IPBES Global Assessment Chapter 6 – Supplementary materials

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6.1 Towards transformative governance

6.1.1 Market-based instruments (MBI)

In the past two decades, there has been a notable shift towards economic, financial and/or market-based policies (often titled "market-based instruments" or MBI) for the conservation and provision of biodiversity and NCP. MBI generally refers to policy tools that involve either mechanisms for changing the price structure in a given market (with the intention to acknowledge and internalize environmental costs or benefits), for creating new commodities derived from environmental features (new markets and products) or for setting up monetary transfers aiming to improve the environmental performance of the targeted agents. These include a broad range of instruments and incentives (e.g. see http://oe.cd/pine), such as tradable permits, eco-certification, biodiversity compensation, payments for ecosystem services, and according to some definitions, even taxes and subsidies (Froger et al., 2015; Pirard, 2012a). Stakeholders using these economic instruments often tend to assume that in one way or another, a monetary value can be established for biodiversity protection measures (either the cost of the action or policy itself, or a value for the underlying NCP being conserved), and that this valuation can be used to leverage positive conservation behavior in some form (Laurans, Rankovic, Billé, Pirard, & Mermet, 2013; Posner, McKenzie, & Ricketts, 2016). Relying solely on monetary value can raise controversy, including on the kind of valuation methods used and their assumptions, the data and accuracy requirements, and the areas of application (Costanza et al., 2017); IPLC groups in particular have noted that economic valuation is not always compatible with their worldviews. Other values, such as relational values (Chan, Satterfield, & Goldstein, 2012), associated with social coherence and other non-material values (i.e. spiritual and cognitive values), as well as those associated with the benefits to be enjoyed by future generations (e.g. bequest value) are examples. These arguments highlight the complexity of identifying suitable value indicators to evaluate whether economic instruments are effective or not as policy tools to motivate behavior and other change. MBI might induce changes not only in the economic performance and behavior of the targeted agents, but also in the social structures and conventions that frame the interaction among those agents, as well as in the type of cognitive framing shaping their relationship with the natural environment. Hence, both co-benefits and unexpected negative impacts have to be considered when designing and implementing MBI (R. Greiner & Stanley, 2013), as these instruments cannot be detached from their social dimensions (R. Greiner, 2013). The functioning or effectiveness of MBI should not be taken for granted, since their performance is very much conditioned by the regulatory framework in which they are inserted (Filoche, 2017). MBI normally are part of broader policy mixes, in which regulations play a very important role (Russi, Margue, Oppermann, & Keenleyside, 2016).

6.1.2 Rights-based approaches

Human rights based approach: States have obligations to effectively protect against environmental harm that interferes with the enjoyment of human rights including a duty to protect against environmental harm from private actors (J. Knox, 2013, 2017; Jhon Knox, 2018). However, there has been a rise of persecution and killings of environmental defenders around the world (Global Witness, 2017). Decision makers at the national level are often unable to strike a balance between environmental protection and other legitimate societal goals, leading to inconsistent decisions and infringements of human rights (J. Knox, 2013, 2017). A human right based approach is well established within the work programme of the UN agencies (United Nations, 2003), and has recently been used to formulate a proposed Declaration on Human Rights and Climate Change, which would use legal arguments for the intrinsic rights of nature and future generations to motivate climate action now (K. Davies et al., 2017). While corporate actors are increasingly incorporating human rights considerations within their business commitments (GRI, 2001; United, 2011), the human rights performance of corporate actors varies based on their size, revenues and geographic location (CHRB, 2017).

Land rights: Land rights include land ownership rights and land-use rights. Land ownership may be vested with the state or private landholders, and such ownership can create tensions where the allocation of land rights to commercial operators or powerful local actors impinges on IPLC rights. In many countries, subsoil resources are owned by the government, which has the power to allocate such rights to extractive industries. National law may formally protect the local land rights, but they are often construed as conditional use rights, rather than ownership. Unclear land-use rights as well a lack of collaboration among stakeholders often result in unsustainable land uses and power asymmetries (Kashwan, 2016; Lawry et al., 2017; I. Ring et al., 2018; Robinson et al., 2017; Willemen et al., 2018). Collaborative arrangements, based on land use rights, between different stakeholders (e.g., community and forest industry) have played a crucial role in conflict management (Holden, Otsuka, & Deininger, 2013; Naughton-Treves & Day, 2012; I. Ring et al., 2018).

Customary rights: Customary rights promote adaptive governance including the fair and equitable sharing of benefits from biodiversity and ecosystem services (IPBES Secretariat, 2015). Customary rights arise from '[l]aw consisting of commonly repeated customs, practices and beliefs that are accepted as legal requirements or obligatory rules of conduct' (IPBES, 2017). If fully incorporated into national law, norms, values, habits, practices and traditions become part of enforceable customary right (I. Ring et al., 2018; Willemen et al., 2018).

Rights of nature: The rights of nature reflect a transition from a juridical anthropocentric orientation to more ecocentric approaches (IUCN, 2016). Policy options include co-management of a resource (e.g., *Te Awa Tupua Whanganui River Claims Settlement Act* 2017 in New Zealand that grants the Whanganui River legal personhood), and integration of the rights in the national Constitution (e.g., the 2008 Constitution of Ecuador codifies the rights of nature) or national legislation (e.g., The Bolivian Law of the Rights of Mother Earth, 2010) (Kauffman & Martin, 2017; Rogers & Maloney, 2017) and local policies (Sheehan, 2015). Rights of nature are still evolving as only a few policies treat human as subjects equal with nature and implementation of such right remains a challenge (UKELA, 2009) (See also "Governance recognizing the role of IPLCs and ILK" below).

Animal rights: Discourse and practice on animal rights are strongly influenced by normative thinking on interspecies justice originating in political philosophy (Garner, 2008; Godlovitch, Godlovitch, & Harris, 1974; Kymlicka & Donaldson, 2011; Nussbaum, 2006; Regan, 1983; Schlosberg, 2007; Singer, 1987). These accounts do not plead for legal equality with human beings but for animals' right to self-determination and an obligation to protect animals, especially those with a consciousness. Moral arguments build the basis for the animal rights movement and for activism on animal welfare (Dawkins, 2015; Einwohner, 1999; Fraser, 2009; Regan, 1983; Singer, 1987; Whay, 2007; Wrenn, 2013). Both have influenced legislative changes, e.g., New Zealand guaranteeing basic rights to great apes species in 1999 and Germany guaranteeing rights to animals in its Constitution in 2002 – often without major practical implications. As seen above, recent developments show that ecosystems, including non-human species, are granted legally enforceable rights (e.g., Colombia's Atrato River) (Biggs, Goldtooth, & Lake, 2017; D. R. Boyd, 2017; Cullinan, 2003; Stone, 1972), usually involving a guardian making decisions for the benefit of the ecosystem (Sanders, 2017).

6.1.3 Governance recognizing the role of IPLCs and ILK

The conservation of a substantial proportion of the world's biodiversity and NCP largely depends on the customary institutions and management systems of IPLCs (Gorenflo, Romaine, Mittermeier, & Walker-Painemilla, 2012; Maffi, 2005; Renwick et al., 2017). There is well-established evidence that IPLCs can develop complex, sophisticated, innovative and robust institutional arrangements for successfully governing the management of watersheds, coastal fisheries, forests and grasslands and a variety of biodiversity-rich landscapes around the world (Arun Agrawal, 2001; Basurto, Bennett, Weaver, & Dyck, 2013; Berkes, 1999; Bodin & Crona, 2008; Colding & Folke, 2001; Fernández-Llamazares et al., 2016; Gadgil, Olsson, Berkes, & Folke, 2003; Lu, 2001; Ostrom, 1990; P. Pacheco, Barry, Cronkleton, & Larson, 2008; C. Stevens, Winterbottom, Springer, & Reytar, 2014; Toledo, 2001; Waylen, Fischer, McGowan, Thirgood, & Milner-Gulland, 2010) to govern their land- and seascapes in ways that align with biodiversity conservation (Blackman, Corral, Lima, & Asner, 2017; E. J. Ens, Daniels, Nelson, Roy, & Dixon, 2016; Emilie J. Ens et al., 2015; Inuit Circumpolar Council, 2008, 2010; Schleicher, Peres, Amano, Llactayo, & Leader-Williams, 2017; C. Stevens et al., 2014; Trauernicht, Brook, Murphy, Williamson, & Bowman, 2015; Vierros, 2017).

These customary institutions and management systems are based on ILK and encoded in complex cultural practices, usufruct systems, rituals, spiritual beliefs, taboos, sanctions, and principles of stewardship ethics (Berkes, Colding, & Folke, 2000; Bhagwat & Rutte, 2006; Bobo, Aghomo, & Ntumwel, 2015; Borrini-Feyerabend & Hill, 2015; Gadamus & Raymond-Yakoubian, 2015; Jones, Andriamarovololona, & Hockley, 2008; Parotta & Trosper, 2012; Rudel, Bates, & Machinguiashi, 2002; Ruiz-Mallén, Fernández-Llamazares, & Reyes-García, 2017; Anders H. Sirén, 2017; Anders Henrik Sirén, 2006; Von Heland & Folke, 2014). The customary institutions of IPLCs to a large extent acknowledge the connection between nature and people in an integrated manner (Borrini-Feyerabend, 2010; C. W. Chen & Gilmore, 2015; Parotta & Trosper, 2012), based on relational values (Clark & Slocombe, 2009; Jeeva, Mishra, Venugopal, Kharlukhi, & Laloo, 2006; Pierotti, 2011; Samakov & Berkes, 2017), kinship-oriented philosophies (Aniah & Yelfaanibe, 2016; Bird, 2011; Salmón, 2000) and a powerful stewardship ethics (Gammage, 2011; Kohn, 2013).

After a long history of conservation conflicts (Arun Agrawal & Redford, 2009; Brockington & Wilkie, 2015; Curran et al., 2009; Dowie, 2009; Gauthier & Pravettoni, 2017; M. J. Goldman, 2011; West, Igoe, & Brockington, 2006), IPLCs and ILK are increasingly recognized in biodiversity governance and at multiple levels and scales (E. S. Brondizio & Le Tourneau, 2016; Forest Peoples Programme, International Indigenous Forum on Biodiversity, & Secretariat of the Convention on Biological Diversity, 2016; Nasiritousi, Hjerpe, & Linnér, 2016; Schroeder, 2010; Tengö et al., 2017; Wallbott, 2014), and the importance of mainstreaming IPLC issues in global environmental policy, including the integration of ILK, has been firmly asserted in international conventions such as CBD and UNFCCC (E. S. Brondizio & Le Tourneau, 2016; Forest Peoples Programme et al., 2016; Tengö et al., 2017). As a case in point, Article 8j of CBD explicitly recognizes the unique role of IPLCs in conserving life on Earth (Convention on Biological Diversity, 1992). Such recognition is underpinned by the UN Declaration on the Rights of Indigenous Peoples (Gavin et al., 2015; S. Stevens, 2010). Yet, the contributions of ILK and IPLCs to the SDGs and the Aichi Targets are still often overlooked in environmental decision-making bodies (Berkes, 2007, 2009a; Castleden et al., 2017; Fox et al., 2017; Gavin et al., 2015; R. Hill et al., 2015; S. Jackson, 2011; Martin et al., 2016; Mistry, Bilbao, & Berardi, 2016; Shrubsole, Walters, Veale, & Mitchell, 2017; Sunderlin et al., 2014; Welch, Brondízio, Hetrick, & Coimbra, 2013). In particular, non-indigenous local communities still confront the challenges of low representation and visibility (Fontana, 2014; Hooker, 2005; Penna-Firme & Brondizio, 2007).

Granting IPLCs with appropriate recognition and policy support can serve a purpose in maximizing IPLC contributions to biodiversity conservation and sustainability (Borrini-Feyerabend, 2010; Fox et al., 2017; Gavin et al., 2015; Pretty et al., 2009; Richardson & Lefroy, 2016; Smyth, 2015; Tipa, 2009; Ziembicki, Woinarski, & Mackey, 2013), but only when supported by robust, participatory and inclusive decision-making processes (D. B. Bray et al., 2008; D. Bray, Duran, & Molina-Gonzalez, 2012; Buntaine, Hamilton, & Millones, 2015; Kerekes & Williamson, 2010; Kothari, Camill, & Brown, 2013; Kothari, Corrigan, Jonas, Neumann, & Shrumm, 2012; Mooney & Tan, 2012; Ojha, Cameron, & Kumar, 2009). More inclusive approaches towards knowledge co-production in decision-making can enhance the legitimacy and effectiveness of environmental policies (E. S. Brondizio & Le Tourneau, 2016; Sterling et al., 2017; Wehi & Lord, 2017). As such, collaborative partnerships between scientists, policy makers and IPLCs can guide innovative ways of conceptualizing and advancing towards sustainability (Athayde, Silva-lugo, Schmink, Kaiabi, & Heckenberger, 2017; Austin et al., 2017; Dobbs et al., 2016; S. E. Jackson et al., 2014; Mistry & Berardi, 2016). Yet, the substantial contributions that IPLCs are making to achieve the Aichi Targets and the SDGs are still largely overlooked in environmental decision-making bodies (Castleden et al., 2017; S. Jackson, 2011; Mistry et al., 2016; Shrubsole et al., 2017; Welch et al., 2013).

IPLCs are increasingly represented in global environmental policy platforms (Nasiritousi et al., 2016; Schroeder, 2010; Wallbott, 2014), although more robust, participatory and inclusive decision-making processes are still largely called upon in order to develop and meet the next generation of biodiversity targets at multiple levels (Berkes, 2007, 2009b; Fox et al., 2017; Gavin et al., 2015; R. Hill et al., 2015; Martin et al., 2016; Sunderlin et al., 2014). In particular, non-indigenous local communities still confront the challenge of low representation and visibility in global environmental governance (Fontana, 2014; Hooker, 2005; Penna-Firme & Brondizio, 2007). Enhanced participation of IPLCs in environmental agencies can advance the recognition of the social, spiritual and customary values of IPLCs in environmental management decisions, thereby strengthening their legitimacy (Brugnach, Craps, & Dewulf, 2014; Redpath et al., 2013; Schroeder, 2010; Wallbott, 2014). Such recognition is underpinned by the UN Declaration on the Rights of Indigenous Peoples (Gavin et al., 2015; S. Stevens, 2010). Global policy arenas such as IPBES and CBD can facilitate knowledge co-production for enhanced environmental governance (Forest Peoples Programme et al., 2016; Tengö, Brondizio, Elmqvist, Malmer, & Spierenburg, 2014; Tengö et al., 2017; Turnhout, Bloomfield, Hulme, Vogel, & Wynne, 2012). Long-term capacity development, empowerment and continued funding support are critical conditions to ensure IPLC involvement in biodiversity conservation, including specifically women, youth and non-Indigenous communities (Thomas M. Brooks, Wright, & Sheil, 2009; Eallin, 2015; Escott, Beavis, & Reeves, 2015; R. S. Reid et al., 2016; Reo, Whyte, McGregor, Smith, & Jenkins, 2017; Ricketts et al., 2010).

Increased participation of IPLCs in environmental policy circles has resulted in a wealth of new approaches and tools to support discussion of collaboration, co-management and power sharing around conservation initiatives, for reasons of social justice and more inclusive biodiversity governance (E. S. Brondizio & Le Tourneau, 2016; Fernández-Llamazares & Cabeza, 2017; S. J. Green et al., 2015; Pert et al., 2015; Schreckenberg, Franks, Martin, & Lang, 2016; Zafra-Calvo et al., 2017). These include sets of IPLC-led codes of ethical conduct in conservation (e.g., Akwe: Kon Guidelines and The Tkarihwaié:ri Code of Ethical Conduct; (Convention on Biological Diversity, 2004, 2011) and tools for dialogue such as the Whakatane Mechanism (Freudenthal, Ferrari, Kenrick, & Mylne, 2012; Sayer, Margules, & Boedhihartono, 2017), which provide a collaborative framework to ensure the full involvement of IPLCs in conservation while respecting their rights and institutions.

Moreover, new legal approaches drawing inspiration from ILK and customary institutions have had broad significance for the recognition of IPLCs rights in environmental governance (Akchurin, 2015; Archer, 2014; Humphreys, 2015; Hutchinson et al., 2014; Strack, 2017). In this vein, the laws promoting the recognition of the Rights of Nature have been, in most cases, heavily influenced by IPLC philosophies placing nature at the center of all life (Akchurin, 2015; Borràs, 2016; Díaz et al., 2015; Kothari & Bajpai, 2017). For example, the Bolivian Law of Mother Earth draws deeply on nature-embracing Andean spiritual traditions seeing Mother Earth (or *Pachamama*) as a sacred home, and it entitles nature with rights as a collective subject of interest (D. Pacheco, 2014). By the same token, the 2008 Constitution of Ecuador codifies the Rights of Nature as part of Sumak Kawsay ("Buen Vivir" or "good living"), a notion rooted in the worldview of the Quechua indigenous peoples, connoting a community-centric and ecologically-balanced approach to human development (Arsel, 2012; Gudynas, 2011; Kauffman & Martin, 2017). Similarly, by granting legal personality to the Whanganui River, New Zealand found an innovative way to honour and respect the Maori traditional worldview that sees the river as "an indivisible and living whole" (Archer, 2014; Hutchinson et al., 2014; Strack, 2017). Placing the intrinsic value of nature alongside that of human beings reflects a transition from a juridical anthropocentric orientation to more ecocentric approaches, aligning more closely with IPLC holistic conceptions of living in harmony with nature (Gordon, 2017; Kotzé & Calzadilla, 2017; Rühs & Jones, 2016). However, there is evidence that the implementation of some of these laws on the ground raises complex questions of interpretation and potential for misuse deserving attention (e.g., India; (Kothari & Bajpai, 2017).

It is well established that when governments and NGOs act to recognize and support the customary rights, knowledge and institutions of IPLCs, including specifically those of non-indigenous local communities, environmental governance becomes more inclusive and is thus strengthened (Alexander, Andrachuk, & Armitage, 2016; Berdej & Armitage, 2016; Berkes, 2009a; Brodt, 1999; J. Davies et al., 2013; Gavin et al., 2015; Robinson, Holland, & Naughton-Treves, 2014; C. Stevens et al., 2014; K. R. Young & Lipton, 2006). Recognition of IPLC ideas and customs in global laws and policies can help lay the groundwork for intercultural dialogue (Martin et al., 2016), and leverage local support to enforce conservation regulations, given that IPLCs often have strong incentives to protect their livelihood base against external pressures (Chhatre & Agrawal, 2008, 2009; Gavin et al., 2015; Gibson, Williams, & Ostrom, 2005; T. M. Hayes, 2006, 2010; Ostrom, 1990; Porter-Bolland et al., 2012; Reyes-García et al., 2012; Waylen et al., 2010). Moreover, granting land and sea rights to IPLCs is a critical means for connecting IPLCs with environmental protection policies, including economic instruments such as carbon offsets, REDD+, PES and micro-credits (de Koning et al., 2011; Duchelle et al., 2014; Gray, Bilsborrow, Bremner, & Lu, 2008; Larson et al., 2013; P. D. McElwee, 2012; Sunderlin et al., 2014; Van Dam, 2011). Yet, when governance of IPLC lands happens in isolation from the dynamics surrounding them, environmental degradation and social-environmental conflicts may ensue (Eduardo S. Brondizio, Ostrom, & Young, 2009; Le Tourneau, 2015).

Although IPLCs have a wide range of legitimate social, political, cultural and economic aspirations for their lands, which may not always align with conservation interests (Kohler & Brondizio, 2017; Sylvester, Segura, & Davidson-Hunt, 2016; Tengö et al., 2017), they increasingly chose to engage in global forums on the state and future of the planet, given their direct material and cultural ties to the environment (Brugnach et al., 2014; Redpath et al., 2013; Schroeder, 2010; Wallbott, 2014). Moreover, their management systems based on ILK often lead to sound conservation outcomes (Blackman et al., 2017; Fernández-Llamazares et al., 2016; Renwick et al., 2017; C. Stevens et al., 2014; Trauernicht et al., 2015; Vierros, 2017). As such, there are many opportunities for collaboration between conservation organizations and IPLCs to achieve mutually beneficial outcomes (Barber & Jackson, 2017; E. Ens, Scott, Rangers, Moritz, & Pirzl, 2016; Emilie J. Ens et al., 2015; Gavin et al., 2015; Painter, Duran, & Miro, 2011; Sterling et al., 2017; Thomas, 2009). Such partnerships have shown success only when based on the development of mutually agreed-upon goals through Free, Prior and Informed Consent (FPIC) and equitable benefit sharing (Crook, Douglas, King, & Schnierer, 2016; E. Ens et al., 2016; Huntington, 2017; Schwartzman & Zimmerman, 2005).

6.1.4 Transformative governance in the Arctic

Interacting forces of climate change and globalization are rapidly transforming the Arctic and awakening new interests in exploiting the region's natural resources (J. R. Bennett et al., 2015; P.A. Berkman & Vylegzhanin, 2012; Holland, Bitz, & Tremblay, 2006; Smith, 2010; Smith & Stephenson, 2013). The interconnected and complex challenges faced by the Arctic have been argued to be better addressed through transformative governance, including stronger transboundary cooperation and globally-coordinated policy responses (Aksenov, Kuhmonen, Mikkola, & Sobolev, 2014; Armitage et al., 2017; Burgass, Lowndes, Hara, Afflerbach, & Halpern, 2018; Chapin, Sommerkorn, Robards, & Hillmer-Pegram, 2015; Edwards & Evans, 2017; Hossain & Morris, 2017; A. E. Nilsson & Koivurova, 2016; Sommerkorn & Nilsson, 2015; Van Pelt, Huntington, Romanenko, & Mueter, 2017). Strategies are being sought that will promote renewed international cooperation and reduce the risks of discord in the Arctic, as the region undergoes new jurisdictional conflicts and increasingly severe clashes over the extraction of natural resources in a region that is critical to the prevision of globally important NCP (Paul Arthur Berkman & Young, 2009; Harris, Macmillan-Lawler, Kullerud, & Rice, 2018; Hussey, Harcourt, & Auger-Méthé, 2016; Keil, 2015; O. R. Young, 2010).

The Arctic Council (AC), established in 1996, is the leading intergovernmental forum promoting cooperation, coordination and interaction among the Arctic States, Arctic Indigenous communities and other Arctic inhabitants on common Arctic issues, with an overall focus on encouraging transformative change towards sustainability (Axworthy, Koivurova, & Hasanat, 2012; Bloom, 1999; A. E. Nilsson & Meek, 2016; O. R. Young, 2012). One of the main achievements of AC has been its work to recognize the Arctic as an integrative unit of governance, a distinct political region and a large, complex and dynamic social-ecological system (Koivurova & Vanderzwaag, 2007). Such recognition is serving a purpose in maximizing the fit between the rapidly changing circumstances of the Arctic and the institutional arrangements needed to steer the region towards a sustainable future (O. R. Young, 2012). Direct policy responses derived from this integrative approach is the application of new, multi-sector frameworks for integrated ecosystem management (Pinsky et al., 2018), the establishment of a circumpolar network of Protected Areas (Fredrickson, 2015; Hossain & Morris, 2017) and the proposal for the creation of a global Arctic sanctuary in the high seas (European Parliament, 2014; Greenpeace, 2014).

Part of the success of AC has been attributed to the principles of informed and inclusive governance that are explicitly endorsed in its institutional framework. Informed governance is reflected in the extensive knowledge base has been established under the auspices of the Council. To take this case in point, AC has invested heavily in the conduct of policy-relevant scientific assessments designed to document emerging issues arising in the region, to frame these concerns as items to be placed on the Arctic policy agenda, and to draw the attention of key policymakers to their significance (Kankaanpää & Young, 2012; Reiersen, Wilson, & Kimstach, 2003; Tesar, Dubois, & Shestakov, 2016). Examples of this include the influential Arctic Climate Impact Assessment (ACIA, 2004), or the Arctic Biodiversity Assessment (CAFF, 2013), both of which have achieved considerable success in generating policy-relevant knowledge about the Arctic, as well as bringing Arctic issues to the attention of global policy platforms. Several important legally binding agreements among Arctic states (e.g., Agreement on Cooperation on Marine Oil Pollution Preparedness and Response in the Arctic, Agreement on Enhancing International Arctic Scientific Cooperation) have been negotiated based on the assessments of the state of the Arctic environment (Tesar et al., 2016). Similarly, the work of AC has been key to the implementation of an international environmental protection treaty on persistent organic pollutants, with a prominent inclusion of Arctic perspectives in negotiation processes (Koivurova & Vanderzwaag, 2007).

The inclusive nature of the governance structure of the AC is best reflected by the unique formal status accorded to Arctic Indigenous Peoples as Permanent Participants, sitting at the table alongside State representatives (Bloom, 1999; O. R. Young, 2005). This has certainly contributed to increase the legitimacy of AC and to push for a new way of articulating Indigenous Peoples' participation in international policy making (Koivurova & Heinämäki, 2006). Along these lines, all assessment reports of AC have featured powerful mechanisms to ensure the inclusion of Indigenous Knowledge and expertise, such as the position of Indigenous representatives in the steering committees of the different constituencies, task forces and working groups of AC (Kankaanpää & Young, 2012).

Whether or not the features of Arctic governance are adaptive is object of caustic scholarly debate (e.g., (Axworthy et al., 2012; Kankaanpää & Young, 2012; Śmieszek & Koivurova, 2017). While some research argues that Arctic intergovernmental cooperation has been resistant to change and that AC continues to be firmly built on fixed governance fundaments (e.g., soft law nature, ad hoc funding; (Koivurova, 2010), other works claim that the rapid changes occurring in the Arctic have driven several governance adjustments and Arctic cooperation process has become more institutionalized with time (Paul Arthur Berkman & Young, 2009; O. R. Young, 2005). For instance, the scientific assessments sponsored by AC have consolidated the view of the 'Arctic in change', which has in turn energized the redrawing of Arctic policies by Arctic actors and agencies in the face of possible regime change (Koivurova, 2010).

Thinning and disappearing sea ice, melting permafrost, and circumpolar climate change, however locally and regionally varied, are commonly identified as playing their part in rapidly unsettling the geographies of Arctic governance (Hussey et al., 2016; Overland & Wang, 2013; Smith & Stephenson, 2013; Stephenson, 2018). As such, several authors and organizations have advocated for the negotiation of a harder law regime for the Arctic (Kankaanpää & Young, 2012), including firmer institutional, financial and regulatory foundations for AC (Paul Arthur Berkman & Young, 2009) and improved transboundary conservation planning (Edwards & Evans, 2017; Greenpeace, 2014; Harris et al., 2018; Hussey et al., 2016). As one of the fastest changing regions on Earth (ACIA, 2004; Cowtan & Way, 2014; Wassmann, Duarte, Agustí, & Sejr, 2011), the Arctic will face vast social-ecological challenges in the next decades, requiring all levels of governance – particularly the Arctic Council – to constantly adjust their modes of operation, ensuring a governance system that is transformative, flexible across issues and sectors, and adaptable over time (Axworthy et al., 2012; Chapin et al., 2015; Ford, McDowell, & Pearce, 2015; O. R. Young, 2012).

6.2 Integrated approaches for sustainable landscapes

6.2.1 Background

Land-use change is one of the major drivers of global biodiversity loss caused by the loss and fragmentation of species' habitats. The extent and impact of human activities over the past six decades have caused rates of species extinctions that are higher than pre-human background rates (Barnosky et al., 2011; Ceballos et al., 2015; Pimm et al., 2014). Land-use change is caused by the conversion of natural and semi-natural ecosystems, the intensification of use in agriculture and forestry, and the biological homogenization of landscapes (see e.g. (Salafsky et al., 2008).

First, the *conversion of natural and semi-natural ecosystems* is driven by the expansion of the production of food, bio-energy, fibres and other natural materials. Agriculture has already cleared or converted 70% of grassland, 50% of the savanna, 45% of the temperate deciduous forest, and 27% of the tropical forests (Foley et al., 2011). In the case of forests, roughly half of the global deforestation takes place for production of commodities, such as soy, palm oil, beef, and timber and paper. For instance, palm oil plantations for commercial purposes have been the major driver of forest loss in Southeast Asia, but vast tracts of land highly suitable for oil palm production in Africa and South America are still covered by tropical forests, mainly due to higher production costs, which have prevented an explosive expansion of oil palm production in these regions (Carrasco, Larrosa, Milner-Gulland, & Edwards, 2014). Other factors, such as unsustainable harvesting of wood and non-timber products, infrastructure and urban expansion, energy, mining, and forest fires also contribute to deforestation and forest degradation to varying degrees (FAO, 2015; UN, 2014).

Intensification of agriculture and forestry systems also result in land-use changes leading to landscape and biological homogenization (Newbold et al., 2015; Pepper et al., 2017), the decrease of the capacity of soils to maintain ecosystem services (Schiefer, Lair, & Blum, 2016; Vitousek, Mooney, Lubchenco, & Melillo, 1997), the abandonment of production in marginal areas (Rudel et al., 2005), and environmental problems associated with eutrophication and pollution of water. Critical NCP such as fostering genetic diversity in crop systems to build in resilience to cope with climate change and pest and disease outbreaks, pollination of crops and wild plants (Garibaldi et al., 2016; Kallioniemi et al., 2017; Larsen, Williams, & Kremen, 2005), control of pests (Lundin, Rundlöf, Smith, & Bommarco, 2016), organic matter decomposition and recirculation of soil nutrients (Fornara & Tilman, 2008), water purification (Lathuillière, Miranda, Bulle, Couto, & Johnson, 2017; Sun, Tong, & Liu, 2017), and the capacity to capture and store carbon, are threatened by the types of agricultural intensification that have taken place over the past 60 years.

The debate about increased yields in agriculture and forestry harmonized with biodiversity conservation considers two main competing pathways that represent different paradigms, i.e. the 'land-sparing' and 'land-sharing' pathways (Balmford, Green, & Scharlemann, 2005; Fischer et al., 2008; Phalan et al., 2016; Phalan, Onial, Balmford, & Green, 2011), see also Chapter 5, box 5.1). There is however increasing consensus in that the choice of options for sustainable land-use systems needs to consider the specific social, economic, ecological and technological context in which these options will be embedded (Tscharntke et al., 2012).

Under land sparing, land-system transitions can follow a combination of both increased food production and abandonment of marginal lands, following agricultural intensification and migration of rural inhabitants to urban areas (Rudel et al., 2005). Farming is practised at the highest sustainable yields (see bridging the 'yield gap' in chapter 5, section 5.3.2.1) and linked to retaining or restoring native vegetation elsewhere (Balmford, Green, & Phalan, 2015; Cui et al., 2018; Lamb et al., 2016). Several studies indicate that especially narrowly-distributed species, including around 1800 species of birds, trees, grasses, daisies and dung beetles, would be favoured by the land-sparing model (Balmford et al., 2015; Dotta, Phalan, Silva, Green, & Balmford, 2016; Gilroy et al., 2014; Hulme et al., 2013; Phalan, Balmford, Green, & Scharlemann, 2011; Williams et al., 2017). Land-use transitions towards land-use intensification and abandonment have taken place in the Northeastern USA and in Europe, and have created opportunities for expanding the area, and connecting networks, of protected areas, re-wilding (Ceausu et al., 2015; Fernández, Navarro, & Pereira, 2017; Navarro & Pereira, 2012) and exploring livelihood and business opportunities provided by wilderness areas.

However, it has been argued that land sparing as a general model of a land-use transition towards sustainability fails to account for real-world complexity (Tscharntke et al., 2012) and presents negative side effects that require careful consideration. Land-use intensification could be implemented through unsustainable and inequitable practices (Carrasco et al., 2014) and may be an incentive for agricultural expansion, and thereby, deforestation and/or degradation of natural systems in many contexts (Angelsen & Kaimowitz, 1999; Barbier, Burgess, & Grainger, 2010; Carrasco et al., 2014; Lambin & Meyfroidt, 2010; Perfecto & Vandermeer, 2010), especially when remaining natural habitats are not managed or protected effectively to avoid leakage effects (Ewers, Scharlemann, Balmford, & Green, 2009; Lim, Carrasco, McHardy, & Edwards, 2017), or when there are telecoupling effects of food and materials demand, for example in the case of soybean and palm oil production whose demand in Europe have driven extensive land-use transformation in South America and Asia (Carrasco et al., 2014; Lim et al., 2017).

The *biological homogenization of landscapes, crops and livestock* is a dominant feature of the past period of land-use intensification. Eighty percent of the world's arable land has increasingly been planted with a few grains and animals (wheat, corn, rice and potatoes account for 60% of the world's vegetable food sources, and 14 species of animals provide 90% of all animal protein (Altieri, Nicholls, Henao, & Lana, 2015)), narrowing the genetic diversity of global agricultural systems. This level of homogenization of crops makes food production systems particularly vulnerable to climatic events such as droughts and extreme rainfall (Altieri et al., 2015; Jarvis, Hodgkin, Sthapit, Fadda, & Lopez-Noriega, 2011; Lin, 2011; Vernooy, Sthapit, Otieno, Shrestha, & Gupta, 2017) and attacks by pathogens (Jarvis et al., 2011). The impacts from climate-related extremes, including heat waves, droughts, and wildfires, reveal significant vulnerability and exposure of many human systems to current climate variability (IPCC, 2014). Land-use transformation will need to address foreseen patterns of climate change, which include higher frequency of extreme weather events, concentrated rainfall (IPCC, 2014, p. 7), higher risks of erosion of fertile agricultural soil, floods, crop damage and losses of harvest in some areas; and longer drought spells due to higher temperatures, and, in some cases, a decrease in rainfall (C. D. Allen et al., 2010; Garreaud et al., 2017). In addition, the manner in which further intensification of agriculture is achieved will differ markedly from the past and will need to rely on new exploitation of genetic diversity, because the exploitable gap between average farm yields and genetic yield potential of crops is closing. There is increasing evidence that much of the observed gain yield during the past 50 years can be attributed to greater stress resistance of crops rather than an increase in yield potential (Cassman, 1999).

The rate of *introduction of species* to new regions has increased dramatically due to the extensive trade of agricultural supplies and products. These introductions have the potential to become invasive and difficult to control, colonize natural ecosystems and produce important ecological shifts, and turn into new pests for crops, pastures and forestry. Land-use change and species invasions are the drivers with most impact on species losses (24.8 and 23.7%, respectively), compared to eutrophication and climate change (8.2% and 3.6%, respectively) (Isbell et al., 2017). Overharvesting of forest and other resources are obstacles for sustainable management and can be driven by different reasons such as short-term economic gains or higher demand of products (Pokorny et al., 2016).

6.2.2 Feeding the world without consuming the planet

Expanding and enhancing sustainable intensification

New practices of intensification need to be developed to overcome the limitations in terms of resource-use efficiency and environmental impacts of land intensification practices over the past six decades. These include the levels of annual global flows of nitrogen and phosphorus, which are currently more than double the natural ones, while the uptake by crops is only 40-50 % (FAO & ITPS, 2015) of input amounts. Water sources have been depleted by irrigation, and other agro-chemical products cause severe problems of pollution (Siebert & Döll, 2010), affecting freshwater and marine biodiversity (Carpenter et al., 1998) and NCP, including the quality of water for drinking and recreational purposes (Foley et al., 2011). In some instances, agrochemicals (i.e. loads of nitrogen and phosphorus) have exceeded the safe operating limits for humans (Rockström et al., 2009). The use of agrochemicals also affects soil biodiversity and the NCP soils generate, but there are large knowledge gaps on these impacts (FAO & ITPS, 2015). Over-use of nitrogen chemical fertilizers also accelerates decomposition of soil organic matter, leading to further soil degradation in over-fertilized soils ((Ju et al., 2009; Tian et al., 2011) cited in (FAO & ITPS, 2015)). In addition, over-fertilization has impacts on global climate through the decomposition of organic matter, which increases emissions of GHGs, and through the release as nitrous oxide (N2O), a gas that has 300 times the warming power of carbon dioxide (FAO & ITPS, 2015).

Sustainable intensification of agriculture has been defined as a form of agricultural production where "yields are increased without adverse environmental impact and without the cultivation of more land" ((The Royal Society, 2009), in (Schiefer et al., 2016)), and should minimize uses which cannot be reversed within 100 years or four human generations (e.g. like sealing, excavation, sedimentation, severe acidification, contamination, and salinization (Schiefer et al., 2016). Practices can enhance resource efficiency (nutrient and water retention in crops), minimize nutrient leakage and soil erosion, and reduce the use of pesticides to avoid locking food production systems in agrochemical dependency and vulnerability. The adoption of these practices can be supported by investment in technological development and outreach, regulations, and public and private quality standards such as voluntary certification schemes and roundtables.

Alternatives for sustainable agricultural intensification include different forms that aim to bridge yield gaps ((Mueller et al., 2012), see chapter 5, section 5.3.2.1) by increasing resource use efficiency (Duru et al. 2015) and are generally in line with the 'land-sparing' paradigm. For instance, *precision agriculture* is a form of agricultural intensification aiming to reduce nutrient leach and increase crop uptake, which depends on sophisticated management of soil and water resources and applied inputs, based on remote sensing and information technology. This kind of site-specific management is most relevant to large-scale agriculture in which field size and within-field variation are great enough to justify the cost of needed equipment (Cassman, 1999). In most developed countries, this technology is presently available. *Vertical farming* is an extreme form of agricultural intensification that is detached from the agroecological system and uses indoor farming techniques, controlled environments and technology. This form of agricultural intensification requires high levels of investment and technology.

Policies that promote land-use intensification need to address leakage effects. For instance, in cases like the expansion of oil palm cultivation there is a danger that improvements in tropical crop yields will further transfer agricultural production from temperate to tropical regions, leading to more tropical deforestation and thereby limiting the social and ecological benefits of high-yielding varieties (Carrasco et al., 2014). The land-sparing model also fails to address the conservation of semi-natural grasslands and other open habitats that require management to be maintained (P. F. Donald, Green, & Heath, 2001; Regina Lindborg et al., 2008; Milestad, Ahnström, & Bjrklund, 2011). Direct payments to farmers (e.g. agri-environmental schemes), combined with PAs (e.g. the European Natura 2000 network), habitat restoration and re-wilding are options to support the protection of open habitats.

Encouraging ecological intensification and sustainable use of multi-functional landscapes

Land-use systems consisting of mosaics of cropland, grasslands and pastures, and forests, are wide-spread globally. They form landscapes, which in some cases include protected areas of natural and semi-natural systems for biodiversity conservation (Regina Lindborg et al., 2008; Takeuchi, 2010). These mixed systems of intensive and extensive forms of land use are critical for food security and rural livelihoods (Herrero, Thornton, Gerber, & Reid, 2009; Tscharntke et al., 2012), can harbour considerable biodiversity (Bengtsson et al., 2003), are critical to the long-term survival of species and ecosystem services (Andersson et al., 2015; R. Lindborg, Plue, Andersson, & Cousins, 2014; Regina Lindborg et al., 2008), can store significant amounts of carbon, especially in agroforestry and silvo-pastoral systems and rangelands (Herrero et al., 2009; Montagnini, 2017), and fill many functions pertaining GQL (Milestad et al., 2011), including help build resilience of food and material production systems (Knoke et al., 2016). Livestock production is a key component in many mixed land-use systems as well, and the largest land-use system globally, occupying 70 percent of the agricultural area (FAO & ITPS, 2015). The food and materials production capacity of mixed systems can be enhanced with improved agriculture and sylvicultural practices and ecological intensification (Bommarco, Kleijn, & Potts, 2013; Doré et al., 2011; Tittonell, 2014; Tscharntke et al., 2012).

Improving certification schemes and organic agriculture

Organic certification in agriculture is based on the international regulation of organic farming by the Codex Alimentarius Guidelines (established by the FAO and the WHO) and the International Federation of Organic Agriculture Movements' (IFOAM) Basic Standards (Tuomisto, Hodge, Riordan, & Macdonald, 2012). IFOAM Accreditation (IFOAM https://www.ifoam.bio/en/ifoam-accreditation-program) is the only fully international accreditation program for certification bodies active in organic agriculture. Standards vary in the scope of application (i.e. one country to international), but national standards tend to be stricter than the IFOAM Basic Standards (Tuomisto et al., 2012). Most current schemes encourage soil and water conservation and regulate agro-chemical use with the aim of protecting both workers and ecosystems (Tayleur et al., 2017). The governance system of certification schemes varies among countries.

The total cropland area covered by certification schemes is low. By 2012, it averaged 1.1% but covered considerable area of tropical commodities, i.e. 23% of all coffee production areas and 15% of all cocoa areas (Tayleur et al., 2017). These patterns have been driven mostly with consumer, company, and civil society demand for certified products in developed countries, but there are fewer incentives for farmers in developing countries to adopt certification for domestic crops (Tayleur et al., 2017). Recent trends show increasing adoption of sustainable behaviours of consumers (e.g. a survey including 30 000 respondents from 60 countries show that 66% consumers indicate that they are willing to pay more for certified products, an increase from 55% in 2014) (The Nielsen Company, 2015), which could lead to considerable effects on supply chains, including those of widely traded tropical crops (e.g coffee, bananas, tea, cocoa).

Early *voluntary certification schemes (VCS)* focused primarily on environmental standards (J. Potts et al., 2014), but in the past decades, and particularly in the case of certification of tropical food commodities, certification programs have broadened to include other criteria (e.g. GlobalGAP 1997, Rainforest Alliance/SAP 1987, UTZ Certified 2002, RSPO, etc) (DeFries, Fanzo, Mondal, Remans, & Wood, 2017), including social and environmental sustainability such as human health, environment, fairness, and adoption of technology (DeFries et al., 2017; Tuomisto et al., 2012). Several tropical crops (e.g coffee, bananas, tea, cocoa) are widely traded commodities, and certification programs could have a large effect on international supply chains. A recent comprehensive review shows that the schemes can sometimes play a role in meeting the SDGs, but that the programs are not a panacea to improve social outcomes or overall incomes of smallholder farmers (DeFries et al., 2017). The lack of social outcomes can be attributed to the fact that voluntary certification schemes tend to privilege industrial interests and the most resourceful farmers and exclude smallholder producers. Although VCS may have strong standards, implementation, and verification mechanisms, the evidence about adoption and level of impact on biodiversity and livelihoods is limited. This is to a large extent due to the many different criteria set in the standards, which make it difficult to compare outcomes among cases and challenges of confounding factors (Milder, Newsom, Lambin, & Rueda, 2016). In order to maintain market demand and the scope of governing authority, market-based environmental certification schemes need to balance competing needs for rigor and transparency of standards, legitimacy of rule-making procedures, and accessibility for clients (S. R. Bush, Toonen, Oosterveer, & Mol, 2013). Legitimacy is often constructed through standard-setting processes that are open and democratic and incorporate the input of a variety of actors, including NGOs, scientists, consumers, public stakeholders as well as industry actors, which generates concerns about the possibility for the assertion of business interests in the development of standards to the detriment of environmental rigor or marginalized actors (Eden, 2009; Hatanaka, Bain, & Busch, 2005; Havice & Iles, 2015; D. Klooster, 2010).

Over the past years, comprehensive consumer surveys show that many consumers have adopted sustainable behaviors, although with varying degrees of commitment, and sales of brands with sustainability commitments have grown. These trends open up opportunities to companies to rise above the competition if they can position their brand as sustainable in their markets (The Nielsen Company, 2015), and thus provide a basis for certification. However, the efficacy of environmental certification also depends on whether there is indeed demand for certified products. While there is evidence of demand for certified products in some markets, for example for coffee (Raynolds, Murray, & Heller, 2007) and some fish (Roheim, Asche, & Santos, 2011; Teisl, Roe, & Hicks, 2002), price premiums and improved market access for clients are elusive for other products such as forest products and many fisheries (Anderson & Hansen, 2004; Anderson, Laband, Hansen, & Knowles, 2005; D. J. Klooster, 2006), despite evidence that consumers state they are willing to pay a premium for environmentally sustainable certified products (Forsyth, Haley, & Kozak, 1999; Jensen, Jakus, English, & Menard, 2004; Vlosky, Ozanne, & Fontenot, 1999). One challenge is that consumers may not fully understand what certification labels indicate about the product (Eden, 2009; Yiridoe, Bonti-Ankomah, & Martin, 2005) or are confused by the multiplicity of standards and labels on the market (Font, 2002).

Improving agri-environment schemes (AES)

Agri-environment payments encourage farmers to adopt agricultural activities or levels of production intensity that deliver positive environmental outcomes, while not being necessarily the first choice from the point of view of profitability. They have particularly been applied to the inclusion of environmental concerns into the EU Common Agricultural Policy (CAP). These are generally voluntary schemes aiming at providing incentives to farmers to conserve and better provision ecosystem services on their individual farmlands. Examples of AES are: 1) environmentally favourable extensification of farming, 2) management of low-intensity pasture systems, 3) integrated farm management and organic agriculture, 4) preservation of landscape and historical features such as hedges, ditches and woods, and 5) conservation of high-value habitats and their associated biodiversity (EC Agriculture and rural development, https://ec.europa.eu/agriculture/envir/measures_en, accessed 2018/10/23).

The majority of European farmers are dependent on agricultural subsidies from EU (Jakobsson & Lindborg, 2015), which is mostly regulated by the CAP as well as complementary national legislations. In the last 25 years several CAP reforms have been undertaken, including the last major changes in 2013 which aimed at: 1) viable food production; 2) sustainable management of natural resources and climate action; and 3) balanced territorial development (European Commission, 2013). The main financial instruments within CAP are direct payments under Pillar 1 (basic and greening payments), and rural development payments under Pillar 2, the latter one also including agri-environmental payments (European Commission, 2013). For a detailed discussion on the structure and functioning of CAP, please check the IPBES European and Central Asia Regional report, chapter 6, section 6.5.2 (I. Ring et al., 2018).

In the EU CAP, farmers are required to make a five-year obligation to use environmentally friendly farming practices (for example, conservation set-asides, organic agriculture, low-intensity systems, integrated farm management; preservation of landscape of high-value habitats and biodiversity), and they receive payments to cover the cost of these enhancements or income lost from doing so. These types of agri-environment measures can be applied at multiple scales (there are examples at national, regional, and local levels) and across different types of ecosystems and farming systems, making them a flexible approach. The literature points to key challenges in defining the required standards for compliance that are aligned with effective conservation of farmland biodiversity. AES in general, and agri-environmental payments of the CAP in particular, are reported to have only a moderate positive impact on biodiversity (e.g., (Capitanio, Gatto, & Millemaci, 2016; D. Kleijn et al., 2006; Overmars, Helming, van Zeijts, Jansson, & Terluin, 2013; Primdahl, Peco, Schramek, Andersen, & Oñate, 2003; Whittingham, 2011), and evidence has also been found for perverse effects of such payments. For instance, the regulation limiting the number of trees on woody pastures is reported as contributing to decreasing diversity (Jakobsson & Lindborg, 2015). Others highlight that in case of organic farmers, agrobiodiversity might decrease if farmers decide to plant only subsidized species (Nastis, Michailidis, & Mattas, 2013). The limited effectiveness of AES, which can be traced back to policy and management failures as well as ecological and socio-economic contextual factors, asks for improvements (Babai et al., 2015). General recommendations, derived from published scientific literature (see e.g. (Batáry, Dicks, Kleijn, & Sutherland, 2015), include among others: 1) applying more result-oriented schemes (Burton & Schwarz, 2013); 2) providing better guidance of application to farmers and locating AES in landscape with higher biodiversity (Whittingham, 2011); 3) integrating effects on non-farmed habitats (Paul F. Donald & Evans, 2006); and 4) differentiating objectives according to common and rare species (D. Kleijn et al., 2006).

Regulating commodity chains

The Brazilian Soy Moratorium provides an example of regulated commodity chains in tropical mid-income countries under strong land conversion pressure to produce agricultural commodities. In 2014, 306.5 million tons of soy were produced and 28% was from Brazil, the second biggest soy producer and the biggest exporter in the world (FAO, 2017). From the early 2000s, with the implementation of agricultural techniques and varieties adapted to Amazonian conditions, farmers took advantage of cheaper land prices to expand soy plantations. This new condition pushed other land uses, especially pastures, to new areas, creating indirect pressure for forest conversion (D. C. Nepstad, Stickler, & Almeida, 2006). In 2006, after the release of the Greenpeace's report "Eating up the Amazon", which highlighted the acceleration of the forest destruction and linked the supply chain with illegal activities (Greenpeace, 2006), civil society and the private sector engaged in negotiations that culminated in the first voluntary zero-deforestation agreement for tropical forests (Gibbs et al., 2015; Rudorff et al., 2011). The Soy Moratorium was signed in July 2006 with the purpose of halting deforestation associated with soy production in the Amazon biome, which represents more than 4 million km^2, or half of the Brazilian territory. The biggest traders operating in the country stopped purchasing soy from new deforested areas, pushing the farmers to compliance. In 2008, the federal government joined the agreement and started to validate the satellite monitoring data.

This agreement came up with a set of public policies and other initiatives that lowered deforestation in the Brazilian Amazon by 70% (Arima, Barreto, Araújo, & Soares-Filho, 2014; Assunção, Gandour, & Rocha, 2015; D. Nepstad et al., 2014). The last report of the Soy Working Group, the governance structure behind the Moratorium, pointed to 99% compliance with Moratorium terms in the municipalities covered by the agreement, which means trading soy only from areas deforested up to 2008 (GTS, 2016). Despite the good results, there are still threats to the moratorium, especially related to market shares, including the lack of engagement of other traders and importers and the competition with farmers not covered by the moratorium. While these threats may demise the motivation of the private sector in keeping the agreement, there is an opportunity to grow soy production in degraded pasture areas without increasing deforestation. Annual crops represent only 6% of the deforested area in the Amazon, while pastures represent 65% (Almeida et al., 2016). Combined with the identification of suitable areas, pasture intensification techniques and controlling new deforestation, the soy supply chain in the Amazon may become a good example of reconciliation of forest conservation and agricultural production. There are leakage risks due to Moratorium restrictions (Arima, Richards, Walker, & Caldas, 2011), but recent analysis is showing no evidence for this (le Polain de Waroux et al., 2017). The Soy Moratorium also set the stage for other initiatives to improve the sustainability of soy production and raise the awareness of the markets, like the Round Table on Responsible Soy (RTRS) and the Soja Plus Program. These initiatives are additional to zero-deforestation agreements and include other issues related to environmental compliance, social justice and economic viability at the farm and the supply chain level.

Managing large-scale land acquisitions (LSLA)

Concerns about LSLA (also sometimes called "land grabbing") have increased considerably over the past decade (Balehegn, 2015; Borras, Suárez, & Monsalve, 2011), including issues of food security, equity, environmental impacts and leakage effects. LSLA encompass expansion of industrial tree plantations; state-led investments in land purchasing or renting in other continents (dominated by, but not limited to, acquisitions in Africa); rapid expansion of lands used for biofuels following an EU biofuel mandate and the US Renewable Fuel standard; rapid growth in demand for cash crops like palm oil and expansion into more marginal lands; and large-scale acquisitions for industrial-scale food production (Aguilar-Stoen, 2016; Baglioni & Gibbon, 2013; Borras et al., 2011; Hofman & Ho, 2012; B. White & Dasgupta, 2010; Zoomers, 2010).

There is inconclusive evidence on whether the food price crisis of 2008 was linked to expanding biofuel land deals (R. Bush & Martiniello, 2017; Kugelman & Levenstein, 2009). A recent meta-analysis has shown that undernourished areas tend to export more "embodied agricultural lands" in foodstuffs for trade than they import (Marselis, Feng, Yu, Teodoro, & Klaus, 2017). Because many land investments in African countries exceed the documented cultivable land area for the country, many of these LSLA are likely expanding cultivation into forest, wetlands and grasslands, which will have negative impacts on biodiversity and NCP (Balehegn, 2015; D'Odorico, Rulli, Dell'Angelo, & Davis, 2017). Water demands for intensification of LSLAs are also likely to increase with impacts on other users of water (Lazarus, 2014).

6.2.3 Managing multifunctional forests

Improving use of Payments for Ecosystem Services (PES)

PES originally developed from calls to use direct payments to resource users for the adoption of biodiversity-friendly practices as a strategy to tackle the drawbacks of "indirect" strategies for nature protection, like integrated conservation and development projects (P. J. Ferraro & Kiss, 2002). Such call was based on the argument that by means of creating direct economic incentives to agents, such payments would help to solve "markets failures" - namely biodiversity loss or the under-provision of ecosystem services – in a more cost-effective way, in comparison to any indirect approach (P. Ferraro & Simpson, 2002). PES schemes have rapidly expanded in size and scope in the past 20 years.

An early "market-based" definition of PES emphasized that these should be voluntary economic transactions between buyers and sellers of a well-defined environmental service, through a conditional payment (Derissen & Latacz-Lohmann, 2013; Engel, Pagiola, & Wunder, 2008; Engel & Palmer, 2008; Sven Wunder, 2007). However, subsequent research has shown that most cases from the real world do not match this theoretical definition, and has called for acknowledging the great variety of institutional arrangements, scales and scope of projects and policies that commonly fall under the PES label (Muradian, Corbera, Pascual, Kosoy, & May, 2010; Pirard, 2012b; Van Noordwijk et al., 2012; Vatn, 2010). Some researchers consider ecotourism and certification forms of PES (Ingram et al., 2014), although we treat them separately in this assessment.

Ecosystems and NCP most frequently focused on by PES

Economic incentives as a conservation tool have a long history in agro-ecological areas in developed countries (such as the Conservation Reserve program in the US, the Common Agricultural Policy in the EU, or similar environmental stewardship plans in Australia and New Zealand). PES approaches have exploded in popularity particularly with regard to water and forest management in the developing countries of the global South. Currently, watersheds and forests providing water supply and carbon sequestration dominate in the PES literature, with much fewer PES for freshwater or marine, or alpine and arctic systems (Goldstein, 2015; Madsen, Carroll, & Moore Brands, 2010; Stanton, Echavarria, Hamilton, & Ott, 2010). By far the most commonly encountered PES schemes are for watershed management for water flow, quality, or flood control (Ivan Bond & Mayers, 2010; Brauman, Daily, Duarte, & Mooney, 2007; Brouwer, Tesfaye, & Pauw, 2011; Stanton et al., 2010). Forest protection for ecosystem services, including water flow, but also encompassing biodiversity conservation and carbon sequestration, comes in a close second (Madsen et al., 2010). Other environmental services protected by PES include soil erosion control (Liu, Zhou, & Hauger, 2013); wildlife conservation (Asquith, Vargas, & Wunder, 2008; Clements et al., 2010); and sustainable agro-ecological systems (J. Börner, Wunder, Borner, & Wunder, 2008; R. Yin & Zhao, 2012). There are also strong regional trends: Latin America is well advanced in PES, with Asia having a few national programs, and Africa almost none (Balvanera et al., 2012; D. E. Bennett, Gosnell, Lurie, & Duncan, 2014; Carroll, Fox, & Bayon, 2007; Egoh, Reyers, Rouget, & Richardson, 2011).

PES in different world regions and ecosystems

Latin America is the world region that has the most cases of PES schemes. Most of the schemes deal with incentives to landholders for the provision of hydrological services. The cases reflect a great diversity of institutional configurations in the field, and the predominance of state-led programs. Several studies have found that factors beyond the search for economic compensation (such as social norms, environmental values) have an important role in determining participation in PES schemes. Recent impact evaluation studies, though still limited, also report a diversity of outcomes in relation to environmental performance and social issues, such as livelihoods and equity. Comprehensive and rigorous impact assessments can be considered as still incipient. The lack of enforcement of conditionality, monitoring of ecosystem services outcomes and evaluation of impacts are reported as recurrent caveats of PES design in Latin America. Considerable knowledge gaps still remain with regards to: (a) How to address the uncertainties associated with the relationship between land use and the provision of ecosystem services (and in particular hydrological services); (b) The extent to which PES schemes are inducing additional effects; (c) How different payment modalities influence rules about the management of common pool resources; and (d) the long-term relational and behavioral implications of the payments among the involved stakeholders. The design of the next generation of PES in Latin America could incorporate the knowledge generated during the past decade, and in particular pay attention to the incorporation of equity and behavioral considerations, how to deal with environmental uncertainties and with the trade-offs that arise between pursuing ideal design principles, on one hand, and transaction costs and the need to reconcile different policy goals on the other.

The Asia Pacific region has also witnessed a sizable expansion in the number of PES schemes in recent years. Vietnam and China both have major national PES programs, while other countries in the region have regional programs (Australia), while still others have pilots at city or watershed scales (Indonesia, Thailand). The effectiveness of these programs in terms of quality of life, provisioning of services and biodiversity has been mixed. There is particularly little assessment of the effects on biodiversity in the literature. Further, key lessons are that state involvement remains necessary for functioning (most programs do not operate as markets, with the exception of Australia's Bush tender program, which uses auctions). The size of these national-scale projects varies widely: China's Sloping Land Conversion Program has over 12 million ha under contracts (M. T. Bennett, 2008; Z. Xu et al., 2006) and Vietnam protects 4 million hectares of forest under PES. There are also an increasing number of smaller-scale PES plans, often initiated by donors or conservation organizations (E. Boyd, May, Chang, & Veiga, 2007; Clements et al., 2010; Milne & Niesten, 2009; Reynolds, 2012; Sommerville, Milner-Gulland, Rahajaharison, & Jones, 2010).

PES is emerging as a promising instrument for addressing challenges to sustainable natural resource management in Africa. It has the potential to help raise new sources of sustainable finance where they are greatly lacking and improve the efficiency of conservation interventions. Nonetheless, PES schemes are not yet very well developed in Africa. The existing PES schemes cover a wide range of goals in line with the diversity of environmental services such as carbon services, biodiversity services and water services. (Namirembe, Leimona, Van Noordwijk, Bernard, & Bacwayo, 2014) reviewed 50 PES projects comprising 27 focused on carbon sequestrations and emission reduction, 17 on biodiversity conservation, 2 on watershed management, and 4 bundled ES. They found out that various mechanisms are utilized to implement these projects, which they categorized as commoditization (30%), compensation (12%) and co-investment (58%) (see Table 6.1 for examples of PES in different countries).

Out of the three common ecosystem services, carbon sequestration and emission reductions PES mechanisms are the most widespread. This is mainly due to the available international demand for this service in carbon markets and the huge potential offered by avoided deforestation in the Congo basin. Carbon PES is developing more in the moist tropical carbon-rich forest of Congo Basin and east Africa following the operationalization of REDD+ (I. Bond et al., 2009). These programs are usually implemented by NGOs or private companies trying to provide financial incentives to communities or individuals while drawing finance from carbon markets. However, as part of the national REDD+ processes, some African countries, such as the Democratic Republic of Congo (DRC), are also implementing a national PES scheme to channel REDD+ funds (AfDB, 2015). Biodiversity-oriented PES schemes have also enjoyed some development in Africa, in particular in East Africa, Southern and Central Africa. Their development has been supported by the expansion of community-based NRM and ecotourism. Though water scarcity is an important concern in Africa, payment for watershed services schemes seem to be the lesser developed type of PES in the continent. This is due in particular to the lack of demand at the local and national levels (AfDB, 2015).

Governance structures and stakeholders

PES programs are very diverse in terms of users and suppliers. Direct users, like water-consuming businesses and households, often require national or subnational authorities to serve as intermediaries to coordinate the transfer of user fees to service supplying households (Goldman-Benner et al., 2012). For many large-scale national PES projects, users or buyers are often taxpayers in general as they use general taxes, rents, or user fees on all citizens for financing (Pagiola, Zhang, & Colom, 2010), thus making some PES more akin to regulatory approaches than a true market mechanism. Many donor-supported PES projects also involve the transfer of funding and resources to service providers and do not involve direct 'users' of these services at all (Thuy T. Pham, Campbell, Garnett, Aslin, & Hoang, 2010; Sommerville et al., 2010). In most PES schemes, there remain significant roles for national and subnational governmental intermediaries, in addition to donors and NGOs, in governance (Bosselmann & Lund, 2013; Thuy T. Pham et al., 2010; Vatn, 2010). Overall, there are very few instances of direct market mechanisms that set variable prices for PES schemes in developing countries (such as through auctions) (Fletcher & Breitling, 2012; P. D. McElwee, 2012; Pirard, 2012a; Shapiro-Garza, 2013), and instead, most PES payments are determined by local or national regulations.

Table 6.1. Countries where national or subnational PES is operational

Country	Level of governance	Name	Start	Coverage	Financing	References
Brazil	Sub-national	Bolsa Floresta	2007			(Newton, Nichols, Endo, & Peres, 2012; Viana, Cenamo, Ribenboim, Tezza, & Pavan, 2008)
Costa Rica	National	Fondo Nacional de Financiamiento Forestal/ Pago por Servicios Ambimentales (PSA)	1997	~900,000 ha forest	~ US$64 to 80/ha for forest protection; ~US$200-300/ha for reforestation - funded primarily by fuel tax surcharge and donors; a few private transactions with hydropower companies	(Montagnini & Finney, 2011; Porras, Barton, Miranda, & Chacon-Cascante, 2013)
Ecuador	National	Socio-Bosque	2008	525,000 ha	$30 and below/ha funded by central govt transfers	(de Koning et al., 2011; Jean Carlo Rodríguez-de-Francisco & Boelens, 2016)
Guate-mala	National	Programa de Incentivos Forestales	1997			(Pagiola et al., 2010)
Mexico	National	Program of Payments for Environmental Services (PSAB)		2.5 mill ha degraded watershed	US$27-36/ha, funded by national water fees, central transfers and donors	(FONAFIFO, CONAFOR, & Ministry of Environment, 2012)

China	National	"Grain for Green"/Sloping Land Conversion Program		12 mill ha	US$ 20-40 equiv/ha, up to max of $600/ha in watersheds funded by central govt transfers	(R. S. Yin, Liu, Yao, & Zhao, 2013)
Vietnam	National	Payments for Forest Environmental Services	2010	4 mill ha watershed forest	US $30 and below/ha: funded by mandatory payment levels on public water and energy use	(P. D. McElwee, 2012; P. McElwee, Nghiem, Le, Vu, & Tran, 2014)

Impacts and effectiveness of PES

Overall the literature indicates that while promising, PES are not a panacea (Muradian et al., 2013). They are often expensive to set up (E. Uchida, Xu, & Rozelle, 2005) and can have high transaction costs (Alston, Andersson, & Smith, 2013); conflicts over the societal value of ecosystem services are often hard to resolve (Clements et al., 2010; Kari & Korhonen-Kurki, 2013); and projects simply may fail to reach people responsible for degradation of environmental services (Brouwer et al., 2011; Minang & van Noordwijk, 2013). A review noted that despite a voluminous literature, analysis has yet conclusively answered, "Does PES work better than no PES intervention in delivering environmental services?" (Pattanayak, Wunder, & Ferraro, 2010). A recent review has argued that while PES is one potential source for biodiversity funding, it will need to be supplemented with other non-market sources like aid and state budgets (Hein, Miller, & Groot, 2013). Concern has been expressed that PES funding may be crowding out traditional biodiversity funding, although some argue the two approaches are complementary (R. L. Goldman, Tallis, Kareiva, & Daily, 2008).

There have been few studies that have tried to compare the impacts of PES on quality of life, so this is still an area of on-going research (Bhim Adhikari & Agrawal, 2013; Mahanty, Suich, & Tacconi, 2013; Mayrand & Paquin, 2004; Tacconi, Mahanty, & Suich, 2013). Total payments in individual case studies have ranged from the order of a few dollars per household per year to as much as thousands of dollars, often dependent on land size (FONAFIFO et al., 2012; Mahanty et al., 2013). There are also many cases of PES being paid to communities rather than households, but there is no clear evidence that one method is better than another (Reynolds, 2012; Tacconi et al., 2013). There are also PES projects that do not make use of cash payments for participation, but rather provide other types of compensation and rewards (van Noordwijk & Leimona, 2010). (R. Greiner & Stanley, 2013) have pointed out that additional co-benefits are an important part of PES, such as the development of social capital and psychological benefits from participation.

Increases in household income are reported in a number of comparative studies where households have received payments (Tacconi et al., 2013), while in other cases, benefits have been mixed. Some PES projects have reported low participation rates and unequal benefit distribution (B. Adhikari, 2009; Clements et al., 2013; Schomers & Matzdorf, 2013). In some reported cases, cash income may increase due to payments, but agricultural production may decline when required land changes are made, leading to no net benefit or even losses. For example, (Yang et al., 2013) report that households in China's Sloping Land Conversion Program faced forest restrictions and crop losses to wildlife that were not compensated for sufficiently by the payments. In tree planting projects for carbon, some studies report positive household incomes. For example, converting agricultural lands to forest freed household labourers for other activities, including migrant wage labour (W. Xu, Yin, & Zhou, 2007), although other studies report that PES benefits were often captured by richer households, larger landowners, or well-connected industries (Jan Börner et al., 2010; Corbera & Brown, 2010; Lansing, 2014; Zbinden & Lee, 2005). In some PES studies, net negative results, such as restrictions on forest use and declining household food security and income, have been documented (Beymer-Farris & Bassett, 2012; Ibarra et al., 2011; Liang & Mol, 2013; Osborne, 2011), as well as community conflicts between PES receivers and non-receivers (Rodríguez de Francisco, Budds, & Boelens, 2013; Tacconi et al., 2013). In these cases, conservation restrictions that have been required to receive PES payments have resulted in trade-offs that have fallen hardest on the poor and women (J. Boyd, 2002; J. Kerr, 2002). Active involvement, particularly from peasant and indigenous communities and organizations, have succeeded in shaping PES programs toward social objectives (P. McElwee et al., 2014; Shapiro-Garza, 2013).

PES projects can be broadly characterized as falling under either “use-restricting” or “asset-enhancing” approaches (Sven Wunder, 2007). “Use-restricting” PES would pay participants to not do something, such as convert forests to agriculture, or hunt wildlife (Milne & Adams, 2012), while “asset-enhancing” would instead focus on active management, such as in reforestation or managing invasive species (Van Noordwijk et al., 2012; S. Wunder, 2005). Asset-enhancing PES projects appear to have made more positive impacts than use-restricting approaches, as policing negative behaviour is more difficult to implement and imposes costs on households (Pirard, Billé, & Sembrés, 2010). In a review of 26 PES cases, (Bhim Adhikari & Agrawal, 2013) assert that environmental outcomes have generally outweighed the social outcomes. Most reports indicate that PES has contributed to modest improvements in forest conservation and ecosystem service provision (Alix-Garcia, Shapiro, & Sims, 2012; R. Arriagada, Ferraro, Sills, Pattanayak, & Cordero-Sancho, 2012; T. M. Hayes, 2012; Robalino & Pfaff, 2013; Scullion, Thomas, Vogt, Pérez-Maqueo, & Logsdon, 2011). Larger PES projects (like in China) have reported larger gains from ecosystem service provision (Ouyang et al., 2016). However, many PES projects report problems with conditionality (e.g., payments only being made for actual conservation actions) in that providers of services are not strictly monitored to make sure they are providing the paid-for action, and there is often no consequence if negative actions, such as deforestation, do occur (Brouwer et al., 2011; Daniels, Bagstad, Esposito, Moulaert, & Rodriguez, 2010; Minang & van Noordwijk, 2013; Pattanayak et al., 2010; Porras et al., 2013; Van Noordwijk et al., 2012).

Only few studies have assessed the impacts of PES on equity. (Hedge & Bull, 2011) show that the forest-conservation PES project they assessed in Mozambique resulted in higher cash income and consumption expenditure of the participant households. However, benefits were differentiated and biased towards a particular profile of households. Female-headed and relatively poorer households did not benefit much from the project. This payment scheme thus could have contributed to exacerbating local inequality between households. (L. R. García-Amado, Pérez, Escutia, García, & Mejía, 2011) also report evidence showing that the PES scheme in Mexico has increased inequality between social groups (landholders and landless people in the community), and (Lansing, 2014) found that access to the PES schemes in Costa Rica is very biased, favoring large landholders and excluding certain types of smallholders. (J. C. Rodríguez-de-Francisco & Budds, 2015) argue that PES might enable the continuation of significant power inequalities between social groups. After reviewing 36 PES schemes, (Bhim Adhikari & Agrawal, 2013) report that most cases score low to medium in social outcomes, such as livelihoods and equity, which raise some questions about the long-term social sustainability of the programs. They point out that securing property rights, the adoption of a community-based approach, inclusiveness, transparency, capacity building and trust in the intermediary organizations are among the factors determining effectiveness of PES. Therefore, the contribution of PES to social welfare and fairness cannot be taken for granted, and would depend on the design and the local institutional setting.

Impact of PES on NCP and biodiversity

Rigorous impact evaluation studies of PES are still incipient and limited in number. However, most of the studies that have been carried out to evaluate the performance PES focused on case studies schemes aiming to prevent forest loss. These studies report a great diversity of environmental outcomes, when taking forest cover as dependent variable. Most of the studies have been conducted in one world region, namely Latin America. In order to assess in a comprehensive way the impacts of the interventions on forest-cover changes, we need to consider not only the propensities to forest loss in treatment versus control parcels, but also the likelihood of deforestation if the payments would have not taken place (baseline situation) and leakage effects (forest loss due to the displacement of the population or productive activities induced by the payment scheme). For example, when assessing the Mexican national program for ecosystem services, (Alix-Garcia et al., 2012) found a significant reduction in the probability of deforestation in the treatment parcels, when compared to control ones, However, the average expected deforestation without the program is reported to be very low. This outcome could be explained by the fact that changes in the drivers of deforestation were already taking place outside of the scope of the PES scheme. For instance, regions that enrolled in PES schemes could have already experienced a forest transition due to demographic or structural productive changes that are not affected by the payments. Treatment (land under PES) and control areas are exposed then to the same type of drivers that have reduced deforestation. In such situations, the overall impact of the program on land-use changes cannot be expected to be significant, and actually there is a risk of "paying for nothing". These authors also conclude that leakage effects were differentiated. While they found evidence of leakage in poorer ejidos (which could erase the program's impacts), in wealthier ejidos the effects were the opposite (the program reduced deforestation even beyond the contracted areas).

Overall, the reported effects of PES on forest cover in the available literature are very diverse, from negligible (Robalino & Pfaff, 2013) to significant (Sims & Alix-Garcia, 2017). (Ezzine-De-Blas, Wunder, Ruiz-Pérez, & Del Pilar Moreno-Sanchez, 2016) argue that PES effectiveness is influenced by three design features: spatial targeting, payment differentiation and strong conditionality. However, though payment differentiation, good targeting and setting in place enforcement mechanisms might likely positively affect the environmental effectiveness of PES, the literature about the design and institutional features that determine payments' effectiveness is still too incipient to draw robust conclusions about the key determinants of effectiveness in achieving environmental additionality. The assesment of additionality is also dependent on the adopted scale. For instance, (Daniels et al., 2010) conclude that the national PES scheme in Costa Rica has had no additional impact on lowering deforestation rates at the national level, but sub-national analyses reveal that it has reduced deforestation rates in some places, as compared to the business as usual scenario.

The number of rigorous impact assessment studies dealing with socio-economic outcomes of PES is even lower and the results equally diverse. (Ibarra et al., 2011) report negative impacts of PES on local food security in an indigenous community in Mexico. When looking at wealth and self-reported well-being, (R. A. Arriagada, Sills, Ferraro, & Pattanayak, 2015) could not find any evidence of impact of participating in the PES program of Costa Rica. Nevertheless, when assessing the impacts of the Grain for Green PES scheme in China, (Emi Uchida, Xu, Xu, & Rozelle, 2007) found that, despite the fact that most households did not participate voluntarily and payments often fell short of the original promised amount, the program had a (moderate) positive impact on income derived from livestock activities and some asset holdings. There might be also trade-offs between the achieving environmental and social objectives in PES. For instance, (Goh & Yanosky, 2016) found that the PES scheme in a location with Atlantic forest in Paraguay has induced low additional effects with regards to deforestation rates, but significant social benefits.

There is less literature on the biodiversity impacts of PES schemes; most reviews treat biodiversity as an assumed co-benefit of forest or land conservation without quantifying the actual biodiversity gain or protection (Nagendra, Reyers, & Lavorel, 2013). There are critiques in the literature that PES programs are often instituted without attention to ecological principles and metrics (Naeem et al., 2015). Monitoring of biodiversity has been particularly challenging for many PES programs (Sommerville et al., 2010). Few papers have tried to address how biodiversity targeted PES might operate, particularly on global levels (Wünscher & Engel, 2012). However, in reality in most cases biodiversity or species targets are set at local community levels (subnational), in either donor or tourist markets (Ingram et al., 2014). Some species-focused PES have reported better conservation of biodiversity after incentive payments have been made, although the number of these projects is very small (Clements et al., 2013; Ingram et al., 2014; Sommerville et al., 2010).

Though the empirical evidence is still incipient and incomplete, a first preliminary conclusion from the available literature is that setting in place a PES scheme does not automatically entail neither environmental effectiveness nor sizable economic benefits for the participants. Also, using direct payments as policy tool does not necessarily encompass more cost-effectiveness in achieving forest conservation, or always has a win-win outcome. The relationship between PES design and environmental and socio-economic outcomes is considerably mediated by the institutional and policy setting in which the scheme is implemented. The choice of the instrument does not ensure a particular type of outcomes. The key choices are actually the institutional and policy settings in which these instruments are unraveled. The literature actually reveals a great diversity of institutional configurations in the field, and the predominance of state-led PES programs. Intermediaries play an important role in the configuration of PES, due to informational and other types of asymmetries between the agents involved. Another key feature that influence PES performance is the profile and motivation of participants. Because PES is basically voluntary, there might be considerable selection biases in PES participation among the target population. Several studies have found that factors beyond the search for economic compensation (such as social norms, environmental values) have an important role in determining participation in PES schemes (Grillos, 2017; Hack, 2010; Hendrickson & Corbera, 2015). The long-term effectiveness of PES schemes is expected to rely on its capacity to appeal to intrinsic motivations (Van Hecken, Merlet, Lindtner, & Bastiaensen, 2017). This however seems contradictory with the proposition that payments can actually induce behavioral changes, and make the adoption of conservation practices contingent on utilitarian calculations and monetary compensation (R. L. García-Amado, Ruiz Pérez, & Barrasa García, 2013). The available literature nonetheless does not yet provide much evidence on the behavioral effects of PES. There is also a significant knowledge gap with regards to the extent to which and in which ways payments can strengthen (or undermine) governance structures for governing common pool resources (T. Hayes, Murtinho, & Wolff, 2015; J. M. Kerr, Vardhan, & Jindal, 2014), as well as on whether induced land-use changes can be sustained after the payments cease (Pagiola, Honey-Rosés, & Freire-González, 2016).

Improving REDD+ policies

"Avoided deforestation" policies, referred to as Reduced Emissions from Deforestation and forest Degradation, or REDD+, have been discussed since the 2007 Bali COP, where the concept was endorsed for the first time by the signatories of the UNFCCC (A. Agrawal, Nepstad, & Chhatre, 2011; Corbera, Estrada, & Brown, 2010; Gupta, Lövbrand, Turnhout, & Vijge, 2012). Following the Cancun COP in 2010, the working group on Long Term Cooperative Action agreed to support the development of REDD+, and encouraged countries to begin to contribute to future implementation by taking a number of steps, including in monitoring, reporting and verification (MRV) and social safeguards. IPLC groups were successful in lobbying for some of their concerns, such as regarding safeguards, to be included at UNFCCC meetings (Wallbott, 2014). Since IPLC control nearly 25% of the tropical forests, their involvement in REDD+ is crucial for success of the program (Bluffstone, Robinson, & Guthiga, 2013).

"REDD+ readiness" pilot programs to prepare countries for REDD+ have emerged in many different nations, funded by bilateral and multilateral donors, such as the Norwegian Development Agency, which has been a large supporter of bilateral REDD+ readiness actions, including pledges of $1 billion to Indonesia, $250 million to Guyana, and $30 million to Vietnam, among other countries, and involving new institutions like the UN-REDD program and the Forest Carbon Partnership Facility (FCPF) of the World Bank (Cerbu, Swallow, & Thompson, 2011). Many organizations have asserted that REDD+ activities need to be combined with co-benefits, such as biodiversity conservation or sustainable development, and using REDD+ to tackle poverty among forest dwellers has been a commonly proposed approach (Luttrell et al., 2013; Tacconi et al., 2013), and that mitigation and adaptation need to be considered together (see e.g. (Locatelli et al., 2015). The literature remains somewhat unclear about how REDD+ will actually work, given that nations themselves will determine much of the on-the-ground activity towards meeting international benchmarks (Corbera & Schroeder, 2011; Lyster, MacKenzie, & Mcdermott, 2013; Visseren-Hamakers, Gupta, Herold, Peña-Claros, & Vijge, 2012). To date most REDD+ activities that have involved actual payments to forest-protecting communities have been through: 1) individual voluntary carbon projects; 2) donor-led projects of limited scale; or 3) as part of a handful of subnational REDD+ implementation (Acre, Brazil, for example) (Duchelle et al., 2014).

***GQL*:** The literature has focused on three main issues relating to IPLCs: 1) how they are included via participatory mechanisms for local involvement in forest management in REDD+ projects; 2) how benefits from REDD+ will be shared with them, especially for poverty reduction; and 3) how safeguards will be designed to prevent abuses towards IPLCs (Abidin, 2015; P. D. McElwee, 2016; Visseren-Hamakers, Gupta, et al., 2012).

Participation: Questions of good governance, particularly in the form of formal arrangements for participation in the development of REDD+ policies, have not been well addressed in most country readiness plans for REDD+, according to early analysis. Despite the fact that many donors, such as UN-REDD, have called for clear systems of information access and local participation, reports to date have indicated that participation has generally been weak in pilot activities, with many communities only consulted, rather than being involved in a systematic manner in all aspects of REDD+ planning (M. I. Brown, 2013; A. Hall, 2012). Capacity to meet many of the technical requirements of REDD+ is often low among IPLCs as well (Cerbu, Sonwa, & Pokorny, 2013). To date, there is no clear UNFCCC guidance on how local participation or equity should be fostered or promoted through REDD+, leaving this question to individual projects and county programs to tackle, with mixed results (Krause, Collen, & Nicholas, 2013; Sunderlin et al., 2014). Some literature sees potential for REDD+ to motivate more participatory governance structures (Aguilar-Støen, 2015; Fujisaki, Hyakumura, Scheyvens, & Cadman, 2016), while other assessments argue that many national-level REDD+ readiness projects have primarily proceeded in a top-down fashion (Atela, Quinn, Minang, & Duguma, 2015), and have focused mostly on technical issues, such as carbon monitoring (Swan, McNally, Grieg-Gran, Roe, & Mohammed, 2011). Many IPLCs have expressed distrust of REDD+ projects managed by government or outsiders (Evans, Murphy, & de Jong, 2014; D. White, 2014). Where indigenous peoples have been included from the beginning in asking for and designing REDD+ projects, such as among the Surui in Rondonia, Brazil, projects are stronger (Chernela, 2011). There is also potential for inclusion of ILK in community-based carbon monitoring at low cost (Danielsen et al., 2013; McCall, Chutz, & Skutsch, 2016; Pratihast, Herold, De Sy, Murdiyarso, & Skutsch, 2013).

Benefit sharing: How benefits will be used for improving livelihoods, especially for the poorest, has been assessed by several studies, with some concerned about the potential for negative livelihood effects (Corbera, 2012), especially from potential land enclosures (Dressler, McDermott, Smith, & Pulhin, 2012; Eilenberg, 2015). A comparative study by CIFOR in 12 countries with REDD+ activities found that in most cases tenure claims by IPLCs had been enhanced by REDD+ participation, but were often insufficient as compared with legal land reform (Larson et al., 2013). Other studies confirm land tenure needs to be resolved before, instead of during or after, REDD+ implementation (Corbera, Estrada, May, Navarro, & Pacheco, 2011; Ludwig, 2012). Further, the integration of livelihoods into REDD+ formal mechanisms have largely been equated with technical discussion of benefit-sharing arrangements (Chapman et al., 2015; Lawlor, Weinthal, & Olander, 2010; Luttrell & Fripp, 2015; Thuy Thu Pham et al., 2014). To date, the experience of forest carbon projects on livelihood indicators is mixed; some carbon projects have increased smallholder incomes, diversified livelihoods and built capacity and skills (Atela, Minang, Quinn, & Duguma, 2015), while other projects have had minimal or negative impacts (E. Boyd et al., 2007; Caplow, Jagger, Lawlor, & Sills, 2011; Lawlor, Madeira, Blockhus, & Ganz, 2013; Reynolds, 2012). In some cases, there was little local benefit because REDD+ projects did not anticipate difficulties surrounding tenure and rights (Awono, Somorin, Eba'a Atyi, & Levang, 2014; Howson & Kindon, 2015).

Safeguards: The literature focuses on whether safeguards will be sufficient to avoid abuses of IPLCs, ensure participants' rights are protected, and avoid adverse impacts on involved communities and households (Chhatre et al., 2012; Visseren-Hamakers, McDermott, Vijge, & Cashore, 2012). These safeguards include use of such actions as FPIC in advance of REDD+ planning, although some have argued FPIC is structurally weak (Dehm, 2016) and must be adapted to specific country contexts

(Thuy Thu Pham et al., 2015). The Cancun COP in 2010 agreed to the principle of safeguards, although details were lacking; many COP participants found the final decision too weak, as it only requires from participating nations "a system for providing information" on how governments are addressing the problem of safeguards in REDD+. SBSTTA has been working through possible approaches for reporting on safeguards in the future, but guidance is still somewhat unclear. Consequently, different REDD+ projects have developed their own approaches to safeguards, including the UN-REDD's Principles & Criteria (P&C); the World Bank's FCPF Strategic Environmental and Social Assessment (SESA); and the Community, Conservation and Biodiversity Alliance (CCBA) REDD+ Social & Environmental Standards (SES). Yet, while these safeguard standards all refer to the idea that local communities must be involved in REDD+ development, experience on the ground suggests that many of the private carbon projects certified by CCBA, for example, fail to meet stated goals for participation, information access or tenure rights (De La Fuente & Hajjar, 2013; Melo, Turnhout, & Arts, 2014; Suiseeya & Caplow, 2013). Other literature has also argued that justice and equity concerns need to be better incorporated into REDD+ rather than just safeguards (Suiseeya, 2016).

Impact on NCP: Given the heterogeneous approaches to REDD+ and the high diversity of countries that plan to participate, it is unclear if REDD+ will actually reduce carbon emissions from deforestation in a cost-effective way – more implementation and experience is necessary to judge REDD+. The literature is currently mixed on the success rates of forest carbon projects in general. Many REDD+ projects have been expensive to set up, and therefore it is not clear if they will be cheaper than other approaches to forest protection (Luttrell, Sills, Aryani, Ekaputri, & Evnike, 2016), and whether direct payments for labor may be more efficient (Bottazzi, Cattaneo, Rocha, & Rist, 2013). Assessing co-benefits besides carbon is one way to increase the value of forests under REDD+ as well (Ojea, Loureiro, Alló, & Barrio, 2016).

***Impact on biodiversity*:** Despite proposals to include co-benefits of biodiversity in forest carbon projects, this has been implemented only loosely. Several carbon projects have demonstrated no benefits to biodiversity (Duque, Feeley, Cabrera, Callejas, & Idarraga, 2014; J. M. Hall, van Holt, Daniels, Balthazar, & Lambin, 2012; Murray, Grenyer, Wunder, Raes, & Jones, 2015; Phelps, Webb, & Adams, 2012; Venter et al., 2013), and overall assessments that map biodiversity onto areas of dense or high forest carbon have not yet translated into specific projects of successful co-benefits (Entenmann, Kaphegyi, & Schmitt, 2014; Jantz, Goetz, & Laporte, 2014; Strassburg, Turner, Fisher, Schaeffer, & Lovett, 2009). Some have therefore called for decoupling, rather than joining, carbon and biodiversity benefits in REDD+ projects (M. D. Potts, Kelley, & Doll, 2013).

Expanding afforestation and reforestation for NCP and GQL

Reforestation projects have contributed to reversing the deforestation trend and increasing forest cover in some countries. The forest area in China and India has increased by 31% and 10% respectively from 1990 to 2014, although mainly due to plantations (FAO, 2016). Carbon forestry and PES schemes have also contributed to expansion of reforestation and afforestation projects in recent years (Carnus et al., 2006; Madsen et al., 2010). Carbon forestry projects have expanded particularly rapidly in Latin America (Corbera & Brown, 2008, 2010; Osborne, 2011) and Africa (Jindal, Kerr, & Carter, 2012; Namirembe et al., 2014), while PES payments have been used to fund afforestation in other parts, such as in China, where the Natural Forest Conservation and Grain for Green programs have spent more than USD 15 billion on the re-conversion of 9 million ha of cropland to forest and grasslands (FAO, 2015; Ministry of Natural Resources of the People's Republic of China, 2013; UNEP, 2012).

Reforestation can have positive biodiversity impacts depending on the local context, such as the creation or improvement of habitat corridors, which in turn can improve long-term survival of forest-dependent species (Carnus et al., 2006). However, there is limited data on the link between reforestation and changes in other NCP (Stringer et al., 2012), and although proponents of carbon forestry have argued that there can be benefits to biodiversity when carbon is bundled with other ES. However, areas and trees with high carbon density and value do not necessarily coincide with high biodiversity areas, and climate mitigation afforestation programs can be in conflict with biodiversity conservation of open habitats such as natural and semi-natural grasslands (FAO, 2015).

Promoting plantation forestry in degraded farmlands can help mitigate the repercussions of water shortages on rural livelihoods, while sustaining energy needs, income, and food security (D. R. Brown, Dettmann, Rinaudo, Tefera, & Tofu, 2011; Djanibekov et al., 2013; Khamzina, Lamers, & Vlek, 2012; Miranda, Porras, & Moreno, 2004; Trabucco, Zomer, Bossio, van Straaten, & Verchot, 2008; Zomer, Trabucco, Bossio, & Verchot, 2008). Some plantation projects, however, have been economically unsustainable for local people because of high opportunity cost of land and labor, and delayed and low benefits (Aggarwal, 2014; Eraker, 2000; Glomsrød, Wei, Liu, & Aune, 2011; Jindal et al., 2012; Lang & Byakola, 2006). Overall, property rights, land availability, social organization and political networks constitute key factors in accessing and benefiting from afforestation projects (Awono et al., 2014; E. Boyd et al., 2009; Corbera & Brown, 2010; Howson & Kindon, 2015; Jindal, Swallow, & Kerr, 2008; John Kerr, Foley, Chung, & Jindal, 2006; Reynolds, 2012).

6.2.4 Protecting nature

Improving management of protected areas
Nearly two-thirds of the global population relies directly on PAs for freshwater provision (Dudley & Stolton, 2003; Harrison et al., 2016), and PAs support other NCP including pollination services for food production (IPBES, 2016), air purification and temperature regulation (Baró et al., 2014; Escobedo, Kroeger, & Wagner, 2011; Kibria, Behie, Costanza, Groves, & Farrell, 2017), protection of medicinal plants that sustain both local and global pharmacopeia (Amjad et al., 2017; S. L. Chen et al., 2016; Jobstvogt, Hanley, Hynes, Kenter, & Witte, 2014; Kala, 2005), numerous cultural services (Ament, Moore, Herbst, & Cumming, 2017; I. D. Wolf, Stricker, & Hagenloh, 2015; Isabelle D. Wolf & Wohlfart, 2014), and foster GQL (Chun, Chang, & Lee, 2017; Lemieux et al., 2015; Maller, Townsend, Pryor, Brown, & St Leger, 2006; Stolton & Dudley, 2010) (Maller et al. 2005; Stolton & Dudley 2010; Lemieux et al. 2015; Chun et al. 2016). These benefits are shown to be higher in areas of greater biodiversity (Fuller, Irvine, Devine-Wright, Warren, & Gaston, 2007; L. J. Wolf, zu Ermgassen, Balmford, White, & Weinstein, 2017)). There is also increasing understanding of the role of protected areas for human health, and the importance to integrate health and conservation policies (Redford, Myers, Ricketts, & Osofsky, 2014; Terraube, Fernández-Llamazares, & Cabeza, 2017).

While PA networks keep expanding at the global level, a trend towards PA Downgrading, Downsizing and Degazettement (PADDD) occurs in many developing countries (Bernard, Penna, & Araújo, 2014; Mascia & Pailler, 2011; Mascia et al., 2014; Pack et al., 2016; Symes, Rao, Mascia, & Carrasco, 2016). Biodiversity is not evenly distributed (T. M. Brooks et al., 2006; Butchart et al., 2015); some of the top conservation priorities at the global level are found in developing countries where the level of resources available for conservation is low (Balmford, Gaston, Blyth, James, & Kapos, 2003; Barnes et al., 2016; Di Minin et al., 2017; Eklund, Arponen, Visconti, & Cabeza, 2011; Lung, Meller, van Teeffelen, Thuiller, & Cabeza, 2014; McCreless, Visconti, Carwardine, Wilcox, & Smith, 2013; Miller, Agrawal, & Roberts, 2013; Ricketts et al., 2005). More efficient support mechanisms are needed to address governance challenges of establishing or managing PAs in developing countries, with a high conservation responsibility at the global level (Balmford & Whitten, 2003; Di Minin & Toivonen, 2015; Miller et al., 2013).

Different studies have evidenced that although PAs do not completely stop habitat loss, they significantly contribute to reducing it across the tropics (Andam, Ferraro, Pfaff, Sanchez-Azofeifa, & Robalino, 2008; Gaveau, Wandono, & Setiabudi, 2007; J. M. H. Green et al., 2013; Nolte, Agrawal, Silvius, & Soares-Filho, 2013; Schleicher, 2018). Similarly, several studies have shown that PAs are effective at maintaining wildlife populations within their borders, both in terrestrial (Barnes et al., 2016; Coetzee, Gaston, & Chown, 2014; Geldmann et al., 2013) and marine environments (Edgar et al., 2014; Lester et al., 2009; Selig & Bruno, 2010). Species that are well represented in protected areas are sliding towards extinction at only half the rate shown by species poorly represented in protected areas (Butchart et al., 2012).

Expanding ecosystem restoration projects and policies

Ecological restoration is the process of assisting the recovery of an ecosystem that has been degraded, damaged, or destroyed (Society for Ecological Restoration International Science & Policy Working Group, 2004). Restoration can occur across scales, from limited and localised to mega-scale projects. It may include returning damaged ecosystems to a more ‘natural’ state (Keulartz, 2009; Van Der Heijden, 2005), creating ecosystems *de novo* (S. J. Hall & Zedler, 2010), or species reintroduction (see also (Fernández et al., 2017). Restoration often aims to restore both ecosystem services and biodiversity and their interactions (e.g. through seed dispersal, pollination, pest control, and invasion resistance) (Bullock, Aronson, Newton, Pywell, & Rey-Benayas, 2011; Montoya, Rogers, & Memmott, 2012).

Freshwater ecosystems have been modified and impaired through pollution, changes in hydrological regime and hydromorphology, overexploitation, IAS and climate change (Gilvear, Spray, & Casas-Mulet, 2013; Malmqvist & Rundle, 2002; UK National Ecosystem Assessment, 2011). This has led to an immense decline and loss in biodiversity (Postel & Richter, 2003; Ricciardi & Rasmussen, 1999; Sand-Jensen, Riis, Vestergaard, & Larsen, 2000) that occurs faster than in any other biome (Millenium Ecosystem Assessment, 2005). At the same time the cumulative total of ecosystem goods and services that watercourses and wetlands provide to society has clearly been diminished (Chicharo, Müller, & Fohrer, 2015; Clarkson, Ausseil, & Gerbeaux, 2013; Febria, Koch, & Palmer, 2015; Postel & Carpenter, 1997; Zedler & Kercher, 2005). The most typical measures in river restoration are related to reconnecting floodplains, flow modification, instream habitat improvement and species management, removal of barriers to upstream or downstream migration of fishes, dam removal or retrofit, channel reconfiguration, bank stabilization, water quality amelioration and activities that increase aesthetics, recreation and education (Wohl, Lane, & Wilcox, 2015). Rewetting by constructing dams or filling draining ditches is the most common measure in wetland restoration (Grootjans, Van Diggelen, Joosten, & J.P.Smolders, 2012). There are increasingly standards being developed for restoration in general (T. McDonald, Gann, Jonson, & Dixon, 2016), specifically for rivers (Palmer et al., 2005) and wetlands (Natural Resources Conservation Service, 2010). Despite numerous efforts to evaluate freshwater restoration success, assessment methods and their standardization still need substantial improvement (González, Sher, Tabacchi, Masip, & Poulin, 2015; Jähnig et al., 2011; Kentula, 2000; Kurth & Schirmer, 2014) in order to be able to arrive at reliable conclusions. Existing studies indicate that hydromorphological conditions are often improved (Jähnig et al., 2011; Muhar et al., 2016), but otherwise restoration performance is often still relatively limited, both for wetlands (Moreno-Mateos, Power, Comin, & Yockteng, 2012) and for rivers (Muhar et al., 2016).

Appropriate restoration criteria for conservation and NCP are required to avoid negative impacts that may be caused by restoration projects, for instance when using non-native species (Ruiz-Jaen & Aide, 2005; Society for Ecological Restoration International Science & Policy Working Group, 2004). Consistently and comparably quantifying land degradation and its reversal through restoration across countries requires assessments of both the geographic extent and severity of damage against a reference state. Several works have proposed that the only comparable and fair reference state across countries is the natural state of ecosystems (IPBES, 2018; Kotiaho, ten Brink, & Harris, 2016; UNEP, 2003).

Improving Sustainable Wildlife Management

Financial compensation to IPLCs who are negatively affected by wildlife has been a widespread strategy for promoting SWM (Anthony & Swemmer, 2015; Boitani, Ciucci, & Raganella-Pelliccioni, 2010; Dickman, Macdonald, & Macdonald, 2011; Kaartinen, Luoto, & Kojola, 2009). It aims at deterring retaliatory killings of wildlife, and to improve the quality of life of IPLCs, by paying for crop damage, predated livestock, injury to humans or fatalities (Agarwala, Kumar, Treves, & Naughton-Treves, 2010; Hazzah et al., 2014; Verdade & Campos, 2004). The idea behind these compensation schemes is to provide economic compensation to IPLCs bearing most of the costs of maintaining public benefits associated with biodiversity conservation (Dhungana, Savini, Karki, & Bumrungsri, 2016; Maclennan, Groom, Macdonald, & Frank, 2009; Naughton-Treves, Grossberg, & Treves, 2003; Persson, Rauset, & Chapron, 2015). In the case of large predators, it is estimated that there are at least 138 compensation schemes in 50 countries, with reported costs of at least US$ 222 million since 1980 (Ravenelle & Nyhus, 2017).

Given that SWM involves many sociocultural values (Dickman et al., 2011; L. R. García-Amado et al., 2011; John Kerr, Vardhan, & Jindal, 2012), some research is also emphasizing the importance of creating not only financial, but also social incentives (e.g., (Berkes, 2004; Dickman, 2010; Hazzah et al., 2014; Heikkinen, Moilanen, Nuttall, & Sarkki, 2011; R. S. Reid et al., 2016), aligning more with IPLC rights and cultural values, capitalizing upon intrinsic motivations to conserve wildlife, and avoiding crowding out effects (Hazzah, Borgerhoff Mulder, & Frank, 2009; Hazzah et al., 2014; Infield, 2001). For example, strengthened land and resource rights can be a strong motivating force for IPLCs to conserve wildlife (Bhim Adhikari, Williams, & Lovett, 2007; Clements et al., 2010; Lindsey, Romañach, & Davies-Mostert, 2009; Ruiz-Mallén, Schunko, Corbera, Rös, & Reyes-García, 2015; Solomon, Jacobson, & Liu, 2012). Establishing clear and secure IPLC rights (including land tenure) has been deemed as an essential basis for fostering effective SWM (Bajracharya, Furley, & Newton, 2005; Scanlon & Kull, 2009; Western, Waithaka, & Kamanga, 2015), given that local ownership of conservation interventions is an important condition for success (D. Nilsson, Baxter, Butler, & McAlpine, 2016; Waylen et al., 2010) and that conservation conflicts often underlie social conflicts, including IPLC struggles over recognition of their rights and institutions (Dickman, 2010; C. Greiner, 2012; Madden, 2004; Madden & McQuinn, 2014; Martin et al., 2016). Hence, ensuring clear entry points for IPLC input and effective communication with IPLC networks and dedicated funding to enable IPLC participation (J. Brooks, Waylen, & Bogerhoff Mulder, 2013; R. Hill et al., 2015; Madden, 2004; Redpath et al., 2013; R. S. Reid et al., 2016; Treves, Wallace, Naughton-Treves, & Morales, 2006; Treves, Wallace, & White, 2009) are critical. In particular, recognition of ILK, including e.g., traditional tools to mitigate human-wildlife conflicts, preventive livestock husbandry methods and natural deterrents to repel wildlife from targeted resources, can inform conservation efforts and enhance the effectiveness of SWM (Chang'a et al., 2016; Hazzah et al., 2014; Housty et al., 2014; Jacobsen, 2014; Kolipaka, Persoon, De Iongh, & Srivastava, 2015; Lute, Carter, López-Bao, & Linnell, 2018; Ocholla et al., 2013; Western et al., 2015).

Managing IAS through multiple policy instruments

Lessons can be learned from IAS (national) programs and experiments (Kettenring, Whigham, Hazelton, Gallagher, & Weiner, 2015; Leung et al., 2012). Tax incentives for set-asides for restoration work, such as Landcare & Bushcare policies (Australia) are farmer voluntary policies that encourage community-based strategic restoration projects (Compton & Beeton, 2012), including bush set-asides for recovery from grazing and grants to replant and fence off bushland. Farmers pay for at least half the restoration costs, which can be reclaimed through tax incentives (Abensperg-Traun et al., 2004). The innovative Working for Water Program in South Africa is an example of an approach that combines IAS removal through targeted employment and payments for removal to poorer participants. The project has been credited with success in indigenous vegetation species recovery (Beater, Garner, & Witkowski, 2008; van Wilgen & Wannenburgh, 2016) and increasing water yields (Dye & Jarmain, 2004; Le Maitre et al., 2002; Le Maitre, Versfeld, & Chapman, 2000). Lessons from the South Africa program include the need to for continuous monitoring and frequent follow-up clearing, the need to train clearing personnel and the need for active restoration (and replanting) of indigenous tree species on cleared plots.

There has been some experimentation with tradeable permits for IAS eradication; for example, in Western Australia, the removal of feral sheep, goats and cats has been funded by the Gorgon Barrow Island New Conservation Benefits Fund, which derives payments for development rights for natural gas from Chevron Australia (Holmes et al., 2016). (Norton & Warburton, 2015) suggest that for offsetting for IAS to work, programs must meet 7 criteria: the invasive species must be able to be reduced in number so as to positively impact native biota; the permit must be affordable; the mitigation area must be large enough; the offset needs to use up to date methods of pest control; the program needs to use adaptive management in IAS control; the program needs to have risk assessment and penalties for nonperformance; and the offset needs to have long-term funding mechanisms.

Improving financing

Auctions for voluntary land conservation: Tender-based approaches and reverse auctions are mechanisms in which landowners bid for financial support for conservation actions. In this approach, a government authority invites stakeholders (usually landholders) to submit bids for implementing conservation actions desired by the state; those whose bid is deemed to be most cost-effective are usually awarded the contracts. Australia has been a particular leader, with several programs including *Reef Rescue* by the state of Queensland and the federal government's *Environmental Stewardship Program* (Burns, Zammit, Attwood, & Lindenmayer, 2016). South Africa (Selinske, Hardy, Gordon, & Knight, 2017) and the US (P. J. Ferraro, 2008) have used auctions for habitat conservation.

Reverse auctions for PES schemes have targeted a range of environmental objectives including protection or enhancement of soil, water and vegetation quality, threatened species habitat, carbon sequestration and pest control, or different combinations of these (Selinske et al. 2017). Most commonly, these forms of reverse auctions are sealed-bid simultaneous auctions, in which bidders offer a service for a set period. Auctions are often repeated over time within the same pool of potential bidders (P. J. Ferraro, 2008).

An important element of successful competitive bidding approaches is an explicit focus on cost-effectiveness. The mechanism is theoretically one of the more efficient ways to allocate conservation funds (Whitten, Wünscher, & Shogren, 2017). In paying landholders to provide environmental benefits, there is a risk that information asymmetry (the greater information that landholders have about the costs to them of providing the benefit) will lead to prices that include significant informational rents (P. J. Ferraro, 2008). Well-managed reverse auctions can reveal information about true costs of provision and therefore reduce these rents, improving the cost-effectiveness of public expenditure on environmental benefits (P. J. Ferraro, 2008; Stoneham, Chaudhri, Ha, & Strappazzon, 2003).

The social benefits include financial recognition of landholders as stewards and providers of ecosystem services as public benefits. The approach, often marketed directly to landholders, has often been successful in placing a value on environmental values, encouraging landholders to engage in environmental management who otherwise might not, and building landholders' environmental management capacity (Selinske et al., 2017).

However, general risks that apply to financial incentives for any social action also apply to reverse auction approaches. For example, the move towards a financially motivated approach to private land conservation carries the risk of motivational crowding-out, in which intrinsically motivated conservation ethic is reduced (Kits, Adamowicz, & Boxall, 2014; Pattanayak et al., 2010). There are also problems with evaluating benefit of the activity (Maron, Rhodes, & Gibbons, 2013). Targeted auctions, which are common in PES schemes, allow bids to vary in both the benefits provided and the price. Careful design of assessment of the benefit component is therefore crucial, but has often been suboptimal, resulting in poor additionality (Maron et al., 2013; Pattanayak et al., 2010).

The cost-effectiveness argument for financial incentives for the conduct of stewardship or conservation relies upon the approach funding predominantly the additional stewardship benefits that landholders would not otherwise provide. While initially, new participants may be drawn into such programs, over time, financial support becomes an expectation of all land stewards, and there is a risk that those who previously would have continued to provide services will no longer do so without payment. Combined with other factors such as learning by bidders from past outcomes, the net cost-effectiveness of such programs can decline over time, and administrative costs associated with reverse auction programs can be relatively high (P. J. Ferraro, 2008).

Ecological fiscal transfers: EFT redistribute public revenues from higher to lower levels of governments to help the latter cover their expenditure in providing public goods and services. Comparatively new is the rationale to use fiscal transfers for biodiversity or forest conservation policies, i.e. acknowledging 'ecological public functions' in redistributing tax revenues from national to state or local governments (Droste, Ring, Schröter-Schlaack, & Lenk, 2017; I. Ring et al., 2011; Irene Ring, 2002; Irene Ring, Hansjürgens, Elmqvist, Wittmer, & Sukhdev, 2010a). EFT introduce ecological indicators such as the quantity and quality of protected area or forest area in the redistribution formula of intergovernmental fiscal transfers. To date, there are only a few countries globally that have implemented ecological fiscal transfers, although the transfer potential is high and many countries have the opportunity to do so with rather low transaction costs (I. Ring, Droste, & Santos, 2017; Irene Ring, 2008a; Schröter-Schlaack et al., 2014). EFT complement PES in the conservation policy mix that mainly addresses private actors, while EFT address public actors. In a wider context, EFT schemes complement the instrument portfolio of ecological or environmental fiscal reforms. Thus far, concepts for environmental fiscal reforms mostly focus on environmental taxes, thereby missing the financing and incentive effects of redistributing tax revenues based on conservation indicators back to public actors (I Ring, 2011; I. Ring et al., 2018).

Since the early 1990s, more than half of the Brazilian states have introduced various ecological indicators to distribute a part of the state-level value-added tax revenue back to municipalities, the so-called ICMS Ecológico (Droste, Lima, May, & Ring, 2017; May, Boyd, Chang, & Veiga, 2004; Irene Ring, 2008b). Protected areas for biodiversity conservation are the most important and widely used of these ecological indicators. Comparing states with and without EFT scheme in Brazil, (Droste, Ring, et al., 2017) show that this innovative instrument succeeded in incentivizing municipalities to designate more protected areas. In Europe, Portugal was the first EU Member State to introduce ecological fiscal transfers in 2007. Building on the EU Birds and Habitats Directives, Portugal used Natura 2000 sites and further, nationally protected areas as indicators for redistributing tax revenues from the national level to municipalities (Santos, Ring, Antunes, & Clemente, 2012). France has a small-scale EFT scheme, acknowledging fiscal needs of municipalities lying in strictly protected areas such as national parks (Borie et al., 2014), whereas EFT schemes have been proposed for Switzerland (Köllner, Schelske, & Seidl, 2002), Germany (Droste, Lima, et al., 2017; Irene Ring, 2008a) and the EU (Droste et al., 2016; I. Ring et al., 2017), among others.

More recently, India's 14th Finance Commission added forest cover as an indicator to redistribute tax revenue from the central government to each of India's 29 states, encouraging state governments to protect and restore forests (Busch & Mukherjee, 2018). Forest conservation has previously been discussed in relation to EFT schemes, especially in the context of REDD+, linking biodiversity conservation and climate change for example in Indonesia (Irawan, Tacconi, & Ring, 2014; Irene Ring, Hansjürgens, Elmqvist, Wittmer, & Sukhdev, 2010b). However, with the implementation of its new EFT scheme, India now distributes 7.5% of its national-level tax revenue based on forest cover. The (Government of India, 2015) estimated that in the coming years, between US$ 6.9 and $12 billion annually will be distributed to states proportional to their forest cover. It remains to be seen what kind of incentives such a large-scale EFT scheme will provide and to what extent the Indian states will introduce EFT schemes from the state level to municipalities, acknowledging the local governments' contributions to forest conservation (see also (Busch & Mukherjee, 2018).

The example of the Amazon Fund: One example of financing mechanism with multiple conservation and development objectives is the Amazon Fund in Brazil. It was created in 2008 and aims to raise REDD+ results-based payments to make non-reimbursable investments in projects to prevent, monitor and combat deforestation and to promote conservation and sustainable use of forests. The purpose is to provide an incentive for Brazil and other tropical forested countries to continue and increase voluntary reductions of GHG emissions from deforestation and forest degradation. This fund supports projects in accordance to national policies in the following areas: (i) management of public forests and protected areas; (ii) environmental control, monitoring and inspection; (iii) sustainable forest management; (iv) economic activities developed through sustainable use of the forest; (v) ecological and economic zoning, land-use planning and land tenure regularization; (vi) conservation and the sustainable use of biodiversity; and (vii) recovery of deforested areas.

The Brazilian Development Bank (BNDES) is the manager of the Amazon Fund, which includes analysing, monitoring and evaluating the projects, receiving the donations, making the disbursements, and carrying out fundraising, accountability and assessments. The governance structure is composed of a Steering Committee, with participation of the federal government, state-level governments and civil society, which is responsible for setting investment guidelines and monitoring the results; and a Technical Committee, appointed by the Ministry of the Environment, which certifies the emission calculations from deforestation in the Amazon and sets the fundraising limits. It is an innovative approach based on evidence of results on reducing deforestation, multi-stakeholder governance, managerial autonomy and low management costs (Marcovitch & Pinsky, 2014).

The Amazon Fund started to receive donations from foreign governments and companies in 2009. The Government of Norway is the main donator (96.9%), followed by the German Development Bank KfW (2,5%) and the Brazilian Oil Company Petrobrás (0.6%), but it is able to receive donations from individuals, NGOs and multilateral institutions. The resources transferred up to December 2016 sum up to over USD 1 billion. From that, 28.9% was already applied in 87 projects distributed through the Third Sector, Federal Government, States, Municipalities, Universities and International Organizations (Amazon Fund, 2016).

The resources have been applied in large-scale structuring projects to support public policies, e.g. the Amazon Region Protected Areas Program (ARPA), the Rural Environmental Registry (CAR, in Portuguese) and the National Forestry Inventory; in civil society-lead initiatives to improve capacity building and sustainable production, e.g. supporting priority municipalities in strengthen environmental management, valuing chains of non-wood forest products and developing territorial management plans to indigenous lands; and in research and development projects, e.g developing technology for the recovery of degraded areas, structuring a Center for Advanced Studies on Biodiversity and developing a deforestation detection system using orbital imaging radar (Amazon Fund, 2015).

6.3 Integrated Approaches for Sustainable Freshwater

Threats to freshwater biodiversity: SDG6 focuses on drinking water and sanitation to cover the entire water cycle, including the management of water, wastewater and ecosystem resources. Targets 6.3 to 6.6 address water quality and wastewater management, water scarcity and water-use efficiency, integrated water resources management, and the protection and restoration of water-related ecosystems. The SDG6 targets are linked to ABT 2, 8, 11, and 14 and other SDGs such as goal 8, 12, 13 and 15. Targets 6.a and 6.b of SDG6 acknowledge the importance of an enabling environment, addressing the means of implementation and aiming for international cooperation, capacity building and the participation of local communities in water management. The current major threats to freshwater biodiversity are widely known and include climate change, habitat fragmentation, introduction of invasive alien species, toxic pollutants, dumping of nutrient or organic loadings increasing eutrophication processes, channel modification, water abstraction for human and agricultural consumption and overexploitation (IPBES GA chapter 2 and 3; (Bates, Kundzewicz, Palutikof, & Wu, 2008; Failler et al., 2018; Jiménez, Cortobius, & Kjellén, 2014; OECD, 2015; UNEP, 2016; Vörösmarty et al., 2010; WWAP, 2012). In managing the risk of shortage, the water management regime should aim to maximise the value an individual and society obtain from water resources in terms of economic, environmental and social outcomes (Konikow & Kendy, 2005; Varady, Zuniga-Teran, Gerlak, & Megdal, 2016). There are trade-offs and co-benefits between water quality and quantity management and other sectoral policies such as land, agriculture, energy, biodiversity, urban planning, and climate change (Stringer et al., 2018; United Nations Environment Programme, 2019; UN-Water, 2016; WWAP/UN-Water, 2018).

Challenges faced by IPLCs: It is well established that IPLCs have a high reliance on freshwater environments (Finn & Jackson, 2011; Langton, 2002) and have developed complex customary institutions for governing and managing freshwater resources and the biodiversity therein (Adams, Watson, & Mutiso, 1997; Cooper & Jackson, 2008; Huanca, 2006; S. Jackson & Altman, 2009; Weir, Ross, Crew, & Crew, 2013). Cultural and spiritual ties to freshwater bodies such as lakes, rivers and lagoons, have deep meaning amongst many IPLCs all over the world, and are at the basis of IPLC water management institutions, from which many NCP are derived (Barber & Jackson, 2011; Fernández-Llamazares et al., 2017; S. Jackson & Altman, 2009; King & Brown, 2010; Lorente Fernández, 2006; S. Yu, 2000). However, IPLCs are often not acknowledged as users or owners of water catchments and systems in many national laws (Tan & Jackson, 2013). Indeed, IPLCs are advocating worldwide for greater management control of water resources (Behrendt & Thompson, 2004; M. R. Hill et al., 2013; S. Jackson, 2011; S. Jackson, Tan, & Altman, 2009; O'Faircheallaigh & Corbett, 2005), given that many of their economies depend on freshwater ecosystems and species (S. Jackson, Finn, & Scheepers, 2014).

There is well-established evidence that IPLCs have often been marginalized from water resource planning all over the world (Finn & Jackson, 2011; S. Jackson, 2008, 2011; Weir, 2009) and are one of the most vulnerable groups to the impact of water resource development projects (Finn & Jackson, 2011; King & Brown, 2010), including dams and large-scale irrigations plans (Dell'Angelo, D'Odorico, & Rulli, 2017; Finer & Jenkins, 2012; Winemiller et al., 2016). Water resource agencies, as well as private corporations, have oftentimes overlooked the crucial links between IPLCs and freshwater bodies, making little attempt to understand the values shaping the patterns of IPLC freshwater resource use (Barber & Jackson, 2012; Bark & Jacobs, 2009; Osborn, 2009; N. Singh, 2006).

6.4 Integrated Approaches for Sustainable Cities

6.4.1 Urban planning for sustainability

Options for sustainable urban planning include: bioregional planning; biodiversity-friendly urban development; increasing green space in cities; and protecting land for urban agriculture.

Bioregional planning*:* Inter- and transdisciplinary, collaborative and strategic urban planning and design that integrates with surrounding regions can offer numerous benefits (Ahern, 2013; Breuste, Niemelä, & Snep, 2008; Colding, 2011; Ignatieva, Stewart, & Meurk, 2011; R. McDonald & Marcotullio, 2011; Novotny, Ahern, & Brown, 2010; Pauleit, Liu, Ahern, & Kazmierczak, 2011). As examples of integrated approaches, the various 'river to life' projects, including the celebrated restoration of the C*heonggyecheon River* in Seoul, illustrate that with coordinated planning a great deal can be achieved, delivering not only restoration of urban habitats but at the same time economic success as major new tourist destinations (Revkin, 2009). A second example is Singapore where an increase in urban density and urban landscape has been achieved concurrently (P. Newman, 2011). The contribution of planning in Singapore has been fundamental in requiring consideration of the environment in every development decision for decades (Ho, Ker, & Kiat, 2016). A third example is planning for green and blue corridors that extend across urban and regional landscapes. The planned national capital of Australia, Canberra has a national capital open space system (NCOSS) weaving between the urban centres of a polycentric city. NCOSS was designed ahead of developing the subsequent town centres and is now highly valued and protected by the Canberra community (National Capital Authority, 2012).

Urban planning and management for climate change adaptation and ecosystem services in cities also offer opportunities to adapt to, anticipate and counteract undesired changes. If different units of city and peri-urban governments worked in tandem, a number of synergies in the governance of urban biodiversity and ecosystem services are possible (Raudsepp-Hearne, Peterson, & Bennett, 2010). Developing in consultation with the community an integrated strategic forward plan for the greater region can consider the major issues of among others water, renewable energy, air quality, carbon in a wider landscape (Alexandra, Norman, Steffen, & Maher, 2017; Beatley, 2011; Carmin, Dodman, & Chu, 2013). This requires functioning and dynamic science–policy linkages at regional scales, challenging the current structure of governance frameworks, practices, and institutions.

Nature-friendly urban development: Some of the most important ecosystem services (e.g., local climate regulation, flood regulation, water purification and supply, recreation) and links between biodiversity and GQL are in cities and urban areas, where most people who consume ecosystem services live. There are limits to urban densification, with thresholds that if exceeded, will not allow for sufficient space for nature and its benefits. Ecosystems are often highly fragmented in urban areas, which can alter the genetic diversity and long-term survival of sensitive species. To ensure viable urban populations, urban planners and designer need to understand species' needs for habitat quality and connectivity among suitable habitat patches(Braaker, Ghazoul, Obrist, & Moretti, 2014).

Ecologically progressive urban planning and policy are already demonstrating how biodiversity conservation and management to enhance local ecosystem services production can be part of urban transitions and transformations for sustainability (Nadja Kabisch, van den Bosch, & Lafortezza, 2017). For example, in the Southeastern State of São Paulo, 133,000 square kilometers of the Serra do Mar State Park cover twenty-four municipalities in the state. The Atlantic Forest is concentrated in the *Serra do Mar*, squeezed between the coastal BSMR (nine cities, 1.6 million people) and the São Paulo Metropolitan Region (19 cities and 20 million people). Despite having been reduced and highly fragmented, the Atlantic Forest is habitat to more than 20,000 plant species – a wealth of diversity greater than that found in North America (17,000 species) and Europe (12,500 species). As part of an intentional effort to link biodiversity conservation with urban development three mosaics (Paranapiacaba, Jureia-Itatins, and Jacupiranga) were created to improve connectivity and allow for buffer zones between urban and native preserved areas in this vast urban region.

Similarly, the City of Jerusalem has assumed the responsibility for improving and maintaining its unique desert and hilly ecosystems to preserve faunal biodiversity in the face of increasing climate change stresses. In 2009, Jerusalem joined the International Council for Local Environmental Initiatives / Local Action for Biodiversity (ICLEI/LAB) Network and established the Gazelle Valley Conservation Program to protect and restore one of the city's unique biodiversity areas and planned and urban nature park for both wildlife preservation and local recreation. The urban park, Gazelle Valley is situated on a sixty-acre undeveloped tract of land in southwest Jerusalem, between two residential neighborhoods and closed in by major roadways. The mountain gazelle (*Gazella gazella*), an indigenous species to the Jerusalem hills, is a highly visible species being managed for co-existence with vibrant urban spaces (McPhearson et al., 2018).

***Increasing green space and greenbelts throughout cities*:** A growing number of cities, local communities and city planners are collaborating to create new green spaces and improve existing ones using GIS and other holistic spatial planning tools and technologies (Pickett & Cadenasso, 2008). These urban green spaces can take the form of public parks, open space, natural preserves, community gardens, private yards, and other spaces. Deliberate preservation of green space through planning and policy is often highly necessary because municipalities regularly encourage development of higher housing densities that lead to the reduction or loss of green spaces and gardens. Green corridors can be incorporated into city planning to connect natural areas and are crucial for enhancing biodiversity on a multidimensional level throughout smaller green spaces (Vergnes, Le Viol, & Clergeau, 2012). For example, urban planning and design that promotes habitat connectivity through linkages or clustering of landscapes, parks, and green infrastructure can increase the provision of multiple ecosystem services such as recreation, stormwater management, and biodiversity preservation (Colding, 2011).

Such green space initiatives can be found in even poorer urban areas of the global South. For example, in Burkina Faso, the municipality of Bobo-Dioulasso has mobilized municipal budgeting to cover the functioning and activities of a greenway management committee and to support maintenance and replication of greenways (Sy, Baguian, & Gahi, 2014). However, despite ample evidence for a positive link between urban green spaces and health benefits, issues of equity in access remain important. Trees and urban green spaces are unequally distributed, often concentrated in wealthier neighborhoods (Depietri & McPhearson, 2017). In addition, socio-economic factors have been found to affect physical activity, with low-income households being less likely to participate in physical activity, and hence use green spaces less (Lee & Maheswaran, 2011a).

Green space planning provides not only recreational benefits, increases in property values, and potential reductions in GHG emissions and urban heat island effects, but can also be a proactive strategy for climate change adaptation. In the US, the case of the Boston's Charles River Basin is exemplary in this aspect. The city was threatened by disastrous floods, since urban expansion and industrial development converted land in large parts of the floodplain during the 1950s-60s (Platt, 2006). By 1983 the US Army Corps of Engineers acquired the Charles River Natural Valley Storage areas, a total of about 32.8 km^2, and, after a decade of improvements and ecosystem restoration along greenways, the Charles River Water Association (CRWA) could measure significant benefits in terms of flood reduction, and improvements in water quality, and recreation opportunities (Platt, 2006).

***Protecting land for urban agriculture and food security*:** Food supply is one of the most important provisioning services provided by ecosystems. Most urban areas only produce a limited amount of food for its inhabitants, done on a much smaller scale than in rural areas. However, increasingly some cities have been successful in providing adequate supplies of food by way of urban and peri-urban agriculture. In the 1990s, the United Nations estimated that 800 million urban residents were involved in some sort of commercial or subsistence agriculture in cities around the world (Smit, Ratta, & Bernstein, 1996). Urban food production can come in the form of community or private gardens and allotments, vegetated rooftops, or more innovative designs such as greenhouse skyscrapers and multifunctional vertical gardens

Urban and peri-urban agriculture can address both increasing food security and conserving biodiversity. (Lee-Smith, 2006) found that urban agriculture provided up to 90% of leafy vegetables and 60% of milk in Dar es Salaam, Tanzania as well as 76% of vegetables in Shanghai and 85% of vegetables in Beijing (K. H. Brown & Jameton, 2000). In Bologna, a case study showed rooftop gardens produced not only 77% of the urban vegetable requirement, but also increased the food system's overall biodiversity as a result of such factors as scale, density, and connection of green corridors (Orsini et al., 2014).

Limited amount of space for urban agriculture is a driving force behind cities adapting to more creative ways in which to grow food. This is difficult in cities that restrict agricultural activities to land zoned as rural. Showing officials that urban agriculture reduces environmental degradation, increases food security, produces jobs, and connects communities could push towards rezoning efforts (Smit et al., 1996). Cities have begun to embrace and realize the potential for urban agriculture, and urban agriculture is making its way into city planning. For example, Minneapolis has written an urban agriculture plan, Accra has created a municipal agricultural department, Havana passed a city ordinance that allows for vacant municipal land be used by urban farmers, Rosario has created a land bank which leases out land to local farmers, Beijing and Dakar are among many cities who have entirely rezoned to make space for urban agriculture, Seattle has legalized the keeping of bees and livestock, and city property has been provided for urban farms in Baltimore (Cohen, 2014; K. J. Morgan, 2010). A recent collaboration between UN-Habitat and the Resource Centers on Urban Agriculture and Food Security (RUAF) Foundation-International has assisting interested cities and other local actors to integrate urban agriculture into local climate change and land-use policies and strategies and to initiate pilot actions that showcase replicable urban agriculture models. Such increased diversification of food and income sources also helps to increase the resilience of poor urban households, which are generally vulnerable to increases in food prices (Dubbeling, 2014).

Furthermore, urban agriculture can be combined with flood or stormwater mitigation. Support for urban agriculture as a flood risk mitigation or stormwater management strategy has grown; for example, in East and Southeast Asia the protection of paddy lands acts as flood mitigation effort and is seen as a benefit to ecosystem services and protection of property and life (Chang & Ying, 2005; Cohen, 2014; Walsh et al., 2016). In Sri Lanka, government offices together with support of UN-Habitat, and a local NGO implemented a pilot project to rehabilitate abandoned paddy lands by promoting the production of traditional varieties of salt-resistant paddy rice combined with the growing of vegetables on raised beds. By re-establishing the flood regulation and ecosystem services of these areas, this strategy not only contributed to reducing flood risk but also to increasing urban food production and income generation and self-sufficiency for the involved households. Support for urban agriculture as a flood risk mitigation or stormwater management strategy has also been taken up in cities like Bangkok after the 2012 flooding (Boossabong, 2014) and similarly in the last decade in New York City (Cohen & Wijsman, 2014).

6.4.2 Nature-based solutions and green infrastructure

Specific options using Green Infrastructure (GI) approaches to address urban problems include the following.

GI to counterbalance temperature effects**:** The role of some types of GI (trees, green roofs and green walls) in regulating temperature is well established (also see Figure 6.1). Ecosystems have the ability to reduce the effects of urban heat islands when there is varied tree coverage and diverse landscape architecture, and when there are urban green spaces and greenbelts, and vegetated roofs and walls - as well as areas with water such as natural or man-made lakes, ponds, reservoirs, rivers, wetlands. Vegetation provides shade, which prevents surfaces such as streets and sidewalks from absorbing heat (McPherson, Simpson, Xiao, & Wu, 2011). In a study done on urban forests in Beijing, results showed that "with a tree/shrub cover of 16.4%, the total air temperature decrease by current trees and shrubs was 1.6°C" (Yang 2005). The positive role of GI has been demonstrated for heat wave mitigation in Berlin (Gabriel & Endlicher, 2011), Manchester (Gill, Handley, Ennos, & Pauleit, 2007), Phoenix (Harlan, Brazel, Prashad, Stefanov, & Larsen, 2006) and Cologne (Depietri, Welle, & Renaud, 2013) amongst others.

Figure 6.1: Living walls grace One Central Park in downtown Sydney, Australia (Elmqvist & McPhearson, 2017). Photograph: bobarc (https://www.flickr.com/photos/bobarc/13160592113/) CC BY 2.0, via Wikimedia Commons

***GI for reducing air pollution*:** Vegetation in urban areas plays an important role in air pollution mitigation. Urban vegetation can remove or reduce certain pollutants from the atmosphere, including GHG emissions through carbon sequestration, and trees act as carbon sinks in urban settings (McPherson, 1998; Simpson & McPherson, 1998). However, cities are not planting as many trees to offset the amount of carbon emissions at the current rate of urban car-dependence (Pincetl, 2012). Encouraging civic ecology, defined as "local environmental stewardship actions taken to enhance the green infrastructure and community well-being of urban and other human-dominated systems" (Krasny & Tidball, 2012), could be a participatory stewardship option to help offset carbon emissions in cities through tree-planting programs, community gardens, and green roof maintenance (Krasny, Russ, Tidball, & Elmqvist, 2014).

***GI to provide clean water supplies*:** Provisioning of water is a critical NCP provided by ecosystems, and protecting watersheds and wetlands within cities and in the hinterlands is crucial to this. Many cities have moved away from investing in expensive and elaborate water filtration systems and have instead invested in local watersheds to purify drinking water (Postel, 2005). A study done in 2005 shows that major metropolitan cities in the United States like New York City, Boston, and Seattle have saved a combined total between $6 and $7 billion in capital and operating costs by protecting their watersheds (Postel, 2005). In a study done on the Piracicaba-Capivari-Jundiaí watershed in Sao Paulo, Brazil, protection and restoration of at least 14,300 hectares of sensitive watershed land would lead to a 50 percent reduction in sedimentation, saving $2.5 million every year and reducing water treatment costs by 15 percent (Y. Qin, Gartner, & Otto, 2015). By protecting wetlands and watersheds, an active role is also taken to support other regulating ecosystem services including flood alleviation, nutrient cycling, and habitat conservation. Ponds, for example, filter waste from human activities reducing the level of pollution in urban wastewater (Karathanasis, Potter, & Coyne, 2003), and urban streams retain and fix nutrients from organic waste (Booth, 2005).

***GI for storm-water management*:** The benefits and cost-effectiveness of GI in storm-water management, flood control, coastal protection (Gedan, Kirwan, Wolanski, Barbier, & Silliman, 2011; Hanley et al., 2014; Keesstra et al., 2018; Morris, Konlechner, Ghisalberti, & Swearer, 2018; Narayan et al., 2016), as well as in climate change mitigation and adaptation in urban areas (W. Y. Chen, 2015; N. Kabisch et al., 2016) are *well established*. (Foster, Lowe, & Winkelman, 2011) found that green alleys or streets, rain barrels, and tree planting are 3-6 times more effective in managing stormwater per $1000 invested than conventional methods. Based on this and similar studies, the city of Portland invested $8 million in green infrastructure to save US$ 250 million in hard infrastructure costs while

Philadelphia saved approximately US$ 170 million of reduced Combined Sewage Overflow (CSO) thanks to planning and development of green infrastructures, which provided greater net value for storm-water control than grey infrastructure ($1.94-$4.45 billion compared to $0.06-$0.14 billion over 40 years) (Foster et al., 2011).

New York City also launched a Green Infrastructure Plan in 2010 designed to invest in new and restored green infrastructure for stormwater management instead of traditional gray infrastructure. This included committing US$1.5 billion for green infrastructure development over the next 20 years (NYC Environmental Protection, 2010). As part of this, the Staten Island Bluebelt is a system of created wetlands developed since the 1990s to provide alternative, ecosystem-based stormwater management services in a rapidly developing borough of New York City. The Bluebelt has become a model for providing multiple ecosystem services including stormwater management, water quality improvement, wildlife habitat provisioning, environmental education, and increased property values. Similarly, the city of Taizhou, China, located on the southeast coast of Zhejiang Province with 5.5 million inhabitants, developed a zoning plan (K. Yu, Li, Liu, & Cheng, 2005) that utilized green infrastructure to adapt urban growth to deal with potential impacts of climate change including preventing stormwater related floods and maintaining food production areas. The Taizhou plan incorporated ecological areas at multiple scales (local to regional) to maintain critical natural processes and flows including hydrology and biodiversity while simultaneously protecting cultural heritage sites and recreation areas (Ahern, 2007; Gotelli & Chao, 2013; K. Yu et al., 2005).

GI for storm and flood control: A growing number of cases are demonstrating the effectiveness of ecosystems as nature-based solutions to buffer the impacts of climatological, hydro-meteorological and even some geophysical hazards such as landslides (McPhearson et al., 2018; Renaud, Nehren, Sudmeier-Rieux, & Estrella, 2016). The creation or restoration of wetlands, tidal marshes, or mangroves provide water retention and protect coastal cities from storm surge flooding (Saleh & Weinstein, 2016; Sutton-Grier, Wowk, & Bamford, 2015) and shoreline erosion during storms (Cunniff & Schwartz, 2015a; Gittman, Popowich, Bruno, & Peterson, 2014a; N. M. Haddad et al., 2015; Pontee, Narayan, Beck, & Hosking, 2016a)(Cunniff & Schwartz, 2015b; Gittman, Popowich, Bruno, & Peterson, 2014b; J. Haddad, Lawler, & Ferreira, 2015; Pontee, Narayan, Beck, & Hosking, 2016b). Such large-scale projects have been implemented in several countries including the USA, UK, Belgium and Viet Nam. In Viet Nam, planting of mangrove forest as a climate adaptation measure was achieved with local communities as co-managers (Schmitt, Albers, Pham, & Dinh, 2013a)(Schmitt, Albers, Pham, & Dinh, 2013b). In Belgium, up to 4,000 hectares of reclaimed wetlands within the Schelde estuary were converted back into floodplains, with an expected saving of €400 million in flood damages (Temmerman et al., 2013). Similarly, “sponge cities” in China are rapidly expanding (Li, Liu, Huisingh, Wang, & Wang, 2017). According to a guideline issued by the State Council of China in 2015, the country aims to have 80% of its built up areas meet the sponge city criteria by 2030. Under this 15-year vision, the sponge city is rapidly gaining momentum backed with sizable investments from central and provincial governments (McPhearson, Iwaniec, & Bai, 2016).

6.4.3 Reducing the impacts of cities

Options to reduce the impacts of cities include the following.

Encouraging density: Urban land area is set to expand by 150% to 2050, apparently exacerbated by a de-densifying trend (Angel, Parent, Civco, Blei, & Potere, 2010; Seto, Güneralp, & Hutyra, 2012). Sprawling cities generally require more energy for transport per capita (P. W. Newman & Kenworthy, 1989), more car travel, less travel by mass transit (Kenworthy & Laube, 1999) and accommodate larger floor area in buildings, which consume more electricity (Christopher A. Kennedy et al., 2015). Overall density is a coarse control for government agencies that provides the potential for uptake of energy efficient mass transit and reduced travel. To be an effective intervention for socio-economic and environmental benefit, density must be articulated (Suzuki, Cervero, & Iuchi, 2013): in other words, implemented at key transport nodes, surrounding and linking between activity centres. Density done well has a reason to be where it is.

Planning urban form and transport: There are further nuances of design that are coupled with density. Articulating density and transit-orientated development enables connectivity, but if the city is designed to house residents densely but separately from areas of employment or other activities, then this does little to reduce the transport task. Planners and industry need to create neighborhoods of mixed land use and diverse housing options that pre-empt the need for citizens to travel across the city (Cervero & Guerra, 2011; Ewing, Pendall, & Chen, 2003; Grubler et al., 2012; Marshall, 2009). Functionally and socially diverse neighborhoods also encourage active transport (W. V. Reid et al., 2010), which may be further promoted by NGOs. Other than density and diversity, the urban chapter of the *Global Energy Assessment* (Grubler et al., 2012) offers several other options to reduce transport energy use:

- Internalisation of external costs: parking fees, dynamic road-use pricing and congestion charges with reduced fees for higher occupancy vehicles. These are possible for metropolitan governments in the short term (less than 5 years) with public consultation.
- Making public transport attractive: dense, high frequency mass-transit networks with efficient intermodal, ticketing and crowd control operations. New networks are a long-term strategy issue for planners, but improved efficiency and systems can be implemented in the short term by companies or government operating mass transit.
- Resist extending the road network: every additional kilometer of road induces an increase in motorized private transport and deflects investment away from more efficient modes.
- Cost effectiveness of different options depends on the city: geography, wealth, income, culture, economic function and access to energy sources mediate appropriate choices for mass transit.

Mitigating building energy use and emissions**:** If the developing world proceeds to construct their new cities with the same material intensity and type of infrastructure we see in developed countries, the potential carbon cost is more than a third of the world's cumulative carbon budget to 2050 to retain the 2 degree climate target (Müller et al., 2013). Buildings are the single largest sector in energy use world-wide (Weisz & Steinberger, 2010). Strategies to mitigate building energy use and emissions are also contingent on the circumstances of the city. With denser colder-climate cities district heating may be an option that is not as effective in lower density cities. If the primary energy source for electricity is carbon intensive, then rooftop PV is advantageous (C. A. Kennedy, Ibrahim, & Hoornweg, 2014) but issues of accommodating intermittency are real and, implemented at scale, renewable electricity could require substantial quantities of rare and base metals (Arrobas, Hund, Mccormick, Ningthoujam, & Drexhage, 2017; R. Kleijn, Van der Voet, Kramer, Van Oers, & Van der Giesen, 2011). The resolution is possible in the medium term with coordination between local government, national building regulators and the construction industry (see Box 6.1 for a discussion of biosolar roofs).

Significant operational savings can be achieved from implementing energy efficient building codes (Pauliuk, Wang, & Müller, 2013) and with new urbanisation and replacement of existing stock, there is an opportunity to decouple energy needs from urban growth. At the same time there are several studies that indicate material resource efficiencies as buildings are demolished or refurbished (Bergsdal, Bohne, & Brattebø, 2007; Müller, 2006; Tanikawa & Hashimoto, 2009). The codes and systems needed for these changes can be implemented in the short term with industry government coordination, though their speed of effect will depend on construction activity.

Addressing urban consumption: The indirect impact of urban consumers and resources embedded in their purchases of goods and services are comparable (if not greater) than their direct resource needs (e.g., energy for heating, transport, or material in food, construction) (Baynes, Lenzen, Steinberger, & Bai, 2011; Lenzen, Wood, & Foran, 2008; Pichler et al., 2017). This may appear outside the control of urban managers and policy but it presents an alternative and de-materialised model for businesses that can be encouraged. Selling the services rather than artefacts that provide a service would mean for example requiring building codes to take advantage of passive heating and cooling, so residents do not individually buy air conditioners or heaters, or encouraging mobility through public transportation, instead of building roads and encouraging automobile use. Implemented with the technical changes about building and transport efficiency and alongside the 'circular economy', this collectively can help separate material needs from consumption (IRP, 2018). In doing so, it addresses broader global issues of embedded energy and emissions in trade and production outside cities and should be encouraged by international NGOs (also see Chapter 6, section 6.4 of the Global Assessment on 'circular economies').

Transformative urban governance: The way that change is brought about is important. Engaging citizens in planning is an important role for (local) governments (Grubler et al., 2012; IRP, 2018). In addition to the usual processes of planning, other flexible, experimental or entrepreneurial strategies may be employed involving business. For example, performance-based standards rather than prescribed technologies or processes can be effective. To help citizens understand conceptually new urban form or transport arrangements, temporary experimentation (e.g. car-free days) can help translate the reality and speed up learning. Where change involves complex systems like autonomous vehicles, vehicle sharing, electric vehicles and charging point networks, a more 'entrepreneurial urban governance' shares the investment and implementation load with business and allows them to also be urban 'agents of change'.

6.4.4 Enhancing access to urban services for GQL

Enhancing access to urban services is especially urgent in cities in the global South, where inhabitants of informal settlements, or slums, have access to no or very few services. A quarter of the world's population live in informal settlements, and according to UN-Habitat, there has been a 28% increase since 1990 in absolute numbers of slum dwellers, although proportion of the urban population in developing countries living in slums has declined (UN Habitat, 2015). Informal settlements are defined as "residential areas where: 1) inhabitants have no security of tenure vis-à-vis the land or dwellings they inhabit, with modalities ranging from squatting to informal rental housing, 2) the neighbourhoods usually lack, or are cut off from, basic services and city infrastructure and 3) the housing may not comply with current planning and building regulations, and is often situated in geographically and environmentally hazardous areas" (UN Habitat, 2013). Informal settlements can be caused by a range of factors, ranging from rapid population growth, rural-urban migration, and displacement from conflict; poor housing policy, lack of affordable housing and economic vulnerability; and weak urban governance (UN Habitat, 2009, 2011, 2013).

With the designation of reducing informal settlements as a UN MDG in 2000, countries have been pressed to address the issue (Richards, O'Leary, & Mutsonziwa, 2007). For example, In South Africa, an "informal settlement intervention approach" through government subsidized housing delivered 1.4 million subsidized houses by 2002. However, there were concens that it was a band-aid approach that involved land seizures, unwanted relocations, and little outreach to communities (Huchzermeyer, 2004). Options for addressing informal settlements and increasing access to urban services among the poorest, so as to increase their quality of life, include increasing access to clean water and sanitation; improving management of solid waste; increasing green space in informal settlements; and transforming governance approaches in informal settlements.

Improving access to clean water and sanitation: The World Summit on Sustainable Development held in Johannesburg in 2002 drew attention to the need for increased access to water and sanitation in urban settlements, as at the time, 2.6 billion people lacked access to improved sanitation facilities. By 2015, 2.3 billion people still did not have access to basic sanitation facilities. While this is an increase from 54% of the world's total population to 68% with access to improved sanitation services, there are still hundreds of thousands of people who suffer health consequences due to lack of sanitation. Combined with almost 900 million people who do not have access to clean drinking water, access to sanitation and water are at the forefront of public health issues across the globe (UNICEF & WHO, 2015).

Water filtration and aquifer recharge are two key ecosystem services needed for sanitation and clean water access. Increasing access to sanitation and clean water will increase economic and social activity by fostering healthier communities, but the development of infrastructure to provide these services can also be detrimental to the environment (Bonnardeaux, 2012). Increasing access to sanitation and clean water by fostering partnerships between all actors to encourage a bottom-up, participatory approach could increase effectiveness and socio-economic benefits (Annamalai et al., 2016; Bonnardeaux, 2012; McFarlane, 2008). More integrated approaches that blend public-private partnerships such as microfinancing, microinsurance, and conditional cash transfers (CCTs) have emerged as potential solutions for improving or providing access to clean water and sanitation (Afrane & Ahiable, 2016; Chunga, Jenkins, Ensink, & Brown, 2017; Davis, White, Damodaron, & Thorsten, 2008; Howard, Calow, Macdonald, & Bartram, 2016; Johannessen et al., 2014). As noted in the previous section on GI, investing in natural ecosystem services such as preserving wetlands can help to conserve biodiversity while also helping communities manage their own water supplies (Postel 2005). Other approaches not dependent on GI include a focus on partnership, participation, and cost-recovery in co-manangement of sanitation and public toilet delivery between governments and NGOs in places like India (McFarlane, 2008).

Improving management of solid waste: According to SDG 11, uncollected solid waste can block drains, cause flooding, and spread waterborne diseases. Between 2009 and 2013, only 65% of the urban population was served by municipal waste collection. Possible innovative options to increase solid waste management in areas not served by traditional public services include incentive programs. For example, Curitiba, Brazil hosts a Green Exchange Programme, "which encourages slum dwellers to clean up their surroundings, and improves public health by offering fresh fruit and vegetables in exchange for garbage and waste brought to neighborhood centers. As of 2012, Curitiba has 96 exchange sites. Each month more than 6,500 people are exchanging an average of 255,416 kilos of collected garbage for 92,352 kilos of fruits and vegetables" (Secretariat of the Convention on Biological Diversity, 2012). Another approach is tiered trash collection policies. A national survey in the US found "that cities that adopted this policy realized significant reductions in the amount of waste disposed per household and significant increases in the amount of materials recycled, controlling for the effects of other policies and demographic features" (Folz & Giles, 2002). This can significantly help communities battle their solid waste and recycling management issues by forcing consumers to be more mindful of the waste they are producing.

Improving access to transportation: Access to safe, affordable, accessible, and sustainable public transportation systems can help reduce automobile dependence, especially in urban areas, while still allowing for a community to thrive socially and economically. For decades, transportation planning has been focused on automobile-oriented development and many agree that sustainable transportation planning should focus on reducing automobile dependence (Crane, 1996; Deakin, 2001; Kenworthy, 2006; Litman, 2006; P. Newman, 1999, 2006). Car dependency has caused major issues in the amount of congestion and pollution experienced in urban areas and is often inaccessible to poorer households. In 2014, the United Nations appointed a High-level Advisory Group on Sustainable Transport to provide key findings and recommendations on how to develop sustainable transportation infrastructure and systems on a global, national, and local level. A 2016 report noted that transportation planning, policy, and investment decisions should be based on social, environmental (including global climate change), and economic factors. Shifting transportation policies to focus more on access to public transportation systems rather than investing so much on independently operated personal vehicles is the paradigm shift needed to create greener, more sustainable transport options which could be accessible to all (Banister, 2001; Cervero, 1996; Litman, 2013). Other options include temporary road closures and promotion of alternative transportation, such as bicycles. For eample, In Bogota, Colombia, physical activity has increased significantly and GHG emissions have been curtailed by closing 97 kilometers of a major road to traffic on Sundays and during holidays, improving the bus transit system, using cleaner buses, and creating a 334-kilometer bicycle path around the city (Lemoine et al., 2016).

Improve access to green space: As noted previously, green spaces in cities can contribute to NCP provisioning and biodiversity protection, among other advantages. They can also increase GQL. Access to and use of green spaces reportedly reduces stress and mental fatigue. A large study in the Netherlands found that perception of general health depended on the quantity of urban green space (Lee & Maheswaran, 2011a)(Lee & Maheswaran, 2011b). Urban green spaces provide health benefits by offering space for physical activity, which in turn lowers risk for cardiovascular disease, obesity, and mental illness (van den Bosch & Sang, 2017b)(van den Bosch & Sang, 2017a). For children, playing time in or near green or blue spaces has been shown to be positively associated with lower attention deficit and aggression problems (Nadja Kabisch et al., 2017). Conversely, lack of biodiversity and access to ecosystem services has damaging effects on social and cultural relationships, health-related functions, economic values, and environmental factors in urban settings (Dennis, Armitage, & James, 2016; Gómez-Baggethun et al., 2013).

Communities investing in green space also demonstrate a positive relationship between green space and decreased crime (Bogar & Beyer, 2015). Vegetation, mostly in the form of tree coverage and planned landscaping, also has an effective influence on safety. Studies in Portland, Oregon, Chicago, Illinois, and Baltimore, Maryland each independently demonstrate that crime rates were lower in areas with more vegetated coverage, whether it be tree canopy along right of ways or manicured gardens encompassing an apartment complex (Donovan & Prestemon, 2012; Kuo & Sullivan, 2001; Troy, Morgan Grove, & O'Neil-Dunne, 2012). While the type and density of vegetation present in areas of decreased crime rates remains a point of contention, one agreed upon reason for the lower crime rates in places with more green space is simply that those areas attract more recreational use by people and therefore attract more legitimate activity, leaving less inclination of perpetrators to commit a crime (Troy et al., 2012). Crime Prevention through Environmental Design (CPTED) is a multi-disciplinary approach to deterring criminal behavior through environmental design using vegetation, landscaping, and green spaces as a design and planning element to discourage illicit activity (Atlas, 2013).

***Improving participatory planning and governance for inclusion*:** One of the targets of SDG 11 is "to enhance inclusive and sustainable urbanization and capacity for participatory, integrated and sustainable human settlement planning and management in all countries" by 2030 (UN-SDG 11). Brazil's Porto Alegre is a successful case of participatory governance in the face of rapid urbanization, where participatory budgeting can be seen as one of the most innovative areas in urban democracy (Cabannes, 2004). The Porto Alegre case was nominated by the 1996 UN Summit on Human Settlements in Istanbul as an exemplary 'urban innovation' and as "demonstrating an efficient practice of democratic resource management" (World Bank, 2003). Some municipalities in Africa and Europe have adopted participatory budgeting as well, such as Cordoba in Spain and Kerala, South Africa (Heller, 2001; Sintomer, Herzberg, & Röcke, 2008) .

6.5 Integrated approaches for sustainable energy and infrastructure

Several options are available to decision makers at different scales to ensure the sustainability of energy, mining and infrastructure development, as outlined in Section 6.3.6 of Chapter 6 of the Global Assessment. However, many of these options create trade-offs and risks, and raise further challenges of implementation. In this supplementary material, the major obstacles envisaged for the implementation of the different options are discussed.

6.5.1 Financial and environmental risks of biofuel production and potential solutions

Commercial funding of biofuel production includes multiple kinds and strategies, such as pension funds and other private funds, IPOs, joint ventures, and mergers as a way to recapitalize projects (ETIP, 2018; Lane, 2012). Public investors, however, remain central to sector development. National governments provide funding via subsidies, investment incentives, and loans (development banks). Moreover, development aid, foreign investment incentives, and foreign direct investment (FDI) are important investment strategies by the importing states. In addition, multilateral financial institutions offer loans and investments (van Gelder & German, 2011). Biofuels development, however, continues to come with many financial risks, due to the uncertainty of supply chains of feedstocks, rapidly changing (and often poorly understood) commercial conditions and shifting regulatory frameworks (e.g., (Goetz, 2015).

Peer-reviewed empirical literature identifies multiple negative impacts of biofuels production on biodiversity, and related environmental problems, such as air, soil and water pollution (from vinasse, fertilizer and pesticide use, erosion); excess water use or competition with other uses; wetland conversion; deforestation and forest degradation; loss or degradation of wildlife habitat and significant changes in the qualitative composition of species; ineffective riparian zone protection (Ariza-Montobbio & Lele, 2010; Blank, Williams, Sample, Meehan, & Turner, 2016; Danielsen et al., 2009; Fletcher Jr et al., 2011; Hartemink, 2008; Mintz-Habib, 2013; Mohr & Raman, 2013; Z. Qin, Dunn, Kwon, Mueller, & Wander, 2016; Rana, Ingrao, Lombardi, & Tricase, 2016; Reijnders, 2012; Schnoor, 2006; Searchinger & Heimlich, 2015; A. K. Singh et al., 2014; Tomei & Helliwell, 2016; Wani, Garg, & Chander, 2016; Wendimu, 2016; Zah & Laurance, 2008). For example, massive oil palm expansion in Indonesia and Colombia has led to forest conversion, and peatland destruction (Mintz-Habib, 2013); US corn production has reduced vertebrate diversity and abundance compared to non-crop habitats (Fletcher Jr et al., 2011); and the intensive commercial production of sugarcane or soy, and the industrial processing of these feedstocks, have been associated with severe water pollution on the production site as a result of stillage and fertigation, irrigation, plant protection and pest control (Baxter & Mousseau, 2011; Gasparri, Kuemmerle, Meyfroidt, le Polain de Waroux, & Kreft, 2016; Gunkel et al., 2007; Martinelli & Filoso, 2008; Obidzinski, Andriani, Komarudin, & Andrianto, 2012). Research also indicates that biofuels production occurs largely on prime land, reducing food and forest acreage, even in the case of Jatropha which turned out to be economically unviable if grown on marginal land (Rathmann, Szklo, & Schaeffer, 2010; A. K. Singh et al., 2014; Timko, 2014). Next generation biofuels are promoted as a possible way to overcome the negative biodiversity impacts of edible plants-based biofuels. However, 2nd generation biofuels (based on non-edible plant biomass) are confronted with sustainability problems similar to those of the 1st generation: the production of cellulosic ethanol still relies on fossil fuel inputs, and its effect on biodiversity, water, and land conflict is subject to sustainable management and governance of land (Mohr & Raman, 2013). Biodiversity is impacted if co-products were produced from high-yielding food crops, or monoculture energy crops; use of “marginal” land formerly used by the poor tends to aggravate land conflicts; while processing tends to be water-intensive (Mohr & Raman, 2013).

Thus, biodiversity effects of biofuels production are caused by a mixture of natural and anthropogenic processes similar to other agricultural activities, including land conversion and land-use change (direct or indirect), pollution and nutrition enrichment, the introduction of alien species (Bowyer, 2010; Butchart et al., 2010; Lapola et al., 2010; Mintz-Habib, 2013; Sparovek, Barretto, Berndes, Martins, & Maule, 2009). The conversion of forest or other ecosystems to crop plantations tended to reduce biodiversity, whereas the replacement of monoculture row crops with grasslands led to increases in biodiversity (Blank, Williams, Sample, Meehan, & Turner, 2016; Goetz, German, & Weigelt, 2017; Robertson, Hamilton, Grosso, & Parton, 2011). Often overlooked, the conservation focus on tropical forests led to biodiversity losses in dry forests as a result of massive expansion of soy production in Savannas of South America and Southern Africa (where cultivation increased from 20,000 ha in 1970 to 750,000 ha in 2013) (Gasparri et al., 2016). Moreover, biofuels investment related land grabs not only negatively affected customary land users, they also resulted in displacement effects in the form of outmigration of displaced land activities elsewhere, rendering zoning strategies ineffective, both at the national and global scale (de Sá et al., 2013; Godar, Suavet, Gardner, Dawkins, & Meyfroidt, 2016; Hammel, 2015; Schut & Florin, 2015; Van Stappen, Brose, & Schenkel, 2011). At the same time, production often occurs under very poor working conditions (Oliveira, McKay, & Plank, 2017; UNEP, ILO, IOE, & ITUC, 2008).

To better mitigate these risks, in the short-term, production and use should occur at a smaller scale until the detected deficiencies in current regulatory and market-based governance instruments for sustainable biofuels are addressed and corresponding responsibilities and accountabilities clarified (German et al., 2017). Simultaneously, and in the long-term, the negative impacts of land-use changes for the environment and social welfare need to be addressed: e.g., biofuel schemes should take into account the consequences of bioenergy production to other agricultural and socio-economic activities (including forestry and food security); restrict biofuel use to areas with highest efficiency; pro-actively advance labor welfare, and safeguard rural livelihoods.

6.5.2 Limitations of current Environmental Impact Assessment practice

Underlying the value of environmental impact assessment (EIA) is the view that environmental decision-making processes are strengthened when the responsible authority can incorporate the views and opinions of all relevant stakeholders regarding the decision at hand. Since its introduction into US federal law in 1970 (NEPA 1970), EIA has emerged as a near universally adopted policy instrument across all levels of government and international institutions (R. K. Morgan, 2012). EIA is a decision support tool that requires prior identification and assessment of the potential environmental impacts of planned activities through a structured process requiring comprehensive evaluation of impacts and public consultation. EIA processes define "environmental impact" broadly and are intended to require assessment of both direct and indirect impacts across all manner of environmental resources, including biodiversity, as well as inter-related socio-economic, cultural and health related impacts (UNEP 1987).

EIA is well established in international legal instruments and customary international law in relation to environmental impacts generally, and impacts on biodiversity specifically. In connection with the latter, the CBD requires state parties to "introduce appropriate procedures requiring environmental impact assessment of its proposed projects that are likely to have significant adverse effects on biological diversity" (Convention on Biological Diversity, 1992), Article 14). This obligation is intended to be implemented in national and sub-national EIA legislation, and is reinforced by further EIA requirements in other multilateral environmental agreements, such as the United Nations Convention on the Law of the Sea (UNCLOS 1982, Art 206), the Convention on Environmental Impact Assessment in a Transboundary Context (1989), and the Antarctic Protocol on Environmental Protection (1991, Art 8(1) and Annex 1). EIA is also a central issue for negotiation in the proposed multilateral treaty addressing (marine) biodiversity beyond national jurisdictions (UNGA A/RES/69/292, 2015). EIA was recognized as a general principle of international environmental law in the Rio Declaration on Environment and Development (1992, Prinicple 17), and by numerous international courts and tribunals (Craik 2015), as well as the International Law Commission (ILC 2001).

Beyond multilateral treaties, EIA is also integrated into the international finance system through project finance requirements. For example, the World Bank has required EA of Bank financed projects since 1989 (World Bank 1999). The International Finance Corporation and Multi-lateral Guarantee Agency also require EIA for a wide range of project finance activities (PSESS 2012). The project finance activities of private banks may also be subject to EIA through the Equator Principles, a voluntary code of conduct for managing socio-environmental risks of large-scale development projects (The Equator Principles, 2013), Principle 1). At national and sub-national levels, (R. K. Morgan, 2012) indicates that over 180 countries have some form of EIA legislation. It is not uncommon for EIA legislation to exist at both federal and sub-federal levels.

Despite its ubiquity, the effectiveness of EIA as a policy instrument remains a matter of debate within the literature (Cashmore et al. 2010). Two empirical evaluations tentatively concluded that EIA does make a positive contribution to the environmental performance of planned activities (Sadler 1996). Morgan argues that determinations of EIA effectiveness are necessarily context dependant and require careful consideration of the social, economic, political and culture conditions of EIA application (R. K. Morgan, 2012). As a consequence, much of the literature evaluating EIA as a policy instrument of general application tends to focus on identifying best practices in light of particular goals and in relation to specific stages of EIA (IAIA 1999). Nevertheless, challenges and limitations have been identified in recent literature.

A consistently identified challenge for EIA is determining whether the threshold conditions for initiating an assessment - likelihood of significant environmental harm - are present. To address this, EIA instruments may require an EIA be conducted for any activity that is carried out in areas that are proximate to areas of high biological or cultural significance, such as critical habitat for endangered species, World Heritage sites or areas identified as having ecological significance through international or national processes (CBD COP Decision VIII/28, 2006). Another approach is to identify activities (usually in lists appended to EIA regulations) known to have a high likelihood to cause biodiversity impacts, for example, linear infrastructure that causes ecosystem fragmentation, and require EIAs be carried out for those classes of activities. In effect, the determination of a likely significant impact is pre-identified by reference to activities or geographic areas. In addition to using lists, or as an alternative, EIA instruments may identify generic screening criteria that account for biodiversity values at multiple levels, directing decision-makers to consider potential impacts on ecosystem functions, species population levels, as well as effects on sustainable yields or rights of access over biological resources (CBD COP Decision VIII/28, 2006).

Another significant challenge to determining preferred outcomes is that project evaluations involve complex trade-offs between environmental, social and economic goals, but EIA instruments provide limited substantive direction on how competing goals are to be managed (Holder 2004; Gibson 2006)). The evaluation of impacts is often made with reference to hierarchy of planning outcomes that prioritizes avoidance over mitigation, which in turn is preferred to compensation measures, including offsets (IAIA 2005; CBD COP Decision VIII/28, 2006). However, adherence to this hierarchy has been identified as an issue, with EIAs focusing excessively on mitigation and the expense of prevention (Eales 2011). The use of the concept of “no net loss” or “net gain” of biodiversity as a basis for decision-making is gaining purchase in EIA policy (PSESS 2011). In instruments that adopt this approach, proponents are required to demonstrate that the proposed activities does not result in an overall loss of ecological function, but in doing so, may rely on compensatory mechanisms, such as offsets. The use of offsets as a response to identified impacts in an EIA requires assessment of the adequacy of the offset, including its ecological quality and whether the offset is additional (Kiesecker et al. 2009; Brownlie et al. 2013), as well as any distributional consequences that arise from the shift of ecological functions from one location to another (Rajvanshi 2010).

The predictive structure of EIA process has severe limitations in the context of ecological systems that are dynamic and where impacts are non-linear (Holling 1978). For example, EIAs for road and other linear infrastructure development are typically too short-term and superficial to detect rare species or assess long-term or indirect impacts of projects (Flyvbjerg, 2009; Laurance & Burgués Arrea, 2017; Ritter et al., 2017). Such assessments are frequently myopic, considering each project in isolation from other existing or planned developments (Laurance et al., 2014). Hence, EIA alone seems inadequate for planning infrastructure projects and assessing their broader environmental, social, and financial impacts and risks (Alamgir et al., 2017; Alamgir, Campbell, Sloan, Phin, & Laurance, 2018; Laurance, Sloan, Weng, & Sayer, 2015).

The application of the precautionary principle to EIA is accepted across EIA instruments, requiring decision makers to identify areas of uncertainty and to give clear consideration of the implications of knowledge gaps for the assessment. A further response to uncertainty has been the incorporation of adaptive management into EIA instruments through the requirement for post-decision monitoring and follow-up measures (CBD COP Decision VIII/28, 2006). In its most robust form, adaptive management takes advantage of planned interventions in natural systems as the basis for experiments that are used to reduce uncertainty. There is considerable debate respecting the conditions under which projects with uncertain outcomes may proceed on the basis that unanticipated impacts may be addressed through adaptive management (C. R. Allen & Gunderson, 2011) (Rist et al. 2013; Allen & Gunderson 2011). Key factors that have been identified include the degree of risk associated with an activity (should adaptive measures be inadequate), and the ability of resource managers to control an activity once it has been established (C. R. Allen & Gunderson, 2011) (Allen & Gunderson 2011; Craig & Ruhl 2014).

6.5.3 Challenges of implementing compensation and offsetting policies

Biodiversity compensation is an umbrella category of policy instrument under which different mechanisms may fall, including compensatory mitigation, biodiversity offsets, mitigation banking, habitat banking, species banking, and wetlands mitigation, often driven by requirements for ‘no net loss’ (NNL) of biodiversity. These policies have been promoted by different actors, concern different subjects (biodiversity, species, habitat, wetland, marine, etc.) and operate on different scales and with a variety of forms ranging from regulatory to voluntary that aim to mitigate the loss of biodiversity by development.

Much biodiversity compensation is driven by a ‘mitigation hierarchy’: avoid, minimize, restore and offset (Kiesecker et al. 2010). An analysis by (ten Kate & Crowe, 2014) found that 39 countries have existing laws or policies on achieving NNL for biodiversity in the context of development projects. There are some differences between the terms biodiversity mitigation, biodiversity offsets, and biodiversity banks (Bull, Suttle, Singh, Milner-Gulland, & Gordon, 2013). Compensatory mitigation is the regulatory requirement that any loss of biodiversity by development be mitigated in some way, either through payments to a government fund or through funding of protection elsewhere through habitat ‘offsetting’, which is a process by which lands lost to development are ‘replaced’ by restoration (an “enhancement offset”) or protection elsewhere of areas that are susceptible to loss (an “averted-loss offset) (Bull et al., 2013; Norton & Warburton, 2015). The Business and Biodiversity Offsets Programme describes offsets as “measurable conservation outcomes resulting from actions designed to compensate for significant residual adverse biodiversity impacts arising from project development… to achieve no net loss and preferably a net gain of biodiversity on the ground with respect to species composition, habitat structure, ecosystem function and people’s use and cultural values associated with biodiversity” (BBOP 2012). According to Calvet et al. (2015), this is still one of the most-frequently cited non-academic references on biodiversity offsets. A more recent definition by IUCN is that “conservation actions intended to achieve offset outcomes must result in a direct measurable biodiversity gain equivalent to the residual loss arising from the impacts on biodiversity associated with a project in order to be considered a biodiversity offset” (IUCN 2016). These tradeable offsets can allow one sector (such as infrastructure development) to use offsets in another sector (such as nonpoint source pollution) to mitigate impacts (Bull et al. 2013; Bell 2016). Conservation easements can also fall into the category of offsets. Biodiversity banking is a process whereby biodiversity offsets are implemented in advance, by third parties, to be subsequently “sold” to developers with offsetting requirements set by permitting authorities (e.g., Vaissière et al. 2015; Levrel et al. 2017). A “mitigation bank” holds sites where resources are restored or conserved, and which are treated as credits to be purchased by developers with obligations for compensatory mitigation (Froger et al. 2014).

Biodiversity banking and offsetting schemes are increasingly promoted as innovative financing mechanisms to access and direct private sector resources to achieve Aichi target 3 and 4 (OECD 2013, 2016). Although they are often associated with market-based approaches, the literature notes that these offset programs are on the contrary using limited features of market governance (Lapeyre et al. 2015) and require a strong regulatory environment in which to operate (Coralie et al. 2015). In Europe, compensation uses a combination of hybrid mechanisms (Froger et al. 2014), while in Latin America the regulatory framework remains dominant (McKenney & Kiesecker 2010; Saenz et al. 2013), and in Africa, voluntary compensation programs have expanded (McKenney & Kiesecker 2010). The value of these programs was estimated conservatively in 2010 at around US$3 billion, covering around 86,000 ha of land (Madsen et al. 2010). Specific national laws for NNL and biodiversity compensation include:

- US federal and state laws (Clean Water Act Section 404 on no net loss of wetlands; 2012 Presidential Executive Order 13604; state regulations such as Colorado requirement for offsets from oil & gas development (Levrel et al. 2017).
- The EU Habitats Directive states that there be no net loss for Natura 2000 sites, and therefore compensation for loss of habitats/biodiversity of threatened species (Mckenney and Kiesecker 2010), such as in the UK, where port development has required the offsetting of habitats for intertidal wetlands elsewhere (Madsen et al. 2011).
- Canadian fisheries regulations state that any development with impact on fish habitats must be offset (Quigley & Harper 2006a; 2006b), and offsets have been used for boreal forest development as well (Weber et al. 2015).
- Germany has public biodiversity banks which are intended to “offset environmental damage caused by the implementation of public development plans carried out by local authorities” (Froger et al. 2014).
- Australia has both federal and state level (New South Wales, Victoria, Western Australia, South Australia, and Queensland) offset programs, including the BioBanking program involving both species and land credits in New South Wales (Hillman & Instone 2010; Coggan et al. 2013; Miller et al. 2015).
- Brazil has federal legislation that requires industrial developments to offset their environmental impacts through payments to the National Protected Areas System (Brazil Fed. Law 9985, Decree 4340) (Mckenney & Kiesecker 2010).
- Colombia has passed Resolution 1517 of 2012, which says that mitigation and offsetting need to be part of any license granted (e.g., for oil exploration) by the Ministry of Environment and Sustainable development (Saenz et al. 2013).

Offsets have also been suggested as a way to fund protected areas, given generally inadequate levels of financing worldwide; however, this challenges additionality standards for most offsets (Githiru et al. 2015), and thus there are questions about whether offsets should be used to potentially meet Aichi Targets for protected lands (Maron 2015).

In addition to compliance markets, there are also voluntary markets for biodiversity offsets (Brock 2015); for example, large multinational corporations like Rio Tinto have pledged no net loss policies in their own operations, achieved primarily through offsetting (Mckenney & Kiesecker 2010). One of the main drivers of biodiversity offsetting has been conforming with loan conditions for project finance and safeguards policies of at least 80 financial institutions (including many bilateral development banks) that have variously formulated requirements for projects to achieve no net loss (NNL) or net gains (NG) of biodiversity (Rainey et al. 2015). In this context, principles and standards on biodiversity offsets have been developed and are widely available to practitioners and decision-makers. In the context of project financing, biodiversity offsets are generally considered as voluntary; however, referring to them as market-based instruments can be a source of confusion. Most biodiversity offsetting occurs in the context of “command and control” regulations, in the context of government permitting of projects as noted above. Many compensatory schemes allow developers to ‘do it yourself’ for offset or restoration; to pay into a general fund; or to buy a permit or credit from a third party in the form of a biodiversity bank.

The US, which has the longest history with offsets, has moved over time toward third party offset providers as the most successful and efficient. These biodiversity banks have taken multiple institutional forms: private non-commercial, private commercial, hybrid commercial, public commercial and public non-commercial (Froger et al. 2014) (see Table 6.2). A 2010 review found 39 different habitat or land banks around the world, most of which were in developed countries (US, Canada, Australia, New Zealand primarily; Europe has been slower to uptake this instrument, although Germany has an Impact Mitigation Regulation) (Madsen et al. 2010). These biodiversity offsets enable a sector of offset providers to emerge, thus diversifying the stakeholders involved in private sector involvement in conservation. Robust institutional arrangements are required to ensure company commitments are actually implemented, including encouraging avoidance, minimization and restoration, not just offsetting (Vaissière & Levrel 2015; Gordon et al. 2015).

Table 6.2: Examples of biodiversity banks and offset programs

Country	***Name***	***Description***	***Coverage***	***BES goals***	***References***
Australia	BushBroker, BushTender, PlainsTender, BioBanking	Compliance market at state and regional levels	~25,000 ha;	IAS management, preservation of native vegetation	Hillman and Instone 2010
USA	Wetland trading program under CWA	Compliance market – national	280,000 ha restored or protected by 2010	Ecosystem function in wetlands	PAWLICZEK and Sullivan 2011
Malaysia	Malua BioBank	Voluntary	34,000 ha of rainforest	Maintenance of habitat for 50 years through selling biodiversity conservation certificates	Brock 2015

Multiple authors have identified key issues in developing offsets policies and banks: (1) there must be equivalence of project impacts with offset gains, preferably in ‘like for like’ exchange (Bidaud et al. 2014), which has been a complicated and difficult process due to “incomplete measurement, imprecise valuation, and non-interchangeability” (Walker et al. 2009); (2) the location of the offset relative to the impact site often varies, but defined ‘service areas’ near development sites are preferred in most policies; (3) ‘‘additionality’’ in that there should be a new contribution to conservation or restoration made possible by the offset; (4) timing of project impacts versus offset benefits must be resolved, with many policies expressing preference for an offset to be established prior to project impacts; (5) offset duration and compliance, with some policies requiring offsets in perpetuity which others using more flexible definitions; and (6) resolving what the ‘‘currency’’ (or tradeable item) is, and mitigation replacement ratios (Mckenney & Kiesecker 2010; Spash 2015; Maron et al. 2016). There have been additional challenges in developing countries in implementing offsets, including lack of regulations and concerns about enforcement; difficulty raising funds; concerns about the livelihood impacts of shifting conservation actions away from areas of development and into poorer communities (Spash 2015; Maron et al. 2016; Bidauda et al. 2017).

Froger et al. (2014), Van Teeffelen et al. (2014) and Gordon et al. (2015) outline some of the risks involved in banking and offsetting, concluding that each type faces limitations and challenges, and a number of key risks are noted in Box 6.1. For some authors, biodiversity offsetting is seen by some as a “licence to trash” that risks accelerating biodiversity losses (Walker et al. 2009; Spash 2015), with the often-stated goal of no net loss (NNL) or net gains (NG) of biodiversity considered as meaningless (Maron et al. 2015b; Virah-Sawmy et al. 2014). Also, some impacts are so significant that they may not be acceptable to society, or cannot be offset, owing to the high risk of failure (Maron et al. 2012; Pilgrim et al. 2013). Projects in such cases would in principle not be permitted, but there are numerous counter-examples that feed strong opposition to private sector involvement in conservation. Strong criticisms of offsets have been encountered, particularly by those who have argued that biodiversity should not be traded or should not be valued by markets, as it turns nature into a commodity (Madsen et al. 2010; Sullivan & Hannis 2015; Brock 2015). For example, in 2013, more than 140 organizations signed a statement opposing biodiversity offsetting in the UK (http://no-biodiversity-offsets.makenoise.org/list-of-signatories/) including indigenous peoples' groups, and anti-globalization and climate activists (Sullivan & Hannis 2015). The concern is that offsetting contributes to a commercialization of nature, although other authors have noted that much of the work of mitigation banking occurs on private, rather than public lands, and therefore such lands have already been commodified (Levrel et al. 2017). Nonetheless, it is clear that there are often incommensurable views on biodiversity offsets, as revealed in controversies over the UK use of this policy (Sullivan & Hannis 2015).

Box 6.1. Questions about offsets 'design' and operation (Spash 2015)

Baseline scenario: what is the current state of biodiversity?
Additionality: what does the offset site add that would not have occurred anyway?
Comparability: how far is the offset site equivalent with the original site and on what basis?
Measurability: how are characteristics of importance to be measured (i.e. metrics) and what about things that cannot be quantified or measured?
Commensurability: can all the objects of value be measured on the same basis?
Complexity: how much ecosystem complexity is permissible before offsets become infeasible?
Time: over what time period will the offset scheme deliver and be maintained?
Space: where should the offset site be located relative to the original site?
Uncertainty: what approach is taken to the unknowns and the unknowables?
Measure of last resort: is the mitigation hierarchy going to be strictly employed so that offsets only occur after harm has been avoided, mitigated and/or rehabilitated?
Enforcement: what mechanisms are going to ensure monitoring and performance?
Transaction costs: who will cover all the set-up and running costs involved, and are they less than alternatives e.g. direct regulation?
Liability and severance: what will be the responsibility of the developer for ensuring the quality of the offset and can they be held responsible for failure, or will offset purchase be used to claim they complied regardless of any actual change in say biodiversity?
Speculation: will trading of credits result in financial speculation and price manipulation for rent seeking and profiteering.
Financialization: will there be a divorce between traded credit value and the physical reality to which credits relate?

Outcomes of offsets and banking

Surveys of US banks have hown average satisfaction of service buyers with benefits provided by banks (Froger et al. 2014; Fox & Nino-Murcia 2005). However, there is little assessment in the literature of the impacts of offsets on IPLCs, or the role of including cultural values of land in offsets. Only a handful of studies have looked at the local impacts of offset projects, including one study in Madagascar that assessed the impact to a local community of an offset project that required them to restrict their forest use, and thereby was considered a net negative for them (Bidauda et al. 2017). It can be further noted that developers who buy such offsets tend to be more powerful actors than the IPLCs who might be impacted by such actions (Apostolopoulou & Adams 2017). Others have noted that the concept of substitutable biodiversity "reframes biodiversity as lacking locational specificity, ignoring broader dimensions of place and deepening a nature–culture and nature–society divide"(Apostolopoulou & Adams 2017), which is likely to be especially problematic for IPLCs. Several studies also note that the nature of mitigation banking, which shifts protection from one area to another, involves distributional risks. For example, development "impacts tend to be located in urban areas, while compensation sites tend to be located in rural areas, with migration of wetland resources to less densely populated areas" (Levrel et al. 2017). Surveys have indicated local support for offsets that tends to be nearer, rather than farther away, from project sites (Burton et al. 2017).

More participatory processes of offset definitions and politics have been proposed to address these challenges (Mann 2015).

In order to judge environmental effectiveness of offsets, authors note the need for attention to criteria of additionality, equivalence and compliance (Coralie et al. 2015; Mckenney & Kiesecker 2010). Concerns have been raised that purchasing development easements in banks is not equivalent to the original land that may have been destroyed as biodiversity or wetlands are not easily substitutable (Wissel & Wätzold 2010; Quétier & Lavorel 2011). Similarly questions have been raised about the efficiency of banks in achieving no net loss goals (Benabou 2014). For example, reviews of mitigation permits in North America have shown a lack of enforcement, resulting in widespread failures of permits to achieve even the minimal levels of biodiversity restoration or protection (Matthews & Endress 2008). Others have made the argument that the existence of offsets "can facilitate planning permissions that might otherwise have been refused" (Apostolopoulou & Adams 2017), thereby inadvertently encouraging risky development.

There is no comprehensive review of the effectiveness of banks or offsets at halting biodiversity loss or provisioning ecosystem services, and reviews of specific offsetting programs in specific ecosystems have shown mixed results (Quigley & Harper 2006a; Matthews & Endress 2008; Sochi & Kiesecker 2016). For example, a review of the Canadian fisheries no net loss of habitat program found that compliance with biological requirements of offset projects was very low, especially in riverine habitats (Quigley & Harper 2006a). Further, the piecemeal nature of offsets, which often do not fit priority plans for nature conservation, may mean that the highest areas for biodiversity protection are not included in offset priorities (Kujala et al. 2015). In practice, regulations in different countries, corporations, and conservation organizations vary greatly in the biodiversity outcomes that they hope to achieve from offsets. For instance, the scope can range from biodiversity in general (all of biodiversity) to very specific biodiversity components such as a limited number of protected or threatened species, certain wetland types or other designated features (Quétier & Lavorel 2011). In practice, a balance must be struck between limiting substitution and establishing a currency that is fungible enough to facilitate exchange. Thus, biodiversity metrics for assessing losses and gains often rely on surrogates of broader biodiversity (often habitat-based) and indicators for biodiversity of the highest conservation concern (e.g. rare and restricted biodiversity), which may be poorly represented by surrogates (Quétier & Lavorel 2011; Maseyket al. 2016). There is a discussion in the literature on what types of biodiversity values can be substitutable (e.g. calculated and offset) and which should not (e.g. irreplaceable or highly endangered species) (Pilgrim et al. 2013). Thus habitat or ecosystem offsets have been particularly difficult to find 'units' of substitution for, and have proven more difficult than pollution offset markets; for example, 'hectares of habitat' are often accounted for or traded, but these cannot capture all indicators of biodiversity and are often not broadly equivalent (Bull et al. 2013). For impacts with a low significance in terms of biodiversity conservation, a simplified approach can be preferable in order to avoid high transaction costs. Societal values concerning biodiversity should be captured within offset goals, but there are risks in conflating biodiversity and ecosystem services (Jacob et al. 2016).

To achieve no net loss policies, conservation outcomes from biodiversity offsets must also be 'additional', given an explicitly stated frame of reference. NNL can be evaluated compared to some fixed state of biodiversity, such as the present state or a desired future state (this is frequently the case in regulated environments), but a dynamic counterfactual could also be used (what would have occurred through time without the impacts targeted by the policy?) (Maron et al. 2015; Maron et al. 2018). Another frame of reference could be a future in which the NNL policy did not exist, but the impacts occur (i.e. business as usual). This would lead to claims that a net gain is achieved because 99 hectares of forest was removed, rather than 100 had there been no policy. The chosen frame of reference against which NNL is evaluated is therefore crucial in determining the outcome of biodiversity offsetting and banking policies. The issue of additionality explains concerns around cost-shifting when support from the private sector are targeted at pre-existing protected areas (Pilgrim & Bennun 2014, Maron et al. 2015a; Maron et al. 2018).

Given the challenges and stakes in expanding the use of biodiversity offsetting, it would be helpful for additional research to provide a more in-depth and critically rigorous comparative analysis of international experience with offset measures, offering evidence on whether they have achieved their policy objectives, such as no net loss or net gains of biodiversity (Quétier & Lavorel 2011; Maron et al. 2015b; Maron et al. 2016; Kujala et al. 2015) (see Box 6.2).

Box 6.2. Priority actions for improving transparency and effectiveness of biodiversity offset policy (Maron et al. 2016)

(1) Develop and embed in policy clear and specific guidelines for how to implement the avoidance and minimization steps in the mitigation hierarchy, along with examples and a requirement to document the steps taken.
(2) Provide an explicit statement of the frame of reference against which offset goals such as no net loss are to be achieved in order to increase transparency, clarity for developers, and public acceptance and ensuring that counterfactual scenarios are both consistent with this frame of reference and periodically revised.
(3) Encourage more strategic approaches to offsets in situations in which multiple offset trades are likely and create policy structures and incentives to generate a supply of banked offset credits to help reduce uncertainties and time lags.
(4) Establish independent oversight and auditing of offset schemes to improve the transparency and effectiveness of governance and the equitable sharing of costs and risks between parties.
(5) Allow free public access to a register that describes how offset actions are achieving their promised outcomes to encourage scrutiny of policy effectiveness."

Species mitigation banks

Species mitigation banks are a specific form of tradable permits that allow development to destroy habitat of endangered species in one place as long as additional habitat for species is maintained or created somewhere else (and thus are similar to offsetting with regard to land development), or some other actions are taken to conserve species elsewhere (e.g. funding for breeding programs, etc). Species mitigation policies can be found in the United States, Australia, Germany, Canada, Brazil, UK, France, Sweden, Spain, Japan, and Namibia (Madsen et al. 2010). For example, the US Endangered Species Act requires any development that impacts on species to be offset with mitigation obligations elsewhere, generally for habitat protection or restoration, although there are some instances of direct species activities (e.g. funding breeding programs) (Bonnie 1999). Landowners can also apply to create 'species banks' on their own lands and thus profit from conservation activities (Madsen et al. 2010; Carroll et al. 2008), and mitigation banks have accompanied conservation easements in many parcels in the US.

In the US, there are around 60 species total currently available in 120 banks (with a value of $200 million in 2010), including the California red-legged frog, the California gnatcatcher, the San Joaquin kit fox and the Western burrowing owl (Madsen et al. 2010). Certain states (Colorado, California) are also active in allowing species mitigation through the use of offsets and banks (Sochi & Kiesecker 2016). The EU Habitats and Birds Directives have required similar attention to mitigation for threatened species in Europe, such as from wind farm development (Madsen et al. 2010; Regnery et al. 2013). Australia has a Koala offsets program in Queensland to provide for loss of habitat due to development or vehicle mortality (Madsen et al. 2010). There is also increasing interest in offsetting among private companies, such as multinational mining operations, even in the absence of national requirements (Kormos et al. 2014).

Development actions that have required species mitigation include expanding urbanization (Burgin 2008), renewable energy farms (which have impacted endangered bird populations in particular) (Cochrane et al. 2015), oil and gas extractions (Doherty et al. 2010), logging and mining (Kormos et al. 2014), and marine fisheries offtakes of seabirds (Pascoe et al. 2011). Units for sale can include either areas of habitat or species-specific credits (e.g., breeding pairs), although in a survey of the SpeciesBanking.com website, scholars found most units for sale were hectares of habitat (Pawliczek & Sullivan 2011). For the US Endangered Species act species banks, landowners are required to engage in “preservation, management, restoration of degraded habitat, connecting separate habitats, buffering already protected areas, and creation of habitat” (US Fish and Wildlife Service 2003). In offsets created by private companies in Africa to mitigate impacts of mining and dams on great apes, offset activities included monitoring of species, creation of awareness programs, and new hunting controls on local populations (Kormos et al. 2014).

Marine habitats are another site where conservation offsets have been used and these face different challenges than terrestrial habitat. One example of marine offsetting is in Australia, where federal law requires any coastal development impacting on the Great Barrier Reef World Heritage Area to be offset someplace else (Bos et al. 2014). Marine offsets are particularly challenging because many marine species may feed and breed far away from offset areas, and there are seasonal fluctuations in population numbers unrelated to offset activities. It may also be difficult to require long-term easements or protections in marine areas (as compared to land purchases) as marine areas are not generally considered ‘for sale’ with the same property rights as terrestrial habitats (Bos et al. 2014). Another challenge has been developing compensatory mitigation for marine bycatch (CMMB) so that fisheries operators bear the cost via levies for conservation activities elsewhere (Wilcox & Donlan 2007; Wilcox & Donlan 2009); there have been concerns that although this may seem an innovative strategy to reduce bycatch losses, CMMB is likely to be effective only “when applied to short-lived and highly-fecund species (not the characteristics of most bycatch-impacted species) and to fisheries that take few non-target species, and especially few non-seabird species (not the characteristics of most fisheries)” (Finkelstein et al. 2008).

The literature suggests that species mitigation and banking has an unclear impact in preventing local extinctions or population impacts of development (Pawliczek & Sullivan 2011); there are some successes where populations have been better off in new areas than those lost to development (Murata & Feest 2015; Pickett et al. 2013), but this has required extensive monitoring and oftentimes a larger expansion of mitigation area than that which was lost. There is no clear understanding if ‘in-kind’ (habitat lost is restored in a similar place) versus ‘out of kind’ (a loss of one species is traded against another, or a loss of habitat is traded for restrictions on hunting elsewhere) offsets are preferable (Bull et al. 2014), as this depends on species. Methodologies used to assess standards for no-net-loss of species also vary across country or project or policies, making comparisons difficult (Bull et al. 2014). The lack of data on banking remains an important barrier to evaluating these policies. A review of the effectiveness of the US Endangered Species Act permit process (the longest-standing biodiversity bank policy) found there is no centralized tracking of species banks and no clear way to assess biodiversity outcomes or impact across species (Madsen et al. 2010).

Other scholars have questioned if banking or offset programmes provide a “legitimization of degradation due to development elsewhere” (Pawliczek & Sullivan 2011) and that the marketization of species in banks may lead to a decline in more formal regulatory rules (Pawliczek & Sullivan 2011), although this potential has been welcomed by communities and scholars who have previously seen laws like the ESA be too inflexible for property owners (Ruhl 2003). In terms of cost effectiveness, the literature is not clear. In some cases offsets can be much less costly than other forms of regulation (for example, in New Zealand, requiring fisheries closures due to endangerment of other marine species was 10 times as expensive as an offset to reduces seabird mortality at a separate nesting site through rat eradication (Pascoe et al. 2011), while in other cases species protection plans have been more time consuming and costly than anticipated. Skeptics of species banking have argued that lack of understanding of time lags, uncertainty and measurability of the values being offset suggest a pause on offsetting until more is known (Maron et al. 2012).

Biologists have suggested that the cumulative impacts of development on species are under-considered in offset plans; further, several authors suggest that “it is difficult to compensate for loss of threatened species or their habitat, particularly for species with very specific requirements, species’ whose requirements are poorly known, or where suitable habitat cannot readily be recreated” (Vanderduys et al. 2016). Many endangered species require old growth habitat to thrive and thus are unlikely to be protected with offsets aimed at restoring systems or offsetting that replaces old growth with different habitat types (Vanderduys et al. 2016; Maron et al. 2010). Species banking is especially inappropriate for those taxa "that are highly vulnerable, endemic to a small area, exhibit low resilience, and provide essential community benefits" (Bos et al. 2014). One study found species richness in offset areas in France was much lower than in impacted areas, even after development of those sites (Regnery et al. 2013). The fact that any species requiring protection often move through landscapes has required attention to the idea of 'dynamic' and temporary habitat protections which can be challenging to develop (BenDor & Woodruff 2014). The literature suggests key areas of attention unique to biological species to be offset, including: "(1) limits to what can be offset, (2) adherence to the mitigation hierarchy, (3) additional conservation outcomes, (4) landscape context, (5) no net loss, and (6) long-term outcomes" (Kormos et al. 2014).

Innovative private finance for infrastructure: Green bonds

There is a diverse group of issuers of green bonds from both public and private sector players, including from stakeholders as commercial banks; the corporate sector; development banks; municipal sector; asset-backed securities; and sovereign wealth funds. At the initial stage, public institutions, such as development banks, helped to establish the market, thus paving the way for private companies to enter the market (Mathews & Kidney 2012). The bulk of the green bond market is in fact bonds issued by pure-play players, which are environmentally-friendly, but not actively labelled as green bonds (Talbot 2017). Non pure-play players, such as banks, tend to use the green label to signal to investors the environmental-friendly nature of the issuance.

European issuers continue to play a dominant role in the global green bonds market. Western Europe issued a total of US$ 195 billion green bonds (labelled and unlabelled), the highest issuing region in 2016 (Climate Bond Initiative 2017). While market demand drives the green bond market in Europe, supportive policies also play a prominent role. For example, the European Green Securities Steering Committee, which was inaugurated with the support from the Climate Bonds Initiative (CBI), the European Covered Bond Council (ECBS) was set up with the explicit goal of facilitating the growth of a climate finance market in Europe. In recent years Asian issuers, particularly China, India and South Korea, are beginning to assert their presence in the market. In fact, China-based issuances accounted for more than a third of the total issuance in 2016, amounting to an issuance of US$ 32.9 billion (SEB 2017) Industry forecasts points to further expansion in the Chinese green bond markets as the demand for green finance is expected to reach RMB 2,908 billion (~USD 465 billion) a year (Zheng 2015). While Indian issuances only accounted by less than 3% of total issuances in 2016, their share is expected to rise in the next few years in response to their USD 2.5 trillion green investment needs by 2030 to fulfil their Nationally Determined Contribution (NDC) goals (Climate and Development Knowledge Network Report 2016).

Institutional investors, who have an estimated USD 100 trillion assets-under-management (AUM), have an important role to play in creating an environmentally friendly investment landscape (OECD 2016). Recent developments in UNFCCC COP21, wherein a group of institutional investors with over USD 11.2 trillion AUM signed the Paris Green Bonds Statement and reiterated their commitment to establish a function green bonds market, are expected to provide a much-needed push for the nascent market. As an investor-driven move towards green investment mandates and stronger transparency and green requirements are experienced in the West and drive demands in the West, policy will drive investor demand in Asian countries. It is also important to remember that the investor pool for green bonds is not limited to environmental, social and corporate governance (ESG) investors that are driven by green mandates. As such, conventional investors are also seen to be active participants in the market in the future.

There are several keys to expanding use of green bonds. Maintaining the environmental integrity and preventing greenwashing, especially with the active involvement of non pure-play players, remains a concern. This issue has been managed by a mix of public and private sector initiatives. In the private sector, voluntary standards, such as the Green Bond Principles and Climate Bonds Standard have laid out some qualifying characteristics of what is deemed as a credible green bond (ICMA 2016). Such voluntary market-based guidelines have been supplemented by public sector guidelines, such as the

People's Bank of China Green Bond Guidelines and Endorsed Project Catalogue and the Securities Exchange Board of India (SEBI) guidelines for Green Debt Securities (GFC 2015; Agrawal 2015; SEBI 2015). Recognizing the potential of different shades of green in green bonds, the market is also moving to establish ranking methodologies to distinguish between different levels of environmental-friendliness of green bonds. One of the proposed methodologies is the Carbon Yield methodology, which measures the emissions abated through the proposed green bond (Baker 2017). Another decision that will enhance the environmental integrity of green bonds is the transparency of disclosure through a clear accounting trial for measurement, reporting and verification (MRV) purposes. Public issuances from development banks are often held to high standards, establishing best practices for private sector players, and setting standards for domestic regulating entities to follow.

Post-mining Restoration and Performance Bonds

Biodiversity conservation and management is an international sustainable development principle for mining and a key defining element of successful mine closure, especially in remote and rural locations (ICMM 2003, 2008). Mining is a transient land use and post-closure the land should be returned to a safe, stable and productive land use with suitable financial provisioning set aside to cover the costs of closure and restoration (ICMM 2006; Sassoon 2009). While post-mining restoration will normally be required of mining proponents by regulators, additional financial assurance instruments are also necessary to ensure governments and communities will not be left with unfunded environmental and financial liabilities in the event of unplanned closure or abandonment of a mine-site (ICMM 2005; Sassoon 2009; Morrison-Saunders et al. 2016). Performance bonds have been a common financial surety in use worldwide. The aim of such a bond is to make the financial risk to government of the private mining operation negligible. Often however, the money held in a bond will be less than the actual cost of mine site rehabilitation were a government needed to call upon it (Gorey et al. 2016).

An alternative approach is the use of a central pooled fund, with revenue into the fund generated by contributions through annual, non-refundable compulsory industry levies upon tenement holders according to the environmental disturbance existing on a tenement (Gorey et al. 2016). In Western Australia, a Mining Rehabilitation Fund (MRF) was adopted in 2013 in place of performance bonds. The fund has quickly grown because of the number of established operations joining the scheme, reflecting an extensive and well-established industry that has been operating for well over one hundred years. The large number of operational mines makes the central fund sufficiently large to be useful without putting an excessive financial burden on individual mining companies (Morrison-Saunders et al. 2016). Benefits to all stakeholders have accrued from this approach such as:

- substantially reduced financial risk for the Western Australian government (Office of the Auditor General 2014);
- costs to industry are substantially reduced, and operators of existing mine-sites benefitted by having their performance bonds returned to them (Gorey et al. 2016);
- a perpetual funding arrangement is now in place for historically abandoned mines –interest earned on the MRF (which is held by the government as a dedicated fund for mine closure surety) is used to restore legacy abandoned sites (over 11,000 such sites exist) in Western Australia (Gorey et al. 2016; Morrison-Saunders et al. 2016);
- the mining industry is now incentivized to: (i) minimise the extent of land clearing for their operations, and (ii) initiate progressive rehabilitation through the mining lifecycle as the annual fee payable for land under restoration is substantially less than for disturbed areas (Gorey et al. 2016); and
- there is full transparency of the environmental footprint and areas under rehabilitation with annual reporting by the Department of Mines and Petroleum publicly available online (Office of the Auditor General 2014; Gorey et al. 2016).

While insufficient time has lapsed to fully understand the benefits of a central rehabilitation fund for mining environmental securities, initial indications are that biodiversity protection is likely to be enhanced under this approach.

6.5.4 Ensuring access to energy for all by promoting community-led initiatives

Obstacles of community-led renewable energy projects reported in the literature include financial, technical, infrastructural, institutional, governance and socio-cultural obstacles, as well as geographical limitations. High investment costs, lack of financing and suitable credit opportunities, and long pay-back periods appear in both developed and developing countries (Geels et al. 2016; Luthra et al. 2015; Nasirov et al. 2015; Rosso-Cerón & Kafaron 2015). Technical or infrastructural obstacles usually stem from technological lock-ins (i.e. centralized infrastructure (Goldthau 2014)) or the lack of local and national infrastructure (Ahlborg & Hammar 2014; Luthra et al. 2015). In addition, lack of human capital and weak entrepreneurial skills in developing countries often result in low maintenance (Chauhan & Saini 2015; Ahlborg & Hammar 2014). Many studies mention slow and bureaucratic administration, weak institutional frameworks, and missing or unclear (governmental) financial incentives as additional limitations, which might act as barriers for new market entrants in developed countries (Geels et al. 2016; Boon & Dieperink 2014), or dissuade investors in developing countries, especially if accompanied by political instability (Nasirov et al. 2015; Luthra et al. 2015). Research suggests that small-scale renewable development projects often fail because they do not take into account social and cultural characteristics of isolated or marginalized communities (e.g., lack of know-how, awareness and credible information, gendered qualities of energy use etc.) and cannot meet local needs (Byrnes et al. 2015; Urmee & Md 2016; Reames 2016), which might further strenghten donor-dependency, especially in developing countries (Ahlborg & Hammar 2014). Existing studies also show that energy transition in communities can further deepen social conflicts and injustices (Nightingale 2002; Jones & Boyd 2011; Park 2012).

Run-off river hydropower as community-led energy initiative
A large diversity of scholarly studies shows that citizen's inclusion to renewable energy production and distribution contributes to behavioural change towards more sustainable energy consumption and provides more affordable energy access in some countries (Schreuer & Weismeier-Sammer 2010; Rijpens et al. 2013; Kunze & Becker 2015; Islar & Busch 2016). Also, in rural areas, community based management has emerged as a way to ensure access to clean, reliable and appropriate energy and to strengthen the local governance structures. For example, in Nepal, due to strong village organization and forest user groups as well as due to its mountainous terrain, there is a high potential to generate significant amounts of energy through renewable sources, especially from run-off river hydropower. There are several community-based initiatives that promote the adoption of a decentralized renewable energy management. It is claimed that such an inclusive and co-operative approach with local governance structures can extend the access to and availability of renewable energy to all community members with particular attention to vulnerable members like women, Dalits (often viewed as the lowest social caste) and indigenous people (UNDP 2012). However, scholars like Nightingale (2002) and Jones & Boyd (2011) have also pointed out the asymmetrical power relations that exist within communities on the basis of wealth, status, gender and caste might influence the ability to derive benefits from such energy projects.

From a technological point of view, it takes less time and effort to construct and integrate small hydropower schemes into local environments, compared to large-scale hydropower plants (Egre & Milewski 2002). However, its sustainability depends on its implementation. As argued in IPCC (2011) the extent to which run-of-river hydropower projects have adverse effects is highly site-specific (e.g. geographical, seasonal and demographical differences) and also depends on the resources invested into mitigating these impacts. In some cases, such as Turkey, run-off-river hydropower projects are seen as threats to key conservation areas due to ineffective environmental assessments and monitoring and poor project design and implementation. Studies document the environmental damages that some run-off-river hydropower facilities caused by diverting water for several kilometers without releasing sufficient flow to the riverbed. If implemented and designed poorly, run-off-river facilities might lock connections between downstream and upstream parts of rivers and affect river ecosystems, impede fish migrations and ultimately lead to the disintegration of livelihoods for local communities living along rivers (IPCC 2011; Islar 2012; Aksungur et al. 2011; Şekercioğlu et al. 2011).

6.5.5 Obstacles related to inclusive governance for energy, mining and infrastructure

Infrastructure and IPLCs

Geographic isolation and weak legal rights make IPLCs particularly vulnerable to land, sea and coastal grabbing in relation to infrastructure development, including roads, dams and oil pipelines (Brasselle et al. 2002; White & Martin 2002; Hayes 2007; Reyes-García et al. 2012; Bebbington 2013; Stevens et al. 2014; Vergara-Asenjo & Potvin 2014; Acuña 2015; Gasalla & Gandini 2016; Martínez-Alier et al. 2016; Altamirano-Jiménez 2017; Bavinck et al. 2017; Hill 2017). There is well-established evidence of a global increase in the number of socio-environmental conflicts in relation to infrastructure development in lands inhabited by IPLCs (Moore 2000; Finer et al. 2008, 2015; Filho 2009; Kumpula et al. 2011; RAISG 2016; Wilson & Stammler 2016). Such large-scale infrastructures are often planned and implemented without any Free Prior and Informed Consent of IPLCs (Hope 2016; Dunlap 2017; MacInnes et al. 2017; Fernández-Llamazares et al. 2018; Spice 2018).

As a result, IPLCs increasingly find themselves on the frontlines of social-environmental conflicts and pervasive violence (Bebbington & Bebbington 2011; Temper & Martínez-Alier 2013; Hope 2016). From the 501 land and environmental defenders that have been assassinated worldwide from 2014 to 2016, almost 40% were members of indigenous communities, with many more facing threats, attacks and harassment (Global Witness 2015, 2016, 2017). Similarly, at least 40% of all the 2,242 environmental justice conflicts documented globally directly involve IPLCs (EJAtlas 2017). Disputes over the ownership, control or use of land and sea are an underlying factor in all these conflicts (Fabricant & Postero 2015; Temper et al. 2015; Oxfam et al. 2016; Dell'Angelo et al. 2017a, 2017b; RRI 2017). Research in Latin America also shows how the rights of indigenous peoples in voluntary isolation and initial contact are under assault from the expansion of infrastructures related to timber, gas and oil exploitation (Napolitano 2007; Finer et al. 2008; Martin 2008; IACHR 2013; Pringle 2014; Walker & Hamilton 2014; Kesler & Walker 2015).

Social License to Operate

Today, the Social License to Operate (SLO) is increasingly used by the extractive industry to publicly promote itself as a sustainable and socially responsible actor (Boiral et al. 2015; Lynch-Wood & Williamson 2007; Nelsen & Scoble 2006). Current SLO practices articulate and enact specific forms of legitimacy that produce specific social-environmental outcomes and should therefore be scrutinized (Boiral & Heras-Saizarbitoria 2017; Meesters & Behagel 2017; Smits et al. 2017). Critical studies however identify three structural issues related to the engagement of extractive companies with communities. They are: (1) compensation versus prevention of social and environmental harm, where an industry-initiated SLO will privilege compensation of harm over prevention; (2) community 'out-reach' versus company 'in-reach' activities, where the SLO tends towards adapting the livelihood strategies of communities rather than adapting in-company activities and procedures; and 3) core (extractive) versus periphery (community) action, where the SLO tends to privilege extractive activities over other types of resource use (Klein 2012; Parsons et al. 2014; Kemp & Owen 2013; Boiral & Heras-Saizarbitoria, 2017; Meesters & Behagel 2017; Idemudia 2014). Such privileging of extractive industries over communities is not accidental: according to Milliken (1999, p. 229) these binary relations "establish a relation of power such that one element in the binary is privileged".

Formally, an SLO is considered additional to legal licensing processes. In practice, legal licences often lack standards for community engagement (Nielsen 2013) and the SLO is central in defining what levels and kinds of social and environmental harm are acceptable, what actions for compensation, presentation or restoration are appropriate, and how responsibilities for these actions are distributed between industry and community. Thus, the SLO is capable of transforming legitimacy structures (Parsons et al. 2014) to the extent that it may make industries avoid potential negative legal or social consequences (White 2011). Paradoxically, a SLO can in fact be in competition with a legal license to operate (Holcomb 2008) and pose additional harm to local socio-ecological communities by placing responsibility and accountability on these communities (Boiral & Haras-Saizarbitoria 2017; Boutilier 2014; Owen & Kemp 2013).

6.5.6 Trends and challenges of infrastructure development

Road infrastructure

There is no doubt that we are also witnessing the most explosive era of infrastructure expansion in human history. Due to an unprecedented explosion of infrastructure development, extensive areas of the planet are being opened to new environmental pressures (Finer et al. 2008, 2015; van Dijck 2008; Johansson et al. 2016; Gallice et al. 2017; Kleinscroth & Healey 2017), particularly in regions that sustain exceptional levels of biodiversity and crucial NCP (Fearnside & Graça 2006; Barber et al. 2014; Mahmoud et al. 2017; Sloan et al. 2017). This is occurring both because of massive infrastructure-expansion schemes — such as China's One Belt One Road initiative (Laurance & Burgues 2017; Lechner et al. 2018) and the IIRSA program in South America (Laurance et al. 2001; Killeen 2007) — as well as widespread illegal or unplanned road building (Laurance et al. 2009; Barber et al. 2014; Fernández-Llamazares et al. 2018). Indeed, massive new "development corridors", including roads, highways, hydroelectric dams and oil and gas pipelines, are planned to crisscross much of Africa (Laporte et al. 2007; Laurance et al. 2015; Sloan et al. 2017), Asia (Debin & Yahua 2015; Balmford et al. 2016; Yu 2016) and Latin America (Killeen 2007; Perz et al. 2008, 2010; Barber et al. 2014), with high environmental and social costs (Alamgir et al. 2017; Laurance and Burgues 2017). The net effect of this infrastructure expansion can be catastrophic for biodiversity. In Amazonia, 95% of all deforestation occurs within 5.5 kilometers of a road, and for every kilometer of legal road there are nearly three kilometers of illegal roads (Barber et al. 2014). Similarly, it has been estimated that, as currently planned, the development of new roads in Africa will bisect or degrade areas (Sloan et al. 2017). New roads have allowed ivory poachers to invade the greater Congo Basin in recent years, slaughtering two-thirds of all forest elephants (Maisels et al. 2013). For example, the Global Roadmap strategy has been used in parts of Southeast Asia (Sloan et al. 2018), Indochina (Balmford et al. 2016), and sub-Saharan Africa (Laurance et al. 2015) to devise land-use zoning that can help optimize the many risks and rewards of planned infrastructure projects. The dramatic expansion of roads is determining the pace and patterns of habitat disruption and its impacts on biodiversity (Laurance et al. 2009; Laurance & Burgues 2017).

Infrastructure development related to renewable energy (including hydropower)

Renewables are already the fastest-growing energy sources around the world, mainly due to factors like technological improvement, government incentives and policies, civil society pressure and private sector engagement (Solangi et al. 2011; Timilsina et al. 2012; EIA 2017). The resources allocated in renewable power plants represent two-thirds of the global investment in energy infrastructure. Since 2010, costs of new solar photovoltaics (PV) have come down by 70%, wind by 25% and battery costs by 40%, and they have become the least-cost source of new generation for many countries (WEO 2017). Scenarios up to 2040 point that generation from non-hydropower renewables is expected to grow an average of 4.9% by year (EIA 2017). Wind and solar energy sources will increase the most over this period, reaching 2.5 and 1.4 trillion kilowatt/hours, respectively. Rapid deployment of solar PV, led by China and India, will help solar become the largest energy source of low-carbon capacity by 2040, by which time the share of all renewables in total power generation is expected to reach 40%. In the European Union, renewables will account for 80% of new capacity and wind power will become the leading source of electricity soon after 2030 (WEO 2017). The growth in renewables is not limited to the power sector. The direct use of renewables to provide heat and mobility is also increasing, although from a low base. In Brazil, the share of direct and indirect renewable use in final energy consumption will rise from 39% today to 45% in 2040, compared with a global progression from 9% to 16% over the same period (WEO 2017).

Rivers are frequently dammed to regulate the waterflow in order to improve navigation, secure water supply and produce hydropower (Nilsson et al. 2005). Although many dams have been removed in US and Europe over the last years (O'Conner et al. 2015), the trend is different in other regions. Biodiversity and fisheries are critically threatened by numerous new dam constructions in Asia, Africa and Latin America, particularly due to increased hydropower production (Winemiller et al.

2016). Habitat fragmentation due to impoundment and water regulation is one major threat to aquatic biodiversity, but terrestrial ecosystems are also negatively affected by dams, in particular when reservoir fillings cover large areas (Wu et al. 2004). Depending on size and location, dams may have severe negative impacts on human livelihoods, for example by removing food sources and income from fisheries (Doria et al. 2017), or by causing resettlement of whole communities (Beck et al. 2012). The upstream damming of rivers for hydropower can lead to conflicts between countries as in the case of the Mekong (Pearse-Smith 2012) or the Nile (Cascão 2009) where plans to generate energy might affect the livelihoods of thousands if not millions of people in other countries.

The main drivers for dam removal are safety and economics (Magilligan et al. 2017). All dams have limited life spans, and many dams built in the 20th century have reached their end of life (Beatty et al. 2017). A common cause of dam aging is sediment filling, which gradually reduces the purpose of the reservoir and increases the risk for dam collapse and severe downstream damage (Lejon et al. 2009). Climate change may cause more dam removals, either because dams are not designed to handle increased water discharge in wet areas (Lejon et al. 2009), or because they become less cost efficient due to reduced flow in dry regions (Beatty et al. 2017). While the costs of repairing dams often are higher than the costs of removing them, funding is nevertheless a major obstacle to dam removal (Lejon et al. 2009). Additionally, neighboring landowners and people living close to dams frequently oppose removal, for example due to the cultural-historical values of the dam and associated industry, or because they use the reservoir for recreational purposes (Lejon et al. 2009; Magilligan et al. 2017).

Resource extraction

Mining activities are geologically determined and occur in a variety of ecosystems. The extent to which mining activity globally overlaps with areas of environmental sensitivity has been studied in some detail in recent years. Duran and Gaston (2013) have done a study using data from the Raw Materials Group of base metals and found that 6.7% of worldwide recorded mines were located within protected area boundaries. An additional 27% lie within 10 km of a protected area boundary. Such analysis, however, cannot account for artisanal and small-scale mining, which can also have serious biodiversity impacts in terms of deforestation and chemical pollution. Future changes in mineral supply and demand will likely increase the threats towards biodiverse regions and thus magnify conservation requirements. However, the direction and magnitude of these shifts are highly uncertain. An increase in mineral demand is being driven by population and economic growth trajectories of rapidly industrializing countries where infrastructure investment and manufacturing are key drivers of growth. China has embarked on a very deliberate strategy driven by state-owned enterprises for minerals security through strategic investments and development bargains in Africa and South America. The United States has relied largely on private-sector investment to source the mineral needs of its industries and military, and Japan has followed a model of minority holding investments in major mineral deposits worldwide, also facilitated by organizations such as the Japan Oil Gas and Metals Corporation (JOGMEC). Thus, biodiversity impact evaluation and management through the demand driven route of mineral governance is likely to be very fragmented.

Much of the international regulatory framework around mining comprises a series of international standards and certification systems, which are meant to link existing national laws on environmental planning for resource extraction sites. Ecological impact of mining is largely regulated at the national level and there is vast variation in the extent of coverage that is offered by national regulations. A detailed comparative evaluation of mining regulations with reference to biodiversity protection stringency has not been carried out so far. The effectiveness of mining regulations per se has been poorly studied in terms of environmental performance, since most of the biodiversity mitigation issues are captured by broader environmental impact assessment regulations (Wood 2014).

Voluntary mechanisms for improving performance of the mining industry have managed to raise awareness. Such mechanisms provide civil society important benchmarking indicators to hold industry accountable, and also create an area of active research by scholars of corporate governance (Mori et al. 2016). For instance, the International Council on Metals and Mining (ICMM) – an industry group currently comprising 23 of the largest and most influential mining companies in the world as well as 34 national industry and commodity associations (thus covering a large part of the global mining sector) – has worked with the International Union for Conservation of Nature (IUCN) and other partners to develop guidelines for the industry. In 2003, ICMM made a commitment through its position statement on biodiversity and protected areas that all its member companies will "not mine or explore in World Heritage properties" (a designation undertaken under UNESCO's World Heritage Convention). Furthermore, the ICMM has also established a Mitigation Hierarchy for biodiversity impacts in partnership with the Equator Principles and the International Petroleum Industry Environmental Conservation Association (IPECA), listing the four main stages of mitigation as: 1) avoid, 2) minimize, 3) restore, and 4) offset (ICMM 2017). Several old mining companies have additionally signed on to the International Cyanide Code, which is a certification system to ensure safe containment of cyanide, which is widely used in gold leaching operations. The motivation for this code has been concerns of ecosystem impacts from cyanide spills, although its full impact on wildlife conservation is still contested (Donato et al. 2017).

The implementation of these various voluntary standards is monitored by civil society groups, and there are procedures whereby grievances can be brought to the attention of the membership organizations. There are also ombudsman offices at national and multinational levels, which can provide further recourse such as the World Bank Group's Compliance Advisor and Ombudsman's office. Despite such assurance mechanisms, some countries such as El Salvador have decided to not allow mining at all on their lands for environmental concerns raised by communities (Broad & Fischer-Mackey 2016). However, such moratoriums can easily be changed by legislative action and hence changes in national government policies need to be monitored for most current information on such matters.

The regulation of mining activity with reference to biodiversity and ecosystem services requires a broader international governance mechanism, which is also a need identified in general by geoscientists to plan for future mineral scarcity (Ali et al. 2017). This is particularly the case where mines are near political borders and where pollution impacts can transcend borders due to air and watershed mobility. Existing international environmental treaties could have more specific programs of work or even protocols, which focus on mining activity. Thus far, only the World Heritage Convention (1972) has attracted the attention of major mining companies as an impact mitigation planning mechanism. Recognizing the limits of coexistence of mining and protected areas in some contexts, and willing to engage on mitigative measures of impact to allow for coexistence where possible could be a realistic and pragmatic way towards more sustainable mineral resource extraction and use.

6.6 Transformations towards Sustainable Economies

6.6.1 Addressing overconsumption

***Reducing consumption*:** Consumption can be thought of at three scales: one is intermediate consumption (main actor: business), and two constitute final consumption, namely state consumption (main actor: public authorities on all levels) and private consumption (main actor: households). Interest in sustainable consumption programming and policy has been growing significantly since the 1980s, due to a variety of factors including increased awareness of ecological limits, demands from developing nations to take seriously the links between affluence and biodiversity loss, and the growing influence of MBIs (Cohen 2005; Isenhour 2014).

Behavioural change includes rationally reflected decisions, dependent on attitudes towards the behaviour, negotiations between household members, subjective norms and perceived behavioural control (Ajzen & Fishbein 1980; Fishbein & Cappella, 2006). Perceived economic affordability is influenced by factors such as household income, international market trends, prices, fashion and demand (Peterson 2014). Social affordability refers to restrictions by formal and informal institutions such as laws and social norms and experiences (Jackson 2005; Welsch & Kühlin, 2009) and subjective affordability to to the expected impact on identity and self-esteem (Veblen, 1899; Spangenberg & Lorek, 2018). Most behaviour, however, is not based on decisions; humans tend to react habitually, and resort to shortcuts instead of engaging in lengthy reflections (Shove & Pantzar 2005; Røpke 1999). The social practices shaping most behaviour are constituted by materials (e.g.infrastructure, tools, hardware), practical knowledge (shared understandings of good performance, skills required to perform) and meaning, including emotion and motivational knowledge (Shove et al. 2012).

Which policies to change consumption are preferred in a situation depends not least on what is assumed to determine consumption behaviour: if humans are considered rational utility maximisers, information with an emphasis on cost and benefits will be sufficient for behavioural adaptation as described by the theory of planned behaviour (Ajzen & Fishbein 1980; Fishbein & Cappella 2006). If humans are not economically rational but characterised by bounded rationality, the choice architecture approach including nudging is the most promising (Gsottbauer & van den Bergh 2011; Keller et al. 2016). However, as far as humans do not decide as individuals but as social beings embedded in routines, habits and practices they cannot change individually, social practice theory is useful to describe how their behaviour changes and can be changed – or not (Shove 2010).

The literature on whether providing consumers with information can alter their behavior towards more environmentally friendly purchases is mixed, particularly for the average consumer who may not share strong environmental norms (Stern 2000; Spaargaren et al. 2013). Other studies indicate that changing the composition of consumption has limited effects on the overall environmental impact (Røpke 2001) and that reducing the level of resource consumption is what can reduce drivers of environmental damage (Lorek 2010; di Giulio & Fuchs 2014; Lorek & Spangenberg 2014). However, in order to effectively target individual and household consumer behavior, efforts must consider the diversity of consumption motivations (Røpke 1999) including catching-up, conformist, positional (Veblen 1899), and defensive consumption (Beckenbach et al. 2012). Working conditions can stimulate compensatory consumption (Scherhorn 1997) and no-choice consumption enforces high resource use.

Increasingly, recognizing this inability to rely on information provision alone to change consumer action, there is considerable literature coming out of sociology and psychology addressing why people consume (Røpke 1999) and how nudges or tools can be used to reduce unsustainable consumption (Halker 2013; Olander & Thorgersen 2014). The 'nudging' or 'choice architecture' approach builds upon cognitive science and behavioral economics, considering individuals as oriented towards finding 'satisficing' rather than optimal results, reacting habitually in most situations, and resorting to shortcuts instead of engaging in lengthy reflections (Gsottbauer & van den Bergh 2011). Strategies are derived from experimental research showing that pre-set default options can in some cases have a strong influence on consumers' propensity to make desirable choices, such as choosing green electricity (Keller et al. 2016). Nudges can include tailored messaging or offer peer comparisons, provide disclosures or warnings, create default rules, or use social norms to affect consumers' 'choice architecture' in order to affect consumption (Sunstein 2015; Lehner et al. 2015). For example, presenting meat-free meals as a "default" option (one must opt-out in order to get meat) in university cafeterias increases their consumption (Campbell-Arvai et al. 2014). Simply including information about neighbors' energy use in electric bills can motivate some, but not all, households to reduce their own energy use (Costa & Kahn 2013), as can creating a default option for renewable energy (Momsen & Stoek 2015). The rise of "networked home" options (lightbulbs, thermostats) to some degree have evolved to make it easier for consumers to adopt pro-environment behavior, such as reduced energy consumption by ensuring lights are off in an empty room (Stern 2011). However, the emphasis on nudging consumer choice, rather than restriction by legislative or other means, has been seen by some critics as both paternalistic and potentially unethical (Shubert 2017) and a defense of power and privilege rather than an earnest attempt to limit through collective means irresponsible and damaging consumption and production (Lehner et al. 2015).

As both information and nudging address individuals and their (mostly isolated) behaviours as the basic units to be targeted with interventions, seeking to effect social change through inducing individuals to make 'better choices', they fall short of addressing the social and institutional context shaping consumption behaviour. In order to effectively target individual and household consumer behavior, decision makers must take into account the diversity of consumption motivations. *Catching-up consumption* refers to the unmet needs in particular of low income groups; *conformist consumption* responds to the desire to match the status of the social reference group, not being identifiable as an outsider or otherwise discriminated for the absence of certain goods which signal group membership (Beckenbach et al. 2012). *Positional consumption* refers to the same peer groups as conformist consumption, but with the desire not only to conform to common cultural standards, but to be superior (Veblen 1899). In both latter cases, the goods can be owned, rented, borrowed or stolen – visibility is more important than ownership details (Lorek & Spangenberg 2003). *Defensive consumption* is the result of efforts to compensate for the deterioration of the prevailing living conditions (Beckenbach et al. 2012). The mortgage-based consumption binge in the USA following long-term income stagnation is probably the most prominent example. Socially bad working conditions (lack of self-determination, permanent control, interference of superiors, lack of recognition) are empirically linked to compensatory consumption (Scherhorn 1997). Scherhorn (1991) adds another, cross-cutting category, *addictive buying*, which can be catching-up, positional or compensatory. It is characterised by the fact that the consumer has limited rational control over the buying decision (like any addiction, severe debt can be the result). Finally, in certain places, in particular in the USA, *no-choice consumption* plays an important role: for the satisfaction of certain needs only high resource consuming options exist (e.g., purchases of multi-bedroom houses in areas offering good schools).

Each of these situations constitutes a specific social context, involving different formal and informal institutions, social relations, prevailing habits, routines, status symbols, dynamics, values and options, which together constitute social practices as a collective pattern. Thus, changing behaviour, including consumption behaviour, can only be successful if emphasising endogenous and emergent dynamics of social groups, which may be stimulated but cannot be directed or hands-on managed (Shove 2010). Consumption change must then co-evolve with the societal structures and conditions, as an important element of transformative change of the consumer societies. One element of the transformation is overcoming the idea that wealth or GQL are equivalent to more consumption opportunities (Fisher 1906). Human needs can be satisfied with less resource consumption (Steinberger & Roberts 2010) if suitable satisfiers are chosen (Max-Neef et al. 1989). GQL has been shown not to increase above a certain income threshold (Max-Neef 1995) and to be decoupled from income and thus consumption thereafter (Layard 2005) (although the rich seem to be happier than the poor in most societies (Veenhoven 2010). However, this is not easy in the current consumer society (Speck & Hasselkuss 2015): dedicated policies are required to make a resource-light, good life easier (Schneidewind & Zahrnt 2014; Heindl& Kanschik 2016).

***Taxing consumption*:** Environmental policy has a long history of using environmental taxes for pollution management and enhancing resource use efficiency, including in road transportation (road pricing schemes, area licensing schemes, mileage taxes, toll roads and congestion fees), taxes for waste management and recycling (taxes on beverage containers or plastic bags for consumers to discourage use or encourage recycling; charges/taxes for the dumping of solid waste, untreated effluents and for hazardous waste promote resource use efficiency and conservation). However, while these targeted fees and taxes, and VAT more generally, dampen consumption, very few direct consumption or other taxes or levies (sometimes called green taxes) have been designed specifically in order to preserve biodiversity: other tools have been preferred, such as PES or agri-environment schemes, which can be implemented through a variety of instruments such as direct subsidies, tax incentives or tax breaks, cap and trade markets, or auctions and certification programs (Sterner 2003; Cubbage et al. 2007; Pirard 2012; Bryan 2013; Braat & De Groot 2012).

Many taxes on activities or products exerting negative (and often indirect) effects on ecosystems and biodiversity, such as through atmospheric pollution or water pollution, rely either on the polluter-pay principle or on the user-pay principle, depending on the payer effect on the degradation of quality or of the use or depletion of natural resources Ekins (1999). Examples of these include:

- pesticide taxes: In France, despite the increases in the tax rate, the tax on diffuse pollution has not been very effective at reducing pesticide use in agriculture (Sainteny, 2011). French studies and official reports mainly attributed this ineffectiveness to the relatively low tax rate and the weak price elasticity of demand for pesticides. Jacquet et al. (2011) estimated that to reduce pesticide use by 30%, the tax rate would need to be 100% (of the sales price of the product), while for a reduction of use of 50%, the tax rate would need to be 180%. Moderate increases in the tax rate alone appear not to be sufficient to change behaviour.
- diffuse pollution taxes, including water pollution charges and taxes (OECD, 2017b; Hogg et al. (2014) and water extraction charges: McDonald et al. (2012) emphasize the positive biodiversity impacts gained when limiting water extraction by the energy sector.
- air pollution taxes in many European states, like France, Sweden, Norway (Ekins 1999; Hogg et al. 2014): NO_x and SO_2 cause many environmental problems detrimental to ecosystems, including acidification and eutrophication of inland waters, thus pollution taxes have been shown to reduce emissions.
- waste taxes in most OECD countries (Ekins 1999; Hogg et al. 2014); e.g. a packaging tax (in Denmark but abolished recently)

Hogg et al. (2014) provide the most extensive review of the existing environmental taxes in European countries, whereas Kenny et al. (2011) review Canadian programs, of which nearly half use taxes and charges, including water permit fees in national parks, water use permit fees, charges on water discharge, charges on fishing or hunting licences, a Riparian Tax Credit in Manitoba to prevent soil erosion and improve water quality, and an Alberta charge for overcutting. However, no real assessment of the effectiveness of these kind of taxes is found in the literature, while existing studies focus more on other tools (Kemkes et al. 2010; Miteva et al. 2012).

***Consumer and community campaigns against overconsumption or damaging products*:** While consumer-based environmental movements are certainly nothing new, the recent focus on consumer responsibility is notable, spurring a wide range of initiatives undertaken across scales and by both private and public sectors. For example, grassroots and civil society environmental organizations have advocated a wide range of lifestyle modifications and shifts in consumer behaviors, and their work is commonly united by a focus on education initiatives designed to help affluent and environmentally conscious consumers connect their lifestyles to environmental impacts around the world. Often drawing on affective appeals to protect endangered species or biodiversity hot spots, these campaigns urge consumers to make educated consumption choices (including buying less) and to boycott particularly damaging products and brands. Here the new opportunities for protecting biodiversity and NCP offered by the telecoupled information flows can work out forcefully. NGOs and consumers using them can generate strong pressures on national and multinational corporations and on governments by mobilizing the social norms of affluent consumers. Highly visible international boycotts against large multinational corporations are examples of such informational telecoupling and have had some historical success (Conroy 2001; Clouder & Harrison 2006; Munro & Schurman 2008). Greenpeace's 2010 social media campaign and boycott of Nestlé's KitKat brand, for example, highlighted tropical deforestation associated with palm oil production. After more than 1.5 million people viewed the campaign's online ad and 200,000 emails were sent to Nestlé headquarters, the company agreed to a plan for more sustainable palm oil sourcing. While consumer-based activism can provide an important challenge to corporate control of supply chains by leveraging buying power to demand more sustainable alternatives (Conroy 2001), critics point out that these successes are often short lived, have done relatively little to challenge dominant consumption logics or practices and fail to recognize that not all consumers have the purchasing power or access to market alternatives (which are often more expensive).

While there is significant potential for demand-side measures to contribute to biodiversity and natural resource conservation, there is very little evidence to suggest that voluntary, market-based programs have resulted in direct or indirect improvements in, for example, deforestation prevention or biodiversity conservation efforts (Newton et al. 2013). This is partially because while some products are less environmentally impactful, consumers are buying more of them each year, essentially offsetting gains associated with efficiency and conservation efforts. Pointing out the irony of attempting to solve problems of over-consumption with even more consumption, Johnston argues that the "conservation through consumption" approach "maximizes commodity choice, while minimizing the citizen's ecological responsibilities to restrain consumption" (2008:259) – the level of consumption is more environmentally relevant than the pattern. Ecologically-sensitive niche markets still constitute a small portion of total consumer activity. If they were to drive significant changes in supply chains, these markets would need to constitute a greater percentage of demand.

There are also consumer-led and community-led initiatives that focus not on individual consumer behavior, but on aggregate communities, such as towns, to reduce consumption. Eco-localism is one such movement, which rejects large-scale economic organization (noting that it leads to unsustainability, lack of place, and vulnerability to energy shocks) in favor of local place-based living centered on local economies. The Transition Towns movement is one such type of eco-localism, with more than 500 sites across Europe, the US and Australia and New Zealand, and which advocates transitions from high energy use and globalized economic exchange to local resilience (Ganesh and Zoller 2015). It revitalizes the efforts undertaken 20 years earlier by the Local Agenda 21 movement, and again 20 years earlier by environmental "citizens' initiatives" (Garcia-Sanchez, Prado-Lorenz 2008; Bossel 1978). Local and regional governments across the world are also investing in a wide range of programs to encourage more sustainable consumption including elements of sufficiency such as hosting repair cafes, materials exchanges or swaps, and innovating 'collaborative consumption' events like tool lending libraries. Alternative local currency movements, and the advent of new 'crypto-currencies' like Bitcoin, are another approach to removing consumers from formal economies (North 2010), but little is known yet about their environmental impacts (though it is believed that using Bitcoin creates large energy demands in particular due to server power). The sharing economy is also potentially a component of reduced consumption, but more needs to be understood about the biodiversity and NCP impacts (Schor & Attwood-Charles 2017).

***Corporate initiatives to reduce consumption*:** Corporations and industry associations have also responded to concerns about biodiversity and consumer demand for sustainable products and services, sometimes in substantial ways. Some have voluntarily implemented more sustainable sourcing practices and consumer awareness campaigns in the interest of both resource protection and building brand loyalty. Patagonia, for example, has implemented a Worn Wear program, which allows consumers to turn in used gear for reuse and repair. They also offer recycling services for clothing that cannot be repaired, helping to offset demand for new production (and the associated resource extraction). The company claims that since 2005 they have recycled more than 95,000 tons of clothing. Advocates argue that corporate social responsibility programs like Patagonia's can result in a triple win—for sustainable business development, biodiversity conservation and building brand loyalty among ecologically concerned market segments (see further section on CSR below). There tends to be very little academic research on such programs, however. Williamson et al. (2006) found that such voluntary approaches will not alter the behavior of manufacturing SMEs significantly unless they have a positive effect on the bottom line, e.g. by reducing resource or labor cost. The sustainability-related certification programs designed or supported by industry associations to help consumers make more sustainable choices are sometimes disputed by NGOs and development groups (see also certification discussion in section 6.3.2 of the Global Assessment, and Carlson et al. 2018).

Such critics of corporate and industry sponsored initiatives dismiss them as little more than greenwashing, designed to commodify consumer concern and capture market share in new and highly profitable ecologically-sensitive niche markets, rather than any sincere interest in biodiversity conservation. Corporations often must demonstrate movement toward environmental and social responsibility to avoid regulation (Marsden & Flynn 2000) and ensure employee morale (Jacobsen & Dulsrud 2007). But critics draw on empirical evidence to suggest that voluntary pledges, made in response to market pressure and in an effort to protect the standing of their global brands, are less likely to contribute to long-term conservation gains than programs inspired by a sense of a "long-term imperative responsibility" (Newton et al. 2013:7; Andersen & Skjoett-Larsen 2009; Kissinger 2012). Today, however, telecoupled information flows can be used to support and monitor efforts at greening the supply chains, both by management and other stakeholders such as employees, trade unions, consumer, environment and development organizations, and investigative media.

***Governmental policies to reduce consumption*:** As discussed above, the most immediate field of influence is, of course, the government's own consumption, on all levels of administration and specifically for the tasks conducted. Given the urgency of biodiversity loss, a growing number of scholars have advocated moving beyond voluntary, market-based measures and into collective and socially coordinated strategies to reduce total consumption levels. While often considered politically unpalatable and a violation of consumer sovereignty, governmental environmental policies have long included bans on products that are toxic to humans and ecosystems, and those accumulating in the environment. Extending these policies from quality to quantity, instruments to reduce consumption are already in use, and are becoming increasingly popular in a wide variety of contexts. They range from broad ecological tax reforms, to bans of single-use disposable products, disincentives for travel or meat consumption, and public investments in product service agreements or collaborative consumption networks. State and local governments around the world have invested in circular economy strategies, designed to keep existing materials in circulation longer and essentially displacing demand for virgin resource extraction (see circular economy section below). The UK Government's Technology Strategy Board, for example has instituted an innovative design challenge intended to make supply chain interventions that can lead to more sustainable consumption behaviors. Other governments have gone further to more directly address consumer behaviors. Sweden, for example, has implemented a tax credit for households that choose to repair rather than replace goods. Germany (which adopted its circular economy law 30 years ago) and many US states and Canadian provinces have product stewardship laws that require corporations to take end of lifecycle responsibility for the products they produce. This financial responsibility has incentivized businesses to design their products for durability, repair and easy recycling (Van Rossem et al. 2006). Other locales are taking more significant measures to limit environmentally damaging consumption behaviors, including bans of particularly damaging disposable products like plastic bags, water bottles and children's clothing with dangerous toxins. A year-long evidence-based inquiry into changing consumption behaviors in the UK concluded that non-regulatory measures used in isolation, including nudges, are less likely to be effective (Hobson 2013), suggesting a need to complement and support these efforts with policies that can broaden participation, such as regulations (Allen 2010). While it is often assumed that all consumers are opposed to hard policies that limit choice, many consumers, in different contexts from Stockholm to Philadelphia, have suggested that they favor the restriction of choice via the removal of dangerous products from the market and a stronger role for governmental agencies in protecting consumers (Isenhour 2010). In some contexts, US consumers actually preferred soft regulations over voluntary programs (Attari et al. 2009), and UK citizens were even supportive of rationing as an allocation mechanism offering fair access to goods and services (Alcott 2010).

***Reducing state consumption per service delivered*:** In most countries, the aggregate of public procurement is the single largest purchaser of goods and services. This gives public authorities the opportunity to strengthen sustainable suppliers and nudge others towards renewing their offers accordingly. For instance, the German Mail, when not finding a suitable offer for an electric delivery car, joined forces with a start-up company which is now supplying those cars, designed for the user's purposes and at competitive cost (Nießen & Burian 2015). Green public procurement can stimulate the local demand for recycled products or organic food, reducing resource and chemicals consumption. So, from canteen food via recycled paper to energy saving buildings, public authorities on different levels, from local to international, have a plethora of options to reduce their consumption of materials, energy and land and thus to reduce several direct and indirect drivers of loss of biodiversity and NCP (Brammer & Walker 2011; Lutz 2009). Beyond this, authorities have also indirect influences on consumption patterns and levels: city planning influences access to urban biodiversity nearby reducing mobility demand, public transport planning can enhance the accessibility of NCP and the GQL without car ownership, schools and other education institutions influence environmental awareness and attitudes.

6.6.2 Reducing unsustainable economic production

***Creating a circular economy*:** The literature on circular economy (CE) is well established and rapidly growing since the first references to the term in the 1980s (Murray et al. 2017), with a large proportion of peer reviewed papers analyzing the Chinese and European experiences (Andersen, 2007; Ghisellini et al. 2016; Jesus & Mendonça 2017). However, there are still competing definitions and blurriness about the concept (Kirchherr et al. 2017) and how far it can be implemented at the micro (e.g., company, consumer), meso (e.g. industrial park) and macro (regional, national, global) level. According to the most frequently cited definition, CE is "an industrial system that is restorative or regenerative by intention and design. It replaces the 'end-of-life' concept with restoration, shifts towards the use of renewable energy, eliminates the use of toxic chemicals, which impair reuse, and aims for the elimination of waste through the superior design of materials, products, systems, and within this, business models." (Ellen MacArthur Foundation 2013: p.7). The major aim of CE is to decouple economic growth and the deterioration of the environment (Ghisellini et al. 2016), suggesting that economic prosperity and improved environmental quality can be achieved together and at the same time (Kirchherr et al. 2017) through technological, economic, and social innovations (Jesus & Mendonça 2017). Definitions of CE regularly refer to the 3R or 4R or other extended Rs models (Kirchherr et al. 2017), listing most frequently reduce, reuse, recycle and recover (and in extended models repurpose, remanufacture, refurbish, repair, rethink and refuse etc. (c.f. Potting et al. 2017; Reike et al. 2017) as the key functionalities within a CE. These functionalities enable the CE not to use additional natural resources to produce materials and to avoid discarding products as waste (Potting et al. 2017). Therefore, CE can be considered as a business-oriented option that helps protect nature and its benefits to people and good quality of life in various ways, e.g. by saving energy (Cooper et al. 2017), decreasing landfilling (Reike et al. 2017), reducing production costs (Mativenga et al. 2017; Mativenga et al. 2017) or lowering the demand for biomass (Haas et al. 2015).

CE is promoted by policy frameworks in various countries worldwide. In China, CE is implemented as a top-down national policy (the Circular Economy Promotion Law of the People's Republic of China has come into force in 2009), aiming for a green and sustainable growth of the economy (Su et al. 2013; Yuan et al. 2006). In Europe, several of the EU Member States have national level legislation fostering circularity, some of them initiated more than a decade ago (e.g. in Germany there is legislation on circularity since 1996) (Doranova et al. 2016). At the EU level, circularity has been encouraged by regulating both production (directives concerning substance restrictions and product performance) and waste management (Waste Framework Directive, 2008), and most recently by the Circular Economy Package including an action plan to foster a more circular economy in Europe (Hughes 2017). Japan fostered circular economy at the meso level by supporting eco-industrial (urban and industrial symbiosis) parks via the Eco-Town Program between 1997 and 2006 (Ohnishi et al. 2012; Van Berkel et al. 2009). In Australia and New-Zealand professional networks such as Circular Economy Australia and the Sustainable Business Network has been working on a CE action agenda

(Ghisellini et al. 2016). The US lacks federal regulation on CE, although state level and sector specific (esp. focusing on waste management) regulations exist (Ranta et al. 2017). Recycling is emphasised in other countries' waste management policies (e.g., Vietnam and Korea) (Hideto et al. 2011), while there is limited evidence on circular economy policies from Africa, Central and South America, Russia and India. Collections of best practice cases also show that CE is being implemented at micro, meso and macro levels, by governmental bodies as well as by business actors and NGOs (Kalmykova et al. 2017; Potting et al. 2017). Nevertheless, consensus is still lacking on how far the global economy is progressing towards a CE. Cooper et al. (2017) estimated that potential savings of energy used for economic activities worldwide can reach 6-11% (using an input-output model and data from 2007). Haas et al. (2015) carried out a material flows analysis on data from 2005 and estimated that the recycling within the economy as share of processed material reached 6% globally and 13% in the EU. Reasons for these relatively low numbers are thought to be the large proportion of material throughput (Haas et al. 2015), and the accelerating production due to the rebound effect (Zink & Geyer 2017).

Shortcomings, barriers and challenges of CE are extensively discussed in the literature, pinpointing both “soft” (social, regulatory and institutional) and “hard” (technological solutions and financial factors) factors (Jesus & Mendonça 2017; Ranta et al. 2017). Table 6.3 sums up the key challenges identified as well as the options to overcome those.

Table 6.3 Challenges and options of circular economy (CE)

Factors	Challenges/Barriers	Options to overcome	Reference
Soft	The concept overlooks social equity	Sharing economy	Kirchherr et al. 2017
	Limited institutional support for CE principles other than recycling	Applying an extended Rs model	Ranta et al. 2017
	Difficulties of law enforcement at local level (clashing with social norms)		Ranta et al. 2017
	Consumers prefer new products	Changes in the value system	Ranta et al. 2017
Hard	Rational economic decision-making at company level (prices of materials mainly reflect the cost of mining and short-term values but not the costs of depletion or environmental degradation)	Pricing externalities	Andersen 2006
	Fossil energy carriers used as energy sources with limited recycling options	Carbon capture, transition to green energy, cascadic use of fly ash and slag	Haas et al. 2015
	Biomass used for food, feed and fuel with limited recycling options	Closing the loops in agricultural production by sustainable agriculture, changing dietary habits, reducing food waste	Haas et al. 2015
	Limitations to practical implementation (e.g. missing infrastructure to local waste separation)		Jesus & Mendonça 2017

Corporate social responsibility (CSR): Corporate sustainability means that a company considers all of its negative and positive impacts on stakeholders and the natural environment including their interrelations and tries to improve them (Dyllick & Hockerts, 2002; Baumgartner 2014; O’Connor & Spangenberg 2008). The current rate and irreversibility of biodiversity loss is considered to be one of the main environmental challenges facing companies and societies as a whole (Boiral & Heras-Saizarbitoria 2017; SCBD 2010; Cardinale et al. 2012). Besides the fact that many business activities are dependent on biodiversity and ecosystems services, companies can also impact biodiversity and ecosystems services (Lambooy 2011), and companies have the potential to make a substantial contribution to arresting declines in both (Armsworth 2010). This is relevant for companies directly causing impacts but also for companies influencing biodiversity and ecosystems indirectly via its global supply chains.

Empirical results show that companies still use a reactive approach to biodiversity and creating internal commitment for the corporate responsibility on the topic remains challenging (Overbeek et al.

2013). Biodiversity has emerged on the corporate agenda as both a risk and an opportunity that must be managed through more rigorous impact assessments and environmental management systems (Athanas 2005). However, since the inception of the CBD in 1992, little progress has been achieved in terms of involving the business community in protecting biological diversity worldwide. Voluntary corporate activities are helpful, but not sufficient, and critics of this approach argue that campaign successes are often short-lived while proponents emphasize the potential of NGO campaigns to achieve positive environmental and social change in the absence of governmental legislation (Boiral & Heras-Saizarbitoria 2017; Dieterich & Auld 2015). NGOs and civil society can play two particular roles in CSR for biodiversity (Dieterich & Auld 2015): (1) offer technical assistance for companies, share learned experiences and help companies to develop strategies to reduce negative impacts on biodiversity; (2) monitor the business sector to ensure that its commitments are fulfilled by tracking progress indicators.

Life cycle analysis: The framework of LCA (ISO 2006a, b) distinguishes four phases: goal and scope definition establishing the aim and scope; inventory analysis (LCI), compiling and quantifying inputs and outputs for a product; life-cycle impact assessment (LCIA), understanding and evaluating the magnitude and significance of potential environmental impacts from the extraction of raw materials until the final product disposal; and life-cycle interpretation, evaluating the findings to reach conclusions and recommendations. The result of the LCI is typically an extensive table of environmental inputs (natural resources such as fossils, ores, biotic resources, water and land use) and outputs (emissions of chemical compounds to air, water and soil) related to a functional unit. Since the inventory table is too long to handle and contains many items that require expert knowledge, LCIA converts LCI results to common units and aggregates them to potential contributions to a specific impact category ('characterization').

Most LCIA methods today only include compositional attributes of biodiversity (Verones et al. 2017), either indirectly through midpoints for, e.g., climate change, acidification, eutrophication, land use, and ecotoxicity that eventually may affect biodiversity through consecutive impact pathways, or by endpoint indicators related to species loss (Goedkeep et al. 2013), most often as relative losses compared to species richness in a reference situation (deBaan et al 2013). The latter approaches are all based on species-area relationships (Verones et al. 2017; Schenk 2001; Penman et al. 2010; Curran et al. 2011; Koellner et al. 2013; Souza et al. 2015; Winter et al. 2017; Chaundhary et al. 2015) and hence only apply to land-use change (LULUC) impacts (or impacts of water use converted into area loss (Verones et al. 2015). Comprehensive and widely accepted biodiversity approaches have still not yet been developed and debates center around overrepresentation of well-studied species, ways to compare different taxonomic groups (while for example reptiles and amphibians are not included at all) and impacts that constitute a loss in quality instead of area. Biodiversity characterization methods are also often difficult to apply in practice. This is mostly due to the fact that many of these approaches may lack readily available and applicable lists of characterisation factors and/or require specific spatial or temporal information that most LCA studies cannot provide (Winter et al. 2017). The same problem of unavailable characterization factors is faced when including ecosystem services in LCIA methods, a discussion sparked after the publication of the Millennium Ecosystem Assessment reports (Reid 2005). Several authors have discussed options to incorporate ecosystem services into LCIA (Zhang et al 2010 a, b; Bakshi and Small 2011; Koellner and Geyer 2011; Cao et al. 2015; Othoniel et al. 2016; Blanco et al. 2017; Bruel et al. 2016), but debates occur on whether ecosystem services impacts should be complementary or instead of currently adopted impact assessment categories. Although progress has been made, none of the current attempts has yet resulted in sufficiently practical and comprehensive methods.

LCA is widely applied by companies (Frankl & Rubik 2000; Clift & Druckman 2015), to inform consumers (Del Borghi 2013), and for public policy making (Owsianiak et al. 2018). It also constitutes the basis of the so-called carbon footprints (Finkbeiner 2009). The main hurdle for even wider policy implementation is the fact that LCA involves many choices, assumptions and data and that these require standardization (Pré Consultants 2006), although this is often complex and time-consuming (Finkbeiner 2014; Galatola & Pant 2014). Another is the fact that each LCA is a snapshot at a specific point in time which may be outdated by innovation or modified supply chains by the time the data is used; real-time LCA would require a global system (for a blueprint, see http://www.myecocost.eu/) which is nowhere in sight. Last but not least, LCA focuses on reducing the impacts per unit of consumption; it does not deal with reducing consumption levels itself.

6.6.3 Reforming models of economic growth

Universal Basic Income: A universal basic income (UBI) is a specific form of the social protection floor, a periodic cash payment to all, individually, and without means test or work requirement (BIEN n.d.; Van Parijs & Vanderborght 2017). Examples in some countries include conditional cash transfer (CCT) programs of national and subnational governments that provide fixed support to the poor in return for some activity (e.g. maintaining a job, or enrolling children in school). Linking CCTs to biodiversity-conserving activities is an increasingly noted possibility (Rodríguez et al. 2011).

A carbon tax designed to reduce consumption of fossil fuels could be turned into a form of UBI if the revenue could be distributed as a per capita dividend. This partial UBI might be attractive because it can make carbon pricing fair (by countering the regressive effects of a consumption tax) and politically feasible (since a majority of taxpayers will receive a benefit greater than the increased costs associated with the tax) (Boyce 2013). In the longer-term, carbon pricing could be an effective policy tool at the global level, with at least some of the revenue going toward poverty relief in countries that might be hardest hit by rising carbon fuel prices (Segal 2012; Blackburn 2011). One obstacle to implementation of a carbon dividend is competition for other uses of the revenue, including tax shifting, tax relief, green investment, and compensation for workers losing jobs in carbon fuel industries (Pollin 2015; Van Parijs & Vanderborght 2017). In addition, a global carbon dividend will face competition with national uses of carbon pricing revenue. A UBI can also be part of a more ambitious "degrowth" strategy away from fossil fuels (see section 6.4.6 of the Global Assessment). A UBI would address the contraction of employment associated with degrowth by partially decoupling income from employment, and addressing poverty and inequality not by sharing the fruits of growth but rather by sharing a smaller pool of resources more equitably (Andersson 2010; Fitzpatrick 2010; Boulanger 2010; Van Parijs 2009; Van Parijs 2013; Van Parijs & Vanderborght 2017; Gorz 2013).

Natural Capital Accounting tools**:** According to proponents, environmental accounting will enable a better overview of the different contributions of nature to people (Faccioli et al. 2016; Edens & Hein 2013), and allow for better insight into environmental costs and benefits next to financial ones (Boehnert 2016; Faccoli et al. 2016; Obst et al. 2015). However, this economic concept of nature has also been criticized (Sullivan 2013; Turnhout et al. 2013; Bordt 2018) arguing that bringing nature and biodiversity into the economic domain is likely to result in accelerated exploitation rather than conservation.

Natural Capital Accounting (NCA) is used here as an umbrella term that includes different methods and approaches including forms of environmental accounting. In some cases, NCA refers to a system of national indicators of monetary or non-monetary wealth, in other cases as a national or sub-national planning tool. Sometimes, a distinction is made between Natural Capital Assessment and Natural Capital Accounting, where the first refers to measurement and evaluation of natural capital without necessarily expressing it in financial terms or following formal accounting procedures, which are typical of the latter (Natural Capital Coalition 2017). Central to the idea of monetary Natural Capital Accounting NCA is that it offers an approach that "nature's stocks and flows are depicted such that they accord with the format of a standard online current bank account" (Sullivan 2013). Stocks are defined as 'the capacity of ecosystems to supply services to multiple stakeholders' (Hein et al. 2006), and with this definition the degradation of ecosystems can be measured as a decrease of the stock. The flows are the values of the specific services that flow between ecosystems, stakeholders and other capital-holding places (Edens & Hein 2013). According to proponents, NCA will enable a better overview of the different contributions of nature to people (Faccioli et al. 2016; Edens & Hein 2013), and that when these stocks and flows are expressed as capital, they can be compared and exchanged with other goods in the financial market (Boehnert 2016), whereas this was impossible when they were defined as non-use values (Faccoli et al. 2016). This is considered important because in economic market mechanisms, these 'externalities' are not visible in the price of commodities (Obst et al. 2015). For example, an NCA analysis of groundwater withdrawals from the high plains aquifer in Kansas, USA, shows the state lost approximately US$110 million per year of capital value, approximately equal to the state's 2005 budget surplus (Fenichel et al. 2016).

Versions of NCA have been endorsed by the CBD, World Bank and the European Union and taken up in initiatives such as the Wealth Accounting and Valuation of Ecosystem Services (WAVES) program (Agarwala et al. 2014); the Natural Capital Coalition (Natural Capital Coalition 2017) and Mapping and Assessment of Ecosystem Services (MAES) (Maes et al. 2015). A growing number of countries and private companies are establishing environmental accounting programs (Edens and Hein 2013; Kenter et al. 2015; Ruijs & van Egmond 2017), but progress has been slower in developing countries (Recuero Virto et al. 2018). At Rio +20, a Natural Capital Declaration was signed by a coalition of banks, investors and insurers (Boehnert 2016) but there was also considerable criticism from civil society. Later, the concept was endorsed and actively implemented by new initiatives like GLOBE international, which passed the first Natural Capital Protocol at the CBD, and the World Bank, which established the WAVES (wealth accounting and valuation of ecosystem services) program (Agarwala et al. 2014). A growing number of countries are establishing accounting programs for environmental values, and private companies are also paying more attention to this concept (Edens & Hein 2013; Kenter et al. 2015; Ruijs & van Egmond 2017), but NCAs have progressed more slowly in developing countries in particular due to lack of political will and poor data availability (Recuero Virto et al. 2018). While NCA can play an important role in broadening the definition of accounting and in making to organizations and business more accountable for their impact on society and the natural environment (Feger & Mermet 2017), others have been opposed to NCA use of the concept of capital as explicitly economic, monetary and capitalist meaning (Bordt 2018).

There are specific uses of environmental accounting approaches that focus on only one type of capital, for example carbon accounting. Natural capital assessment in physical terms either uses a variety of object-specific units, arguing that different units are incommensurable and cannot be added up (like mass, energy and area) while others focus on specific aspects. Directly measuring (elements of) direct and indirect drivers of biodiversity and NCP loss escapes the criticism directed at monetary accounting. While a number of methods and indivators have been developed, they should be considered as complementary rather than competing approaches (Robèrt 2002).

Industrial ecology analyses the industrial metabolism in terms of physical stocks and flows of matter and energy within and between countries and sectors, with material flow accounting (MFA) or material and energy flow accounting (MEFA) a key method to describe such flows, characterised by the weight of the flows in tons (Bringezu et al. 1997; Haberl et al. 2004). The method as standardised by the European Statistical Office based on work of several European research institutions (e.g. Wuppertal Institute, Germany, SERI, Austria) provides information about the total amount of resources consumed in a country (DMC Domestic Material Consumption), used (TMI Total Material Input), imported and exported (TMI-DMC), and activated (Schmidt-Bleek 2008). The latter, the TMR Total Material Requirement, is of particular importance for nature and NCP as it counts the material flows set in motion throughout the supply chain, characterising one important indirect driver of biodiversity loss (Behrens et al. 2007). Due to data problems, the European Commission uses a simplified indicator called Raw Material Consumption (RMC) in monitoring its Resource Efficiency Strategy (http://ec.europa.eu/archives/commission_2010-2014/potocnik/expert_group/pdf/DGENVResEffIndicsonlyFin.pdf).

Since MFA accounting schemes use Physical Input-Output Tables (PIOT) based on the system of national accounts (SNA) to characterise material flows, they operate on the national economy level where the data are gathered, and they are capable of disaggregating flows into a wide range of substances (Spangenberg et al. 1998) between all business sectors in the system of national accounting, and between trading partners (Dittrich et al. 2012). However, they cannot spatially disaggregate the flows within a national economy, but are closely related to a number of methods calculating the physical requirements of products. In particular, embodied energy or emergy accounting (adding up the total energy used to produce a certain good, Ulgiati et al. 2011), material intensity per service unit (Schmidt-Bleek et al. 1998), energy calculations (Ayres et al. 1996), and different methods of calculating a land use, water and substance footprint are life cycle wide physical assessment methods on the micro level (Steen-Olsen et al. 2012). Other approaches include MuSIASEM, an accounting system developed at the UAB Barcelona, that offers an integrated, multi-level consistent physical accounting system for the industrial metabolism and its exchanges with the respective environment (Giampietro et al. 2014; Lomas & Giampietro 2017), as well as the System of Environmental-Economic Accounting (SEEA), is a framework that integrates economic and environmental data in their respective units to provide a more comprehensive and multipurpose view of the interrelationships between the economy and the environment (https://seea.un.org/).

As an information tool, it is not surprising that most of the NCA literature is of a technical nature. Two issues stand out: data quality and availability and the harmonization of methods and approaches. Data availability and quality are considered major limitations in the use and uptake of NCA (Hein et al. 2015; Mace et al. 2015; Donnelly et al. 2016). Existing indicators are not always fit for purpose, regulating services are strongly underrepresented, and there is a lot of variability in the representation of different habitat types (Faccoli et al., 2016). Access to environmental information is a separate but related challenge, since each service needs a different dataset (MA 2003). Besides that, these datasets need to cover both the flows and the stocks of this specific ecosystem service, which makes even fewer datasets available for this purpose (TEEB 2010). Additionally, there are many different approaches that use different classifications and categories of nature's contributions and assets, different definitions, and different methods for valuation (Day 2013; Faccoli et al. 2016; Bateman et al. 2011). This diversity is considered positive because different approaches are fit for different purposes and because competition between different approaches could speed up their development (Agarwala et al. 2014). Others, however, lament the lack of consistency; the calculated values of different methods can differ greatly and this makes it harder to compare them (Edens and Hein 2013) and impossible to aggregate them into a total value calculation. Several initiatives are working towards more consistency; for example, in the EU MAES initiative, ecosystems are all analyzed and measured using the same procedures (Donnely et al. 2016).

There is as yet no evidence of the effectiveness of the use of NCA for the protection of nature, biodiversity, and nature's contributions to people. As an information instrument, its effectiveness is based on the premise that more information will result in better decision-making (Guerry et al. 2015; Mace et al. 2015) – a premise that is largely unsupported and problematic (Turnhout et al. 2013; Wesselink et al. 2013). Yet, as has been shown for other information tools such as models or indicators (Turnhout et al. 2007; Van Egmond and Zeiss), NCA may be helpful as a tool for the facilitation of dialogue on the diverse values of nature and biodiversity. However, in order to enable this role, it is considered important that it uses a broad perspective that includes non-economic values and that it employs a participatory approach so that relevant stakeholders can contribute to the definition, identification and assessment of Natural Capital (Raymond et al. 2009).

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