation National Action Plan, the development of innovative and sustainable mechanisms for investing in ecosystem restoration are needed. Economic mechanisms and incentives, especially payments for ecosystem services (PES), are increasingly being recognised as a way of strengthening conservation, improving livelihoods, and generating revenue outside of the state budget for conservation. If implemented successfully, PES schemes can be more flexible, cost-efficient and effective than traditional approaches. The South African government is committed to promoting the emergence of PES schemes. National Treasury published a draft policy paper on market-based instruments to support environmental fiscal reform in South Africa (National Treasury, 2006), which has acted as a catalyst for market mechanisms in general, and government has a strong focus on developing South Africa’s green economy through the use of economic mechanisms and incentives such as PES schemes (PAGE, 2017). Indeed, it has been identified that there is an urgent need to factor environmental services into financial instruments in order to create a more sustainable economy and that green economy initiatives in the resource conservation and management sector should be further scaled up (PAGE, 2017).

A payment scheme for ecosystem services can be implemented in the catchment to provide incentives for SLM by making payments available to communities who implement improved rangeland management practices (rotational grazing, fire management) thereby generating ecosystem services (e.g. soil, water and biodiversity conservation) as co-benefits. Such a programme would incur ongoing costs estimated at R950 per ha per year which includes management costs, and an estimated implementation cost of R140 per ha in year 1 (Romero *et al.*, 2012; Turpie, Warr & Ingram, 2014). We assume that such a PES programme would be best implemented on communal land and that stewardship programmes such as certification schemes could be appropriate on private lands. We assume that the per hectare costs of such would be the same as that of an ongoing PES programme.

6 RESULTS AND DISCUSSION

6.1 Estimation of baseline and future potential degradation

6.1.1 Baseline extent and spread of invasive alien plants

Based on the NIAPS data IAPs covered just over 80 000 condensed ha or 2.8% of the catchment area in the LDN Reference year of 2015. We estimated that this would increase to 105 526 condensed ha, an increase of 32%, by 2021, and by 2030 it will have more than doubled to 160 584 condensed ha in the absence of interventions (Table 6.1).

Table 6.1. Estimated condensed area (ha) of the main invasive genera in the study area in 2015 (LDN Reference year), 2021 (start of the additional modelled interventions) and in 2030, the target year for achieving LDN.

	Condensed area (ha)		
Species	**2015**	**2021**	**2030**
Wattle *Acacia* spp.	58 148	76 821	120 452
Gum *Eucalyptus* spp.	17 788	24 447	35 702
Pine *Pinus* spp.	4 170	4 258	4 430
Total	**80 106**	**105 526**	**160 584**
Change from 2015		**25 419**	**80 478**

6.1.2 Bush encroachment

The total area that had changed to a woodier vegetation class between 2005 and 2017 (whether by bush encroachment, IAP spread or both) was 235 733 ha (Table 6.2). This is 10% of the remaining natural areas in the Thukela catchment at that point. Most of the change to woodier vegetation had occurred in what was initially classified as grassland. The predominant pathway appeared to be grassland → medium bush → dense thicket & bush, with significant areas having changed from grassland to dense thicket & bush. A substantial area defined as "Grassland / bush clumps mix" also transitioned to medium bush. Woodland occupies a much smaller area of the catchment but also had fairly sizeable areas changing to medium bush or dense thicket & bush.

We estimated that by 2015 (the LDN Reference year), just under 200 000 ha of the catchment had experienced an increase in woody cover since 2005 (8.4% of remaining natural area; Table 6.3). Based on past rates of change, we predicted that the area with increased woody cover relative to 2005 would reach 262 000 ha by 2021 and 323 000 ha (16.3% of the remaining natural area) by 2030 in the absence of intervention (i.e. 123 000 ha encroachment between 2015 and 2030). Certain sub-catchment areas are expected to be quite severely affected. For example, over 50 000 ha of grassland in the relatively small tertiary catchment V40 could have changed to woody vegetation of varying densities (relative to 2005), and over 35 000 ha could be under woodier vegetation in tertiary catchment V60 (Figure 6.1) by 2017. There is a concern that bush encroachment will reduce environmental flows, although because the majority of bush encroachment is in the middle and lower reaches of the catchment (i.e. V40 and V60), it is not as strongly linked to a reduction in water yields. Past and projected increases in woody cover are described in more detail in Table 6.3, including the estimated change in woody cover for each pathway, based on the increase in canopy cover suggested by the land cover classification. Note that *these* land cover changes *do not* distinguish invasive aliens from indigenous woody plants.

Table 6.2. Matrix showing the net change between each vegetation class between 2005 and 2017 for the Thukela catchment for changes resulting in increased woody cover.

2005 \ 2017	Grass-land	Grassland / Bush clumps mix	Wood-land	Medium bush	Dense thicket & bush	Forest
Grassland	-	26	8 670	**95 926**	**54 972**	739
Grassland / bush clumps mix		-	1 837	17 197	7 384	30
Woodland			-	5 118	5 691	113
Medium bush				-	**37 282**	170
Dense thicket & bush					-	579
Forest						

Table 6.3. Increases in woody cover from 2005 to 2015, and predicted changes to 2030.

Increases in woody vegetation cover	2005 to 2015 (ha)	2005 to 2021 (ha)	2005 to 2030 (ha)	2015 to 2030 (ha)
Total area with increased woody cover (ha)*	195 962	262 155	322 907	126 945
Total area with increased woody cover (con ha)	110 041	147 312	181 655	71 614
Bush encroachment (ha) (after subtracting IAPs)	**104 053**	**158 576**	**201 824**	**97 771**

* Condensed ha calculated using the midpoint of the canopy cover range for each land cover class; grassland = 0%, grassland / bush clumps = 10%, woodland = 40%, medium bush = 55%, dense thicket & bush = 85%, forest = 85%.

To estimate the area experiencing indigenous bush encroachment specifically, increases in woody cover due to invasive alien plants needed to be separated out. We therefore subtracted the areas impacted by IAPs from the areas that had shown an increase in woody vegetation cover to produce a ballpark estimate of the proportion of woody cover gains that were due to bush encroachment. The results suggested that indigenous bush encroachment had occurred across 104 000 ha of the catchment by 2015 since 2005 (4.5% of the remaining natural area). This was projected to increase to 159 000 ha by 2021 and 202 000 ha by 2030 in the absence of intervention, roughly a doubling in area in 15 years. These values are consistent with values elsewhere in the province. For example, Case & Staver (2017) reported remarkable changes in the Hluhluwe-iMfolozi Park (just north-east of the catchment area), where the area under medium tree density increased by 46% in just seven years.

While there is major uncertainty on the extent to which the increase in woody cover can be attributed to IAPs versus bush encroachment, the overall gain in woody cover is likely to be fairly reliable, as it is based on past trends (purely from the land cover).

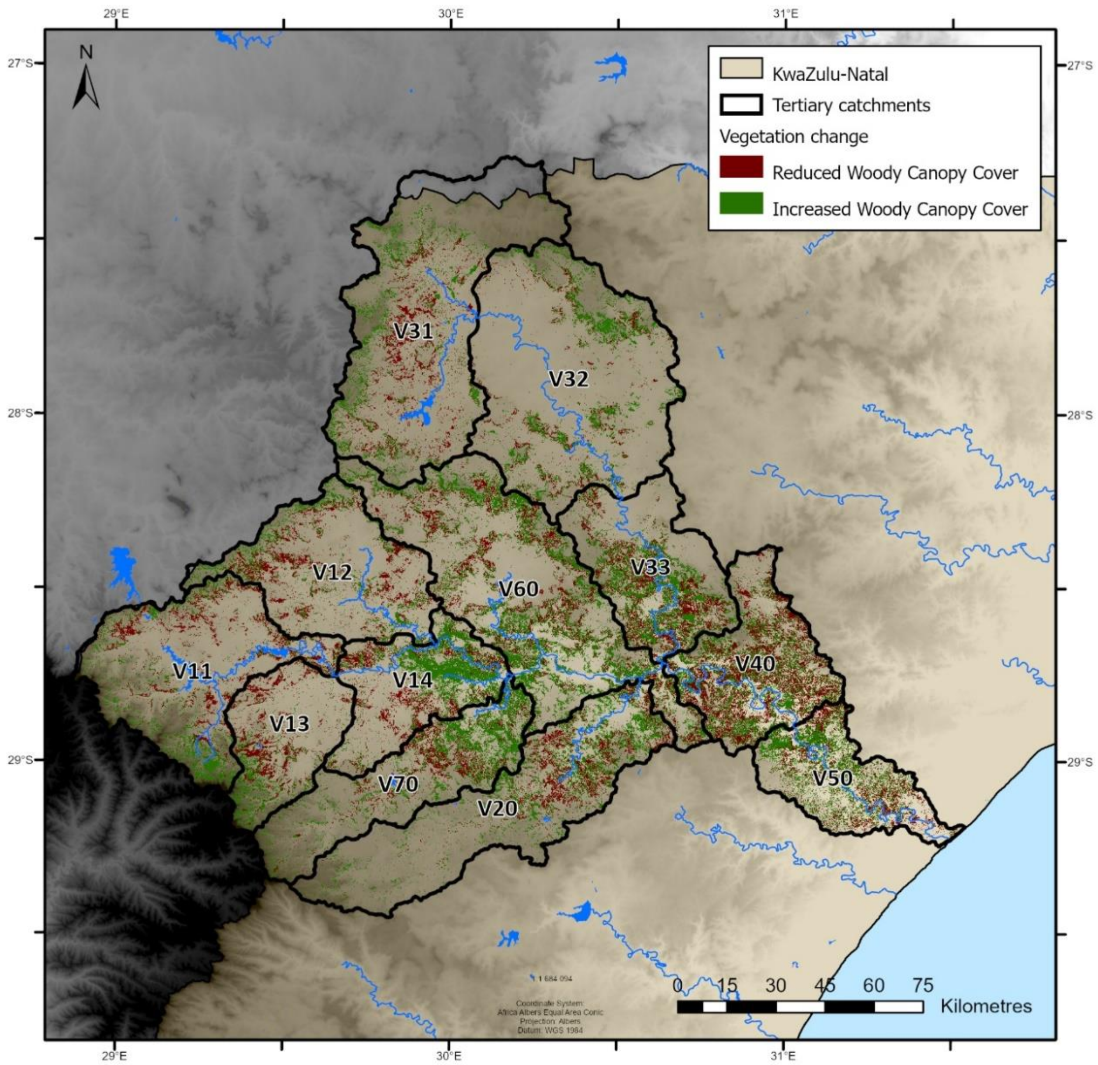

Figure 6.1. Projected changes in woody vegetation[27] cover by 2030, relative to 2005. Source: Derived from the KZN landcover series (2005, 2017).

[27] Excluding plantations

6.1.3 Loss of vegetative cover and erosion

Between 2005 and 2017, just over 202 500 ha of the catchment experienced some decline in woody biomass (Table 6.4). For areas that had reduced woody cover, by far the majority had changed to grassland, which is suggestive of bush clearing, the likes of which is commonly seen in the areas immediately around homesteads in communal lands. Indeed, 60% of all areas shown to have experienced a decline in woody cover in the Thukela catchment were in communal lands, despite comprising only 30% of the Thukela catchment. A substantial area of dense thicket & bush was reduced to medium bush, which could be related to resource use.

Table 6.4. Matrix showing the net change between each vegetation class between 2005 and 2017 for the Thukela catchment for changes resulting in decreased woody cover.

2005 \ 2017	Grasslan-d	Grasslan-d / bush clumps mix	Woodland	Medium bush	Dense thicket & bush	Forest
Grassland	-					
Grassland/bush clumps mix	32 568	-				
Woodland	15 773	3	-			
Medium bush	60 016	4	2 593	-		
Dense thicket & bush	24 855	1	587	64 253	-	
Forest	846	0	40	70	917	-

Between 2015 and 2030 the area of vegetation loss is roughly 94 000 ha or 40 000 con ha (Table 6.5). By 2030, about 74 ha of forest is expected to change to vegetation without or with less than 40% canopy cover, per year, and 2 071 ha of dense thicket & bush is expected to change to grassland. Based on past trends, forest and woodland loss from 2015 to 2030 is expected to be greatest in tertiary catchment V31. The conversion of Dense thicket & bush, grassland / bush clumps, woodland and medium bush to more open vegetation is expected to be most extensive in V60 but proportionately greatest in V40 (Figure 6.1). The causes of this are likely to be several but it is likely that deforestation and harvesting for various wood products, from subsistence to commercial scale are the reason for these changes. That said, literature detailing woody vegetation loss in mesic savannas is scarce and what exists is usually focused at the level of village or municipality, with a specific focus on fuelwood harvesting and use. Substantially more attention is paid to woody encroachment, particularly in protected areas. While wood harvesting most certainly occurs, there is evidence from recent censuses that fuelwood harvesting is declining in KwaZulu-Natal more rapidly than elsewhere (such as the area bordering the Kruger National Park, Madubansi & Shackleton, 2007) and the province has lower use of wood for energy purposes than other predominantly rural provinces (Statistics South Africa, 2012; Lloyd, 2014; Russell & Ward, 2016). While thicket and woodlands remain susceptible to harvesting and browsing, indigenous forests are well protected in the Thukela catchment and less vulnerable.

Table 6.5. Decreases in woody cover from 2005 to 2015, and predicted changes to 2030.

Decreases in woody vegetation cover	2005 to 2015 (ha)	2005 to 2021 (ha)	2005 to 2030 (ha)	2015 to 2030 (ha)
Total area with decreased woody cover (ha)	168 771	221 030	262 664	93 893
Total area with decreased woody cover (con ha)*	70 327	92 340	110 224	39 897

Gully erosion, a major driver of vegetation loss, was predicted to increase from 2017 to 2030, albeit by only 2.5% to 61 444 ha, but still constitutes 2.1% of the total catchment area. The greatest percentage increases are in the V50 tertiary catchment (132%) and V40 (119%), while the greatest net increase is in V32 (1 088 ha), followed by V60 (1 061 ha). The two lowest tertiary catchments account for 32% of all eroded areas in the Thukela catchment in 2030. Sheet erosion, which is difficult to determine from remote sensing techniques is likely to be a significant problem but could not be isolated in this study. The total degraded areas still occupy 8% of the catchment, of which 3.7% is degraded grassland.

There is considerable uncertainty regarding both past and future changes in land cover in the study area. Some of the apparent changes in woody cover over the period since 2005 may have been due to variability in the quality of satellite data used in the land cover datasets, notably the lack of multi-seasonal imagery available in earlier LULC product iterations. Comparing images from different times of the year can have significant effects on readings of canopy cover. We acknowledge that vast improvements to the mapping in more recent years has led to more accurate classifications. However, this may reduce comparability with potentially less accurate older classifications, such as the 2005 LC data. The 2008 LC report indicates that the users accuracy for natural vegetation classes are very low, with values of 41.7% for grassland / bush clumps mix, 53.6% for grassland and 65.5% for medium bush (GeoTerra Image, 2010). Areas of undulating and steep topography such as the Drakensberg Mountains may also lead to distortions of imagery if orthorectification of imagery is not successful, and misclassifications in areas of shadow. Unfortunately, the 2005 LC remains the best baseline available for determining LULC change, particularly for woody vegetation change at the catchment-level scale.

The above estimates of the area of degradation were based on land cover changes from 2005 to 2017, but did not take NDVI changes into account. We compared our results to those obtained for the study area over the same time period using Trends.Earth, which takes NDVI, SOC and land cover changes into account using global datasets (Figure 6.2).

The total degraded area estimated by Trends.Earth was markedly higher than we obtained based on changes in the KZN land cover, mainly due to widespread reductions in NDVI. Conversely, Trends.Earth did not detect the same level of degradation based on changes in land cover. NDVI allows a level of fine tuning, in that it would pick up changes within a land cover class. Our use of a refined and highly detailed, partially ground-truthed LULC product compared with the ESA CCI global land cover maps used by Trends.Earth resulted in our mapping of degradation based on land cover being somewhat intermediate between the land-cover and NDVI results of Trends.Earth. This further highlights the challenges with respect to measuring land degradation based on satellite data alone.

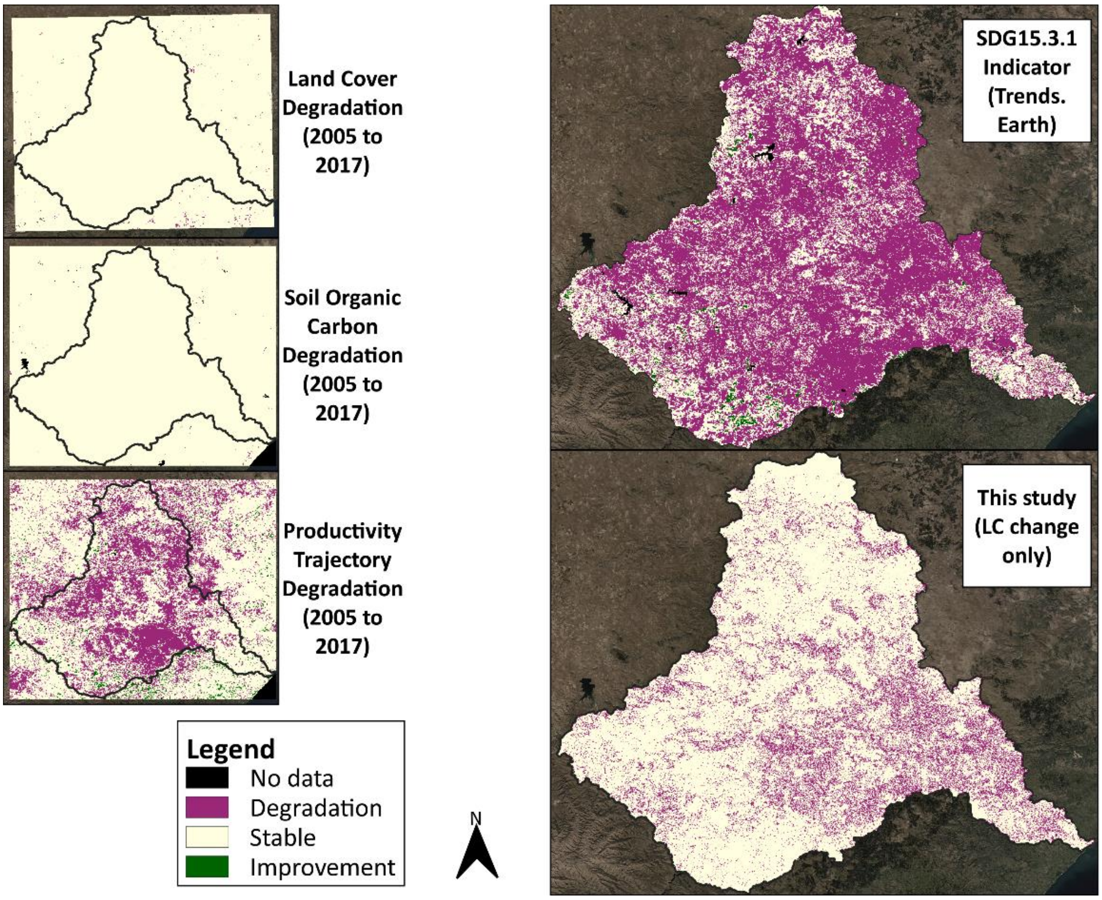

Figure 6.2 The Trends.Earth results for the Thukela catchment of the three sub-indicators for determining land degradation with respect to SDG Indicator 15.3.1, between 2005 and 2017, and the final Trends.Earth results compared to the results from this study (using land cover change only). Note that only the productivity trajectory is shown in the bottom left map for visual purposes as productivity performance and state showed no major changes)

The reduction in NDVI in areas where there has not been a land cover change is suggestive of more gradual loss in vegetation condition that might eventually become a more obvious form of degradation. These areas should ideally also be considered in stemming land degradation. Based on the result from Trends.Earth, this affects the entire study area, including some parts of protected areas. The latter finding is surprising and warrants further research.

6.1.4 Summary of land cover under the alternative scenarios

The study required modification of the 2017 land cover data to include better estimates of the extent of different types of land degradation for the 2017 Baseline, and from this, estimates of the extent of degradation in 2015 (the LDN Reference condition), and in 2030 under a Business-as-Usual scenario, as well as an estimate of a hypothetical Restored catchment in 2030.

Just over 555 000 ha, or 26% of the *remaining natural areas* of the Thukela catchment (i.e. not under cultivation, mining, human settlement or other infrastructure) were classified as degraded as at 2015 (Table 6.6). At least 270 000 of natural vegetation had changed to a woodier or less woody class by 2015 since 2005, thus representing some form of degradation of the original vegetation. There is limited spatial information on the condition of cultivated lands, and these were not considered in this study. The most dominant degraded areas were those classified as experiencing vegetation change (excluding IAPs), particularly loss of woody vegetation and canopy cover reduction. This is followed by "gully erosion" with nearly 60 000 ha or 11% of the degraded area and 2.1% of the catchment as a whole. The single largest classified degraded LULC class was degraded grassland (part of the degraded grassland area in Table 6.6) which accounted for 4% of the total catchment area and 20% of all degraded areas.

Table 6.6. Summary of degraded and denuded areas in the Thukela catchment. Areas are in hectares.

Degradation class	Degraded by 2015	Projected *further* degradation 2015 to 2030
Area with reduced vegetation cover		
as per land cover classification#	114 806	421
changes to less woody LC classes (ha, excl. IAPs)*	168 771	93 893
Old plantations, fields and rehabilitated mines	7 336	898
Bush encroachment (ha, excl. IAPs)*	104 053	97 771
Gully erosion (ha)	59 680	2 003
Areas impacted by light IAP invasion	87 058	-5 227
Areas impacted by medium IAP invasion	12 363	-1 024
Areas impacted by dense IAP invasion	2 032	25 881
TOTAL	**556 784**	**213 930**

* Since 2005

Degraded vegetation in terms of canopy cover loss

Dense wattle, gum and pine invaded areas occupy roughly 2 000 ha, which is just under half a percent of the total degraded area, with wattle being the most widespread genus, prolific across the entire catchment. Gums tend to be denser in the lower reaches and north-east of the catchment while dense stands of pines tend to occur in the south of the catchment. As there is little commercial silviculture and mining in the catchment, the areas classified as old plantation and rehabilitated mines is low.

Without intervention, it was estimated that the degraded areas would increase by some 213 930 ha by 2030 (Table 6.6, Figure 6.3). While the areas classified as degraded in the land cover were not projected to increase very much based on past trends, other forms of degradation were a significant threat. The extent of areas under dense infestations of IAPs is expected to increase from 0.1% in 2017 to 1% by 2030 as light and medium areas become denser, with the overall area impacted by IAPs with densities over 5% increasing from 3.8% to 4.2% (Figure 6.3). This only takes the three most common IAPs species into account. Almost 98 000 ha are expected to change to woodier land classes, while nearly 94 000 ha of woody vegetation loss is predicted to occur. There is also a substantial proportion of land which is expected to become eroded. In total, 27% of the *entire* catchment is predicted to be considered degraded to some degree by 2030, of which 7.5% of this is predicted to occur between 2015 and 2030 under the BAU scenario.

In the 2030 Restored scenario, degraded vegetation areas, erosion and invaded areas are effectively eliminated (Figure 6.3). This means natural grassland vegetation increases from 41.6% in the 2015 Baseline to 56.5% owing to the large areas of erosion and degraded grassland that are restored.

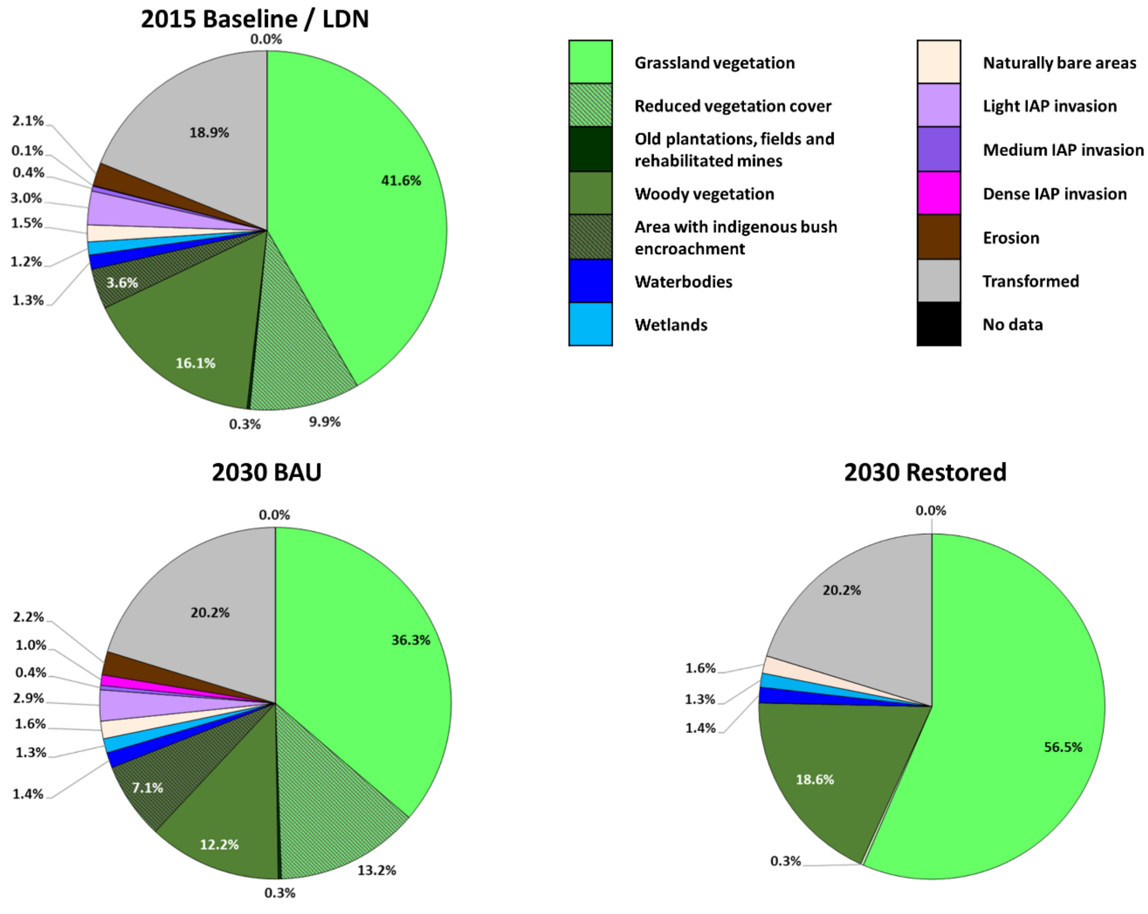

Figure 6.3. Proportion of different LULC classes under the three scenarios. Note that the 2015 Baseline approximates the LDN scenario at 2030, and that the grassland vegetation and woody vegetation areas do not account for vegetation changes prior to 2005.

6.2 Impacts on ecosystem services and values

Achieving LDN would maintain ecosystem service supply at 2017 baseline levels. The demand for certain services would change slightly though, as population grows and changes. If no additional investment is made, then the BAU scenario results in further degradation, resulting in a reduction in supply of most ecosystem services. Conversely if efforts are made to restore the catchment over the next decade, then there would be an increase in the supply of many ecosystem services. In this section we compare the outputs of ecosystem services under each scenario.

6.2.1 Water supply

Natural habitats exert a significant influence on hydrological regimes, both in terms of the overall amount of streamflow, and in terms of the temporal dynamics of streamflow through the year. In catchments like the Thukela, a major concern is reduction in water yield from the spread of IAPs. There is also concern that bush encroachment will reduce environmental flows, although this usually occurs lower in the catchment and is therefore not as strongly linked to yields.

According to the NIAPS data (Kotzé *et al.*, 2010) there are just over 81 000 condensed ha of IAPs in the Thukela catchment. Le Maitre *et al.* (2016) estimated that these reduce streamflow by some 100.87 Mm^3 per annum, 2.6% of the mean annual runoff (MAR). This equates to a unit area flow reduction of 1243 m^3/condensed ha. Similarly, Cullis *et al.* (2007) estimated that IAPs in the Thukela Catchment reduced MAR by about 98 Mm^3, and water yields by about 48 Mm^3 (Table 6.7), which could increase to a 261 Mm^3 decline in yield if the catchment were fully invaded (to 87.5% of remaining "invadable areas").

Table 6.7. Total reduction in surface water yields in the Thukela catchment, based on data in Cullis et al. (2007). Some of figures have been interpolated from the information provided in the paper.

2004	Catchment	Riparian	Total
Condensed hectares	94 000	17 400	111 400
Reduction in MAR (Mm^3)	20	77.6	97.6
Reduction in dam yields (Mm^3)	4.2	16.3	20.5
Reduction in run of river yield (Mm^3)	5.6	21.7	27.5
Total reduction in yield Mm^3	9.8	38	48
Registered use Mm^3			390

Using the estimated condensed ha of IAPs in the catchment under each scenario, unit area flow reduction estimate for the Thukela catchment from Le Maitre *et al*. (2016) and the predicted change in yield associated with MAR estimates from Cullis *et al.* (2007), who estimated changes in yield due to alien plants in the Thukela catchment to be around 49% of the reduction in MAR, we calculated predicted yield increases of 15.5 Mm^3 and 64.3 Mm^3 under the LDN and Restored scenario respectively, relative to BAU. The clearing of an additional 80 107 condensed ha of IAPs through full restoration gives rise to an additional yield of 48.8 Mm^3.

These avoided losses of yield were costed using the municipal sales price of water in the Thukela WMA. The avoided losses were estimated to be R171 million under the LDN scenario and R709 million under the Restored scenario in 2030 (in 2020 Rands).

6.2.2 Erosion control

Erosion and sedimentation within watersheds can become a major issue as it causes structural damage to reservoirs, flooding, affects the quality of drinking water and increases water treatment and maintenance costs at water treatment works. Vegetation can reduce erosivity by stabilising soils and intercepting rainfall, thereby preventing erosion. Vegetated areas also capture the sediments that have been eroded from agricultural and degraded lands and transported in surface flows, preventing them from entering rivers. In KwaZulu-Natal, soil erosion is a serious problem in many of the upper catchment areas and on communal land, and in the Thukela catchment there are a number of areas that are particularly vulnerable and susceptible to erosion which has resulted in the loss of natural vegetation cover and the formation of dongas.

The average sediment yield for the Thukela catchment was estimated to be 0.59 t/ha/y (0.01-5.21 t/ha/y) under the LDN scenario (Table 6.8). Under the BAU scenario with increased levels of degradation, these values increase to 0.68 t/ha/y (0.01-6.05 t/ha/y). Full restoration of the catchment would result in a decrease in average sediment yields when compared to the BAU and would also result in lower yields than those estimated under the LDN Scenario (0.52 t/ha/y, 0.01-5.05 t/ha/y). The total amount of sediment exported from the catchment amounts to 2.29 million tonnes under the LDN scenario, increasing to more than 2.65 million tonnes under the BAU Scenario, a difference of over 362 000 tonnes (Table 6.8). A fully restored catchment is estimated to generate a total of 2.06 million tonnes of sediment per year, which is retaining approximately 588 000 tonnes of sediment more than the BAU landscape, essentially preventing this amount of sediment from reaching rivers and dams, and downstream environments.

There is evidence showing that ground cover plays a significant role in controlling the rates of runoff and sediment loss in grassland and savanna landscapes (Bartley *et al.*, 2006). The degradation of grassland through overgrazing of livestock exposes the soil layer which leads to increased soil damage and erosion. Furthermore, trampling by livestock leads to soil compaction, which in turn increases soil surface erosion, especially via water runoff. Therefore, bush encroachment related to anthropogenic changes to fire regimes or elevated levels of CO_2 will have less of an impact on sediment retention, whereas increased woody cover as a result of overgrazing will result in increased soil erosion as overgrazing drives both of these processes (Turpie *et al.*, 2019). Our results show that while accelerated erosion is usually a result of decreased vegetative cover, it is also likely to be higher in situations of bush encroachment (i.e. the BAU) than under more natural conditions.

Table 6.8. Sediment yields (t/ha/y), total sediment exported (t/y), sediment retention (t/ha/y) and total value (R million/y) in the Thukela catchment under each of the three scenarios.

	BAU (2030)	LDN (2030)	Restored (2030)
Average sediment yield (t/ha/y)	0.68	0.59	0.52
Total sediment exported (t/y)	2 652 951	2 290 841	2 065 663
Sediment retention (t/ha/y)	9.71	9.80	9.86
Total value R million/y (2030)	286.6	289.3	290.9

The value of erosion control, valued as the replacement cost of lost storage capacity, was estimated to be R291 million in 2030 for the Restored scenario and R289 million for the LDN scenario. The value of erosion control under the BAU was less than 2% lower than the Restored Scenario at R287 million in 2030 (2020 prices). The average per ha value was R73 (R1-R638) for the Restored scenario compared to R72 per ha for the LDN scenario (R1-R639), and R71 (R1-R636) per ha for the BAU. This is due to some increase in erosion under the BAU as well as impacts of bush encroachment where overgrazing has resulted in soil erosion. A spatial analysis of erosion control in KwaZulu-Natal as part of the pilot accounts, found that the upper sub-catchments of the Thukela catchment are particularly important for retaining sediments (Turpie *et al.*, 2020b).

6.2.3 Carbon storage

All ecosystems make contributions to global climate regulation via the sequestration and storage of carbon. This is especially true for natural densely vegetated systems but even cultivated and highly altered land cover contributes through belowground biomass and SOC. Degradation of natural ecosystems and a loss of woody biomass, carbon-rich soil and peat releases carbon and contributes to global climate change with impacts on biodiversity, water supply, droughts and floods, agriculture, energy production and human health (IPCC, 2007), whereas restoration or protection of these habitats mitigates or avoids these damages, respectively.

Reducing degradation thus has a net benefit in terms of avoiding and mitigating damage caused by impacting the carbon balance. That said, while bush encroachment is a severe form of ecosystem degradation, there is consensus that it can be positively seen in terms of being beneficial to carbon sequestration and enhances woody biomass provisioning, with gains of between 4.3-28.5 t/ha in bush encroachment in areas affected by bush encroachment in South Africa (Turpie *et al.*, 2019). At the same time, woody IAPs expansion may also be seen beneficially in terms of carbon sequestration and storage, with enhanced costs imposed by removing IAPs (Hassan & Mahlathi, 2020). The net social and environmental benefits of carbon storage by both IAPs and expansion of native woody vegetation is widely considered to be outweighed by the negative biodiversity, hydrological and tourism impacts (Turpie *et al.*, 2019; Hassan & Mahlathi, 2020).

Total ecosystem carbon was estimated as 354 Tg C under the LDN scenario, which equates to approximately 1 299 Tg CO_2 (using molecular weight of CO_2/molecular weight of carbon; United States Environmental Protection Agency, 2015). Under the BAU scenario the total ecosystem carbon was estimated as 357 Tg C, which equates to approximately 1 310 Tg CO_2. This therefore shows a slight increase in carbon stocks. This is likely due to the increased woody vegetation density due to bush encroachment and areas densely invaded by IAPs, particularly pine and gum encroached areas. The biggest single increase is in the large expansion of dense thicket & bush area between 2017 and 2030 owing to modelled bush encroachment while the biggest losses are in reduced stocks in modelled grassland areas, including those with light IAP invasions, followed by substantial areas woody vegetation declines in the reduction of medium bush areas. In the 2030 Restored scenario the total ecosystem carbon was estimated as 363 Tg C, which equates to approximately 1 333 Tg CO_2. The total tonnes of carbon are summarised in Table 6.9.

Table 6.9. Total estimated ecosystem carbon (Tg C = million tonnes of carbon) under the LDN, BAU and Restored Scenarios

	BAU 2030	LDN 2030	Restored 2030
Natural Vegetation	271.8	272.3	307.8
Degraded Natural Vegetation	14.6	14.5	1.0
Cultivated areas	54.4	54.4	54.4
Areas impacted by IAPs	16.3	12.8	0
Total	**357.1**	**354.0**	**363.2**

Further deterioration as modelled in the 2030 BAU Scenario appears to result in slightly higher carbon storage by 2030, with annual gains of roughly 233 770 tonnes relative to the LDN scenario. Under the 2030 Restored Scenario, 6.12 Tg C of carbon is gained in the catchment area relative to the BAU as large areas of grassland are restored (increasing total grassland storage by 37.1 Tg C). This exceeds the carbon lost from reduction in dense thicket & bush and the reduction in woody IAP-impacted areas which are reduced to nil, often to land covers with lower levels of total ecosystem carbon storage.

Using an average SCC value based on Nordhaus (2017) and Ricke et al. (2018), in 2030, the retained carbon stocks of 357 Tg under the BAU, 354 Tg under the LDN scenario and 363 Tg under the Restored scenario had an annualised global value of some R261 billion, R259 billion and R266 billion per year of which national benefits amount to R2.06 billion, R2.04 billion and R2.11 billion per year, respectively.

6.2.4 Livestock production

Livestock farming is an important livelihood activity, both on private and communal land, across large areas of the Thukela catchment. Indeed, approximately 30% of the communally owned and 45% of the commercially owned large stock units (LSUs) within KwaZulu-Natal are farmed in the Thukela catchment. This is unsurprising given the vast areas of grassland available for forage.

On communal land, the total amount of grazing land increased by just under 2% for the LDN scenario when compared to the BAU and under a fully restored catchment increased by over 6% when compared to the BAU (Table 6.10). The resource rent value of communal livestock production in the Thukela catchment was estimated to be R322.5 million under the LDN scenario, decreasing to R301.9 million under the BAU as a result of the loss and degradation of rangeland areas. Under a fully restored catchment, the value increases by just under R47 million to R348.5 million relative to the BAU.

There was a smaller change in the amount of grazing land lost and gained on commercial land across the three scenarios (Table 6.10). Grazing land increased by 1.6% under the LDN scenario compared to the BAU and under a fully restored catchment increased by 2.2%. This resulted in a difference in the value of livestock production of some R18 million under the LDN and R46 million under the Restored scenario relative to the BAU. The total resource rent value of commercial and communal livestock production in 2030 was R826 million under the BAU, increasing to R865 million and R918 million under the LDN and Restored scenarios, respectively. This is a difference of some R39 million per year under the LDN scenario and R92 million per year under the Restored scenario relative to the BAU.

In physical terms, the number of livestock (based on LSU/ha in the base year and adjusted to the total grazing area) was estimated at just over 496 000 under the BAU Scenario, increasing to over 534 000 under LDN and further to just over 571 000 under the Restored Scenario. Communal livestock numbers are affected to a greater extent under the BAU where there are 22.7% the total number of LSU in communal areas compared to commercial areas, which increases to 24.5% under LDN and 28.3% under the Restored Scenario.

These results demonstrate how even a small loss in grazing land can have a significant impact on livestock production. Poor rangeland management in addition to changes in traditional cultural norms and practices, particularly within communal areas, has over the past few decades resulted in declining livestock numbers. Indeed, the DAFF quarterly livestock statistics show that in KwaZulu-Natal cattle have decreased by 20%, sheep by 33% and goats 16% between 1996 and 2018. The number of heads of livestock of these animals declined from roughly 5.1 million to 3.9 million in that time. This has had a significant impact on those farmers that depend on livestock production as their main livelihood activity. This trend continues under the BAU scenario as more rangeland areas become degraded and encroached by woody vegetation. Restoring degraded rangeland or preventing any further loss through improved rangeland management can have significant positive impacts on grazing capacity and associated livelihoods.

Table 6.10. The extent of grazing land within the Thukela catchment and the total value of livestock production in 2030 (R million, 2020) under each scenario.

	BAU 2030	**LDN 2030**	**Restored 2030**
Communal grazing land (% of catchment)	20.6%	22.4%	26.8%
Commercial grazing land (% of catchment)	31.4%	33.1%	33.6%
Communal livestock production (LSU/y)	91 840.2	104 987.4	126 189.1
Commercial livestock production (LSU/y)	404 750.0	429 174.4	445 236.0
Total livestock production (LSU/y)	**496 590.2**	**534 161.4**	**571 425.1**
Communal livestock production (R million)	301.9	322.6	348.5
Commercial livestock production (R million)	524.1	542.2	569.6
Total livestock production (R million)	**826.0**	**864.7**	**918.1**

6.2.5 Harvested resources

Millions of South Africans harvest wild foods, medicines, raw materials such as thatch and building poles, and fuel for both subsistence and commercial use, especially in rural areas where economic opportunities are limited. The use of these resources depends on household income, and the availability and cost of alternatives. Within the Thukela catchment, only 30.3% of the land falls within communal areas, which is limited to the lower catchment areas around Nkandla and small parts of the upper catchment around the Southern Drakensberg. Since natural resources are generally much more available for subsistence harvesting in communal areas, changes in the availability of these resources are more strongly dependent on resource stocks in these regions. Under the LDN scenario it was estimated that 370 000 m^3 of woody resources were used across the catchment, the bulk of which was firewood (352 700 m^3), followed by poles (15 600 m^3) and timber (1 700 m^3). In terms of non-woody plant resources, we estimated that 13 300 t of thatching grass and 10 050 t of plant foods and medicines were harvested. Lastly, we estimated that 860 t of wild meat were harvested across the catchment.

The impact of the various scenarios on quantities of harvested natural resources varied according to the resource in question. We predicted that harvesting of firewood would be highest under BAU (Table 6.11). Conversely, we predicted a decrease in the availability of firewood under the restored scenario, relative to LDN and BAU. The increase in firewood under BAU is not surprising, given the net increase in woody vegetation occurring in the catchment, from both indigenous bush encroachment and invasive alien plants. Reversing these processes would thus reduce the availability of firewood, as the coverage of woody habitats is decreased. Timber shows a different trend, with highest harvesting predicted under the Restored scenario, followed by LDN and then BAU. This reflects the fact that wattle, and to a lesser extent gum, are not suitable for timber harvesting. Hence, timber stocks would be expected to decline where these species continue to replace indigenous forest and woodland habitats under BAU. In the restored scenario, invasions of woodland and forest are reversed, while degraded woodland and forest habitats are rehabilitated. Both of these processes contribute to the higher timber stocks predicted under the restored scenario. Additionally, while indigenous bush encroachment might increase stocks of firewood and poles under BAU, the spread of bushland and thicket makes little contribution to timber stocks, since few to no large trees are associated with these habitats.

Table 6.11. The total estimated amount of resources harvested per year from the Thukela catchment under each of the three scenarios.

Resource	Unit	BAU 2030	LDN 2030	Restored 2030
Fuelwood	m^3	394 635	352 754	334 852
Poles	m^3	14 714	15 580	15 108
Timber	m^3	1 583	1 723	2 205
Thatching grass	t	12 485	13 326	15 973
Wild plant foods & medicines	t	8 867	10 046	11 420
Wild meat	t	784	860	1 084

Stocks of non-woody plant resources were predicted to increase with restoration of the catchment (Table 6.11). Thatching grass harvesting was predicted to increase significantly with restoration, while stocks would decline relative to LDN under BAU. This reflects the clearance of indigenous bush encroachment and alien species from naturally grassy habitats, promoting the expansion of grassy habitats. Additionally, the restoration of degraded grassland and woodland would increase thatching grass stocks in these habitats. Conversely, thatching grass stocks would decline further under BAU, due to unchecked indigenous bush encroachment and alien invasion of grassy habitats, and continuing degradation of grassland and woodland habitats.

We similarly predicted an increase in harvesting of wild plant foods and medicines with restoration under both LDN and Full Restoration (Table 6.11). Alien plants are generally not used for food or medicine, so increased stocks of these resources would be expected where indigenous vegetation replaces alien species. Restoration of degraded natural vegetation types would also increase stocks of plant foods and medicines under the restored scenario. Harvesting of wild meat was also predicted to increase with restoration of the catchment. Alien invasion reduces the suitability of habitats for most wild animal species, which are adapted to indigenous vegetation. Thus, full alien plant clearance and restoration of degraded habitats again increases stocks of harvested wildlife species under LDN and Full Restoration.

Provisioning of wild resources was estimated to be worth some R1.60 billion under the LDN and R1.74 billion under the Restored scenario, which is higher than the value of R1.59 billion under the BAU (in 2020 prices, Table 6.12). The higher values under the LDN and Restored scenarios compared to the BAU is due to the restoration and rehabilitation of degraded areas which would then generate more output, particularly non-woody resources, as a result of more grassland areas.

Table 6.12. The total value (R millions) of harvested resources for each scenario in 2030.

Resource	BAU 2030	LDN 2030	Restored 2030
Fuelwood	688.7	615.7	584.4
Poles	21.5	22.7	22.0
Timber	4.3	4.7	6.1
Thatching grass	605.3	646.1	774.4
Wild plant foods & medicines	249.6	282.4	321.8
Wild meat	23.8	26.1	32.8
Total	**1 593.2**	**1 597.6**	**1 741.5**

6.2.6 Tourism

Encompassing all tourist activities related to nature, nature-based tourism is an important component of the overall tourism sector in KwaZulu-Natal. Activities include visits to national parks, nature, game and forest reserves, and outdoor activities such as hiking and trekking, mountain-biking and birdwatching. In the Thukela catchment there are a number of important protected areas that are popular tourist destinations, including uKhahlamba Drakensberg Park, a UNESCO World Heritage Site comprising several wilderness areas, and the numerous state- and privately-owned game and nature reserves such as Chelmsford Nature Reserve and Nambiti Private Game Reserve.

The resource rent value of attraction-based tourism in KwaZulu-Natal in 2017 was R672 million (in 2020 Rands). The tourism value was spatially allocated in proportion to photo density (using the density of geotagged photos uploaded to Flickr) and apportioned based on land cover data (2017 Baseline). Based on this, the Thukela catchment accounted for R101 million (15% of KwaZulu-Natal's contribution). Further analysis of the spatial pattern suggested that the value of natural areas within the Thukela catchment was some R79 million in 2017 (2020 Rands), i.e. accounting for about 78% of the attraction-based tourism value in the catchment. Approximately R35 million, or 45%, of the nature-based tourism value is found within protected areas.

The uKhahlamba Drakensberg Park is by far the most valuable of the protected areas overall, contributing 87% of this value. While the uKhahlamba Drakensberg Park is vast, comprising several wilderness areas that are difficult to access and thus attract few tourists. Three protected areas had a higher mean tourism value per hectare than the uKhahlamba Drakensberg Park: Robinson's Bush Nature Reserve, Isandlwana Heritage Reserve and Bill Barnes Crane and Oribi Nature Reserve. Most of the nature-based tourism value fell within the grassland biome which is the dominant biome within the main protected areas of the Thukela catchment (Figure 6.4). Outside of the protected areas, most of the nature-based tourism value can be found around the Lower Drakensberg Foothills from Bergville to Winterton, Estcourt and Mooi River.

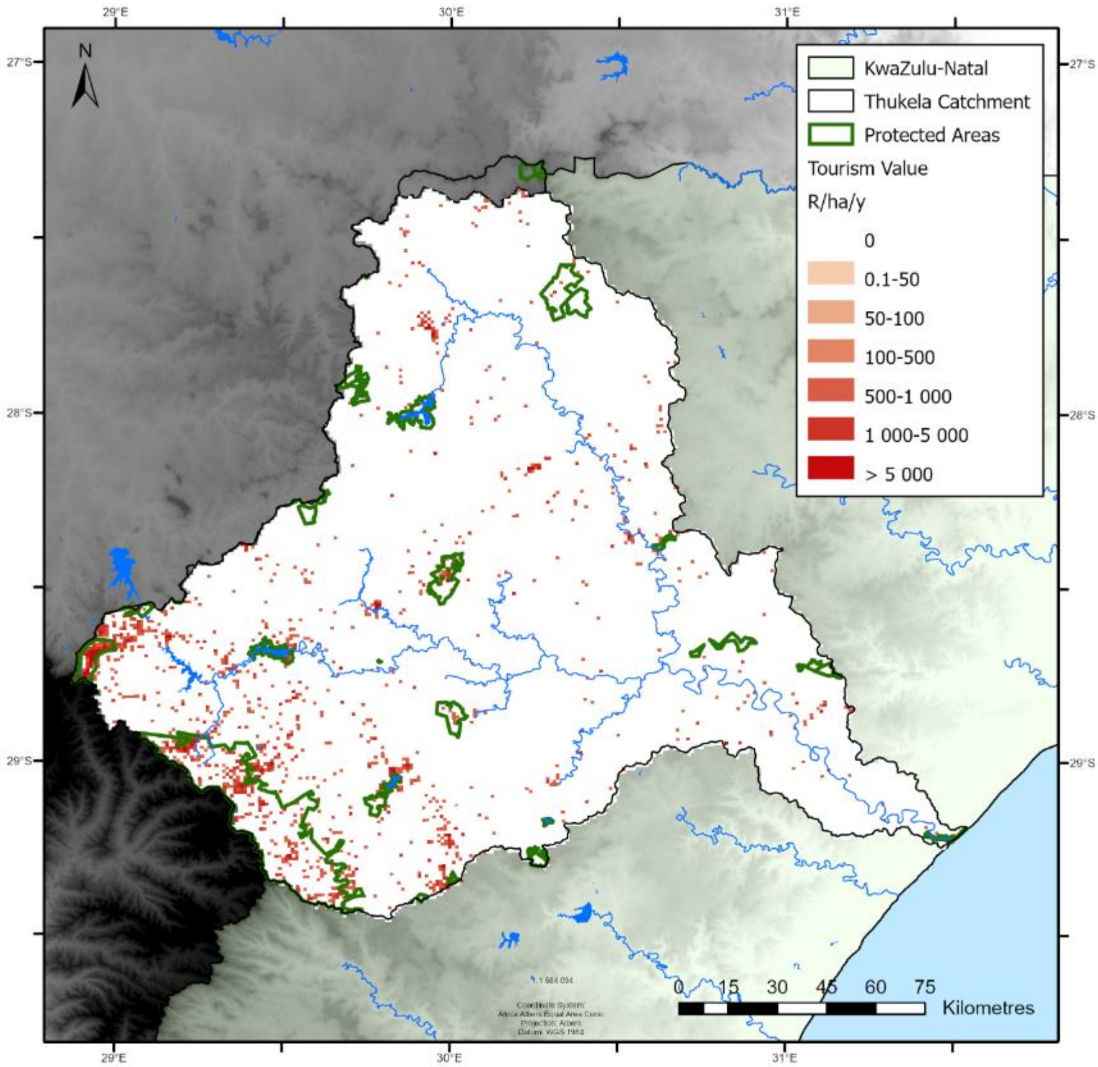

Figure 6.4. Nature-based tourism value for the year 2017 across the Thukela catchment based on the distribution of geo-referenced photos uploaded to Flickr and the location of protected areas across the catchment.

Note that the resource rent (an accounting measure of the value of the ecosystem service) is very much smaller than the direct and total contribution to GDP of the tourism dependent on these attractions, and also does not include the domestic and international consumer surplus (another element of welfare value) that is generated by this tourism.

KwaZulu-Natal remains a top destination for domestic travel,[28] but it has experienced a reduction in the number of foreign visitors (and associated total foreign direct spend) over the

[28] While domestic visitor numbers have also decreased over recent times, this appears to be a trend across the country and not just in KwaZulu-Natal. This decline is likely due to fewer South Africans holidaying as a result of poor economic growth and the generally tough economic climate in the country.

past decade (Figure 6.5). While there appears to be some recovery over the last three years, total numbers remain relatively stable and total spend is still significantly lower than it was some ten years ago. This is surprising given that South Africa has experienced overall growth in tourism over the same time period (Figure 6.6).

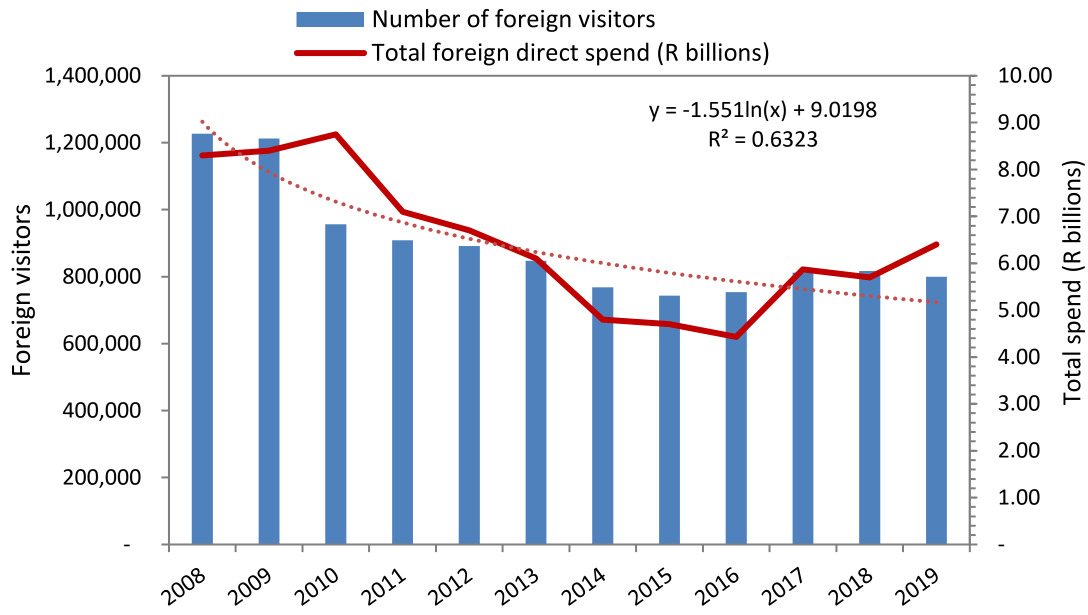

Figure 6.5. The number of foreign visitors and total foreign direct spend (R billions) in KwaZulu-Natal from 2008-2019. Source: Department of Tourism, KwaZulu-Natal Tourism.

This suggests that foreign tourists are opting to visit other provinces, such as the Eastern Cape, Mpumalanga and Limpopo, which may offer similar attractions, rather than choosing to visit KwaZulu-Natal as part of their itinerary to South Africa. Reasons for this decline in visitation appear to be largely unrelated to changes in the environment but rather the result of changes in tourist preferences as well as clear failure in the management of various aspects of the conservation areas in KwaZulu-Natal by EKZNW.[29] Once rated as a leading wildlife conservation organisation globally, it is now in disarray following turmoil at executive-level and upper-management, accusations of nepotism, fraud and financial misconduct, which have resulted in poor service filtering down and reduced operational budgets which is apparent in the decline in quality and maintenance of tourist offerings (infrastructure, facilities and accommodation) within in their parks and reserves. As a result, tourists may be choosing to visit parks and reserves in other provinces where management and tourist facilities are in line with what foreign tourists are expecting or offer better value for money.

[29] See recent news articles: KZN's sinking conservation ship: Part's 1 and 2, and SCOPA blasts Ezemvelo.

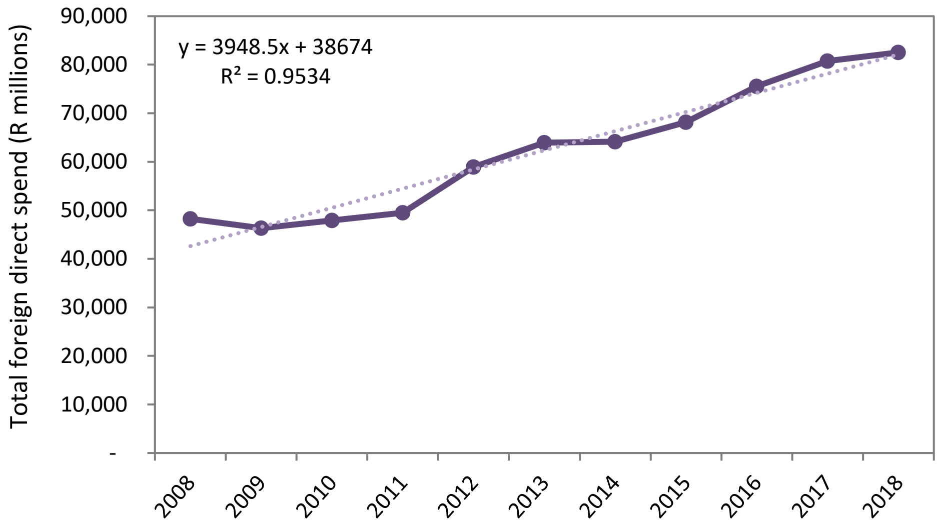

Figure 6.6. Total foreign direct spend (R million) in South Africa from 2008-2018. Source: Department of Tourism.

Continued degradation of the remaining natural areas in the Thukela catchment under the BAU scenario could further reduce the tourism potential in conservation areas. There is evidence to suggest that not only does bush encroachment have a significant impact on wildlife numbers and densities (particularly grazers such as blue wildebeest, zebra, buffalo, eland and sable), but also impacts on visitor numbers to conservation areas and their overall satisfaction (Gray & Bond, 2013; Arbieu *et al.*, 2017; Dube & Nhamo, 2020). When natural grassland or savanna landscapes become encroached, they become less suitable for a number of wildlife species and game viewing is negatively affected due to lower visibility (Arbieu *et al.*, 2017). The results of a survey of visitors to Hluhluwe-iMfolozi Park have shown that woody plant encroachment does influence visitor experience, in particular foreign visitors, and that a large proportion of potential future visitors to the Park could be lost if animal viewing becomes more difficult (Gray & Bond, 2013). Arbieu *et al.* (2017) found that when shrub cover reached 30%, attitudes were found to become negative with respect to game-viewing, particularly at Hluhluwe-iMfolozi Park. At the same time, little to no vegetation cover was also perceived negatively. Dube *et al.* (2018) found that visitors to protected areas are worried about climate change impacts (with many areas of bush encroachment being in part due to climate change, (Sankaran *et al.*, 2005); Eamus & Palmer (2007)), and confirmed that many visitors would likely adapt their travel plans based on perceived impacts on their enjoyment, rather than accepting it as a natural process nonetheless. While maintaining open vegetation in game-viewing areas is recommended by some authors, the cost of managing bush encroachment in protected areas can be prohibitively expensive, especially if density is beyond a certain threshold. Protected area managers may need a unique funding mechanism whereby it may, as an example, be included in visitor fees, as ensuring a better visitor experience free of a problem that is no fault of the management authority.

Not only will continued degradation have an impact on existing conservation areas but it will limit any opportunity for developing the wildlife sector in areas that would otherwise be suitable for wildlife conservation. As part of the Wildlife Economy, these areas have significant potential to contribute to socio-economic development by supporting and enabling emerging wildlife ranchers in entering the sector, particularly in rural, former homeland areas. Indeed, if efforts are made to restore the catchment over the next decade then there is great potential to expand the wildlife sector in the Thukela catchment. Empowering emerging wildlife farmers and entrepreneurs is recognised as a key Wildlife Economy intervention to facilitate transformation within the wildlife sector and there has already been significant investment by the South African government to achieve such outcomes[30]. The growth goal set out in South Africa's National Biodiversity Economy Strategy (NBES) states that by 2030 the South African biodiversity economy will achieve an average annualised GDP growth rate of 10% per annum and is aligned to the National Development Plan, Vision 2030.

Therefore, under the BAU Scenario we assume that the growth in tourism is constrained over the next decade, considering the impacts of degradation on existing conservation areas and the lack of opportunity to develop the wildlife sector further as a result of the loss of wildlife suitable areas across the catchment (Table 6.13, Figure 6.7). Conversely, we expect that restoration of the catchment and protection of existing conservation areas could have a positive impact on tourism in the catchment. The restoration of already-degraded areas and the protection of existing natural vegetation could provide significant opportunity for developing the wildlife sector further in this part of the province.

Table 6.13. The estimated resource rent value of nature-based tourism in the Thukela catchment in 2030 for the LDN, BAU and Restored Scenarios. All values are R millions, 2020 Rands.

	BAU 2030 R million	LDN 2030 R million	Restored 2030 R million
Nature-based tourism value Thukela catchment	85.3	94.7	104.2

[30] See the National Biodiversity Economy Strategy (NBES) published in 2016.

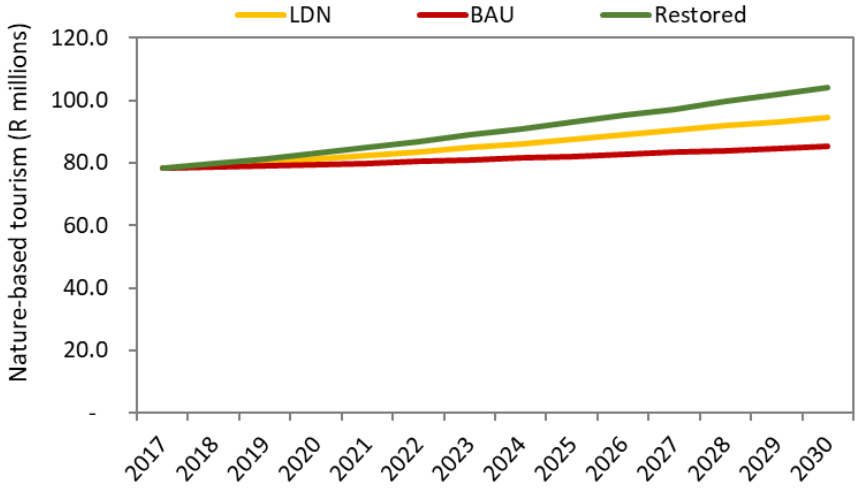

Figure 6.7. Forecasted nature-based tourism value (R millions) in the Thukela catchment for the LDN, BAU and Restored Scenarios 2017-2030.

6.2.7 Summary

A summary of the ecosystem service values in both physical and monetary terms for the BAU, LDN and Restored Scenarios is provided in Table 6.14.

Table 6.14. The total biophysical supply and total value (R millions in 2030) for each of the ecosystem services per scenario.

Biophysical supply	**BAU 2030**	**LDN 2030**	**Restored 2030**
Yield increase (Mm^3 relative to BAU)		16	64
Sediment retained (t/ha/y)	10	10	10
Ecosystem carbon (Tg C)	357	354	363
Livestock production (LSU/y)	496 590	534 161	571 425
Wood products (m^3)	410 932	370 057	352 165
Non-wood products (t)	22 136	24 232	28 477
Nature-based tourism value (R million)	85	95	104
Value (R millions in 2030)	**BAU 2030**	**LDN 2030**	**Restored 2030**
Water supply (relative to BAU)		171	709
Erosion control	287	289	291
Carbon storage (global)	261 317	259 093	266 006
Carbon storage (national)	2 064	2 047	2 101
Livestock production	826	865	918
Wood products	689	616	584
Non-wood products	22	23	22
Nature-based tourism value	85	95	104

6.3 The costs of SLM and restoration interventions

6.3.1 Clearing invasive alien plants

Achieving LDN or full restoration by 2030 would require undertaking clearing and follow up treatments over the period from 2021 to 2030. The hypothetical extremes are to undertake all the clearing in the first or last year. The latter choice would mean bearing the costs later but would involve tackling a much larger problem. The costs of these extremes are shown in Figure 6.8 for achieving LDN, along with the most likely pattern of clearing, which is to take a phased approach that requires a relatively constant cash flow and level of employment. Expenditure breakdown of the phased approach is given in Figure 6.9. This pattern also requires the lowest overall capital input for both restoration scenarios (Table 6.15). Note that IAPs cannot be reduced through SLM, and so the costs are the same for both LDN scenarios.

These figures are derived using a threshold of 5% change in density by 2021 relative to the Reference year (2015). In other words, it assumes that action is only taken when the increase in density was greater than 5 percentage points. Setting a zero threshold would lead to a five-fold increase in costs and would be inefficient. Indeed, this would incur a similar cost to the full restoration of the catchment, i.e. removal of all IAPs in areas where they occur at densities of greater than 5%, which would cost an estimated R546 million if implemented immediately. This is subtly different, but would be a better option in terms of managing future spread.

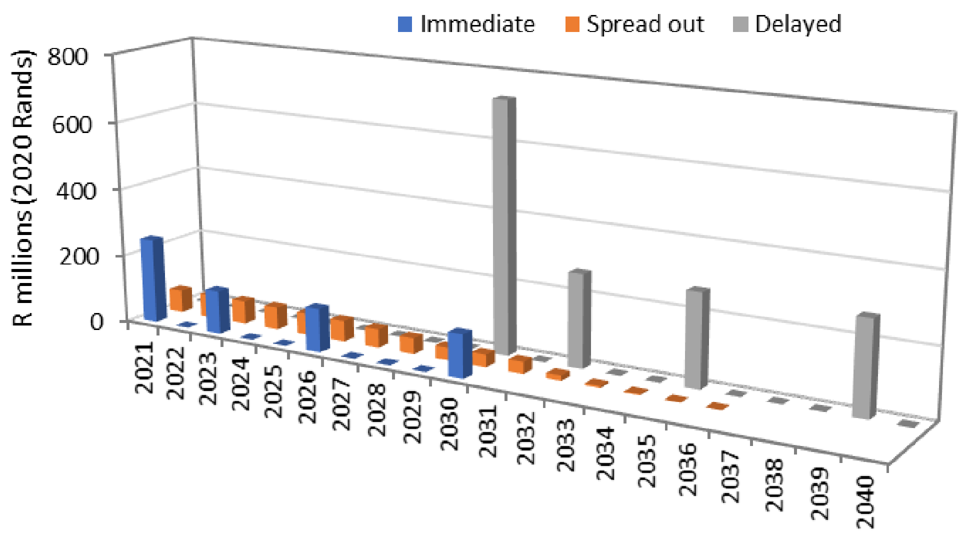

Figure 6.8. Two extreme options and a phased option to achieve LDN by 2030. Note that the graph includes necessary follow up action into the future required to secure the results.

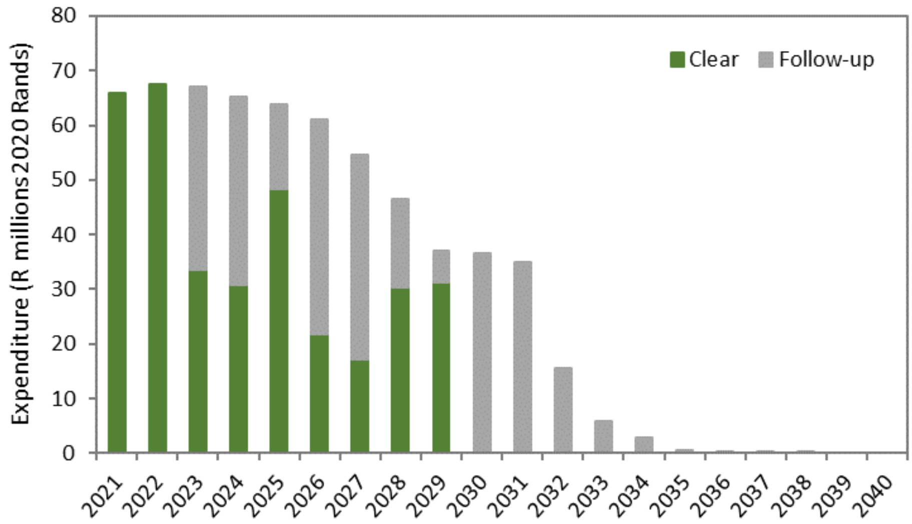

Figure 6.9. Smoothed expenditure on clearing and follow up in an unequal 10-phased clearing plan with clearing in year 1 and follow ups at years 3, 6 and 9. Treatment area in each phase diminishes over time.

Table 6.15. Net present value (R millions in 2020 Rands) of the investment required for clearing IAPs under different timings of intervention, based on costs over 20 years and a discount rate of 3.66%.

	Area (con ha)	Immediate	Phased	Delayed
LDN	25 419	545.1	514.4	980.4
Full restoration	105 526	2 388.1	2 355.2	2 427.4

6.3.2 Addressing bush encroachment

Achieving LDN by 2030 would require undertaking restoration activities and preventative action over the period from 2021 to 2030 that reduce the encroached bush back to levels recorded in 2015. The costs here refer to manual clearance, which we assumed is required to reverse moderate to severe forms of bush encroachment. For the optimistic LDN scenario, the interventions were designed to clear woody encroachment that had occurred from 2015-2021, assuming that the SLM measures (see below) would stop any further encroachment. Under the pessimistic LDN scenario, interventions were designed to clear the same area as all projected degradation from 2015-2030, under the assumption that SLM will not make any measurable difference. We also experimented with the timing of the clearing action. As at 2021, 44 781 ha needed treatment under the optimistic LDN scenario, at a cost of about R237.6 million whereas under the pessimistic LDN scenario, 95 571 ha would need treatment (to account for expected further encroachment as well), at a cost of R507.2 million (Table 6.16). Under the Restoration scenario, 130 241 ha would need to be cleared at a cost of R691.1 million. These costs assume a phased approach. In this case a phased approach would be some 25% more expensive than immediate action because of the rapid rate of encroachment. However, it is more realistic in terms of cash flow and availability of resources.

Table 6.16. Net present value (R millions in 2020 Rands) of the investment required for addressing bush encroachment under different timings of intervention, based on costs over 20 years and a discount rate of 3.66%.

	Area (ha)	Immediate	Spread
LDN pessimistic	95 571	406.6	507.2
LDN optimistic	44 781	190.5	237.6
Full restoration	130 241	554.1	691.1

6.3.3 Sustainable land management

In order to prevent further degradation as a result of overgrazing (e.g. loss of grassland productivity, erosion or bush encroachment), as well as to maintain the condition of areas where bush encroachment had been cleared, it was assumed in the optimistic LDN scenario that SLM could be achieved with a combination of improved extension services and non-monetary and monetary incentive measures including stewardship programmes and payments for ecosystem services (PES). To achieve LDN, the total area targeted would be the 53 771 ha threatened with degradation between 2021-2030, at a cost over 25 years of R2.0 billion. Under a full restoration scenario, this would be a much larger area of 335 309 ha (including follow up of a more extensive bush encroachment effort), at a cost of R6.1 billion (Table 6.17).

Table 6.17. Net present value (R millions in 2020 Rands) of the investment required to achieve sustainable land management under the LDN and Restored scenarios. The investment includes the cost of increasing the number of extension officers in the catchment as well as implementing PES and stewardship programmes over a 25-year period (2021-2045).

	Total area (ha)	Investment (R million)
LDN optimistic	53 771	1 981.0
Full restoration	335 309	6 093.6

6.3.4 Active restoration of rangelands through reseeding and exclusion

Under the pessimistic LDN scenario it was assumed that SLM would not be successful in preventing further degradation in the catchment, which would require restoring an area equivalent to all projected degradation from 2015-2030. This would entail active restoration of all degraded areas through reseeding and replanting as well as protection of these areas through reduction of livestock or managed wildlife densities where necessary. The per hectare cost of restoring denuded and degraded rangeland areas was estimated to R12 500 (2000 Rands), based on estimates from Mander *et al.* (2007). The total area that would need to be actively restored under the LDN pessimistic scenario would be 53 771 ha, at a cost of some R673 million. Maintenance costs were estimated to be 20% of the implementation cost, to be carried out until 2030.

6.3.5 Summary

A summary of the main land degradation issues in the Thukela catchment and the assumptions made for each under each of the scenarios in determining the total areas for costing are shown in Table 6.18.

*Table 6.18. The five main land degradation issues targeted in the Thukela catchment, the assumptions made for each and the total area targeted under the LDN pessimistic, LDN optimistic and Restored scenarios. *condensed hectares.*

Problem	Assumptions	Area (ha)
LDN pessimistic		
Degraded/denuded grassland, erosion, gullies & dongas	All areas between 2021-2030 to be actively restored through reseeding/replanting and protection through exclusion of herbivory	53 771
Light BE <25%	Clear 2015 to 2030	7543
Med/dense BE >25%	Clear 2015 to 2030	88 208
IAPs	Clear all IAPs >5% between 2015-2021 + follow up	25 419*
LDN optimistic		
Degraded grassland	All areas between 2021-2030 to be managed through SLM	243
Erosion, gullies & dongas		1 204
Light BE <25%		7 543
Med/dense BE >25%	Clear 2015 to 2021, + SLM	44 781
IAPs	Clear all IAPs >5% between 2015-2021 + follow up	25 419*
Full restoration		
Degraded grassland	All degraded areas as at 2030 (prior to 2021 too) to be managed through SLM	107 120
Erosion, gullies & dongas		61 687
Light BE <25%		53 770
Med/dense BE >25%	Clear 2005 to 2021 +SLM	130 241
IAPs	Clear all IAPs >5% as at 2021 + follow up	105 526*

6.4 Cost-benefit analysis

The estimates derived in this study suggest that the implementation of restoration interventions and SLM in the Thukela catchment would result in a net benefit under the optimistic LDN scenario and with full restoration. Using a discount rate of 3.66%, the net present value over 25 years was estimated to be R435.5 million and R6 389.6 million, respectively for optimistic LDN and Restored scenarios (Table 6.19). The higher costs under the pessimistic LDN scenario resulted in a net loss. This was further tested under varying assumptions of costs, benefits and discount rate (Table 6.20). This result confirms that SLM is more cost effective than active restoration and that preventing further degradation from happening is cheaper than restoring already degraded areas.

Table 6.19. Present value of the costs of interventions and ecosystem service benefits relative to BAU under the LDN and Full Restoration scenarios (2020 R millions, 3.66% discount rate, 25 years).

	Present value (R millions) base estimate		
	LDN Scenario		
Costs	Pessimistic (Upper bound costs)	Optimistic (Lower bound costs)	**Full Restoration Scenario**
Clearing IAPs	514.4	514.4	2 355.2
Addressing Bush Encroachment	507.2	237.6	691.1
Active restoration of grasslands, erosion	2 623.6	-	-
Sustainable land management	-	1 981.02	6 093.62
Total present value of costs	**3 645.18**	**2 733.09**	**9 139.98**
Benefits			
Water supply	2 591.4	2 591.4	10 757.2
Sediment retention	38.9	38.9	63.1
Tourism	121.8	121.8	243.6
Carbon storage (avoided national cost)	-274.91	-274.91	597.5
Harvested resources	70.6	70.6	2 391.3
Livestock production	620.7	620.7	1 476.9
Total present value of benefits	**3 168.6**	**3 168.6**	**15 529.6**
Net Present Value	**-476.6**	**435.5**	**6 389.6**
BCR	**0.9**	**1.2**	**1.7**

The noticeable difference between the LDN scenarios and the Restored scenario is the difference in benefits gained relative to the BAU for carbon storage, harvested resources, livestock production, and water supply. Under the Restored scenario large areas of degraded and denuded grassland become restored giving rise to significant ecosystem service benefits in the form of additional carbon storage, higher stocks of wild non-woody resources, more productive rangelands for livestock farming, and the clearing of over 100 000 condensed ha of IAPs gives rise to significant water supply gains. Under LDN, while there is some restoration of grassland, the focus is more on clearing IAPs and woody encroachment and the difference between the LDN and BAU is minor in terms of the amount of grassland and savanna areas that are restored. This result suggests that the protection and restoration of grassland areas could yield significantly more benefit than the benefits gained through addressing bush encroachment.

It should be noted that the benefit estimates only include the tangible benefits that can be monetised and that these estimates are conservative. Firstly, the tourism benefits associated with the restoration today are likely to understate future values of these benefits, which are expected to increase as demand and incomes rise. Secondly, the cost-benefit analysis does not take into account a range of other potential benefits of the restoration interventions. These include water quality amelioration, as well as intangible benefits that may arise from restoring the biodiversity of the area. The restoration of the Thukela catchment would lead to an improvement in its flora and fauna, which is something that many members of society, even beyond South Africa, would value. These kinds of values, referred to in the literature as non-use or existence values, are intangible and difficult to quantify, even with best practice stated preference methods. While this study has not attempted to estimate existence value, this benefit is expected to be significant and should be acknowledged.

Furthermore, the results presented here include the avoided national costs in terms of carbon storage and not the avoided global costs which are orders of magnitude greater. Including the avoided global costs would result in a negative NPV of -R35 255 million under the optimistic LDN (BCR of -11.9) due to the net loss in carbon relative to the BAU and a positive NPV of R81 509 million (BCR of 9.9) under the Restored scenario due to the net gain in carbon relative to the BAU. This result is difficult to grasp as it suggests that South Africa needs significant investment to reach LDN but at a substantial cost to the rest of the world in the form of lost carbon. When restoration is taken further and includes restoring rangelands degraded prior to 2015 then there are carbon gains due to the restoration of grassland areas, resulting in a net gain to the rest of the world.

Of importance is that the results show a significant benefit in terms of livestock production under both restoration scenarios relative to the BAU. The present value of livestock production for the LDN scenarios was R621 million greater than under the BAU and R1 477 million for full restoration relative to the BAU (Table 6.19). From a livelihood perspective, especially in the communal areas of the catchment, these results are particularly promising.

Looking at the results of the best-case scenario where we assumed lower costs and higher benefits the results are more favourable with all three scenarios have a positive net benefit. There is a net benefit of R1 459 million, R2 448 million and R14 774 under the pessimistic LDN, optimistic LDN and Restored scenarios, respectively (Table 6.20). The benefit cost ratio under the best-case scenario was estimated to be 2.4 for optimistic LDN and 4.1 for the Restored scenario, much higher returns on investment than under the base estimate. Under the worst-case scenario assuming higher costs and lower benefits the results are expectedly far less favourable with net losses of -6.3, -R6.3 and -R17.1 billion under the pessimistic LDN, optimistic LDN and Restored scenarios, respectively.

Table 6.20. Present value of the costs of interventions and value of ecosystem service benefits under the expected best case, base estimate and worst case for LDN and Full Restoration (2020 R millions, 3.66% discount rate, 25 years).

	Present value (R millions)		
LDN Pessimistic	**Best-case**	**Base estimate**	**Worst-case**
Total present value of costs	2 707.7	3 645.2	5 760.3
Total present value of benefits	4 166.9	3 168.6	- 547.0
Net Present Value	**1 459.2**	**- 476.6**	**- 6 307.3**
BCR	**1.5**	**0.9**	**- 0.1**
LDN Optimistic	**Best-case**	**Base estimate**	**Worst-case**
Total present value of costs	1 718.5	2 733.1	5 723.5
Total present value of benefits	4 166.9	3 168.6	- 547.0
Net Present Value	**2 448.4**	**435.5**	**- 6 270.5**
BCR	**2.4**	**1.2**	**- 0.1**
Full restoration			
Total present value of costs	4 788.7	9 140.0	25 812.2
Total present value of benefits	19 563.1	15 529.6	8 796.6
Net Present Value	**14 774.4**	**6 389.6**	**- 17 015.6**
BCR	**4.1**	**1.7**	**0.3**

A sensitivity analysis of net present value using discount rates of 2% and 6% shows that under full restoration and LDN optimistic scenarios, a positive net outcome under all three discount rates for the base estimate and best-case situation are generated. The NPV only becomes negative under the worst-case situation for these two scenarios and these losses are substantially larger under the fully restored scenario compared to LDN as a result of the extent of SLM and restoration interventions implemented in achieving a fully restored catchment (Table 6.21). The LDN pessimistic scenario has a negative net outcome under all three discount rates for the base and worst-case situations.

Table 6.21. NPV Sensitivity Analysis using discount rates of 2%, 3.66%, 6% for LDN and Full Restoration (2020 R millions, 2021-2045).

Net present value (R millions)	**Discount rate**		
	2%	**3.66%**	**6%**
Pessimistic LDN Best	1 981.0	1 459.2	942.2
Pessimistic LDN Base estimate	- 305.0	- 476.6	- 630.4
Pessimistic LDN Worst	- 7 429.6	- 6 307.3	- 5 148.7
Optimistic LDN Best	3 059.2	2 448.4	1 833.5
Optimistic LDN Base estimate	641.3	435.5	238.4
Optimistic LDN Worst	- 7 458.7	- 6 270.5	- 5 034.4
Fully restored Best	18 230.4	14 774.4	11 272.8
Fully restored Base estimate	8 150.1	6 389.6	4 637.5
Fully restored Worst	- 20 182.4	- 17 015.6	- 13 702.1

7 DISCUSSION

7.1 Challenges in quantifying past degradation and condition

Countries may adopt a broader definition of degradation for LDN than the typical definition based on desertification and loss of soil carbon (Orr *et al.*, 2017). In South Africa, both IAPs and native bush encroachment have been recognised as forms of degradation that should be included in LDN targets (Department of Environmental Affairs, 2018; Turpie *et al.*, 2019; von Maltitz *et al.*, 2019). In fact, these were set as the highest priority forms of degradation to address in the country (von Maltitz *et al.*, 2019). While there is a strong case for their inclusion, this means that the existing remote sensing products designed for detecting degradation such as Trends.Earth are not sufficient for detecting degradation (von Maltitz *et al.*, 2019).

Although the UN indicator includes change in land cover as well as loss of vegetative productivity and loss of soil carbon, it does not typically include changes to woodier land cover types. This may be because an increase in woody cover (even if detectable as a land cover change) is not commonly recognised as a form of degradation. Indeed, in some parts of the world, tree planting has been encouraged as a measure to offset degradation, even though this sometimes involves alien trees or tree planting in landscapes that were not originally wooded (M. Braack, DFFE; A. Driver, SANBI, *pers. comm*). Given that the problems of IAPs and bush encroachment are widespread on the African continent and elsewhere, and that that they have serious impacts on ecosystem services, they should be included in the types of degradation described in the LDN framework, and indicators should be devised to accommodate this. This is especially important for indigenous bush encroachment, which is much less known, and which is linked to poor land management.

Our most important challenge was the lack of detailed, high resolution spatial data on degradation, largely as a result of the lack of field-verification. This spanned all types of degradation. Firstly, while the desertification of rangelands was mapped to an extent, overgrazed rangelands that had become vegetated with unpalatable species could not be picked up from satellite or aerial imagery. Thus, the extent of this type of degradation was likely underestimated.

Secondly, there were no detailed spatial data on IAPs. This necessitated the integration of coarse resolution data on IAP densities with fine scale land cover data, which had many challenges. Further work is urgently required to accurately map the extent of IAP invasions in South Africa at a resolution more compatible with land cover datasets. The lack of progress in this regard has also been noted by van Wilgen, Munyai & Wilson (2018: 81). Considering the extent of the problem, there is a need to integrate IAP data into national land cover datasets.

Thirdly, there are no reliable maps of the extent of bush encroachment. This necessitated the estimation of woody vegetation encroachment from land cover time series, and then subtracting the estimated proportion that could be IAPs. These problems were compounded by the changing reliability of LULC products over time. There is a need for a more thorough analysis of the aerial photography available for South Africa since the 1940s as a potentially more reliable way of "ground-truthing" earlier satellite imagery. In this study we found evidence of very poor levels of consistency in the identification of land cover classes, which made the identification of past trends very difficult. There were many cases of inconsistent changes through the KZN LC datasets that were highly unlikely to have been real, and which undermined the reliability of the projections made in this study.

The pilot accounts and this study used the KwaZulu-Natal LULC product because of its inclusion of ground-truthed measures of degradation (based on vegetative loss and erosion). However, the earliest in this time series was 2005. The National Land Cover (NLC) products go back to 1990 but do not include any measures of condition, and replication of the KZN approach would be too challenging at national scale.

Because of the emphasis on degradation, this study extended the methods used in the pilot accounts to provide more detailed information on condition. This highlighted some of the challenges of assessing terrestrial ecosystem condition, a critical component of NCA. Satellite data alone, and tools such as Trends.Earth which derive data from remote sensing cannot be used exclusively as measures of ecosystem condition (e.g. relative to natural or reference state) as these can only detect changes over the past two decades. In the Thukela catchment, large areas were already degraded before the turn of the century. Thus, research is needed to estimate terrestrial condition in a base year (e.g. 1990), based on the earliest available imagery and descriptions. For example, aerial photography is available from the 1940s. Given the magnitude of the task, this will likely require some sort of machine learning approach. Once the reference condition is properly described, it will be easier to track changes in condition over time.

That said, the continued tracking of land cover and ecosystem condition over time, as required for NCA, will also need to rely on ground-level or aerial data in conjunction with satellite data. In the study area, the invasion of IAPs is a key form of degradation that needs to be tracked in detail. Ideally, a combination of co-ordinated citizen science and automated methods use aerial and/or satellite surveillance and technological innovation is needed to map IAPs at high resolution and monitor changes over time..

If NCA is used to monitor progress in terms of LDN targets, then it is important to consider how the trends will be computed and interpreted. In the NCA, the condition of natural ecosystems (largely rangelands) will be measured in terms of their similarity to the Reference condition, and can be expressed as a percentage, and/or translated to a qualitative scale, e.g. from A (natural/near-natural) to F (critically modified).[31] In contrast, Indicator 15.3.1 assesses degradation, and a unit of land is deemed to be degraded if it has experienced a negative trend

[31] Of potential relevance, in South Africa, water policy suggests that aquatic ecosystems may not be allowed to degrade beyond some threshold state (D, or 40% similar to Reference, on the above scale).

of any of three parameters. Thus, the NCA could inform the indicator by computing land area that has experienced a negative change in condition. Based on the current binary computation of the indicator (degraded or not degraded), an ecosystem that changes from an A class to a B class would be deemed degraded, as would a system that has changed from a B class to an E class. Following SEEA guidelines, the NCA will also come up with a metric to evaluate the condition of lands used for crop production. It would be useful to separate the analysis of natural/rangeland and cultivated lands in the indicator.

Such methods, if applied consistently, can also be used to inform the environmental degradation criterion of the IUCN Red List of Ecosystems (RLE). The RLE identifies ecosystems that are "undergoing loss or disruption of key biotic processes or interactions," and assists in categorising the risk of ecosystem collapse based on abiotic and biotic degradation (both of which are fundamentally different in their mechanisms, Keith *et al.*, 2013; Bland *et al.*, 2017).

7.2 Uncertainties in projecting future degradation

The projections of land degradation were not only uncertain because of the lack of clarity on past trends, but because of the unavoidable uncertainties of projecting into the future. This comes from lack of knowledge about ecological responses to pressures, as well as lack of certainty about how some of those pressures may change in the future.

For example, much of the degradation that has been observed has been attributed to overgrazing. However, these pressures may well be reduced in future. Both livestock and cultivation appear to be on the decline in the region, as people increasingly move away from traditional rural livelihood strategies or are put under strain by climatic events such as droughts (see Vetter, Goodall & Alcock, 2020). This would explain the apparent declining rate in the denudation of lands and development of erosion gullies.

There is also uncertainty as to how climate change may affect process such as bush encroachment. There have already been marked changes in the area's climate in recent decades (Blignaut, Ueckermann & Aronson, 2009), and the continued increase in temperatures and carbon dioxide concentrations may accelerate bush encroachment (O'Connor *et al.*, 2014). As a result, poorly managed ecosystems may become more susceptible to bush encroachment than might be predicted from historical land cover changes, resulting in even more substantial alteration of the catchment in the absence of intervention.

It is also difficult to predict future demand for harvested natural resources. Past trends suggest that the demand for these resources has declined with increasing incomes and as people move away from traditional rural livelihoods, and through increased provision of services such as electricity. If South Africa meets its national development goals, it could happen that our study has overestimated future demands for these resources. However, it is well known that these natural resources also provide a fallback in the time of shocks. The recent Coronavirus pandemic is a reminder of the unpredictability and potential severity of such shocks. These uncertainties have implications for the estimation of ecosystem service benefits as well as rates of degradation.

7.3 Estimating the impacts on ecosystem services and values

Key to this analysis was the estimation of changes in the value of ecosystem services under different scenarios. This required an understanding of how ecosystem capacity to supply services varies in relation to ecosystem type and condition, as well understanding the demand for the services. These are challenging requirements, especially when applied at scale and based solely on existing information.

For example, our study used a relatively sophisticated spatial model to estimate natural resource use, but its accuracy was dependent on good information on both resource stocks and household demands. Information on how stocks of non-woody resources change as a result of IAP infestation and woody encroachment is scarce. We assumed that dense encroachment and/or infestation of single invader species has an impact on diversity and functionality of ecosystems, resulting in significantly reduced stocks in these degraded areas. Based on this assumption, the predicted harvesting of wild plant foods, medicines and wild meat was lower under the BAU compared to the LDN and Restored scenarios. Field based research is needed to more accurately assess how resource stocks vary with ecosystem characteristics, IAP infestation and woody encroachment.

There is also some uncertainty in the projection of tourism, which was based on a conservative estimate of growth. The tourism industry is influenced by a number of local and global factors including culture, crime and security, developed infrastructure, natural attractions, attitudes of the people, global pandemics, and economic recessions. Therefore, predicting how demand for tourism might change in the future is very challenging.

One of the key challenges in this study was estimating the impacts of different scenarios on water supply. There is an extensive literature on the effects of IAPs on water flows, but almost none that considers a range of forms of degradation at the same time. Modelling hydrology at scale is also particularly challenging. We set up a hydrological model using SWAT. SWAT is a widely used model and has been applied at various geographic scales. However, addressing complex changes in land cover is usually modelled at much smaller scales. For example, Scott-Shaw, Hill & Gillham's (2020) conducted their study on a 76 ha catchment within Kwazulu-Natal. Modelling a small catchment area allows the user to build a skeleton model, which can be further defined and developed over the course of many years as they study for instance a particular land class; thereby allowing them to refine their parameters to local conditions. Many of the difficulties and limitations within large-scale hydrological modelling study were aligned with the limitations as described by Abbaspour *et al.* (2015) in their continental-scale hydrology and water quality model for Europe. (1) Limited and unevenly distributed flow, sediment and nutrient data, and discharge stations with varying time series lengths and quality of record, (2) limited available knowledge of the attributes and especially the management of reservoirs, as well as the lack of detailed records of water transfers (urban and agricultural) within and out of the catchment area, (3) lack of data on regional scale soil moisture and/or deep aquifer percolation, which made a calibration/validation of these components impossible overall and most notably during the parameterization of IAPs, (4) lack of knowledge of South African agricultural management operations and their translation into SWAT, and (5) the application of a global conceptual model within South Africa and the inherent assumptions and simplifications necessary to adapt a model to a South African environment at a large scale. Where possible, local academic efforts, such as the recent study published by Scott-Shaw *et al.*, (2020) should be used in parametrizing geographically large SWAT models (as was done in this study). The development of large-scale SWAT models designed for complex scenario analysis is a major undertaking and can be challenging when there are time or resource constraints. In this study, the model did not yield satisfactory results for water supply, and so we relied on existing estimates from studies that were focussed on IAPs. IAPs were likely to account for most of the variation in water supply under the different scenarios. Although this approach did not include the negative impacts of bush encroachment on water supply, the removal of bush encroachment was largely balanced by the restoration of woody habitats in other areas, which may have cancelled out those impacts. Finally, it should be noted that the existing estimates of the impacts of IAPs on water supply are themselves based on relatively simple models and assumptions, and that there is uncertainty around these estimates as well.

Further research is needed on many of the ecosystem services considered in this study. In particular, a better understanding is required of how the supply of these services varies geographically and in response to changing land cover, ecosystem types and condition. Hydrology, sediments, water quality and natural resource stocks should be priorities for research, so that the models that have been set up to estimate these services can be refined. Related to this, there is a need for far more accurate, high resolution spatial data on land cover, land use and IAPs.

7.4 Methods and costs of measures to address degradation

The primary focus of LDN is to minimise degradation through implementation of SLM practices. However, very little research has been conducted in South Africa on such practices and their costs and effectiveness in different contexts. The study therefore relied on information from the international literature and on expert opinion, including that of experienced NGO staff who were not optimistic about the effectiveness of SLM interventions. In South Africa, in many instances, the causes of land degradation are compounded by poor regulatory frameworks, planning and implementation around land tenure and farming activities, and socio-economic drivers such as high population densities, market access, cultural norms and poverty (Hoffman & Todd, 2000; Scholes & Biggs, 2004). This often makes finding an effective and efficient solution very challenging and can contribute to the failure or collapse of interventions that are implemented without due consideration for social and political factors. Recognising the high level of uncertainty in this regard, it was necessary to set up both a pessimistic and an optimistic LDN scenario.

In order to achieve SLM, it was assumed that public extension services would be strengthened in the catchment by increasing the ratio of extension officers to the number of farmers and by providing improved training and renumeration. In addition, we included budget for an incentive-based mechanism in the form of payments for ecosystem services (PES) and for a stewardship programme. There are various other possible mechanisms that could have been considered, such biodiversity offset mechanisms, environmental fiscal reforms and taxation models, markets for green products, business-biodiversity partnerships, new forms of charity, new and innovative sources of international development finance, and synergies with funding mechanisms for climate change. However, their relative costs and effectiveness are not well understood. This lack of research is a potential barrier to decision-making and the eventual roll-out of interventions, especially given that the South African government has expressed a desire to develop incentive mechanisms to encourage green growth and support the biodiversity economy. There are already some funds that have been set up, such as the "Green Fund", supporting initiatives to assist South Africa's transition to a low carbon, resource efficient and climate resilient development path, the "Drylands Fund" (since 2011) to support programmes and interventions, foster partnerships and scale these up to ensure SLM, especially in poor communities who are most susceptible to land degradation (Republic of South Africa, 2018), and the "Jobs Fund" (since 2011) to co-finance projects by public, private and non-governmental organisations, focusing on labour-intensive activities such as ecosystems restoration.

In particular, there is very little information on the amount of investment in these options that is required to stem degradation. Our estimates suggested that a considerable, sustained investment is required to achieve SLM. Long-term, sustainable financing solutions are therefore needed to adequately address SLM at the landscape level. One opportunity is the Land Degradation Neutrality Fund (LDNF) which has been created by the Global Mechanism of the UNCCD. Labelled as an 'impact investment fund', it aims to mobilise private capital for investment (to be blended with public finance) into addressing land degradation through the LDN approach (Chancellor, 2019). Indeed, achieving LDN requires investment from both public and private sectors (UNCCD, 2015a). Public investment should focus on enabling mechanisms such as extension programmes, PES schemes, stewardship programmes and biodiversity offsets, and to facilitate private sector investment. Further research is needed to get a better understanding of the costs associated with implementation of SLM interventions and the amount of investment required to achieve target outcomes. The importance of this understanding is highlighted by the difference in outcomes under the optimistic versus pessimistic LDN scenarios.

Long-term, sustainable financial solutions are needed to adequately address SLM at the landscape level. Indeed, given the scale of the problem, innovative and integrated funding solutions are needed to achieve LDN, as well as investment from both the public and private sector and critically, political will. Public investment should focus on providing appropriate enabling structures such as policies and strategies after which the private sector can provide the investment needed to adequately address land degradation, with the idea that landscape level restoration projects will then generate revenues.

The estimated costs of achieving SLM were orders of magnitude lower than those of rehabilitation or restoration. Preventing future land degradation is generally far more cost-effective in the long run, than aiming to reduce or reverse past degradation. This understanding underpins the scientific conceptual framework for LDN which can be used for developing planning and implementation of LDN measures (Cowie *et al.*, 2018). Indeed, more is known about the costs and effectiveness of restoration. There is a wealth of information pertaining to the management and clearing of IAPs in South Africa and a number of studies have been undertaken to better understand the costs and benefits of restoring alien infested areas. Less is known on the costs of addressing bush encroachment, and the study relied on estimates from elsewhere in Southern Africa. Nevertheless, more research is also required on the costs of restoring the loss of vegetative cover and palatable grasses in rangelands.

8 CONCLUSIONS

8.1 Halting and reversing ecosystem degradation has positive net economic benefits

Achieving LDN should primarily focus on stopping the process of degradation through SLM practices, but recognising that this is not likely to be completely successful, it also involves offsetting any new degradation by actively restoring currently degraded lands (Lal, Safriel & Boer 2012). LDN does not address past degradation, except inasmuch as required for this offsetting. This study also explored the costs and benefits of a full restoration scenario, to test a much more ambitious alternative to the minimum obligation of LDN.

Implementing LDN in South Africa is already an imperative at any cost. Implemented timeously, it simply means halting further degradation and undertaking some restoration to make up for where degradation has not been completely halted. This would be expected to be far less costly than restoring past degradation. In South Africa, where there has been some delay in stepping up action to stem degradation, achieving LDN will now also involve restoring the degradation that has taken place in the six years since 2015. Nevertheless, the benefits of achieving LDN in 2030 relative to 2015 are likely to outweigh the costs, especially if SLM measures are implemented in an effective manner. Moreover, it would be even more beneficial to go above and beyond the minimum requirements of LDN, to address degradation that has taken place before 2015. In the Thukela catchment, much of the degradation of grassland areas took place well before this. These areas have the potential to make a valuable contribution to biodiversity, ecosystem services and livelihoods. Thus, in the case of the Thukela catchment, restoration above and beyond the obligations under LDN would be desirable.

Nevertheless, this study has also shown that, while addressing land degradation can be shown to be worthwhile from a societal point of view, it still comes at a considerable financial cost. Innovative and integrated funding solutions will be needed to achieve LDN or more ambitious restoration goals. Public investment should therefore include a focus on providing the enabling conditions that foster the private sector investment needed.

These results are conservative, in that they do not include all the values associated with healthy ecosystems, and do not consider the increasing scarcity value of intact natural systems. In particular, we have not included estimates of the non-use or intangible values associate with the restoration of habitats and biodiversity, which are likely to be considerable and increasing over time.

The conclusions were also supported by the sensitivity analysis, which was necessary due to some of the uncertainties involved in the mapping and projection of land cover and ecosystem condition, modelling and valuation of ecosystem services, and the costs and effectiveness of measures to address the drivers and effects of degradation. Measures to reduce these uncertainties are discussed below.

8.2 Ecosystem accounting will be a useful tool for informing policy and strategy

This study benefited from the pilot ecosystem accounts for KwaZulu-Natal, which include standardised spatial information on ecosystem extent, condition and ecosystem services in physical and monetary terms. We made use of the spatial architecture (i.e. reference grid and projection), and applied or adjusted the methods, models and value estimates from that study to the 2017 LC used in this study. The compilation of more accurate ecosystem accounts over a series of years will be invaluable in guiding future interventions and monitoring their outcomes.

8.3 Priority areas for research

The uncertainties in this study could be reduced with further monitoring and research to advance ecosystem accounting as a monitoring tool and to inform national strategy and priorities to address land degradation, including:

- Developing high resolution time series data of IAPs at national scale using field data and technological innovation;
- Defining the reference condition for terrestrial ecosystems, and determining ecosystem condition from a defined base year and going forwards that takes all forms of degradation into account;
- Undertaking further research to improve modelling of the effects of changes in ecosystem condition on hydrological and other ecosystem services; and
- Undertaking scientifically rigorous research into the efficacy of different interventions addressing land degradation in different biophysical and social contexts.

For example, even with the very important and increasing contribution of remote sensing, some of the challenges faced in this study have emphasised the importance of collecting ground and/or aerial baseline and monitoring data to improve the accuracy and consistency of classifying land cover and ecosystem condition. This includes improving the techniques for assessing and mapping the extent of IAP invasions in South Africa to generate a dataset that is consistent with the Standard for National Land Cover structure and resolution.

Ecosystem accounting will also be better able to inform such policy decisions through further baseline research on the supply of ecosystem services. For example, there are few data on the stocks of resources in different habitats or how they vary with ecosystem condition. One of the key challenges was modelling the hydrological effects of complex land cover changes at large scale. This did not yield satisfactory results within the time available, and we instead relied on existing estimates derived with simpler models. Future studies will need to start by exploring these complexities at smaller scales. While it will be necessary to account for ecosystem services at national scale, and this may initially be done at a relatively coarse level, this study has shown that even working at a basin scale can make it difficult to quantify important local effects of ecosystem changes. It will be important to use a scale that captures processes at a local, tangible level for most citizens, scientists and policy-makers, and then to gradually scale up, catchment-by-catchment, to build a more accurate national perspective.

While availability of information allows for relatively robust estimates of restoration costs, there is still considerable uncertainty regarding the methods, costs and effectiveness of achieving SLM. Land degradation in the study area occurs as a result of poor regulatory frameworks, planning and implementation around land tenure and farming activities, as well as socio-economic drivers such as high population densities, market access, cultural norms and poverty. Without addressing the institutional issues behind land degradation, SLM efforts carry a risk of failure. For this reason, our study placed included upper and lower bound estimates of costs to encompass the range of possibility in this regard. Strategies for achieving LDN should assume a middle road, and place strong emphasis on addressing the underlying issues.

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A1. APPENDIX 1: 2017 KWAZULU-NATAL LAND COVER CLASSES

Table 9.1 KwaZulu-Natal Land Cover Classes (GeoTerra Image, 2018).

Value / Code	Land Cover Class	Description
0	No Data	
1	Water natural	All areas of natural open water, excluding estuarine, and coastal waters.
2	Plantation	All areas of non-natural timber plantations.
3	Plantation clear-felled	All temporary clear-felled stands awaiting re-planting within non-natural timber plantations.
4	Wetlands	All permanent, near permanent or daily freshwater, brackish or saline wetland areas (primarily associated with topographically defined flood plain areas).
5	Wetlands-mangrove	Mangrove wetlands.
6	Permanent orchards (banana citrus) irrigated	Permanent, irrigated orchards comprising primarily banana and citrus's trees and shrubs. Also includes tea plantations.
7	Permanent orchards (cashew) dryland	Permanent, non-irrigated orchards comprising primarily cashew nut trees.
8	Permanent pineapples dryland	Permanent, non-irrigated orchards / plantations comprising primarily pineapple crops.
9	Sugarcane - commercial	Commercial, large scale sugarcane cultivation, including both irrigated and dryland crops.
10	Sugarcane - emerging farmer	Commercial, small scale sugarcane cultivation, including both irrigated and dryland crops. Emerging farmers are defined on the basis of field sizes being typically larger than subsistence field units but smaller than commercial field units, on a locally defined basis.
11	Mines and quarries	Major surface-based mineral and rock excavation sites.
12	Built up dense settlement	All major urban and built-up areas, irrespective of associated residential, commercial or industrial use, defined in terms of local high building densities. Also includes associated covers such as land-fills, rubbish dumps and cemeteries, and other built-up features such as chicken and pig battery farms.
13	Golf courses	Golf courses and golf estates (includes all grass and tree areas within boundary), and other major areas of non-agricultural improved grasslands such as sports fields and race tracks.
14	Low density settlement	Areas of low-density settlement, typically in rural or urban periphery locations, that do not in terms of size or density belong in the denser Built-Up settlement. Often associated with subsistence cultivation activities.
15	Subsistence (rural)	Identifiable areas of scattered or clustered, small-scale, dryland cultivation for local or household consumption, typically associated with rural dwelling cover classes. Can include some subsistence level dryland sugarcane fields, if field sizes are small, and the sugarcane crop cannot be defined as a "pure" unit in each case.
16	Annual commercial crops dryland	Commercial, medium-large scale dryland cultivation of annual crops.
17	Annual commercial crops irrigated	Commercial, medium-large scale irrigated cultivation of annual crops.

Value / Code	Land Cover Class	Description
18	Forest	Dense, tall tree dominated forest communities with > 70% canopy closure.
19	Dense thicket & bush (70-100 cc)	Dense, medium / tall, tree and shrub dominated communities with > 70 % canopy closure.
20	Medium bush (< 70cc)	Medium / tall shrub dominated communities with 40 – 70 % canopy closure.
21	Woodland	Tree-based communities with an open grass layer, with tree canopy closure between 10 – 70 %.
22	Grassland / bush clumps mix	Grassland dominated areas with scattered bush and thicket clumps.
23	Grassland	Open grassland areas.
24	Bare sand	Natural non-vegetated areas of exposed sand (e.g. river sand,). Specifically excludes coastal beach and dune deposits, which are mapped as a separate sub-class.
25	Degraded forest	Areas of Forest (class 18) that show a significant loss of tree and shrub canopy cover, when compared to surrounding areas of natural Forest.
26	Degraded bushland (all types)	Areas of Bushland (all types, classes 19, 20, 22)) that show a significant loss of tree and/or shrub canopy cover, when compared to surrounding areas of natural Bushland. If tree loss is not significant, “degraded woodland and wooded grassland” areas will be included in this class.
27	Degraded grassland	Areas of Grassland (class 23) that show a significant loss of grass canopy cover, when compared to surrounding areas of grassland. If tree loss is significant, “degraded woodland and wooded grassland” areas will be included in this class.
28	Old cultivated fields - grassland	Old fields, not recently cultivated, which are identifiable on the basis of remnant fence-line effects, and which appear to have been previous grassland areas.
29	Old cultivated fields - bushland	Old fields, not recently cultivated, which are identifiable on the basis of remnant fence-line effects, and which appear to have been previous bushland areas.
30	Smallholdings - grassland	Semi-rural areas on the fringes of major urban areas that contain a combination of large residential cadastral parcel and / or “recreational” semi-commercial farming activities, within a previously grass or bushland-dominated landscape.
31	Erosion	Non-vegetated areas (or areas of very low vegetation in comparison to the surrounding natural vegetation), that are primarily the result of gully-type erosional processes, occurring through either natural and / or anthropogenic actions.
32	Bare rock	Natural non-vegetated areas of exposed hard rock (e.g. sandstone paving, cliffs).
33	Alpine grass-heath	Communities of low shrubland and grassland typically associated with the high-altitude Drakensberg Escarpment Plateau regions.
34	KZN national roads	National class road lines as defined within the KZN Provincial Dept of Transport’s GIS database.
35	KZN main & district roads	Main & District class road lines as defined within the KZN Provincial Dept of Transport’s GIS database.
36	Water dams	All areas of open water within man-made impoundments, ranging from farm dams to major reservoirs.
37	Water estuarine	All areas of natural open water, associated with the estuarine reaches of a river.

Value / Code	Land Cover Class	Description
38	Water sea	All areas of natural open water, associated with the coastal and sea areas.
39	Bare sand coastal	Natural non-vegetated areas of exposed sand associated specifically with coastal dunes and beaches.
40	Forest glade	Naturally occurring open grassy regions, enclosed within closed canopy indigenous forests.
41	Outside KZN boundary	Areas not classified since they fall outside the KZN Provincial boundary.
42	KZN railways	All railway lines located within the KZN Provincial, and visible on the SPOT5 imagery.
43	Airfields	Rural airfields and airstrips (often grass).
44	Old plantation- high vegetation	Former tree plantations that have been cleared and are now covered in tall regrowth vegetation.
45	Old plantation - low vegetation	Former tree plantations that have been cleared and are now covered in low regrowth vegetation.
46	Rehabilitated mines - high vegetation	Former mining areas that are now covered in tall regrowth vegetation.
47	Rehabilitated mines - low vegetation	Former mining areas that are now covered in low regrowth vegetation.
48	Wetland – Drainage / Riparian	All permanent, near permanent or daily freshwater, brackish or saline wetland areas (primarily associated with riparian lines outside of topographically defined flood plain areas). Note: class not delineated in 2005, 2008 or 2011 LC products.

A2. APPENDIX 2: RULES FOR DEVELOPING THE INTEGRATED CLASSES IN THE LAND COVER DATASETS

Table 9.2. Rules for developing the integrated classes in the land cover datasets. For infested classes, the base land cover in the integrated class remains when IAPs are removed.

KwaZulu-Natal LC class	IAP level	Integrated class / Restored (= without IAPS)
Forest	<5% IAPs	Forest
Forest	Light IAPs	Forest + Light IAPs
Forest	Med IAPs	Forest + Med IAPs
Forest	Dense IAPs	Forest + Dense IAPs
Dense bush	<5% IAPs	Dense bush
Dense bush	Light IAPs	Medium bush + Light IAPs
Dense bush	Med IAPs	Bush clumps/Medium bush + Med IAPs
Dense bush	Dense IAPs	Grassland/Woodland + Dense IAPs
Medium Bush	<5% IAPs	Medium Bush
Medium Bush	Light IAPs	Grassland/Woodland + Light IAPs
Medium Bush	Med IAPs	Grassland/Woodland + Med IAPs
Medium Bush	Dense IAPs	Grassland/Woodland + Dense IAPs
Woodland	<5% IAPs	Woodland
Woodland	Light IAPs	Grassland + Light IAPs
Woodland	Med IAPs	Grassland/Woodland + Med IAPs
Woodland	Dense IAPs	Grassland/Woodland + Dense IAPs
Grassland / bush clumps mix	<5% IAPs	Grassland / bush clumps mix
Grassland / bush clumps mix	Light IAPs	Grassland + Light IAPs
Grassland / bush clumps mix	Med IAPs	Grassland + Light IAPs
Grassland / bush clumps mix	Dense IAPs	Grassland + Med IAPs
Grassland	<5% IAPs	Grassland
Grassland	Light IAPs	Grassland
Grassland	Med IAPs	Grassland
Grassland	Dense IAPs	Grassland
Forest glade	Any	Forest glade
Alpine grass-heath	Any	Alpine grass-heath
Wetland - Drainage / Riparian	<5% IAPs	Wetland - Drainage / Riparian
Wetland - Drainage / Riparian	Light IAPs	Wetland - Drainage / Riparian + Light IAPs
Wetland - Drainage / Riparian	Med IAPs	Wetland - Drainage / Riparian + Med IAPs
Wetland - Drainage / Riparian	Dense IAPs	Wetland - Drainage / Riparian + Dense IAPs
Degraded bushland	<5% IAPs	Degraded bushland

Degraded bushland	Light IAPs	Degraded bushland + Light IAPs
Degraded bushland	Med IAPs	Degraded grassland + Med IAPs
Degraded bushland	Dense IAPs	Degraded grassland + Dense IAPs
Degraded forest	<5% IAPs	Degraded forest
Degraded forest	Light IAPs	Degraded forest + Light IAPs
Degraded forest	Med IAPs	Degraded forest + Med IAPs
Degraded forest	Dense IAPs	Degraded forest + Dense IAPs
Degraded grassland	Any	Degraded grassland
Old cultivated fields - bushland	<5% IAPs	Old cultivated fields - bushland
Old cultivated fields - bushland	Light IAPs	Old cultivated fields - bushland
Old cultivated fields - bushland	Med IAPs	Old cultivated fields - grassland + Med IAPs
Old cultivated fields - bushland	Dense IAPs	Old cultivated fields - grassland + Dense IAPs
Old cultivated fields - grassland	Any	Old cultivated fields - grassland
Old plantation - high vegetation	<5% IAPs	Old plantation - high vegetation
Old plantation - high vegetation	Light IAPs	Old plantation - low vegetation + Light IAPs
Old plantation - high vegetation	Med IAPs	Old plantation - high vegetation + Med IAPs
Old plantation - high vegetation	Dense IAPs	Old plantation - high vegetation + Dense IAPs
Old plantation - low vegetation	<5% IAPs	Old plantation - low vegetation
Old plantation - low vegetation	Light IAPs	Old plantation - low vegetation + Light IAPs
Old plantation - low vegetation	Med IAPs	Old plantation - low vegetation
Old plantation - low vegetation	Dense IAPs	Old plantation - low vegetation
Rehabilitated mines - high vegetation	<5% IAPs	Rehabilitated mines - high vegetation
Rehabilitated mines - high vegetation	Light IAPs	Rehabilitated mines - low vegetation + Light IAPs
Rehabilitated mines - high vegetation	Med IAPs	Rehabilitated mines - high vegetation + Med IAPs
Rehabilitated mines - high vegetation	Dense IAPs	Rehabilitated mines - high vegetation + Dense IAPs
Rehabilitated mines - low vegetation	Any	Rehabilitated mines - low vegetation