Chapter 5
Maintaining Biodiversity during Biofuel Development

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5.1 Introduction

This chapter focuses on four key issues:

- Understanding the opportunities and threats biofuels pose to biodiversity.
- Understanding how impacts can be predicted and modelled as a component of multi-criteria decisions surrounding strategic decision making on whether to undertake a biofuels programme or not.
- Operational planning at the biofuel plantation level to minimise negative biodiversity impacts.
- Minimising the risk of invasive alien species (IAS) resulting from biofuel production.

Most of this chapter is focussed on the impact of feedstock plantations for liquid biofuels. However, the same techniques can usually be applied to feedstock for other types of bioenergy. This is illustrated in some of the provided examples.

5.2 Why Consider Biodiversity Impacts?

5.2.1 What is biodiversity?

Biological diversity, normally referred to as biodiversity, is defined by the United Nations Convention on Biological Diversity (UNCBD 1973) and the Millennium Ecosystem Assessment (MA 2005) as: *"the variability among living organisms from all sources including, inter alia, terrestrial, marine and other aquatic ecosystems and the ecological complexes of which they are part; this includes diversity within species, between species and of ecosystems."* The term is used to cover all forms of life, but for practical purposes is often expressed with reference to specific taxa, e.g. biodiversity of plants, biodiversity of mammals, biodiversity of insects, etc (see section 5.5.2). In most common usage it is the disappearance or decrease in abundance of naturally-occurring (endemic or indigenous) species that is implied when 'loss of biodiversity' is being discussed. When considering biodiversity it is often convenient to subdivide the landscape into units of area which have similar biodiversities such as habitat types or ecosystems[15]. The habitat type is typically defined by eco-regions, biomes or broad vegetation type such as lowland forest, dry deciduous forest, grassland or wetlands when working at a global or national level. More detailed local classifications could be used when working at a smaller scale, such as at a plantation level.

Biodiversity can be expressed at a number of different scales, with the following three scales (or levels) of diversity commonly considered:

1. Genetic diversity is the differences in the genetic composition of individuals of the same species. A sister and brother are of the same species (humans, or more correctly *Homo sapiens*), but have differences in genes that make them different. Unrelated people of different regional origins will have greater differences in genes than closely related people. The same applies to non-human organisms.
2. Species diversity is the variety of different species. For example, buffalo and elephants are different mammals, *Eragrostis curvula* (love grass) and *Eragrostis gummiflua* (Gum grass) two grass species of the genus *Eragrostis*.
3. Ecosystem diversity is diversity between different habitats or ecosystems.

It is useful to think of diversity at this and other levels as having three attributes:

- Diversity in composition (i.e. which ecosystem types are present)
- Diversity in structure (are the patches large or small, tall or short, connected or fragmented?)
- Diversity of function (do they all work the same way and produce the same ecosystem services?).

Depending on the application, ecosystem diversity may compare broad habitat types such as tropical forests to tropical grasslands, or might be measured at a finer scale of different types of forests within tropical forests. The diversity could also be expressed as diversity of different functional types of organisms rather than as difference in species. This is useful due to the fact that very different species may functionally play very similar roles within the ecosystem. For some applications it is the diversity of functions that species play in an ecosystem that may be more important to the ecosystem's integrity than the diversity in species. In this regard, species can be grouped into functional types based on the role they play in the ecosystem or into response types based on the way they respond to different stresses and disturbances. Biologists, when describing or measuring species biodiversity, use the terms alpha (α), beta (β), and gamma (γ) diversity to describe different attributes of the diversity. Alpha (α)-diversity is the biodiversity within a patch of a given size (usually expressed as the number of different species present, or 'species richness'). Beta (β)-diversity is a measure of the degree of change in species composition along a gradient – in other words, if you were to measure another patch near to the first patch, how many shared species would there be? (γ)-diversity refers to the total species richness over a large area or region, and is strongly influenced by how many different patch types there are. This is

[15] Biome, vegetation type, habitat type and ecosystem can all be used to describe unique assemblages of biodiversity. Key differences relate to the scale of analysis and the basis for the grouping (e.g. based on vegetation or ecological process). Precise definitions are beyond the needs of this text, and the key consideration is that these are used as ways of grouping areas of similar biodiversity for analytical purposes.

best explained by considering the hypothetical species turnover along a hypothetical environmental gradient as illustrated in Figure 5.1. The 10 species in Figures 5.1a and 5.1b are artificially divided into two habitat types. In Figure 5.1a all species are common to both habitat types, but in very different proportions. In Figure 5.1b, habitat type A has three unique species, whilst habitat type B has five unique species and only two species are common in both habitats. As will become important in later sections, this uniqueness of biodiversity within a single habitat has consequences when considering conservation status and impacts of habitat destruction. Totally transforming all of habitat B in Figure 5.1a would not result in the total loss (*extirpation*, or *extinction* if that was the last representative of the species on Earth) of any species, but would result in some common species becoming extremely rare. By contrast, losing the same area in Figure 5.1b would result in the total loss of five species, with a sixth species becoming extremely rare. Figure 5.1a also shows how some species can have very wide habitat tolerances - these are often referred to as generalist species. Species in Figure 5.1b, by contrast, have

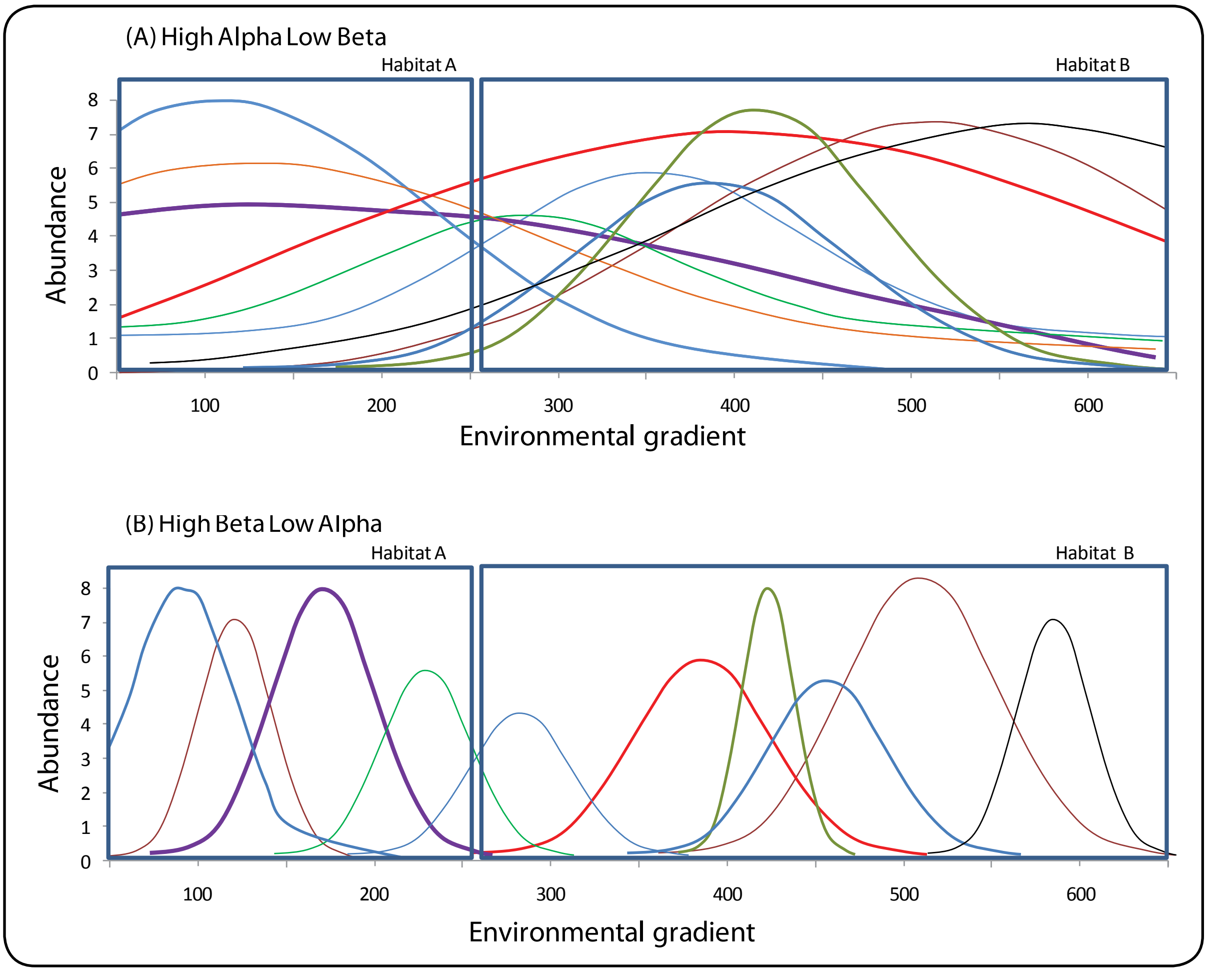

Figure 5.1: **A symbolic illustration of biodiversity changes over an environmental gradient, illustrating the difference between alpha and beta diversity. γ-diversity is the same in both figures as the same 10 species are found in both. Each species is represented by a coloured line, with the height of the line representing abundance (individuals of the species per unit area) at each point on the environmental gradient. The figure also illustrates how species can have wide environmental tolerances (generalists) as in (A) or narrow tolerances (specialist) as is in (B) and how this can lead to habitats with unique species, versus habitats that though different, share all of the same species. In both figures the environmental gradient is split at 250, two artificially defined habitats A and B.**

narrow habitat tolerances and are referred to as specialists. If a species range is naturally restricted to only one area in the world then it is referred to as an *endemic* species (to that area, and an *alien* outside of that area).

Since biodiversity is multi-faceted, quantifying biodiversity and its changes is non-trivial. Clearly, measuring biodiversity comprehensively requires more than simply making a list of the species that are present. Simply listing the species present tends to create a bias toward the well studied and easy to find and identify taxa. A large number of indices on biodiversity have been proposed (see Magurran (2004) for a recent summary). It is beyond the scope of this Chapter to go into details of all the potential biodiversity measures. Nevertheless, it is important to select one or a few indices that meet the needs of the specific task. Many biodiversity indices impose unattainable data needs, focus on a single scale and aspect of the biodiversity hierarchy, or are scale-dependent and thus hard to interpret in a comparative context (Biggs et al. 2004). For the purposes of this chapter we focus on the Biodiversity Intactness Index (BII), for reasons summarised in section 5.5 and expanded in Scholes and Biggs (2005) and Biggs et al. (2006).

Biodiversity is not spread evenly around the globe. In general, species richness increases from the Polar Regions toward the tropics. Developing countries, which are largely within the tropics, therefore tend to have far higher biodiversity than is found in more temperate regions of developed countries. As a general rule tropical regions are historically less transformed than temperate regions, and hence contain more of their original biodiversity. Recent accelerated levels of land transformation in the tropics, including land transformation for biofuels, is placing an increasing threat on tropical biodiversity. As the level of biodiversity is so high in these reasons, so is the potential for large amounts of biodiversity loss (MA 2005).

5.2.2 Why is preserving biodiversity important?

The extinction of species has occurred over the period of historical record (the last few hundred years) at a rate estimated to be one hundred times higher than the long-term average rate calculated from the fossil record (MA 2005). This accelerated rate is attributable almost entirely to anthropogenic (human-induced) causes, principally habitat loss and overharvesting (Sala et al. 2000; Sala et al. 2005). Modelled predictions of extinction rates in the twenty-first century predict a further acceleration (MA 2005; GBO3 2010).

The loss of biodiversity can have direct negative consequences on humankind. Preserving biodiversity is therefore both an ethical and an economic consideration. Biodiversity loss is considered of such global significance that a UN Convention is in place to facilitate the conservation of biodiversity, which has been signed by virtually every country in the world (United Nations Convention on Biological Diversity UNCBD 1973). Understanding the direct importance of biodiversity to humans is best explained though the concept of ecosystem services. Ecosystem services are the benefits people obtain from nature. The Millennium Ecosystem Assessment (MA 2005) distinguishes four key clusters of ecosystem services: provisioning; regulating; cultural and supporting (Figure 5.2). All our food, and much of our fuel and fibre, is derived from living organisms in ecosystems (including highly modified ecosystems, such as croplands and plantations) – these are examples of provisioning services. Though only a small

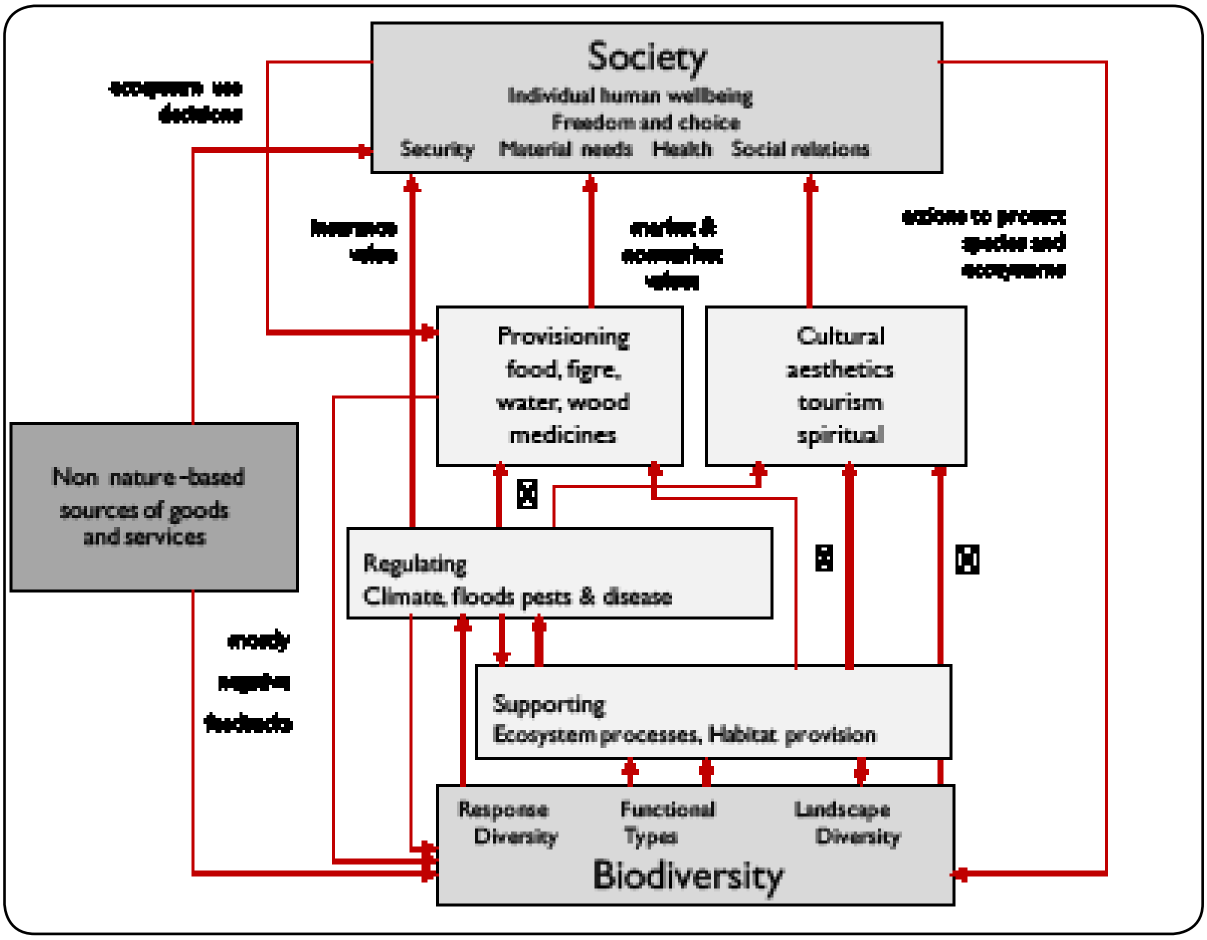

Figure 5.2: **The pathways and processes by which biodiversity influences ecosystem services, and ecosystem services influence human wellbeing. The value of supporting services, most of the value of regulating services and most of the aspects of biodiversity is contained within the value of the directly-used provisioning and cultural services. These underlying elements can influence the direct services through altering the mean magnitude of the service (μ) or its variability in time (σ) or its variability in space (γ).** From Kinzig A., C. Perrings and R.J. Scholes unpublished.

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fraction of organisms have been domesticated for human use, the gene pool in wild relatives and in as-yet unused organisms is important in ensuring that we maintain our ability to adapt crops and livestock to changing environmental conditions, and to discover and develop new medicines, compounds and structures. Many widely-used medicines originated from plants or other organisms and it is almost certain that many more useful compounds will be discovered over time.

The deliberate simplification of ecosystems, for instance through mechanised monocultural cropping using high inputs of nutrients, water and pesticides, has been the key mechanism for increased provisioning services such as food, and fuel over the past century. This has generally been at the cost of other services - even of other provisioning services such as water and biodiversity (MA 2005).

Loss of regulatory services can have devastating impacts. An example is the 'dustbowl' in the American Midwest in the 1930s, a consequence of converting diverse, perennial natural grasslands to annual cropland. The loss of ground cover to bind the soils, coupled with drought, gave rise to extensive wind erosion and large dust storms. In other examples, degradation or invasion by alien species has resulted in changes to the hydrological function of a catchment which has in turn led to increased flooding and increased river pollution in one case and decreased river low-flows in the other. The phenomenon of global climate change is to some extent a loss of regulatory services. About a quarter of CO_2 emissions are linked to land use change, including deforestation (IPCC 2007).

Cultural services from the environment are important for both the spiritual, physical and psychological wellbeing of people, as well as having economic significance in aspects such as tourism and recreation.

Supporting services underpin all other environmental services. A breakdown in the supporting services (which include aspects such as soil formation, primary production and nutrient cycling) will reduce the ability of the environment to generate provisioning, regulating and cultural services.

Biodiversity is important in all ecosystem services, directly or indirectly, although the relationship is often quite complex and subtle. There is firm evidence that diverse ecosystems, in general, are both more productive and more resilient to stress than less diverse ecosystems (MA 2005).

In addition to the direct human benefits derived from biodiversity (the so-called 'utilitarian' value of biodiversity), there are also ethical reasons as to why humans should maintain biodiversity (sometimes referred to as 'intrinsic value' arguments).

5.3 Likely Impacts of Biofuel Production on Biodiversity

Biofuel expansion, if not carefully regulated, has the potential to have very high impacts on biodiversity, especially as a consequence of habitat loss. It is counter-productive to fight one global environmental problem, climate change, and simultaneously exacerbate a second global environmental problem by increasing biodiversity loss. This is, however, a complex tradeoff since climate change is also predicted to have profound impacts on biodiversity (Thomas et al. 2004). Changes in temperature and rainfall regimes will displace habitats. Since temperatures are predicted to rise this will displace the zone of climate preference for most species polewards or to higher altitude. It is likely that a significant fraction of species will totally lose their current habitats and will thus ultimately become extinct unless intervention steps are taken (Hannah et al. 2002; Thomas et al. 2004). Though biofuels can in part mitigate climate change impacts, this positive impact is likely to be very small compared to the high negative land transformation costs. The synergistic impact of both land transformation and climate change will have a double blow to biodiversity with transformed habitats making it much harder for species to adapt to climate change.

5.3.1 Habitat loss

Land cover change (both direct and indirect) is the single biggest biodiversity concern from biofuel feedstock production. To grow biofuels will require land, and since biofuels are grown mostly as monocrops, this will result in the loss of most existing biodiversity from the area planted. When replaced by a biofuel monocrop, the structural, functional and compositional diversity of the original habitat is replaced, with a single functional response, highly reduced structural diversity and very limited species diversity. This impact will be greatest where previously intact natural landscapes are transformed. Land clearing results in habitat loss. In response, some policies encourage biofuels to be grown on so-called degraded land or land formerly used for other monocultural crops, and in this case the additional impact on biodiversity loss is small. But if vast volumes of biofuel are to be produced, as would be required to meet the more than 50 biofuel policies and mandates worldwide (Peterson 2008), then this cumulative demand will require the opening up of new land for biofuel plantations.

5.3.2 Impacts of iLUC

Only locating biofuel plantations on land already used for agriculture or grazing does not automatically reduce the risk of biodiversity loss since there is a very real threat of causing indirect land use change (iLUC). This process is also known as 'leakage' or 'displacement' and is shown in Figure 5.3. Put simply, because current agricultural land is converted to biofuel, new agricultural land needs to be sought to make up for the agricultural shortfall resulting from the reduced agricultural production. iLUC is difficult to quantify because the impacts of iLUC are by definition expressed in spatially separate locations from the biofuel production area itself. These locations could be remote being in other countries, or even on the other side of the world. There is strong circumstantial evidence that biofuel expansion has resulted in iLUC (see Chapter 4 this volume). For example, indirect land use change attributed to biofuels is considered one of the drivers for the current high rates of Amazon deforestation (Morten et al. 2006). Ways to reduce the risk of causing iLUC include:

- Increasing agricultural productivity (reducing the need for increased agricultural area)
- The use of biofuel by-products (biofuels might be able to provide food or feed as well as fuel)
- 'Second generation' biofuels (the so-called second generation biofuels make use of crop residues, allowing fuel and food to be come from the same land)

The biodiversity assessment methods discussed in this chapter relate predominantly to direct land use impacts, but are also applicable to situations where indirect land use impacts can be quantified.

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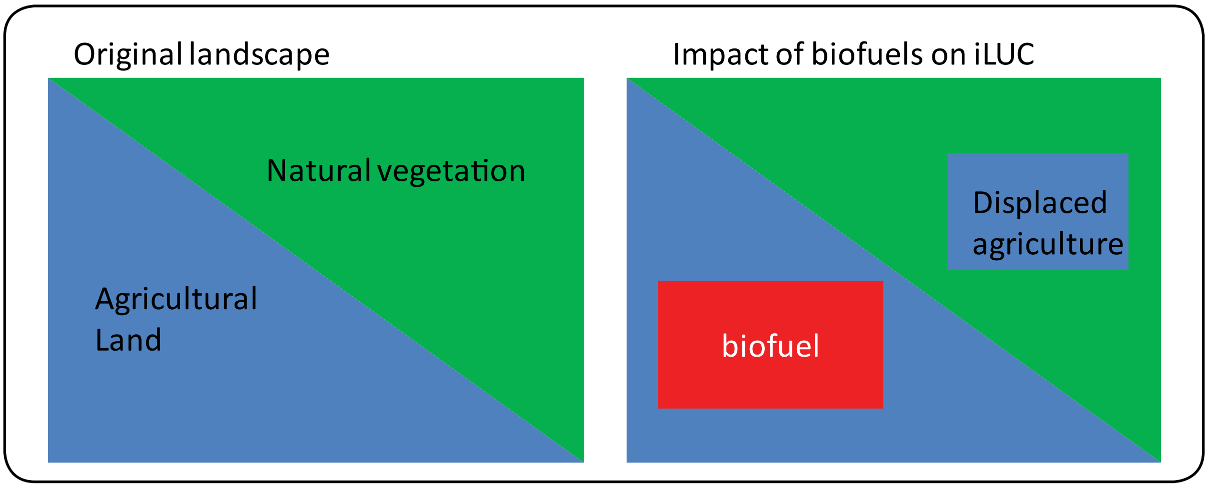

Figure 5.3: **A simple diagrammatic example of iLUC on land transformation of natural vegetation. Though biofuels feedstock growing took place on agricultural land, natural vegetation was converted to make up the agricultural production loss due to biofuel. Note: this displaced agriculture need not occur locally and could be of a different spatial extent due to different productivity levels.**

5.3.3 High diversity importance of developing countries

As stated above, most developing countries are located in the tropics, and hence in intrinsically biodiverse areas. This therefore increases the potential for biodiversity loss from habitat change. In addition some of these tropical habitats have been identified as areas with high levels of threatened species. For instance, the Millennium Ecosystem Assessment found tropical and sub-tropical moist broadleaf forest to be the terrestrial habitat type with by far the highest number of threatened vertebrate species (MA 2005). This habitat type is undergoing rapid transformation to oil palm plantations in SE Asia and to soybean fields in Brazil. Both these crops are potential biofuels, though currently only a small percentage of their oil is used for biofuel with the remainder being used for food or fodder. Tropical and subtropical dry broadleaf forest, and tropical and subtropical grassland, savanna and scrubland all have high levels of threatened vertebrate species (MA 2005). These ecosystems are potential locations for expansion of the oilseed *Jatropha curcas* as well as for numerous other grain and oilseed crops. Sugarcane is also a potential biofuel crop in these habitats, especially where water is available for irrigation. This potential for high biodiversity loss places an added burden on tropical areas when considering potential biofuel expansion.

5.3.4 Impacts from invasive alien species

The introduction of invasive alien species (IAS) is a direct and indirect threat to biodiversity though it has received little attention compared to other sustainability issues associated with biofuel production. Alien species are plant or animal species not native to a specific location. If these species are introduced and can reproduce, establish and expand on their own, then they are considered invasive. That is to say they becoming naturalised and a pest in their new environment (Pheloung 2003). The concern with IAS is that they are able to competitively displace the indigenous species, largely because they lack natural predators in their new environment. They might even alter the habitat by being of a different structure (e.g. tree species in grassland), through impacting on fire regimes (for instance by being highly flammable and tolerant of fire), soil fertility (e.g. through fixing nitrogen), soil hydrology (through sustained high transpiration) or other aspects of the environment. The total economic cost of IAS can be enormous and controlling IAS is costly and difficult. Alien invasive species have, for instance, been estimated to cost the USA agricultural industry $77.8 billion per year, and the cost to restore the Cape Floristic region to its pristine state without aliens is estimated as $2 billion (Pimentel et al. 2005; Turpie and Heydenrych 2000). Despite the fact that very few introduced species become invasive (Pimental et al. 2005; Zaveletta 2001) it is far better to prevent invasion rather than attempt to eradicate or control a species once it has invaded (Lockwood et al. 2001). Biofuels, and especially what are termed second generation biofuels, hold a high risk of becoming IAS, specifically because the very features that make for a good biofuel are the same features that are common in invasive species. These include rapid growth, aggressive colonisation of space, ease of establishment, wide habitat tolerance and resistance to pests and diseases. In addition to potential biofuels feedstocks themselves being invasive, the uncontrolled movement of biofuel products can act as a vector for the transportation of other potential pests and pathogens that might be invasive (IUCN 2009). Section 5.8 of this Chapter will focus on methods for limiting invasion risk.

5.3.5 Additional biodiversity risks

Though of lesser importance than land use change and invasion, there are a number of other mechanisms through which biofuels can impact on biodiversity during both the growing of feedstock and the processing of biofuels. These include:

- Pollution in waterways from fertilisers and pesticides applied to the biofuel fields, sediments washing off of them, and salts draining out of irrigated biofuels. This can impact on downstream waterways and wetlands causing eutrophication and toxin accumulation.
- Reduction in streamflow resulting from growing perennial, deep-rooted biofuel species in formerly seasonal grasslands. This has the further effect of exacerbating the water quality impacts noted above, through loss of dilution potential.
- Impacts of pesticides and herbicides on target and non-target species, as well as impacting on predators of these species.
- Pollutions from processing plants that are discharged into river systems. This includes adding organic matter to rivers which results in high Biological Oxygen Demand (BOD).
- Changes in hydrology leading to drying out of wetland systems.
- Impacts on soil micro-organisms through cultivation and the removal of food sources.
- Habitat fragmentation which impacts on species movement and dispersal.

These additional threats to biodiversity will not be specifically considered in this Chapter, but their potential impact on biodiversity should not be ignored.

5.3.6 Can biofuel plantations provide the same ecosystem services as natural forests?

Biofuel plantations have been proposed as a mechanism for reclaiming degraded land, reforesting deforested areas and as crops for marginal areas. In effect the suggestion is made that biofuel plantations may be a mechanism for increasing the flow of environmental goods and services (or even increasing biodiversity) in these damaged areas (Ghosh et al. 2008). Can biofuel plantation be beneficial to biodiversity? In general the answer would be no, but under specific conditions it is feasible that biofuel plantations may be a more favourable land use option than the prevailing land use from a biodiversity and/or ecosystem service perspective. This would be very situation specific: it depends on the current land use and condition, which biofuel is to be grown and how it will be managed. Research data to back up this claim is relatively limited. In some circumstances biofuel crops may be less environmentally detrimental than other agricultural crops (for instance, where the pesticide or fertiliser inputs are lower or the rotations are longer). But where biofuel crops replace other crops, iLUC impacts are likely. Furthermore, it is wrong to assume that marginal lands are low in biodiversity; low-productivity lands are often extremely biodiverse, especially where the high-productivity lands have already been transformed.

A number of studies have compared aspects of biodiversity in oil palm plantations to natural forest and degraded forest. For most taxa and functional groups the oil palm plantation had far lower diversity of indigenous forest species than the adjacent indigenous forest, though may have higher values that heavily degraded forests such as when transformed to *imperat*a grasslands (Danielsen et al. 2009; Koh and Wilcove 2008; Fitzherbert et al. 2008). Some examples include:

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- Oil palm plantations have a 77% reduction in forest bird species and 83% reduction in forest butterfly species compared to adjacent mature forest. In the oil palm plantations these taxa are fewer than in comparable logged forests, or rubber plantations (Koh and Wilcove 2008).
- A mean across a number of studies found only 31% of forest invertebrates and 23% of forest vertebrates in oil palm plantations. Total species numbers in oil palm plantations were 89% and 38% of forest species density for invertebrates and vertebrates respectively (Danielson et al. 2009).
- Palm plantations were found to have few or no forest trees, lianas, epiphytic orchards or indigenous palms, but had a higher diversity of pteridophytes (ferns) than mature forest (Danielsen et al. 2009).
- The diversity that does exist is mostly of non-forest species (Danielsen 2009).
- Biodiversity in oil palm plantations tends to be dominated by a few generalist species and often by exotic invasive species (Danielsen et al. 2009; Koh and Wilcove 2008).
- Palm forests, have more forest species in some taxa than *Imperata cylindrica* (itself an IAS in this circumstance) grasslands (Fitzherbert et al. 2008).

Sugar cane is known to support relatively limited biodiversity (Oliver 2005). There are a few taxa that do well in cane plantations - including rats, snakes, spiders and ants. Due to the vigorous growth of cane, very few other plant species are found in cane plantations. The plant species that do occur are mostly weeds, and quite commonly invasive aliens. Bird diversity in cane plantations is also low (Petit et al. 1999; Martin and Catterall 2001).

Almost no data are available on the biodiversity impacts of Jatropha plantations. One of the selling points of Jatropha for famers is the toxicity of the fruit and leaves to mammals. It is probable that the management practices applied to the Jatropha plantation will have a big impact on the biodiversity. Some Jatropha projects such as GEM Biofuels in Madagascar plant Jatropha directly into degraded savannas or grasslands with relatively limited immediate impacts on current biodiversity. Other projects, such as ESV Bio-Africa Limitada in Mozambique, plough the site before planting, effectively destroying most existing biodiversity. Everson et al. (In Press) have found that fully clearing the herbaceous layer during early years of the plantations greatly increases oil yield, so clearing may well become a common practice. If an indigenous herbaceous layer is maintained in Jatropha plantations and the trees are widely spaced, then the habitat will maintain some characteristics of the original, and will clearly support greater biodiversity than if all indigenous vegetation is cleared and Jatropha is grown as a monoculture.

Biofuel plantations fall short of indigenous forests, savannas, shrublands, wetlands or grasslands in terms of their ability to provide habitat for biodiversity. Oil palm plantations and Jatropha plantations probably maintain more natural biodiversity than sugar cane plantations or annual crops such as soybean. They might in extreme cases also maintain more biodiversity than badly degraded landscapes or alternative annual crops. Other tree crops such as coffee, cocoa, rubber or *Acacia mangium* tend to maintain greater forest diversity than oil palm (Fitzherbert et al. 2008).

Biofuel plantations, though not equivalent to indigenous vegetation, might maintain certain environmental services to a degree, especially in contrast to other alternatives. Services which are partly supported include; soil formation and stabilization, carbon sequestration, stream flow regulation and flood mediation. The degree to which these services are maintained will be dependent on actual management practices employed and will vary between feedstock crops. For instance maintaining an understory of indigenous grass within a Jatropha plantation will support more ecosystem services than if the understory is kept totally clear of vegetation. Mechanical harvesting of sugar, where the toppings are returned to the soil as a mulch maintains greater soil carbon, and reduces erosion when compared to manual harvesting after burning (Noble et al. 2003). In general a perennial crop is likely to maintain more ecosystem services, and more reliably, than annual crops.

A common feature in biofuel literature is the notion that biofuel feedstock, especially Jatropha, can be grown on degraded, waste, unproductive or marginal land. Even if grown in degraded areas there is limited data to substantiate if this will have positive biodiversity impacts or not; and the reality is that Jatropha will more likely be grown in good areas because this improves the economics of production. While Jatropha can survive in degraded lands, it is at the expense of higher yields that would be obtained with optimum soils, water and nutrient inputs. Unfortunately, definitions of what constitutes these different land categories are seldom given, and the terms are often used interchangeably. From a biodiversity perspective there is a concern that what might be considered as 'marginal' or 'waste land' from an agronomic or livestock grazing perspective, might be a highly biodiverse area from a biodiversity perspective. For example, the Succulent Karroo biome of South Africa is an area of very low rainfall (almost exclusively falling in winter) located along the South African western coast. This area has very limited agricultural value, but has a long evolutionary history and exceptionally high species diversity of endemic flowering plants (Cowling and Hilton-Taylor 1994; 1997), contributing about half of the world's succulent flora.

5.4 How to Identify Areas of Potential Biodiversity Concern for Biofuel Development?

Land transformation to plant biofuels is the single biggest biodiversity concern. This concern is especially relevant to situations where near-intact indigenous habitats are transformed. The degree of biodiversity impacts from the growing of biofuels will differ between different habitats and land use change scenarios. Some habitats are more of a concern in terms of potential habitat transformation and the resulting biodiversity loss than others. In addition, the current status of the land in terms of degradation and transformation is an important determinant of potential biodiversity loss.

Two aspects underpin the severity of biodiversity impacts. One is the importance of the habitat for biodiversity protection, and the other is the degree to which the proposed land is degraded or already transformed. A simple matrix (Figure 5.4) illustrates that it is untransformed areas of high biodiversity importance which are likely to have the greatest biodiversity conservation value. However, determining what constitutes 'important' from a biodiversity protection perspective is non-trivial and may well change over time. The following are some of the features that will indicate that a specific habitat is likely to have a high biodiversity conservation value:

- The area has been identified as a region of global biodiversity importance (Myer et al. 2000)
- The species richness of the habitat. Habitats with high species richness are likely to have high biodiversity conservation importance

Quality of land (whether it has been degraded or transformed)	Biodiversity importance: Low	Biodiversity importance: High
Good	Good condition natural habitat of low conservation importance Low overall conservation value – but large scale conversion could alter conservation state	Good condition natural habitat of high conservation importance Very high biodiversity conservation value
Bad	Totally transformed or badly degraded land of an original habitat type of low conservation importance Very low biodiversity conservation value	Degraded or transformed land in high conservation value habitat Conservation value dependent on degree of degradation and possibilities of reclamation

Figure 5.4: **Two way matrix illustrating the interplay between land degradation and biodiversity importance when determining conservation importance**

- The degree to which the habitat supports endemic and unique species. Endemics, because of their restricted range, are more likely to be driven to extinction, especially if they are relatively uncommon species
- The number of species present in the target areas that are considered as having a high conservation status due to their rarity or likelihood of being driven to global extinction (IUCN red data species e.g. panda bear or tiger)
- The degree to which the habitat is protected elsewhere. If the habitat is overall well protected, developments in the unprotected areas are less likely to be a biodiversity threat.
- Rate of habitat loss. In habitats where the rate of loss is high, additional drivers of habitat destruction should be discouraged
- Extent of habitat transformation compared to its historic extent. In habitats that are already highly transformed, any additional transformation should be discouraged
- The total spatial extent of the habitat. Transformation in small unique habitats will have disproportionate impacts on biodiversity
- The degree to which the area is a large, natural, undisturbed, unfragmented, fully functioning ecosystem; i.e., does it still maintain a 'wilderness' nature[16]

[16] The words 'pristine', 'virgin' or 'wilderness' are sometimes inaccurately used to describe such situations. Nowhere in the world is pristine anymore, and a certain low level of disturbance is in fact beneficial for biodiversity. The key issue is whether the ecological processes that allow the habitat to persist and regenerate without external inputs are still in place, and the full suite of functional types are present in more-or-less their natural proportions.

- The extent to which the habitat provides important ecosystem services, especially where they may impact on other habitats (e.g. a wetland provides clean and regulated water to downstream river and estuarine habitats).

Where biofuels are grown on already-transformed agricultural lands, abandoned mine dumps, highly degraded landscapes etc., the biodiversity impacts many be minimal or even positive. However, if agriculture is being displaced by biofuels then there is a very real possibility of indirect land use change (iLUC) and its resultant biodiversity impacts (see Figure 5.4). Growing biofuels on abandoned agricultural lands and other degraded land will have relatively low impacts on biodiversity or possibly even positive impacts.

It is important to remember that not growing biofuel feedstock also has consequences to biodiversity since current and alternative future trends in land use may also have biodiversity consequences which could be either negative or positive. Further, if a biodiversity project is not implemented in a region, this does not automatically imply that the area's biodiversity will be preserved. The opposite could also be true. Current degraded land, if not converted to biofuel, might undergo successional changes back to secondary and eventually mature forest; but it may also continue to degrade, or be used for other agricultural or exploitation purposes that are even more damaging to biodiversity than a biofuel plantation.

5.5 Strategic Assessment of Likely Biodiversity Impacts (the BII Approach)

From a strategic national or regional (provincial[17]) perspective a biodiversity assessment tool is required by policy decision makers to investigate likely impacts from large scale biofuel expansion. The tool needs to be able to investigate likely consequences of different scenarios such as the type of biofuel crop envisaged and where the plantings will take place in terms of habitat types and current land use options. In this regard 'mean species abundance' approaches, such as the Biodiversity Intactness Index (BII) are regarded as an appropriate tool. BII has been widely tested and is well documented (e.g. Scholes and Biggs 2005; Biggs et al. 2006). A simple user manual has been produced by Nickless and Scholes (2009) and is available for download[18]. Extracts from the manual are given below as well as an example of its use to investigate possible impacts from large scale biofuel expansion in the Eastern Cape, South Africa.

In determining consequences of biofuel expansion there are likely to be complex tradeoffs between many different aspects of biofuel impacts. Clearly many of these impacts, such as the biodiversity impact, cannot be accurately and rigorously expressed in monetary value terms. To compare these different types of impacts, expressed in non-commensurate terms, some form of multi-criteria decision analysis is the favoured analytical tool. In this regard the BII is an appropriate method for determining relative biodiversity impacts of a number of competing land use options as an input into multi-criteria decision making.

[17] The term regional is used throughout to refer to regions within a country such as provinces or states. The BII could also be used for assessments of regions consisting of multiple countries.
[18] http://www.ceg.ncl.ac.uk/reimpact/

The Biodiversity Intactness Index (BII) is a measure of the abundance of individuals, averaged across a wide range of well-known elements of biodiversity, relative to their abundance in a defined reference case (Scholes and Biggs 2005; Biggs et al. 2006). It is an indicator of the average abundance of a specified set of organisms (or functional groups of organisms) in a given geographical area (Scholes and Biggs 2005).

The BII was created as part of the Southern African Millennium Ecosystem Assessment to provide an easy-to-understand overview of the state of biodiversity for policy-makers and the public (Biggs et al. 2006). Specifically, the BII was designed to fulfil the requirements set out by the Convention on Biological Diversity (CBD) which stipulated that an indicator for biodiversity change should be scientifically sound, be sensitive to changes at policy-relevant spatial and temporal scales, allow for comparison with a baseline situation and policy target, be useable in models for future projections, and be amenable to aggregation and disaggregation at ecosystem, national and international levels (CBD 2003a; Scholes and Biggs 2005; Biggs et al. 2006). In addition it requires that the index be easy to understand and use, broadly accepted and measurable with sufficient accuracy at affordable cost (Biggs et al. 2006). The BII is intended to provide a single, integrated measure of biodiversity, for instance in assessing progress towards the CBD goal to "achieve by 2010 a significant reduction of the current rate of biodiversity loss at the global, regional and national level" (CBD 2003b; Mace 2005). The ability to use the index to explore impacts from future land use scenarios makes it of particular importance for assessing potential biodiversity impacts from large biofuels programmes. In this regard, the biodiversity score comparing a landscape with and without biofuel plantations can be easily computed.

The BII is an aggregate index. It is weighted by the area subject to different land use activities, which can range from complete protection to extreme transformation (e.g. in the case of urbanization), and the number of species occurring in the particular area (Scholes and Biggs 2005). Because of the area and species weighting, the BII is essentially scale-independent, and taxonomically unbiased. It can thus be aggregated and disaggregated in many ways. It can be expressed by ecosystem or political unit, or by taxonomic group, functional type, or land use activity. This capability provides the BII with transparency and credibility (Biggs et al. 2006, Scholes and Biggs 2005). The BII can be used to describe the past (Biggs and Scholes 2007) or project into the future (Biggs et al. 2008). The BII also has an associated error bar, allowing the user to monitor the degree of uncertainty (Biggs et al. 2004; Scholes and Biggs 2005, Hui et al. 2008). Critiques of BII, and suggested improvements, can be found in Rouget et al. (2006) and Faith et al. (2008).

BII is not the appropriate tool for examining impacts on rare and endangered species, since the changes in these species will be obscured by the variations in the more abundant species. For this purpose, it is suggested that approaches such as the 'Red List Index' are used, which focus on threatened species (IUCN 2003). As a general principle, it is unlikely that a single indicator will suffice for all purposes. But to avoid a proliferation of indicators, we suggest that a minimalist approach is to have one that reflects changes in the mean state of biodiversity (such as the BII) and one that looks at the fringe of the abundance distribution (i.e. rare and threatened species).

5.5.1 The BII Algorithm

The principles underlying the BII are discussed in Scholes and Biggs (2005) and Biggs (2005). The BII gives the average richness- and area-weighted impact of a set of activities that can be associated with a defined spatial domain on the population of a given group of organisms in an assessment area, which can contain many such activities. The BII is the estimated average population size of a wide range of organisms relative to their baseline populations for a given area (Biggs 2005; Scholes and Biggs 2005). A value >1 would indicate an increase in biodiversity and a value <1 a decrease of biodiversity with reference to the chosen baseline populations. The BII is calculated by:

$$\mathrm{BII} = \frac{\sum_{i}\sum_{j}\sum_{k} R_{ij} A_{jk} I_{ijk}}{\sum_{i}\sum_{j}\sum_{k} R_{ij} A_{jk}}$$

where

R_{ij} = Richness of taxon *i* in ecosystem *j*

A_{jk} = Area of land use *k* in ecosystem *j*

I_{ijk} = (Size of population of taxon *i* under use *k* in ecosystem *j*) / Size of population the reference time

(Biggs 2005; Scholes and Biggs 2005).

'Taxon' means a group of organisms that are expected to react in a similar way to the activities associated with various land uses. Typically, the definition of a taxon for the purposes of BII begins with a traditional high-level taxonomic approach (i.e., mammals, birds, amphibia, reptiles and plants are treated separately), but below this follows a 'functional type' approach rather than a strictly phylogenetic approach (e.g. 'trees' rather than a particular family or genus).

Three basic input factors are needed to calculate the BII: Richness (R_{ij}), area (A_{jk}) and relative population size (I_{ijk}), defined in terms of specific taxa (*i*), ecosystems (*j*) and land uses (*k*) (Biggs 2005). Biggs (2005) discusses the considerations that need to go into the definition of *i*, *j*, and *k* and the determination of R_{ij}, A_{jk} and I_{ijk}. This is summarised below.

5.5.2 Taxa (*i*)

The BII should be calculated across all indigenous species within the broad taxonomic groups that are reasonably well described. This usually includes plants and vertebrate species, such as mammals, birds, reptiles and amphibians. The invertebrates and microbes are typically excluded at regional, national and global scale because, although diverse, they are generally poorly documented at these scales (it is estimated that less than 10% of the probable number of species in these groups have been scientifically described). However, if at a project scale a reasonably stable and complete species list exists for any group it can, and should, be included in the calculation of the BII. The idea is to reduce bias in the estimate by making it as broad-based as possible. Alien species should in general not be included if they were not present at the time of the baseline establishment. An increase in the abundance of aliens is not generally regarded as a 'good thing' for biodiversity. Invasion by a diverse array of aliens could increase the apparent BII, sending the wrong interpretive signal. Where their population is zero in the baseline case, their inclusion can lead to mathematical problems (division by zero). However, in certain circumstances alien species can be regarded as "naturalized". These species can then be included if the baseline population level is not zero and can be viewed as the equilibrium population level. As an example, for long-established agricultural landscapes, organisms that originated elsewhere may be used.

5.5.3 Land uses (*k*)

Land uses are defined by the major human activities impacting on biodiversity in a particular region. These land uses are expressed in terms of their 'footprint': the area that they affect. The number of classes of land use should generally be limited to ensure that the number of I_{ijk} estimates is manageable: less than ten might be a good guideline. Scholes and Biggs (2005) used six categories of land uses in their application of the BII to the southern African region: protected, moderate use, degraded, cultivated, plantations and urban. The land use classes need to be defined clearly in order that the estimates of impacts on populations can be unambiguously assessed. An example of the land use definitions used in the southern African example (Scholes and Biggs 2005) appears in Table 5.1. A land use map should be created using available information, such as satellite or aerial photo images or ground-derived land-cover maps and land tenure boundaries. Where different data sources lead to an overlap of different land use activities, the highest impact land use class should be assigned.

The resolution of the land use map will affect the estimation of the I_{ijk} and, if the information obtained for land uses is too coarse, it can result in a significant decrease in the accuracy of the BII (Rouget et al. 2006). The impact of habitat fragmentation, as opposed to habitat area loss, can be incorporated in the definition of the land use categories if it occurs at a resolution much smaller than the land parcels under consideration. Or, if it occurs at scales larger than the resolution of the land use classes, it could be incorporated using a species-area curve approach suggested by Faith et al. (2008). If certain impacts are not associated entirely with one class, then multiple classes need to be defined. For example, in the southern African case, the protected areas land use class can be divided into large and small protected areas, thereby accounting for fragmentation. A separate estimate of I_{ijk} would then be obtained for each of these classes. Rouget et al. (2006) recommend carrying out detailed land use surveys in order to ensure that the BII scores calculated are reliable.

Activities that have an impact which can be given an aerial footprint, but are additional to the direct and local effects of the land use (for instance climate change or air pollution) are not dealt with by the standard definition of BII (Biggs and Scholes 2005). Where they are important, they should be applied as a multiplier to the I_{ijk} score due to land use, wherever they apply. Say, for instance, that there is robust information that climate change has caused a 30% reduction in population abundance, then the land-use I_{ijk} for all the affected species, under all land uses within the affected area, would be multiplied by 1- (30/100) = 0.7.

5.5.4 Ecosystems (*j*)

Broad-scale associations of organisms with particular abiotic environments can be defined as ecosystems, and can typically be arranged into several hierarchical levels. For BII purposes, these ecosystems need

Table 5.1: **Example of land use definitions from Scholes and Biggs (2005)**

Land use class	Description	Examples	Data source
Protected	Minimal recent human impact of structure, composition or function of the ecosystem. Biotic populations inferred to be near their potential.	Large protected areas, national, provincial and private nature reserves, 'wilderness' areas.	World Database on Protected Areas. All designated protected areas of IUCN categories I-V.
Moderate use	Extractive use of populations and associated disturbance, but not enough to cause continuing or irreversible declines in populations. Processes, communities and populations largely intact.	Forest areas used by indigenous people or under sustainable, low-impact forestry; grasslands grazed within their sustainable carrying capacity.	All remaining areas not classified into one of the other five categories.
Degraded	Extractive use at a rate exceeding replenishment and widespread disturbance. Often associated with high human population densities and poverty in rural areas. Productive capacity reduced to approximately 60% of 'natural' state.	Clear-cut logging, areas subject to intense harvesting, hunting, fishing or overgrazing, areas invaded by alien vegetation.	All areas falling below 75% (forest, grassland and savanna) or 50% (shrublands) of expected production as estimated by nonlinear regression (Michaelis-Menten function) of maximum annual NDVI on growth days. Degraded areas not estimated for desert, wetland and fynbos.
Cultivated	Natural land cover replaced by planted crops. Most processes persist, but are significantly disrupted by ploughing and harvesting activities.	Commercial and subsistence crop agriculture, both irrigated and dry land, including planted pastures and fallow, or recently abandoned cultivated areas. Orchards and vineyards.	SADC Landcover Data set, filled with GCL2000 fpr Namibia and Botswana.
Plantation	Natural land cover permanently replaced by dense plantations of tress. Unplanted areas assumed to constitute approximately 20% of class.	Plantation forestry, typically Pinus and Eucalyptus species.	SADC Landcover Data set.
Urban	Land cover replaced by hard surfaces such as roads and buildings. Dense populations of people. Most ecological processes are highly modified. Remnant semi-natural cover assumed to constitute 10% of class.	Dense human settlements, industrial areas, transport infrastructure, mines and quarries.	Urban extents.

to be defined at a spatial level so that the appropriate richness weighting (R_{ij}) can be allocated to each area for the calculation of each I_{ijk}. All three of these classifications (R_{ij}, A_{jk} and I_{ijk}) can be expressed at different hierarchical levels. In the example of calculating BII for all of southern Africa, the WWF Ecoregions (Olson et al. 2001) were used as the basic information on the spatial extent and species richness of ecosystems. These ecoregions (about 30) were aggregated into six high-level biomes for the subcontinental study (forest, savanna, grassland, shrubland, fynbos and wetland), since this led to a manageable number of cases to consider. It was found that if too many ecosystem types were defined, the experts ended up giving them exactly the same impact scores anyway. In this case, R_{ij} and A_{jk} were determined in terms of the disaggregated WWF ecoregion data, while I_{ijk} estimates were determined at the biome level, and then associated with each of the ecoregions.

5.5.5 Richness (R_{ij})

In the calculation of the BII, R_{ij} refers to species richness across the landscape at the reference point in time (normally, prior to the onset of the land transformation process that is under consideration). In general, species richness is available as total species counts for each broad taxonomic group, per ecosystem type, for the 'well-known' biodiversity. Species-by-species distribution data is usually not available for more than a few species for large areas, but it may be available for smaller areas, or places with relatively low biodiversity. Where the potential geographical distributions for individual species are available, these can be used in conjunction with ecosystem-level distributions for other species. There are two approaches for calculating the BII: the raster (pixel) method and the vector (polygon) method. If individual species distribution data is used, the BII should be calculated on a raster basis.

5.5.6 Area (A_{jk})

The area of a particular land use within a specific ecosystem type (A_{ijk}) is determined by overlaying ecosystem and land use maps in a Geographic Information System, after first ensuring they are in the same projection and at the same scale, and correctly geo-referenced.

5.5.7 Relative population size (I_{ijk})

The first step in estimating I_{ijk} (the population size of taxon group *i* under land use activity *k* in ecosystem *j* relative to a baseline population in the same ecosystem) is to define a meaningful and practical reference or baseline population. This reference point will influence the interpretation of the BII. In the southern African context, pre-modern (pre-1700) populations were used as the 'conceptual' baseline, but the practical reference was the current population density in large protected areas, which was assumed to broadly reflect the pre-modern abundances, which are unknown. However, some species have large home ranges, so their levels in large protected areas could conceivably be impacted compared to the conceptual baseline levels. In this case, the effect on the BII of these few species was considered sufficiently small as to be negligible. Parts of the world that were already highly transformed by the start of the modern era can use alternative reference points: for instance a time within record or reliable memory, or even the initiation year of a project. The same baseline should be used when comparing BII between different time points or between different regions.

Field data can be used to estimate I_{ijk} where available, but this will normally only be the case for a few species in a few locations. The value for I_{ijk} calculated from field data is simply some indicator of population density (i.e. some proxy of individuals or biomass per unit area) for an area affected by the land use, divided by the same indicator for the reference area. Since these data are relatively rare and spotty, it is perhaps best to keep them for validation purposes.

Population models can also be used to generate the I_{ijk} matrix where a great deal is known about the underlying drivers of population change. Alternatively, expert judgement can be used to generate this matrix, which can then be validated against field data, as was done in the southern African example.

Where estimates of I_{ijk} are collected using an expert interview process, this can be assisted by subdividing the taxonomic groups into functional types, in consultation with the experts. The same subdivision must be used by all the experts within a taxon. Species in the same functional type should respond in a similar way to the selected land uses. It has been found that body size (or height of the bud, in the case of plants), trophic niche and reproductive strategy are all good criteria for defining functional type. The number of functional types is a practical consideration. In the southern African example about 10 functional types per broad taxonomic group were defined. Classifying plants according to the Raunkier classification (Raunkier 1934) worked well, and the birds and mammals were classified according to size and feeding strategy (i.e. 'herbivore', 'carnivore', and 'omnivore'). Frogs were classified according to breeding strategy as they all were of similar size, and had a similar trophic niche.

Experts specialising in each broad taxonomic group are asked to estimate the impact of each land use activity within the context of each ecosystem type (e.g. biomes) on each functional type in their speciality taxonomic group. In the southern African example, land use (*k*) was considered to be the overriding factor impacting on population abundances, and in this case, the differences in the magnitude of the impact between ecosystem types was small relative to the differences between land use types within an ecosystem. For example, the impact of cultivation in savanna as opposed to cultivation in grassland areas is small compared to the difference in impact between cultivation and urbanisation within a grassland or a savanna. This justified the aggregation of ecosystem types, simplifying the estimation process. Due to the coarseness of the current knowledge regarding I_{ijk}, and the practical constraints of collecting or generating this data, it was in this case sufficient to define I_{ijk} at a relatively broad ecosystem level. Where finer level ecosystem data are available for R_{ij} and A_{jk}, it is recommended that these be used in calculating BII, even if broad-level I_{ijk} estimates are used.

If expert opinion is used in the generation of I_{ijk}, it is important to thoroughly discuss, define and illustrate the land use categories with the experts before beginning the judgement exercise. This ensures that their judgements of the impact of these land uses are accurate and appropriate for each land use category. It is advisable to give the experts examples of each land use category in the area under examination. It is also important that the resolution of these land use classifications is taken into account. For example, if one considers a regional study where the resolution of the land cover and ecosystem maps is 1 × 1 km, and 'cultivated' is a land use category, it needs to be understood that an area mapped as falling into this category is not going to be completely cultivated, but have small inclusions of uncultivated land as well – field edges, contour bunds, riparian strips and set-asides. This needs to be taken into account in the I_{ijk} estimates. In the southern African example, at the 1 x 1 km resolution, areas classified as cultivated were assumed to contain approximately 20% uncultivated land, areas classified as plantations were assumed to contain approximately 25% non-plantation land (in the form of e.g. riparian buffer strips and rocky outcrops), and areas classified as urban were assumed to contain approximately 10% non-built land (e.g. parks with natural vegetation within cities, or undeveloped land such as steep areas, locations within the floodplains of rivers, or rocky outcrops).

Estimates of I_{ijk} were aggregated up from the functional type level to the broad taxonomic level by weighting the estimates for each functional type (*f*) by the number of species in that group in the particular ecosystem type ($I_{ijk} = \sum_f \left(I_{fijk} \times \frac{R_{fij}}{R_{ij}} \right)$). Alternatively, if richness (R_{ij}) spatial data can be disaggregated to functional type level, BII can be calculated directly at the functional type level.

5

It is recommended that at least three experts are interviewed independently for each broad taxonomic group. Experts should not be shown or informed about the magnitude of the estimates provided by others or jointly interviewed to obtain a consensus estimate. Interviews with the experts typically last several hours and result in a few hundred estimates per expert. All estimates should then be entered into the database, and the variability (standard deviation) in expert estimates then enables the determination of the uncertainty around estimates. The range of estimates obtained in this way can then be used to derive a confidence interval for the I_{ijk} estimates (a topic discussed in more detail in Nickless and Scholes 2009) and for the BII as a whole (Hui et al. 2008). For operational applications, this is probably not a necessary step.

5.6 Using the BII to Investigate Provincial Level Impacts of Biofuel Expansion in the Eastern Cape South Africa

The Eastern Cape province (176 377 km^2) of South Africa has been identified as a potential area for canola production for biodiesel. Suggestions have been made that 5 000 km^2 of land be transformed from grassland, savannah or abandoned agricultural fields to canola in an initial project. In addition the Eastern Cape has the potential for forestry expansion with 1 200 km^2 of land being potentially available for this purpose. Though there is no current intent to use this expanded forestry land for bioenergy, it is one potential use of the product.

The Eastern Cape is one of the poorest provinces in South Africa, and has extensive areas under customary land tenure (the ex-homeland areas of the Ciskei and Transkei) as well as extensive areas under private freehold land tenure. The province has relatively high agricultural potential, but in the customary-tenure areas commercial agriculture has almost ceased, and current agriculture is largely small-scale, to supplement household food security. The region therefore has a large area of abandoned agricultural fields, many of which are now reverting to grasslands. Overgrazing means that much of the area is classified as degraded in the land cover maps of the region (NLC 2005) (see Figure 5.5).

The region has areas of high biodiversity value. It includes the Pondoland centre of endemism. The area under consideration for biofuels has three main ecoregions: Drakensburg Montain grassland, woodland and forest (grassland); Maputuland-Pondoland bushland and thicket (bushland) and Kwazulu-Natial coastal forest mosaic (coastal) (Olsen et al. 2001). Though biofuels may not be planted in all three ecoregions due to climatic or social constraints, all are included in the analysis to demonstrate impacts on habitats with limited distribution. The WWF ecoregion classification is used rather than the more recent and detailed classification of Mucina and Rutherford (2006) because it matches the Southern African coverage used by Biggs et al. (2005; 2006) which predated the new biome maps. This enables direct utilisation of the BII scores as derived by Biggs et al. (2006). The Wild Coast region, a strip running along the northern coastline, is an area identified as an area of national conservation priority by the National Biodiversity Conservation Strategy and Action Plan (Driver et al. 2005). Much of the Wild Coast Region coincides with the proposed canola growing region.

The vegetation classes, land use classes and land use biodiversity impact scores of Biggs et al. (2006) are used to investigate scenarios of biofuel expansion in the Eastern Cape province (see Tables 5.2 and 3). A number of alternate scenarios are considered relating to the way in which land is allocated to biofuel.

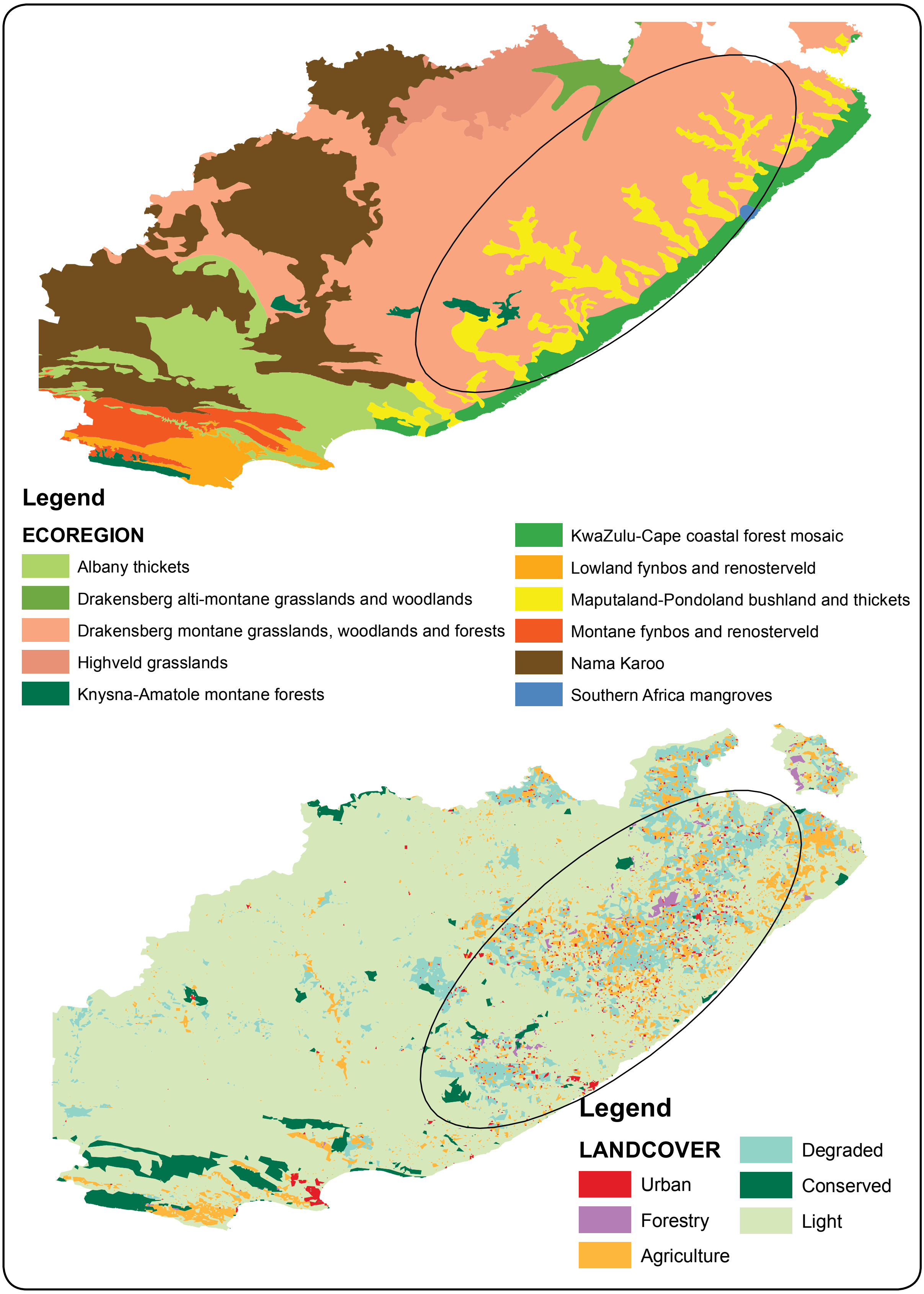

Figure 5.5: **Ecoregions and land cover of the Eastern Cape. The black oval approximates the area being targeted for biofuel expansion.**

Table 5.2: **Species richness of taxa in the Eastern Cape. Note, for the analysis taxa can be further sub-divided into functional groups (from Biggs et al. 2006)**

Ecoregion	Full ecoregion name	Plants	Mammals	Birds	Reptiles	Amphibians
AT0115	Knysna-Amatole montane forests	1000	52	272	36	24
AT0116	KwaZulu-Cape coastal forest mosaic	2000	80	373	88	40
AT1003	Drakensberg alti-montane grasslands and woodlands	800	71	288	29	18
AT1004	Drakensberg montane grasslands, woodlands and forests	3700	152	450	118	47
AT1009	Highveld grasslands	1900	115	397	68	29
AT1012	Maputaland-Pondoland bushland and thickets	2100	92	351	63	38
AT1201	Albany thickets	1200	68	280	55	14
AT1202	Lowland fynbos and renosterveld	3000	75	296	68	27
AT1203	Montane fynbos and renosterveld	6300	88	311	74	31
AT1314	Nama Karoo	1100	106	300	70	11
AT1322	Succulent Karoo	4850	75	225	94	15
AT1405	Southern Africa mangroves	200	26	224	6	2

Table 5.3: **Distribution of area between ecoregions of the Eastern Cape, and the distribution within each ecoregion to different land cover classes (from Biggs et al. 2006). See Table 5.2 for the ecoregion names. The three ecoregions where we simulated biofuel expansion are these with short names. All ecoregions were considered when determining overall BII scores**

Ecoregion	Short name	% Protected	% Light Use	% Degraded	% Cultivated	% Urban	% Plantation	% Total	TOTAL in km²
AT0115		27.4	52.3	5.5	12.7	1.6	0.4	0.8	1343
AT0116	Coastal	4.1	78.5	3.1	11.9	1.4	1.0	4.3	7302
AT1003		1.0	88.9	7.9	2.0	0.2	0.0	1.5	2542
AT1004	Grassland	1.3	71.5	14.5	9.7	1.7	1.3	46.6	79319
AT1009		0.0	79.1	13.5	6.6	0.8	0.0	4.2	7179
AT1012	Bushlands	1.3	67.7	16.4	11.6	2.5	0.5	5.7	9718
AT1201		5.1	86.0	2.1	5.2	1.6	0.1	8.1	13753
AT1202		2.7	65.0	1.0	28.8	2.0	0.5	2.0	3432
AT1203		47.1	50.2	0.7	1.8	0.0	0.1	3.2	5518
AT1314		1.5	94.0	3.1	1.2	0.1	0.0	23.5	40098
AT1322		12.5	87.5	0.0	0.0	0.0	0.0	0.0	16
AT1405		0.0	91.1	0.0	8.9	0.0	0.0	0.1	157
Total %		**3.43**	**77.68**	**9.52**	**7.46**	**1.23**	**0.68**	**100.0**	
Total		**5842**	**132349**	**16215**	**12717**	**2093**	**1161**		**170377**

- **Scenario 1:** Agricultural land is allocated first, until it is used up, then degraded land is allocated, and finally lightly utilised land (no conservation) land is allocated. This is done per ecoregion, or for all ecoregions at an equal rate proportional to the ecoregion total extent.
- **Scenario 2:** As per scenario 1, but starting with degraded land, then moving to lightly utilised land. No existing agriculture is allocated.
- **Scenario 3:** As per scenario 1, but allocating all biofuel to lightly used land.

The option of using a perennial tree crop instead of an annual agricultural crop as bioenergy feedstock was also investigated. No BII impact is available for Jatropha, but a factor has been derived for plantation forestry based on eucalyptus, pine and wattle (Biggs 2005, Biggs et al 2006). Though there are clearly differences between Jatropha and plantation forestry species, at this provincial scale this provides an initial estimate of possible Jatropha impacts. It is also useful as a scenario generation exercise if forestry is to be considered for bioenergy in the future.

5.6.1 Results

The results give total provincial BII impacts to the Eastern Cape, and given that biofuel was given the equivalent impact factor as crop agriculture, biofuel development has no biodiversity impact when allocated to agricultural land, provided there is no iLUC. Note that, though the overall forestry impact seems to not differ from the agricultural impact, as will be explained below, the impact is different for different taxa, though when combined for all taxa is remarkably similar for the specific ecoregions under consideration. In both the agriculture and forestry scenarios the impact increases as land allocation shifts to degraded, with the highest impacts when allocating to untransformed lightly used land (Figure 5.6). Overall BII impacts for forestry as a feedstock are almost identical to the impacts from crop agriculture on cultivated land.

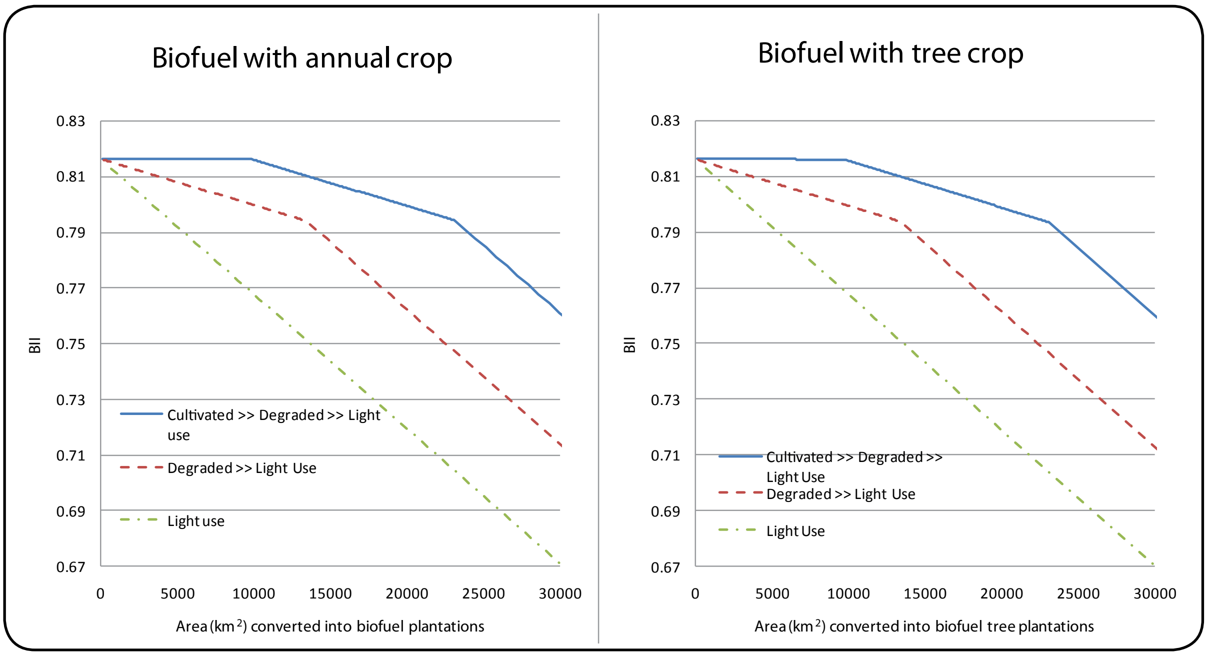

Figure 5.6: The influence of 'type of land allocated to biofuel' on 'the BII impacts', for both annual and tree biofuel crops. The land allocation rules allocate all cultivated land before allocating degraded land, and all degraded land before allocating lightly used land, in the scenarios where this is applicable.

The potential impact of biofuels on biodiversity is considerable, and for 20 000 km^2 (i.e. 12% of the total provincial area) the biodiversity loss ranges from 6 to 14% depending on the land allocation scenario. This is a substantive impact considering that the province has only lost an estimated 19% of its biodiversity over the preceding 300 years. In addition biofuel feedstock plantations are only being allocated to 3 ecoregions in the province and these three ecoregions in total are only 57 % of the total provincial area. This biodiversity loss is not allocated evenly by ecoregion or taxa as is illustrated in Tables 5.4 and 5.5. In this scenario where two million ha of lightly used land is allocated to crop agriculture, both amphibians and plants lose 11% each of their species overall. For plants each ecoregion looses 15% of their plants, whilst for amphibians the loss is far higher in the costal and grassland areas than in the bushlands. Overall, mammals in the grasslands have the lowest BII. This is largely due to a combination of an initially low BII as well as a relatively high estimated future loss. Birds show the highest resilience to loss of species compared to other taxa.

Table 5.4: **Impacts on BII per taxa of converting 20 000 km^2 annual cultivated biofuel (blue) or plantation forestry (red) in total from previously lightly used land to bioenergy for three of the ecoregions in the Eastern Cape. The proportion of area converted per ecoregion was the same for each of the three ecoregions. The number in black is the current BII score**

	AT0115	Coastal	AT1003	grass-land	AT1009	Bush-lands	AT1201	AT1202	AT1203	AT1314	AT1405	Grand Total
Plants	0.78	0.77	0.88	0.78	0.82	0.78	0.89	0.70	0.95	0.87	0.87	0.80
Cultivation 2 m ha		0.63		0.64		0.64						0.70
Forestry 2 m ha		0.61		0.63		0.63						0.70
Mammals	0.79	0.76	0.66	0.59	0.61	0.65	0.74	0.79	0.95	0.71	0.77	0.64
Cultivation 2 m ha		0.66		0.53		0.60						0.60
Forestry 2 m ha		0.66		0.49		0.59						0.58
Birds	0.92	0.93	0.96	0.90	0.92	0.88	0.97	0.92	0.97	1.07	0.91	0.94
Cultivation 2 m ha		0.79		0.77		0.76						0.86
Forestry 2 m ha		0.85		0.85		0.82						0.91
Reptiles	0.87	0.87	0.89	0.82	0.86	0.83	0.92	0.77	0.96	0.94	0.88	0.85
Cultivation 2 m ha		0.72		0.70		0.71						0.77
Forestry 2 m ha		0.74		0.71		0.72						0.77
Amphibians	0.85	0.86	0.99	0.88	0.93	0.91	0.97	0.79	0.95	0.98	0.92	0.89
Cultivation 2 m ha		0.67		0.70		0.75						0.75
forestry 2 m ha		0.71		0.73		0.86						0.79
Total	**0.81**	**0.79**	**0.89**	**0.79**	**0.83**	**0.79**	**0.90**	**0.72**	**0.95**	**0.90**	**0.89**	**0.82**
Cultivation 2m ha		0.65		0.65		0.66						0.72
Forestry 2 m ha		0.65		0.65		0.66						0.72

Table 5.5: **Percentage biodiversity loss from a scenario of 20 000 km² new biofuel plantations in untransformed land in the Eastern Cape, transformed to annual biofuel feedstock production. Current biodiversity minus future biodiversity expressed as a percentage of original biodiversity**

	Costal	Grass	Bush	Total
Plants	15	15	15	11
Mammals	10	10	06	06
Birds	08	05	06	03
Reptiles	12	11	11	08
Amphibians	15	15	05	11
Total	**14**	**14**	**13**	**10**

Table 5.6: **The difference in BII biodiversity loss if the conversion of land is to plantation forestry rather than an annual crop. A negative number indicates plantation forestry has a more negative impact. As an example total grassland bird impact in forestry is 13 versus 5 for agriculture giving - 8**

	Costal	Grass	Bush	Total
Plants	1.4	0.8	1.1	0.6
Mammals	-0.1	3.8	-1.1	2.2
Birds	-5.9	-8.0	-6.2	-5.0
Reptiles	-1.5	-0.8	-0.7	-0.6
Amphibians	-3.5	-3.6	-11.4	-3.4
Total	**0.1**	**-0.1**	**-0.1**	**-0.1**

Though forestry had the same overall BII as crop agriculture, the nature of this impact on different taxa is very different. Plants fare slightly better under forestry than cultivation, as do mammals in grasslands, but all other taxa in all other ecoregions fare substantially worse, with bird biodiversity in particular being disadvantaged by plantation forestry (Table 5.6).

Comparing total provincial biodiversity scores per ecoregion for scenarios where cultivation is restricted to a single ecoregion provides interesting results (Figure 5.7). Providing only existing cultivated and degraded land is used, using land from grasslands has the least impact on provincial level biodiversity. However if lightly used land in the grasslands is transformed then this has the highest impacts. This is despite the fact that grasslands are by far the largest ecoregion, and that there would still be extensive untransformed areas. There is only about 600 000 ha of coastal and 800 000 ha of bushlands in total, excluding a small percentage under conservation. If, as in these scenarios, almost the entire ecoregion were to be transformed to biofuel plantations then there is a very real probability that a number of species get driven to extinction since there are a number of species endemic to the ecoregions that are not found outside the Eastern Cape. As stated previously, the BII algorithm, because it is weighted by area, is relatively insensitive to these losses. The BII is not the appropriate tool to consider this potential loss of what might be rare and endangered species and the analysis would be strengthened by considering this aspect separately.

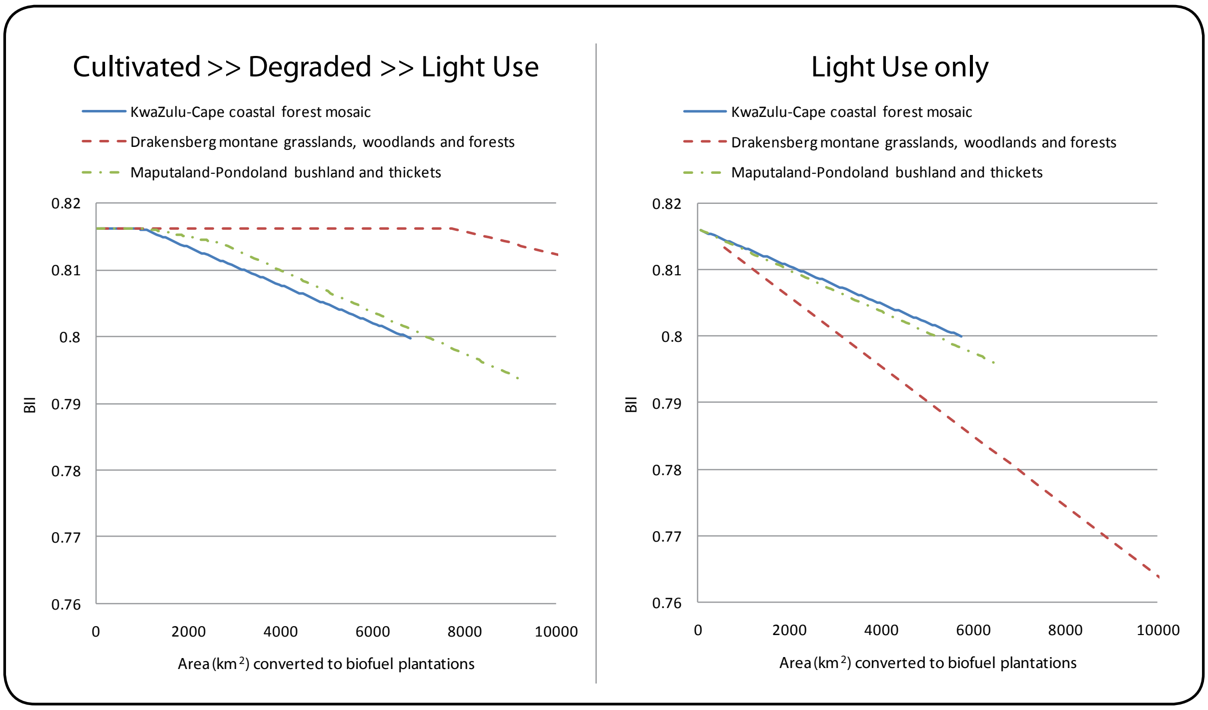

Figure 5.7: Impacts of area converted to agricultural biofuel crops on the Eastern Cape; total BII scores if land transformation is limited to a single ecoregion. Two scenarios are show, one where cultivated then degraded land is used first, and one where only lightly used land is transformed.

The BII analysis highlights a number of important issues. Transforming lightly used land to biofuels will have substantive impacts, especially if it is to take place on what would appear to be relatively abundant grasslands. This confirms the viewpoint that land transformation is a big threat to biodiversity. It is interesting to note that in this situation the use of forestry crops or annual crops would have very similar overall impact, but very different impacts on select taxa. Using currently cropped areas and degraded land has low impacts, however no consideration was made on potential iLUC impacts caused due to the displacing of agriculture.

This analysis demonstrated that, if even relatively coarse BII data is available, the BII approach can be used for rapid scenario generation. Only about 2 to 3 days of analysis time was required to conduct this exercise. However the analysis as presented is very crude and if finer level scenarios were required then the following could improve the results:

- Developing biofuel crop specific impact factors, but this should only be considered if these are likely to differ substantially from general agriculture or forestry impact factors;
- More detailed ecosystems could be considered (such as the bioregions of Mucina and Rutherford (2006)) for the Eastern Cape example. The use of detailed vegetation types could be used, though experience suggests that specialists are likely to give the same impact factor to all vegetation types in a bioregion, and though detailed species data may be available for some taxa, e.g. trees, it is unlikely to be available at this level of detail for all taxa;
- The process would be strengthened by linking it to a matrix of impacts on rare and endangered species.

5.7 Planning for Biodiversity at the Project Level

At the level of specific project implementation, biodiversity is an important consideration, but methods for planning for biodiversity conservation move away from the strategic conservation planning toward operational issues such as how best to configure plantations in the landscape. Having said this, it is very difficult to implement a rigorous operational biodiversity plan at the plantation level in the absence of a national or regional understanding of biodiversity priorities. In this section we propose a step by step approach to biodiversity assessment with a number of screening techniques and tools, with increasing levels of complexity, for ascertaining if biodiversity issues are likely to be a priority concern during project planning. We also provide links to tools that may assist to minimise biodiversity impacts during the implementation phase of biofuel plantations.

5.7.1 A first cut screening

The two-way matrix in Figure 5.8 is designed as a crude and simple scan to ascertain if biodiversity impacts are likely to be a critical issue. It is not designed to give an exact result, but rather to indicate if further investigation is required. This matrix could be used to assess the entire plantation, but it is probably best to apply it to each and every unique habitat within the proposed plantation footprint. No precise definition is suggested as to what would constitute 'an area of high biodiversity importance'; a list of some of the features likely to indicate high biodiversity interest was provided in section 5.4.

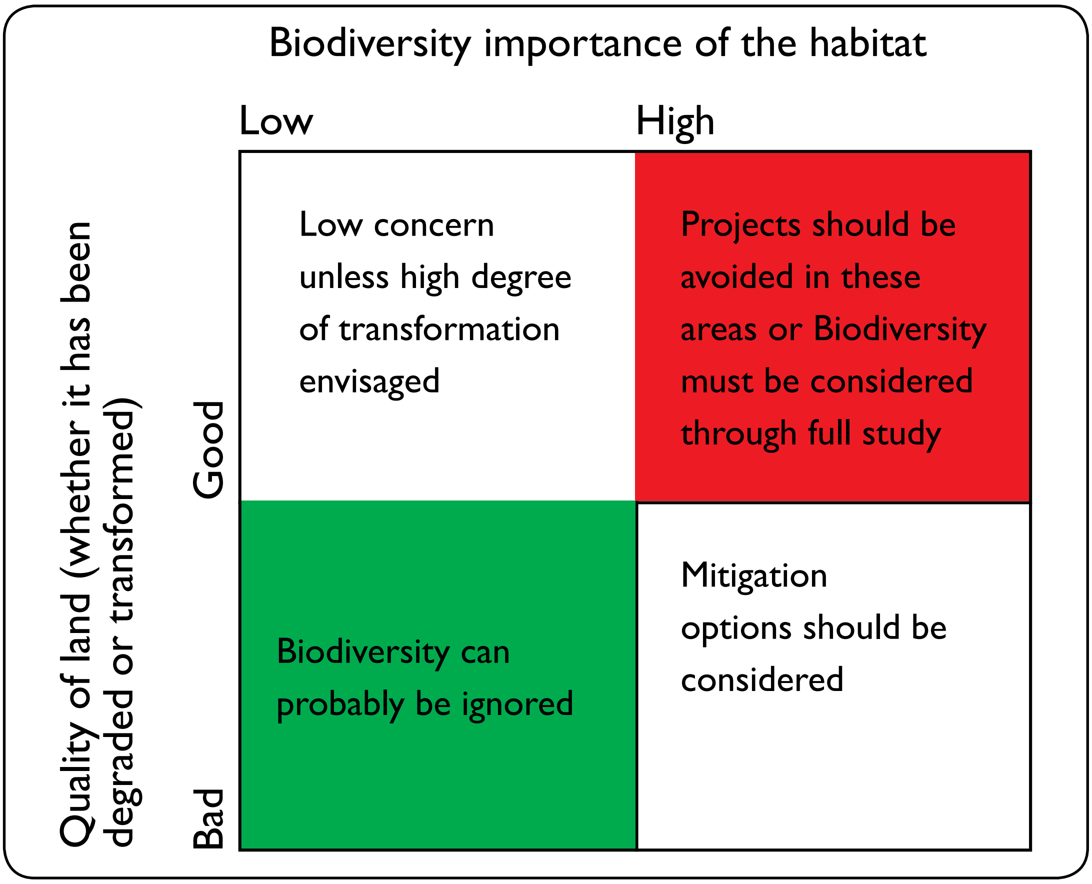

Figure 5.8: **A rapid screening to identify habitats where biodiversity is likely to be an important concern. Habitats that fall into the red area should be avoided. If they are to be considered then detailed impact assessments should be undertaken.**

Near-intact habitats[19], especially those with high biodiversity importance, should be avoided where possible. If such habitats cannot be avoided, full studies on biodiversity impacts should be undertaken. Where near-intact habitats have a relatively low biodiversity importance currently, it is important to consider if their biodiversity importance may change in the near future due to rapid land transformation as this could potentially change their conservation status. In areas with high conservation status, but that have already been badly degraded, it is possible that biofuel plantations may be able to use mitigation methods that can help overall conservation of the habitat type. One mechanism to achieve this is for the biofuel plantations to set aside some of the habitat and assist in rehabilitating it.

5.7.2 Decision tree approach

A decision tree approach provides a slightly more sophisticated method of screening than a two way matrix. It is based largely on the same logic, but provides more detail on the specific criteria that may be considered as important. The relatively generic model of Figure 5.9 could be improved to represent national conservation priorities.

5.7.3 A simplified BII approach

At the planning phase it is often useful to compare the proposed action (biofuel development) with alternative land use options (doing nothing or using the land for some other type of agricultural crop, forestry etc.). In these circumstances the BII as described in section 5.5 is an appropriate tool, and can be rapidly and cheaply used if appropriate impact factors have been developed for the habitats and land uses under consideration. If calibrations, at the appropriate scale, are not available for specific habitats and land uses then a simplified BII can be used where taxa are rated against a three point (better, no change, worse) subjective scale, or a five point scale (much better, better, no difference, worse, much worse). This would be best done through 'expert judgement': the consensus of a small number (3-5) of biologists with an understanding of the ecology of the area and the taxa involved. Species should be divided into broad taxon groups (e.g. plants, birds, amphibians, rodents, large mammals), and then subdivided into functional groups, if feasible (e.g. trees, shade tolerant grasses, grazing large mammals, large mammal predators, seed eating birds etc). Functional groups should represent organisms that are likely to respond in a similar way to the change in vegetation between the natural vegetation, current land use, bioenergy plantation and any other land use option being compared. Scoring can be done as per Table 5.7. Since most projects are not in complex landscapes, a separate scoring can be conducted for each habitat type that is likely to be impacted.

For example, a proposed biomass-gasification electricity plant in northwest Uganda would require large plantations of fast growing trees. Currently the land is large scale farmland that was previously used for cattle ranching. In the recent past herds of large game including buffalo and elephant used to wander through the farm. At present the area is being re-populated following 20 years of warfare. If this land is not used for biofuel plantations, it is likely that the land will slowly be broken up into small scale allotments for subsistence and commercial farming. Three scenarios are compared against the current

[19] The words 'pristine', 'virgin' or 'wilderness' are sometimes inaccurately used to describe such situations. Nowhere in the world is pristine anymore, and a certain low level of disturbance is in fact beneficial for biodiversity. The key issue is whether the ecological processes that allow the habitat to persist and regenerate without external inputs are still in place, and the full suite of functional types are present in more-or-less their natural proportions.

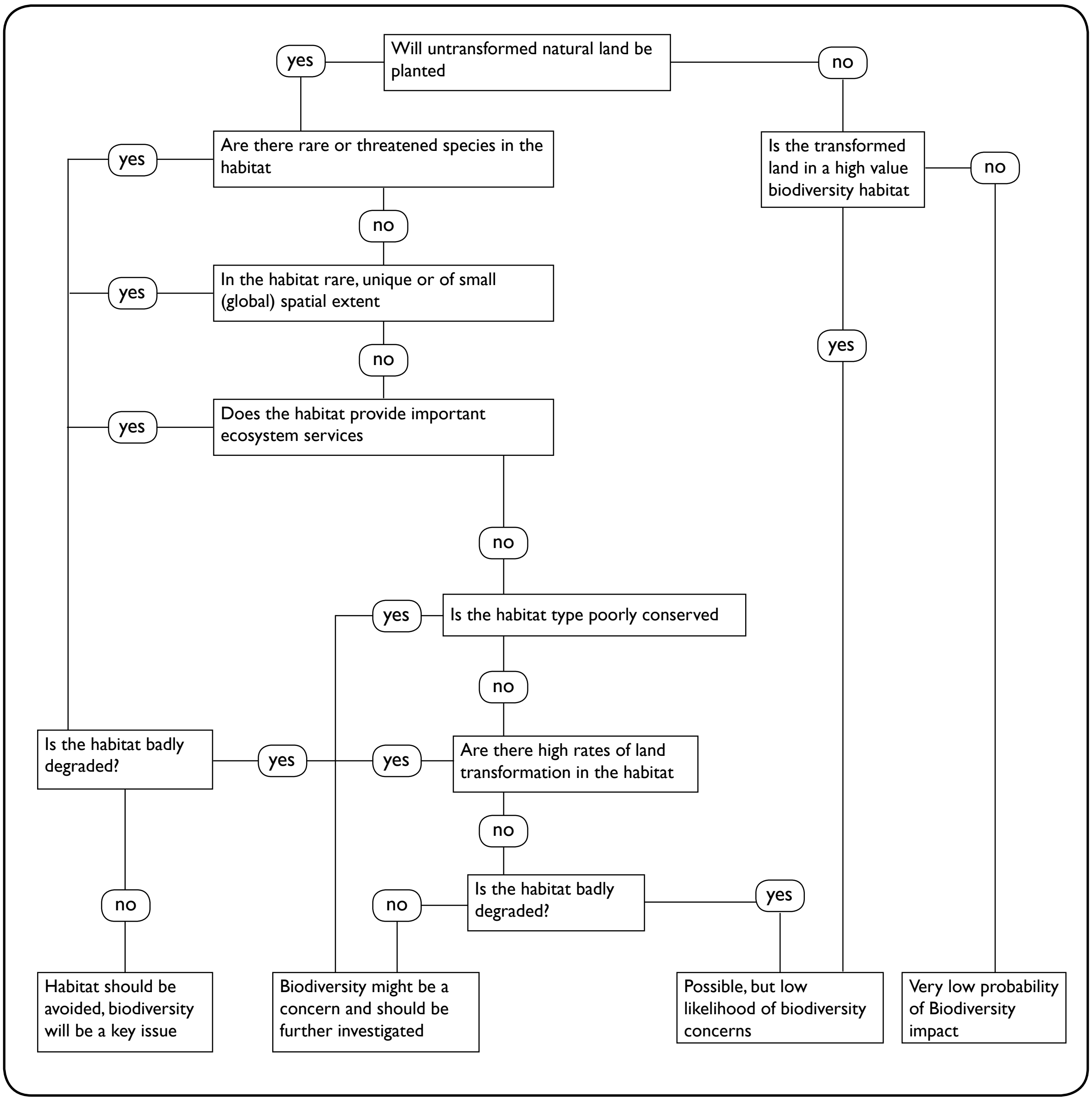

Figure 5.9: **A simple logical tree approach for screening habitats for likely biodiversity impacts.**

baseline: a plantation of fast growing trees; total conversion to small scale subsistence agriculture and gradual transformation to smaller commercial and subsistence allotments (the projected current situation in 10 years).

The results from Table 5.7 suggest that the current land use or a tree plantation would both have negative impacts on biodiversity, but that the impacts would be in very different functional groupings of plants and animals. Both these options have far less biodiversity impacts than the land being fully transformed to small scale cropland. Though a simple summing of impacts gives a trend, data are not normalized and individual functional taxa may carry disproportionately high weightings.

Table 5.7: **A rapid screening of biodiversity impacts per plant and animal functional types for a proposed gasification plant in north western Uganda. – represents a strong negative impact, - a week negative impact, 0 no impact, + a week positive impact and ++ a strong positive impact. (Data for demonstration purposes only and would require further verification from a panel of experts before being finalised).**

Plant or animal functional group	Subsistence agriculture	Current situation in 10 year trend	Fast growing timber plantation
Large mammals	–	-	-
Forest birds	–	-	++
Grain feeding birds	+	+	-
Small mammals (non pest)	-	0	+
Small mammals (pest)	+	0	-
Indigenous trees	–	-	–
Indigenous grass	–	0	-
Total	**-7**	**-2**	**-3**

5.7.4 The HCV approach

For projects wanting a packaged solution for planning around biodiversity, one option is the High Conservation Value Network Approach (HCV). This approach has the advantage that it is backed by methodology manuals, a network of practitioners and it is likely to be directly recognised and accredited by some of the biofuels sustainability standards such as the Round Table on Sustainable Biofuel Production (RSB), Round Table on Responsible Soya (RTRS) and Round Table on Sustainable Palm Oil Production (RSPO).The HCV approach considers conservation value beyond just biodiversity, and incorporates both environmental as well as related socio-economic and cultural issues. In essence, the HCV approach considers 6 aspects of conservation value as given in Table 5.8. Full details of the HCV approach can be obtained from their web site[20]. Unless this approach is backed by good definitions of what constitutes a 'High Conservation Value', it can easily lead to inappropriate planning (Koh et al. 2009). In this regard, the approach is strengthened if there are existing nationally defined biodiversity conservation plans and priorities in place. Preserving only small isolated fragments of areas of high conservation value should be avoided: larger blocks or corridors of indigenous vegetation are preferable. Collaboration with adjacent plantations should be sought to maximise an overall integrated conservation strategy.

5.8 Reducing Risks of Alien Invasion

Due to the vast number of plant species requiring permission to enter a country, the use of a screening tool can help distinguish which species are likely to be actual threats from those which are not. Most countries are signatories to the International Plant Protection Convention of 1952, and the UN Convention on Biological Diversity of 1993 and as such should already have appropriate quarantine and introduction protocols in place, though may be too understaffed and under-resourced to properly fulfil

[20] http://www.hcvnetwork.org

Table 5.8: **The HCV network uses 6 criteria in determining what constitutes high conservation value areas. Their definitions, though largely based on biodiversity concerns, also include some cultural issues.**

High Conservation Value category		Examples
HCV1	Areas containing globally, regionally or nationally significant concentrations of biodiversity values (e.g. endemism, endangered species, refugia).	The presence of several globally threatened bird species within a Kenyan montane forest.
HCV2	Globally, regionally or nationally significant large landscape-level areas where viable populations of most if not all naturally occurring species exist in natural patterns of distribution and abundance.	A large tract of Mesoamerican flooded grasslands and gallery forests with healthy populations of Hyacinth Macaw, Jaguar, Maned Wolf, and Giant Otter, as well as most smaller species.
HCV3	Areas that are in or contain rare, threatened or endangered ecosystems.	Patches of a regionally rare type of freshwater swamp in an Australian coastal district.
HCV4	Areas that provide basic ecosystem services in critical situations (e.g. watershed protection, erosion control).	Forest on steep slopes with avalanche risk above a town in the European Alps.
HCV5	Areas fundamental to meeting basic needs of local communities (e.g. subsistence, health).	Key hunting or foraging areas for communities living at subsistence level in a Cambodian lowland forest mosaic.
HCV6	Areas critical to local communities' traditional cultural identity (areas of cultural, ecological, economic or religious significance identified in cooperation with such local communities).	Sacred burial grounds within a forest management area in Canada.

their mandate. Biofuel practitioners, in their enthusiasm for the rapid establishment of biofuel projects, may well inadvertently or deliberately bypass the formal introduction protocols. This practice needs to be strongly discouraged. Although invasiveness is relatively rare, its consequences are severe, and can completely cancel out any economic or environmental benefits that were intended for the project.

5.8.1 Predicting invasiveness

It has long been a goal of invasion ecologists to identify a specific suit of traits that would identify an invasive plant species. Recently these traits or plant attributes have been summarised based on the correlation of traits with known invasive species, via experimentation and general theory (Pheloung et al. 1999; Pattison and Mack 2008). For example species that share common traits known to increase the risk of invasion include:

- Fast growth and ability to outcompete local vegetation
- Abundant seed production, especially of long-lived and resistant seeds
- Tolerance of a wide range of conditions

While important, the use of traits alone cannot adequately identify new invasive species. A successful invader is the result of a multiple-step process. Before a species can become invasive a series of barriers need to be overcome (Richardson et al. 2000). For example, a necessary condition for plant establishment is habitat compatibility (also known as habitat invisibility) (Rejmanek 2000). A habitat is known to be invasibile when a non indigenous species is able to establish, persist or expand (Burke and Grime 1996).

Current assessment schemes using a combination of approaches have proved most accurate to distinguish between known invasive species and non-invaders. These non-experimental predictions are most accurate when the biological attributes of the species and the climatic variables of both the source and recipient regions are included in the assessment process. In a recent survey, an accuracy level of 80% was considered acceptable (Gordon et al. 2008). Further examples for determining invasiveness include measuring stochastic events such as time since introduction, evaluating specific taxon (e.g. *Pinus*) or experiments (Rejmanek 2000).

5.8.2 Weed risk assessments

Risk assessments follow a non-experimental approach and are based on the biological information of the species, bioclimatic features of the source and recipient regions and the evolutionary history of both (Richardson et al. 1990). The risk assessment phase is intended to be applied at various stages of plant movement. For example, it can be applied to species already present in a country, to determine the risks of increasing the range (Rouget et al. 2002) and whether a species could be moved from one region to another with minimal risk of invasion (Barney and DiTomaso 2008). However in most instances risk assessments are adopted during border control to determine the risks of new species entering a country (Pheloung et al. 1999).

A risk assessment should ideally form part of a decision support system to help determine whether the risk of introducing a non-indigenous species is acceptable (Cousens 2008). An example of its use is described in Figure 5.10 where the risk assessment is part of a three-tiered decision support system adopted by most countries (Pheloung 2003). One of the main aims of the risk assessment process is to find an objective and standardized process for evaluating the risk of introducing an organism to a new environment. This is important as decisions based on expert opinion may be swayed by potential economic opportunities resulting in a subjective bias towards acceptance (Pheloung et al. 1999).

Several methods exist to determine the risk of invasion from plant species (Tucker and Richardson 1995; Reichard and Hamilton 1997; Pheloung et al. 1999). These screening systems either use a decision tree format or a questionnaire coupled with a scoring system to evaluate various sources of information including biogeography, life history, plant traits and specific regional characteristics to draw conclusions about whether to accept, reject or further evaluate a species in question.

One of the challenges regarding the available methods is that they were developed for specific geographical regions, narrow taxa (e.g. pines) or broad taxonomic groups (e.g. woody plants) to facilitate decision making regarding the threat of new species or existing plant species. Therefore, adopting any of these methods as a general screening tool requires slight modifications to the original format before being applied elsewhere (Daehler and Corrino 2000).

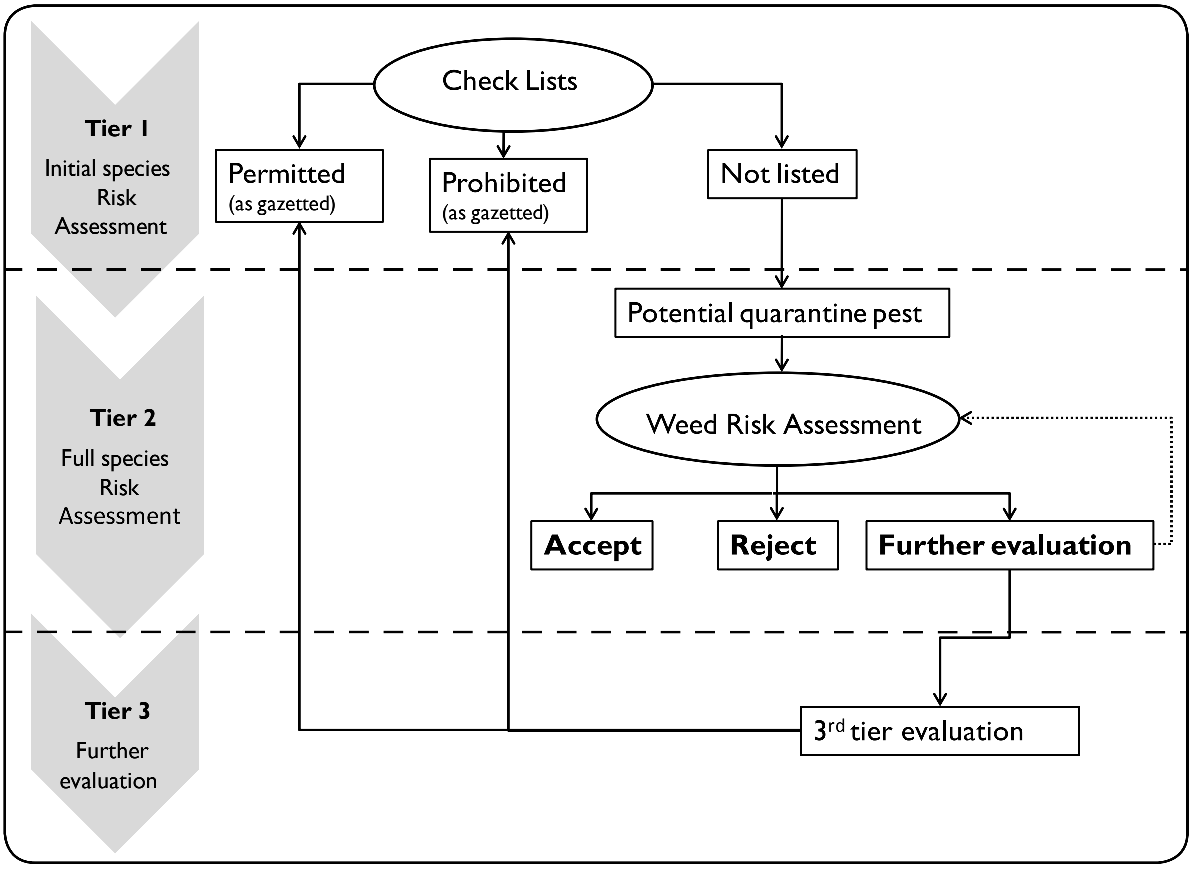

Figure 5.10: **An example of a three-tiered screening system proposed for plant imports to Australia (based on Pheloung 2003).**

The ability of the screening system to adequately identify weeds is a major indicator of the efficacy of such a model. Several recent studies have compared the various methods, for regions other than the region of development, to correctly identify invasive and non-invasive species from predetermined sample of plants from the region including known invaders. Examples of screening systems include (1) Australian weed risk assessment (WRA) scheme (Pheloung et al. 1999); (2) WRA with additional analysis by Daehler et al. (2004); (3) decision tree scheme of Reichard and Hamilton (1997); and (4) altered versions of existing methods (See Jefferson et al. 2004). These comparisons have revealed the WRA to most accurately determine the likelihood of invasive species (Daehler and Corrino 2000; Krivanek and Pysek 2006). For example, an American sample of plants of unknown invasive potential was correctly accepted or rejected over 80% of the time (Gordon et al. 2008). More recent testing of the WRA system sees it as an acceptable tool for assessing weed species beyond the region it was intended for (Crosti et al. 2010; Dawson et al. 2009).

5.8.3 Steps for reducing risks of invasion from biofuel plantation

All species that are to be introduced to an area must be screened for invasiveness if they are a new species to the area. This would apply even to species being moved to new locations within a country[21]. Screening and all appropriate phyto-sanitary requirements as specified by the countries national legislation must be adhered to.

[21] The term 'indigenous', ie a plant that historically occurs within a given country, is not typically very helpful, since the country boundaries seldom coincide with ecological boundaries. 'Endemic' to a particular habitat is more helpful. It is entirely possible for an indigenous species to be an alien invasive species in its own country, but in a new habitat.

Once approval has been obtained for planting a particular species in a biofuel plantation, a number of options are available to reduce the risk of unintended invasions. The biofuel crop itself might have the potential to invade, and if there is any possibility of this, then strict measures should be undertaken to prevent this happening. Monitoring must be conducted to give early warning of any start of invasion. It is also possible that when moving the biofuel products around that this could facilitate the transfer of other species, including pest species of the biofuel or other crops. IUCN has produced a simple and short manual to help government officials and biofuel plantation managers better understand how to minimise invasive risks (IUCN 2009).

Five key recommendations on reducing invasiveness from IUCN (2010) are:

1. Follow a precautionary approach when choosing feedstocks
2. Work with stakeholders to build capacity
3. Comply with local, national and regional regulations
4. Develop and follow Environmental Management Plans
5. Extend planning, monitoring and assessments beyond the field

This manual is freely available online[22]. Four key areas of intervention are proposed by IUCN as in Table 5.9. The IUCN report gives more detail on the proposed methods of intervention (IUCN 2009).

Table 5.9: **Proposed actions for key areas of intervention to reduce risks of invasion (IUCN 2010)**

Area of intervention	Proposed actions and tools
Planning	Cost benefit analysis Strategic environmental assessments Projects Environmental Impact Assessment Contingency fund
Importation	Quarantine process Phytosanitation regulation and action Comply with regulations Remember the pest associated with biofuel
Production	Follow best practices A contingency plan if an "escape" A contingency fund to pay for eradication, containment, management, or restoration monitoring system that checks for escapes and the presence of pests and pathogens. EMPs should ideally be audited by a neutral third party
Transportation	Reduce distance Process before transportation Monitoring of routes Awareness

[22] http://cmsdata.iucn.org/downloads/iucn_guidelines_on_biofuels_and_invasive_species_.pdf

5.9 Conclusions

Biofuel expansion carries with it a real risk of resulting in biodiversity loss. This risk is especially high for developing countries in the tropics where there is both a high concentration of biodiversity as well as high levels of threat to the biodiversity due to a multitude of land use pressures.

From a national strategic policy perspective, careful assessment is needed as to whether the biodiversity loss from biofuel is justified relative to the potential gains from biofuel programmes. In this regard strategic multi-criteria assessments are needed in which projected biodiversity impacts are one of the variables. The direct and indirect economic, human wellbeing and ethical consequences of biodiversity loss must not be forgotten. The extent of biodiversity loss can be mitigated to some extent by limiting the size of a proposed biofuel industry and by defining the habitats and land uses in which it is permissible. Impacts of indirect land use change must not be ignored.

Careful planning both at the strategic level and at the plantation level can greatly reduce the level of biodiversity loss. Mitigation measures may also be used to enhance biodiversity overall, so that even if some biodiversity is being lost from specific locations, the overall strategic biodiversity conservation objectives for the region can potentially be increased. Biodiversity impacts from biofuel expansion need to be considered against biodiversity impacts from alternate land use options. In this regard 'doing nothing' also has a biodiversity consequence, which may be greater or less than the consequences from introducing biofuels. Feedbacks between biofuel expansion and other drivers of biodiversity loss need consideration, as biofuel production could potentially reduce or enhance other drivers of biodiversity loss. The way these interactions between sectors are managed could greatly enhance overall biodiversity conservation.

Measuring or monitoring biodiversity and biodiversity impacts is complex and can be extremely costly. For a strategic perspective the BII tool is recommended as an appropriate method to consider overall biodiversity consequences of different biofuel and competing land use scenarios. Because the tool can be based on specialist input rather than raw data, it is relatively inexpensive to undertake a BII assessment, especially in areas of relatively limited data availability. It can, however ultilize more rigorous data sources if available. The BII approach is scalable and results can be disaggregated in a number of different ways. The BII approach is, however, poor at picking up impacts on rare and endangered species and should be run in parallel with a red data approach if impacts of this nature are anticipated.

Numerous techniques are available for local level assessment at the project level. It must, however, be stressed that any local biodiversity plan should be aligned with strategic conservation objectives. Simple screening can give a first cut as to if biodiversity impacts are likely to be an important issue. If biodiversity is likely to be important, then either the development should be abandoned, or more detailed approaches for biodiversity protection should be considered such as the High Conservation Value approach. In many situations, strategic conservation of specific areas within a biofuel estate which have high conservational value can greatly mitigate the overall biodiversity impact.

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