Environmental Impact Assessment Review 59 (2015) 1–9

Contents lists available at ScienceDirect

Environmental Impact Assessment Review

journal homepage: www.elsevier.com/locate/eiar

Tools for integrating environmental objectives into policy and practice: What works where?

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ARTICLE INFO

Article history:
Received 8 October 2015
Received in revised form 4 March 2016
Accepted 4 March 2016
Available online xxxx

Keywords:
Environmental policy integration
Mainstreaming
Integration
Tools

ABSTRACT

An abundance of approaches, strategies, and instruments – in short: tools – have been developed that intend to stimulate or facilitate the integration of a variety of environmental objectives into development planning, national or regional sectoral policies, international agreements, business strategies, etc. These tools include legally mandatory procedures, such as Environmental Impact Assessment and Strategic Environmental Assessment; more voluntary tools such as environmental indicators developed by scientists and planning tools; green budgeting, etc. A relatively underexplored question is what integration tool fits what particular purposes and contexts, in short: "what works where?". This paper intends to contribute to answering this question, by first providing conceptual clarity about what integration entails, by suggesting and illustrating a classification of integration tools, and finally by summarising some of the lessons learned about how and why integration tools are (not) used and with what outcomes, particularly in terms of promoting the integration of environmental objectives.

1. Introduction

'Integration' has long been advocated as a way to promote more sustainable policies and planning. By including sustainability objectives into sectoral policies and planning, inconsistencies between these and sectoral objectives that often result from institutional 'compartmentalisation' can be avoided and synergies be achieved (Lafferty and Hovden, 2003; Runhaar et al., 2014). Moreover, in sectoral policies and plans, the driving forces of environmental pressure can be targeted (think of urbanisation or agricultural intensification; Adelle and Russel, 2013).

Policy integration, and its synonym 'mainstreaming', are most often associated with environmental objectives (Lafferty and Hovden, 2003; Jordan and Lenschow, 2010; Runhaar et al., 2014), and, in the last decade with a growing emphasis on climate policy integration (CPI) within the international policy literature on climate change (Huq and Reid, 2004; Adelle and Russel, 2013; Uittenbroek et al., 2014). But the principle is also used in relation to disaster risk reduction (Wamsler, 2006; Fischer, 2014), gender equality (Pollack and Hafner-Burton, 2000), health (Fischer et al., 2010; Carmichael et al., 2012) and, more broadly, sustainability (Rival, 2012; Rietig, 2013; Velázquez Gomar, 2014). Concepts such as 'integrated coastal zone management' (Shipman and Stojanovic, 2007), 'integrated pest management' (Kogan, 1998), and 'integrated water resources management' (Biswas, 2004) are specific operationalisations of policy integration that already have a long history.

In this paper the focus will be on *environmental* policy integration rather than on the integration of sustainability objectives. In order to support the integration of environmental and sustainability-related objectives into sectoral policies and plans, a wide variety of approaches, strategies, and instruments – in short: tools – have been developed. Examples are environmental indicators that facilitate monitoring and policy evaluation; Environmental Impact Assessment (EIA) and Strategic Environmental Assessment (SEA); Poverty Social Impact Analysis; valuation of ecosystem services; sustainability appraisal etc. (Baker and Wong, 2006; Gillingham, 2008; Obst et al., 2015). These tools have an analytical starting point and aim to steer towards integration by the provision of information. Other tools are more procedural in nature, and focus more on mobilising actors and stimulating the creation of support for achieving some sort of policy integration, such as area-based participatory planning tools (Runhaar and Driessen, 2011). Institutional tools focus on reform of e.g. state departments such as of the establishment of environmental units within sectoral departments (Jordan and Lenschow, 2008). And then there are what policy analysts call 'policy instruments' that are more regulatory of nature: environmental taxes, licences, green budgeting, payments for ecosystem services, etc. (Runhaar et al., 2014). Although these tools are rather different in terms of target actors and strategies, their aim is the same: ensure that environmental or sustainability objectives are incorporated in sectoral policies and plans.

The abundance of integration tools available can assist planners and policy-makers who aim for more policy integration in whatever sense, but also may raise the question of what tools to use, in what situation, and for what purpose. In other words: what works where? This question is far too ambitious to answer in one paper but nevertheless provides clear direction and inspiration for future research (Bressers,

http://dx.doi.org/10.1016/j.eiar.2016.03.003
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2004; Lafferty and Ruud, 2008; Runhaar et al., 2014). In this paper I aim to make a modest contribution towards an answer to the above question by providing some conceptual clarity about the concept of integration (Section 2), by suggesting and illustrating a classification of integration tools (Section 3), and by drawing lessons from studies about the use, non-use and sometimes abuse of integration tools (Section 4). The paper concludes with some conclusions and reflections (Section 5).

2. 'Integration' defined and operationalised

Integration refers to bringing things together, linking them, making them part of a larger system (Runhaar et al., 2009). Comparable concepts are 'holistic' (planning etc.) or 'mainstreaming' (Uittenbroek et al., 2013). Jordan and Schout (2006: 66, in Jordan and Lenschow, 2008: 11) define environmental policy integration (EPI) as "*a process through which* "*non*" *environmental sectors consider the overall environmental consequences of their policies, and take active and early steps to incorporate an understanding of them into policy making at all relevant levels of governance*". There is however no generally accepted definition of policy integration, or EPI, in particular (Lafferty and Hovden, 2003; Jordan and Lenschow, 2008; Runhaar et al., 2009). Some conceptual clarity is needed in order to assist planners or policy-makers who aim for integrated policies or plans. Moreover, the development of indicators for measuring the nature and degree of integration may facilitate the (ex ante) assessment of the 'success' of integration efforts, and may also help structuring a debate about what integration to strive after. Below I will discuss three basic questions that are related to defining and operationalising 'integration'.

2.1. What should be integrated and into what?

In Section 1 I indicated that many environmental objectives – environmental, risk, health, etc. – can be sought to be integrated into a wide variety of sectoral plans and policies. *What* is to be integrated can be determined from the top-down – e.g., Sustainable Development Goals (which include environmental objectives) to which states have committed themselves, CO_2 reduction targets, etc. But integration can also originate from the bottom-up within sectors; think for instance of eco-labels such as the Marine Stewardship Council and pressure from consumers and NGOs on companies to reduce particular environmental pressures (Runhaar et al., 2014; Wolf, 2014).

There is some evidence that the degree to which integration takes place depends on what is to be integrated – and how it is framed. For instance, regarding the integration of climate change in urban planning, Wejs et al. (2014) and den Exter et al. (2015) found that mitigation objectives were integrated to a larger extent than adaptation objectives. The framing of what needs to be integrated could explain such differences. Runhaar et al. (2014), Wejs (2014) and Wejs and Cashmore (2014) suggest that a careful framing of the integration objective in such a way that synergies with sectoral objectives are made clear could help to create support for integration (as well as political will - an important factor determining integration ambition levels, as Lafferty and Hovden (2003) suggest). Uittenbroek et al. (2014) suggest that the issue to be integrated can be framed as a solution to another problem, "*For example, climate adaptation can be considered as a problem that requires investments or can be framed as an opportunity for sustaining an attractive and safe city*" (ibid., p. 1046).

The *subject* of integration is often public policy, however, if a broader conceptualisation of integration is adopted, then also environmental measures as part of Corporate Social Responsibility (CSR) policies of companies could be considered as forms of integration (Berger et al., 2007; Knudsen, 2013). The basic idea is the same: incorporating environmental or other objectives into policies or plans in which they normally are not integrated (or at least not beyond what is required by licences or laws), although scope is limited to the company at issue and perhaps its suppliers or customers. More ambitious (perhaps *too* ambitious) concepts are those of 'sustainable supply chains' and 'green economy' (Faisal, 2010; Brand, 2012; Vermeulen, 2015). The scale at which integration takes place hence can differ from an individual organisation (public agency or company) to a particular sector or domain such as development planning, agriculture, transport, or energy (Runhaar et al., 2014).

The extent to which environmental objectives are (or can be) integrated seems to differ across policy sectors. For instance, (Persson et al., 2016) found that in Sweden, in the energy sector a higher degree of environmental policy integration was observed than in the agricultural sector.

2.2. When?

Integration can take place at different moments in the planning process: during the decision-making stage, its implementation, the evaluation or the re-design of policies and plans (Kivimaa and Mickwitz, 2006; Moser and Ekstrom, 2010; Uittenbroek et al., 2013). From several studies it appears that policy integration at the decision-making stage is often (but not always) easier than during the implementation of 'integrated' policies and plans (e.g. Alahuhta et al., 2010; Jordan and

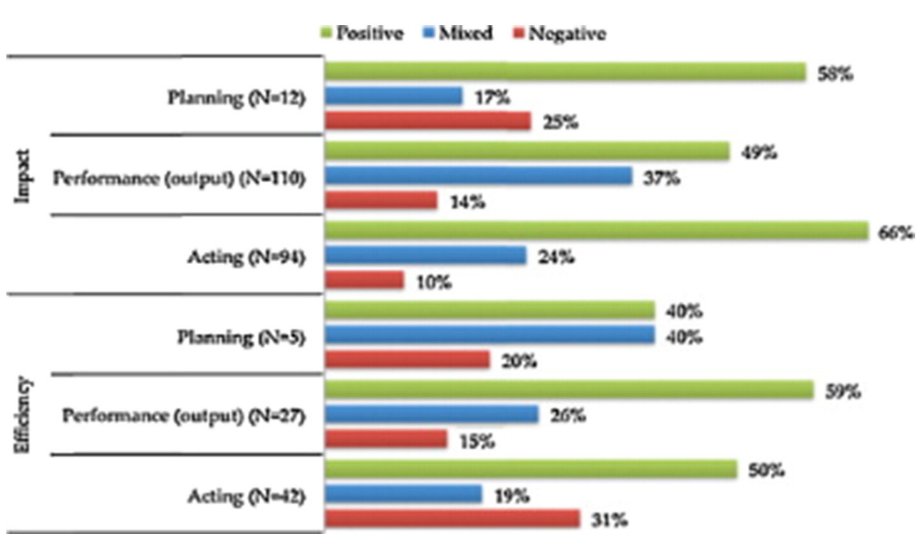

Fig. 1. Impact and efficiency evaluations of low-carbon policies according to the targeted stage.

Lenschow, 2010; Kolhoff et al., 2016; Uittenbroek, 2015). An important explanation is that in implementation, policies and plans are transferred to other 'policy arenas' where different actors are involved who have different interests and routines (Uittenbroek, 2015). Yet, the ultimate impacts of integration efforts are realised during implementation of integrated policies and plans; hence this stage seems particularly important (Kolhoff et al., 2016).

In this context, the following meta-analysis is interesting. In their review of 165 ex post evaluations of low-carbon policy instruments and policies, Auld et al. (2014) find that policies that target the planning stages and the behaviour (acting) of the subjects (mostly firms) have relatively more often positive outcomes than policies that focus on the performance (output) of subjects. An opposite trend is observed regarding the efficiency of these policies (again, see Fig. 1). This implies that at least for some policies and policy instruments, trade-offs exist between these criteria (see also Mees et al., 2014).

2.3. Why and to what extent?

Attempts to integrate or mainstream environmental or other objectives obviously have the aim to bring about change in favour of these objectives. But what is strived after may differ from case to case. The same applies to the level of ambition, or the relative priority of environmental or sustainability objectives vis-à-vis sectoral objectives.

In literature on EIA, often distinction is a distinction is often made between 'procedural' and 'substantive' purposes of this integration tool (e.g. Cashmore et al., 2004; van Doren et al., 2013). In the first case, the emphasis is on compliance with the formal procedure and policy outputs such as Environmental Impact Statements (EIS) and their quality and explicit consideration in decision-making. In the latter case, the focus is on the influence of the tool in terms of how it changes policies and plans in more environmentally friendly ways. Substantive change is more ambitious, but can also be operationalised in different ways. Runhaar et al. (2009), building on Lafferty and Hovden (2003), distinguish between three substantive levels of integration:

- *Coordination*: aimed at avoiding contradictory sectoral policies or at compensating for adverse environmental consequences of these policies. This is a rather basic level of integration;
- *Harmonisation*: aimed at bringing environmental objectives on equal terms with sectoral objectives; here also the search for synergies (e.g. 'green growth') is important;
- *Prioritisation*: aimed at giving priority to environmental objectives in sectoral policies.

Note that this distinction bears some resemblance with that between 'strong' and 'weak' sustainability (Jordan and Lenschow, 2008; Neumayer, 2013; Mullally and Dunphy, 2015). The above categorisations clearly differ in terms of ambition level (see also Persson, 2007). In the development of integration plans, the deliberation about ambition levels could be a logical starting point.

2.4. Reflection

In this section I provided conceptual clarity about the concept of integration tools. In terms of the organising question of "what works where", this section particularly elaborated on the 'works' part of the question. In the next section I will identify and classify tools that promote integration – related to the 'what' part of the question. In Section 4, where I will review evaluations of integration tools, I will explicitly focus on the 'where' part of the question as well as the relation between the three sub questions.

It should be noted that in this paper a broad understanding of 'integration' is adopted, in attempt to draw lessons from as many forms of integration as possible. This implies first that I move from *policy integration* to integration at the level of the *sectors* and *practices in these sectors*, that beforehand I do not exclude particular actors who aim to promote integration, particular tools that they can employ, or what environmental objectives exactly are to be integrated (environmental objectives or sustainability objectives). I therefore do not employ the concept of EPI – which predominantly or exclusively takes *public* actors into account – but will use integration in a broader meaning (cf. Runhaar et al., 2014). I acknowledge that whereas the potential benefit is to draw from a richer set of studies and literature, a potential limitation is that incomparable phenomena are compared. I will explicitly reflect on this potential limitation in the concluding section of this paper.

3. Tools for integration: a classification

In the last two decades or so, a wide variety of EPI integration tools have been implemented or suggested. It is not possible to present an exhaustive list of these. For instance, Runhaar and Driessen (2011) identify no less than 75 tools in the Netherlands that support the integration of environmental objectives into spatial planning – a specific field of policy integration. In this section I will provide a framework that enables a systematic inventory of EPI integration tools and give some examples. For more exhaustive overviews, I refer to e.g. Jacob et al. (2008), Jordan and Lenschow (2008), Zuidema et al. (2012), Runhaar and Driessen (2011), Runhaar et al. (2014), Mullally and Dunphy (2015) and Turnpenny et al. (2015).

From the literature various dimensions of EPI tools can be derived that could form the basis of such a framework. Mullally and Dunphy (2015) distinguish EPI tools regarding their degree of voluntariness among other things. Jordan and Lenschow (2008) distinguish between communicative, organisational and procedural tools. Runhaar et al. (2014) propose a slightly different distinction between legal (or regulatory) tools, economic tools and communicative (or informational) tools. Runhaar et al. (2009) classify integration tools according to their underlying logic and distinguish between substance-oriented, process-oriented and 'hybrid' tools. Authors such as Jordan and Lenschow (2008) also distinguish tools regarding their governance mode, i.e., top-down or bottom up and, with that, between policy levels (e.g. UN, EU, nation state, regional authorities), and between actors involved (public and private) and their roles. Given the broad interpretation of the concept of integration in this paper, the set of tools discussed in this paper is broader than those in the literature referred to above (i.e., including tools initiated by public and private actors and covering a wide variety of mechanisms).

In this paper I consider integration tools primarily as *governance* tools – i.e., tools that aim to steer particular actors in such a way that they are stimulated (or forced) to incorporate environmental objectives in their policies or practices (cf. Arts et al., 2012). I employ a relatively simple framework based on two governance dimensions, Table 1 provides an overview of some integration tools based on the dimensions of governance mode and steering strategy (cf. Runhaar et al., 2014). Regarding the former dimension, distinction is made between classical, government-led governance, interactive forms of governance based on 'horizontal' relationships between governments and societal partners (companies, NGOs, interest groups, etc.), and self-governance (steering by societal actors themselves). The second dimension relates to the logic of steering: in what ways are the subjects of governance steered towards integration. Regulatory tools for instance are based on restricting or allowing behavioural options, economic tools aim to change cost-to-benefit ratios of these options, whereas communicative, informational or analytical tools aim to steer by means of providing information about the consequences of policies or about available options. Organisational tools finally aim to bring about more structural change in the way in which organisations work, and in this way promote integration by changing procedures, incentives, or routines (Runhaar et al., 2014).

Table 1
A framework for classifying integration tools.

Mode of governance[a] → Steering strategy ↓	Top-down government S M CS	Interactive governance S M ← CS	Self-governance M ← CS S
Regulatory			
	EIA and SEA	Voluntary agreements and covenants (e.g. on energy efficiency or reducing packaging volumes)	Voluntary sustainability environmental protection principles and standards with (limited) enforcement (e.g. UN Global Compact or the Equator principles)
	Environmental reporting or green accounting obligations		Voluntary disclosure arrangements (e.g. Global Reporting Initiative)
	Legal requirements (e.g., mandatory use of Best Available Technology in environmental permits)		
Economic			
	Subsidies and taxes	Green procurement	Premiums for green suppliers based on particular indicators
	Green budgeting	Biodiversity conservation markets	
Communicative/informational/analytical			
	Environmental information campaigns	Eco-labels	Sustainability Environmental reporting
	Environmental indicators	Design or planning tools for integrating urban and environmental planning	Voluntary information sharing
	Voluntary EIA and SEA	Capacity building and awareness activities	NGO 'naming and shaming' campaigns
Organisational			
	Environmental units within sectoral departments	Information centres or helpdesks supporting companies to 'go green'	CSR units within companies, HSE officers
	Specific coordinating minister for sustainability		
	Environmental units within sectoral departments	Public-private partnerships	Environmental Management Systems, e.g. ISO 14001 and EMAS
		Green think tanks	Requirements and guidelines, e.g. related to Design for Environment or Ecodesign

[a] The pictures show how actors from the different domains (s = state, m = market, cs = civil society) relate, based on Driessen et al. (2012: 146–147). Examples taken from Morrow and Rondinelli (2002), Jacob et al. (2008), Jordan and Lenschow (2008), Mees et al. (2014), Runhaar et al. (2009, 2014), and Mullally and Dunphy (2015).

As with every classification, also this one leaves out particular details, such as the specific purpose of the tools, the actors usually involved in their application, capacities and resources required for their effective use, the degree of voluntariness of their use, etc. These are further refinements that can be made within the classification if needed. Another critical note is that the distinction made between the categories is an analytical one; some tools may fit into two or even more categories. An example is EIA; a regulatory tool because it is mandatory for major project, policies and plans where substantial environmental impacts can be expected. Yet its rationale is informational/communicative.

As stated above, the value of the classification resides in its broad overview of available tools; in other studies usually the focus is on one particular type of integration tool (e.g. initiated by public actors or applicable to one particular type of policy integration).

4. What works where: insights from the literature

In this section I will review literature on the use and influence of integration tools that were identified in the previous section. In view of the disclaimer made earlier about the incompleteness of the overview of tools, the literature review in this section cannot be complete, either. I will first discuss analyses and evaluations of the use and effects of the four categories of tools, focusing on those tools for which empirical evidence is presented in literature. As far as possible findings in the literature are presented in terms of the levels of integration discussed in Section 2. I will end with some more general observations.

4.1. Regulatory tools

This category consists of tools that aim to regulate the choices, behaviour etc. of its subjects. Regulation can take place in a formalised way (for example legal prescriptions, e.g. to conduct EIAs for particular types of projects or plans), or in a more voluntary way (e.g. compliance with particular disclosure principles by companies).

Typical regulatory tools associated with top-down governance are legal requirements; think for instance of the mandatory use of Best Available Technology in environmental permitting. Legal requirements can be very effective in terms of reducing environmental pressures; however, as they limit degrees of freedom of companies involved, they may raise legitimacy concerns (Mees et al., 2014).

Other well-known regulatory tools are EIA and SEA. Nowadays these tools are adopted by a majority of countries worldwide (NCEA, 2015;

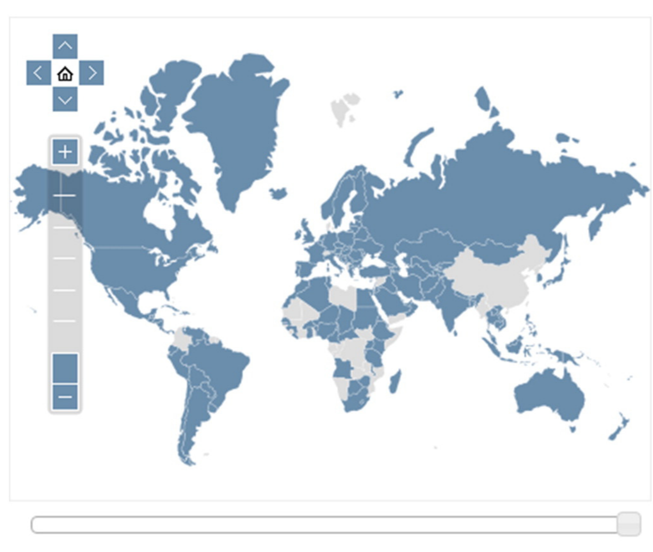

Fig. 2. World map of EIA.

see Fig. 2). EIA and SEA are mandatory procedures that aim to integrate environmental and other considerations into policy-making and planning by demanding the ex ante assessment of environmental and other impacts of particular types of policies and plans. Sometimes also the development and assessment of alternatives is part of the legal procedure. EIA and SEA are found to have an influence on policies and plans at two moments: before the procedure is started (anticipating the EIA or SEA, proponents are perceived to pay specific attention to potential environmental impacts) and during the procedure (when the Environmental Impact Statement (EIS) is delivered and a decision about the EIS should be made). The general observation however is that EIA and SEA usually do not have a large impact on decisions (Arts et al., 2012; Lyhne et al., 2016). In terms of the three levels of policy integration discussed in Section 2, the impression is that seldom levels beyond 'coordination' are achieved (Runhaar et al., 2014). An important explanation for the observation that EIA and SEAs have an impact is found to be its legal base (Arts et al., 2012). This however also often impedes the creative use of EIA and SEA as tools to optimise plans and policies (Arts et al., 2012). Often-heard complaints about EIA and SEA are its too broad scope (i.e., too many aspects are taken into account) and its quality. On the other hand, EIA and SEA are considered to provide transparency to the decision-making process and thus are perceived to have added value. (Runhaar et al., 2013) Kidd and Fischer (2007) finally observe that a broadening of EIA and SEA to sustainability assessment may come at the expense of the attention being given to environmental impacts.

Regulatory tools based on interactive modes of governance mainly include voluntary agreements and covenants. This category of tools has become relatively popular in countries such as the Netherlands and are sometime seven considered as tools that can make up for governments' failure to enforce environmental laws (Glasbergen, 1998; Ruysschaert and Salles, 2014; Cagno et al., 2015). However, many scholars are critical about voluntary agreements in terms of achievement of the objectives agreed upon. From evaluations of voluntary agreements addressing different themes (energy, palm oil, etc.) the following reasons emerge: too much flexibility/room for interpretation; a lack of enforcement mechanisms (no control systems and rewards/sanctions); if applicable financial compensations that are considered to be too small; fundamental sustainability issues, such as trade-offs between environment and economy are not made (Glasbergen, 1998; Bizer, 2014; Ruysschaert and Salles, 2014; Cagno et al., 2015). An important lesson is that voluntary agreements should not be too voluntary – there should be some form of enforcement.

Relatively little evaluation work has been conducted on how and to what extent self-governance oriented integration tools that support CSR policies of individual companies contribute to environmental protection or sustainable development. The UN Global Compact (UNGC) is one of the widest used tools, and frequently discussed in literature. The UNGC specifies ten sustainability principles (partly environmentally oriented) which participating companies have to adopt and report upon (risking delisting in the case of noncompliance), and offers support to subscribing companies in terms of, among other things, best practice examples and regional networks. There are now about 8000 companies worldwide participating in the UNGC (Rasche and Waddock, 2014). There are critical voices about what the UNGC has achieved in terms of realising the ten sustainability principles (e.g. Sethi and Schepers, 2014), which are not commonly shared however (e.g. Rasche and Waddock, 2014). Yet when it comes to empirical evidence the literature remains relatively silent. In a survey by McKinsey and Company (2004) it was found that the UNGC did not triggered many companies to initiate CSR strategies, but rather that it had facilitated the further development of existing ones (except for in non-OECD countries where CSR is relatively new). Cetindamar and Husoy (2007: 172) found that "*all companies indicate that being a UNGC participant completely influences their sustainable development efforts*" but how remains unclear. Then there are some case studies. For instance, Runhaar and Lafferty (2009) found that the UN Global Compact is of modest use to frontrunning companies in the telecommunications sector. Another initiative is the Equator principles that apply to the banking sector. Also this tool has been subject to scientific analysis, but there are few if any evaluations of the outcomes and impacts of these tools in terms of contributing to environmental protection and sustainable development. In this respect, the study by Macve and Chen (2010) suggests these impacts of the Equator principles are rather modest.

Vermeulen (2015) addresses the more general question of whether self-governance (of which CSR is a manifestation) is enough for transforming companies and global supply chains into sustainable ones. He concludes that progress has been made, in part through what are listed as tools in Table 1, but that there remain at least three weaknesses: only particular products and services are targeted; efforts are limited to the first links in the value chain (farming, production); and sustainability issues (including environmental objectives) are selectively addressed.

4.2. Economic tools

Well-known market-based integration tools include subsidies and taxes, tradable permits, liability schemes, biodiversity conservation markets, cap and trade systems such as the European CO_2 Emission Trade Scheme etc. (EEA, 2005; Runhaar et al., 2014). These tools promote integration by putting a price on the environment or the sustainability objective at issue. They are in-between top-down tools and interactive tools; although they often are decided upon and implemented in (more or less) a top-down manner, much depends on the (voluntary) behaviour of the target group (companies, consumers, households). Therefore, below I will discuss experiences with top-down and interactive tools without making an explicit distinction between these two categories of tools.

Various studies have shown that market-based tools can be very effective in promoting environmental protection (e.g. Alvarado-Quesada et al., 2014; Lu et al., 2012; Manni and Runhaar, 2014), although under specific conditions. The effectiveness of these tools first of all strongly depends on the price set and on their enforcement (Wissel and Wätzold, 2010; Ward and Cao, 2012). Also the legitimacy of this tool is an important thing to be taken into account; when applying market-based instruments, distributional effects that follow from their implementation may compromise their effectiveness (Jacka et al., 2008). In addition, much depends on the specific context in which these tools are implemented in terms of design and the potential trade-offs between the environmental and other objectives at issue (Jacka et al., 2008). The European Environmental Agency (EEA, 2005: 7) in its review of a range of market-based tools concludes: "*Evidence suggests that instruments where they have been applied work better if*: *they are well-designed in themselves and as part of a wider package of instruments*; *the reasons for having them and how revenues will be used are clearly communicated*; *the levels at which 'prices' are set reflect both an incentive to producers and consumers to change behaviour and a realistic analysis of affordability*." In conclusion, market-based tools seem to have large potential for the integration of environmental and sustainability objectives into sectoral policies and practices, but there are several considerations that have to be taken into account for this tool to be effective.

Green procurement is a specific economic tool that can be adopted and implemented by both public actors and private actors (companies, NGOs, consumers). In a meta-analysis of scientific papers on green private procurement, Appolloni et al. (2014) find that green procurement usually focuses on reduction of emissions, waste and energy, and a decrease in accidents and positively contributes to environmental improvement in these areas (although it is difficult to quantify these impacts). Gimenez and Sierra (2013) indicate that not only the assessment of suppliers but also collaboration with them as part of green procurement contributes to the environmental performance of procurement.

4.3. Communicative/informational/analytical tools

Although information as a tool to influence behaviour leaves the targeted audiences with a large freedom to act upon the information at issue, Auld et al. (2014) in their meta-analysis of low-carbon policies referred to earlier in this paper observe relatively many positive experiences with this tool in terms of impact (see Fig. 3).

However, other studies arrive at different conclusions. For instance, environmental indicators (including assessments such as ecological footprints) do not seem to be very actively used in planning (Brown, 2003; Higginson et al., 2003; Lehtonen, 2015). In part this is explained by differences in 'languages' between planners and policy-makers on the one hand, and developers of indicators on the other hand. The active involvement of users of indicators in the development of these indicators may therefore be favourable for their use (Brown, 2003). Another reason for the limited use of indicators resides in a mismatch between the geographical scale at the indicators are defined and what their potential users need (Graymore et al., 2008). According to Lehtonen (2015: 86–87) "*relevant factors may include excessively loose linking between reporting schemes and policymaking*; *lack of trust of potential users in the indicators* (*government actors may be institutionally prevented from using 'unofficial' data sources, while external actors may mistrust government data*); *lack of resources within the administration*; *or neglect of user concerns in the design of indicator systems.*"

The use of ecological knowledge in coastal management and projects – a particular tool for, and form of, environmental integration – is found to differ in part along the voluntariness of the production and use of the knowledge at issue. Broadly speaking, the less voluntary the arrangement at issue, the larger its use and its impact on policy-making and planning, but this may come at the expense of the legitimacy of the knowledge at issue (Runhaar et al., 2016). Similar findings are reported by Auld et al. (2014) in their meta-analysis of low-carbon policies regarding environmental or sustainability reporting. These authors found that in only 5% of evaluations of voluntary reporting positive experiences are reported, as opposed to 57% in the case of mandatory reporting.

In their analysis of two particular informational tools – NGO campaigns and eco-labels – Jacquet and Pauly (2007) report mixed experiences in terms of the quality of these tools and impact. Naming and shaming – a specific form of NGO campaigns – can be a very effective tool, but also have rebound effects for the NGOs involved (as for instance in the case of the Brent Spar controversy in the 1990s; see Huxham and Sumner, 1999). Limitations of eco-labels as discussed by van Amstel et al. (2008) include their ambiguity about environmental effects and their (perceived) reliability.

EIAs and SEAs that are conducted on a voluntary basis seem to contribute more to the integration of environmental and other sustainability objectives than mandatory ones. The voluntary use is more often associated with actors open to environmental values and willing to use EIA and SEA not only as assessment tools but also as design tools for policies and plans (Arts et al., 2012; Runhaar et al., 2013).

Various researchers have observed that the use of planning and appraisal tools (in general and tools that encompass an integration component) is rather limited, and that tools that are used, usually do not make much of a difference in policy-making and in planning (Vonk et al., 2005; Jacob et al., 2008; Nilsson et al., 2008; Te Brömmelstroet and Schrijnen, 2010; Runhaar and Driessen, 2011; Arts et al., 2012; Geertman, 2013; Lyhne et al., 2016; Turnpenny et al., 2015). In terms of the level of integration achieved, the literature gives the impression that seldom levels beyond 'coordination' (see Section 2) are achieved (Runhaar and Driessen, 2011; Runhaar et al., 2014). There are multiple reasons (Vonk et al., 2005). Partly, there often seems to be a technical mismatch between what planners need and what the tools can provide (i.e., level of abstraction, a lack of insight into the financial consequences of policy options). For another part, planning tools seem to ignore the political aspects of policy and planning (i.e., a lack of room for involving multiple interests or debate or a lack of guidelines about how to deal with controversies during the application of the tools). (Runhaar et al., 2009).

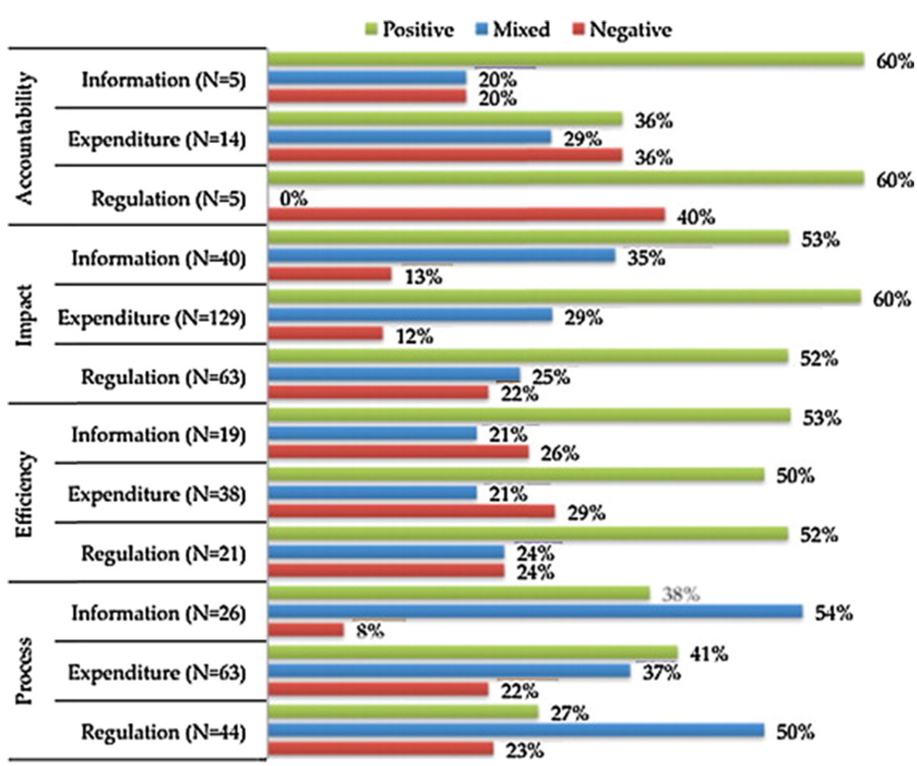

Fig. 3. Evaluations of low-carbon policies according to the type of instrument.

In a recent evaluation of planning tools for supporting Dutch municipalities in developing climate adaptation plans, Tijhuis (2015: 83) observes that these tools "*seem suitable to support municipalities in their initial adaptation endeavors, particularly for acquiring knowledge. However, the tools in itself are often not suitable for incentivizing adaptation planning, and for the actual implementation of adaptation actions. A main reason for this is that the tools are not specific enough.*"

Finally, regarding Environmental Management Systems, Pawliczek and Piszczur (2013) found that ISO 9000 and ISO 14000 have an impact on companies' awareness of sustainability environmental priorities. I found no evidence of the extent to which these tools contribute to a reduction in environmental pressures (either in a qualitative sense, or in a more qualitative way; see Section 2).

4.4. Organisational tools

This last category of tools aim to bring about more structural change in terms of the integration of environmental and other sustainability objectives by changing organisational structures. Although this 'tool' is proposed by different authors (e.g., Lafferty and Hovden, 2003) and is implemented both in the public domain and in the private domain, I found little empirical evidence of the performance of these tools. An exception are partnerships; voluntary cooperations between state actors, NGOs and companies or between NGOs and companies only. Regarding business-NGO partnerships, Bitzer and Glasbergen (2015: 35) state "*partnerships seem to solve some problems but also create new ones*". The same seems to apply to government-stakeholder partnerships (see e.g. Visseren-Hamakers, 2013; Lamers et al., 2014). Holman (2013) in a study of urban planners engaging in partnerships, suggests that the continuity of partnerships as well as the interconnectivity between partnerships are problematic. Lamers et al. (2014) emphasise the active governance of partnerships.

4.5. Overall observations

In this section I provided an overview of analyses and evaluations of a variety of tools that can be used to promote the integration of environmental and sustainability objectives. Although surprisingly little empirical evidence of their performance is found, let alone comparative evaluations of different types of tools (cf. Runhaar et al., 2014), the evidence presented above suggests that we should be modest in our expectations regarding what these tools can achieve.

At various times it was observed that there may be trade-offs between (potential) effects of integration tools. An important trade-off seems to exist between effectiveness and legitimacy. Various economic tools and mandatory tools have been proven to be potentially effective in realising their objectives, albeit seemingly not above levels of 'coordination' (see Section 2). Their effectiveness however may come at the cost of their legitimacy or acceptability. This can be a serious political hurdle in decision-making about the implementation of economic and mandatory tools. In addition, the less voluntary tools are, the less a creative use of these tools seems to be provoked. On the other hand, voluntary tools leave room for a more creative use, but also for simply disregarding these.

The existence of trade-offs may however be an argument to combine tools. For instance, when stakeholders put pressure on supermarkets to 'green' their supply, governments may help by providing information to supermarkets or consumers about 'green' products or NGOs may start partnerships round particular product groups (e.g. the Marine Stewardship Council for fish and seafood). Combinations in time are also possible; for instance subsidies for clean technology followed by legal requirements for those companies that do not invest in this technology on a voluntary basis (Mees et al., 2014).

Specific attention should be paid to the (usually context-specific) conditions that influence the impact that tools can have. Based on van Enst et al. (2014) distinction can be made between two main categories of conditions:

- *Operational conditions*: e.g., the fit between the type of information that integration tools yield and what the target audience (e.g. planners) want, the moment in the planning process the tool is employed, the fit between the tool and organisational routines, a fit between the implicit or explicit ambitions of the tool and its users, etc. (e.g. Vonk et al., 2005; Runhaar and Lafferty, 2009; Te Brömmelstroet and Schrijnen, 2010; Pelzer et al., 2015);
- *Strategic conditions*: the deliberate ignoring or selective use of integration tools or their outcomes seems to be related to whether or not the tools in terms of their purposes and outcomes align with actors' interests (Nilsson et al., 2008; Turnpenny et al., 2015). This should be taken into account in the tool selection and their design and use.

A critical condition regarding the effectiveness of integration tools that emerges from various studies is their *enforcement* (in whatever form). This condition thus should be given specific attention when implementing a tool or a combination of tools.

5. Conclusions and reflection

Integrating environmental objectives into sectoral policies, planning, and practices has long been advocated as a promising strategy for promoting environmental protection sustainable development. Not only can integration avoid trade-offs and inconsistencies between environmental and sectoral objectives, also synergies might be achieved. Moreover, driving forces of environmental problems may be targeted more directly than in 'stand-alone' environmental policies (Adelle and Russel, 2013; Runhaar et al., 2014).

Over the last decades, a wide variety of tools have been developed that facilitate or promote the integration of environmental objectives into policies, planning, and practices. In this paper, I aimed to explore their effective use, organised round the question of "what works where". First, I defined the key concept of 'integration' and identified various forms and levels of integration. Second, I identified and categorised different types of integration tools. These two steps provided conceptual clarity regarding the 'what' and 'works' parts of the organising question of this paper. Third, I made an inventory of evaluations and experiences with these tools (addressing the organising question as a whole). My main conclusion is that integration tools can help promoting more integrated policies, plans, and practices, but that their performance usually is modest, which implies that expectations should be realistic. Mandatory tools and economic tools seem more effective than voluntary ones. However, this may come at the expense of their legitimacy. Political support for and legitimacy of tools therefore seem important preconditions for the selection and implementation of less voluntary tools ('where'). Also related to the 'where' part of the question, is the observation that the effective use of tools is rather context-specific. But perhaps the most critical condition seems to be the monitoring of how tools are used and what they achieve, on a voluntary basis in the case of tools such as environmental indicators or planning tools, or in terms of follow-up and enforcement in the case of mandatory tools such as EIAs, SEAs and environmental permit requirements.

I have included a broad range of integration tools, developed by both public and private actors, aimed at integration a wide variety of environmental objectives. Whereas on the one hand this allows for bringing lessons and findings together, on the other hand it may hide intrinsic differences between these tools as well as more nuanced differences in the performance of different variants of tools (e.g. EIA versus SEA). I therefore recommend that the above findings are further tested and refined in comparative analyses of tools, concentrating on specific environmental objectives and their characteristics (e.g. their complexity, certainty of knowledge basis, and degree to which they are contested) and/or specific sectors. Such an analysis may also provide a better understanding of the 'where' part of the "what works, where" question: what tools are selected for what particular integration issues and sectors, which can tell us more about the *appropriateness* or *feasibility* of tools – something that was not explicitly addressed in this paper.

Acknowledgments

This paper is based on a paper and presentation commissioned by the Division for Sustainable Development, UN Department of Economic and Social Affairs.

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