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ASSESSMENT OF THE MONETARY VALUE OF THE EXTERNAL EFFECTS OF ENVIRONMENTAL USE IN ESTONIA, ANALYSIS

STAGE 1

LISA 1

ANNEX 1

Ülevaade DPSIR kontseptsiooni kasutamisest Balti riikide vaates

A review of the DPSIR framework: perspectives for the Baltic States

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ABSTRACT

This brief review summarises the approaches taken by DPSIR studies internationally. It is clear from the literature that DPSIR has been employed in a diversity of settings across a wide variety of topics across the globe.

The review pays particular attention to state-of-the-art use of DPSIR, the types of indicators being used, key case studies, and the limitations of the DPSIR methodology and what to do about it. Given the requirement to predominantly focus on *Pressure*, *State* and *Impact* within the context of the 1st stage of external cost analysis in Estonia, the review pays special attention to these aspects.

A distinction is made between a characterisation of biophysical and socioeconomic *Impact*. The latter is to account for human welfare dimensions. The review draws on economic theory particularly where it concerns external factors, public goods, environmental valuation, ecosystem goods and services and, in this regard, provision is made of case study material involving biodiversity, tourism, protected areas, managed landscapes, and marine and water resources.

Key words: driver-pressure-state-impact-response, impact, biodiversity, Baltic states, Estonia, externalities

ÜLEVAADE

Käesolev lühike ülevaade võtab kokku lähenemisviisid DPSIR metoodika kasutamisel rahvusvahelisel tasemel. Läbitöötatud allikate põhjal võib väita, et DPSIR kontseptsiooni on erinevates riikides kasutatud erinevate teemade ja valdkondade käsitlemisel.

Ülevaates pööratakse tähelepanu eelkõige erinevatele DPSIR kasutusrakendustele, seejuures ka kasutatud indikaatoritele ning olulistele uuringutele. Lisaks käsitletakse DPSIR metoodika rakendamise plusse ja miinuseid ehk tugevaid ja nõrku külgi ning eelkõige seda, kuidas väljatoodud nõrkusi kompenseerida. Kuna analüüsis, mille raames käesolev ülevaade on koostatud, keskendutakse eelkõige survele, seisundile ja mõjudele, pöörab ka käesolev ülevaade põhitähelepanu neile komponentidele.

Vahet tuleb teha biofüüsikalistel ja sotsiaalmajanduslikel mõjudel. Viimased on seotud inimese heaolu mõõdikutega. Ülevaade käsitleb majandusteoreetilisi aspekte, eelkõige läbi teemade, mis puudutavad väliseid tegureid, avalikke hüvesid, keskkonna hindamist, ökosüsteemi hüvesid ja teenuseid. Nendeks teemadeks on elurikkus, turism, looduskaitsealad, majandatavad maastikud ning mere- ja mageveeressursid.

Võtmesõnad: vallapäästev jõud-surve-seisund-mõju-vastumeede, mõju, elurikkus, Balti riigid, Eesti, välismõju

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INTRODUCTION

This brief review of the literature serves to summarise the state-of-the-art approaches that encapsulate the DPSIR methodology, the theory behind the technique, empirical approaches used in its deployment, and case study examples used in its application. Above all the intention is to evaluate how studies document the *Impact* aspects of DPSIR and provide examples of actual measures of indicators used to specify impact.

The broad goal of the study which was conducted in the Baltic States can be defined as follows: it is to define environmental pressures, state, and impact in accordance with the DPSIR methodology. In phase I this involves the identification of a number of relevant indicators to establish the amount of pressure on the environment across ten categories. Phase II entails specifying pressures, where feasible in monetary terms and, finally, guidance will be given to policymakers concerning the regulation of negative external costs, those that may already be taxed and those which not (or perhaps to ascertain whether the level of regulation is adequate, or has not been over or underestimated).

A literature review summarising state-of-the-art approaches regarding how *Impact* has been used and defined in the DPSIR literature will be conducted. In the present study *Impact* is hereby defined as a change in status. Impact can be thought of as affecting two aspects - firstly, the *biophysical environment* and secondly the socioeconomic aspect on humans directly in terms of personal or social welfare. The DPSIR method is a useful technique but it does have some shortcomings. The review will identify key limitations of the methodology and pinpoint possible solutions to these as suggested by the DPSIR literature. The specific objectives are to:

1. Identify the types of measures that have been used to determine biophysical *Impact*;
2. Determine the types of indicators that have been used to specify socioeconomic or human *Impact*;
3. Depict key weaknesses in the DPSIR methodology and suggest solutions to overcome these based on the available literature.

We begin with some definitions of the approach. The abbreviation, DPSIR, refers to a conceptual framework that describes environmental problems and their relationships with a socio-economic domain in a policy-relevant manner. The technique is defined following Maxim et al (2009) as follows:

Social and economic developments (driving forces (D), impose pressures (P) on the environment and, as a consequence, the state (S) of the environment changes. This gives rise to impacts (I) on ecosystems, human health, and society, which may lead to a societal response (R) which feeds back driving forces, on state or on impacts via various mitigation or adaptive actions.

In this sense the method is depicted as a causal framework for describing the interactions between society and the environment. DPSIR was initiated to indicate cause-effect relationships between environmental and socioeconomic systems. The framework was designed to enable policymakers to better understand information being provided by indicator reports.

These interactions use indicators in a causal framework. Environmental indicators include physical, biological and social factors which can be represented in causal chain frameworks. Three causal chain frameworks are depicted in **Figure 1** following Niemeijer and De Groot (2008), pressure, state-response (PSR), driving force state-response (DSR), and driving force-pressure-state-impact-response (DPSIR).

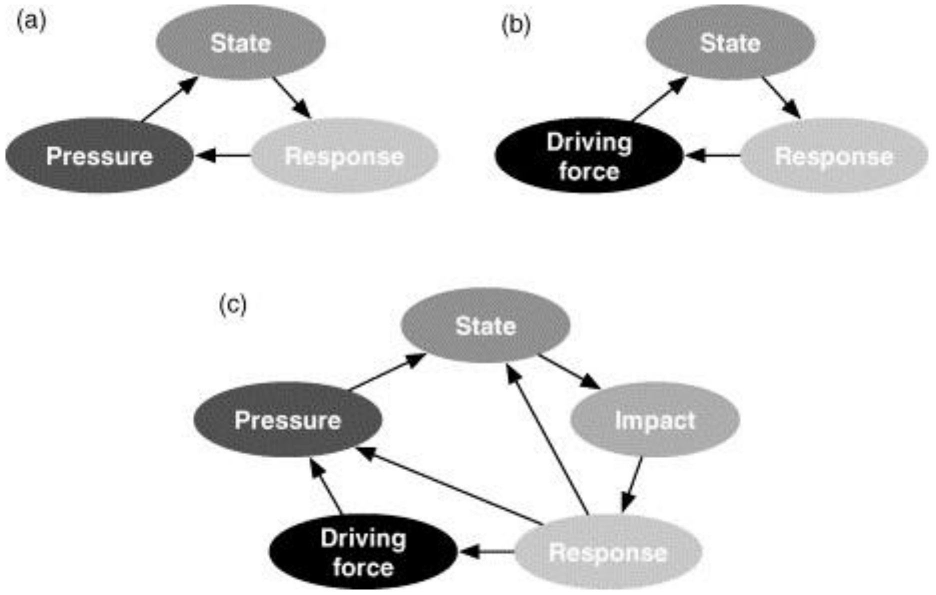

Figure 1. The (a) PSR, (b) DSR, and (c) DPSIR frameworks (Niemeijer and De Groot, 2008).

In (a), pressure on the environment gives rise to changes in state or environmental conditions and may initiate responses to deal with that pressure or state. In (b), driving force replaces pressure since economic activity can have detrimental and beneficial effects and it encompasses a broad range of issues, including government policies and economic and cultural factors. In (c), a distinction is made between indirect driving forces such as social and economic development and effects such as emissions/pollutants that impact directly upon the environment. Bell (2009) provides a good example of this in his tourism case study. The point is made that direct responses are those with a direct link to identified drivers. The inadequate treatment of sewage (from increased tourist pressures) leads to a response for more water treatment plants. Yet the growth of international tourist movements is linked to personal choices and expectations of increasing travel and tourism. A direct response is not possible. An indirect response in the form of an Environmental Impact Assessment Directive can be used. Crucially (c) also differentiates between impacts upon the state of the environment (pollutant concentration) and changes in the impact upon human wellbeing or health.

1 ANALYSIS OF STUDIES USING THE DPSIR FRAMEWORK

1.1 A review of the literature

A structured review of the literature was conducted and an initial search (from a variety of sources including Web of Science-all data bases, Scopus, OvidSP, Science Direct, and Google Scholar) revealed nearly 2,000 documents potentially linked to DPSIR in all fields and all countries. Sources were restricted to those which had been published in English. Many of these papers had a national and a regional focus (see **Table 1**, below) which were published in peer-reviewed international journals and also in some books. A more focussed screening by the author narrowed this down to about thirty candidate papers (again see **Table 1**). This was restricted to high quality papers (published between 2001-2017) which used the DPSIR framework, linked to the key indicator groups (air quality, noise, etc). Selected papers **included coverage of the Baltic States** and are providing good case study examples of how to address some of the problems associated with the technique. **The studies reflected a diverse and global use of DPSIR.** Early studies focussed on water management (watershed, estuarine, coastal, marine, lagoon, river systems), but later focussed on biodiversity, land degradation, farming systems, climate change, renewable energy, recreation and tourism, stakeholder interaction, and training, and in an urban setting.

Table 1. Candidate DPSIR studies identified in the study.

Author	Year	Topic	Rank
Berger and Hodge	1998	Overview	4
Bidone and Lacerda	2003	Water management	2
Bowen and Riley	2003	Water management	4
Newton et al	2003	Water management	2
Mander and Kuuba	2004	Landscape biodiversity	
Pirrone et al	2005	Water management	2
Borja et al	2006	Water management	2
Karageorgis et al	2006	Water management	2
Rapport and Singh	2006	Ecosystem health	4
Smalling and Dixon	2006	Agriculture	
Mangi et al	2007	Water management	3
Niemejer and De Groot	2008	Water management	5
Svarstad et al	2008	Biodiversity	3
Bell and Etherington	2009	Water management and tourism	5
Binimelis et al	2009	Agriculture	3

Author	Year	Topic	Rank
Haberl et al	2009	Biodiversity	2
Kuldna et al	2009	Agriculture, biodiversity	3
Maxim et al	2009	Biodiversity	5
Maxim and Spangenberg	2009	Biodiversity, chemical risks	4
Nobre	2009	Water management	3
Omann et al	2009	Climate change	4
Skoulikidis	2009	Water management	2
Skowrońska et al	2009	Water management	2
Spangenberg et al	2009	Biodiversity	2
Vihervaara & Kamppinen	2009	Forestry	2
Kimmel et al	2010	Wetlands	
Kulig et al	2010	Agri-food	2
Ness et al	2010	Water management	3
Piirimäe et al	2010	Water mangt, oil shale	
Atkins et al	2011	Water management	3
Hou et al	2011	Biodiversity	5
Rozīte and Vinklere	2011	Biodiversity and tourism	
Wolfslehner and Vacik	2011	Forests	3
Bell	2012	Water management	3
Tscherning et al	2012	DPSIR Overview	4
Yee et al	2012	Human health	3
Cooper	2013	Water management	5
Gregory et al	2013	Water management	3
Cavalho-Santos et al	2014	Forests	3
Hou et al	2014	Biodiversity	5
Melecis et al	2014	Biodiversity	
Spangenberg et al	2014	Agri-food, chemical, biodiversity	4
Sundblad, et al	2014	Water management	3
Gari et al	2015		3

Author	Year	Topic	Rank
Kubacka et al	2016	Agri-environment/protected area	4
Lewison et al	2016	Water management	4
Mohammadizadeh et al	2016	Air quality	2
Osterwind et al	2016	Water management	3
Sun et al	2016	Water management	2
Zhang et al	2016	Water management	3
Dempsey et al	2017	Water management	4
Hubeau et al	2017	Agri-food	4
Pires et al	2017	Water management	2
Rodriguez-Gonzalez et al	2017	Forests	2

1.2 Candidate literature by study topic

A number of topics of analysis have been defined in this study across ten categories, **including ambient air, odour, vibration, noise, water pollution, water abstraction, dams, soil pollution, waste, and land take.** To facilitate consultation and reference to the candidate literature, studies have been grouped according to these categories as shown in the table below. The water category includes water pollution, water abstraction and dams. Biodiversity studies where they overlap with soil and water have been included in those categories.

Table 2. Candidate DPSIR studies identified in the study according to the study topics.

Author	Year	Air/odour	Vibration/noise	Water	Soil	Waste	Land take
Berger and Hodge	1998			X			
Bidone and Lacerda	2003			X			
Bowen and Riley	2003			X			
Newton et al	2003			X			
Mander and Kuuba	2004			X			X
Pirrone et al	2005			X			
Borja et al	2006			X			
Karageorgis et al	2006			X			
Rapport and Singh	2006						
Smalling and Dixon	2006						X
Mangi et al	2007			X			
Niemejer and De Groot	2008			X			
Svarstad et al	2008						X
Bell and Etherington	2009			X	X		
Binimelis et al	2009						X
Rozīte and Vinklere	2011			X			X
Wolfslehner and Vacik	2011						X
Bell	2012			X			
Tscherning et al	2012						
Yee et al	2012	X					
Cooper	2013			X			
Gregory et al	2013			X			
Cavalho-Santos et al	2014			X			
Hou et al	2014						X
Melecis et al	2014			X			X
Spangenberg et al	2014				X		X
Sundblad, et al	2014			X			
Gari et al	2015			X			

Author	Year	Air/odour	Vibration/noise	Water	Soil	Waste	Land take
Nassi and Löffler	2015	X					X
Zhou et al	2015	X					
Wang et al	2015			X			
Baldwin et al	2016			X			
Dolbeth et al	2016			X			
Kubacka et al	2016						X
Lewison et al	2016			X			
Mohammadizadeh et al	2016	X					
Osterwind et al	2016			X			
Sun et al	2016			X			
Zhang et al	2016			X			
Dempsey et al	2017			X			
Hubeau et al	2017						X
Pires et al	2017			X			
Rodriguez-Gonzalez et al	2017			X			

Baltic States using the DPSIR framework

A number of DPSIR studies have focussed on the Baltic States/the Baltic region (see **Table 3** below). Mander and Kuuba (2004) provide an account of the marginalisation-intensification relationship that has been taking place in the Baltic States since the early 1990s. This is a useful paper in terms of summarising the main trends in landscape change for the Baltic countries, but the use of DPSIR is treated as an afterthought. Skowrońska et al (2009) is a useful account of the use of DPSIR for the Baltics and is worth reviewing.

This is a regional analysis with a focus on drivers pressures and state. It is a useful source with respect to identifying data sources and includes a summary of data paucity and how this is dealt with. Kimmel is definitely worth reviewing and is very relevant. This is a good example of an application of the DPSIR framework in terms of land use. Some valuable points are made in terms of steps towards optimal land use planning. The paper by Ness et al (2010) represents a regional analysis of the Baltic Sea. This is a useful paper in terms of data sources used to derive indicators for the Baltic, but the focus is on Swedish agriculture rather than the Baltic States. Rozīte and Vinklere provide an account of tourism pressures and drivers in Latvia but there is no analysis or in-depth evaluation of changes in *State* or *Impact*.

Melecis et al (2014) provide quite a comprehensive account of the use of the DPSIR framework in the Baltic States. The paper is worth a review and provides a good account of problems facing the region, and an attempt is made to refine and adapt the DPSIR framework by treating local and external drivers as two separate groups. Piirimäe et al (2014) is the only application of cognitive mapping and the use of causal networks in the Baltic States of which the author is aware. An attempt is made to apply these techniques to refine the DPSIR framework and the paper provides a good overview of how to deal with transboundary external factors associated with landscape change.

Table 3. DPSIR studies conducted in the Baltic States.

Author	Year	Topic	Country
Mander and Kuuba	2004	Landscape biodiversity	Estonia
Skowrońska et al	2009	Baltic Sea (chemical pollution)	Regional
Kimmel et al	2010	Wetlands	Estonia
Ness et al	2010	Baltic Sea (agricultural pollution)	Regional
Rozīte and Vinklere	2011	Biodiversity and tourism	Latvia
Melecis et al	2014	Biodiversity	Latvia
Piirimäe et al	2014	Water management. Oil shale	Estonia/Russia

The paper by Bell and Etherington (2009) is really an overview paper on the DPSIR technique. Very little is said about the Baltics as a case study. Kuldna et al (2009) is very relevant. Although the authors are from the Baltic States, very little is said about the Baltic States *per se*.

1.3 Advantages of the DPSIR methodology

Maxim et al (2009) suggest that the technique serves to structure the indicators with reference to political objectives which are related to the environmental problem at hand, and identifies supposed causal relationships in a clear manner which appeals to policymakers and makes it easy for them to grasp the environmental concern that is of relevance. The real strength of the DPSIR method is its capacity to identify and visualise in a simplified manner complex cause-and-effect relationships between society and environment. Agyemang et al (2007) showed how DPSIR can be used very effectively to organise complex environmental relationships and present them in a simplified way to policymakers. They showed how DPSIR enhanced the efficacy of land degradation control measures because processes and linkages could be readily linked to impacts and responses. Spanó et al (2017) reveal that the DPSIR framework was highly valuable when it came to identifying the most critical issues in land use planning.

The method pinpointed key threats to a loss of ecosystem services (air and water purification, water flow regulation, and biodiversity) were highlighted through the causal relationships of the DPSIR framework. One important goal of the DPSIR framework is to bridge the gap between scientific systems and the public regarding environmental issues and concerns. In this vein it is also an effective tool when it comes to improving communications between researchers from different disciplines (Svarstad et al, 2008). Bell and Etherington (2009) suggest DPSIR sends relatively clear messages to policymakers and individuals. Bidone and Lacerda (2008) found the DPSIR framework to be an effective method when it came to evaluating policies in Brazil. The use of the framework reduces the complexity of indicators and makes the interrelations between them readily understandable. A further strength of DPSIR which was highlighted by Bell and Etherington (2009) is that it is good at identifying the scale of predicted driver increases and in pinpointing the level of decoupling required, but is less effective at the response. The technique is used widely by the European Union, its member states, and elsewhere (Iannucci et al, 2017). The methodology does not model the environment as such. Instead it provides a systems framework in order to better understand the semantics between the DPSIR components and can be useful as a means of processing big data. These issues are summarised in Table 4.

Table 4. Strengths of the DPSIR framework.

DPSIR - advantages
Identifies cause-and-effect relationships between environment and human systems (Smalling and Dixon, 2006; Ness et al, 2010)
Enables project-related structuring of indicators (Svarstad et al, 2008; Tscherning et al, 2012; Spangenberg et al, 2014)
Integrates knowledge from diverse disciplines (Spanó et al, 2017)
Scope for incorporating stakeholders in decision-making using participatory processes (Tscherning et al, 2012)
Useful for framing international monitoring and reporting (UNEP, 2012)
Useful for developing indicator sets (Kristensen, 2003)

1.4 Disadvantages of the DPSIR methodology

However, several papers suggest that the method has a number of flaws. From the key literature reviewed and depicted above, in broad terms these can be depicted as follows:

- ***Impact***: a need to separate socioeconomic impacts from environmental impacts, to identify a welfare or human impact category, and the formulation of indicators representing socioeconomic impacts in a manner which is amenable to economic analysis;
- **Complexity**: the inherent complexity of socio-ecological systems;
- **Scale effects**: local regional and global concerns and their interrelationships;
- **Data limitations:** data paucity;
- **An interdisciplinary approach:** the incorporation of different disciplines and a DPSIR framework that is multidisciplinary in its perspective;
- **Stakeholders:** the inclusion of a DPSIR approach that is not confined to 'experts in the field' but which embraces multi-stakeholder interests, particularly where stakeholders have important knowledge or are impacted by the process.

Environmental and socio-economic impacts

Depending upon the disciplinary perspective, the notion of *Impact* can have a very different focus. The distinction between biophysical impact and human impact can sometimes seem blurred. For example, a state change of the system (eg. benthos in a water column) takes place and results in an impact upon society (a loss of biodiversity, a degraded habitat), which when linked to human welfare results in positive or negative implications. According to some studies (Maxim et al, 2009), although the effects of human activities are related and occur in a sequence, it is only the final step in the sequence (that which directly affects changes in environmental use functions by humans) that should be considered as being an impact. There is an issue of including the effects of social systems in the same category and of double counting. Cooper (2013) suggests that many DPSIR studies express impacts in terms of the effects on environmental quality on both social and ecological systems. This is problematic for a number of reasons. Firstly, it obscures the boundary between the *Impact* and *State* categories. With reference to an example of eutrophication, Cooper (2013) gives an example showing that ecosystem changes could either be treated as a manifestation of *Impact* or, alternatively, as an aspect of *State*. Secondly, it becomes problematic if the effects on social and ecological systems are included in the same *Impact* category. The bioscience impact is discussed first.

Bioscience impact

The literature appears to indicate that, in the biosciences, an *Impact* refers to the **biotic and abiotic components of ecosystems** (aquatic, terrestrial, atmospheric). Examples may be changes in the chemical composition of air or water, or changes in ecosystem productivity, stability, or resilience.

Socioeconomic impact

In contrast a socioeconomic perspective deals with human systems: with consequential impacts on human welfare through changes in productivity, health, amenity, and existence value (Cooper, 2013) and impacts upon humans, ecosystems, and man-made capital (EEA, 2012). Also with human health and welfare, longevity, morbidity, the value of ecological goods and services, and other, non-use values. These are associated with changes in environmental functions or services such as resource provision, water and air quality, soil fertility, physical and mental health, effects on physical capital, and social relations and social cohesion. This can be extended to include effects on future generations.

Bowen and Riley (2003) give a good account of **early pressure-state-response models which have been developed by the OECD**, and go on to show that the early use of this approach was problematic for three reasons:

1. because it did not factor natural causes into the pressure category (natural variability and episodic events find no easy place in the model),
2. it did not identify the underlying reasons for the pressures, and
3. it failed to define specific indicators regarding social impact.

The inherent complexity of ecosystems

Berger and Hodge (1998) suggest that the **DPSIR framework works best when an environmental problem has clearly been identified and linked to a causative group of human activities**. Responses are established to address the known relationship and indicators are identified to monitor the efficacy of the response. The problem they suggest is that the sequence is predominantly reactive rather than anticipatory in nature. Environmental changes are inherently unpredictable.

Niemeijer and De Groot (2008) find that the **DPSIR framework is inadequate for complex interrelationships,** with an emphasis on narrow causal chain one-to-one relationships. Svarstad et al (2008), using the example of biodiversity, argue that the DPSIR method is not a tool for generating neutral knowledge but, instead, the framework reproduces the discursive positions associated with the applicant. Using examples, they show how the framework tends to favour conservationist or preservationist positions with regards to biodiversity. Bell and Etherington (2009) make the point that ecosystems are complex, as are responses to the problems of issues such as global warming. They suggest that in the case of global warming responses can involve multiple instruments including European ETS, policies for renewables, and carbon capture and storage. These policies are designed to act as reverse drivers (decoupling energy production from emissions) but the latter responses may themselves induce impacts (such as marine habitat loss). Biofuel production may induce increased nutrient loads into river and coastal systems.

Kubacka et al (2016) provide a helpful summary of the **weaknesses of the DPSIR framework in dealing with complex systems and in addressing sustainability concerns**. They assert that DPSIR is limited in dealing with systems dynamics, is reliant on linear unidirectional causal chains, and can ignore key non-human drivers involved in environmental change (Berger and Hodge, 1998). They cite Rapport et al (1998) which is instructive in dealing with this theme. David Rapport and colleagues provide a useful critique of the DPSIR framework and make an important contribution to an understanding of Ecosystem Health and the role played by DPSIR in state of environmental reporting (SOER) in general (see Rapport and Singh, 2006 for further details).

Rapport and Singh (2006) provide an excellent critique of the PSR framework (pp415-416) in their paper on state of the environment reporting (SOER) and ecosystem health. They include a number of topics on this theme including:

1. the lack of a bottom line,
2. an allowance for options other than negative human impacts on the environment,
3. the lack of a dynamic picture and historical aspects,
4. insufficient focus on the properties of the ecosystem, including ecosystem health and ecosystem services,
5. inadequate focus on ecosystem complexities and uncertainty,
6. a need to account for the root causes of environmental transformation.

Scale effects

Bell and Etherington (2009) suggest that DPSIR's value as a predictive tool *per se* is limited. The simple causal relationships fail to capture the complexity inherent in social and natural systems. A specific impact can be caused by a number of *State* conditions, and by responses to other state impacts (ie. intervention failure) in a synergistic manner. A clear understanding of this is problematic when each of these is dealt with separately. There is a need for clear anchors to specify the method to make it relevant to the biosciences or the social sciences. Bell and Etherington (2009) provide a useful summary of the deficiencies of the DPSIR method, particularly regarding limits to data, the complexity of ecosystems and regional limitations, and the fact that the tool does not address local-global interrelationships using tourism as an example.

In their study of the Grand Bank ecosystem, Dempsey et al (2017) caution that changes in the drivers and pressures are often not immediately apparent in the structure of the fish community structure (state variables). They also developed a series of indicators and built these into the DPSIR framework. In so doing they warn that the categorisation of indicators is often contextual and is by no means a straightforward exercise. They suggest that responses do not directly affect the state. Instead they influence drivers and pressures which then result in changes to the state. There is a useful section in their paper (pp337) at the end of section 4.2, highlighting some of the subtleties of incorporating indicators into the DPSIR framework.

Kubacka et al (2016) find that the DPSIR framework was not particularly effective at revealing relationships between the system components. This is consistent with several other studies (Carr et al, 2007; Niemeijer and De Groot, 2008). They showed instead that the framework can be very useful for monitoring and evaluating the landscape at the local level, rather than at the national or global level. Kubacka et al (2016) is a useful reference in terms of developing agri-environmental indicators to monitor landscape change.

Relatively few examples exist regarding the use of the DPSIR framework where it is applied to forests. Rodriguez-Gonzalez (2016) provide a useful account of how to monitor forest dynamics using the DPSIR framework in Spain. The value in this study is the integration of remote sensing and field data over time within the framework. The focus remains entirely on environmental aspects and there is no attempt to evaluate *Impact* from a social perspective. The work also does not focus on factors which are related to forest composition such as biodiversity.

Data limitations

Bell and Etherington (2009) point out that the **DPSIR framework requires extensive statistical work in terms of physical and socioeconomic factors which may not always be available**. The use of the DPSIR framework within the context of biodiversity is fraught with difficulties given the frequent lack of data at different spatial and temporal scales in many studies (Hou et al, 2014).

Haberl et al (2009) raise a number of points regarding mismatches between the scale(s) at which biodiversity is monitored, the scale(s) of its management, and its scale of conservation. Migratory species serve as an important example. The authors suggest (pp1800) that a combination of GIS, remote sensing, statistical and survey data is often required.

An interdisciplinary approach and stakeholders and their inclusion

One branch of the literature suggests that the DPSIR framework is too hierarchical, resulting in a marginalisation of informal responses to pressures and drivers and is too exclusive of a range of stakeholders. In DPSIR studies there is also the question of stakeholder involvement. In many decisions regarding the governance of ecosystem the focus is typically on agents with vested interests in the ecosystem itself (farmers, fishers, the oil industry), or stakeholders who are affected or impacted by the ecosystem in some way (Armstrong and van den Hove, 2008). Stakeholders may also have valuable local knowledge about resources which external experts do not have and can make valuable contributions to a DPSIR analysis. These may be fishermen, oil producers, or farmers for example (Levin et al, 2009).

There are a number of DPSIR papers that discuss stakeholder involvement and impact and which employ participatory approaches to stakeholder involvement (Carr et al, 2007; Svarstad et al, 2008). Tscherning et al (2012) provide a good overview of the literature on this topic and suggest that the DPSIR framework can be a useful tool when it comes to integrating knowledge across disciplines, but that many studies fail to achieve this.

1.5 Overcoming problems with the DPSIR method

In what follows we provide a number of examples of how various analysts have dealt with these issues. This includes a short case study description as well as a summary of the types of indicators used in the literature, and how these were applied in the case study and why. This is followed by a short discussion of the types of indicators of relevance for the present Baltics DPSIR study and a short explanation of why these indicators were identified. To begin, see the table below for a summary of the deficiencies of the DPSIR framework as identified by the relevant literature, as well as suggested approaches to remedy these shortcomings.

Table 5. Problems identified with the DPSIR methodology and suggested solutions identified from the relevant literature.

Issue	DPSIR Problem	Solution
Impact	Early PSR models did not account for the social impact of environmental change (Bowen and Riley, 2003)	Later DPSIR models explicitly account for *Impact*
	Clarity of the definition subject to interpretation. Obscure boundary between ecological and social Impacts, particularly human welfare/wellbeing (Cooper, 2013; Spangenberg et al, 2014)	Separate category for environmental and welfare effects (Atkins et al, 2011; Cooper, 2013; Bowen and Riley, 2003). Includes a denominated 'welfare' category to account for human impact
	Principal focus on environmental factors not social issues (Bowen and Riley, 2003; Kulig et al, 2010)	Additions to incorporate social, institutional and natural systems pressures (Bowen and Riley, 2003)
Complexity	Early PSR models limited, natural variability and episodic events not accommodated explicitly in the model. Problems associated with separating system components (Berger and Hodge, 1998)	Focus on interconnections between different indicators. Replace uni-directional causal chains with multiple causal networks. Narrowing down of the indicators.
	Problem over-simplification, does not focus on complexity of system & its interactions (Bowen and Riley, 2003). Niemeijer and De Groot, 2008; Bell and Etherington, 2009; Maxim et al, 2009). Inadequate for complex interrelationships, emphasis on narrow causal chain one-to-one relationships.	Development of a double-DPSIR scheme to ensure careful selection of responses to address drivers (Spangenberg et al, 2014)
	Cannot take account of systems dynamics, reliant on linear unidirectional causal chains and can ignore key non-human drivers involved in environmental change (Berger and Hodge, 1998)	
	Difficulties with dealing with factors that represent both response and drivers without incorporating different levels of responses and drivers (Ness et al, 2010; Tscherning et al, 2012; Spangenberg et al, 2014; Dempsey et al, 2017). Narrow definition of causality chains can be problematic. Responses from one loop are the drivers for another (Spangenberg et al, 2014).	
	Regional limitations indirectly linked to globalisation (Bell and Letherington, 2009)	Shows how to contextualise forces of globalisation linked to regional impacts Bell and Letherington, 2009

Scale	The link between socioeconomic drivers of biodiversity and related pressures is often incomplete and reliable indicators are often absent in the field (Haberl et al, 2007)	Haberl et al (2009) identify a typology and ranking system for identifying data gaps in terms of drivers, pressures, and states at different scales.
Data limitations	DPSIR lacks interdisciplinary. DPSIR confined to specialists, not widely available to impacted stakeholders (Bell and Etherington, 2009; Tscherning et al, 2012)	Development of conceptual models integrating different disciplines. Making the results more accessible to different disciplines and stakeholders (Tscherning et al, 2012)
Interdisciplinary approach	DPSIR framework too hierarchical, hierarchy of elements can lead to a hierarchy of actors and can limit its use in sustainability research (Carr et al, 2007). Hierarchy also in terms of marginalisation of informal responses to pressures and drivers and too exclusive of stakeholders (Carr et al, 2007; Bell, 2012). DPSIR excludes normative perspectives (Svarstad et al, 2008)	Integrate local stakeholder knowledge and narratives into cause-effect relationships using discourse analysis (Svarstad et al, 2008; Tscherning et al). Employs participatory approaches to account for public value and to incorporate local knowledge (Bell, 2012; Spanó et al, 2017)
	DPSIR model is ambiguous when used as an analytical tool in value-laden situations (Binimelis et al, 2009)	Redefine DPSIR categories by allowing different opposing groups to convey own narratives (Binimelis et al, 2009)
Stakeholders		

1.5.1 *Environmental impact*

According to the EEA, impacts are the consequences of changes in the state of the environment when it comes to environmental functions (EEA, 2005). **Cooper makes the point that natural scientists find it difficult to think in terms of impact relating to social systems** because they think about biophysical changes. He argues this makes a clear link between the causes and effects of environmental change. Indeed, some DPSIR studies focus mainly on environmental impacts. See for example Cave et al (2003) which focussed on ecosystem not social effects.

Berger and Hodge (1998) suggest that the language of DPSIR may not help to make a clear link between public policy and decision-making. **They refer to issues concerning the 'language gap' between natural scientists and policymakers.** Ecosystem perturbation may degrade ecosystems but may also help them to rejuvenate (such as a wild fire, for instance) as Holling (1986) has argued.

1.5.2 Socio-economic impacts

Arguments have been made in a number of studies for isolating impacts upon social systems as a separate category (Bowen and Riley, 2003: Atkins et al, 2011; Cooper, 2013; ELME, 2007). DPSIR has been modified where the impact element is solely concerned with human impact (mDPSIR = modified DPSIR) (ELME, 2007). Cooper's (2013) paper points to a modified version of DPSIR in which he argues for and includes a welfare category encompassing and replacing *Impact*, with the approach becoming DPSWR (Cooper, 2013). This eliminates the diffuse boundary between state and ecological impact by singularly focussing on human welfare when dealing with impact. The ecological impact is moved to the state category. In DPSWR impact refers to state and welfare changes.

Bowen and Riley (2003) show that early PSR models did not explicitly account for *Impact*. They have argued for the specific need for socio-economic impact indicators which record the specific impacts on ecosystems as well as on human welfare. In their study, *Impact* explicitly accounts for welfare effects not impacts upon the ecosystem by measuring changes to human health. Cooper (2013) and others (Bowen and Riley, 2003; Atkins et al, 2011) propose a modified DPSIR framework which explicitly includes a 'welfare' component to account for the impact upon human agency as opposed to conditions which relate primarily to ecosystems. He argues that this accounts for the welfare effects of impacts, improves definitional clarity, and is better aligned with the needs of economic analysis.

Cooper (2013) also suggests that separating the two forms of impact can be a useful way of relating this to a driver and response category. The use of fertilisers increases farm yields and the welfare of A, but leads to ecosystem impacts (eutrophication, say), measured in biophysical terms as B, which give rise to external costs C. In this example, we may not be too concerned about B since A and C are directly commensurable and, in this case, represent the benefits and costs (social costs) of fertiliser use. In this case C may include the welfare impact upon humans directly as well as the reduced productivity of the ecosystem in terms of its capacity to provide goods and services, but these are also measured as welfare effects. So we use the final step in the sequence, in this case C, as the *Impact*. Measured in this way there is direct correspondence with the components and requirements of cost benefit analysis. We can weigh the social benefits with the social costs and then think in terms of a response based on these trade-offs.

The welfare category can be defined in terms of monetary units, but also conceptually (especially if valuation data is lacking) by referring to, for instance, reductions in employment, levels of morbidity, life expectancy, or by setting certain ethical standards of ecosystem provision.

Rather than measuring the impact, a number of studies have suggested identifying risk categories for policymakers without directly specifying the cost to humans. With respect to the latter, these impacts can either be measured directly, using valuation techniques, for example or, alternatively, reference can be made to certain targets, the risks involved in not meeting the Water Framework Directive targets (in terms of low, medium, or high), for example, and the costs of meeting these goals is then considered as part of a response. As another example, a study in Brazil which set certain targets with respect to water quality and then established the costs of meeting these targets using a Cost Benefit Analysis.

Bowen and Riley (2003) also suggest that it is very important in DPSIR studies that a clear performance measuring system is outlined at the outset which must stipulate that each action can be measured using input, output, outcome, and impact indicators and design objectives with this in mind.

Cavalho-Santos et al (2014) provide an example of how the ecosystem services concept can be mapped into the DPSIR framework in their study on the provision of forest and hydrological services in Portugal. The study does not consider the dynamic aspects of forest change or provide much detail about how DPSIR can be refined *per se*. It does nevertheless represent a useful example of how ecosystem services can be incorporated into DPSIR. This is done by incorporating the 'properties' which are associated with forests (which include biophysical structure, processes, functions, services, and benefits) into the *State* component as shown in the figure below.

Hou et al (2014) is noteworthy for its attempt to quantify both the social and environmental impact of biodiversity loss and to try and integrate these aspects in a quantitative manner. They find that the linkages between the DPSIR components is better depicted as a causal network than a causal chain.

1.5.3 Solutions presented in this study

The inherent complexity of ecosystems

Svarstad et al (2008) suggest that it is imperative that the DPSIR framework accommodates non-preservationist perspectives. Rapport and Singh (2006) provide a comprehensive list of points in order to address some of the limitations of PSR frameworks. They propose the idea of monitoring natural capital stocks, the use of the concept of the ecological footprint, and an analysis of ecological integrity to address concerns raised earlier regarding the establishment of the 'bottom line'. They raise the point that human activities can have both negative and positive impacts upon the environment. They employ the term 'eustress' to explain how human activity can also enhance the organisation, the resilience of ecosystems, and also have the opposite effect (distress) as shown in the figure below. They provide some examples of how to build dynamics into the framework by incorporating non-linear effects and thresholds to account for the fact that ecosystems may exist which are far from being in a steady state.

The concept of ecosystem services, they argue, provides a useful pathway in which to link transformed environments which may be affecting human welfare. Adaptive management can be employed to try and deal with fundamental uncertainties, along with elements of surprise and the unexpected. They suggest some thought needs to be given in SOER to broader issues which relate to scale, including the effects of population growth, globalisation, inequality, and consumerism.

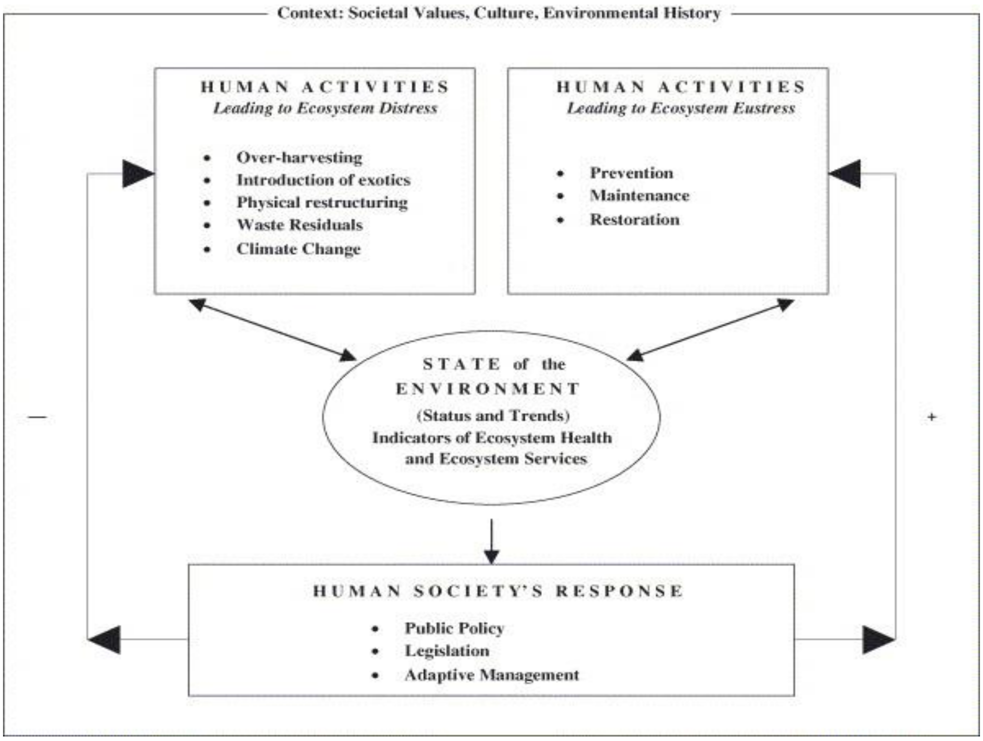

Figure 2. The effects of human activities can result in ecosystem distress and *eustress* (being defined as a perturbation that enhances system function) (Rapport and Singh, 2006).

Wolfslehner and Vacik (2011) in their study on sustainable forest management in Austria use network analysis to define the causal relationships between DPSIR components. This paper is instructive insofar as it develops quantitative tools involving cognitive mapping and causal networks to demonstrate the links between components and indicators in forest management.

Scale effects

The paper by Bell and Letherington (2009) is worthy of note in two regards. It contains a useful discussion on globalisation, scale, and issues which relate to consumer sovereignty and impact, and also the complex policy issues that are likely in defining *Impact* and developing responses. It has an excellent section on decoupling. They suggest that the monitoring of the effectiveness of responses is a critical issue in ascertaining whether decoupling - the separation of the growth of economic goods from negative environmental impacts - is in fact taking place (pp80-81).

Haberl et al (2009) give several examples indicating that an impact in one area may be affected by a driver many hundreds of kilometres away. A large proportion of the food eaten within a defined area, between 100-10,000km^2, is likely to come from agricultural activities which take place outside this area, and the production of this food will have an impact upon biodiversity outside this area. The impact of climate change is likely to be outside an area of study. Shifts between local and global pressures can make it very difficult to pinpoint the precise source of pressures which serve to impact upon biodiversity. Bridging the gaps between the scale at which data is amassed at the field level alongside socio-economic and political data represents a considerable challenge.

Data limitations

The link between the socioeconomic drivers of biodiversity and related pressures is often incomplete, and reliable indicators are often absent in the field (Haberl et al, 2007). Haberl et al (2009) suggest that sourcing data on biodiversity at different spatial scales represents a particular challenge. They identify a typology and ranking system for identifying data gaps in terms of drivers, pressures and states at different scales.

Consideration needs to be given to the target audience. The implications depend upon whether this is viewed from the perspective of society as a whole or in terms of one particular group (Atkins et al, 2011). Some studies use DPSIR to look at groups of fishermen or farmers. This requires the identification of the user community and, where feasible, a scientific analytical and/or monetary evaluation of the social benefits and costs. Here, defining *Impact* is easier than dealing with global public goods such as climate change. Lupp et al (2016) present evidence to show that stakeholders find it very difficult to put forward day-to-day solutions towards the adaptation goals for climate change unless clear tangible realistic goal-setting is put in place.

Vihervaara and Kamppinen (2009) use DPSIR within the context of forest management decision-making and ecosystem-based management. The study involves a number of stakeholders and their views are incorporated into the DPSIR framework. The paper is instructive insofar as it discusses a number of relevant pressures affecting the forest industry in Finland. The integration of components and the use of the DPSIR framework is weak, however.

Bell (2012) suggests that although the DPSIR is limited in scope it can, if applied in a participatory manner, help to create outcomes which are of value to local populations and the wider public. Workshops and participatory approaches are used to define indicators. The method does not establish DPSIR criteria first. Instead workshop participants are asked for their views and resulting measures are then framed in terms of the DPSIR. Tscherning et al (2012) provide a good overview of the necessary steps to ensure that the DPSIR method is interdisciplinary and participatory in perspective. Spanó et al (2017) suggest that holding local workshops with stakeholders serves as a good basis for sound decision-making through an inclusive, legitimate, informed participation of stakeholders, while also being a useful way of gaining their attention and input, and ultimately leads to sustainable land-use solutions using a green infrastructure-planning approach.

Kuldna et al (2009) include a number of European eco-toxicology experts in order to evaluate the impact of pesticide use on pollinators. This paper is very relevant to this study. It explores the relationship between pollinators and agrochemical use, and is a useful reference regarding the problem of how to regulate agrochemical use in the Baltic States. It makes an interesting link between the employment of agri-environment schemes and impact on environment as a response to the problem.

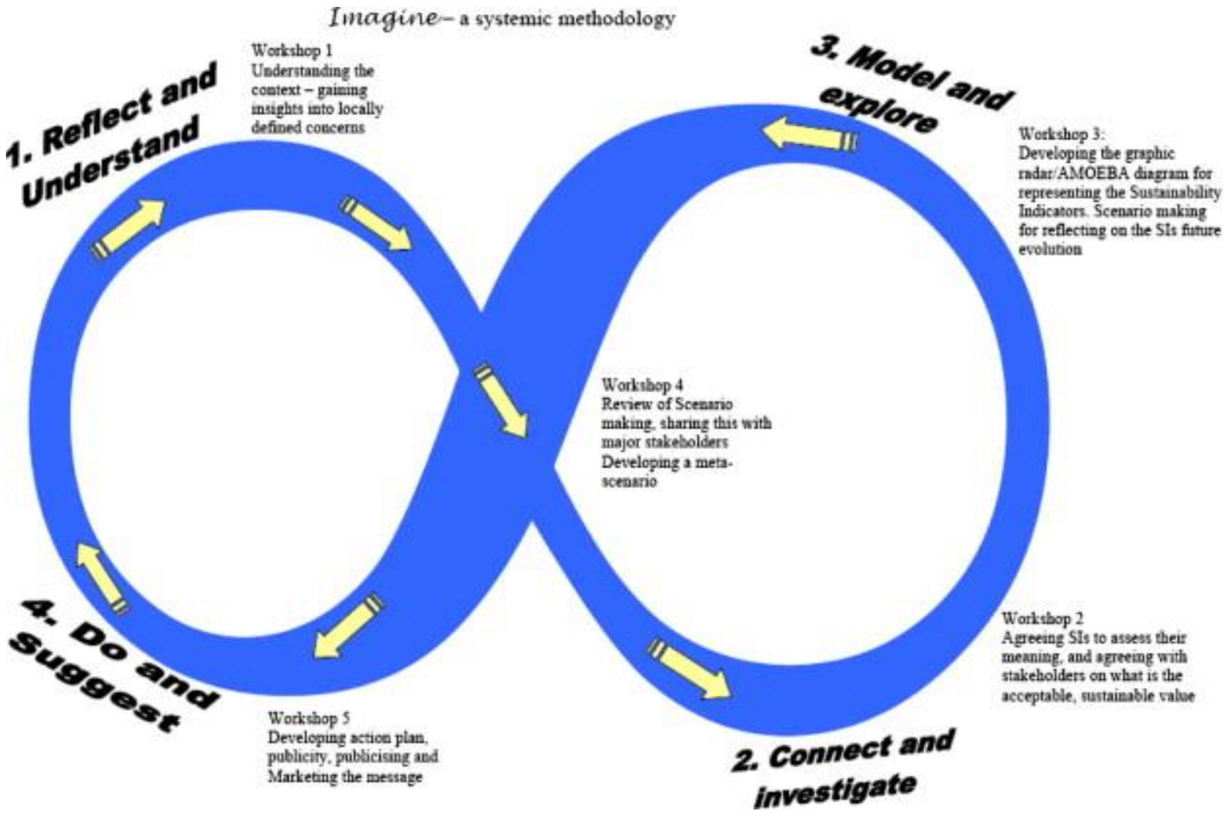

Figure 3. Imagine - a multi methodology (Bell, 2012).

Indicators used to define *State* and *Impact*

Given the requirement to predominantly focus on *State* and *Impact* within the context of DPSIR frameworks, the review pays special attention to these aspects and includes indicator categories on state and environmental impact and social impact. Candidate papers from the available literature provide some examples, and these are highlighted according to the ten topics described above. As the table below shows, these are split into biophysical or social impact categories. A decline in fish productivity due to pollution can be estimated in tonnes but the direct effect on humans can be estimated in terms of the decline in a fish catch per unit effort for a stakeholder group.

These driver categories are important, they argue, since they address the root causes of society, economy, and the state of the environment.

A number of case study topics which are of relevance to the study are presented in the next chapter. These include biodiversity, forests, land abandonment, land fragmentation, and chemicals used in agriculture. This section is followed by economic and policy aspects.

2 A DESCRIPTION OF CASE STUDIES BY ENVIRONMENTAL TOPIC

2.1 Biodiversity

2.1.1 Genetic and functional diversity

Several definitions have been proposed to capture the multifaceted nature of biodiversity (ecosystems, species, and genes). This is acknowledged in the definition which has been developed in the convention on biodiversity as follows: *'Biological diversity' means the variability amongst living organisms of 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 amongst ecosystems* (UNEP, 1995).[1]

A distinction is made in the literature between **genetic diversity and functional diversity**. **Genetic diversity** usually refers to the genetic variation that exists within a species (the *gene pool*). Genes are the fundamental unit of biodiversity and the ultimate source of all variation amongst all animal and plant species (Dobzhansky, 1970; Soulé and Wilcox, 1980). Genetic diversity has been proposed as the basis on which to take conservation decisions using the evolutionary distinctiveness of taxa when assigning them priorities for preservation (Vane-Wright et al*,* 1991; Crozier, 1992; Solow et al*,* 1993; Weitzman, 1998). Here, relative ecological value is based on how far away species are from one another genetically, and an objective value is assigned to the taxonomic distinctiveness or degree of 'independent evolutionary history' (IEH) that is vested in a species (Vane-Wright et al*,* 1991).

Conservation groups and professional conservationists often exploit certain species and ecosystems to further their own conservation goals. **Conservationists have referred to charismatic species that win the hearts of the general public as *flagship species***. This may be at the expense of less well-known species that may be critical for the functioning of ecosystems (Metrick and Weitzman, 1994). How then does functional diversity differ from genetic diversity, and if so, why do these differences matter? Firstly, genes are after all just chemicals which have no value in and of themselves. Instead, genes have value in what they do - control the structure and function of life, instead of what they are. Measures of genetic distance may not capture the relative values of species such as the complex functional relationships that are embodied in ecosystems. Two species may be very similar with respect to genetic distance but they may perform very different functions within the ecosystem. One may be a keystone species which is vital to the wellbeing of the ecosystem whilst the other is 'functionally redundant'. Species diversity is relevant because some species appear to play a more important functional role than others. An individual who is evaluating a species in terms of its functional role would be more sensitive to a change in the ecosystem's productivity than would a person focussing on biodiversity in terms of its amenity value. A person who is assessing an ecosystem from the perspective of ecosystem function would be more likely to focus on key species and processes and may overlook the disappearance of a rare or charismatic or flagship species. There is greater emphasis on the biological integrity of the system than simply ensuring that all the biotic elements are present.

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[1] Biodiversity therefore represents the diversity of all life, being a characteristic property of nature rather than a resource. The term also has a broader meaning for the set of organisms themselves. For example, a biodiverse tropical rainforest, therefore, refers to the quality or range of diversity within it.

Functional diversity refers to the characteristics of ecosystems and includes ecosystem complexity at different levels of organisation such as trophic levels (Cousins, 1991). This approach uses trophic-level analysis to relate species diversity to functional ecosystem parameters such as food web structure or the transfer of energy, water, and chemicals between different trophic levels. Functional diversity can be interpreted as the number of species required for a given ecological process.[2]

Ecologists acknowledge that some species **have a greater ecological impact than one may expect from their abundance or biomass, and these have been referred to as *keystone species*** (Power et al, 1996). Some ecosystem studies indicate that only a small number of the numerous species found in ecosystems perform key functions or so-called keystone roles, while most species perform a perfunctory role (Holling, 1992). For example, beavers have been shown to have a profound impact upon streams, forests, and wetlands through dam construction. Many species may play keystone roles which involve interdependencies with other species (Daily et al, 1993). The elimination of any single component of an ecosystem could lead to an unanticipated unravelling of community structure and to local extinctions of some species.

The 'keystone role' of a species may also depend upon whether a number of other species can assume its functional role within the ecosystem (Schindler et al, 1989). Functional redundancy is known to occur if other species can perform similar roles (Hutchinson, 1961: Walker, 1992).

Most research has focussed on which species are important here and now. However, millions of species have not even been identified let alone evaluated for their potential value to humans. There are difficulties in predicting which species will be important in the future since the present functions which are being performed by a species may provide no clues as to its role when environmental conditions change (Main, 1982; Lovejoy, 1988).

However, studies on ecosystem function may reveal clues about the most sensitive components of food webs, as well as nutrient and energy flows. Research reveals that the most sensitive components of ecosystems are those in which the number of species which are performing a particular function is thought to be very small (Schindler, 1990).

Most ecologists recognise that some species play a more important functional role than others. But what does this imply in terms of the properties of an ecosystem? The next section provides a review of the relationship between biodiversity and the stability, resilience, and productivity of ecosystems.

2.1.2 Biodiversity, properties of ecosystems: stability, resilience, productivity

Holling (1973) refers to **stability as a characteristic of the individual populations of an ecosystem**. For example, stability is defined as the propensity of a population to return to some kind of equilibrium following a disturbance. The stability of ecosystems may be linked to their biodiversity and it has long been hypothesised that more diverse ecosystems are more stable. A clue as to why this may be the case is illustrated by a natural disturbance that deleted some species from the ecosystem. A diverse system may be little affected by the impact because other species with similar niches can perform functions which are similar to those of the missing species. Early advocates of this theory include MacArthur (1955) who postulated that a highly diverse ecosystem would change less upon the removal or addition of a species than would an ecosystem with fewer species. Elton (1958) also suggested that less diversity resulted in less ecological stability.

However, these theories were not without their critics. May (1973) challenged this argument and showed that a highly connected system (higher biodiversity) may be less stable than simpler ones and more vulnerable to disturbance because all of its components closely interact and are therefore subject to the effects of perturbations. A drought which eliminates key species in a complex ecosystem, for example, will have widespread repercussions for the animals that depend upon them.

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[2]

More recent work (Tilman, 1996) has shown that there exists an important distinction between the properties of a community and its individual species, so although diverse ecosystems are more stable than less diverse ecosystems the populations within them can have great variability. From this perspective, what matters is the stability of the community or ecosystem **not** their individual populations. There is some experimental evidence to support these assertions. Tilman and Downing (1994) have shown that an ecosystem with many species is more likely to be stable even though the populations of individual species may experience considerable fluctuations.

Resilience is a further factor that refers to the properties of the stability of a system. The traditional concept of resilience is a measure of the speed of return to a state of equilibrium after an ecosystem has been disturbed (Pimm, 1984; O'Neill et al*,* 1986). Alternative definitions have been proposed by Holling (1973). He describes resilience as the propensity of an ecosystem to retain its functional and organisational structure following a disturbance. Expressed another way, resilience is the amount of disturbance that can be absorbed before the system changes its structure by changing the variables that control how the system behaves (Holling, 1973). A characteristic feature of 'Holling resilience' then is that although the system parameters (net primary production, or system growth rates, or species composition) may change after disturbance, a resilient community will return quickly to equilibrium after disturbance is removed. A resilient ecosystem does not necessarily imply that all of its component populations are stable. Environmental perturbation may result in the extinction of an individual species without affecting ecosystem function or resilience. Holling (1973) distinguishes between stability as a property associated with individual populations of an ecosystem, and resilience as a property of an ecosystem. Early work by Holling (1973) has suggested that, in general, the resilience of an ecosystem is an increasing function of the diversity of that system. There is some empirical evidence to support this view. In a series of field experiments in drought-affected grasslands in Minnesota, Tilman (1996) has shown that species-poor plots were less productive in terms of biomass than species-rich plots (see Figure 1a). He also demonstrated that species-poor plots were more greatly harmed by drought (they were less resistant), took longer to return to pre-drought conditions (they were less resilient), and were less stable than species-rich plots. Tilman et al *(*1997) also demonstrated that plots with lower functional diversity had lower productivity in biomass terms than plots with high functional diversity (see Figure 4).

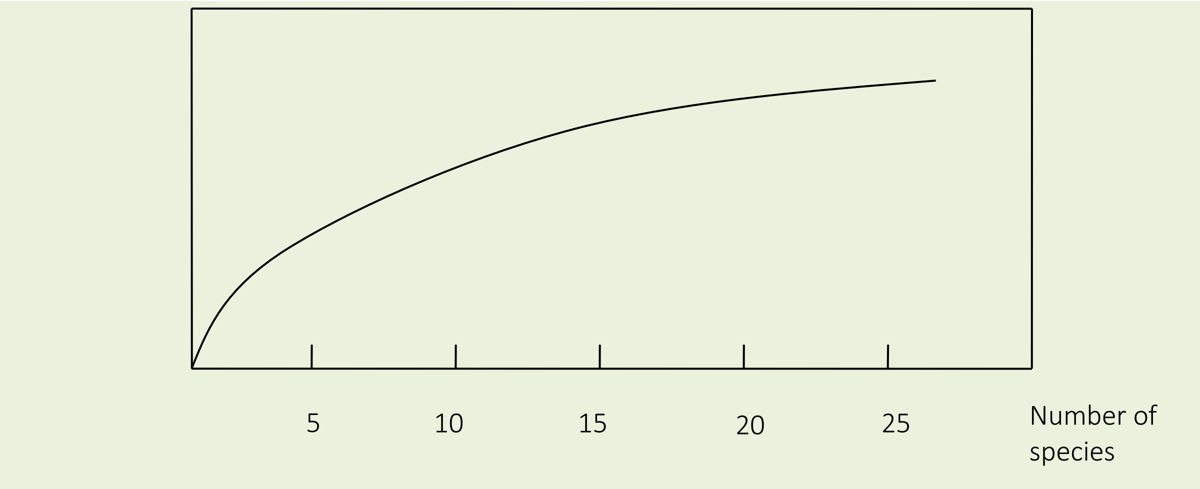

Figure 4. Conceptual relationship between the number of species and ecosystem functioning (Source: Sala et al*,* 1999; Vitousek and Hooper, 1993).

Bell (2012) uses a DPSIR approach in which he evaluates the notion of equilibrium and resilience with stakeholders in Malta.

Indicators:

- Alien species
- Endemic species
- Species changes
- Red data book species, species of conservation concern
- Species richness, biodiversity indices (Shannon, Berger-Parker, Margalef)
- Species abundance
- Population of key species
- Alpha diversity, beta diversity, gamma diversity, landscape diversity
- Functional diversity
- Keystone species, engineer species
- Ecosystem properties: stability, productivity, resilience
- Habitat fragmentation, distribution area of important or rare species
- Genetic diversity
- Animal breeds
- Crop varieties

Ecosystem health and biodiversity

A large body of literature now exists on ecosystem health, its links to biodiversity, and the DPSIR literature (Rapport et al, 1998). According to Rapport et al (1998) a healthy ecosystem is defined as being stable and sustainable, maintaining its organisation and autonomy over time and its resilience to stress. An assessment of the properties of ecosystems has been conducted by way of specific indicators (for vigour, resilience, and system organisation) for large-scale systems for coastal areas, forests, arctic ecosystems, and grasslands (Rapport et al, 1998; Yazvenko and Rapport, 1997).

2.1.3 Ecosystem disturbance and biodiversity

Considerable insight into the understanding of conservation biology has been gained through knowledge of the effects of human-induced disturbance on biodiversity (Wilson and Johns, 1982). There is a substantial literature which shows that **human-induced disturbance and habitat degradation can result in a decline in biodiversity and species extinction.** Highly intensive agricultural practices which reduce spatial habitat complexity, leading to a homogenisation of the landscape, may lead to biodiversity loss. The decline in most of Europe's SPECS (Species of European Conservation Concern) has been linked to land use and management changes, with agricultural intensification being cited as the most significant threat to bird populations (Tucker et al, 1994). Arable farming systems in parts of Europe are thought to have played a part in the decline of many species. For example, as a consequence of changing conditions in agricultural fields in Britain many bird species have undergone significant population declines. Fuller et al (1991) report that many British farmland birds have declined dramatically over the last three decades as agricultural land use has altered, hedgerows have declined, and farms have developed to form larger contiguous areas. A reduction in forest area due to agricultural expansion can also reduce species diversity. Studies of the avifauna of fragmented forests have shown that some species are absent or infrequent in very isolated sites and that smaller woodland size gives rise to less bird species diversity (Lynch and Whigham, 1984; Opdam et al, 1985; Ford, 1987; van Dorp and Opdam, 1987).

Persistently high levels of disturbance are also thought to affect ecosystem function, particularly where these eliminate important functional groups that affect ecosystem processes. Groups of grass species may be significant in maintaining the productivity of savannah ecosystems (Walker et al, 1981). Walker et al (1981) found that grasslands with persistent intensive grazing by settled peasant farmers had lower levels of productivity than moderate opportunistic grazing practices employed by nomadic pastoralists. In the former case, productive functional groups declined because herbivores showed a preference for the most palatable species, whilst in the latter case these preferred species were able to persist in the sward and adapt to change and instabilities caused by grazing and drought, thereby maintaining structural resilience.

MacArthur and Wilson (1967) suggest that some **disturbance can promote diversity because different species respond to disturbance in different ways.** They first characterised species as either *r* or *K* strategists which have evolved mechanisms to optimise resources in quite different environments. The former (the *r* strategist) refers to species which attempt to maximise growth in an unconstrained environment, reproduce quickly, disperse widely, and are of a smaller size and shorter lifespan. On the other hand, *K* strategists include species which optimise growth in a climax successional phase or a crowded environment, are highly adapted to stable equilibrium conditions, are less flexible and more vulnerable to change, are generally longer lived, and do not disperse as well. High levels of disturbance may lead to species-poor habitats since they favour the persistence of competitive, opportunistic *r* species which are better adapted to coping with disturbance (Miller, 1982). Conversely, undisturbed environments which do not undergo change may support less diversity because they favour the persistence of dominant *K* strategists. Linder et al (1997) examined the effects of fire history on structure of forest stands and plant diversity in Swedish forest reserves. They concluded that the reintroduction of fire represents an important means of disturbance that was necessary to promote the diversity of flora and fauna in the area. Continued fire suppression has changed successional patterns and altered stand structure. Late successional species such as spruce dominate due to the lack of fire, and pioneer species such as pine, silver birch and aspen are decreasing in number because they require fire disturbance to regenerate. This appears to accord with MacArthur and Wilson's 1967 theory in which undisturbed environments may therefore support less diversity because climax species are favoured. Linder et al (1997) recommend prescribed burning to ensure a relatively wide range of successional stages to promote biodiversity over the longer term.

Higher habitat diversity due to moderate disturbance can also be explained by niche relations and the manner in which species divide up limited resources for their survival (Schmida and Wilson, 1985). They may divide up the available space (for instance, by selecting different habitats) or energy resources (for instance, by adopting different diets). Some studies serve to demonstrate that moderate levels of human activity may enhance biodiversity by opening up new niches, providing new food or protection from predators, and by diversifying micro-habitats. For example, structural heterogeneity is thought to be important for bird species diversity, and vegetation indexes have been developed to quantify structural diversity particularly in relation to bird species (MacArthur and MacArthur, 1961; Willson, 1974; Erdelen, 1984). Several studies indicate that a decline in structural diversity (James and Wamer, 1982; Terborgh, 1985; Ratcliffe, 1993; Telleria and Carrascal, 1994) and floristic diversity (Lynch and Whigham, 1984) leads to less bird species diversity.

Recent developments on the functional complexity of ecosystems show that small disturbances may actually enhance ecosystem function and increase resilience. Holling et al (1986; 1994) suggest that some natural disturbances which are initiated by fire, wind, and herbivores are an inherent part of the internal dynamics of ecosystems and in many cases set the timing of successional cycles. These natural perturbations are part of ecosystem development and evolution and seem to be crucial for maintaining ecosystem resilience and integrity (Costanza, et al*,* 1993). In the absence of such shocks, the system will become highly connected and this will provoke even larger perturbations that are more destructive to the ecosystem because they reduce the ability of the system to survive similar shocks in the future (Scholes and Walker, 1993).

As described above, human-induced perturbation on managed ecosystems is a critical factor in maintaining biodiversity. However, the application of best scientific practice may not in itself be sufficient to achieve biodiversity conservation goals. This is because markets may fail to account for the value of biodiversity to society. It is essential therefore that land managers and policymakers are aware of the limitations and opportunities of the market with respect to an understanding of biodiversity.

In a coastal system in Portugal, Nobre (2009) used a DPSIR approach and estimated disturbance regime as the impact upon the ecosystem based on changes in bivalve production rates and changes in macroalgal growth and dissolved oxygen.

Indicators of disturbance in coastal systems:

- Algal blooms, bivalve production
- Eutrophication symptoms
- The number of red tides
- Overgrazing
- Wetland loss, loss of peatlands and bogs, drainage
- Habitat degradation and loss
- Hedgerows
- Forest cover
- Diversity of forest habitats
- Listed habitats
- Habitats of conservation status (SAC) (IBA) (NHA)
- Scheduled protected areas
- Abandoned agricultural land
- Marine conservation, protected areas
- Bathing water quality (the number of points obtained from faecal coliform readings)
- Gastroenteritis outbreaks
- Beach closures
- Leaked water
- The number of claims of storm damage
- Recycled water
- Enforcement actions by the EPA
- Pollution in ground water
- The quality of drinking water (levels of chloride and nitrates)

2.1.4 Biodiversity and productivity

An important question for this paper is how diversity affects farm productivity, profitability and profit efficiency. Increased diversity has been suggested by a number of studies to enhance productivity of grassland systems (Hulme et al*,* 1999; van Ruijven and Berendse, 2003; Bai et al, 2007; Bullock et al, 2007). Explanations for the productivity-diversity hypothesis include an enhanced complementary approach when it comes to resource use (Chavas and Di Falco, 2012), the more efficient use of water and improved nutrient recycling (Caldeira et al, 2001; Niklaus et al, 2006), and increased resistance to pests and diseases. A limited number of plant species makes them more vulnerable because pests easily spread through herbaceous grasses which have the same genetic base (Altieri and Lieberman, 1986). At the level of individual farms this can reduce costs which are related to agrochemical inputs and thereby improve the farm's income. Using long term data from an agricultural ecosystem Schläpfer et al (2002) found a reduced need for fertiliser and lower fertiliser costs amounting to between US$3.50 and US$6.00 an acre in diverse swards in comparison to species-poor grasslands.

Additionally, animal genetic diversity may influence the productivity of livestock systems (Sall et al, 1993; Fraser et al, 2014).[3] Mixed grazing using sheep and cattle is thought to improve livestock productivity and vegetation quality in comparison to sheep-only systems (Sall et al, 1993; Fraser et al, 2014).

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[3] Animal genetic diversity includes mixed species systems and the use of different animal breeds. Mixed species systems involve the use of two or more animal species to graze an area simultaneously.

2.2 Biodiversity and management: terrestrial examples

We now turn to some examples of how land managers from around the world manage systems which conserve biodiversity. In many managed landscapes good conservation practice succeeds because it is perceived to coincide with the interests of land managers whose support is vital for conservation initiatives. Such conservation practices may also have been developed to avoid the over-utilisation of the resource on which the human population depends. Consequently, most biodiversity exists in human-dominated ecosystems (Pimmental et al*,* 1992). In areas in which human populations have long been an integral part of the landscape and have had a good deal to do with its recent evolution, species may have adapted to 'managed' landscapes. The development of species-rich raised coastal dune and bog habitats in north-western Europe, known as machairs, is also thought to be strongly associated with agriculture and human activity, particularly fire and grazing (Mate, 1992; Edwards, et al*,* 2005). Floristic diversity is high in the grasslands of south-western Spain which are remarkable for maintaining some of the most species-rich grasslands outside of the tropics, with as many as sixty plant species per square metre having been recorded (Marañon, 1986). The Mediterranean basin acts as a transitional biogeographical location. It has been suggested that its flora, which comprises several different genetic elements, has been enriched by historical climatic fluctuations during the Quaternary period, by complexity of mountain relief and by altitudinal heterogeneity and historical human disturbance (Zohary, 1973; Whittaker, 1977; Marañon, 1986). Defoliation by domestic herbivores and the occurrence of frequent fires in association with periodic droughts are also thought to have promoted plant diversification, particularly of annual species and initiated adaptations to drought, fire, and grazing (Pignatti, 1978; Naveh, 1994).

Biodiversity may be important under highly variable environmental and socioeconomic conditions. In circumstances in which there are large differences in climatic, geological, and topographical gradients which contribute to a considerable degree of variation in productivity across regional landscapes we often find that land managers use biodiversity as a means of ensuring stability. The individual components that comprise this complex ecosystem including tree, herbaceous and shrub, and livestock components.

In Mediterranean grasslands, groups of perennial herbaceous species may be significant for maintaining productivity because they are able to utilise nutrients and moisture more effectively. These include *Agrostis castellana, Dactylis glomerata, Lolium perenne, and Phalaris aquatica* which were all found more frequently beneath tree canopies than in the open field (Joffre, et al*,* 1988). Joffre et al (1988) hypothesised that **differences in nitrogen utilisation occurred between annual and perennial species and that the efficiency of nitrogen utilisation by herbaceous species was affected by the tree canopy**. They report higher nitrogen mineralisation in grasslands with perennials when compared to annuals and greater nitrogen mineralisation below the tree canopy. Short grass (concentrate) grazers benefit from the modification of sward structure brought about by long grass (bulk) grazers; for example, sheep generally perform better when grazed in mixed systems than when grazed alone (Nolan, and Connolly, 1977). Moreover, in savannas (Walker et al*,* 1981) and *Agrostis-festuca* grassland in Britain (Hulme et al*,* 1999), groups of grass species are important in maintaining the system's productivity.

In livestock systems there are a number of factors that are considered to affect biodiversity. It is generally thought that grazing has a positive effect on biodiversity (Fraser, et al, 2014). Defoliation by domestic herbivores maintains an open landscape, may also promote plant diversification (Marriot et al, 2009; Reinhammar, 1995), and pastoral systems are acknowledged as a means of maintaining and restoring open managed landscapes, preventing natural succession to woodland, and a decline in herbaceous diversity (DeGabriel et al, 2011; Mavromihalis et al, 2013). Species-rich Machair habitats are thought to be strongly associated with agriculture, particularly grazing (Mate, 1992; Edwards, et al*,* 2005). However, the level of disturbance appears to be important and stocking rates are considered to influence biodiversity and productivity (Dumont et al, 2009; Marriot et al, 2009;

Mavromihalis et al, 2013). Hulme et al (1999) found that high levels of grazing resulted in the replacement of productive *Agrostis-festuca* grassland by less desirable species such as *Nardus stricta* and *Molinia caerulea.*

2.2 Biodiversity, land abandonment, and forests

In western Europe, abandonment appears to be driven mainly by rural depopulation, industrialisation, urbanisation, market-orientation, and off-farm employment (MacDonald et al, 2000; Gellrich et al, 2007). Macdonald, et al (2000) suggest that land abandonment is related to rural depopulation, something to which isolated and poorer regions are more vulnerable. A number of studies have highlighted the fact that land is being abandoned due to a lack of successors (Visser et al, 2007), an aging workforce (Glauben et al, 2004), and the exodus of a younger generation of better educated people away from farming. Research also suggests that land may be abandoned as a result of a reduction in farming income (MacDonald et al, 2000). Gellrich et al (2007) finds that land abandonment is more prevalent in places in which part-time farming is common. Other work points to the fact that local demand and the supply of labour and land have affected land abandonment decisions (Gellrich et al, 2007).

The move towards part-time farming and the abandonment of certain agronomic practices may have important implications for biodiversity loss and environmental degradation. **The abandonment of low-intensity traditional farming systems is now seen as a threat to High Nature Value (HNV) farming** because agronomic practices which are thought to support biodiversity are being abandoned (Bokdam & Gleichman, 2000). These practices include the abandonment of upland hay meadows (Jefferson, 2005) and hedgerows (Keena, 1998), stonewall and field margin maintenance, and a tendency to replace mixed grazing with specialised systems. Walther (1986) points out that many regions in which land abandonment is known to occur also coincide with economically marginal areas that are known for their biodiversity and amenity values.

Haymaking in Europe is a traditional agricultural practice which sees grass or legumes being cut, dried, and stored for use as animal fodder. Upland hay meadows have significant aesthetic and recreational value. However, many of the farm practices which are associated with haymaking are being abandoned and upland hay meadows are becoming increasingly scarce as a result of a cessation of this activity (Jefferson, 2005; Critchley et al, 2007). Smith et al (2000) and Bignal & McCracken (2000) showed that as livestock production moves away from haymaking, there is a greater reduction in plant species diversity.

Mixed livestock systems have had a profound effect on landscape and biodiversity and are thought to be beneficial for biodiversity (Sanderson et al, 2009).

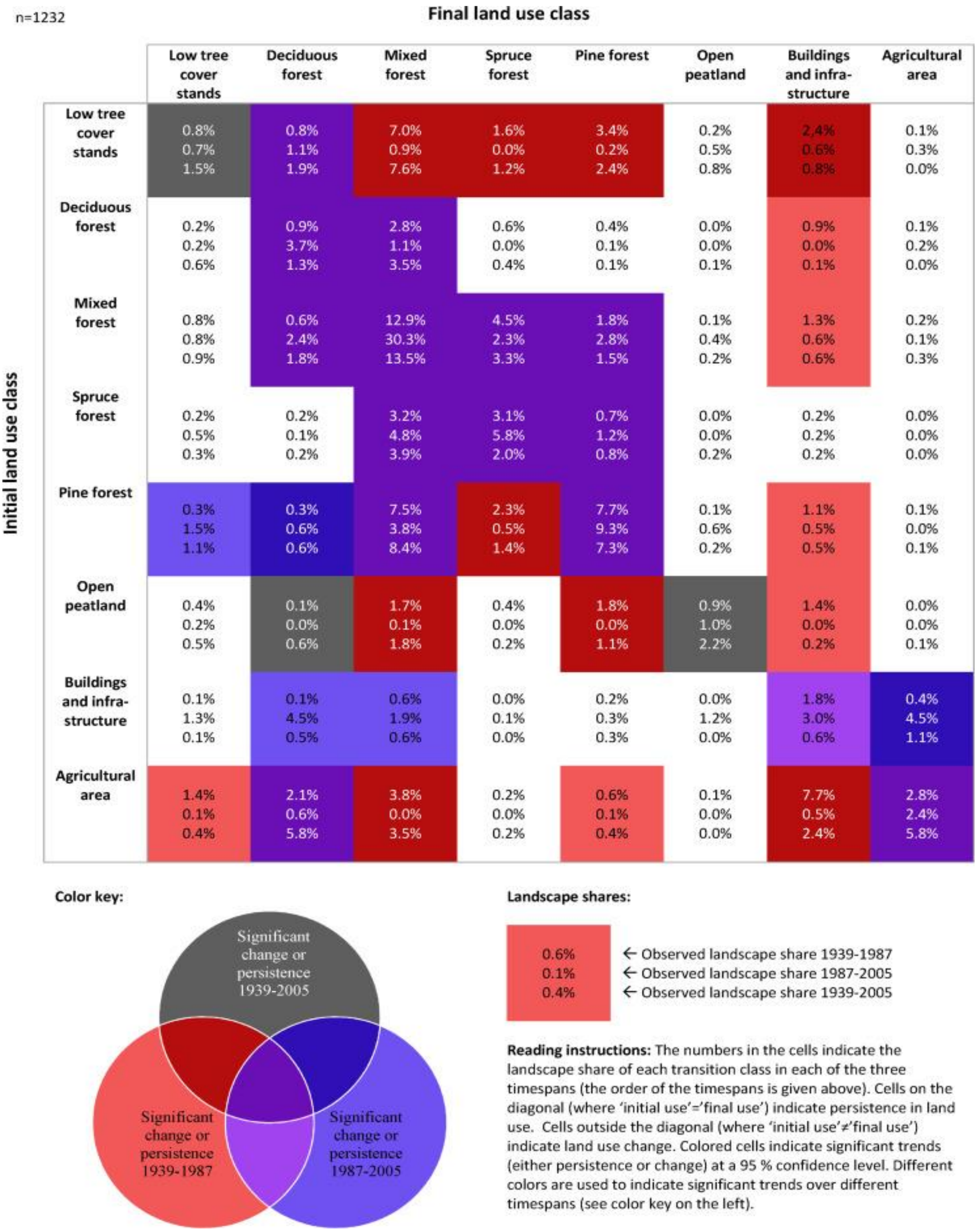
In post-Soviet Eastern Europe institutional changes led to widespread land abandonment after the collapse of the Soviet Union during the decade after transition but one study reveals big differences in the rates of abandonment across countries (Prischepov, et al, 2012). Where institutions governing land, use changed little (Belarus) or where institutional adaptation was rapid and adaptable (Poland) abandonment was low. In contrast countries where the establishment of new agricultural institutions and regulations involved long delays (Latvia, Lithuania and Russia) abandonment was high. After more than two decades many abandoned agricultural lands have reverted to forest. This has widespread repercussions on biodiversity as well as the carbon balance but also may increase the environmental and economic costs of re-cultivating agricultural land. The paper suggests that the opportunities for conservation and for increasing agricultural production on abandoned lands may be significant.

Although the study by Rautiainen et al (2016) does not cover the Baltic States it has been included because it provides a useful case study example of landscape change in the context of land abandonment and land use change. It depicts landscape shares and the composition of forests over time and reflects on forest type, biomass and forest carbon. The Isthmus of Karelia also provides an interesting case study contrasting the consequences of agricultural land abandonment on natural succession versus relatively intensive forest and agricultural management (Finland). Rautiainen et al (2016) reveal that agricultural land diminished and gave way to forest. Prior to the collapse of the Soviet Union in 1991 logging was banned but resumed thereafter. Forest development followed a natural successional sequence and some deciduous forests were transformed to spruce forests and mixed forests.

Clear cuts became rare as did stands of saplings and old growth forest increased as did the forest biomass carbon stock. This is contrasted with Finland where land use remained stable, private forest owners manage the land and increasing forest biomass is achieved by intensifying agriculture and forestry not by land abandonment. The authors suggest that species associated with old growth forest may have benefited but that not all species are supported by old growth and that the overall long-term effects of biodiversity are difficult to gauge. The figure below can provide some examples of how they report changes from initial to final land use classes and the transitions involved.

We should also consider what can happen to biodiversity when land and management activities are abandoned. Land abandonment can have important implications for biodiversity. Londo (1990) reports that, in the absence of management, semi-natural grassland communities revert by processes of natural succession to natural woodland and forest and the diversity of herbaceous species falls. Many traditional extensive farming practices have been shown to maintain plant and animal diversity (González Bernáldez, 1991; Naveh, 1994), and where these activities cease, susceptibility to disturbances, especially fire, can be increased. Fire in turn can have a negative effect on biodiversity (Faraco et al*,* 1993). Landscape homogenisation can also result from the abandonment of agricultural/pastoral land (Fernandez-Alés et al*,* 1992). Without human management diverse plant communities in the Mediterranean basin, for example, become overgrown, and displaced by relatively few, shrubby unproductive species. Livestock may play a positive role in influencing the system.

Bokdam and Gleichman (2000) have suggested that abandonment is a major threat to traditional pastoral landscapes and their wildlife in Europe. They report that increased labour costs have undermined traditional herding systems, which are being replaced by free-ranging grazing systems leading to a decline in species-rich open heathland.



n=1232

Final land use class

Initial land use class

	Low tree cover stands	Deciduous forest	Mixed forest	Spruce forest	Pine forest	Open peatland	Buildings and infra-structure	Agricultural area
Low tree cover stands	0.8% 0.7% 1.5%	0.8% 1.1% 1.9%	7.0% 0.9% 7.6%	1.6% 0.0% 1.2%	3.4% 0.2% 2.4%	0.2% 0.5% 0.8%	2,4% 0.6% 0.8%	0.1% 0.3% 0.0%
Deciduous forest	0.2% 0.2% 0.6%	0.9% 3.7% 1.3%	2.8% 1.1% 3.5%	0.6% 0.0% 0.4%	0.4% 0.1% 0.1%	0.0% 0.0% 0.1%	0.9% 0.0% 0.1%	0.1% 0.2% 0.0%
Mixed forest	0.8% 0.8% 0.9%	0.6% 2.4% 1.8%	12.9% 30.3% 13.5%	4.5% 2.3% 3.3%	1.8% 2.8% 1.5%	0.1% 0.4% 0.2%	1.3% 0.6% 0.6%	0.2% 0.1% 0.3%
Spruce forest	0.2% 0.5% 0.3%	0.2% 0.1% 0.2%	3.2% 4.8% 3.9%	3.1% 5.8% 2.0%	0.7% 1.2% 0.8%	0.0% 0.0% 0.2%	0.2% 0.2% 0.2%	0.0% 0.0% 0.0%
Pine forest	0.3% 1.5% 1.1%	0.3% 0.6% 0.6%	7.5% 3.8% 8.4%	2.3% 0.5% 1.4%	7.7% 9.3% 7.3%	0.1% 0.6% 0.2%	1.1% 0.5% 0.5%	0.1% 0.0% 0.1%
Open peatland	0.4% 0.2% 0.5%	0.1% 0.0% 0.6%	1.7% 0.1% 1.8%	0.4% 0.0% 0.2%	1.8% 0.0% 1.1%	0.9% 1.0% 2.2%	1.4% 0.0% 0.2%	0.0% 0.0% 0.1%
Buildings and infra-structure	0.1% 1.3% 0.1%	0.1% 4.5% 0.5%	0.6% 1.9% 0.6%	0.0% 0.1% 0.0%	0.2% 0.3% 0.3%	0.0% 1.2% 0.0%	1.8% 3.0% 0.6%	0.4% 4.5% 1.1%
Agricultural area	1.4% 0.1% 0.4%	2.1% 0.6% 5.8%	3.8% 0.0% 3.5%	0.2% 0.0% 0.2%	0.6% 0.1% 0.4%	0.1% 0.0% 0.0%	7.7% 0.5% 2.4%	2.8% 2.4% 5.8%

Color key:

Landscape shares:

0.6% ← Observed landscape share 1939-1987
0.1% ← Observed landscape share 1987-2005
0.4% ← Observed landscape share 1939-2005

Reading instructions: The numbers in the cells indicate the landscape share of each transition class in each of the three timespans (the order of the timespans is given above). Cells on the diagonal (where 'initial use'='final use') indicate persistence in land use. Cells outside the diagonal (where 'initial use'≠'final use') indicate land use change. Colored cells indicate significant trends (either persistence or change) at a 95 % confidence level. Different colors are used to indicate significant trends over different timespans (see color key on the left).

Figure 5. Land use transitions in the Isthmus over three periods: 1939-1987; 1987-2005; 1939-2005

2.3 Land fragmentation

Land fragmentation, where a single farm comprises numerous individual parcels of land is a common agrarian feature of many transition economies including the Baltic States (Blarel et al, 1992; Dijk, 2002; Sabastes-Wheeler, 2002; Todorova and Lulcheva, 2005; Niroula and Thapa, 2007; Jürgenson, 2016). During the 1990s, Central and Eastern European countries conducted land reforms. The main elements of reform were land restitution, privatisation and dissolution of large centrally run agricultural enterprises (Lerman, 1999; Davidova, 1997; Dijk, 2002: Kopeva, 2001; Jürgenson, 2016). Land fragmentation is often considered to be an obstacle for improving agricultural productivity and land abandonment (Theesfeld, 2005; Dirimanova, 2006; Jürgenson, 2016). It is thought to impede growth and prevent efficiency gains in the agricultural sector and many governments including the Baltic States have sought to promote a more rational spatial allocation of land and formulated policies aimed at encouraging land consolidation (Blarel et al, 1992; Jürgenson, 2016).

Although it has been argued that land fragmentation may be detrimental to both farmers and the economy, there are a number of reasons why farmers may benefit from land fragmentation. Land fragmentation provides a means of exploiting land parcels of differing quality. This facilitates crop diversification, spreads labour requirements, reduces production and price risks and better matches soil types with necessary food crops (Bently, 1987; Blarel et al, 1992). Land fragmentation is thought to promote crop and agricultural diversity[4] (Bellon and Taylor, 1993; Hung, 2006). The Baltic States are prime examples of such a situation. After the land reform in 1991, agricultural production characteristics have changed dramatically. The mode of agricultural production has gone from large specialised production units to highly fragmented private farms (Buckwell and Davidova, 1993; Kopeva et al, 1994).

In Eastern Europe an aim of land reform policy by the state has been to restore land to those who owned it in the year 1940s (Kopeva et al, 1994; Jürgenson, 2016). An aspect of post-socialist transformation under the Washington Consensus has been the privatisation of state owned and/or co-operative farms (Van Dijk, 2007). On the one hand this has meant restoring land to those who owned it in 1940s (or to their heirs), who not only have very small and scattered holdings, but who often live in an urban setting at considerable distances from the claimed land with no tradition of farming. On the other hand, farming has been altered by the liquidation of collective farms and the emergence of privately managed farms (Kopeva et al, 1994).

Jürgenson (2016) provides a very good overview in the case of Estonia. The pattern of land reform in Estonia following the Estonian Land Reform Act is shown in the figure below. The paper suggests that 93% of the territory has been reformed (37% in state ownership, 33% transferred through restitution and 22% privatised). However, the study reveals that the process has actually led to more fragmentation in land tenure compared to 1940. One of the reasons for this is that several eligible individuals had the right to the same land parcel and as a consequence the land became further subdivided. In two case study areas the author notes an increase in the number of plots and a reduction in their size. Jürgenson (2016) finds that the current Land Readjustment act is outdated and requires reform but that there is a lack of political will to guide this process. Very little is said in the paper about the impact upon biodiversity or indeed the environment in general of the land reform process in Estonia.

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[4] Agricultural biodiversity (or agrobiodiversity) is defined as a component of biodiversity, referring to all diversity within and amongst species found in crop and domesticated livestock systems, including wild relatives, interacting species of pollinators, pests, parasites, and other organisms (Qualset et al, 1995; Wood and Lenné, 1999).

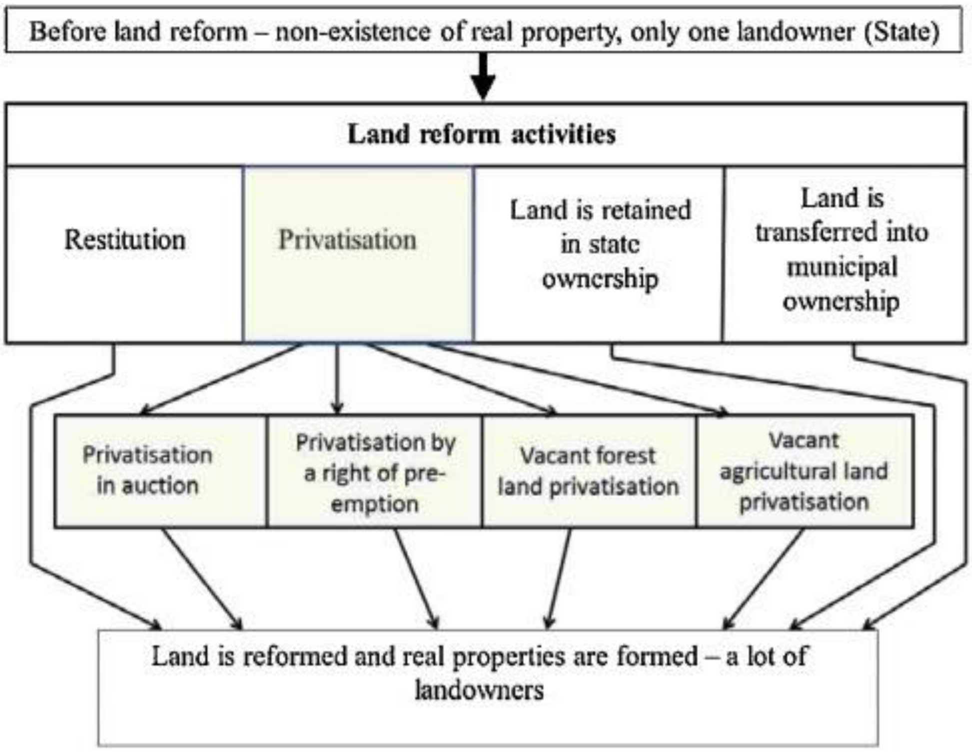

Figure 6. Land reform in Estonia after 1990 following the Estonian Land Reform Act (Source: Jürgenson, 2016)

A number of notable changes to the cropping system took effect during the collectivisation process (post 1945) whereby all village land was subsumed within cooperatives which eventually became (in 1971) large Agro-industrial Complexes (AIC). Traditional crops were replaced by high yielding varieties; decision-making became progressively more centralised and each village specialised in a narrow range of crops. Seed production was confined to specialist farms, supply was centrally controlled, and only laboratory tested varieties could be used for production. Notably, decisions over the spatial location and precise mix of crops were no longer made by the household.

After the reforms of the early 1990s, the system of seed control collapsed. An unregulated wholesale market for seed quickly developed but these were often expensive and the quality is not always consistent.

With respect to land fragmentation there is a trade-off been increased productivity and biodiversity (crop diversity based on an index that accounts for both the number of crop species and crop varieties). On the one hand farms with fragmented land are less profitable. This is probably due to inherent inefficiencies arising from the spatial distribution of land. Fragmented fields are problematic to cultivate, it is difficult to use machines; space is lost along field boundaries and there are problems with development and management of irrigation systems (Bently, 1987, Jürgenson, 2016).

On the other hand, there is an indirect positive effect. More fragmentation increases farm crop diversification, and this, in turn, has a positive effect on profitability. This effect has not been studied in any detail for the Baltic States but a study from one CEEC country revealed the reduction in revenues due to higher levels of land fragmentation, which can therefore be buffered by the positive role that fragmentation has on diversity (Di Falco et al, 2009).

As far as the author is aware, there are no DPSIR studies that focus on land fragmentation and its social and environmental effects for the Baltic States. But clearly the social and environmental impact of land reform for these countries is significant. According to Jürgenson (2016) the reform process has stalled and a response is urgently needed. Carefully crafting such a response is outside the scope of this review, but clearly this is a topic that merits further enquiry across the region. A list of possible indicators which are used to assess and monitor land fragmentation is provided in the table below.

Table 6. Indicators for assessing and monitoring land fragmentation.

Index
1. Land plots before 1940 2. Land plots in 2017 3. Unreformed land plots 4. Number of plots 5. Total land area 6. Average plot size 7. Restitution (%) 8. Privatisation (%), vacant forest land privatisation, vacant agricultural land privatisation 9. Retained in state ownership 10. Transferred to municipality 11. Intensity of input use 12. Fragmentation 13. Land quality 14. Distance (to nearest houses from plots) 15. Number of land owners 16. Year of reform 17. Experience (of farmer/forester/land manager) 18. Farm crop diversity (Margaelf, Shannon, see Magurran, 1988) 19. Forest biodiversity 20. Labour

3 BIODIVERSITY: ECONOMIC CONSIDERATIONS

3.1 Biodiversity and external costs

According to the economic theory of general equilibrium, the search for opportunities for increased private returns can ensure that resources are allocated to the use which returns the highest value possible, so that economic efficiency is achieved. This result depends upon a number of conditions. If these conditions are fully met, the use of biodiversity which is motivated by private profit need not be a cause for concern. **However, biodiversity management, in common with most environmental goods, have characteristics that ensure that the necessary conditions will never be fully met in practice.** In general terms, this failure implies that any resulting allocation of resources is likely to be economically inefficient, meaning that it would be possible to reallocate resources in such a way as to make at least one member of society 'better off'.

Market failure occurs when private decisions based on a set of prices, or lack of them, do not generate an efficient allocation of resources (Hanley, et al*,* 1997). With respect to biodiversity the concern is that market prices are not reliable indicators of social cost. Social cost refers to opportunities which have been forgone by society when committing resources in some way (Coase, 1960), and social cost in this study is taken to mean the true value that society as a whole places on natural resources or biodiversity. Private cost, on the other hand, refers simply to the financial cost faced by the private individual or business which is undertaking the management of the environment, at current and expected market prices.

This divergence between private and social cost occurs because managed biodiverse areas generate benefits to society in addition to those that are transacted in the market system: external benefits. An absence of such external effects is one of the necessary conditions for market efficiency, as is the absence of public goods referred to in the previous section. Typically, the reason these benefits remain external to the market system is that they have the characteristics of public goods; in particular they are indivisible and perhaps also non-excludable, making their exchange in markets unlikely (see **Table 2**). The discussion above relates mainly to what may be called the 'complete set of market' conditions (Common, 1995).

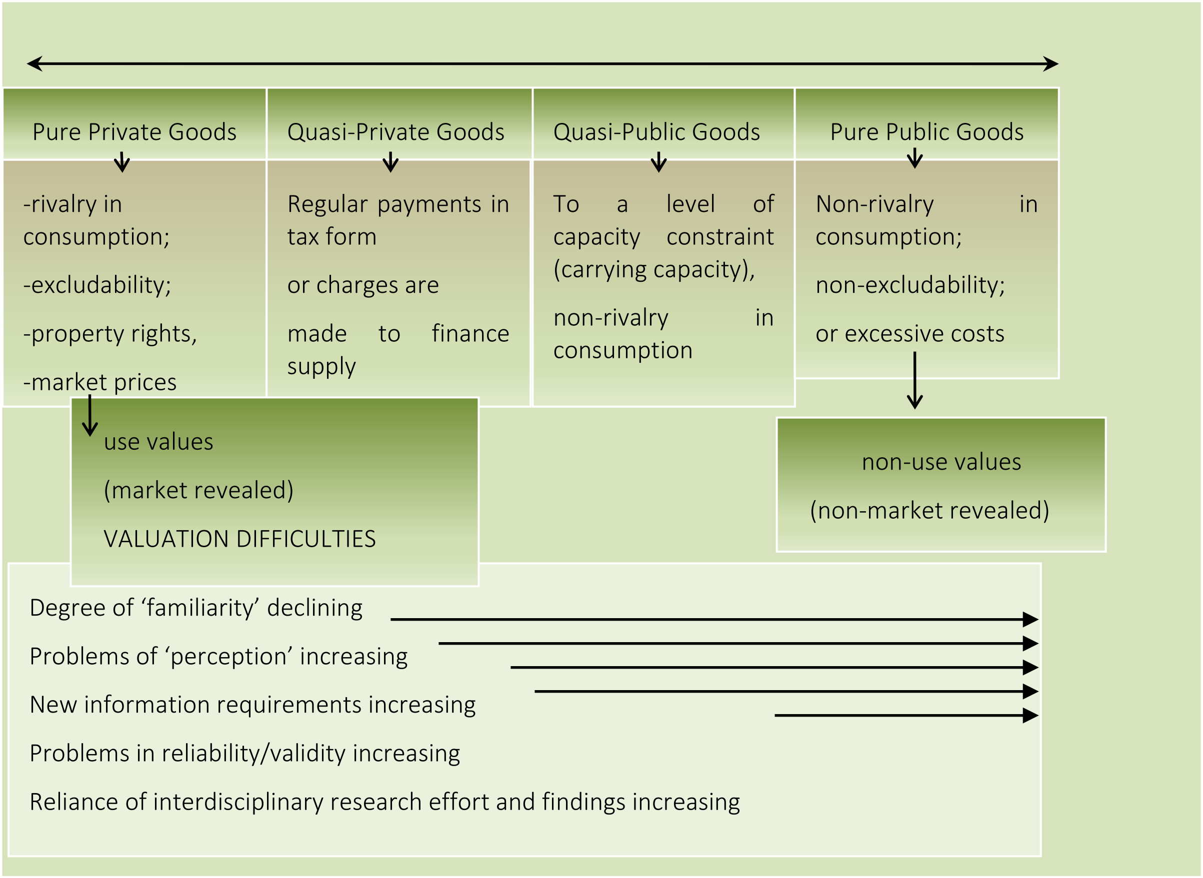

Figure 7. The characteristics of public and private goods. Source: Turner [1993].

Other issues which give rise to inefficiency include the lack of well-defined property rights over biodiversity. Many ecosystems, especially global public goods and commons such as the atmosphere, the high seas, and many Exclusive Economic Zones (EEZs) lack strong property rights, involve difficulties related to excluding users, and entail high monitoring and enforcement costs (Long and Grehan, 2002). The lack of strong and complete property rights is a major cause of external costs and threats to these ecosystems and their biodiversity. Markets may fail to account for the value of their biodiversity to society due in part to the public goods nature of the resources,[5] and the failure of markets to capture any non-market benefits which may be associated with these ecosystems.

Because managed landscapes provide high levels of *un-costed* public benefits, in terms of wildlife and landscape quality, private agents will have no incentive to take account of these benefits in decisions regarding land use. The main point that is frequently made by environmental economists who are working on valuations with regards to market and policy failure is that private resource users do not attribute sufficient weight to biodiversity. Valuation, it is argued, aims to redress this imbalance and sets out to determine what weight should be given to biodiversity in the interests of society as a whole.

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[5] Indeed, fishermen may, due to the mismanagement of fish resources, not even have the incentive to incorporate the ecosystem values of the CWC for the relevant fishery as a whole. For example, since fisheries are often characterised by the so called 'race for fish' where a single fisherman has no incentive to take into account the effect of their own harvesting upon other fishermen, or even upon their own well-being in the future, those fisheries which benefit from the presence of the CWC will face the same fate (Clark, 2006). Hence not only do fishermen not take into account the benefits of the CWC for people outside the fishery, the fisheries' own benefits are not taken into account either.

Market failure also implies that resource users do not pay the full (external[6]) costs of any damage to ecosystems which occur as a result of their actions.[7] External factors occur in different guises and affect a given ecosystem in various ways. A fishing example is given below, as well as an illustration which uses recreational external factors. For example, how do Type 2 conflicts result in the over-harvesting of fish species and impose higher external costs on other fishermen? These effects include possible damage to marine habitat, a reduction in fish stocks, and increased efforts and costs involved in the fish harvest. Type 1 conflicts, whereby fishers damage coral habitat, involves an asymmetrical externality arising in production.

The aforementioned social costs are borne by consumers, other stakeholders, the nation state, and future generations. Because many of the goods and services, including the genetic, functional, and existence value which are associated with biodiversity, are *un-costed*, then resource users have little incentive to take account of these benefits in decision-making. There are three hypothetical equilibria for an agro-ecosystem in relation to economic activity (Q) such as crop production. Under well-defined property rights, when external costs are not taken into account, private users will operate at Q^{Π} where marginal costs (MC) equal marginal benefits (MB). Under open access conditions, private users will operate at Q^{OA} where MB = AC (Average Costs). In cases in which intensive agricultural production involves biodiversity loss this results in an externality. In this case internalising this external effect to achieve a social optimum (Q^{S}) and an efficient allocation of resources results in an intensity effort level that equates marginal external costs (MEC) with MB at a lower level of production intensity.

Economic theory suggests that internalising external costs will result in a level of economic activity that is socially optimal. This requires the external costs which are associated with production or development to be measured in some way in order to derive an MEC curve. Valuation can be used to do this and can assist policymakers in developing regulatory measures to correct for Pareto-relevant external costs.

3.2 Valuing ecosystems and their biodiversity

3.2.1 Methodologies for valuing ecosystems and biodiversity

Assigning monetary values to non-market goods can potentially avoid any undervaluation of biodiversity (Daily, 1997; Loomis et al, 2000; Chee, 2004). The concept of 'Total Economic Value' (TEV) can be used to describe the components of value which are associated with an ecosystem (see **Figure 8**). From an economic perspective, the goods and services which are generated by an ecosystem comprise *use* and *non-use* values.

The literature indicates that a variety of methods have been employed to estimate biodiversity values in managed landscapes and protected areas. Studies have focused on their use and non-use values. These values are based on an individual's willingness to pay (WTP) or their willingness to accept (WTA) compensation. **Gross willingness to pay** may include the cost of travel, the purchase of equipment to participate in the recreation activity, actual fees which are associated with the activity, and any consumer surplus. The concept of 'Total Economic Value' (TEV) has been used to describe the components of value as shown in Figure 8. Use values which are associated with managed landscapes

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[6] An absence of such external effects is one of the necessary conditions for market efficiency. Typically, the reason that these benefits remain external to the market system is that they have the characteristics of public goods; in particular they are indivisible and perhaps also non-excludable, making their exchange in markets unlikely.

[7] If an individual can benefit from any benefit, irrespective of their contribution towards that benefit, or can avoid paying some of the costs which are inflicted upon others, there is a tendency in both cases to maximise short-term self-interest and to free-ride.

refer to the actual and/or planned use of a service by an individual, and they include recreational activities such as bird watching or hunting. Use values also include the following: option value, ie. the value of the option to guarantee the use of the service by the individual in the future (Weisbrod, 1964); the quasi option value, ie. the value of future information which is protected by preserving the resource now, given the expectation of future growth in knowledge relevant to the implications of development (Arrow and Fisher, 1974; Perman, et al, 2003).

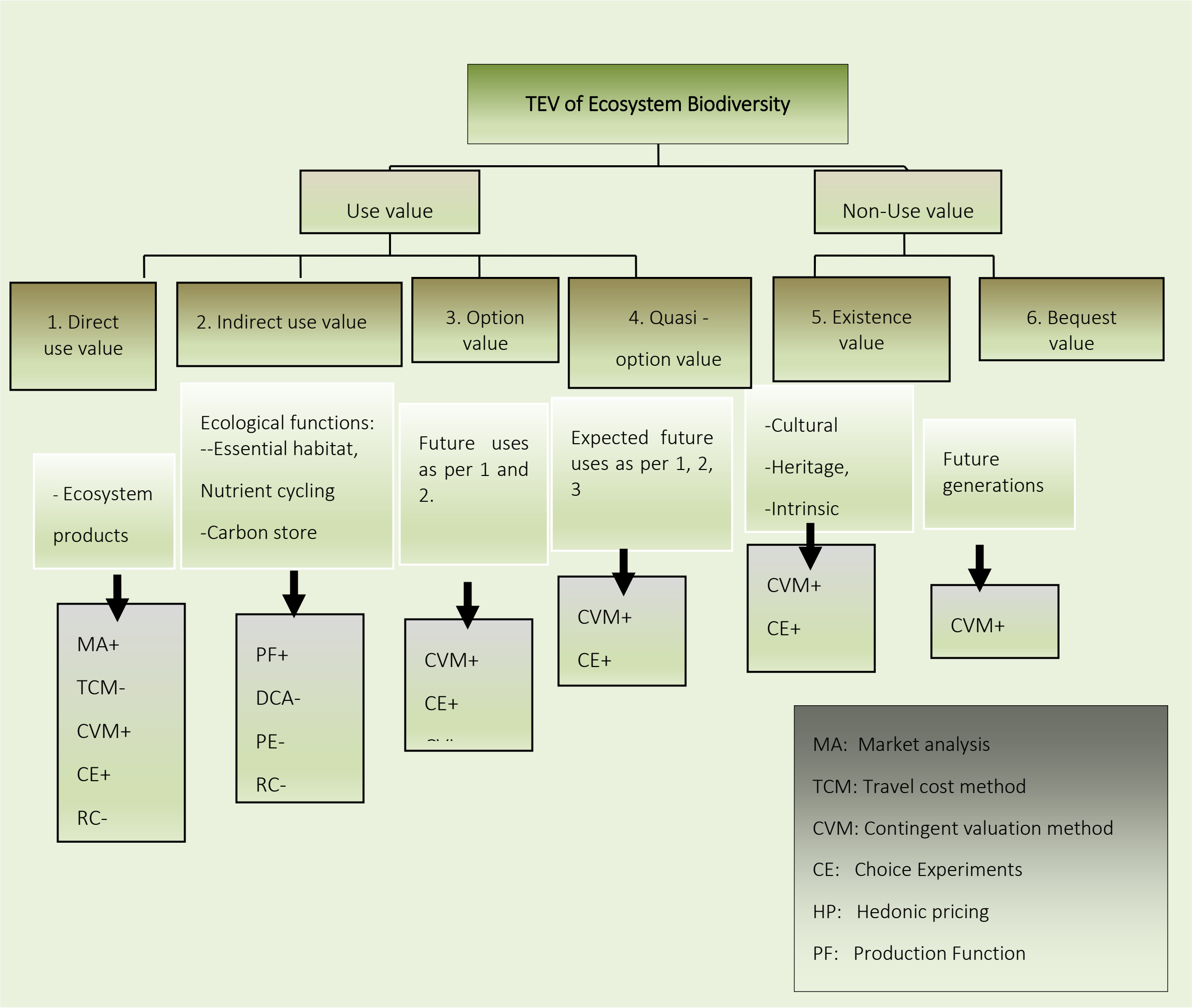

Figure 8. The components of Total Economic Value associated with ecosystem biodiversity. The + indicates high relevance and - indicates low relevance in the use of the valuation technique for the goods or service in question. Source: adapted from Barbier et al (1994).

The valuation of any aspect of biodiversity which is not traded is more problematic. In the absence of market prices for biodiversity values, certain non-market valuation techniques need to be used. Two broad approaches are relevant to the valuation of CWC ecosystems: the surrogate market valuation (revealed preference), and the stated preference methods. The surrogate market valuation methods which are of relevance include the **travel cost method (TCM)**, and the **production function (PF) approach**. Services which are provided by ecosystems and their biodiversity include non-market use values such as scientific, aesthetic, and educational information, and the potential recreational viewing of biodiversity and its amenity aspects.

The contingent valuation method (CVM) or choice experiments (CE) could be employed to estimate the aesthetic and scientific value of biodiversity. The contingent valuation method (CVM) and choice experiments are tried and tested techniques which are capable of measuring the non-use values associated with biodiversity. These methods can be used to forecast preferences for biodiversity or to determine whether consumers are willing to pay a price premium for ecosystems that are managed using sustainable production methods. For example, a number of studies indicate that consumers prefer organic or eco-labelled products, as long as price premiums for the eco-labelled products are not excessive (Johnston et al, 2001). The use of high nature value farming or fair-trade coffee production represents other such examples. Less intensive or more socially responsible production in this case is considered to have less impact but still allows commercial production to continue, thereby protecting commercial industries whilst still ensuring that social costs are internalised.

TCM has been widely used for valuing the non-market benefits associated with forest parks, nature reserves, hiking areas, visiting marine parks, protected areas, and dive sites (Soderqvist et al, 2005). The application of TCM typically involves users travelling to the site.

Ecosystems and their biodiversity may influence supply side production. Indirect use values include nutrient cycling, carbon sequestration, and enhanced productivity due to biodiversity and its habitat. The production function approach represents an important means of quantifying functional values that are associated with biodiversity. The method would link biodiversity crop or livestock production, or to fisheries, identifying to what degree profits from commercial species are affected by the presence or absence of biodiversity. Given the identification of such a link, this could then be modelled in order to ascertain the losses involved when this link is not included in management or conservation decisions. The method can be used to take account of how changes in the level of biodiversity itself, or indeed the habitat area or its quality, affect production (Barbier, 2000; Knowler, 2002). A number of studies on other ecosystems have been conducted using this approach to determine the indirect value of mangroves and marshlands as inputs in fishery production (Ellis and Fisher, 1987; Barbier and Strand, 1998; Barbier, 2000;). Barbier and Strand (1998) established a value for one of the non-market functions of mangroves by exploring the relationship between mangroves and shrimp production in Campeche, Mexico.

Although there are problems associated with double counting (Aylward and Barbier, 1992), the production function approach may be highly relevant to DPSIR and biodiversity studies where the properties of the ecosystem (productivity, stability, resilience) are affected in some way.

More recently, empirical studies on marine values have placed an emphasis on non-use value (see the right-hand side of Figure 4). These refer to situations in which an individual knows that a biological resource exists and will continue to exist, independent of any actual or prospective use by the individual and where that individual would feel a sense of ‘loss’ if the resource were to disappear (Ledoux et al, 2001). Non-use values include the following: *bequest value*: the value of ensuring that the resource remains intact for one’s future heirs (Perman et al, 2003); *existence value:* the value that arises from ensuring the survival of a resource (Perman et al, 2003).

The main beneficiaries of existence values are probably the general public. A number of international environmental organisations and NGOs have been particularly vocal in pressing for CWC conservation; for example the WWF, Oceana, and UNEP. Although this has not been quantified, it provides some evidence of the importance of non-use values in relation to CWC. It is clear from NGO involvement that deep-water corals have both existence and bequest values.

3.2.1 Indirect use values and ecosystem function

As an example, the stability of a managed ecosystem constitutes an indirect use value and represents an important function to land managers. As seen above, biodiversity may mitigate large inter-annual variation in productivity (McNaughton, 1985; Walker, 1988). For instance, the economic value of a change in diversity can be evaluated from the change in livestock liveweight gain which is associated with a decline in forage biomass as a result of a decline in grassland diversity. Plot studies indicate that intercropping can reduce the probability of absolute crop failure and that crop diversification increases crop income stability (Walker et al*,* 1983). Therefore, the greater the diversity between and/or within species and functional groups, the greater is the tolerance to pests. This is because pests easily spread through crops which have the same genetic base (Sumner, 1981; Altieri and Lieberman, 1986). Crop diversity may enhance farm productivity, stabilise farm income, and reduce the risk of outright crop failure (Long et al*,* 2000). Di Falco and Perrings (2003; 2005) found cereal diversity to be positively correlated with yields and negatively correlated with revenue variability in two studies in southern Italy. Similarly, the importance of bird species used as a biological control agent can be captured from increased timber sales associated with insect pest reduction. Takekawa and Garton (1984) used the substitution method to determine the value of a bird species, the evening grosbeak (*Hesperiphona vespertina*) in controlling spruce budworm populations affecting stands of Douglas fir (*Pseudotsuga menziesii*) in Washington. They substituted the cost of insecticides to produce the same mortality rates that birds cause and established that it would cost at least $1,820 per square km per year over a one hundred-year rotation.

Di Falco et al (2007) revealed that wheat diversity increases farm productivity in the highlands of Ethiopia. A number of studies in India (Manjunatha et al, 2013), South Africa (Nel and Loubser, 2004), and Bangladesh (Rahman and Rahman, 2009) all demonstrate that crop diversification significantly improves profit efficiency. In Australian rangelands, Moloney et al (2011) employ a portfolio optimisation technique to demonstrate that mixed grazing improves profitability, reduces risk, and increases biodiversity. Poshiwa et al (2013) also use portfolio theory to show that including the management of native herbivores reduces rainfall-induced fluctuations in a household's income, and can be used as a hedge asset to offset risk without compromising income. Martin and Magne (2015) conduct a farm-scale simulation study to show the potential for increasing the adaptive capacity and reducing the vulnerability of livestock systems to weather variability by increasing their agricultural diversity.

3.2.2 Biodiversity and output: a complementary or competitive relationship

The fact that land users benefit directly from biodiversity is thought to depend upon whether the provision of biodiversity involves a complementary or a competitive relationship (Wossink and Swinton, 2007; Chavas and Di Falco, 2012; Gullstand, 2014). Biodiversity and livestock production are closely intertwined. Without defoliation by domestic herbivores the landscape would revert from an open landscape to natural woodland and the diversity of species-rich grassland may fall (Reinhammar, 1995; Moog et al, 2002). However, beef, lamb, and biodiversity are all obtained from the use of livestock as inputs into the production process. Competition between outputs in the use of fixed inputs is likely, such as in a fixed area of land between single species and mixed systems or between dairy and dry stock farming. The relationship between milk and beef production and biodiversity may also be affected by the intensity of production (Wossink and Swinton, 2007), such as the stocking rate. At some point the grass sward becomes too short with increasing grazing intensity, and as a result biodiversity may fall. In the British uplands, Young et al (2014) employ a state-transition model (STM) to show that overgrazing may trigger the transition from productive dwarf shrub/heath to less productive and less resilient grassland/degraded wet heath. Gullstrand et al (2013) show that biodiversity provision can impose costs on Swedish farmers and that biodiversity increases the costs of market commodities such as milk and beef.

The relationship between biodiversity and productivity may exhibit complementary or competitive effects, but it can be characterised as non-linear. Harvey (2003) reports a positive linear relationship between biodiversity and productivity but suggests that the relationship becomes non-linear as productivity intensifies. This has important implications for the farmers' bottom line. It indicates that although a farmer may benefit from diversity, there may come a point where the provision of biodiversity imposes costs on the farmer and where they may face trade-offs over biodiversity and production.

This implies that farmers face trade-offs between the provision of biodiversity and market produce if they intensify their operations. Increasing stocking rates is an obvious way to enhance productivity but our results indicate that high stocking rates are detrimental to biodiversity. Given the positive marginal effects of biodiversity on profit efficiency, increasing livestock numbers may come at a cost and may reduce the private value of biodiversity to the farmer.

3.3 Tourism values, protected areas, and biodiversity

Many empirical studies which are applied to wilderness areas indicate **that the value of recreational and other non-marketed direct values which are derived from areas of high nature conservation value can be significant and may compare favourably to competing commercial uses of the same resource.** For example, Hanley and Craig (1991) contrasted the trade-offs which were implicit in permitting or prohibiting afforestation with respect to the flow country, in northern Scotland (the largest body of blanket peat bog in the northern hemisphere). The development would generate employment and produce timber but would also displace extensive populations of internationally rare breeding birds. They demonstrated that the total recreational value of the resource exceeded the benefits which could be derived from afforestation at discount rates of 6%, 4%, and 3%. Similarly, Willis (1991) established that the total recreational value of the Forestry Commission estate in the UK exceeded the value of timber sales.

A study conducted by Brown et al (1994) values the northern spotted owl and its ancient old growth forest habitat using the contingent ranking approach. In this study, respondents were offered five different policies. Associated with each policy were the cost of the policy itself, the area being preserved, the estimated number of owl pairs being preserved, and their probability of survival. They estimated the existence values for conserving the northern spotted owl at about US$20 per person per year. Probabilistic theoretical models have been used to determine the benefits of important wildlife species such as the northern spotted owl in old, naturally regenerated red wood forests, and have demonstrated the high marginal cost of preservation (Montgomery et al*,* 1994). Estimates based on the probability of survival and a reduction in supplies of timber stumpage, provide an estimated welfare cost of US$21 billion to ensure an 82% chance of the species surviving. Increasing the chance of survival from, say, 90% to 95% was estimated to cost an additional US$13 billion.

Nobre (2009) suggests that a TEV approach can be used to evaluate *Impact* in regard to changes in *State* which are caused by different pressures on a coastal ecosystem.

A review of the literature covering the demand side estimates serves to indicate that there may be a number of attributes that will affect whether individuals visit protected areas for recreational use. Some of these may be useful as indicators in the DPSIR study. These can be grouped into a handful of umbrella dimensions including *Control*, *Endowment*, *Infrastructure*, *Access,* and *Social*. The *Control* dimension reflects socio-demographic attributes, simply the number of visitors, their income, family composition, number of children, respondent gender, age (in particular <12), and education (Bennett et al, 2003, Ja-Choon et al, 2013, Juutinen et al, 2011, Kil et al, 2012, Schirpke et al, 2013, Sevenant & Antrop, 2010, Stamps, 1999). Other attributes relate to stated or revealed ecological and/or environmental attitudes (Bennett et al, 2003), and similarly to the purpose, type, or duration of recreation use (Ja-Choon et al, 2013, Juutinen et al, 2011, Kil et al, 2012). Many studies have included control attributes which represent the proximity of trail-to-residence (Ja-Choon et al, 2013, Dhakal et al, 2012, Kil et al, 2012, Lieber & Fesenmaier, 1985), and related to this is whether the respondent is a domestic or foreign tourist/user (Juutinen et al, 2011, Tyrväinen et al, 2013). Other important control attributes include the magnitude of trail entrance prices or fees (Bennett et al, 2003, Juutinen et al, 2011), whether the respondent is an expert or not, and whether the respondent is a member of a special interest group or other organisation (Sevenant & Antrop, 2010, Stamps, 1999).

What may be termed Trail *Endowments* could be important since they capture supply side features such as (Ja-Choon et al, 2013)'s **biotic attributes**, including singular or general natural characteristics such as biodiversity, ecological complexity, composition, and land cover, forest and vegetation characteristics (eg. the forest's age or tree dimensions), aesthetics, ecology, naturalness, and undisturbed landscapes, see (Arabatzis & Grigoroudis, 2010, Bennett et al, 2003, Bestard & Font, 2009, Dhakal et al, 2012, Graefe & Burns, 2013, Ja-Choon et al, 2013, Juutinen et al, 2011, Kil et al, 2012, Lynn & Brown, 2003, Mau-Crimmins et al, 2005, Nahuelhual et al, 2013, Neuvonen et al, 2010, Schirpke et al, 2013, Sevenant & Antrop, 2010, Tveit et al, 2006, Tyetal, 2013, van den Bergh et al, 1998). Endowments may also capture abiotic attributes such as scenic beauty, riparian, forest, or waterway features, the depth of visual perception, spatial scale, openness and views, fresh air, peace and quiet, park or trail shape and size, soil conditions, park history and/or age, landscape composition in terms of agricultural versus valley settings, and landscape cohesiveness (Bennett et al, 2003, Bestard & Font, 2009, Mau-Crimmins et al, 2005, Kil et al, 2012, Lieber & Fesenmaier, 1985, Lynn & Brown, 2003, Nahuelhual et al, 2013, Neuvonen et al, 2010, Nielsen et al, 2012, Reichhart & Arnberger, 2010, Schirpke et al, 2013, Sevenant & Antrop, 2010, Tveit et al, 2006). **Other abiotic endowment** attributes may be related to geophysical/biophysical indicators which are often quantified by GIS metrics, such as fragmentation, visibility, shape, elevation, and slope (Bestard & Font, 2009, Reichhart & Arnberger, 2010, Schirpke et al, 2013, Sevenant & Antrop, 2010, Tveit et al, 2010, Ólafsdóttir & Runnström, 2013). **A third category** of endowments relate to anthropogenic attributes such as trail use impacts, disturbed landscapes, and environmental conditions, eg. litter, muddiness, unplanned trail widening, vegetation damage, and exposed roots (Lieber & Fesenmaier, 1985, Lynn & Brown, 2003, Moore et al, 2012, Ólafsdóttir & Runnström, 2013, Reichhart & Arnberger, 2010, Sevenant & Antrop, 2010). Other anthropogenic attributes include trail history, archaeological sites, local place meanings (Kil et al, 2012, Neuvonen et al, 2010), tourism attractiveness, park capacity (Nahuelhual et al, 2013), human artefactualism, and distance to roads and/or cars (Lynn & Brown, 2003, Nahuelhual et al, 2013, Reichhart & Arnberger, 2010). Note that there may be definitional crossovers or interactions amongst the biotic, abiotic, and anthropogenic categories of endowments, eg. naturalness and visual scale. **Fourthly and lastly**, several authors have reported an important endowment attribute category which is related to ephemera, eg. sunlight though the canopy falling upon historical sites, creating transitory and subjective coincidences of indicators that provide positive respondent experiences (Nielsen et al, 2012, Tveit et al, 2006).

The *Infrastructure* which is associated with a protected area or recreational amenity can also be important, and this represents a third dimension which is associated with supply side features and is related to trail or park. One category of infrastructure indicators describes management elements including the management of usage impacts (little clean up or management of disturbed landscapes), as well as forest or park management (eg. clear cutting or forest thinning) - see (Ja-Choon et al, 2013, Reichhart & Arnberger, 2010, Sevenant & Antrop, 2010, Tyrväinen et al, 2013). A second category of infrastructure relate to trail or park amenities such as park services and recreational facilities which are available to users on site or in the local community, such as accommodations, toilets, customer service personnel, or parking (Arabatzis & Grigoroudis, 2010, Bennett et al, 2003, Buckley et al, 2009a, Dhakal et al, 2012, Graefe & Burns, 2013, Ja-Choon et al, 2013, Juutinen et al, 2011, Neuvonen et al, 2010, Reichhart & Arnberger, 2010). The final category of infrastructure which has been identified in earlier studies characterise walking trails themselves and describe attributes such as the provision of, or numbers of, existing or improved hiking and cycling trails, trail information boards, trail markings, resting places, trail surface types and preparation, the spatial configuration of trails (loops or one-way), trail design (eg. the number of curves and/or sight distance), and proximity to anthropogenic features - see (Bennett et al, 2003, Dhakal et al, 2012, Ja-Choon et al, 2013, Howley et al, 2012, Juutinen et al, 2011, Mau-Crimmins et al, 2005, Neuvonen et al, 2010, Nielsen et al, 2012, Reichhart & Arnberger, 2010, Tyrväinen et al, 2013).

A fourth dimension of attributes relate to *Accessibility* and social features. These have previously been described as institutional features (access agreements), and can capture demand side features such as proximity to population centres or high population density areas, or proximity to other recreational walking sites (Buckley et al, 2009a & 2009b, Graefe & Burns, 2013, Nahuelhual et al, 2013). This group of attributes also includes social features. One category of attributes within this dimension could be related to crowding and solitude. This includes the actual or expected likelihood of unwanted social interactions as well as the numbers of large user groups (Arnberger et al, 2010, Graefe & Burns, 2013, Ja-Choon et al, 2013, Juutinen et al, 2011, Kil et al, 2012, Lieber & Fesenmaier, 1985, Reichhart & Arnberger, 2010). Another social category relates to the type of use conflicts (eg. walking versus cycling and, related to this differential in users' speeds when on the trails) (Arnberger et al, 2010, Reichhart & Arnberger, 2010). Lastly, attributes which are related to tourists' usage aptitude as well as to security concerns have also been cited as important social considerations (Nahuelhual et al, 2013, Ja-Choon et al, 2013).

The supply side provision of tourism facilities and biodiversity may be important too (Mulder et al, 2006; Buckley et al, 2009). Previous research has examined good public provision by landowners to forests (Bateman et al, 1996; Alavalapati et al, 2004; Sullivan et al, 2005; Shaikh et al, 2007) and for environmental services (Garrod and Willis, 1996; Kline et al, 2000; Vanslembrouck et al, 2002; Cooper, 2003; Thomas and Blakemore, 2007). Crabtree and Chalmers (1994) investigated the costs involved in public access provision on private farmland in Scotland. Public policy criteria demand that any scheme be delivered efficiently on a cost minimisation basis. In the literature it is taken as a given that decisions over access provision should be guided by allocative efficiency criteria, and that the economic benefits (and costs) should be clearly identified and valued (Hanley and Spash, 1993). One study (Buckley at al, 2009) identified three clear benefits: non-providers, providers for free, and willing providers. Willing providers were operating on marginal soils, located in areas of high recreational demand, and were willing to provide access at low cost, thereby indicating a high cost benefit ratio.

The fees which are generated by a protected area are important, but the wider benefits may be more significant. An analysis of the eco-tourism sector, which focuses solely on the *supply* and *demand* of biodiversity and does not include an assessment of the wider economic values, will not capture the total economic value to the rural economy (Hurley et al, 1994). **The *indirect* economic values which are associated with tourist expenditure in a region may be considerable, due to a multiplier effect of tourist income on the rural economy** (Midimore, 2000; Hurley et al, 1994). These include increased incomes in an area, or an increase in employment brought about by tourist activity - for example, an increase in the sale of walking equipment, food and/or local products, and employment created from the provision of accommodation and meals for tourists. Enhanced visitor management has the potential to promote this external benefit further, and improved provision of goods and services matched to visitor needs may increase the proportion of trip expenditures spent locally.

Expenditures by holidaymakers are significantly higher than those by day trippers (Bergin & Ó Rathaille, 1999; Hurley et al, 1994). It is also clear that direct employment and expenditure represents only a small proportion of the overall economic impact of the tourist visit (SNH, 1998). There may be a strong case for promoting the use of tourism as part of a longer visit in order to maximise the local economic benefits while reducing the environmental impact of travelling to the sites. The infrastructure and size of the local economy may also be important. This is due to the extent to which injections of expenditure are retained in the local economy, and therefore the multiplier effect may depend to a large degree on the size of the local economy (Hurley et al, 1994). For example, smaller accommodation establishments tend to generate higher multipliers than hotels, because a greater proportion of expenditure is on locally sourced goods and services; Green tourism is often more embedded in the local economy, respecting local traditions, using local produce, and employing local people, and therefore it often produces a large local multiplier effect (Ní Mhainnín, 1996).

Public preferences can be elicited directly using stated preference methods (Bateman, et al*,* 2001; Clinch, and Murphy, 2001; van Rensburg, et al*,* 2002). To evaluate *indirect* effects an assessment of the range of expenditure, employment, and income multipliers can be conducted to provide an indication of the wider economic impacts of visitor expenditure (Keane et al*,* 1992; Keane, 1996). Estimates of expenditure averaged across each local job can also be made (Stynes, 2001; Lutz et al*,* 2000; Leones & Dunn, 1999; Hurley et al*,* 1994, Herriges & Kling, 1999).

Respondents may weigh biodiversity and recreation differently, and they can be asked these questions. They can also be asked to indicate the importance of biodiversity or the presence of a trail infrastructure and facilities for their safety and the enjoyment of a walk in the countryside. Attributes tested include the level of biodiversity, stiles and footbridges, an information point, a map or guide, a trail, route signs, a car park, measures to control erosion, or a guaranteed access agreement with the landowners/recreation provider (Buckley at al, 2009). Individuals that are WTP often single out biodiversity or trail attributes as being very important (Buckley at al, 2009). In one study the high-paying group systematically placed a much higher level of importance on all of these attributes (Buckley at al, 2009).

Environmental quality can influence whether the wider community benefits from tourism. Vanslembrouck et al (2005) reports that amenities from agriculture have a positive influence on rental prices in tourist areas, but negative external effects have a negative impact upon rental prices. Vaughan et al (2009) show that there are four components of the tourist multiplier: initial tourist spend, direct impact upon jobs and income, indirect impacts, and induced income resulting from people spending incomes earned as a consequence of visitor spending.

Multi-functionality and the delivery of public goods through agriculture is now at the forefront of the policy agenda in the EU and elsewhere (Brunstad et al, 1995; Brouwer and Slangan, 1998; Hanley et al, 1998; Fleischer and Tsurz, 2000; Randell, 2002; Gerowitt et al, 2003; Hall et al, 2004; Bills and Gross, 2005). Recreational activity, of the type described above, constitutes an important component of the multifunctional role played by agriculture in revitalising and sustaining the rural economy. **The analysis presented here indicates that there is significant scope for policy approaches that support the development of non-consumptive recreational land uses and sustainable tourism** in marginal areas of the Republic of Ireland.

Indicators:

- Cost benefit analysis, Net Present Value weighing the discounted benefits and costs using a social discount rate
- Multiplier effects of tourism income, tourism expenditure, local employment, and expenditure per local job
- The number of tourists engaged in outdoor activities or in visiting parks
- The proportion of special interest activity tourism
- The percentage of the adult population using recreational trails or protected areas
- The number of walking trails, and the length of trails (in kilometres)
- Estimates of total expenditure on travel, food items, entry fees, and accommodation, and expenditure on walking equipment
- Total number of annual domestic trail visits
- Trail attributes, and biodiversity or recreational facilities
- Supply-side estimates of biodiversity amenities
- Multifunctional use of the landscape
- Percent occupancy. Average number of nights stayed by the guest during peak season. Days in the region
- Local B&Bs associated with high amenity areas and their earnings
- Rental prices for accommodation in tourist areas

- Special Areas of Conservation (SAC), Special Protection Areas (SPA), Natural Heritage Areas (NHA) and Bird Protection Areas (BPA)
- Tourist accommodation occupancy
- Employment in tourism
- Beach closure
- Tourist resident ratio

3.4 Uncertainty, ecosystems, and biodiversity

Although valuation is useful in capturing the value of biodiversity, there are a number of problems with this approach. Research on biodiversity by Perrings and Pearce (1994) suggest that there may be a threshold past which resilience is threatened and the ecosystem is irreversibly changed. The problem is fundamentally one of uncertain ecological thresholds which imply that damage functions are not smooth and continuous in relation to changes in economic activity but ‘jump’ once economic activity exceeds a certain level (Q^S) whereby an ecosystem is driven beyond some critical point (a) to a new point involving much higher external costs (b) as shown in Figure 9.

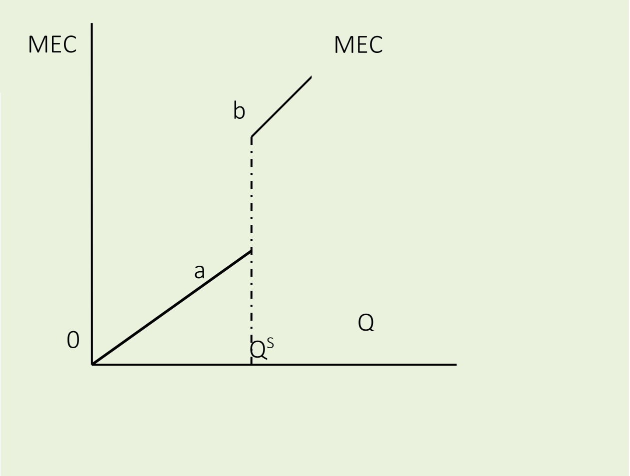

Figure 9: Q^S indicates a critical threshold which, if crossed, leads to a jump in the marginal damage function between points a and b with an increase in economic activities (Q) such as farming, forestry production, or fishing.

Thresholds are known to occur in many terrestrial habitats, as well as marine ones (Perrings and Walker, 1997; Moberg and Folke, 1999; Muradian, 2001; Guttal and Jayaprakash, 2008; Petersen et al, 2008). The issue of uncertainty may matter because it will influence the type of response under the DPSIR framework.

4 POLICY ISSUES

4.1 Policy and market instruments

Conventional economic theory seeks to cast government in the role of an objective and well-informed 'third force' (in addition to individuals and businesses), with some ability to intervene to correct for market failures. **Government or policy failure occurs when policy decisions which are required to correct for market failure are not implemented and fail to fully recognise, or incorporate, the values that are associated with environmental resources.** Policy failure may also arise where government decisions themselves induce economic inefficiencies. Poorly formulated policy instruments and incentives may unintentionally distort the allocation of resources. Subsidies that aim to promote cash crops to secure export revenue may result in land degradation, soil nutrient losses, and a reduction in the resilience of ecosystems (Grainger, 1990). Royalties in forestry can lead to excessive rates of deforestation (Repetto, 1989; Barbier, et al*,* 1991).

A comprehensive review of policies for the Baltic States is beyond the scope of this study. However, a short description is included below of three themes of relevance. These are the nitrates directive reform of the CAP, agri-environment schemes, and the safe minimum standards and precautionary approaches.

Nitrate pollution is a serious problem throughout the EU, and agriculture is one of the main contributors to the problem. The regulation of nitrates on farms in the EU is governed by the 1991 'Nitrates Directive'. The main objective of the EU Nitrates Directive (*Council Directive of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources* (91/676/EEC) is to reduce nitrate concentrations to below an acceptable level of 50mg/litre, Under the directive each member state must implement an action plan which ensures that the applications of nitrogen to farmland are within limits calculated to avoid a level of nitrate emissions into water supplies that would put them above the concentration level specified in the directive. The premise of the action plan is that farmers should take all reasonable steps to prevent or minimise the application to their land of fertilisers which exceed crop requirements.

Economists usually concentrate on the relative cost-efficiency of market instruments, but other criteria also exist: for example, the level of uncertainty regarding policy outcomes; the equitable distribution of costs; political acceptability; and co-ordination with existing policy in the wider sphere (either agricultural or environmental). No policy option is likely to be preferred simultaneously on all of these grounds, therefore 'policy choice is also implicitly a choice among competing criteria' (Hanley, 1997). In practice, command and control (CAC) measures such as input regulations and management practices rather than economic instruments are commonly used to deal with nitrate pollution from agricultural sources (Parsche & Radulescu, 2004; O'Shea, 2002). The EU has adopted a command and control (CAC) approach, in the form of the Nitrates Directive, rather than economic instruments to deal with the problem of nitrate pollution.

Historically, production-based payments under the EU's Common Agricultural Policy (CAP) have led to a steady intensification of farming systems and have provided an incentive for European farmers to increase sheep and cattle numbers and intensify tillage and arable cropping systems which has given rise to severe environmental degradation in some areas. CAP policies, by creating incentives for farmers to expand production, may result in a greater level of degradation in this area than would be the case in the absence of such policies.

To counteract these trends, the European Union introduced significant reforms, under what is known as Pillar II of the CAP, which includes agri-environment schemes (under EU Council Regulation 2078/92) in order to encourage farmers to carry out their activities in a more extensive and environmentally-friendly manner (Emerson and Gillmor, 1999), as well as and a number of initiatives in support of the provision of biodiversity in agriculture (COM, 2006). However, despite these policy reforms, farmers remain reliant on subsidies such as the Less Favoured Areas scheme (the LFA, introduced in 1975), and the single farm payment and agri-environment schemes, and there is a dearth of knowledge regarding the efficacy of agri-environment schemes which relate to biodiversity. At present it is still not clear what effect agri-environment schemes have on biodiversity in European agro-ecosystems (Feehan et al, 2005; Murphy et al, 2011; van Rensburg and Mulugeta, 2016). The single farm payment is based on the number of premium claims made in the historical three year reference period between 2000-2002.

In cases regarding species extinctions or where there is a near irreversible effect of development on biodiversity/environmental resource then this sets the scene for alternatives to cost benefit analysis (CBA) or an Environmental Impact Assessment (EIA), in the form of a precautionary approach which is allied to changes in the use of production methods.[8]

In many ecosystems thresholds are not known with any certainty. Figure 9 (a) shows that, if left unregulated, the external costs which are connected to a resource can be expected to be discontinuous at a particular level of economic activity Q^S and may jump at this point to a much higher level, and then continue to increase with the level of development. If regulation is not introduced to protect the ecosystem or its biodiversity, the private user will operate at Q^{Π} where Marginal Net Private Benefit (MNPB) is maximised.

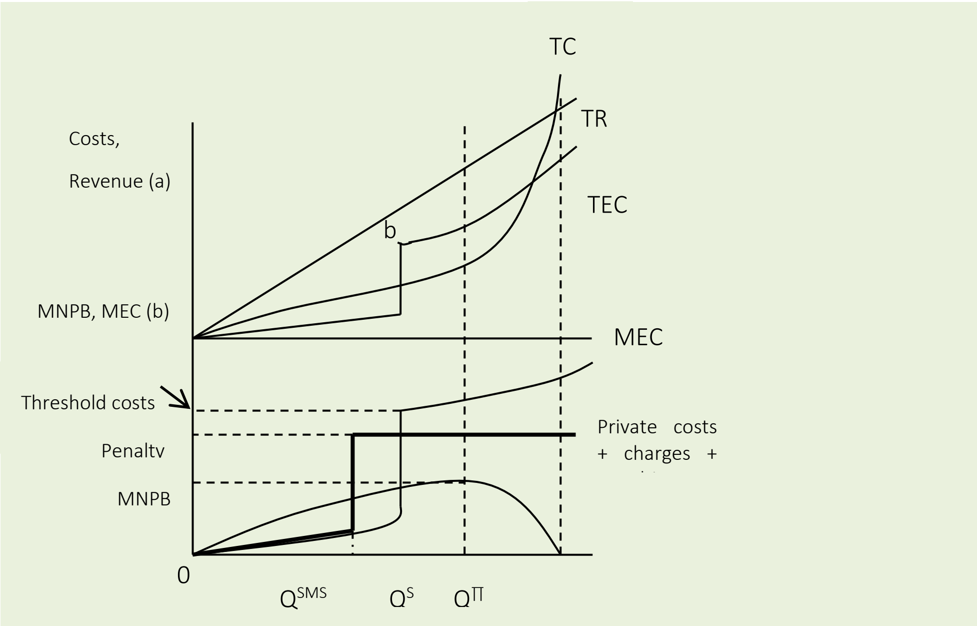

Figure 10: Penalty and threshold costs where thresholds are not known with certainty (adapted from Perrings and Pearce (1994)). Panel (a) indicates Total Costs (TC) and Total Revenue (TR), and shows a discontinuous Total External Cost (TEC) function. Panel (b) shows Marginal External Costs (MEC), Marginal Net Private Benefit (MNPB), and a 'private costs and charges plus penalties' function.

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[8] The policy of taking action before uncertainty about possible environmental damages can be resolved has been referred to as the 'precautionary principle'. One justification for this is that costs which are related to damage to biological resources may exceed the costs involved in taking preventative action (Taylor, 1991). Also, irreversible damage may occur, such as species extinctions. The emphasis is therefore on avoiding potentially damaging situations in the face of uncertainty over future outcomes (Myers, 1992).

The existence of threshold effects invalidates normal tests for efficiency in the allocation of resources. Discontinuities imply that it is no longer feasible to equate marginal net private benefit with marginal external cost. The implication of this being that regulatory measures to protect certain thresholds cannot rely solely on economic criteria. They are set according to ethical judgements about some socially acceptable margin of safety. The precautionary principle is seen as an ethical judgment about how society should respond to risks in the face of collective ignorance about the future impact upon the environment which development activities may cause (Perrings and Pearce, 1994).

4.2 Goods and services provided by ecosystems

Although it is not central to this paper, **the ecosystem services concept** (MEA, 2005) can be used to demonstrate how ecosystems contribute to public well-being and to examine monetary tradeoffs between market and non-priced services.

In the following section ecosystem goods and services are presented, using their supply by cold water coral as an example, based on the categorisation in the Millennium Ecosystem Assessment (MEA, 2005), starting with the more indirect supporting services that can potentially feed into the more direct provisioning, regulating, and cultural services.

Supporting services

Biotic supporting services refer to the functional values which are associated with biodiversity,[9] and the role that ecosystems provide in supporting agricultural or marine production processes such as food production or fisheries (Costello et al, 2005) through what has been termed facultative habitat or 'Essential Fish Habitat' (EFH) (Rosenberg et al, 2000). EFH is defined as 'those waters and substrate necessary to fish for spawning, breeding, feeding, or growth to maturity' (Anon, 1996). Facultative habitat use is defined as the use of habitat for many important life processes, but that the absence of these habitats does not result in the extinction of the species in question.

Provisioning services

Many ecosystems are thought to provide biodiversity which offers new opportunities for pharmaceutical, engineering, medical, and food research (Ehrlich et al, 2006).

Regulating services

The most significant contributor to global warming is anthropogenic carbon dioxide (CO_2). The rapid release of CO_2 poses a fundamental threat to many ecosystems (Turley et al, 2007). The sequestration - that is, the storage or absorption of CO_2, whether naturally or by human intervention - is high on the international climate policy agenda. Ecosystems may sequester CO_2 and thereby remove CO_2 from the atmosphere. The emerging potential for CO_2 storage either in the oceans or in important terrestrial habitats such as forests is an important issue. In this light, national policy for the protection of forest or oceans could potentially be used in a similar vein to those covering afforestation, thereby alleviating climate change.

Cultural services

Many ecosystems and natural environments offer a host various recreational, amenity, or tourism services. Biodiversity is important in terms of amenity values but is also important in terms of other values, including education, and preservation values. Another service that ecosystems may supply are

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[9] Functional diversity refers to the characteristics of ecosystems and includes ecosystem complexity at different levels of organisation such as trophic levels (Cousins, 1991; Barbier, 2000). This approach uses trophic-level analysis to relate species diversity to functional ecosystem parameters such as food web structure or the transfer of energy, water, and chemicals between different trophic levels. Functional diversity can be interpreted as the number of species required for a given ecological process.

their potential as archives which record intermediate-to-sub-surface water temperatures and salinity (Lutringer et al, 2005). Recent concerns about climate change have emphasised the need for long-term proxies of climate change in the oceans (Risk et al, 2005). Ecosystems may indeed provide a unique record of temperature changes and could serve as a good climate change proxy (Puglise et al, 2005). Suggestions of a linkage between some marine ecosystems may lead to the use of an indicator for oil and gas exploration (Rogers, 1999).

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