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Managing whale-watching as a non-lethal consumptive activity.

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Abstract

Marine tourism is a new frontier of late-capitalist transformation, generating more global revenue than aquaculture and fisheries combined. This transformation created whale-watching, a commercial tourism form that, despite recent critiques, has been accepted as non-consumptive activity. This paper uses four academic discourses to critique whale-watching as a form of capitalist exploitation: (1) commercial whale-watching and global capitalist transformation, (2) global capitalist politics and the promoted belief that whale-watching is non-consumptive, (3) the inherent contradictions of non-consumptive capitalist exploitation, and (4) whale-watching as a common-pool resource. These discourses lead us to critique whale-watching practices in relation to the common capitalist sequence of resource diversification, exploitation, depletion and collapse. Using specific impact studies, we conclude that a sustainability paradigm shift is required, whereby whale-watching (and other forms of wildlife tourism) is recognised as a form of non-lethal consumptive exploitation, understood in terms of sub-lethal anthropogenic stress and energetic impacts. We argue the need for a paradigm shift in the regulation and management of commercial whale-watching, and present the case for a unified, international framework for managing the negative externalities of whale-watching. The relevance of the issues raised about neoliberal policy making extends beyond whale-watching to all forms of wildlife and nature-based tourism.

Keywords: Marine tourism, wildlife tourism, whale-watching, capitalism, sub-lethal anthropogenic stress.

Introduction

Capitalist economies attempt to achieve continued economic growth through diversification of the ways in which natural resources are exploited (Harvey, 2011). This is evident in the late-capitalist exploitation of previously untapped marine resources (Fletcher & Neves, 2012). We now exploit marine environments (including wildlife species) directly or indirectly in diverse ways, through industrial fishing/hunting, tourism, shipping, oil and gas exploration and production, military operations, and renewable energy development (Williams, 2014). All of these activities are known to cause behavioural perturbations in many taxa (Berger-Tal et al., 2011), including top-order predators that are vital to ecosystem function (Estes et al., 2011; Myers, Baum, Shepherd, Powers, & Peterson, 2007). Despite now long-standing evidence that repeated exposure to tourism causes disturbances that can affect the conservation status of the targeted species (e.g. Currey, Dawson, & Slooten, 2009), global whale-watching practices continue to be conducted in the general absence of strict and enforced regulations (e.g. Garrod & Fennell, 2004). While scientists seek to define the biological significance of tourism impacts on a case-by-case basis, regulation and management, as limited as it is, continues to be framed by dominant neoliberal capitalist discourses (Neves, 2010).

In this paper, we draw together four existing academic discourses to critique global whale-watching as a form of capitalist exploitation of the marine environment. First, we consider the commercial whale-watching industry as a powerful form of global capitalist transformation (Fletcher & Neves, 2012). We then frame whale-watching within the context of global capitalist politics, which, we argue, has perpetuated the belief that whale-watching is non-consumptive (Neves, 2010; Higham et al., 2014). Thirdly, and in order to highlight the inherent contradictions of non-consumptive capitalist exploitation, we highlight parallels in different forms of lethal and non-lethal human interactions with wild animal species. Finally, we consider whale-watching in relation to common-pool resource (CPR) theory to highlight the common capitalist sequence of resource diversification, exploitation, depletion and collapse. These discourses, we argue (as other have before us), highlight the refusal of capitalism to accept responsibility for the costs and consequences of production (Harvey, 2005) in order to perpetuate the privatization of profit and socialization of the negative externalities of production (Harvey, 2011). As such, it should be acknowledged that whale-watching is a consumptive activity. Recognising whale-watching as a form of non-lethal exploitation, which may impact animal morbidity (e.g., sub-lethal anthropogenic stress) and mortality (e.g., vessel strikes), represents a paradigm shift in thinking away from so-called non-consumptive wildlife tourism (and the assumptions that it supports), with implications for regulation and sustainable management.

Marine tourism and whale-watching

Marine tourism has, in recent years, functioned as a new frontier of late-capitalist transformation (Fletcher, 2011; Fletcher & Neves, 2012). In the last decade or so, it has come to generate more global revenue (2006: US$222 billion[1]) than aquaculture (2006: US$78.8 billion) and fisheries (2006: US$91.2 billion, first sale value) combined (FAO, 2009; Honey & Krantz, 2007; UNEP, 2008). Watching and interacting with cetaceans (whales, dolphins and porpoises) in the wild (hereafter referred to as whale-watching) is an important part of this industry (Higham, Bejder, & Williams, 2014; O'Connor, Campbell, Cortez, & Knowles,

[1] Based on a 1998 global estimate of US$161 billion and given the conservative estimate that the sector grew at the same rate as the annual average growth rate of 4.1% for the global tourism industry.

2009). The growth of whale-watching has emerged from a significant shift in public attitudes toward cetaceans that dates to the 1970s and early 1980s (Corkeron, 2014; Samuels & Tyack, 2000). Once generally considered another inexhaustible resource to be exploited by humans, strong anti-whaling and pro-conservation sentiments, initially in western societies, became prevalent at around this time (Corkeron, 2006; Samuels & Tyack, 2000). With this shift in sentiment in some countries came increasing emphasis on the non-consumptive uses of cetaceans (Barstow, 1986). This terminology describes uses of cetaceans that do not immediately deplete populations, but rather afford tourists experiences of whales and dolphins in the wild (Corkeron, 2004). It stands in contrast to extractive (consumptive) uses that remove focal animals from their natural environment (e.g., trophy hunting, killing for animal products) (Duffus & Dearden, 1990).

Like other forms of wildlife tourism, whale-watching has been perceived uncritically as a non-consumptive activity (Barstow, 1986). The assumption of sustainability has allowed for unregulated and accelerating capitalist production of whale watching to be widely perpetuated (Higham & Bejder, 2008; Neves, 2010). However, the fact that whale-watching can affect the behaviour of targeted individuals has been known for more than 25 years (e.g. Baker & Herman, 1989). Species with varied life history strategies, from migrating baleen whales to resident populations of dolphins, perceive interactions with boats as a risk (Lusseau & Bejder, 2007). They respond by adapting their behaviour to integrate this risk in their ecological landscape when making behavioural decisions. Responses include evasive tactics (e.g. Bejder, Dawson, & Harraway, 1999; Nowacek, Wells, & Solow, 2001), leading to activity disruption (e.g. Christiansen, Rasmussen, & Lusseau, 2013a; Constantine, Brunton, & Dennis, 2004; Lusseau, 2004) or habitat abandonment (Bejder et al., 2006a Lusseau, 2005a).

The prevailing view has been that such effects do not adversely impact the survival or reproduction of those individuals (Neves, 2010). Accordingly, these effects have received lower priority than impacts that pose more immediate threats, such as incidental capture, or by-catch, in fisheries (Reeves, Smith, Crespo, & Notarbartolo-di-Sciara, 2003) and vessel strikes (Lammers, Pack, Lyman, & Espiritu, 2013). Recent research has demonstrated that, in some instances, tourism activities can negatively affect not only the activity budgets (Lusseau & Higham, 2004) and residency patterns of targeted wildlife populations, but also their conservation status (e.g. Currey, Dawson, & Slooten, 2011). These significant developments in our understanding of whale-watching (and wildlife tourism more broadly) highlight the need to consider an appropriate general framework within which to regulate and manage commercial whale-watching for sustainability.

Whale-watching as capitalist transformation

Tourism is a powerful expression of unrestrained neoliberalism (Mowforth & Munt, 2008) and has been a driving force of post-war capitalist transformation (Fletcher & Neves, 2012). The contribution of travel and tourism to global GDP in 2013 was US$7.0 trillion, accounting directly or indirectly for 266 million jobs, US$754 billion in annual investment and US$1.3 trillion in annual exports (WTTC, 2014). Marine resources have been implicated in this transformation. With the demise of whale hunting due to the collapse of target populations (Hammond, 2006; Williams, 2014), wildlife tourism has been at the forefront of new forms of capitalist accumulation (Neves, 2010). As governments have changed their political-economic systems to embrace neoliberalism and global capitalism (Harvey, 2011), and as regional communities have engaged in new forms of economic development (Hall & Boyd, 2004), tourism has become a driving force of economic transition.

The unrestrained growth of commercial whale-watching highlights the capitalist transformation that has occurred. Demand for whale-watching sustained annual growth of 12% per annum through the 1990s (Hoyt, 2001), growing much faster than the broader tourism industry itself (Garrod & Fennell, 2004; Honey & Krantz, 2007). By 2009, the whale-watching industry exceeded revenues of $2.0 billion a year, involved 13 million whale-watchers per annum and supported some 13,000 full time equivalent jobs (O'Connor et al., 2010). The sustainability issues associated with this tourism growth trajectory are multifarious. Capital investment in (marine) tourism has propagated a momentum that is difficult to divert (Britton, 1991; Fletcher, 2011; Harvey, 2011). Driven by capitalist enterprises that, by their nature, are generally engineered to maximize profit, resource degradation has come to be deployed as a means of adding scarcity value in the form of "extinction tourism" (Fletcher & Neves, 2012) and "last chance to see" (Lemelin, Dawson, Stewart, Maher, & Lück, 2011) experiences of nature that may be at risk of disappearing (Leahy, 2008). It may be argued that cetaceans have been subject to the industrialization of wildlife tourism like no other species of free-ranging wild animals (Higham et al., 2014).

This is a discourse dominated by capitalist transformation, neoliberal economic development and exploitation of new forms of natural capital (Fletcher & Neves, 2012). Through this period of transformation, new forms of natural capital themselves (cetaceans) have remained almost entirely without adequate protection from exploitation (Garrod & Fennell, 2004). Legislation, where it exists, has proved to be ineffective in most cases (Lusseau & Higham, 2004). Self-regulation in the face of growing demand is also widely considered to afford inadequate protection to wild animal populations subject to commercial tourism development (Allen, Smith, Waples, & Harcourt, 2007; Garrod & Fennell, 2004; Higham et al., 2014). In light of the rhetoric of non-consumptive wildlife tourism (Duffy, 2008; Neves, 2010), it is important to recognise that whale-watching is anchored in the dominant neoliberal paradigm of resource exploitation, profit maximization and capital accumulation (Fletcher & Neves, 2012).

Whales as economically and politically contested "resources"

Beyond its socio-economic impacts, whale-watching has served a political role in the anti-whaling debate (Neves, 2010), being strongly advocated by non-governmental organisations (NGOs) seeking to end whaling (Cisneros-Montemayor, Sumaila, Kaschner, & Pauly 2010; Corkeron, 2006). This has perpetuated the assumed sustainability of whale-watching as a tourism practice (Fletcher & Neves, 2012) in much the same way that the United Nations World Tourism Organisation (UNWTO) strongly advocated tourism as a 'smokeless industry' in the latter half of the last century. Both falsely imply that the activities are benign. The deliberate association of eco-consumption with biodiversity conservation has been a powerful argument for unregulated growth (Neves, 2010), even though ecotourism, despite its conservation ideals, is highly contested (Cater, 2007; Hall, 1994; Higham, 2007; Wheeller, 2012). There is also an inherent political tension between whaling and whale-watching interests (Higham & Lusseau, 2007). The political position of environmental NGOs, such as the International Fund for Animal Welfare (IFAW, 2014), is founded upon the assumption that commercial whale-watching is a non-consumptive activity that allows for a shift from conflict (hunting) to symbiosis (Budowski, 1976) in tourism and species conservation. This position has been pursued in the Azores (Neves, 2010) and Iceland (Andersson, Gothall, & Wende, 2014; Rasmussen, 2014) in an attempt to engineer a shift from whale hunting to whale-watching (Corkeron, 2014; Cunningham, Huijbens & Wearing, 2012).

In an effort to stop contemporary whale hunting, the focus of environmental NGOs has been to portray whale-watching as a “quintessentially and uniformly benign activity” (Neves, 2010, p. 721). This portrayal is contrary to the significant body of field-based behavioral science (e.g. Bejder et al., 2006b; Christiansen et al., 2013b; Lusseau, 2003, 2004, 2005a; Williams et al., 2006). Without acceptance that altered behaviours could have broader biological and ecological consequences (Corkeron, 2004; Neves, 2010), whale-watching has continued to grow in the almost complete absence of adequate regulatory and management frameworks (Allen et al., 2007; Higham, Bejder, & Lusseau, 2009). Indeed, such a portrayal situates whale-watching alongside what Žižek (2011) describes as “charity capitalism” – the building of an association between commodity consumption and a charitable cause. In this case, the charitable cause that is aligned with whale-watching, regardless of the possibility of sub-lethal anthropogenic stress, is the conservation of whales. It may be argued that these powerful economic and political discourses have hijacked and diverted the debate surrounding whale-watching development and sustainable management. This should not detract from the worthy intentions of those business operators who aspire to the highest standards of sustainable practice, and who may actively and positively influence public opinion. Those businesses that engage in less honorable practices (in terms of the welfare of individual animals and protection of populations) may be benefactors of the charitable intentions of tourists (who may assume that they contribute to the conservation of cetaceans by choosing whale-watching over whale hunting).

Parallels in contrasting forms of capitalist exploitation

Whale-watching, like other forms of wildlife tourism, has been treated uncritically as a non-consumptive activity (IWC, 1983; Knight, 2009), in large part because it has emerged as an alternative to lethal (consumptive) hunting practices (Duffus & Dearden, 1990). While whale hunting and whale-watching practices stand in obvious contrast (Knight, 2009), there are, in fact, problematic contradictions in the treatment of whale-watching as a non-consumptive activity (Tremblay, 2001). These contradictions have become increasingly acute as whale-watching has been subject to industrial-scale growth (Garrod & Fennell, 2004; Hoyt, 2001; O'Connor et al., 2010).

There is a compelling case to be made for whale-watching to be managed and regulated as a form of consumption (Meletis & Campbell, 2007), in recognition of the fact that it may cause sub-lethal anthropogenic stress (Christiansen & Lusseau, 2015). Whale-watching may be considered ocular consumption (Lemelin, 2006), insofar that it requires close proximity of tourists to wild animals, a practice associated with vessel strikes (Lammers et al. 2013), acoustic pollution (Lusseau, 2007) and behavioural disruptions due to anti-predator responses (e.g. Williams, Trites, & Bain, 2002). These may be implicated in animal morbidity due to sub-lethal anthropogenic stress and energetic constraints, which may have lethal cumulative effects (Christiansen & Lusseau, 2015; Christiansen et al., 2015; Bejder, Samuels, Whitehead, Finn, & Allen, 2009; Lima & Dill, 1990). This line of debate cements the view that treating whale-watching uncritically as non-consumptive has been misguided (Higham et al. 2014).

Managing the exploitation of a common-pool resource

This course of capitalist development has raised widespread concerns that the dominant neoliberal framings of environmental governance have failed to afford adequate protection to the environment (Byrne et al., 2004; Castree, 2008; Fletcher, 2011). Within this global context, local whale and dolphin populations are best viewed as common-pool resources instead of public goods (Heenehan et al. 2015; Pirotta & Lusseau 2014). Commercial tour operators compete to extract value from encounters with wild animals (Neves, 2010), which effectively consumes limited resources in order to extract maximum profit. The resource may be defined as the presence of animals in the vicinity of the commercial whale-watch business or, more specifically, the presence of animals on the ocean surface, often for a limited period of time, where they may be viewed. Common-pool resource theory is useful in that the more time one operator spends with an animal at the surface, the less time remains available for others to extract value from the same resource, or the less satisfied customers from other operations will be, which ultimately leads to losses in revenue (Finkler & Higham, 2004; Parrott et al., 2011).

Unregulated or unmanaged access to such common-pool resources, particularly those that can be depleted, can have catastrophic ecological (and, therefore, social and economic) consequences (Ostrom, Burger, Field, Norgaard, & Policansky, 1999). One of the great limitations of the capitalist dictum of growth through ever-greater extraction of value is the tendency for capitalism to destroy the very resources on which it depends (Harvey, 2011). This has been observed in commercial fisheries (e.g. Costello, Gaines, & Lynham, 2008; Jackson et al., 2001; Myers & Worm, 2003). Accelerated capitalist exploitation has occurred in many whale-watching contexts. High-speed whale-watch vessels allow for the accelerated capitalist production of whale-watching, which, simultaneously, has resulted in dramatically increased cases of vessel strikes (Lammers et al., 2013). Activities that exploit common-pool resources tend to fare better if they are publicly regulated, either through cooperation or institutional regulation (Harvey, 2011), so that limited resources can be allocated by a public third-party to individuals or groups. Where resources are shared across geopolitical borders, as is the case for many cetaceans, inter-governmental organisations (IGOs) must function as a third party (Archer, 2001) in order to promote the likelihood of sustaining an economically viable industry, while avoiding resource depletion or collapse (Costello et al., 2008).

Behavioural disturbances as non-lethal takes

In 2006, the International Whaling Commission (IWC) reached agreement that “there is compelling evidence that the fitness of individual odontocetes[2] repeatedly exposed to commercial whale-watching vessel traffic can be compromised and that this can lead to population-level effects” (IWC, 2006 np). This consensus emerged from a series of studies indicating that tour boats elicit avoidance responses from targeted cetaceans and that those responses can disrupt cetacean energy budgets. Repeated disturbance can lead to displacement and reduced population fitness (e.g. Bejder et al., 2006b; Lusseau, 2005a; Lusseau et al., 2006; Williams et al., 2006).

For example, interactions with tour boats caused avoidance responses and disrupted the activity budgets of bottlenose dolphins living in the Fiordland region, New Zealand (Lusseau & Higham, 2004). These interactions, in turn, increased the dolphins’ energy expenditure, decreased the amount of time they could spend near their basal metabolic rate (resting) (Lusseau, 2003, 2004), and decreased their foraging efficiency (Symons, Pirotta, & Lusseau, 2014). This region sustains three populations of the species, each using different fiords and exposed to differing levels of tourism. One population ranges over several small fiords, only one of which is subject to commercial tourism. When tour boat density was such that the inter-boat interaction interval for a dolphin was less than 70 min in that population, dolphins

[2] Odontocetes (approximately 70 species) are toothed cetaceans, which include dolphins and porpoises. Mysticetes (approximately 10 species) are larger baleen whales, which, unlike toothed whales, feed by filtering prey through their baleen plates.

abandoned that fiord (Lusseau, 2005a). In this instance, the population adapted to the habitat degradation caused by tourism. Another population, however, is exposed to tourism throughout its home range. When tour boat density increased such that the inter-boat interaction interval for a dolphin reached the 70 min threshold, a step-change in (reduced) calving success was detected (Currey et al., 2009; Lusseau et al., 2006; Symons et al., 2014).

Changes to activity budgets constrain the decisions that individual cetaceans can make about energetic allocation to survival and reproduction (Christiansen, Rasmussen, & Lusseau, 2013a; National Resource Council, 2005; New et al., 2013). Cetaceans, being long-lived and slow to reproduce, will prioritize survival over calving (New et al., 2013). The decline in the size of the Fiordland dolphin sub-population has been attributed to a reduction in calf survival, with the impact of repeated tour boat interactions identified as one of the most likely causal factors behind the decline (Currey et al., 2009). This, effectively, equates to tourism's "take", or "consumption". The Fiordland sub-population was subsequently listed as Critically Endangered by the International Union for the Conservation of Nature (IUCN) (Currey et al., 2011). In response, the New Zealand government formulated the Doubtful Sound Marine Mammal Code of Management (Department of Conservation, 2008), including the establishment of Dolphin Protection Zones (DPZs) (Lusseau & Higham, 2004). While not being completely closed to vessel traffic, tour operators were not permitted to enter the DPZs if dolphins were detected in them. Even though kayaks could also elicit behavioural disruptions (Lusseau, 2003), they were exempt from this exclusion.

This Code acted as a provision under which tour operators could obtain consent to interact with dolphins. Previous research had shown that operators were likely to behave more responsibly and, hence, have less impact on the dolphins, if they had progressed through the consent process (Lusseau, 2005b). The increase in dolphin-watching tourism in Doubtful Sound was driven by an increase in non-consented operators (Lusseau, 2005b). Operators without the consent were not obliged to adhere to this Code. Thus, such a management regime does not appear to address the problem posed by over-exposure of dolphins to tour boat interactions in Doubtful Sound. This situation was paralleled in Port Stephens, Australia. The voluntary Code of Conduct for tour operators proved ineffective in reducing exposure of dolphins to multiple operators and other vessels to which the Code did not apply (Allen et al., 2007).

Tourism-caused activity disruptions have now been well documented in both odontocetes (as per above) and mysticetes (e.g. Christiansen et al., 2013b; Heckel, Reilly, Sumich, & Espejel, 2001), yet they do not always lead to population-level consequences (e.g. Gulesserian, Slip, Heller, & Harcourt, 2011). Some populations are sufficiently large and wide-ranging that the proportion of individuals exposed to whale-watching is relatively small and, hence, any effect on that cohort will not affect the population's growth rate (nor its conservation status). This is the case in Kaikoura, New Zealand, regarding commercial tourist interactions with dusky dolphins (Lundquist, 2014). Alternatively, the ecological conditions in the range of the animal population and the site where interactions take place may be such that individuals can compensate for any behavioural disruptions (Lundquist, 2014). While we would not anticipate population-level effects in these cases, there could still be ecosystem-level effects. For example, predators would have to feed more in order to fuel the added energetic costs of interactions, which, in turn, could cause a depletion of their prey base (Williams et al., 2011).

Across many other taxa, individuals from tourism-exposed populations/colonies have lower fitness than those that are not. Detected impacts include lower reproductive success, lower fledging weights, and reduced body and health condition (e.g. Amo, López, & Martín, 2006; McClung, Seddon, Massaro, & Setiawan, 2004; Müllner, Linsenmair, & Wikelski, 2004). However, these demographic studies are often difficult to reconcile with short-term behavioural impact studies that do not consider the context in which the observed behavioural responses, or lack of responses, occur (Beale & Monaghan, 2004; Bejder et al., 2006a; Bejder et al., 2009). While these studies inform an understanding of the population-level impacts of wildlife tourism, it is difficult to apply the results to inform management.

The sheer scale of industrial whale-watching highlights the absence of an overarching mechanistic framework to understand the context of the impacts it perpetrates (Garrod & Fennell, 2004; Higham et al., 2014; O'Connor et al., 2010) and the prevailing *status quo* in the way marine wildlife tourism is managed (Corkeron, 2006). Recent advances in state-space modeling and individual-based simulation techniques provide one approach to linking these context-dependent behavioural disruptions to vital rates and, ultimately, population dynamics (New et al., 2013, 2014). Such approaches provide an empirically informed simulation platform that can be used to assess the wider consequences of behavioural disruptions. However, these simulations still require clearly defined management aims.

Given the demonstrated potential for adverse population-level impacts, we propose that whale-watching and, arguably, the whole wildlife tourism sector, should be considered a consumptive activity. This argument is supported by the synthesis of 25 years of impact assessment studies, which echoes previous propositions that wildlife tourism should not be considered different from any other form of wildlife exploitation (Meletis & Campbell, 2007; Tremblay, 2001). Indeed, the IWC Scientific Committee noted "in the absence of these data it [the relationship between an individual's fitness and exposure] should be assumed that such effects [compromised fitness] are possible until indicated otherwise" (IWC, 2006). This means that whale-watching should be managed in a precautionary, science-based manner with the aim to minimize its risk to both individual animals within a population, and the conservation status of the species.

A science-based adaptive management framework

The call for a science-based adaptive management framework that recognises whale-watching as a consumptive activity must acknowledge the complexity of the management context. Regulation and management of marine common-pool resources is fraught with difficulty given the complexities of jurisdiction and open access (Heenehan et al., 2015). Despite efforts to designate marine protected areas (MPAs), coastal and pelagic environments have few barriers to access (both public and commercial) and are notoriously difficult to police (Lusseau, 2005b). Marine environments are also subject to varying socio-political-economic contexts, which operate at multi-scalar levels from the global to the local (Higham et al., 2009; Dimmock, Hawkins & Tiyce, 2014). The complex and dynamic interplay of global, national and local process should not be underestimated (Young, 1999). In terms of stakeholder theory, it has also been asserted that members of local communities are more likely to protect natural resources if they stand to gain financially from the conservation of those resources (Honey, 1999). Within the neoliberal paradigm, ecotourism is intended to incentivize conservation without the need for explicit regulatory frameworks. However, the challenge remains to recognise and act upon whale-watching as a consumptive activity, and to engage in adaptive management practices (Farrell & Twining-Ward, 2004).

Whales and dolphins are still considered an open-access public good in most jurisdictions (Cisneros-Montemayor et al., 2010; Peterson, 1992). Most whale-watching activities lack formal management or have operated under local operator self-regulation (Allen et al., 2007; Wiley, Moller, Pace, & Carlson, 2008). Such a management approach has failed to ensure the sustainability of other industries (e.g. see the meta-analysis of fish stock exploitation patterns under different management regimes in Costello et al., 2008). In the few locations where compliance to self-regulatory measures has been assessed, the results are not encouraging (Allen et al., 2007; Lusseau, 2005b; Whitt & Read, 2006; Wiley et al., 2008). On the northwest coast of the United States of America (USA), for example, the National Oceanic and Atmospheric Administration (NOAA) responded to non-compliance issues and impacts by moving to impose public regulations to decrease interactions between the whale-watching industry and killer whales, where self-regulation has been in place for several decades. This includes attempting to set vessel exclusion zones (NOAA, 2009).

The question of self-regulation is critical to this discussion. There are many cases of whale-watching taking place in accordance with codes of conduct, best practice guidelines and certification (Tyne, Loneragan & Bejder, 2014). Indeed, some of these self-regulation practices are the initiative of commercial operators who seek to respond directly to issues of sustainability (Allen et al., 2007; Garrod & Fennell, 2004). While voluntary guidelines may have merit (e.g., they may raise awareness of the impacts of whale-watching on cetaceans), they lack the enforcement that could come with legislation and regulation (Tyne et al., 2014). Self-regulation has failed at key, mature destinations, primarily because conditions to foster cooperation were not present. In this regulatory environment, a defecting operator (i.e. an operator that does not consistently conform to agreed guidelines), is unlikely to incur any significant negative consequences of conduct that deviates from a stated code of best practice (Pirotta & Lusseau, 2014).

Accordingly, a paradigm shift is necessary: moving to public regulation of whale-watching. Sustainable thresholds, based on critical elements of carrying capacity, could be defined using an integrated and adaptive management model based on Limit of Acceptable Change parameters (Corkeron, 2006; Higham et al., 2009). This approach has proven successful in other contexts, such as fisheries (Costello et al., 2008) and agriculture (Ostrom et al., 1999), although obtaining the appropriate scientific data is not a trivial task (Ahn, Lee, & Shaffer, 2002). This public regulation need not be implemented by a governmental institution, but does require costs arising from lack of compliance to be mandatory.

Several countries have taken precautionary steps and have been licensing whale-watching activities under Marine Mammal Protection Acts, State legislation or Marine Protected Area regulations for some time (e.g. Australia and New Zealand). In general, however, these licensing frameworks do not have explicit mechanisms to define sustainable thresholds on the number of licenses that can be issued for given sites (thresholds to maximize economic viability and minimize threats to population viability). Thus, these regulatory frameworks may not prevent whale-watching fleets reaching overcapacity. As is the case for commercial fisheries, socio-political constraints preclude downward adjustments when allocated quotas/licenses exceed carrying capacity (Worm et al., 2009).

The one exception to this is the management of the whale-watching industry in Hervey Bay, Queensland. Within two years of the industry's start in 1987, a spatially explicit management scheme was put in place, with defined management goals. Management started with a mathematical model for the number of licenses issued; a transferable scheme of licensing that included enforcement (individual transferable quotas, ITQs); and the capacity for industry consolidation over time (Jeffery, Postle, & Simmons, 1994; Smith, Newsome, Lee, &

Stoeckl, 2006). Management lessons that can be drawn from the decades of practice provided by this example remain ignored elsewhere. Current industry growth is now driven by the expansion of destinations in developing countries, where regulatory processes to manage natural resource exploitation are generally limited and often non-existent (e.g. Beasley, Bejder & Marsh, 2014; Coria & Calfucura, 2012; Mustika, Birtles, Everingham, & Marsh, 2013; Mustika, Birtles, Welters, & Marsh, 2012). These governance issues cannot be ignored in setting conservation priorities (Eklund, Arponen, Visconti, & Cabeza, 2011). Further, the fact that whale-watching development has been so strongly advocated and encouraged by international organisations for short-term political gains has been to the detriment of long-term sustainable resource use (Cisneros-Montemayor et al., 2010; Corkeron, 2004; Neves, 2010).

We propose that, in the interests of sustainability, the overarching management aim should be maintenance of the favourable conservation status of targeted populations. To achieve a more proactive approach to management, we need to: (i) understand how disturbances to cetaceans elicited by whale-watching boats are interacting with local ecological conditions and life history strategies to relate to population-level consequences; (ii) develop mechanisms to define the upper level of whale-watching activities that animal populations can sustain; and (iii) clearly define the uncertainties surrounding these thresholds and assess the sensitivity of management decisions to these uncertainties. This will aid decision-making processes and further guide research efforts (Punt & Donovan, 2007). Such an iterative assessment process can then form the basis for rational decisions in an adaptive management scheme that can be used to guide the development and sustainability of whale-watching. It would require international coordination, given that (as has been noted) many cetaceans cross national maritime borders and move between international jurisdictions on a regular (e.g., seasonal) basis. Several IGOs have the remit to coordinate such a scheme. It also requires a paradigmatic change in thinking, towards recognition and acceptance of whale-watching as a form of consumptive capitalist resource exploitation (Higham et al., 2014), based on acceptance that whale-watching may cause sub-lethal anthropogenic stress.

An internationally coordinated adaptive management scheme

Evidence exists that tourism exposure can have negative, population-level consequences for cetaceans, as it can for a variety of other taxa. We do not argue that tourism exposure will have population-level consequences in all cases. Instead, we argue against waiting to detect adverse population-level consequences in cetacean populations before developing and implementing management frameworks (Corkeron, 2004; Wilson, Hammond, & Thompson, 1999). Due to current scientific uncertainties and evidence of resource degradation in some situations, we need to implement a regulatory approach to the management of whale-watching based on the precautionary principle (Sandin, 1999, Fennell & Ebert, 2004). Whale-watching should, thus, be considered a consumptive activity, requiring a more appropriate regulatory framework to manage the industry for the benefit of the public and the marine environment (Champney, 1988). The tools used to ensure that management aims are met should be tailored to the location and the behavioural ecology of the targeted population/s. Time-area closures can be useful in a mixed industry, where whale-watching companies share cetaceans with recreational boats or other tourism ventures. Establishing conditions under which boat interactions with cetaceans are restricted and where governance issues are present (Corkeron, 2006; Costello et al., 2008) could ensure a more sustainable local approach to management.

The current state of whale-watching can be equated to that which we faced with the management of cetacean bycatch[3] in fisheries two decades ago (Read, Drinker, & Northridge, 2006), a pervasive threat to many cetacean populations today (e.g. Slooten, 2013). Bycatch causes population-level consequences in certain instances and not in others (Allen et al., 2014). However, logically, the variation in the propensity for population-level consequences did not preclude the development of management frameworks to address the biological impacts of bycatch. This saw the development of the Potential Biological Removal equation in the USA (Wade, 1998), allowing scientists and managers to calculate the acceptable human-caused take of cetaceans impacted by fishing activity (Wade, 1998; Taylor, Wade, DeMaster, & Barlow, 2000).

The fact that the viability of cetacean populations can be affected by whale-watching suggests that some political strategies – polarising whaling and whale-watching, while providing uncritical support for whale-watching, for example – require urgent revision (Corkeron, 2014). Many communities now involved in whale-watching activities have already undergone substantial socio-economic trauma after the collapse of local fisheries (Orams, 2002; Simmons, 2014). Ensuring the sustainability of marine tourism in general, and whale-watching in particular, is, therefore, of paramount importance in both ecological *and* socio-economic terms.

Conclusion

Tourism has been at the forefront of late-capitalist transformation of the marine environment. Capitalism is characterized by the privatization of profits, resource exhaustion and failure to

[3] The term bycatch describes unwanted non-target fish (and other marine species) caught in commercial fisheries (most of which die and are disposed of at sea), and the associated adverse ecological consequences of such practices.

accept responsibility for the costs of production (Harvey, 2011). Those costs, in the case of whale-watching, need to be understood in terms of sub-lethal anthropogenic stress and energetic impacts (e.g. Christiansen et al., 2014). This path of capitalist transformation demands a fundamental shift in policy and management thinking. Such a shift requires the development of a unified, evidence-based framework to manage all threats – lethal and non-lethal - that marine animal populations now face, on par. The reformulation of whale-watching as a form of consumption and as a cause of sub-lethal anthropogenic stress signals the need for commercial whale-watching to be managed within an architecture of strong national and international regulation. Despite the unique challenges of managing marine common-pool resources (Heenehan et al., 2015), an urgent need exists to move away from an open-access management paradigm for whale-watching (Corkeron, 2006).

Given the regularity and scale of movements across international boundaries by many cetacean populations, the coordination of tourism management plans will be challenging, though not insurmountable (Dietz, Ostrom, & Stern, 2003). We now need to identify the suite of generic management mechanisms that can be applied according to the socio-ecological characteristics of each whale-watching destination (Ostrom, 2009). IGOs, such as the IWC or the Convention on the Conservation of Migratory Species of Wild Animals, offer possible fora and already possess precautionary and adaptive management mechanisms to deal with the capitalist exploitation of long-lived, slow reproducing shared resources (Punt & Donovan, 2007). Other IGOs, such as the IUCN, the United Nations Environment Programme, and the United Nations World Tourism Organisation, should also be involved in defining sustainability targets for this industry.

In this paper, we present the case for a unified and international framework for managing whale-watching impacts. We highlight the importance of understanding contemporary commercial whale-watching practices in terms of capitalist transformation (Fletcher & Neves, 2012), political contestation (Neves, 2010) and common-pool resource management (Ostrom et al., 1999). Drawing insights from specific behavioural studies, we argue that a sustainability paradigm shift is required, whereby whale-watching is accepted as a form of non-lethal consumptive exploitation and actively managed based on the definition of sustainable thresholds. Several management tools such as time-area closures or spatially explicit, transferrable quotas are available to implement an internationally coordinated management approach that can be locally implemented in accordance with site-specific socio-ecological pressures and planning/management structures (Dietz et al., 2003). This represents an urgent challenge to the sustainable management of commercial whale-watching, which is itself in need of paradigmatic transformation. It also highlights the need for new thinking about issues of neoliberal policy making that are not confined to whale-watching, but rather apply to all wildlife tourism practices and to nature-based tourism more broadly.

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