

# Chapter 10

## Ecological Impacts of Biomass Production at Stand and Landscape Levels

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### 10.1 Introduction

In Chaps. 4, 5 and 6 of this book, we discussed the production and procurement of biomass from various sources, including extensively managed systems such as woodlands, and much more intensively managed systems such as short-rotation bio-energy plantations. It is generally accepted that intensive, production orientated land uses will have an impact on carbon stocks, biodiversity, growth resource use or resource quality (Achten and Verchot 2011; German et al. 2011). Some of these impacts may be exacerbated if exotic species are used in the system. Conversely, the ecological impacts on areas of low management intensity or near natural vegetation are usually lower. This chapter will focus more specifically on the ecological impacts of intensively managed tree-based biomass production systems, and it is structured according to the scale of impacts, i.e. the landscape level and site level. At both levels, we have attempted to highlight the major threats to sustained productivity and the provision of ecosystem services. However, even intensive biomass production systems can arguably be managed in ways that mitigate the

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ecological impacts of such systems. This chapter will therefore also focus on some case studies where ecological impacts could be limited or mitigated through the adoption of specific management strategies.

## 10.2 Impacts at the Landscape Level

The conservation of biodiversity and the use of scarce resources such as water are issues that belong at the landscape scale. The challenge is then to balance these impacts at the landscape scale and to ensure biological corridors/linkages in the indigenous vegetation that can facilitate the movement of individuals and genetic material. Evaluation of landscape scale impacts will also consider the effects of biomass production systems on water and air quality.

### 10.2.1 Water Use

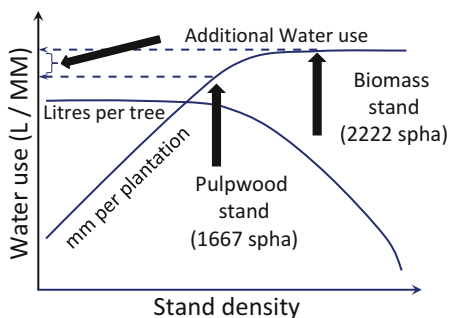
Large-scale changes in land-use, such as those proposed for intensive biomass production, constitute a change in the structure, functioning, species composition and management of the vegetation growing on the land. This, in turn, signifies a change in how water is intercepted, infiltrated, transpired and evaporated from the land surface. The resultant impacts on the availability and quality of water in rivers is of great importance to the downstream users of that water. Consequently a good understanding and quantification of land-use driven water resources impacts is required when land-use changes are proposed. Stream-flow changes associated with vegetative land-use changes may be described using a simplified water balance equation, namely:

$$Q = P - Et \pm \Delta S$$

where  $Q$  = streamflow,  $P$  = precipitation,  $Et$  = evapotranspiration and  $\Delta S$  = changes in soil water storage.

This equation is best applied over a suitably long time period (e.g. several years), where changes in soil water storage are assumed to balance out, and rainfall is representative of the long-term mean for the area. In this case, changes in  $Et$  caused by vegetative land-use change equate to changes in streamflow at the landscape level, if the water use of the replacement vegetation is significantly different to that of the existing land-use. In South Africa, plantation forestry with introduced (exotic) species is an extensive and profitable land use in many of the high-rainfall regions of the country (Chamberlain et al. 2005) with an area of approximately 1.25 million hectares currently under commercial plantations (FSA 2010). Growth in the industry

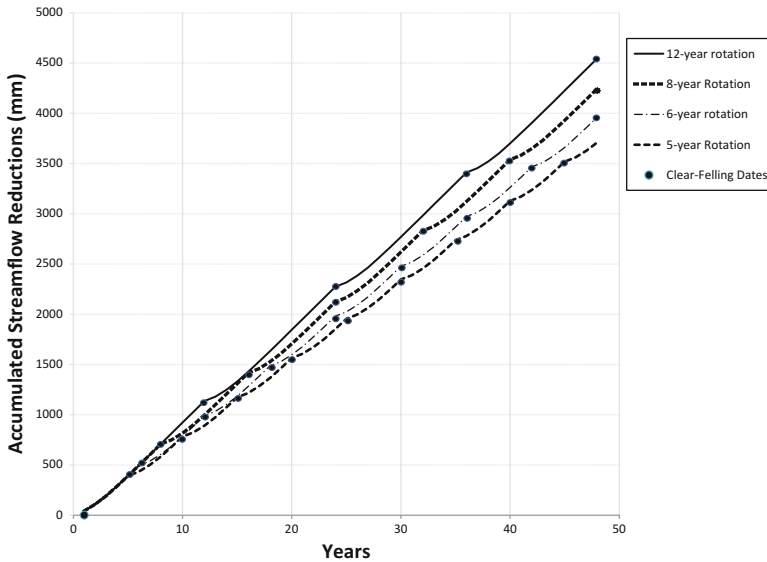
**Fig. 10.1** Hypothetical changes in the water-use of individual trees ( $L \cdot tree^{-1}$ ) compared to a commercial plantation (mm), relative to increases in stand density. Current and future scenarios under intensive biomass production are illustrated



is restricted by legislation (National Water Act of 1998, Section 36), which, amongst other aspects, declares commercial plantation forestry to be a Stream-flow Reduction Activity (SFRA) due to the high water-use of forest plantations and their impact on catchment water yields (Dye and Versfeld 2007). However, with the resultant efforts to maximise biomass production in intensively managed commercial forests, it is important to consider the associated hydrological impacts. This section considers the potential hydrological impacts, particularly streamflow changes, likely to be associated with intensive woody biomass production at a landscape scale. The primary land-use changes predicted to occur under intensified biomass production, and for which hydrological impacts need to be considered, include increased stand densities, shorter rotation lengths, earlier canopy closure and changes in site/species preferences.

### 10.2.1.1 Stand Density

Increases in tree densities are usually associated with increases in water-use, due to greater competition for resources driving increases in leaf area per unit of land, higher root intensities and colonisation of greater soil volumes by roots. While it is true that the water-use of individual trees of a given age within a plantation will decrease as tree densities increase (due to competition), over-all water use for the plantation is likely to increase until a threshold is reached where water availability is the limiting factor and the water use levels off. This threshold tree density will vary depending on the site and species, but in general for intensively managed tree plantations of a given age, where effective understorey/weed control is practiced, moving from a typical pulpwood stand density of  $3 \text{ m} \times 2 \text{ m}$  (1,667 spha) to a spacing of  $3 \text{ m} \times 1.5 \text{ m}$  (2,222 spha), envisaged for intensive biomass production, will increase overall water-use of the stand (Fig. 10.1). A potential exception to this is when understorey vegetation with a particularly high water-use is suppressed by an increase in trees with relatively lower water-use rates (e.g. certain indigenous tree species).



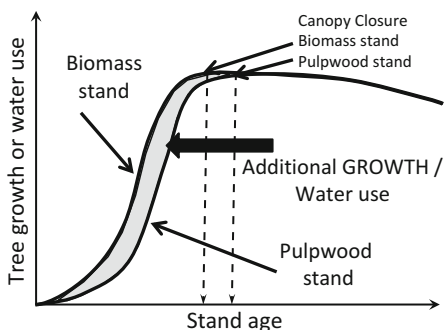
**Fig. 10.2** Accumulated streamflow reductions for Eucalypts simulated under a range of rotation lengths (Data from Gush et al. 2002)

### 10.2.1.2 Rotation Length

Tree water-use also changes relative to the age of the tree, typically following a sigmoid curve, with a low initial water use followed by rapid increases flattening out to a plateau once canopy cover is achieved (Scott and Smith 1997; Dye and Bosch 2000). In certain long-rotation sawtimber stands (>30 years for pines, and >15 years for eucalypts), tree water-use (streamflow reduction) has even been observed to tail off with age (Scott and Prinsloo 2008). However, with the advancement of tree improvement programmes and genetic selection for faster growth rates, rotation lengths have generally decreased, particularly in eucalyptus plantations (Verryn 2000). Rotation lengths are predicted to shorten even further with the advent of intensive biomass production plantations, typically reducing from 6 to 5 years on good quality pulpwood sites, and from 12 to 8 years on sawtimber sites. Combined with higher stand densities, these plantations will produce trees of smaller individual volumes at harvesting, but greater biomass production per land unit overall. Due to the shape of the tree water-use curve relative to tree age, reductions in rotation length are likely to decrease the overall water-use of the plantation, resulting in lower streamflow reductions (Fig. 10.2).

A challenge in evaluating the wider scale hydrological impacts of rotation length changes associated with biomass plantations is that individual compartments within a commercial forest plantation go through a growth cycle from planting to clearfelling and back to planting again, with constantly changing water use impacts. However, at a landscape scale it may be helpful to consider the planted area as

**Fig. 10.3** Hypothetical changes in tree growth and water-use relative to stand age, before canopy closure within a pulpwood stand compared to an intensive biomass production stand



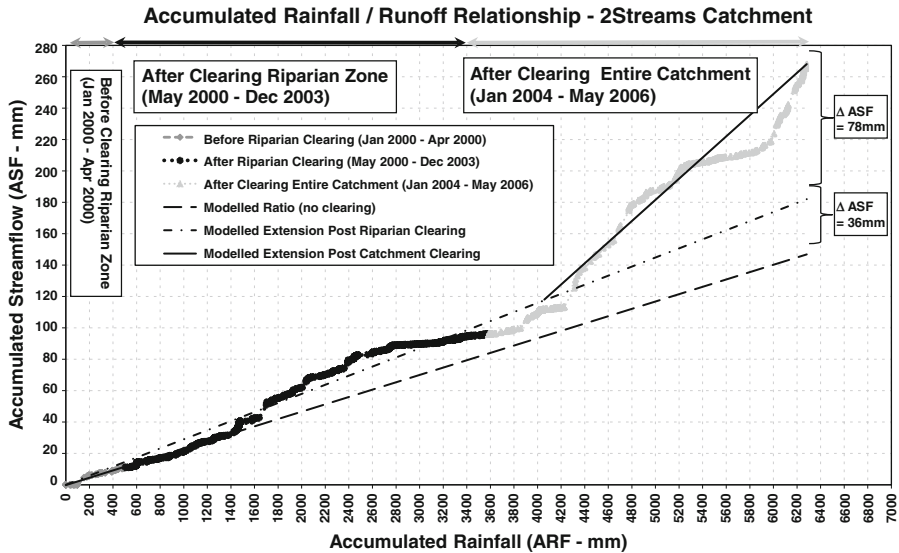
a mosaic of compartments representing all ages from seedlings to mature trees, growing simultaneously, and cycling through the various stages of growth and water use. For areas growing just one species under one rotation length, the net water use impact will be that of plantations at the “water-use mid point” of their rotation. However for areas growing multiple species at different rotation lengths water use impacts may need to be weighted relative to species and rotation length predominance.

### 10.2.1.3 Canopy Closure

Tree improvement programmes and advances in site management have increased the growth rates of commercial forestry tree species. This has also led to earlier canopy closure in plantations. Under increased stand densities and shorter rotation lengths associated with intensive biomass production, canopy closure, with an associated Leaf Area Index (LAI) of 5 is now likely to occur after just 2 years on a favourable site. There are well established links between growth and water-use in trees, so an increase in the gradient of the water-use curve during the first few years of the rotation may be expected under faster growing trees. However, at canopy closure, competition for light, as opposed to water, may become the limiting factor to further increases in leaf area and hence water-use. Consequently, under intensive biomass production, the water-use curve is likely to peak and plateau earlier than before, resulting in an overall increase in water-use relative to tree age (Fig. 10.3).

### 10.2.1.4 Site and Species Choices

The location within the landscape of commercial forestry plantations; be they existing pulpwood/saw-timber stands or proposed future intensive biomass production stands, undoubtedly has an impact on their water-use. This is most pronounced in the distinction between riparian and non-riparian sites. In a study quantifying the effect of changes in riparian zone vegetation on catchment water yield (streamflow),



**Fig. 10.4** Changing relationships between accumulated streamflow and rainfall in response to progressive clearing of *Acacia mearnsii* trees from riparian and upslope sites respectively (Everson et al. 2007)

**Table 10.1** Calculation of relative contributions of riparian and upslope areas to streamflow following clearing of *Acacia mearnsii* (Black Wattle) stands

Zone	Area (ha)	% of total	Streamflow gain (mm)	% of total	Streamflow gain (mm ha <sup>-1</sup> )
Riparian Zone (RZ)	7.5	11.5	36	31.5	4.8
Non-RZ	58	88.5	78	68.5	1.34
Total cleared area	65.5	100	114	100	1.74

Based on data from Everson et al. (2007)

Everson et al. (2007) showed significant responses in streamflow following clearing of *Acacia mearnsii* (Black Wattle) trees from riparian and upland areas in a small catchment in KwaZulu-Natal (Fig. 10.4).

During the 6 year period of the study (May 2000 to May 2006), increases in streamflow associated with the clearing of the *A. mearnsii* trees, which had initially been planted throughout the catchment including the stream channel, were monitored. Based on the areas cleared and the resultant streamflow changes observed, these results indicate that streamflow gains following clearing operations were 4.8 mm for every hectare of riparian area cleared, and 1.34 mm for every hectare of upslope area cleared (Table 10.1). A unit of land in the riparian area under *A. mearnsii* consequently represented the hydrological equivalent of 3.58 times the upslope area ( $4.8/1.34 = 3.58$ ) when cleared.

While legislation currently prohibits the establishment of commercial forestry plantations in riparian areas (FIEC 1995), the above findings illustrate the

importance of focusing on riparian areas when clearing invasive exotics through activities such as the Working for Water programme (Turpie et al. 2008). They also help to quantify the water released by such activities, particularly when harvesting for biomass/bio-energy production. A further site-related hydrological consideration is the utilization of more marginal forestry areas, particularly in terms of rainfall. Intensive biomass production in such areas is likely to have lower *absolute* water-use impacts (mm), but significantly greater *relative* water-use impacts (%), compared to optimum sites. Linked to this is the need to make distinction between impacts on total flows and impacts on low flows (e.g. driest 3 months of the year). Scott and Smith (1997) argued that low flows may be of greater relevance to decision makers than reductions in total flows, and several South African studies have focused on this aspect, most recently Jewitt et al. (2009). The significance of low flows is also attributable to the emphasis placed by the National Water Act (NWA 1998) on the human and ecological “reserve”, both of which are critical during periods of low flow.

As far as species selection is concerned, intensive biomass production is likely to favour coppicing Eucalyptus species due to their rapid growth, despite a relatively high ash content after combustion. However, pines will still be considered due to better pellet quality producing less ash, particularly where multiple-use of tree biomass is practiced (e.g. quality saw timber used for conventional sales, with off-cuts and branches used for biomass production). The implications of this in terms of water-use are that Eucalyptus species use more water than pines and wattle in turn. Allocation is made for these differences in the current SFRA water-use licensing system, however, changes from one species to another (e.g. Pine to Eucalyptus) will constitute an increase in water-use (greater streamflow reduction) and hence will be subject to species exchange adjustments to existing water-use licenses.

Given the scenarios discussed in this section, the evidence suggests that higher stand densities and faster growth rates (earlier canopy closure) will increase water-use while shorter rotations will reduce water-use. In general, however, a move from conventional pulpwood and saw-timber plantations to intensive biomass production plantations is likely to result in increases in water-use per unit area. If large-scale changes to this form of land-use are to be approved innovative solutions will be required to offset the increased water resource impacts. Options include accelerated clearing of high water-using invasive exotic trees from riparian areas, and possible replacement with low water-using indigenous tree species of high economic and ecological value (see Gush et al. 2011; Wise et al. 2011).

### ***10.2.2 The Biodiversity Threat of Commercial Timber Plantations***

With the growth of the global human population, the demand for food, wood and fuel will increase, so more areas of the world will turn to intensive agriculture and timber cropping systems like plantation forestry (Cubbage et al. 2010). Plantation

forestry is a serious risk to global biodiversity, as the plantations themselves are often non-native and contribute little to biodiversity (Samways and Moore 1991; Pryke and Samways 2009; Bremer and Farley 2010). Biodiversity can be directly impacted by the plantations themselves, especially as large amounts of natural habitat are often transformed into plantations. Even so there are often natural areas left between plantation blocks in planted areas. These are left aside as areas of high conservation value such as protected grasslands, wetlands or indigenous forests, or for management requirements such as firebreaks, power lines and vehicle tracks (Samways et al. 2010). While not directly affected by the plantations these natural areas are often indirectly impacted through landscape fragmentation. This fragmentation isolates populations and leads to ecological relaxation, which is the loss of species from these fragments due to stochastic events or loss of ecological interactions leading to further losses of overall biodiversity. Abiotic disturbances caused by timber plantations such as water loss and soil nutrient depletion also complicate the conservation of biodiversity in and around plantations. Ecological networks (ENs) are a way to mitigate many of the adverse effects of plantation forestry on the local biodiversity (Samways et al. 2010). ENs work by connecting as many of the fragments together, although they need to be designed properly and be well managed to optimise their conservation value.

#### **10.2.2.1 The Role of Ecological Networks in Conserving Biodiversity in Highly Transformed Landscapes**

ENs are systems of natural or semi-natural landscape elements that are configured to **best** conserve and maintain biodiversity and ecological function (Fig. 10.5) (Bennett and Witt 2001). ENs consist of core natural patches that comprise of either existing nature reserves, areas of high conservation value or even areas within plantations that for various reasons remain unplanted, and are then connected by natural linkages (Jongman 2004). These linkages usually are either stepping stone patches or corridors of natural vegetation (Jongman 1995). Corridors are often simply defined as movement corridors for focal species (Hilty et al. 2006), but they also can function as habitats per se, especially when connected among themselves to form ENs. As the aim of ENs is to conserve biodiversity, they also need to include the inherent abiotic and biological complexity of the whole ecosystem (Fig. 10.1) (Jongman 1995). Conceptually much work has gone into the biodiversity value of ENs, although only a few areas of the world have actually implemented them, most notably the Pan European Ecological Networks (Jongman et al. 2011), the greenways in China (Yu et al. 2006) and the South African timber industry (Samways et al. 2010).

These ENs reduce the isolation of populations or even individuals, allowing better gene flow and reduce founder effects (loss of genetic variation that occurs when a new population is established by a very small number of individuals). ENs also allow species to recolonize areas after localized extinctions. This reduction of





**Fig. 10.5** Ecological networks need to conserve ecological processes and services such as hydrology

isolation and fragmentation helps prevent ecological relaxation (the loss species and their interactions) and so prevent further biodiversity loss. This means that when ENs are designed and managed correctly, with large-scale interconnecting corridors and reserve areas, they can play an important role in ensuring connectivity between habitat patches for organism dispersal on evolutionary as well as on ecological time scales (Beier and Noss 1998; Samways et al. 2010) (Fig. 10.5).

#### 10.2.2.2 Designing Ecological Networks

The design of ENs is a major determinant of their effectiveness. For example, stepping-stone patches within ENs, provided they are large enough, are useful for some of the larger animals using the ENs (particularly large mammals and birds). However, continuous corridors of good quality habitat in ENs are still preferable, as they allow smaller, less mobile, animals, such as frogs, insects and spiders, to use the linkages. In fact, these small animals use these corridors as habitats in their own right, giving the corridors themselves their own inherent biological value (Pryke and Samways 2011).

ENs function at the landscape scale. So, to be effective, they need to incorporate as many landscape features as possible. The inclusion of as many habitat types, as well as landscape features, is fundamental to good EN design. For example, grassland ENs with indigenous forests embedded within them have high biological value, not only because of the additional species in the forests themselves, but also

because there are more species in the associated grasslands (Pryke and Samways 2011). This is due to grassland-indigenous forest interface, which is seemingly so essential for some species.

Recently, there has been much interest in the edge effect between the transformed plantation blocks and the corridors of the ENs. One reason for this is that ENs have more edge than occurs naturally because of the linear nature of corridors (Koh et al. 2010). Understanding these EN edge effects is important for conservation planning, in that it determines minimal width of corridors. Edge effects are caused by structural changes along the edge boundary (Cadenasso et al. 2003; Harper et al. 2005), as well as through changes in soil moisture and nutrients (Li et al. 2007). Over time, secondary effects, such as roads and invasion by exotic plants and animals, can further deteriorate the habitat along the edge.

The influence that transformed areas have on the ENs is often a two-zoned effect: the edge zone, which is influenced by the interface between a transformed area and a natural one, and the interior zone, where species richness, abundance and assemblage composition are no longer influenced by the distance to the edge (Cadenasso et al. 2003; Ries et al. 2004). Disturbance on the edge allows generalist species to disrupt natural systems (Pinheiro et al. 2010; Ivanov and Keiper 2010), although given enough space, it gives way to a more valuable interior zone (Slawski and Slawska 2000; Hochkirch et al. 2008).

Edge effects of exotic plantation blocks on indigenous grasslands are larger in size than that between natural forest patches and grasslands (Wilson et al. 2010), while there seems to be no general edge effects between natural Afromontane forest and its associated grassland (Kotze and Samways 2001). The type of transformed landscape also contributes to the extent of the edge, and determines those species found in it, as has been shown with changes in edge zones in rural versus urban contexts (Vallet et al. 2010) and in edges between different age classes of timber plantation blocks (Armstrong and van Hensbergen 1994).

Although some biodiversity responds positively to the edge, and many species have their habitat at the edge (van Halder et al. 2011), it is the interior zone which is of most concern. The reason for this is that the interior is harder to conserve, as it requires enough space for edge zones to completely surround it. When corridors are too small, they consist entirely of edge zone, without the important interior zone. When edge effects for a variety of arthropods are tested between plantation blocks and adjacent grasslands, there are many different responses, but overall edge effects for all arthropod groups are absent beyond 32 m into the grassland corridor (Pryke and Samways 2011).

Although this 32 m edge zone is a conservative estimate of grassland edge effects around timber plantation blocks, this result suggests that corridors of less than 64 m will be mainly edge and have specific conservation value only as disturbed sites. In fact, interiors of the corridors have similar biodiversity to reserves, suggesting that corridors with widths over 64 m have a biodiversity profile similar to that of nearby reserves (Pryke and Samways 2011). Provided corridor width is wide enough, these

corridors have considerable biodiversity value. The 250 m suggested by Pryke and Samways (2001) is appropriate, as this incorporates a great deal of interior space for more sensitive species. Furthermore, a network of larger habitat corridors, as suggested by Samways et al. (2010), will reduce the area of edge zones across the entire network. When planning agroforestry landscapes with conservation in mind, we need to consider the edge zone around intensive land-use areas as a transitional area from a transformed to a natural ecosystem.

The concept of corridor and ENs is based on connectivity to enable organisms to move through the fragmented landscape (Hilty et al. 2006). For arthropods, these concepts need to be put into perspective, especially as dispersal in most species is strongly linked to resource-searching behaviours (foraging, mate or lek location etc.) (Baguette and Van Dyck 2007). Corridors need to be habitats that allow less mobile arthropods to use them as pathways for dispersal. This means that these corridors need to be of high enough habitat quality to encourage resource use (e.g. feeding, breeding etc.). The best way to ensure this high quality habitat is to manage the ENs optimally.

### 10.2.2.3 Managing Ecological Networks

Along with design, management considerations are vital for ensuring that the ENs function optimally. Such management includes using correct fire regimes, grazing densities and controlling invasive plants, to ensure availability of high quality habitat. Using grasshoppers as sensitive indicators, management of the ENs was found to be three to five times more important than EN design (Bazelet and Samways 2011). This means that all the expense and time put into designing and setting up ENs can be undone if ENs are not managed correctly.

Using the correct fire regimes, grazing densities and the clearing of invasive plants are the three most important management considerations for the ENs to work correctly. Fire regimes are critically important for grasslands and need to simulate natural fire regimes as closely as possible (O'Connor et al. 2004). The simulation of natural fire regimes is also important to savanna and fynbos, although fire intervals are usually longer in these biomes. This management technique is often problematic in timber production areas, as fire is a major risk to plantations (Kirkman and Pott 2002). Because of this, managers in fire prone areas tend to burn entire ENs too early, so essentially using the ENs as fire breaks. The consequences of continuous annual burning are not fully understood, but generally it seems that it is not ideal for biodiversity and it would be better to rest the land occasionally (Chambers and Samways 1998; Uys et al. 2004). This would require fire protection zones to be set up next to the plantations, requiring bigger corridors to accommodate them. In fact, bigger corridors would allow half the corridors to be alternately burned between years and so have fire protection value allowing more space to bring fires under control, while being easier to manage, as well as being more accommodating for



**Fig. 10.6** Ecological network with an unburned corridor in the foreground, a burned corridor in the middle and a reserve in the background

biodiversity (Fig. 10.6). An alternative to annual burning of ecological corridors would be to alternate it with prescribed burning operations within plantation blocks where this is possible (e.g. semi-mature plantations).

A way to reduce the fuel load is to allow grazing on the ENs. Ideally native fauna should be used (Fig. 10.7). However, in areas where this is not an option, domesticated animals could be used, as long as their densities are controlled (Fynn and O'Connor 2000). Where fire and grazing are not available as realistic options, then mowing has been shown to have some success for at least grasshopper diversity (Chambers and Samways 1998).

Invasive plants are to some degree controlled in grasslands by fire and grazing. If invasive plants are allowed to take over corridors then they lose their ecological value and essentially become transformed areas themselves (Magoba and Samways 2011). This can lead to the breakdown of connectivity and of the optimal functionality of the EN.

The effectiveness of ENs within the South African timber industry has been assessed for a wide range of different organisms from the habitat base of plants, through to large mammals and birds (Joubert 2011). Much of this research has been based on arthropods, as they are small, hyperdiverse, habitat sensitive, resource dependent, ecologically important and can be sampled in large numbers (Bazelet 2011; Pryke and Samways 2011). Although there is much variation among these groups, they all benefit from ENs, provided that the ENs are well designed (corridors wide enough and all landscape feature are considered) and good quality habitat is maintained (Figs. 10.6 and 10.7).



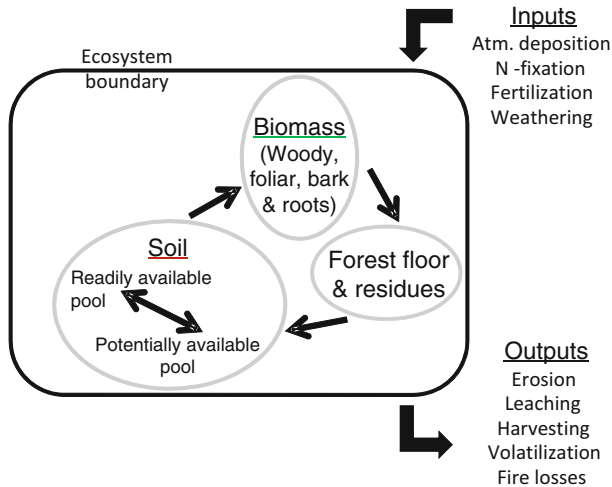
**Fig. 10.7** Native animals grazing in an ecological network

### 10.3 Impacts at the Site Level

Management of the ecological impacts at the site level has the goal to ensure sustainable productivity over the long term. Site-level impacts such as nutrient depletion, soil loss or degradation, pest and disease outbreaks or uncontrolled fires are likely to impact negatively on the sustainable production capacity of the system. Careful monitoring of intensively managed systems is thus needed, along with strategies to mitigate against negative site impacts that may take place. It is in this regard that agro-forestry systems and mixed cropping systems with nitrogen fixing trees can contribute to mitigate against nutrient depletion. Silvicultural systems have to be so designed to offer a degree of buffer against biotic or abiotic risks. The discussion on mitigation strategies and ecological buffering is supported by information from case studies in plantation and agro-forestry systems.

#### 10.3.1 *Nutritional Sustainability*

The biogeochemical cycling of nutrients in forest ecosystems has been well described for several forest ecosystems (Jorgenson et al. 1975; Likens and Bormann 1995; Ranger and Turpault; du Toit and Scholes 2002; Laclau et al. 2005, 2010a; Dovey 2012). Nutrients reside within forest and plantation systems in several nutrient pools that differ in size and in the form in which a particular plant nutrient is held. Figure 10.8 is a simplified representation of a number of basic nutrient pools as well the movement of nutrients (usually termed nutrient fluxes) (a) into



**Fig. 10.8** Schematic representation of a simplified forest biogeochemical cycle (nutrient pools and fluxes within a forest ecosystem) as well as inputs/outputs from such a system (After Ackerman et al. 2013)

and out of the system (as part of the so-called input–output budget), and (b) among pools within the system (Ranger and Turpault 1999; du Toit and Scholes 2002). Forests are at risk of malnutrition and subsequent decline in productivity if the biogeochemical cycles of nutrients are decoupled in time or space. For example, an ecosystem can systematically be depleted of nutrients if outputs exceed inputs by a large margin. Several short rotation commercial forestry systems where only stemwood is harvested, are not at risk of nutrient depletion (du Toit and Scholes 2002; Ackerman et al. 2013). However, in the context of this chapter, increased nutrient removals in bio-energy plantations resulting from the harvesting of tree crowns, bark (and even roots) in addition to the conventional stem wood harvest is probably the biggest concern (see examples in Fig. 10.9 below). Forest productivity can also be negatively affected if nutrients are not necessarily lost from the system, but end up in pools that are (temporarily) decoupled from their usual cycle and thus unavailable to the stand during a particular phase of growth. An example would be the lock-up of nutrients in the forest floor (Morris 1986) or the precipitation of a micronutrient in a plant-unavailable form following measures to raise soil pH, such as liming. In practice, management operations that affect nutrient pool sizes usually have a direct or indirect effect on nutrient fluxes too. For example, slash removal for bio-fuels will affect (for example) the content of nitrogen on the site, but also the rate of mineralisation which transforms N in a form that can readily be taken up by trees (du Toit and Dovey 2005; Deleporte et al. 2008; Gonçalves et al. 2008b; Mendham et al. 2008). Similarly, fire will affect the quantity of N and P oxidised during burning, but it may also strongly influence the availability of soil N through changes in mineralisation rates after the fire, as well as P availability (through changes in soil pH caused by the increase in pH (the ash-bed effect).



### 10.3.1.1 Gauging Nutritional Sustainability

The only way in which a comprehensive understanding of nutritional sustainability of a particular forest system can be gained, is by studying the majority of the more important nutrient fluxes in the system as brought about by specific management regimes or operations, (e.g. Ranger and Turpault 1999; du Toit and Scholes 2002; Laclau et al. 2005, 2010a; Dovey 2012). However, it has been known for a long time that nutrient dynamics may change significantly with the stage of stand development (Miller 1995). It follows that such studies has to be repeated in time, (or perhaps be done in a chronosequence approach) to paint the full picture. This fact, coupled to the reality that many different site types (and potentially even more than one management regime per site type) will have to be studied, makes it a very daunting task.

Nutrient fluxes consist of some processes in the biogeochemical cycle that are comparatively easy to monitor, e.g. uptake from the soil to the plant or return to the slash/soil layer following clear felling, thinning or through litterfall and fine root turnover. However, it also includes more complex processes that are difficult to quantify, such as transformations in the soil that are specific to each nutrient and the prevailing microclimatic and edaphic conditions in a specific soil, which are largely responsible for plant nutrient availability (e.g. nitrogen mineralisation, sorption and desorption processes on soil phosphate, oxidation/reduction processes on soil sulphur, etc.) This chapter deals with these fluxes in a simplistic way (see Fig. 10.8), and the reader is referred to soil chemistry texts for more detailed descriptions of these processes.

### 10.3.1.2 Nutrient Input–Output Budgets and an Index of Nutritional Sustainability

A simple index of nutritional sustainability has been proposed by du Toit and Scholes (2002) to gauge the nutritional sustainability of a variety of management regimes across different site types. While this is a fairly coarse indicator (it does not take transformations within the system into account) it is comparatively easy to use because it requires estimates of only (a) the larger input–output fluxes and (b) the major system nutrient pools sizes of the macronutrients. These can be estimated to an acceptable degree of accuracy in many regions of the world. Minor nutrient fluxes (such as weathering rates in very old soils) does not have to be gauged to high degrees of accuracy as they will not materially influence the system. Du Toit and Scholes (2002) proposed to express the net nutrient output from a system as a fraction of either (1) readily available or (2) potentially available nutrient pools in the system, to judge potential short- and long term effects. The index of nutritional sustainability thus developed carries the acronym pINS, where:

$$p(INS) = -\log_{10} \left( \frac{\text{Net annual nutrient loss}}{\text{Nutrient pool}} \right)$$

**Table 10.2** Scenario's for biomass harvesting intensity per genus and per silvicultural regime in the case study of Ackerman et al. (2013)

Scenario	Genus and silvicultural regime	Harvesting intensity
A	Eucalypt pulpwood	Regular (75 % of stem wood only)
B		As above plus slash burning
C		Whole tree harvest with 75 % efficiency
D	Pine pulpwood	Regular (75 % of both stem wood and bark)
E		As above plus slash burning
F		Whole tree harvest

In its original form, du Toit and Scholes (2002) made provision for the nutrient pool to be either can be calculated as the readily plant available fraction or the long term (potentially) plant available fraction. We have used the fraction of the nutrient pool that is likely to be available to trees on a time scale of months to several years, because estimation of the long term potentially available pool sizes requires more developmental work.

Notes:

- In most intensively managed forestry and agricultural systems, there is a net loss of nutrients over time until such time as ameliorative action is taken.
- If the net input–output budget does not constitute a loss, it is simply reported as a gain and the pINS index is not calculated.
- A value of 1 (log scale) has been tentatively chosen as a value that should raise a red flag (i.e. if the net nutrient loss is more than 1/10th of the readily available nutrient pool) as defined by du Toit and Scholes 2002, the site may be at risk of nutrient depletion if the current management regime continues to be implemented.
- A feature of the pINS index is that different scenario's can be developed, for example where the portion of biomass harvested is increased or the rotation length is shortened (as is likely to happen in bio-energy crops), and the new scenario's can be compared with conventional systems.

This approach was developed further by Dovey and du Toit (2006) and by du Toit and Dovey as part of a more comprehensive study reported by Ackerman et al. (2013) dealing with nutrient fluxes and nutrient pools, respectively, in South African short-rotation plantation systems. Ackerman et al. (2013) chose three scenario's each for short-rotation pine and eucalypt systems as shown in Table 10.2. These scenario's were applied to 28 short-rotation pine sites and 21 short-rotation eucalypt sites in Southern Africa for which adequate data was available.

The pine sites in the study of Ackerman et al. (2013) virtually all showed net gains in N and P with relatively higher pINS values for K, Ca and Mg than the eucalypt sites. The main reason for this is twofold: Firstly, the longer average rotations of pine pulpwood (18.0 years as opposed to 7.1 years for eucalypt sites tested) has the effect that harvesting outputs are offset by a larger number of years' worth of atmospheric deposition. The regions where most of the pine test sites are



located receives higher loads of atmospheric deposition than other remote rural locations, due to its proximity to a large number of coal-fired power plants (Olbrich 1993; Lowman 2004). Secondly, the pine sites are mostly located on more fertile sites in the region (clays and loams with high organic matter contents in the topsoils) whereas many of the eucalypt test sites in the test battery are located on sandy soils with low levels of organic matter. Nonetheless, the case study does illustrate the relative resilience from the most vulnerable to the more resilient sites in the region.

A summary of the results for the eucalypt sites is presented in Fig. 10.9, where it can be seen that the pINS value frequency curves all shift to the left (lower pINS values) when moving from Scenario A via B to C. This means that under scenario C (whole tree harvesting with 75 % efficiency), a large number of stands are coming close to a situation where nutrients may be depleted over the scope of several rotations unless corrective action is taken. Many tropical soils under short-rotation plantations have undergone more intensive leaching and are substantially poorer in nutrient pools and organic carbon, than Southern Africa's eucalypt sites presented in the case study in Fig. 10.9 (Gonçalves et al. 1997; Deleporte et al. 2008; Tiarks and Ranger 2008). It follows that intensively harvested bio-energy plantations on infertile sites are at much higher risk of nutrient depletion than is the case for the Southern African eucalypt data set.

It is important to keep in mind that increasingly intensive harvesting regimes and shortened rotations may result in a net loss of nutrients in many plantations, yet this may not have any immediate effect of decreasing the subsequent rotation's productivity. This may happen because the new rotation may still have access to sizable pools of readily available nutrient reserves on the site. Furthermore, there may not always be a positive growth response when the net loss of nutrients are replaced (e.g. by fertilization). Most intensively managed, short-rotation plantation forests respond mainly to macronutrient additions of N and P (Gonçalves et al. 1997). Indeed, it is only after several rotations of intensive biomass harvesting in plantations and/or plantations grown on poor soils that widespread responses to the addition of base cations started to become common (Gonçalves et al. 2008a). In short rotation tree stands where fertilization regimes are very basic or non-existent, there may be a net loss of several nutrient elements, and although there may not be an immediate growth response to (say) replacing Ca lost during harvesting, there will still be a constraint on the ability of the site to supply Ca in successive rotations. Furthermore, Laclau et al. (2010a) have presented evidence to show that short-rotation plantations of eucalypts may be capable of extremely efficient nutrient conservation and cycling, but that many such plantation systems in the tropics apparently depend on soils having been pre-enriched with nutrients by the natural vegetation before the plantations were established. For these reasons, and because bio-energy plantations are usually grown on very short rotations with large percentages of the biomass harvested, it would be wise to monitor nutrient exports in bio-energy plantations very intensively, and to upgrade the fertilization regime where necessary.

One of the most important measures to ensure sustained productivity on infertile sites, is the conservation of organic matter in the system (Laclau et al. 2010a, b).

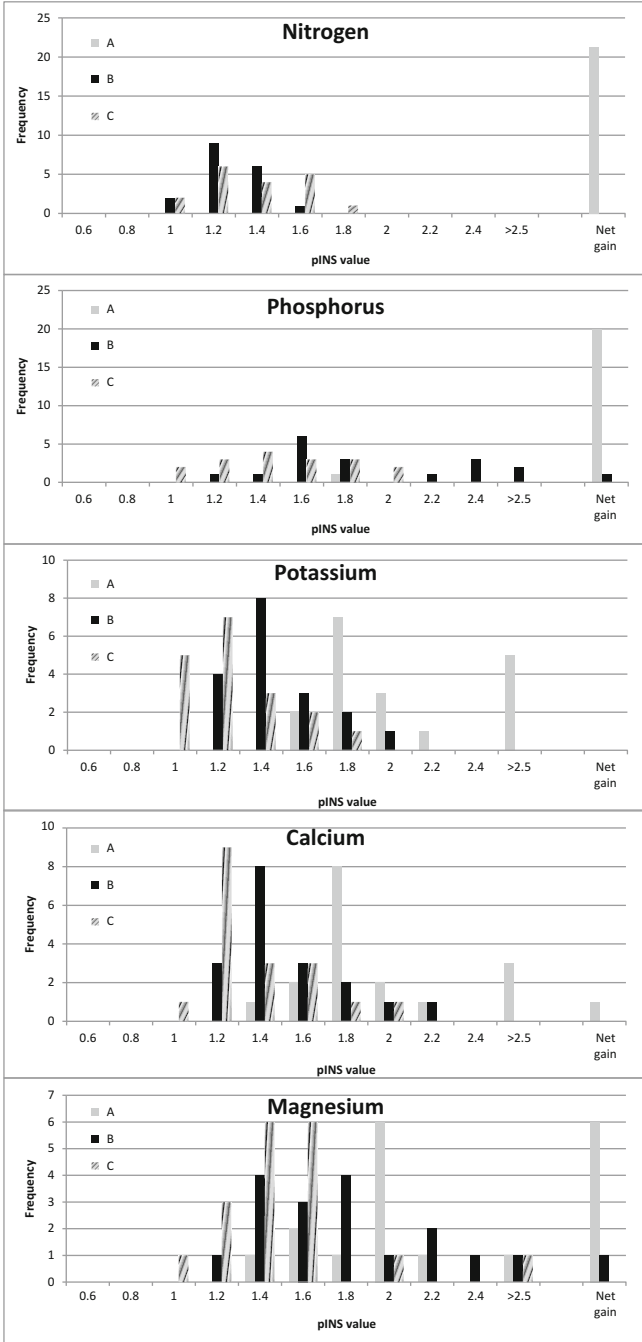
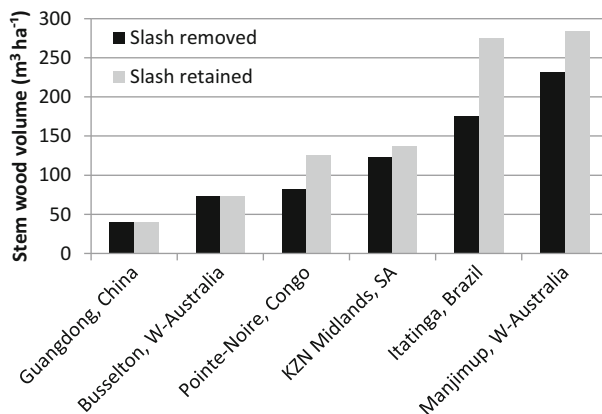


Fig. 10.9 pINS indices for five macronutrients (after Ackerman et al. 2013) calculated for 21 short-rotation eucalypt crops under scenarios A, B and C (Refer to Table 10.2 for scenario explanation)



**Fig. 10.10** Stemwood volume production in short rotation eucalypt case studies under slash retention and slash removal scenario's (After Nambiar and Kallio 2008)

Several experiments in a tropical network study (reviewed by Nambiar and Kallio 2008) tested the effects of slash management (and in particular, slash removal) on short rotation stand productivity. The slash removal treatments in this trial series constituted the removal of the slash plus the un-decomposed portion of the forest floor, which is a more intensive treatment than whole aboveground tree harvesting (which effectively only excludes the return of harvesting residue to the soil). However, it does give an indication of what can potentially be the result after successive rotations of either whole tree harvesting or some form of intensified biomass harvesting. The stand productivities (stem wood volume production at rotation end) of eucalypt case studies in this network of trials under slash retained and slash removed scenario's are given in Fig. 10.10.

The case studies by (Deleporte et al. 2008 (Congo); du Toit et al. 2008 (South Africa); Gonçalves et al. 2008b (Brazil); Mendham et al. 2008 (Manjumup, Australia)) all showed decreases in forest productivity following removal of harvesting residue and un-decomposed material in litter layers. The largest decline in stand productivity due to slash removal occurred in sandy soils (arenosols) and dystrophic loams (oxisols), both with low topsoil organic matter contents. This result underscores the point that a combination of poor soils, short rotations and intensified biomass harvesting means that many bio-energy plantation systems in the warm climate countries will not be nutritionally sustainable in the long run unless significant additional nutrient inputs are made. Inputs may be in the form of fertilizers, and/or ash replacement (from biomass burners), and/or incorporation of N fixation, either through mixed cropping (Binkley and Giardina 1997; Bouillet et al. 2012), or crop rotations with symbiotic N-fixers in the broader silvicultural management system.

### 10.3.2 *Soil Conservation and Protection*

The major threats to the ability of soils to sustain highly productive forest, other than nutrition-related issues discussed in Sect. 10.3.1, is soil displacement, soil erosion and soil compaction.

When viewed simplistically, soil compaction sensitivity by mechanical equipment is strongly related to machine mass, soil texture and soil water content at the time of impact. Coarse textured soils such as sands can be trafficked by fairly heavy loads with low risk of compaction in both wet and dry conditions (Smith et al. 1997a, b; du Toit et al. 2010; Ponder et al. 2012). Soils with very sandy textures seldom suffer from compaction problems and may even experience improved water holding capacity and sometimes improved growth following moderate compaction (Smith et al. 1997b; du Toit et al. 2010; Ponder et al. 2012). Conversely, soils with a fairly even mixture of particle size classes such as sandy loams, loams and sandy clay loams are moderately compactable when dry but strongly compactable when moist (Smith et al. 1997a, b; du Toit et al. 2010). Harvesting operations should where possible be scheduled to avoid soils with an even particle size distributions during wet conditions. Furthermore, machines travelling on plantation soils could be matched with the soils load bearing capacity, which is strongly related to texture, organic matter and initial bulk density (Ampoorter et al. 2012). Most compacting occurs in soils with the first few passes of machines over a specific area. This fact, combined with efforts to limit the spatial extent of compaction, calls for controlled vehicular movement on designated skid trails (Ampoorter et al. 2010). The effects of compaction in short-rotation bio-energy plantations can be thus minimised (a) by using designated skid trails, (b) by matching machine mass with soil texture and thus with load bearing ability, (c) by limiting harvesting operations on fine textured soils during wet conditions and (d) by retaining as much of the harvesting residue as is possible, given the harvesting system chosen.

The sustainability of bio-energy plantations will be severely compromised if erosion rates significantly exceed soil formation rates. Soil erosion is generally affected by rainfall & runoff, slope gradient and length, soil erodibility, vegetation cover and soil surface cover, soil tillage, and any other man-made support practices to limit erosion, e.g. contour banks or windbreaks.

On a site with a give soil, slope and climate, forest management practices can play a major role in limiting erosion. The single factor giving the most effective protection against wind, rain-splash and water erosion is the presence of absence of a soil cover layer, e.g. a mulch layer in agricultural fields or, the forest floor/slash layer in plantations and forests (Morgan 1995). This is echoed by several case studies where forest floor layers have been removed or destroyed by intensive fires (du Toit 2002; Miura et al. 2003; Fernández et al. 2004). Soil loss through erosion in plantation-based systems therefore depends very strongly on management of the slash and the forest floor. If destruction of the slash/forest floor is combined with other factors that favour soil erosion, it often results in an increase in erosion by orders of magnitude. Sherry (1953, 1954, 1961, 1964, 1971) documented the effects

**Table 10.3** Average soil loss over two crop cycles of short-rotation *Acacia meurnsii* plantations under varying management regimes

Number of erosion enhancing factors present	Description	Soil loss (tons ha <sup>-1</sup> )
One	Steep slopes	Nil
	Slash burning	0–0.8
	Hoeing to control weeds	0–0.4
Two	Burning on slopes	11.4
	Burning and hoeing	10.1–17.6
	Hoeing on slopes	4.6
Three	Burn + Hoe on slopes	113.7

From du Toit (2002); after the work of Sherry (1953, 1954, 1961, 1964, 1971)

of three factors: slash burning, slope steepness and soil tillage (and combinations thereof) on soil erosion. The results have been re-analysed by du Toit (2002) and show an order of magnitude increase in soil erosion with the number of factors present (Table 10.3).

In non-planted areas, road design, construction and maintenance is critically important to minimize erosion because cuttings and road construction will lay the soil bare and poor road drain maintenance may cause the water flow to be concentrated in certain areas.

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