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Effects of Renewable Energy Production and Infrastructure on Wildlife

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Introduction

The high levels of human demands of resources—from food to space and energy—are one of the main drivers of global change and are causing large negative impacts on ecosystems functioning worldwide (Vitousek et al. 1997). Global change components range from climate change to habitat destruction, species invasions, pollution and eutrophication. Although these factors can produce ecosystem changes independently, the final

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descent is often driven by synergistic processes. The resulting amplifying feedbacks can be disconnected from the original driver of change, leading to a state shift in the biosphere with unexpected consequences (Barnosky et al. 2012). Climate change seems to have taken prominence over other drivers of global change, leading to larger funding and attention with respect to other major components of global change (Veríssimo et al. 2014).

Human population growth and growing per capita consumption are causing an increasing energy demand to support industrial and domestic activities. Reducing the emissions of greenhouse gases responsible for current climate change is the main goal of renewable energy production. Major efforts have been devoted to the development of this kind of energy, including longstanding sources such as hydroelectric infrastructure along with more recent technologies such as wind farms and solar plants (Johansson et al. 1993). Renewable energy aims to provide humans with sustainable resources (Dincer 2000), although the development of infrastructure aimed to produce and distribute it may also have detrimental effects on ecosystems. Paradoxically, the development of renewable energy might also jeopardize biodiversity by increasing extinction rates of endangered species (Hooper et al. 2012). Thus, it is important to reconcile the production of renewable energy with the conservation of biodiversity to meet the primary objective of sustainable development.

No energy source is entirely 'biodiversity-neutral': nuclear power has obvious waste-disposal problems, the manufacture of solar cells can create hazardous waste, hydropower alters freshwater ecosystems, and wastewater from geothermal plants affects aquatic ecology. The transmission lines, roads, and other infrastructure associated with all 'clean' power projects can also have extensive impacts on ecosystems. Given that energy consumption is set to double by 2030, the environmental and business communities must find ways to assess and manage the tradeoffs between energy generation and use and biodiversity conservation. Conservationists are racing to gather the necessary information about the impacts and make sure it feeds into the decision-making process. In this chapter, we review previously described as well as less explored effects of renewable energy production and infrastructure on biodiversity, from species to populations and communities, and discuss the main guidelines to mitigate their impacts.

Hydroelectric Production Infrastructures: Overview

Hydroelectricity refers to the electric energy generated by the movement of water though turbines. Hydroelectric production most often requires damming a river to simultaneously ensure the availability of water and provide the water jump necessary to move the turbines. Worldwide, there are almost 10,000 large dams (height >15 m) devoted to hydroelectric production, constituting almost 20 % of all built dams (ICOLD 2014). The number of smaller dams (<15 m) is not precisely known, but they are thought to outnumber larger ones by some tenfold (Carpenter et al. 2011). China alone has almost 50,000 hydroelectric facilities (Kosnik 2008). Hydropower is used to produce over 16 % of total electricity globally, constituting over 75 % of the overall renewable electric production (REN21 2013). The main advantage of hydroelectric production over other renewables is the possibility of

responding quickly to demand peaks. During periods of low demand (e.g. at night) electricity can be used to pump water to reservoirs at higher altitudes, thus ensuring water availability for production during high demand periods.

Hydroelectricity has been presented as a "green" energy source, based on the lack of greenhouse gas emissions directly related to electricity production (Kosnik 2008). However, this view has been challenged in recent times due to the accounting of emissions from reservoirs and downstream water reaches (Fearnside 2004; Kemenes et al. 2007). Reservoir-linked emissions can be related to the deforestation of the area occupied by the water mass and the decay of submerged organic matter (Fearnside 2000), and by the production of methane due to microbial metabolism in anoxic environments (Bastviken et al 2011). Hydroelectric development also has socioeconomic impacts on local communities, being cited as one of the main drivers of forced displacement and resettlement (Scudder 2005). Resettlement usually involves a change in living means due to environmental constraints, e.g. when people are moved from fertile rural areas to cities (Wilmsen et al. 2011), and most often results in the impoverishment of affected households (Scudder 2005). Flow regulation for hydroelectric production can negatively affect downstream agricultural lands (Kuenzer et al. 2013) and fisheries (Silvano et al. 2009), while the creation of hydroelectric reservoirs can have negative health implications in some areas (Yewhalaw et al. 2009). Nevertheless, we will not further discuss these global climate and social issues here and will henceforth focus on the ecological impacts of hydroelectric production on aquatic systems and their biota. Many of these impacts are related to the presence of dams and are thus not exclusive of hydroelectric infrastructures, being shared with those of dams devoted to irrigation, urban and industrial water supply, or other uses.

Barrier Effect

Dams constitute impassable barriers for the movement of most strictly aquatic animals. The fragmentation of river networks by dams has led to the collapse of migratory fish populations in many areas of the world (e.g. Holmquist et al. 1998; Limburg and Waldman 2009; Hall et al. 2012). A search in the IUCN red list (www.iucnredlist.org) provides a list of 452 threatened fish species that are negatively affected by dams (threats 7.2.9, 7.2.10 or 7.2.11 in the IUCN threat classification), including 85 critically endangered and 8 extinct taxa. These numbers underestimate the real number of fish species affected by dams, because fish species have not been globally assessed and the red list covers less than 50 % of existing freshwater fish species.

The impacts of river fragmentation can be extreme for anadromous migratory fish, such as migratory salmonids (Fam. Salmonidae) or clupeids (Fam. Clupeidae), because the establishment of a dam can completely impede reproduction. Sturgeons (Fam. Acipenseridae) are perhaps the clearest example of highly threatened anadromous fish affected by river fragmentation, with all 25 sturgeon species assessed by the IUCN negatively affected by dams. For example, the construction of the Gezhouba (in 1981) and Three Gorges (2003) dams in the Yangtze river has led to

large reductions in the ranges of the critically endangered Chinese (*Acipenser nudiventris*) and Yangtze (*A. dabryanus*) sturgeons (Zhang et al. 2015).

Catadromous fishes, i.e. those that reproduce in the sea but spend an important part of their life in freshwater systems, are often less severely impacted by dams than anadromous ones. However, the barriers can impede the occupancy of an important proportion of the original habitat of a species, which can in turn be translated into significant population declines. For example, dam construction and associated loss of freshwater habitats is one of the main drivers of the recent collapse of the European eel (*Anguilla anguilla*) (Kettle et al. 2011). Dams also act as effective barriers for tropical catadromous fish assemblages, even precluding the upstream migration of climb-adapted gobies (Fam. Gobiidae; Cooney and Kwak 2013). Dams can also constitute important or absolute barriers for species that move along river systems, using different habitats within the year and/or during different life-history stages. This is the case for several fish species in the Amazon basin, which have been severely affected by hydropower development in Brazil (Godinho and Kynard 2009), and also of different river dolphins in Asia (Dudgeon et al. 2006).

The impediment of downstream movement of the river biota is also a relevant impact of hydroelectric infrastructures. A key element of the downstream barrier is not the dam itself but the reservoir behind it. Reservoirs are stagnant water masses that differ radically from river systems, and thus can eliminate most of the ecological cues followed by migrating fishes, causing disruptions in their migrations (Pelicice et al. 2014).

Most, if not all, of the outflow of hydroelectric dams passes through a turbine system to produce electricity. Fish mortality in these turbines can be high, even approaching 100 %, although it is dependent on the type of turbines, their operation, and the different species and life history stages involved (Larinier and Travade 2002). Because the risk posed by turbines increases with body length, long-bodied species such as freshwater eels (Fam. Anguillidae) often are more severely affected by the downstream barrier effect of hydroelectric infrastructures (Calles et al. 2010).

For more than a century there have been important technological developments to facilitate the movement of fish across dam barriers (Katopodis and Williams 2012). These structures have proven useful in several situations for good swimmer (e.g. salmonids) or good climber (e.g. eels) species (Laine et al. 2002; Feunteun 2002), especially for relatively small barriers. However, passing facilities designed for northern hemisphere fish (most commonly salmonids) often fail to be effective in other environments (e.g. Roberts 2001; Mallen-Cooper and Brand 2007). On the other hand, although technical solutions have been relatively successful in facilitating upstream fish migration, the elimination of the barrier effect for downstream movement is often much more complicated (Feunteun 2002). Proposed actions include the design of modified turbines (Čada 2001) or the implementation of lateral bypasses (Gosset et al. 2005), but their effectiveness and widespread applicability has not been proven. Some successful restorations of fish populations by eliminating barrier effects exist, but failures have also been reported, even in cases where enormous efforts had been invested.

An analysis of anadromous fish restoration programs in large North American rivers fragmented by hydroelectric dams concluded that strategies based on fish passes had failed and that dam removal was the only viable and realistic option (Brown et al. 2013).

Flow Regulation and Ecosystem-Level Impacts

The alterations of natural flow regimes (Poff et al. 2007) are one of the most widespread human impacts on aquatic systems, with the overall potential for water retention in existing dams being more than five times larger than the total volume of water of all the rivers in the world (Dudgeon et al. 2006). Flow regulation affects more than 60 % of the world's large river systems, and there are regions (e.g. Europe) where unregulated large rivers do not exist anymore (Nilsson et al. 2005). In natural conditions, flow regimes differ among regions and among rivers within regions (Poff et al. 1997). This diversity in the functioning of river systems drives the high intersystem variability (or β-diversity) of the river biota. Flow regulation has homogenized this originally diverse scenario so that river systems are nowadays much more alike in their functioning than they were before dam construction (Poff et al. 2007).

Reservoirs retain water during flood periods and often increase baseline flows during naturally dry seasons, thus smoothing the temporal variability in flow conditions (Poff et al. 1997). Of particular relevance is the buffering or elimination of peak floods due to their importance for river ecosystems, including their floodplains, riparian ecosystems, estuaries and deltas, as well as for their biota (FitzHugh and Vogel 2011). After the closure of the High Aswan dam in the mid-1960s, the discharge of the Nile River during the annual flood period decreased by around tenfold (Tockner and Stanford 2002) with severe consequences for floodplain and estuary areas, even affecting marine fisheries in the Eastern Mediterranean (White 1988). Elimination of floods reduces the lateral connectivity of river systems, often precluding the inundation of floodplains that are critical habitats for the reproduction of several aquatic and semiaquatic organisms (Nilsson and Dynesius 1994; FitzHugh and Vogel 2011). The impact of flow regulation extends beyond quantity of water and also affects water quality. Dams often release hypolimnetic water, which is cold and has little dissolved oxygen, with severe consequences on biological communities in downstream areas close to the dam (Ward and Stanford 1982; Jurajda et al. 1995; Zhong and Power 1996).

In water-abundant regions throughout the world, hydroelectric dams, especially small ones, are run-of-the-river facilities (Richter and Thomas 2007), in which the discharge running through the turbines is similar to that entering the reservoir. Dams operated in this way may have small effects, if any, for the flow regimes of river systems. However, hydroelectric facilities in drier areas and (especially) large hydroelectric dams do store water and disrupt hydrological regimes. Hydroelectric dams may also have specific, short-term impacts on river flow related to the

variability in electricity demand (Bevelhimer et al. 2015). The daily demand fluctuations can be translated into large, completely unnatural daily variations in flow below hydroelectric dams, with strong negative impacts on the aquatic biota (Cushman 1985).

Upstream Habitat Modifications

World's reservoirs occupy a combined area of about 500,000 km^2, similar to that of France (Nilsson 2009). The loss of terrestrial habitat affects valley bottoms especially, which are the most productive environments in many areas (e.g. mountains, high latitudes) and often host diverse and abundant biological communities (Nilsson and Dynesius 1994; Nilsson and Berggren 2000). The new water masses can also constitute effective barriers precluding the movement of terrestrial species and disrupting migration behaviors (Nellemann et al. 2003). The loss of generally diverse riparian ecosystems is almost never compensated by the establishment of riparian vegetation around the reservoir shores, due to the high water-level fluctuations. Whenever there is a chance for the formation of a riparian vegetation fringe, it is most often dominated by invasive plants and/or those typical of eutrophic systems (Hill et al. 1998; Nilsson 2009). However, reservoirs (including some hydroelectric facilities) can also constitute relevant feeding and breeding habitat for threatened species (McCartney 2009) and become refuge habitat in semi-arid territories during water shortages (Prenda et al. 2001).

The impoundment of a river generates a radically different and simplified stagnant water mass that results in a hostile environment for an important part of the fluvial biota (Baxter 1977). The lack of water currents and turbulence, and the loss of complexity in the substrate due to the deposition of fine sediments, produce a radical homogenization of the water landscape. As a consequence, species linked to running water or those needing complex habitat structures at some stage of their life history tend to be absent from reservoirs (Gido et al. 2009). The fluctuation in water levels, which in hydroelectric dams can be large within each day (Nilsson 2009), hinders the occupation of the littoral zone by many slow-moving animals, including many invertebrate taxa (Baxter 1977). In general, the aquatic biota occupying reservoirs is simplified and impoverished when compared to rivers (Gido et al. 2009; Clavero and Hermoso 2011), and results in low quality habitats for aquatic top predators (Palmeirim et al. 2014).

Reservoirs, Flows and Invasive Species

A relevant ecological impact of reservoirs is their relationship with invasive aquatic species (Havel et al. 2005). Reservoirs host more invasive species than rivers (Clavero and Hermoso 2011) and natural lakes (Johnson et al. 2008). The

invasibility of reservoirs can be related to two phenomena. First, reservoirs are stable environments and very homogeneous when compared to rivers (Clavero et al. 2013). These characteristics may facilitate the establishment of many widely introduced species such as the common carp (*Cyprinus carpio*) or the zebra mussel (*Dreissena polymorpha*). Second, reservoirs are associated with many of the activities related to the release of non-native aquatic organisms, such as sport-fishing and recreational navigation (Havel et al. 2005; Johnson et al. 2008).

Reservoirs are not only easily invaded, but also act as facilitators for the invasion of associated river systems (Havel et al. 2005). Reservoirs may function as a source of individuals of invasive species to upstream river stretches (Rincón et al. 1990; MacIsaac et al. 2007). Naturally flowing rivers are suboptimal habitats for many invasive species, especially in areas with high environmental fluctuations for which invasive species may lack adaptations. However, reservoirs, being more stable and thus less hostile systems for invasive species, may compensate the environmental harshness of rivers by being a constant source of colonizing individuals. Reservoirs can also be a source of individuals of non-native species for downstream reaches (MacIsaac et al. 2007), the invasion process being in this case reinforced by the alteration of flow regimes. Most frequently, non-native species are not adapted to cope with the specific natural flow regimes of the river systems where they are released. The softening of discharge extremes due to flow regulation eases their establishment in river sectors downstream from reservoirs (Hermoso et al. 2011). Thus, apart from the barrier effects and habitat changes, reservoirs may exert an indirect negative impact on native species (both within the reservoir and in associated river systems) by enhancing populations of harmful invasive species (Hermoso et al. 2011; Clavero et al. 2013)

Solar and Wind-Energy Infrastructures: Overview

During the last decades, the number of wind and solar energy developments has increased substantially all over the world, receiving support as alternative energy sources that can achieve substantial avoidance of greenhouse gas emissions. By the end of 2013, worldwide installed capacity for wind and solar energy were 318.1 and 138.9 GW, respectively. China, USA, Germany and Spain accounted for more than 65 % of the worldwide installed capacity of wind energy (GWEC 2014), whereas solar energy installation has been led by Europe, followed by China, Japan, and the United States.

In this section, we review the impacts of solar and wind energy production and infrastructure. We consider direct impacts on wildlife populations through habitat transformation and demographic unbalances (e.g., Langston and Pullan 2003; Baerwald et al. 2008; Garvin et al. 2011) as well as changes in ecosystem functioning at different scales, which can be a source of further concern for communities and populations of both plant and animals. We include not only terrestrial infrastructure but also the increasingly implemented and less studied offshore developments.

Effects on Landscapes and Ecosystems

The occupation of the territory by solar and wind energy systems (SWES) involves a change in land use, and their operational characteristics can locally modify mass and energy fluxes, with potential effects on soil-plant processes and ecosystem services such as soil formation and nutrient cycles (supporting services), climate and hydrology (regulating services), water and food supply (provisioning services), and recreational and aesthetic activities (cultural services). Some of these changes have been extensively reported, whereas others, mainly related to indirect impacts, still require further assessment (Tsousos et al 2005; Saidur et al. 2011; Leung and Yang 2012; Aman et al. 2015).

The main environmental impact of SWES is land use change. On average, the footprint area associated to SWES is estimated at 13–20 m^2 for a 5-MW wind turbine, 1.9 m^2 for a 160-W solar photovoltaic system, and 1.9–2.4 km^2 for a 100-MW concentrated solar power system (Jacobson 2009), which results in 1 m^2 of land being required to produce 0.38–0.25 MW (wind energy) or 83–42 W (solar energy). Vegetation is usually removed and the soil is graded, which enhances soil erosion or aeolian sediment transport, as well as loss of organic carbon and nitrogen, especially in semiarid/arid conditions. Nonetheless, SWES impacts in terms of landscape (use and cover) disturbances are much lower than those due to conventional energy systems (Tsousos et al. 2005). Photovoltaic systems, for example, have higher land use efficiency than other renewable energy sources and surface coal mining (70 % of global United States coal extraction). These numbers highlight the need for addressing solar energy environmental impacts (and energy issues, in general) in the global framework of substitution, and not in their absolute values (Jacobson and Delucchi 2011).

Microclimatic conditions can be influenced by SWES (Baidya 2011; Hernandez et al. 2014). Ground-mounted photovoltaic arrays directly intercept precipitation and atmospheric deposition, change surface albedo and increase shading on a local basis, greatly altering the radiation budget on the ground surface. Moreover, their presence affects the wind speed and enhances turbulence at the near-surface atmospheric boundary layer. All these factors may significantly modify energy fluxes over the affected surface and cause changes in local temperature, air and soil moisture, thus affecting evaporation dynamics. Wind farms have little influence on the radiation components, but they strongly change the wind profile distribution and magnitude, and affect turbulence and mixing within the atmospheric boundary layer, which also modifies temperature and air moisture profiles, and again the final energy and water budgets (Baidya et al. 2004).

These effects are also dependent on the diurnal cycle of solar radiation. Photovoltaic panels reduce the shortwave reflected solar radiation due to their lower albedo when compared to ground surface, but they increase the ratio of diffuse/direct shortwave radiation below the panel (Scherba et al. 2011). Changes of soil conductance and air warming below panels are not clear, because the effects on wind must be taken into account and different results can be obtained depending on

different additional factors (Taha 2013). Finally, the rainfall interception results in an enhanced draining along the points from the lower end of the panel surface, which significantly alters the infiltration/runoff ratio over the ground beneath, especially for light and intermittent rainfall events. Drop erosion processes may also occur due to this redistribution of rainfall over the ground. At larger scales, the different importance of each hydrological component in the water cycle would be altered, but the significance of this change is dependent on the time distribution pattern of precipitation at the area and the local arid/wet regime (Pisinaras et al. 2014). Since solar energy facilities are efficient in regions with high insolation rates, arid and semiarid areas would be more affected by these impacts, not all of them being negative (Turney and Fthenakis 2011).

As for wind-farms, the impacts on temperature have been observed in both directions. Night-time surface temperature may be increased downwind from turbines, since warmer air eddies would mix into the cooler air; the inverse effect would be observed during daytime. Under stable atmosphere conditions, an increase in evapotranspiration would be expected. Direct effects on microclimate may also induce indirect impacts on the physical environment. Besides water in air and soil, other gases like CO2, methane, and nitrous oxide could modify their concentration profiles over the area affected by SWES, not only due to mixing condition alterations but also to the forcing action of changes in temperature. Changes of 0.7–3.5 °C have been measured within the influence area of wind farms (Baidya and Traiteur 2010; Zhou et al. 2012), and increases of 2.5–26.0 °C in the area surrounding solar facilities in the developed environment (Scherba et al. 2011). Plant and soil processes involving water and energy exchanges, microbial activity, C and N cycling, and other biogeochemical cycles could be significantly altered depending on the initial conditions, and the extent and location of the SWES, with further consequences on the ecosystem dynamics (Armstrong et al. 2014).

Potential effects on climate may work at larger scales. For example, Wang and Prinn (2010) concluded from modelling that the induced changes by wind farms in surface heat fluxes and temperature could result in modified cloud cover distributions, while Fiedler and Bukovsky (2011) obtained an increase of rainfall after modelling on a large scale. However, global circulation models and their downscaling to surface models still have a significant uncertainty related to the adequate inclusion of interactions between physical and biological processes (Cramer et al. 2001; Hawkins and Sutton 2011).

Effects of Solar Plants and Wind Farms on Wildlife

Solar plants are relatively new and their effects on biodiversity have been scarcely documented (but see DeVault et al. 2014). On a local scale, impacts are associated with the above-mentioned habitat transformation and wildlife mortality (Lovich and Ennen 2011; Hernandez et al. 2014). For example, avian mortalities at a 10 MW concentrating solar thermal power plant in California, USA averaged 1.9–2.2

individuals per week, and were mainly caused by collision with site infrastructure (81 %), particularly with heliostats, and to a lesser degree, burning when heliostats were oriented towards standby points (19 %), especially for aerial foraging species (McCrary et al. 1986). Moreover, mortality rates can be increased because of the soil degradation and the creation of roads, and exotic species invasions could even occur.

One potential solution proposed to reduce the negative impacts of solar energy production on wildlife is to locate some solar facilities on unused portions of airport lands (DeVault et al. 2012, 2014). Airports represent one of the only land use types where reduction in wildlife occurrence is justified and socially acceptable, due to the risk of wildlife-aircraft collisions (DeVault et al. 2013). Thus, wildlife conservation is largely discouraged at airports because of safety concerns (Blackwell et al. 2013). Further, airports often occupy extensive areas and much of the land is undeveloped; for example, airports in the USA collectively contain well over 3300 km^2 of idle grasslands (DeVault et al. 2012). Also, photovoltaic solar facilities are largely compatible with airport operation from a safety perspective (Barrett and DeVita 2011), and solar energy production using photovoltaic arrays is generally economically advantageous for airports (DeVault et al. 2012).

Concern about wildlife mortality at wind farms began to surface in the late 1980s and early 1990s, and it was mostly focused on the Altamont Pass Wind Resource Area (APWRA), a 165 km^2 wind farm near San Francisco, California. Hundreds to thousands of birds, including more than 40 species, some of them endangered, died there every year (Asmus 2005; Thelander and Rugge 2000; Smallwood and Thelander 2005; Thelander 2004; Smallwood and Thelander 2008). Such fatalities are not limited to California or the USA (Erickson et al. 2001; US GAO 2005). European countries such as Spain and Belgium also have reported hundreds of birds killed by turbines (Lowther 1998; Everaert and Stienen 2007). Thus, much wind farm-wildlife research has been devoted to investigating how wind farm developments impact bird populations (e.g., Langston and Pullan 2003; Baerwald et al. 2008; Garvin et al. 2011), in particular collision rates of birds with turbines as well as factors influencing interspecific and local variability (reviewed in Drewitt and Langston 2006, 2008; Kuvlesky et al. 2007; Stewart et al. 2007; Loss et al. 2013; Marqués et al. 2014). These studies show that the effects of wind farms on birds are highly variable (while most wind turbines actually kill none or very few individuals, some turbines kill many), and depend on a wide range of factors, including the development type, the topography of the surrounding land, the habitats affected and the number and species of birds present (Barrios and Rodriguez 2004). In general, risks are higher when turbines are placed on ridges and upwind slopes, built close to migration routes, or operated during periods of poor visibility such as fog, rain, and at night (Sovacool 2009). Also, modern, monopole wind turbines might kill fewer birds than older, lattice turbines, such as those that were located at APWRA (Loss et al. 2013; but see Barrios and Rodriguez 2004).

There is a general acceptance of the idea that collision mortality would increases with bird abundance (Musters et al. 1996; Osborn et al. 2000; Drewitt and Langston 2006; Tellería 2009a, b). Although linearity in this relationship could be, a priori, a

simplistic assumption because of interspecific differences in susceptibility to this infrastructure, higher abundance of individuals of species sensitive to collision at wind farms would increase fatality rates. In particular, raptors, grouse, gulls and terns tend to collide more often than expected from their occurrence and numbers (Carrete et al. 2009, 2012). Thus, the most effective measures to minimize negative effects on birds is to identify the dangerous locations and avoid locating wind turbines there. Most accidents with birds occur in places where the more vulnerable species concentrate, so the use of reliable estimates of abundances (such as those derived from the location and size of breeding or roosting sites; Carrete et al. 2012) should be preferred over highly variable punctual, year estimates (Barrios and Rodriguez 2004; de Lucas et al. 2008). Otherwise, as is case for most current risk assessments studies in countries such as Spain, it is not surprising to find weak relationships between the predicted risk and the recorded mortality at wind farms (Ferrer et al. 2011).

A major difficulty in assessing the impact of wind farms on bird populations is the scarcity of long-term studies at operational wind farms. Thus, there is a widespread belief that wind farms have, at most, a low impact on animal populations (Marris and Fairless 2004). However, the few studies evaluating the long-lasting effects of wind farms on wildlife advise caution. A recent study carried out in the UK using long-term data of breeding bird abundances show that wind farm developments may result in significant reductions in habitat usage (from 100 to 800 m from the turbines after construction) by some species. This result in the decline in the abundance of some breeding birds such as red grouses *Lagopus lagopus scoticus*, snipes *Gallinago gallinago* and curlews *Numenius arquata* (in some cases by up to 50 % within 500 m of the turbines during construction), without general recoveries after the first year of operation (Pearce-Higgins et al. 2009, 2012). Another study focused on the impact of wind farm mortality on the demography of an endangered species, the Egyptian vulture *Neophron percnopterus*, show that even very low levels of additional mortality can be significant for this long-lived species with low productivity and slow maturation rate (Carrete et al. 2009; Sanz-Aguilar et al. 2015), as is the case with many endangered or rare long-lived species (Saether and Bakke 2000). Very low reductions in survival rates of territorial and non-territorial birds (−0.015 and −0.008, respectively) associated with wind-farms can have significant population impacts (Carrete et al. 2009; Bellebaum et al. 2012; Sanz-Aguilar et al. 2015). Regrettably, this a widespread scenario affecting many other long-lived endangered species also killed at wind farms in different European countries (e.g., at least 10 white-tailed sea eagle *Haliaeetus albacilla* per year in Norway; http://www.statkraft.com/pub/wind_power/feature_articles), in the USA (e.g., 65 golden eagles *Aquila chrysaetos* in California per year; Smallwood and Thelander 2008) and in Australia (e.g., at least 12 Tasmanian wedge-tailed eagles *Aquila audax fleayi* in 4 years; http://www.windaction.org/news/17683).

The other taxonomic group highly affected by wind farm mortality are bats. The first reports of bat fatalities at wind farms occurred in North America and Europe during the 1990s (Kunz et al. 2007; Rydell et al. 2010). However, the occurrence of bat fatalities at wind farms is a global phenomenon with cases

described from numerous countries including Australia (Hull and Cawthen 2013) and South Africa (Doty and Martin 2013). The lack of information on the impact of wind farms on bat populations reflects the severity and poor understanding of the problem. This is concerning because in a country such as the United States, which produced 51,630 MW of wind-energy in 2012, some studies have estimated that wind farms kill between 600,000 and 888,000 bats per year (Hayes 2013; Smallwood 2013). Spain is one of the world leaders in wind farm technology and in 2010 produced 19,148 MW of wind-energy. Despite deficiencies in post-construction monitoring of the impact of wind farms on wildlife (e.g. biases in searcher efficiency and carcass scavenging), the estimated number of wind-energy related bat fatalities in Spain can be comparable to the highest estimates available from North America (Camina 2012). In fact, bat mortality at wind farms tends to be higher than that of birds (Barclay et al. 2007). For example, in USA, estimates are 888,000 bats killed per year *vs* 573,000 birds (Smallwood 2013). This is of great concern because bats are extremely long-lived for their size and they have a low reproductive potential (Barclay and Harder 2003). Thus, the cumulative impacts of this new hazard could result in long-term population declines or even extinctions in certain areas (Kunz et al. 2007), especially given that many bat populations are already under severe stress due to mortality from white-nose syndrome (Foley et al. 2011).

Bats may be attracted to wind turbines for many different reasons, including curiosity, searching for food or potential roosts, or social interactions (Cryan and Barclay 2009). The presence of aviation warning lights on the turbines does not increase mortality rate (Johnson et al. 2004; Bennett and Hale 2014); however a recent study has shown that tree roosting bat species from North America were attracted to the turbines (Cryan et al. 2014). This attraction could be due to the visual confusion of the turbines silhouettes with trees, reinforced by other cues such as similar downwind airflow patterns. Attraction of tree bats to other tree-like structures such as tall communication towers only appears to occur in the late summer and autumn, possibly because of social rather than foraging behavior, as they emit fewer feeding buzzes around these sites (Jameson and Willis 2014).

About half of dead bats examined around wind turbines do not show any external injury caused by direct collisions with turning blades. Instead, most of them have pulmonary lesions and internal hemorrhages compatible with barotrauma caused by rapid air-pressure reduction near fast moving turbine blades (Baerwald et al. 2008). Barotrauma has not been suggested as a cause of bird mortality because of differences in their respiratory anatomy. Although the ability to echolocate allows bats to detect and avoid turbines blades, it seems they cannot detect rapid pressure reductions and may die even if they do not come in contact with the blades.

Out of the 23 species of bats reported to be affected by wind turbines in North America (http://www.batcon.org/), about 80 % of fatalities involve migratory tree-roosting bats (mainly *Lasiurus cinereus, L. borealis* and *Lasionycteris noctivagans*; Arnett et al. 2008). In Europe, 27 bat species have been reported as killed by turbines (http://www.eurobats.org/), and 98 % of these fatalities belong to a "high-risk" species group formed by bats included in the aerial-hawking guild (i.e., species

that catch their prey in open spaces; Rydell et al. 2010. Although it could seem that different bat species groups are susceptible to wind turbines in America and Europe, it should be taken into account that all migratory tree-roosting bats from America are aerial-hawking, while mortality of migratory species also occurs in Europe but in smaller proportions (64 % in NW Europe, Rydell et al. 2010; 56 % in Greece, Georgiakakis et al. 2012; 40 % in Portugal, Amorim et al. 2012). However, a significant proportion of migratory killed bats in Europe belong to resident populations (Lehnert et al. 2014).

In both North America and Europe most bat fatalities (90 %) occur during the late summer and early autumn with another minor peak occasionally seen in the spring (Kunz et al. 2007; Rydell et al. 2010). There are no consistent sex or age biases in bat mortality during these periods (Arnett *et al*. 2008; Rydell et al. 2010; Hull and Cawthen 2013). Young bats are not more vulnerable despite their typical dispersal behavior and expected lack of experience. In North America, the mortality peak coincides with the migration period of tree-roosting bats.

As among birds, some studies have attempted to generalize bat mortality patterns associated with wind farms. Highest bat mortality has been observed along forested ridge tops in the Appalachian Mountains and forested hilltops in southern Germany, with lowest records in flat agricultural landscapes. Although local concentrations of mortality at specific turbines has been occasionally described (Piorkowski and O'Connell 2010; Georgiakakis et al. 2012), most bat fatalities are randomly distributed across turbines (Arnett et al. 2008), making it difficult to draw clear guidelines for conservation planning (Kunz et al. 2007; Rydell et al. 2010). However, most fatalities occur during low wind nights (<6 m/s) in late summer and the first half of autumn, thus increasing the cut-in speed of the turbines on nights with high risk of bat collision would be an effective management tool to reduce mortality. Indeed, these methods have achieved reductions of bat fatalities from 50 to 90 % with marginal power loss ($\leq$ 1 % of total annual output) (Baerwald et al. 2009; Arnett et al. 2011). Conversely, other management measures such as acoustic deterrents are less effective, in part due to rapid atmospheric attenuation of ultrasounds (Arnett et al. 2013).

Offshore wind energy is developing rapidly and is rapidly occupying marine areas to produce low carbon energy. Whilst acknowledging that research into the impacts of the offshore renewable industry is still in its infancy, it is widely regarded that the risk for impacts on the marine environment may not be negligible and must be taken seriously. Noise disturbance, electromagnetic fields, and migration barriers have had some negative effects on fish, marine mammals, birds and seabed communities (Wilhelmsson et al. 2010). On the other hand, these installations create 'no-take zones' around them and their underwater vicinity that can function as artificial reefs, which leads to a greater abundance of many species (Wilhelmsson et al. 2010). But if not properly planned and managed, these installations can adversely affect marine biodiversity through habitat loss, collisions with turbines, deviation of the migratory routes, noise and electromagnetic disturbance and navigational hazards for ships (Desholm and Kahlert 2005; Larsen and Guillemette 2007; Wilhelmsson et al. 2010).

In spite of the previous information, the understanding of the potential implications of large-scale renewable energy developments has not kept pace with the recent rise in the number of development proposals. The risk that an animal could be killed at a wind turbine is probably small compared to the risks faced from other human activities (Calvert et al. 2013; DeVault 2015), and some not very successful attempts have been made to compare wind turbine mortality with fatality rates associated with energy sources (for example, by calculating a number of birds killed per kWh generated for wind electricity, fossil-fuel, and nuclear power systems; Sovacool 2009). However, the point here is not to ascertain that turbines are or are not the leading cause of bird deaths, something that can change when considering the current pace and scale of wind energy development (Loss et al. 2013). The point is to recognize that renewable energy can help to reduce greenhouse gas emissions and slow-down climate change, but we should develop them in ways that account for and minimize their impacts on wildlife. Unlike fossil-fuel and nuclear power plants, which spread their wildlife-related impacts across large scales, most of the impact from wind farms occurs locally, so solutions are relatively straightforward.

Energy Infrastructure: Power Lines and Wildlife

Renewable energy produced by wind, solar and hydroelectric facilities not only impact biodiversity during the production stage; these facilities also need power lines to transport the electricity to final consumers. An extraordinarily dense network of power poles and lines is located around cities and industrial areas, and have impacts on wildlife in various landscapes around the world. Power lines have significant potential impacts on biodiversity, mainly through changes in habitat structure and wildlife mortality.

The presence of poles and wires introduces lineal anthropogenic structures that alter the visual natural quality and create division lines on the landscape (Arriaza et al. 2004). This applies especially to transmission lines, the higher voltage power lines (> 66 kV). Their presence causes severe changes in habitat structure, increasing fragmentation as a consequence of the removal of natural vegetation below the lines (Luken et al. 1992; Forrester et al. 2005). This change does not always have negative consequences; some species could benefit from the new habitats created (Askins et al. 2012), e.g. forest ungulates could benefit from foraging in power line rights-of-way where there is increased availability of pastures compared to adjacent forest (Bartzke et al. 2014). However, transmission power lines can behave as barriers to animal movements by disrupting migratory routes and promoting the development of avoidance strategies, as described for the reindeer (*Rhandifer tarandus*) (Reimers et al. 2007; Vistnes et al. 2004). Moreover, as a consequence of electric transmission, power lines generate strong electromagnetic fields, UV discharges and acoustic pollution which can affect animal health and behavior (Phernie et al. 2000; Tyler et al. 2014) and have also been identified as causes of wildfires (Tenforde 1992; Haas et al. 2005).

Probably the most serious environmental impact of power lines is avian mortality caused by electrocution, entangling, and collision (Bevanger 1998; Gangoso and Palacios 2002). Mortality associated with power lines can accelerate the declines of several species and affect occupation patterns (Sergio et al. 2004) or population dynamics (Schaub et al. 2010). Power lines are currently considered one of the main human-related causes of bird mortality worldwide (Bevanger 1998; Loss et al. 2014).

Electrocutions and Collisions

Electrocution and collision with power lines are among the main causes of population declines for some species, mainly raptors (Lehman et al. 2007). These include the Cape Vulture (*Gyps capensis*) in South Africa (Ledger and Hobbs 1999), the Egyptian vulture (*Neophron pernopterus*) in Canary Islands and East Africa (Donázar et al. 2002; Angelov et al. 2011), the griffon vulture (*Gyps fulvus*) in Israel (Leshem 1985), the eagle owl (*Bubo bubo*) in France (Bayle 1999) and Italy (Rubolini et al. 2001), the golden eagle (*Aquila chrysaetos*) and the bald eagle (*Haliaetus leucocephalus*) in USA (Harness and Wilson 2001) and Canada (Wayland et al. 2003) and the Spanish imperial eagle (*Aquila adalberti*) (González et al. 2007) and Bonelli's eagle (*Aquila fasciata*) (Real et al. 2001) in Spain.

Several studies have found that bird mortality at electric facilities is not randomly distributed, but concentrated in a very small percentage of pylons (Mañosa 2001; Guil et al. 2011). For example, electrocutions mostly occur in distribution power lines (<66 kV), where the dimensions of the supports are conducive to animals simultaneously touching the wires and the support. Other factors affecting electrocution risk are the characteristics of the landscape (topography, vegetation, prey abundance), that of the pylon (cross harm design, material) and weather conditions (external factors), with all of them usually being spatially correlated (Bevanger 1998; Haas 1980; Ferrer et al. 1991; APLIC 1996; Janss and Ferrer 1999, 2001; Mañosa 2001; Lehman et al. 2007). Identifying the most dangerous pylons and correcting or replacing them can significantly reduce the number of fatalities (Tintó et al. 2010; López-López et al. 2011; Guil et al. 2011).

Collisions occur when a flying bird hits any of the wires (conductors or ground wires). Collisions can occur at any type of power line, and even at other lines such as telephone and telegraph wires or railway catenary (Bevanger 1994, 1998). Many studies have reported annual estimates of bird mortality due to collisions with power lines, and extrapolations from these studies produce estimates ranging from hundreds of thousands to millions of dead individuals (Manville 2005; Rioux et al. 2013; Loss et al. 2014). Although estimates may be biased upward due to the lack of random selection of sampling sites (Bevanger 1999; Jenkins et al. 2010), there is a general consensus that this impact is one of the main causes behind the population declines of some endangered species either locally or globally (Bevanger 1998; APLIC 2012). This is the case for the Whooping crane (*Grus americana*) and the

California condor (*Gymnogyps californianus*) in the USA and the Great bustard (*Otis tarda*) and the Little bustard (*Tetrax tetrax*) in the Iberian peninsula (BirdLife International 2004; Silva et al. 2010; APLIC 2012). More than 350 bird species are considered susceptible to collisions with power lines (Manville 2005; Prinsen et al. 2011a), including more than 50 % of the Spanish (Pérez-García and Botella 2012) and 17 % of the Italian breeding bird species (Rubolini et al. 2005).

Mortality rates due to power line collisions depend on biological, environmental, and engineering-related factors (Loss et al. 2014). Collision vulnerability varies between species due to several biological traits such as size, wing loading, flocking and flight behavior, habitat use, maneuverability in flight and vision. Species with high wing loading such as herons, cranes, swans, vultures and condors tend to be more frequently reported in collision casualties (APLIC 2012, and references therein). Flocking species, like waterfowl, and colonial species that move daily between resting or breeding sites to foraging areas are more vulnerable than solitary ones because these individuals have less space to maneuver and limited vision of the obstacle (Bevanger 1998; Janss 2000; Martin and Shaw 2010; Martin 2011). Vision in bird species and its relationship to collisions have been reviewed by Martin (2011), who suggested that bird collisions may be the result of both visual and perceptual constraints. Environmental factors such as weather conditions and visibility interact with biological characteristics (see above), enhancing risk of collisions. Stormy weather, fog, and wind can alter flight patterns (i.e. altitudes) and affect visual detection, increasing the probability of accidents (APLIC 2012). The location and technical design of power lines are among the most studied factors affecting bird collisions. The presence of shield wires above the conductor phases in transmission lines (> 60 kV) and its smaller diameter increases the risk of collisions and ground wires are involved in most bird fatalities (Bevanger and Brøseth 2001; Prinsen et al. 2011b; APLIC 2012). The location of power lines in areas with high bird abundances (i.e. wetlands) and landscape features can affect the path of bird flight directing them to wires (Janss and Ferrer 2001; Martin and Shaw 2010). Lines crossing migration, frequent displacement routes, or mountain ridges also can increase collision risk.

Mitigation Measures

The design and implementation of mitigation measures to avoid or reduce fatalities with power lines peaked in 1990s (Bevanger 1999; Janss and Ferrer 1999; Lehman et al. 2007). Despite the extensive literature during the last 25 years, and that modification of power lines has proven to be an effective method for reducing mortality in dangerous power poles (Janss and Ferrer 1999; Harness and Garrett 1999; Guil et al. 2011; López-López et al. 2011), power lines still remain an important bird mortality source worldwide (Bayle 1999; Rubolini et al. 2005; Lehman et al. 2007). The first step to reduce the negative effect of power line installation should be an efficient planning of electric transmission and distribution, in order to minimize the

extension of the actual and the future electric network. This can be achieved through a spatial aggregation of distribution and transmission lines or by bringing power generation closer to users (Prinsen et al. 2011a, b). The burial of the lines is the most effective solution to prevent the majority of the impacts of power lines on biodiversity and is the safest modification for birds. In fact, it is the only measure which eliminates the risk of electrocution and collision (APLIC 2006). But unfortunately, the economic cost is 3–20 higher than traditional overhead lines (APLIC 1994; Prinsen et al. 2011a, b) and can only be performed under certain conditions (e.g., low relief, medium voltage lines). Indeed, only in some countries of Central Europe it has been widely implemented as a common practice (Netherland, Belgium, Germany, and Norway).

The most widely used measure to mitigate avian electrocutions are the use of deterrents and modification of the supports, increasing the distance between conductors, and isolating the supports or spreaders to ensure that there is no contact between birds and wires (Harness and Garrett 1999; Haas et al. 2005; APLIC 2006; Prinsen et al. 2011b). Flight diverters (spirals, plates or spheres) attached to the wires are widely used to increase their visibility and reduce collisions. Effectiveness of flight diverters has been evaluated with diverse results, ranging from no reduction in mortality (e.g. Scott et al. 1972; Janss and Ferrer 1998; Anderson 2001) to slight (9 %, Barrientos et al. 2012) and strong reductions (60–80 %, Alonso et al. 1994; Bevanger and Brøseth 2001).

Conclusions

The production and transportation of renewable energy has several environmental impacts, ranging from the population to the ecosystem level (Table 1). Hydroelectric production is the major source of renewable energy worldwide and probably the most impacting one, not only because of its geographical extent but also because it affects ecosystem processes at the large scale. Migratory species like fishes are the most dramatically impacted taxa by hydroelectric infrastructure. Wind energy production is also an emerging source of environmental impact at both local and regional scales, with strong effects on certain bird and bat populations. Solar facilities impact mostly at the local scale through habitat alteration, although their effects on wildlife have rarely been studied. All these sources of energy share the need for transportation by means of power lines that have significant negative effects, particularly on bird populations at local and regional scales.

Fighting climate change is one of the major challenges of contemporary society and renewable energies are a key instrument to reduce greenhouse emissions. However, the greener energy is the one that it is not consumed, so reducing energy consumption should be the highest priority to minimize the effects of energy production on ecosystems and wildlife. Nevertheless, the increasing demand and even the need to turn from conventional fuel-dependent to renewable energies require the understanding of the potential effects of the latter on the environment. Under this

Table 1 Summary of the main effects of renewable energy on ecosystems and wildlife

Energy source	Ecosystem/habitats	Scale	Processes	Wildlife
Hydroelectric	Freshwater Riverine Coastal	Large Regional Local	Flow regime Barrier Habitat alteration Invasive species Regional climate	Anadromous migratory fish Catadromous fish
Wind	Terrestrial Marine	Regional Local	Habitat alteration Mortality Local climate	Birds (mainly raptors, waders, seabirds, passerines). Bats
Solar	Terrestrial	Local	Habitat alteration Mortality	Birds (mainly passerines)
Transmission	Terrestrial	Regional Local	Habitat alteration Mortality	Birds (mainly raptors, steppe birds, storks)

scenario, we consider that reducing energy consumption, planning infrastructures, and adopting mitigation measures should be, in that order, the key strategies to minimize the effects of renewable energy production and transportation. We would also recommend improving research on the emerging wind and solar facilities through more comprehensive assessments that require large spatio-temporal data sets. The scientific evidence of the long-term effects of hydroelectric production on species, populations, and ecosystems might help to visualize the potential effects of the other emerging renewable energy sources.

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